THESIS

MOVEMENT AND NESTING ECOLOGY OF TWO ENDANGERED HAWAIIAN WATERBIRDS

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ABSTRACT

MOVEMENT AND NESTING ECOLOGY OF TWO ENDANGERED HAWAIIAN WATERBIRDS

Waterbirds rely on the preservation of wetland habitats to support key life-history cycles. Emerging threats to wetland habitats from climate change include increased variability in precipitation, salinity intrusion, and sea level rise; these changes can exacerbate wetland drying and flooding periods and erode the extent and quality of coastal wetlands, making it more difficult for waterbirds to find suitable habitat across their annual cycle. Hawaii's waterbirds are especially at risk, as the island of O‘ahu has already lost 65% of its wetlands to development. Before remaining habitat is lost to sea level rise changes to water availability can affect water quality, food abundance, and plant communities for Hawaii's waterbirds. The Hawaiian Common Gallinule (‘alae ‘ula, *Gallinula galeata sandvicensis*) and the Hawaiian Coot (‘alae ke‘oke‘o, *Fulica alai*) are the only two Hawaiian Rallid endemics found in the world, both endangered and conservation-reliant. However, there is limited information on how Hawaiian coots and gallinules may respond to changing environmental conditions of water availability, either through movement or nesting habitat selection. I tracked population and individual movements through waterbird surveys and satellite GPS transmitters from January of 2023 to April of 2025 at Kailua, HI, USA, and evaluated the effects of water availability and quality on abundance, space use, and documented the timing of long-distance movements. I also determined habitat characteristics affecting nest site selection and hatching success of the Hawaiian gallinule.

Hawaiian coots had a higher propensity to movement perhaps linked to seasonal changes in environmental conditions, while Hawaiian gallinules remained sedentary despite fluctuations in water availability. I found synchronous changes in the abundance of Hawaiian coots at three wetland sites in Kailua, HI, indicating immigration from wetlands outside the local area and seasonality in abundance with increases in the wet season. Hawaiian coot abundance was affected by water level, rainfall, and water quality, suggesting coots were moving to find favorable habitat. In contrast, gallinule abundance was not impacted by water-related variables; rather, declines in abundance may have been from golf course hazards. Through satellite GPS transmitter tracking, 24% of coots and 10% of gallinules tracked moved away from their home wetland, but individual movements did not appear seasonal. Both Hawaiian coots and gallinules had similar home range sizes during the wet and dry seasons, and little effects of sex, and site on Hawaiian coot home range, and no effects on Hawaiian gallinule home range. Nesting success of Hawaiian gallinules was high (83%), likely due to the absence of predators at the nest and flooding events. Failures were attributed to conspecific egg destruction, abandonment, and unknown causes. Hatching success averaged 53.4% and increased with water depth nearest the nest; deeper water can provide abundant invertebrate food sources, thus reducing time off the nest, resulting in consistent incubation and nest protection. I found that native vegetation around the nest was the only nest-site characteristic significantly associated with nest-site selection in a landscape dominated by non-native pickleweed (*Batis maritima)*. My results show that Hawaiian coots have a higher propensity to movement than gallinules, and water availability influences successful nesting of gallinules. Management of nesting habitat where sedentary Hawaiian gallinules occur may increase population productivity, and preservation of landscape-wetland habitat connectivity may aid Hawaiian coots in safe passage in local and regional movements. As sea level rise and climate change increase freshwater scarcity and degrade wetland habitats, understanding Hawaiian coot and gallinule movements, population abundance, and factors affecting nesting is critical to their conservation.

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CHAPTER 1

ENVIRONMENTAL INFLUENCES ON HAWAIIAN WATERBIRD MOVEMENT AND ABUNDANCE

Introduction

Wetland systems are one of three major ecosystems on earth and are very susceptible to habitat loss from climate change and development: 87% of the world's wetlands have been lost since 1700 CE (Gardner and Finlayson 2018, Chen et al. 2019). Recent studies and satellite imagery capture the decline in wetland extent, primarily driven by human disturbance and climate change, which has increased rapidly over the past 200 years (Gong 2010, Davidson 2014). Sea level rise threatens inundation of coastal wetland habitats and may exceed critical rates in coastal zones by 2100 if warming exceeds 2 °C (Saintilan et al. 2023). Additionally, climate-driven weather changes (e.g., hotter, more frequent heat waves, greater precipitation variability, and more frequent extreme weather events) are key drivers of wetland surface water loss (MacDonald 2010, Dai 2011, Stott 2016). Wetland surface water loss reduces available habitat, which can increase salinity and water temperatures. These effects can cause algal blooms and disease outbreaks, as well as changes in food resources, all of which reduce the functionality and productivity of a wetland (Nielsen and Brock 2009, Van der Kolk et al. 2020, Prosser et al. 2023). Despite existing wetland conservation efforts, many government agencies lack sufficient wetland protection plans (Mojica Vélez et al. 2018). The effects of higher temperatures exacerbate habitat limitation and issues with water composition, which greatly affect wetland wildlife populations and biodiversity (Walther et al. 2002).

Such alterations in wetland hydrology and quality have significant consequences for wetland-dependent species, prompting shifts in distribution, behavior, and habitat selection. Despite changing wetland conditions, some waterbird species have shown adaptations through behavioral plasticity. Locally, Eurasian oystercatchers (*Haematopus ostralegus*) in the Netherlands have been increasing their nesting elevation by 5mm/year from 1995-2016 in response to higher tides associated with rising sea levels (Van De Pol et al. 2023). Migratory waterbirds (i.e., shorebirds, wading birds, and waterfowl) in Oregon and California are losing permanent wetland habitat, but are shifting to utilize temporary or seasonal wetlands instead (Donnelly et al. 2022). However, habitat changes may be difficult for nonmigratory waterbirds like rails (Rallidae) to adapt to, and not much is known about their ability to use movement to find suitable habitat in a changing environment (Fournier et al. 2017, Kolts and McRae 2017). This lack of movement data is particularly concerning for island rail species (Takano and Haig 2004), such as those in Hawaiʻi, where limited habitat availability and isolation may heighten vulnerability to environmental change (van Rees and Reed 2018).

Hawaii's freshwater and coastal wetlands are diminishing due to climate change, human development, and non-native plant invasions, which affect native populations and diversity (USFWS 2011). The island of Oʻahu has lost an estimated 65% of its wetlands, the most of any Hawaiian island (van Rees and Reed 2014). The Hawaiian Gallinule (ʻalae ʻula, *Gallinula galeata sandvicensis*) and the Hawaiian Coot (ʻalae keʻokeʻo, *Fulica alai*) are two of four extant waterbird species endemics to the main Hawaiian islands and are federally listed as endangered (USFWS 2011). All Hawaiian waterbird species are conservation-reliant, and need continued, active management for their populations to persist (Underwood et al. 2013). Hawaiian waterbird numbers have declined primarily due to habitat loss and the introduction of predatory mammals (USFWS 2011). Hawaiian gallinules only occur on the islands of Oʻahu and Kauaʻi, while Hawaiian coots inhabit all major Hawaiian Islands except for Kahoʻolawe (Reed et al. 2012). The Hawaiian coot population average is estimated at 1564, while the Hawaiian gallinule has an estimated average of 712 (Gorreson et al. 2024). The Hawaiian waterbird recovery plan designates core and supporting wetlands for preservation (USFWS 2011), but it is unknown how these wetlands are connected through movement.

The movements of Hawaiian gallinules and Hawaiian coots remain poorly understood, particularly in relation to environmental factors and climate change impacts. Hawaiian gallinules and Hawaiian coots are territorial, wetland-dependent birds that inhabit natural and artificial wetland habitats (e.g., golf courses, ponds, and agricultural fields) (USFWS, 2011). A VHF (very high frequency) telemetry study recorded limited intra-island movements of Hawaiian coots on Oʻahu and no movements of gallinules (Paxton et al. 2022). A key limitation of VHF tracking is that it cannot detect movements outside of the receiver's range, so movements to remote areas or off island would go undetected. Separately, resightings of banded individuals and landscape genetics documented intra-island movements of Hawaiian gallinules (van Rees et al. 2018a, van Rees et al. 2018b) and inter-island movements of Hawaiian coots (Riggs 2016), perhaps in response to rainfall events (Engilis and Pratt 1993). Nesting peaks March-August, but both species breed year-round depending on water availability (Byrd and Zeillemaker, 1981). With more variability in weather and rainfall due to climate change (Frazier and Giambelluca, 2016), coots and gallinules may adjust their movements and habitat use; however, little is currently known about their individual and population responses to changes in water availability and quality. Further, estimating home range size and population abundance in shrinking habitats could provide useful information for the management of these species.

In this study, we investigated the movement patterns of the Hawaiian gallinule and Hawaiian coot in relation to water availability and weather conditions. Biannual waterbird surveys have documented rapid changes in coot abundance at some wetland sites (USFWS 2011), which are assumed to result from movements between wetlands; however, the seasonal timing of these changes and their environmental correlates have so far been unexplored. At an individual level, we captured and uniquely marked a subset of Hawaiian coots and gallinules and monitored large-scale movements (defined as movements off home wetland) with satellite-GPS transmitters, to quantify seasonal movement patterns, if any. At a population level, we tracked coot and gallinule abundance at three wetland sites to identify seasonal movements in relation to water availability and water quality. We expected to see spikes in large-scale movements and decreases in site abundance during inclement weather (droughts, heavy rainfall events, etc.). Additionally, we estimated the home range size of our focal species during the wet (breeding) and dry (non -breeding) seasons through satellite-GPS transmitter tracking. We also expected to see larger home ranges during the dry season due to increased foraging from reduced food availability (Conway et al. 1993), and larger male home ranges than female home ranges due to their larger body size (van Rees et al. 2021). As coastal wetlands at MCBH are projected to be gradually inundated by sea-level rise over the coming decades (EA Engineering, Science, and Technology, Inc. 2024), understanding coot and gallinule movements and habitat use is needed to inform their conservation.

Methods

Study Area

Our main study area was Marine Corps Base Hawaiʻi (MCBH), located on the east side of Oʻahu at the Mōkapu peninsula in Hawaiʻi, USA (21.4398° N, 157.7552° W). Our study sites consisted of seven brackish and freshwater wetlands (Figure 1). Brackish water habitats (Paʻakai, Waipuna, Kaluapuhi, Sag Harbor) are tidal wetlands (except Sag Harbor) that flood with the tide and with rainfall. Freshwater habitats included Klipper Golf Course (hereafter Klipper; ʻAlae, ʻIwa, and Koloa ponds), Percolation Ditch (hereafter Perc Ditch), and Temporary Lodging Facility (hereafter TLF) stream. Study site elevations ranged from 0-4.9 m. The vegetation consisted mainly of non-native pickleweed (*Batis maritima*), native sedges kaluhā (*Bolboschoenus maritimus*) and ʻahuʻawa (*Cyperus javanicus*), woody herbaceous plants Indian camphorweed (*Pluchea indica*), and milo (*Thespesia populnea*) trees in upland areas.

We included two other peripheral wetland sites for population monitoring and capture: Hāmākua and Kaʻelepulu (Figure 2); both sites host the largest coot populations in Kailua outside of MCBH. Hāmākua is a seasonally brackish 15 ha marsh and upland area located on the west edge of Kailua Town (21.3898° N, 157.7417° W). Field work and monitoring of Hāmākua took place in the lower elevation wetlands, which are approximately 3.8 ha, with an elevation range of 0.4 -5.2m. Hāmākua marsh comprises four interconnected basins connected by the Kawainui stream (Works et al. 2024). In the emergent wetland area, the dominant vegetation is pickleweed, with patches of interspersed kaluhā and ʻahuʻawa. Hāmākua is owned and managed by the State Department of Land and Natural Resources (DLNR) Division of Forestry and Wildlife (DOFAW). There, we tagged Hawaiian coots and conducted weekly waterbird surveys. Kaʻelepulu or Enchanted Lakes (21.3823° N, 157.7377° W and 0.1 -5m elevation) is a privatelyowned tidal wetland, where we conducted waterbird surveys but did not capture or tag birds. The majority of the landscape is brackish open water, with small patches of pickleweed and kaluhā.

Trapping Methods.and Banding

Capture efforts of Hawaiian gallinule and Hawaiian coots ran from January 2023 to March 2025. We used walk-in cage traps (Tomahawk, model 107, Hazelhurst, WI, USA) baited with cracked corn and mango and placed on floating platforms constructed from plywood and boogie boards, to trap coots and gallinules. Traps were monitored with a Reconyx (Hyperfire 2.0, Holmen, WI, USA) cellular-enabled camera, which allowed us to remotely check if traps were triggered. Traps were monitored every 15-60 minutes while deployed. At Hāmākua, 12-18 m mist nets were deployed across streams to capture scuttling or flying coots. Nets were not used to target gallinules since walk-in traps were relatively successful. Additionally, coots were captured at night or during the day using dip nets, either while hiding in vegetation or on a nest established for at least 2 weeks to prevent abandonment. Hawaiian coots were more difficult to trap and were captured only in 2024 -2025.

Each bird captured was fitted with a size 6 USGS band and a unique combination of Darvic color leg bands which were soldered on. Gray was the site code for MCBH, and Orange was the site code for Hāmākua; all band combos were recorded top left, bottom left: top right, bottom right (LL:RR) with the site code and aluminum USGS band on the same leg. Morphometrics and body mass (g) were collected. We collected two feathers from the breast and a small blood sample (<1mL) with a 28 -gauge needle for each bird, to determine sex through amplification of sex-specific alleles.

Molecular sexing

Both Hawaiian gallinule and Hawaiian coots are sexually monomorphic, so sex assignment must be determined via molecular sexing. Blood spots and/or feathers were collected during banding of Hawaiian gallinules and Hawaiian coots and were used for molecular sexing (n=68). For each sample, we cut half of a pea-sized blood spot from the filter paper and sliced it into fine pieces with sterile nail scissors into a sterile 2mL Eppendorf tube. Pieces were then extracted with the DNEasy Blood and Tissue Kit (Qiagen) according to the manufacturer's protocol for tissue, but modified to include an overnight digestion period at 56°C to increase DNA yield. We conducted molecular sexing of all samples by polymerase chain reaction (PCR) amplification of sexually-diagnostic CHD1 alleles, using 1237L/1272H/GWR2 primers specifically designed for the closely-related American coot (Shizuka and Lyon 2008); these primers have been confirmed to work on Hawaiian coot (Riggs et al. 2019) and gallinule (S. Sonsthagen, personal communication). We amplified samples in 25ul reactions (0.5ul 1237L, 0.25ul 1272H, 0.25ul GWR2, 2ul DNA template, and 22ul water) with Illustra Pure-Taq RTG PCR beads (Cytiva Life Sciences, Marlborough, MA, USA). Amplification conditions were as described in Shizuka and Lyon (2008). We visualized amplified CHD1 alleles on stained 2% agarose gels under ultraviolet light. Male (ZZ) Hawaiian gallinules and coots showed one or two Z-specific bands, and females (ZW) showed one Z and one W-specific band. Allelic dropout can be a problem in DNA samples with low starting quantities and could cause us to mis-assign females (with W-specific band) as males (no W-specific band); however, extracted blood samples typically yielded high concentrations (>55ng/ul). We checked for allelic dropout by re-running any apparent males.

Transmitter Attachment

Twenty-eight Lotek Solar PinPoint Argos-S Tag (6.6 g satellite GPS tags) and four Geotrak 12 g Solar GPS PTT tags were applied to Hawaiian gallinules (n=15) and Hawaiian coots (n=17) with a modified backpack harness (Dwyer 1972). Geotrak transmitters with harnesses weighed 17-18 g, and Lotek transmitters with harnesses weighed 10-14 g. The total weight of the transmitter was not to exceed 5% of the bird's body weight, so birds weighing less than 360 g and 280 g were not fitted with Geotrak or Lotek transmitters, respectively. Only a subset of birds captured had transmitters attached due to budget limitations and weight restrictions. Lotek transmitters sent three GPS locations (10±m accuracy) and their transmission quality in 8-hour intervals. Geotraks collected 4-5 location points a day at 5-hour intervals. Data from both types of transmitters were sent to the Argos data hosting platform.

Home Range and Movement

For each transmittered bird, GPS points were exported from Lotek or Geotrak in .csv format, then points were converted to latitude longitude format and filtered to exclude transmission failures and outliers using Lotek Argos-GPS Processor (v.4.6). GPS files were imported into Esri ArcGIS Pro (v3.4) for visualization, and points before the deployment date were manually excluded. We used the packages *adehabitatHR* (v0.4.22), *terra* (v1.8.42), and *sf* (v1.0-20) packages in R (version 4.5.0) to calculate monthly bivariate kernel density estimations (KDE). We defined home range as a 95% isopleth KDE and core range as a 50% isopleth KDE, estimated via an ad hoc smoothing parameter (href) (Walter and Fischer 2016). Monthly KDEs were calculated in both dry (May 1 -October 31) and wet seasons (November 1-April 30). KDE home and core ranges were reported in m^2. To utilize datasets with incomplete months (<30 days), we calculated the minimum number of points required to estimate a KDE without a significant difference relative to complete datasets. To do that, we utilized data from three individual coots from Hāmākua, Pa'akai, and Waipuna, and three individual gallinules from MCBH: Klipper, Waipuna, and Pa'akai; coots and gallinules were analyzed separately. Lotek transmitters transmitted an average of 65 points per month, which is lower than the anticipated ~90 points. Therefore, for each bird, we randomly subsampled 5-65 points in intervals of 5 (n=13), at 100 iterations per subsample from each month with complete data. Incomplete months of each individual were excluded from this analysis. Then the average estimated 95% home range from each subsample was plotted for each month of the iterated data to estimate where home range estimates stabilized. Stabilization points for each month were then averaged across individuals, and a total average was calculated across all individuals to ascertain the minimum number of points that can be used in a monthly KDE for each species.

Monthly seasonal KDE for Hawaiian gallinules and Hawaiian coots were log-transformed to approximate a normal distribution. Linear mixed effects models were fitted to the log-transformed home and core ranges using the *lme4* (v1.1-37) package, with Hawaiian coots and gallinules evaluated separately. In each model our fixed effects were sex, season, and an interaction between sex*season; and a random effect of individual bird ID. Site was also included as a fixed effect in the Hawaiian coot model, but could not be retained for Hawaiian gallinules due to the few birds per site. Initially, the year was considered as a random effect, but it accounted for little variance and was therefore excluded. We predicted the same effects of season and sex on home range for both species. Overall, we expected males to have larger home ranges than females due to their larger body size (van Rees et al. 2021), and that dry-season ranges would be larger than wet-season ranges, if individuals of both sexes forage more widely when food availability is lower (Conway et al. 1993). However, given that breeding females may expand their home ranges in advance of nesting to acquire additional resources associated with egg laying (Bluso-Demers et al. 2008, Demers et al. 2008), we considered that females could have larger home ranges than males during the dry season. We were unsure of the effect each site would have on home range but expected home ranges to be smaller at sites with higher population density due to increased conspecific competition.

GPS points to neighboring wetlands (including wetlands within MCBH) or off the wetland boundary were considered large-scale movements. The Euclidean distance of large-scale movements was calculated in kilometers in ArcGIS using the Measure Distance tool. Days spent at new locations were recorded.

Abundance and Environmental Variables

Waterbird surveys were conducted at all wetland study sites, using the single-day waterbird monitoring protocol used biannually by Hawaiʻi DOFAW to count each species at each pond or other pre-defined sampling unit (Henry 2017). Surveys were conducted weekly during the field season (January-July, 2023-2025), and monthly during the off-season in 2024. Additionally, to detect synchronous or asynchronous movements of coots in the region, waterbird surveys were also conducted weekly at neighboring wetlands outside of MCBH at Hāmākua and Kaʻelepulu beginning in April 2023. Waterbird surveys at Sag Harbor, Klipper, TLF, Perc Ditch, Hāmākua, and Kaʻelepulu were conducted by walking along wetland areas and scanning for birds with binoculars. Waterbird surveys at Pa'akai, Waipuna, and Kaluapuhi were conducted using a point count method from a wildlife viewing tower. Tallies of all waterbird and shorebird species were recorded. A secondary waterbird survey, only to count coots at Pa'akai and Kaluapuhi, was conducted weekly during nesting surveys.

We placed water gauges at a visible location in each wetland, except Kaʻelepulu, with markings at 3cm intervals to allow observation of water depth from a distance. We recorded water level during each waterbird survey and coot-specific surveys at Pa'akai. We obtained daily precipitation (mm) measurements from the Moku o Loʻe (Coconut Island) weather station in Kāneʻohe Bay (PacIOOS 2025) - 4km from MCBH - and calculated recent rainfall as the sum of precipitation that fell during the three days prior to each survey date.

Water samples were collected twice a week during waterbird surveys and nest surveys in 2024 and 2025. We used a handheld YSI ProDSS multi-parameter water sampling instrument (Yellow Springs, OH, USA). Water samples were taken at the water gauge by collecting approximately 4 oz of water with the calibration cup from the middle of the water column. if possible. The YSI collected date, time, water temperature (F°), pressure (mmHg), dissolved oxygen (mg/L), specific conductance (μS/cm), salinity (ppt), pH, turbidity (NTU), and location (latitude and longitude). Sample measurements were recorded at 20-30 seconds of YSI reading.

Band Resights

We opportunistically collected location data of band resights of Hawaiian waterbirds (coots and gallinules) during weekly waterbird surveys, nesting surveys, and incidental encounters. For each resighted bird, we recorded species, wetland, date, time, and color band code. We tallied total resights, number of individuals resighted, proportion resighted in home wetlands (where banded) versus different wetlands, and individual movements captured by band resights.

Statistical Analysis

To predict coot abundance at Paʻakai and gallinule abundance at Klipper as a function of water level (cm), recent rainfall (mm), and water quality measures (specific conductance, salinity, turbidity, pH, and dissolved oxygen) we fitted generalized linear models with a Poisson distribution and log link using the *lme4* (v1.1-37) package in R. All covariates were scaled and centered; recent rainfall and turbidity were log-transformed to approximate a normal distribution. Specific conductance was excluded from analysis due to high collinearity with salinity. Paʻakai and Klipper were the only sites evaluated because both sites host the highest populations of coots and gallinules, respectively, at MCBH. We expected rainfall to negatively affect coot abundance during heavy rains due to coot dispersal to ephemeral wetlands (Engilis and Pratt 1993). We were unsure of the effect of water level on coot abundance. On the one hand, similar to our expectation for rainfall, a high water level indicative of greater water availability on the landscape would also decrease coot numbers at Paʻakai. On the other hand, drought periods decrease wetland water quality by concentrating solutes through evaporation (Mayer and Pilson 2019) and may also reduce coot numbers. Thus, we expected fewer coots when turbidity and salinity were high and when dissolved oxygen and pH were low. For gallinules, given prior evidence of limited intra-island movements (van Rees et al. 2018a, van Rees et al. 2018b), we expected gallinules to stay in home wetlands unless water levels and water quality were severely reduced by evaporative water loss. We expected rainfall to have no effect on gallinule abundance due to high site fidelity.

To see if Hawaiian coot populations change synchronously or asynchronously in the Kailua wetland complex, we used the Pearson product-moment correlation coefficient (*r*) of weekly coot abundance between sites: Pa'akai (MCBH)-Hāmākua, Pa'akai (MCBH)-Kaʻelepulu, and Hāmākua-Kaʻelepulu. A positive correlation between these highest-abundance sites would mean that populations are changing synchronously, suggesting that coots are immigrating from/emigrating to other wetlands outside of Kailua. A negative correlation would indicate that coot abundance is inversely related across sites, suggesting movement within the Kailua wetland complex. In line with observations from Pearl Harbor Wetland Complex on Oʻahu, where coot movements were typically to nearby wetlands within the complex (Paxton et al. 2022), we expected negative correlations in abundance between sites.

Results

Home Range

We captured 49 gallinules and 19 coots; of these, 15 gallinules and 17 coots received satellite GPS transmitters. Molecular sexing results indicated a roughly even sex ratio for adult (16M,19F, 5U) and juvenile (4M:5F) captured gallinules. Additionally, Hawaiian coots had an even sex ratio for adults (6M:7F) and juveniles (3M:3F). Of all the birds fixed with transmitters, 16 coots and 10 gallinules were used in analysis. Birds with transmitter failure were excluded from the analysis, and included those with zero transmission from mechanical failure (n=3), those that fell off the bird (n=1), and/or had too few GPS locations to calculate home range size (n=2). A total of 13,716 GPS points were transmitted (from 1/12/2023 to 4/22/2025), with an average of 355.3±253.2 points per coot and 730.1±645.7 points per gallinule. Transmitter durations were between 25-340 days for coots and 24-539 days for gallinules, with an average of 156.3±92.4 monitoring days and 267.4±162.2 days, respectively, for coots and gallinules.

Of the 49 color-banded gallinules, we resighted 30 unique individuals at MCBH (n=297 total resights) and 9 at Hamakua (n=17 total resights); 95% of individuals were resighted exclusively at their home wetland where they were banded.

For monthly home ranges, ≥25 locations were necessary to reach a stable size estimate of KDE for both coots and gallinules. This threshold allowed us to calculate Hawaiian coot home ranges of 16 individuals (10 MCBH (Pa'akai, Waipuna, and Kaluapuhi), and 6 Hāmākua) for 52 wet season months (30 male, 22 female) and 28 dry season months (18 male, 10 female) (Table A1). Monthly KDEs were calculated with an average of 57.4±11.9 GPS points. Hawaiian coot home ranges (Figure 3) and core ranges (Figure A1) appeared slightly larger for both males and females in the wet season. Hawaiian coot home ranges averaged 53,171.6 ± 58,063.5 m^2, and core ranges averaged 10,208.3± 12,648.8 m^2.

Ten Hawaiian gallinules from MCBH were included in the analysis, with four females and four males in the wet season and four females and six males in the dry season for monthly KDE (Table A2). A total of 39 wet season months (20 male, 19 female) and 41 dry season months (16 male, 25 female) were analyzed (Table A2). Monthly KDEs were calculated with an average of 73.3± 42.6 GPS points. Sites included in the analysis were all at MCBH at Pa'akai, Waipuna, Klipper, Sag Harbor, TLF, and Perc Ditch. Hawaiian gallinule home ranges (Figure 4) and core ranges (Figure A2) were similar for males and females across seasons. Hawaiian gallinule home ranges averaged 10,156.8± 6,408.4 m^2, and core ranges averaged 1,734.7± 1,198.5 m^2.

Monthly home and core ranges of Hawaiian coots analyzed by our linear mixed-effects model showed some variation by site and little variation by season or sex. There was a strong effect of the site on KDE: Kaluapuhi (95%: $\hat{\beta}$=1.787±0.735, t=2.430; 50%:$\hat{\beta}$=1.875±0.794, t=2.361) and Pa'akai

(95%:$\hat{\beta}$=1.419± 0.472, t=3.003; 50%:$\hat{\beta}$ =1.432±0.512, t=2.799) coots had significantly larger home and core ranges compared to those at Hāmākua. Home ranges of coots at Pa'akai tracked within the same time period were overlapping and substantially larger than those of gallinule (Figure 5). Coots at Waipuna had the smallest home ranges (95%: $\hat{\beta}$=-0.755±0.672, t=-1.123), but not statistically significantly smaller core ranges than Hāmākua (50%: $\hat{\beta}$=-0.667±0.733, t=-0.909). Our results suggest that wet season home ranges were somewhat larger than dry season home ranges, but this effect was not apparent for core ranges (95%: $\hat{\beta}$=0.544±0.541, t=1.006; 50%: $\hat{\beta}$=0.442±0.582, t=0.760). Example home ranges of a Hawaiian coot at Waipuna in the wet (February) and dry (May) seasons are shown (Figure 6); home range and core range are noticeably smaller in the dry season. Male Hawaiian coot home ranges were marginally larger than females' (95%: $\hat{\beta}$=0.691±0.638, t=1.082), but core ranges did not differ between the sexes (50%: $\hat{\beta}$=0.688±0.689, t=0.999). The interaction between sex and season was not a significant effect when comparing male home ranges in the wet season (95%: $\hat{\beta}$=-0.440±0.639, t=-0.688, 50%: $\hat{\beta}$=-0.366±0.686, t=-0.533).

Hawaiian gallinule monthly home and core ranges were similar across season and sex. We found that sex had no effect on home ranges and core ranges (95%: $\hat{\beta}$ =-0.120±0.301, t=-0.400; 50%: $\hat{\beta}$ =-0.050±0.308, t=-0.161). Additionally, we found wet season home and core ranges were not smaller than dry season KDE (95%: $\hat{\beta}$=0.127±0.175, t=0.725; 50%: $\hat{\beta}$=0.042±0.174, t=0.239). Though there was no difference in home range size between seasons, at least one individual appeared to shift its space use (Figure 7), using more woody wetland area in the dry season and more emergent wetland area in the wet season. Average Hawaiian coot home ranges were fivefold larger than average Hawaiian gallinule home ranges (Figure 5). There was no significant interaction between sex and season for home range size, though ranges were somewhat larger for males in the wet season (95%: $\hat{\beta}$=0.174±0.259, t=0.671; 50%: $\hat{\beta}$=0.262±0.258, t=1.016).

Long Distance Movements

Of the 17 Hawaiian coots fitted with a transmitter, nearly a quarter (24%) exhibited large-scale movements. Three adult coots tagged at Pa'akai and one tagged at Kaluapuhi made long-distance trips across Oʻahu and to neighboring islands (Figure 8, Appendix A1). Timing of movements varied among individuals, with departures from home wetlands in May and November of 2024, and February and June of 2025 (Appendix A1). All Hawaiian coots that made long-distance movements were female. The longest Euclidean distance traveled between fixes was 251km from Pa'akai to Niʻihau. The shortest Euclidean distance traveled between fixes was 35.93km from Pa'akai to James Campbell Wildlife Refuge. Stopover sites included federal and state wildlife refuges like James Campbell Wildlife Refuge, Mānā Plains Forest Reserve, Kawainui Marsh Wildlife Sanctuary, and Pacific Missile Range Facility. Public or private stopover sites also included streams/drainage ditches, agricultural land, golf course ponds, residential areas, and coastal wetlands. Two of the coots (#261781 and #261787) remained at their newly explored wetland for months on Niʻihau and Kauaʻi. Only one Hawaiian coot (#261781) attempted to return to Oʻahu after relocating to northwest Niʻihau for 4.5 months. She returned in September of 2024, but transmissions ended after 8 days on the North Shore of Oʻahu at a Turtle Bay Golf Course pond.

In contrast, 10% of transmittered Hawaiian gallinules in our study exhibited long-distance movements. Adult female Hawaiian gallinule BG:ZA (transmitter#: 239070) had one large scale movement from 5/6/2023-5/10/2023 within Kailua (Figure 9, Appendix A2). During the large-scale movement, stopover sites included residential areas, Kawainui Marsh Wildlife Sanctuary, and wetlands Waipuna and Perc Ditch (MCBH). She returned to her home wetland TLF and traveled a total linear distance of 13.13 km within a five-day span. Additionally, we detected several inter-site movements through band resights. Paʻakai gallinule KY:ZA, a hatch-year female, was found at Klipper Golf Course in June, two months after being banded, and remained near Koloa Pond at Klipper for several weeks. A Hāmākua gallinule LY:OA banded by DLNR was found at TLF on 1/27/25 and later seen on camera nesting at TLF on 3/27/25 with gallinule BG:ZA (female); therefore, we can assume that LY:OA is a male.

Abundance and Environmental Variables

Waterbird surveys at 10 different sites yielded variable population counts throughout the year. A total of 556 waterbird surveys were conducted during the study. Pa'akai hosted the largest number of coots at MCBH ranging from 6-104 coots, with an average of 26.5±2.5 coots. Seasonally, coot populations increased during the wet season (November-April) (Figure 10). Klipper golf course had the largest proportion of gallinules at MCBH, ranging from 0-12 gallinules, with an average of 6.4±0.4 throughout the study (Figure 11). The adult gallinule population at Klipper declined between 2023 and 2025. Hāmākua had a stable coot and gallinule population over the study period (average of 22.2±0.9 coots and 28.5±1.2 gallinules). There were stable populations of gallinules at Kaʻelepulu and fluctuations in coot abundance, especially in the wet season (average:14.9±0.8, 58.1±2.5, respectively). Changes in coot population at Hāmākua were largely independent from those at Kaʻelepulu (r=0.344) and Pa'akai (r=-0.379). By contrast, changes in coot numbers at Kaʻelepulu and Pa'akai were more synchronous (r=0.857).

Water quality and water level data varied between sites and seasons at MCBH and influenced waterbird populations. All water variables tested appeared to influence coot numbers. Log-rainfall ($\hat{\beta}$ =-0.263±0.038, z=-7.152, p<0.001), dissolved oxygen ($\hat{\beta}$=-0.395±0.044, z=-10.499, p<0.001), and turbidity ($\hat{\beta}$ =-0.206±0.033, z=-8.445, p<0.001) each had negative effects on coot abundance (Figure 12). By contrast, pH ($\hat{\beta}$=0.224±0.045, z=6.091, p<0.001), salinity ($\hat{\beta}$ =0.288±0.022, z=12.392, p<0.001, and water level ($\hat{\beta}$=0.148±0.026, z=5.722, p<0.001) each had positive effects on coot abundance (Figure 12). Water quality (log-turbidity ($\hat{\beta}$=0.259±0.139, z=1.863, p=0.063), dissolved oxygen ($\hat{\beta}$ =0.037±0.110, z=0.341, p=0.733), salinity ($\hat{\beta}$=-0.160±0.096, z=-1.663, p=0.096), and pH ($\hat{\beta}$ =-0.034±0.135, z=-0.255, p=0.799)), water level ($\hat{\beta}$=-0.003±0.148, z=-0.019, p=0.985), and log-rainfall ($\hat{\beta}$=0.151±0.120, z=1.260, p=0.208) had no significant effects on gallinule populations at Klipper (Figure 13), despite population decreases over the study period (Figure 11).

Discussion

Fine scale tracking and observations of abundance over time of Hawaiian gallinules confirms previous evidence of limited movements for the species (Paxton et al. 2022, van Rees and Reed 2018). In contrast, Hawaiian coots displayed some population-level seasonal movements related to environmental conditions, though not as synchronized as in continental migratory species (Eddleman et al. 1985), and individual tracking revealed no consistent seasonal patterns. Hawaiian coot abundance increased synchronously at the two largest wetlands monitored, suggesting immigration from areas outside the Kailua wetland complex. Water level, water quality, and recent rainfall all significantly influenced coot abundance.

Synchronous changes in the abundance of Hawaiian coots in Kailua may indicate immigration from wetland sites outside of the local area and seasonality in abundance. Previous VHF studies of Hawaiian Coots in the Pearl Harbor Wetland complex detected local movements in 30% of individuals (Paxton et al. 2022); therefore, we predicted that abundance would have an inverse relationship and that coots would move between wetlands within the Kailua wetland complex. Coot populations at Pa'akai and Hāmākua had a weak inverse relationship (r=-0.379), consistent with some movement between sites. However, populations at Pa'akai and Ka'elepulu were strongly correlated (r =0.857), indicating immigration from other wetland populations outside of Kailua. Synchronous population increases at these sites during the wet season may reflect improved habitat quality and suggest seasonal abundance patterns, and that Pa'akai and Ka'elepulu function as secondary or peripheral wetlands. Supporting this interpretation, the large population influx at the end of 2024 at Pa'akai was not followed by increased nesting rates (L. S. Roerk, unpublished data). Previous rapid changes in coot abundance on O'ahu have been observed at MCBH (L. Bookless, personal communication) and farther west (Honouliuli, K. Goodale, personal communication) and are largely undocumented or explained. Nomadic species may move in response to resources that are sporadic in time and distribution (IUCN 2025). Some individuals captured at MCBH may not have been residents but using the habitat seasonally perhaps in response to favorable local conditions, or unfavorable ones elsewhere. Nomadicism has been documented in other waterbird species and in Virginia rails (*Rallus limicola)* in arid and semi-arid environments, with irruptive congregations in ephemeral wetlands (Takekawa et al. 2015b, Conway and Pyle 2024, McGinness et al. 2024). Predicting population influxes can be difficult, but communication between wetland managers could piece together Hawaiian coot intra-island (and possibly inter-island) wetland connection. If Hawaiian coots are a nomadic species, it will be difficult to predict movements of flocks, especially as Hawaii's weather becomes more variable;

conservation must reach beyond the site scale alone. Conversely, populations had more stability at Hāmākua and a weaker correlation to population changes within Pa'akai and Kaʻelepulu, which could be due to Hāmākua’s year-round habitat suitability from DOFAW’s rigorous habitat management. Hāmākua is identified as a core site (USFWS 2011), with greater annual nesting activity (Works et al. 2024) than Pa'akai (Chapter 2). Increased nesting propensity at Hāmākua could be associated with increased territoriality by nesting individuals, excluding nomadic individuals. Long-distance movements of rails are energetically expensive (Rush et al. 2020); therefore, preserving wetland connectivity by providing suitable habitat and protection from predators may aid in the safe passage of these species.

Coot abundance at Pa'akai was significantly affected by water level, rainfall, and water quality, while gallinule abundance at Klipper was not related to changes in water and weather conditions. We expected low water levels, increased rainfall, and decreased water quality due to evaporative loss to negatively affect coot abundance, but coot abundance was higher during periods with higher water levels. Changes in freshwater input and salinity can alter food resource abundance by affecting the chemical tolerance of aquatic prey species for waterbirds (Haig et al. 2019). Pa'akai is a relatively shallow coastal wetland with rainwater and tidal hydrological influence (Cox and Jokiel 1997); intermediate water depths can provide Hawaiian coots with hospitable foraging depths and abundant aquatic invertebrates. Anthropogenic water drawdowns and climate change related evaporative loss in Hawaii could reduce local coot populations. In the Great Lakes region, increased vegetation interspersion and higher macroinvertebrate abundance associated with higher lake levels positively impacted marsh-obligate and marsh-facultative species richness and greatly increased American coot (*Fulica americana)* and common gallinule (*Gallinula galeata)* population abundance (Hohman et al. 2021). Conversely, Hawaiian waterbird abundance was negatively correlated with rainfall and higher water levels, reflecting decreased habitat conditions (Mayer and Pilson 2019), and Hawaiian coots were predicted to utilize ephemeral wetlands during heavy rains (Engilis and Pratt 1993). Consistent with these findings, recent rainfall negatively affected coot abundance, and coots at Pa'akai may be dispersing to ephemeral wetlands, perhaps in search of better foraging habitat. As climate change increases precipitation variability, populations at wetland sites may exhibit greater seasonal variability.

Changes to water chemistry of coastal wetlands via evaporative loss may directly affect waterbird abundance and indirectly affect habitat suitability by changing plant communities and food abundance (Longcore et al. 2006, Mayer and Pilson 2019, Donnelly et al. 2022). Water quality measures dissolved oxygen, salinity, turbidity, and pH all had significant effects on coot abundance. Salinity had a positive effect on coot abundance, but this effect could be influenced by the population influx in 2024 during higher-than-average salinity levels. Coots are thought to have a limited salinity tolerance and prefer freshwater habitats (Byrd et al. 1985), but at the coastal wetland Kealia Pond in Maui, Hawaiian coot abundance increased with lower water levels associated with higher salinity and turbidity and decreased dissolved oxygen due to invertebrate midge abundance (Mayer and Pilson 2019). We had the opposite outcome, in that increased water level increased coot abundance; invertebrate diversity and abundance may differ between coastal wetland sites in Hawaii. Changes in water quality can have complex effects on coastal wetland environments, drastically shifting trophic cascades (Takekawa et al. 2015a). Typically, evaporative water loss in coastal wetland systems can increase salinity and turbidity and decrease dissolved oxygen, but we see conflicting results: increased water level and increased salinity positively affected coot abundance. We are unsure if the direct effects (e.g., water chemistry tolerance, changes to open water habitat area) or indirect effects (e.g., changes to plant communities and invertebrate abundance) of water level, recent rainfall, and water quality are drivers of coot abundance. Knowing more about the Hawaiian coot's diet and seasonal changes in prey availability would help understand these conflicting effects of water composition and availability.

Throughout the study, gallinule populations appeared consistent despite changes in water level and water quality, except for those at Klipper. Hawaiian gallinules seem to have a low propensity to movement and did not seem to relocate despite changes in water availability and quality at their home wetland. Population changes at Klipper were due to other factors unrelated to water-related environmental conditions. Klipper's adult population declined from 2023 -2025, likely due to golf course hazards.

Through the duration of the study six gallinule carcasses were recovered from Klipper, and others were found injured. One carcass was autopsied, and the individual died from blunt force trauma to the head, likely from a golf ball. All carcasses were either found in front of the tee box or on the fairway, suggesting additional golf-ball strikes. Golf courses can provide habitat for waterbirds, but action must be taken when recreation threatens waterbirds protected by the Endangered Species Act (ESA) and Migratory Bird Treaty Act (MBTA). Changes to golf course structure by creating vegetation buffers, moving tee boxes away from wetland bird habitats, or shutting down holes completely, can prevent take or injury of waterbirds. Bridging the gap between waterbird conservation and recreational management could mitigate take of federally endangered species when their habitats conflict with highly trafficked recreational areas. Further exploring Hawaiian coot and gallinule diet, habitat preferences, and how movement is utilized may aid in understanding factors affecting abundance.

Through satellite GPS transmitter monitoring, we found that Hawaiian coots had a higher propensity to move than Hawaiian gallinules, but individual movements did not appear seasonal. Coots that moved off their home wetland traveled far intra-island and inter-island distances between fixes (8-24 hours), including the longest flight distance (251 km) of a Hawaiian coot on record (Figure 8). Riggs (2016) documented Hawaiian coot movement through band resights and reported three inter-island movements to Kauaʻi and Molokaʻi among 60 banded individuals, while three coots in our study left Oʻahu. Only one coot attempted to return to Oʻahu after its inter-island movement, compared to other studies with consistent returns after local movements (Paxton et al. 2022). Transmitter failure or mortalities may have prevented us from seeing additional returns that could have suggested seasonal migration. Notably, all movements were made by females. Sexual segregation in terms of habitat use is poorly documented in rails, but male king rails (*Rallus elegans*) in North Carolina, USA, remained in their breeding habitats overwinter, while females showed somewhat wider movements after the breeding season (Kolts and McRae 2017). As for changes in coot abundance, habitat suitability may influence individual movement. Hawaiian coots were tracked at Hāmākua (n=6) for up to 10 months, and showed no movement, perhaps because the site offers year-round habitat suitability and supports substantial nesting (Works et al. 2024).

Contrarily, all movement departures were from Pa'akai and Kaulapuhi, and a proportion were captured during the large population influx at the end of 2024 into 2025. Nomadic migrants may be pushed out of unfavorable habitats, while favorable habitats may attract them elsewhere, leading to irruptive movements of birds that depend on climate and food availability (Strong et al. 2015). Pa'akai and Ka‘elepulu could be considered secondary sites that can host nomadic coot populations with a stronger propensity to movement when conditions are favorable, but these sites are not used as primary wetlands for nesting. A variety of habitats were utilized as stopover sites and temporary home ranges for coots and gallinules, including federal and state wildlife refuges, public or private streams/drainage ditches, agricultural land, golf course ponds, residential areas, and coastal wetlands. Previously unknown stopover sites and the use of federal and state wildlife refuges serve as important habitats for waterbird movement corridors. Improving stopover sites through habitat restoration and predator control, alongside increasing habitat connectivity, may aid waterbird survival during movement bouts.

Other Hawaiian waterbird movement studies have recorded very little to no movement of Hawaiian gallinule (van Rees et al. 2018a, van Rees et al. 2018b, Paxton et al. 2022), and our results reflect those patterns. Only one gallinule movement was captured via transmitter tracking; TLF gallinule BG:ZA traveled in a loop over Kailua wetlands, perhaps in an attempt to prospect for alternate habitat (Figure 9). The water level dropped 21.3 cm the week prior to departure, and the journey occurred between nesting bouts. Juvenile dispersals, such as the one observed from Pa'akai to Klipper, may function to establish new territories, while adult forays may result from limited suitable habitat and efforts to conduct habitat reconnaissance. As their use of movement appears low, Hawaiian gallinules may require more local or site-specific conservation measures, especially in areas threatened by sea-level rise.

Both Hawaiian coots and gallinules had strong site fidelity, and little effects of season, sex, and site on Hawaiian coot home range, and no effects on Hawaiian gallinule home range. Originally, we predicted that home ranges would be larger in the dry season due to decreased food availability. Additionally, home ranges may be constricted during the wet season due to increased territoriality and nest incubation (Zurell et al. 2018, Doyle et al. 2023). But Hawaiian gallinule breeding was sporadic and aseasonal (Chapter 2) and gallinules exhibited high territoriality year-round due to little overlap in home ranges (Figure A3); therefore, season did not affect home range. Habitats may shift seasonally with resource availability; woody wetland areas were used in the dry season, while typical emergent wetland habitat was used in the wet season. Emergent wetland expansion may increase hospitable habitat, but secondary habitats, such as woody wetland areas, may be critical when increased evaporative loss from climate change reduces emergent wetland extent. Site-specific conservation is important for gallinules that have constricted home ranges, given their lower movement propensity, even under unfavorable conditions.

Wet season KDE was larger than the dry season for coots, which may be from the shift in early dry season nesting and the large non-breeding population. Of the few nests that we had, the majority of nests were initiated early in the dry season (L. S. Roerk, unpublished data). Many of the coots were tagged at Pa'akai and Kaluapuhi during the wet season in 2024 due to a large population influx. Home ranges of coots monitored during the same period overlapped, were five times larger than those of gallinules, and utilized more open water habitat; signifying a non-breeding population due to flocking behavior. Flocking behavior can increase home range size but also contributes to foraging efficiency, increased protection from predators, and increased vigilance among flock members (Darrah and Smith 2014, Ofstad et al. 2016). A similar study compared the seasonal movements of red-capped larks (*Calandrella cinerea*) and found that non-breeding individuals' home ranges were larger due to flocking behavior (Mwangi et al., 2019). This flocking behavior and lack of breeding suggest that Pa'akai hosts a non-breeding population (L. S. Roerk, unpublished data), implying that Pa'akai may serve as a secondary stopover site or loafing grounds rather than a primary breeding site. Conversely, coots at Hamakua appeared to show site fidelity to the basin in which they were captured. DOFAW’s rigorous vegetation management and predator control, paired with water availability at Hāmākua, can offer year-round habitat suitability with high plant and habitat diversity for nesting. A larger breeding population, driven by higher nesting rates at Hāmākua, led to less overlap in home ranges (Works et al. 2024). Tagging more breeding individuals over longer periods may reveal different patterns in Hawaiian coot seasonal home range, if any. Understanding these patterns can guide wetland management to maintain suitable breeding habitat throughout the year, ensuring that water levels and vegetation structure support both nesting and foraging needs.

As climate change intensifies, shifts in weather and water availability will affect habitat quality, and waterbirds will likely face challenges finding suitable habitat (Dai 2011, van Rees and Reed 2018, Donnelly et al. 2022). It is difficult to predict future adaptations of Hawaiian rails, but information on space use, movement, and population abundance as they relate to weather and water availability can be useful for conservation efforts presently. Hawaiian coots have more mobility than gallinules in terms of searching for suitable habitat, as weather and water composition have significant effects on coot abundance. Movements of Hawaiian coots were significant distances within O‘ahu and to neighboring islands. Improving habitat connectivity and conserving stopover sites can aid in the safe passage of moving birds. Communication statewide among wetland managers can help understand irruptive movements of Hawaiian coot flocks and piece together intra-island and inter-island wetland movement networks. Future studies could implement increased tagging of coots in a variety of managed vs unmanaged wetland sites to help identify movement patterns. Partially sedentary gallinules require more site-specific conservation measures, especially in threatened coastal wetlands. Some information on diet is available for the Hawaiian gallinule (DesRochers et al. 2009, DesRochers et al. 2010), but diet metabarcoding of both species can help identify habitat requirements across all life stages. The lack of seasonality in Hawaii may become more variable with fluctuations in temperature and rainfall as climate change progresses (Hawai‘i Climate Change Mitigation and Adaptation Commission 2021). Sex and season had little to no effect on home range size, but home ranges can provide insight into a wetland's carrying capacity and seasonal shifts in space use. Localized water allocation, where water redirection is available, may provide year-round habitat suitability, especially for Hawaiian gallinules, which are less mobile than coots. Areas at risk of coastal inundation threaten the extirpation of Hawaiian waterbirds (van Rees and Reed 2018). Translocation of the Hawaiian gallinule to a third island may help combat habitat loss related population decline (VanderWerf 2024). As sea-level rise and climate change reduce available habitat for Hawaiian waterbirds, dispersal may be critical for population persistence, while dedicated conservation efforts are vital to slowing habitat loss and preventing local extirpation.

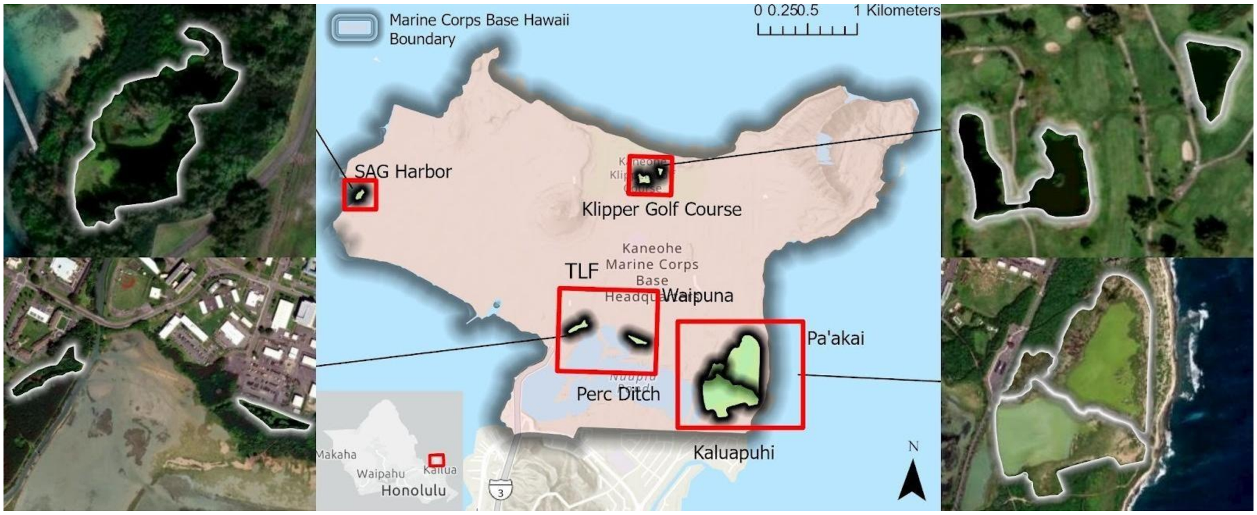

Figure 1. Study area map of Marine Corps Base Hawai‘i (MCBH), Kāne‘ohe Bay, O‘ahu, Hawai‘i, USA. Black shadowed borders indicate MCBH boundary, red boxes indicate locations of 7 wetland study sites, left and right maps indicate wetland extent with satellite imagery. Wetlands include (clockwise from left) Sag Harbor, Klipper Golf Course, Pa‘akai, Kaluapuhi, Waipuna, Percolation Ditch (“Perc Ditch”), and Temporary Lodging Facility (TLF) stream.

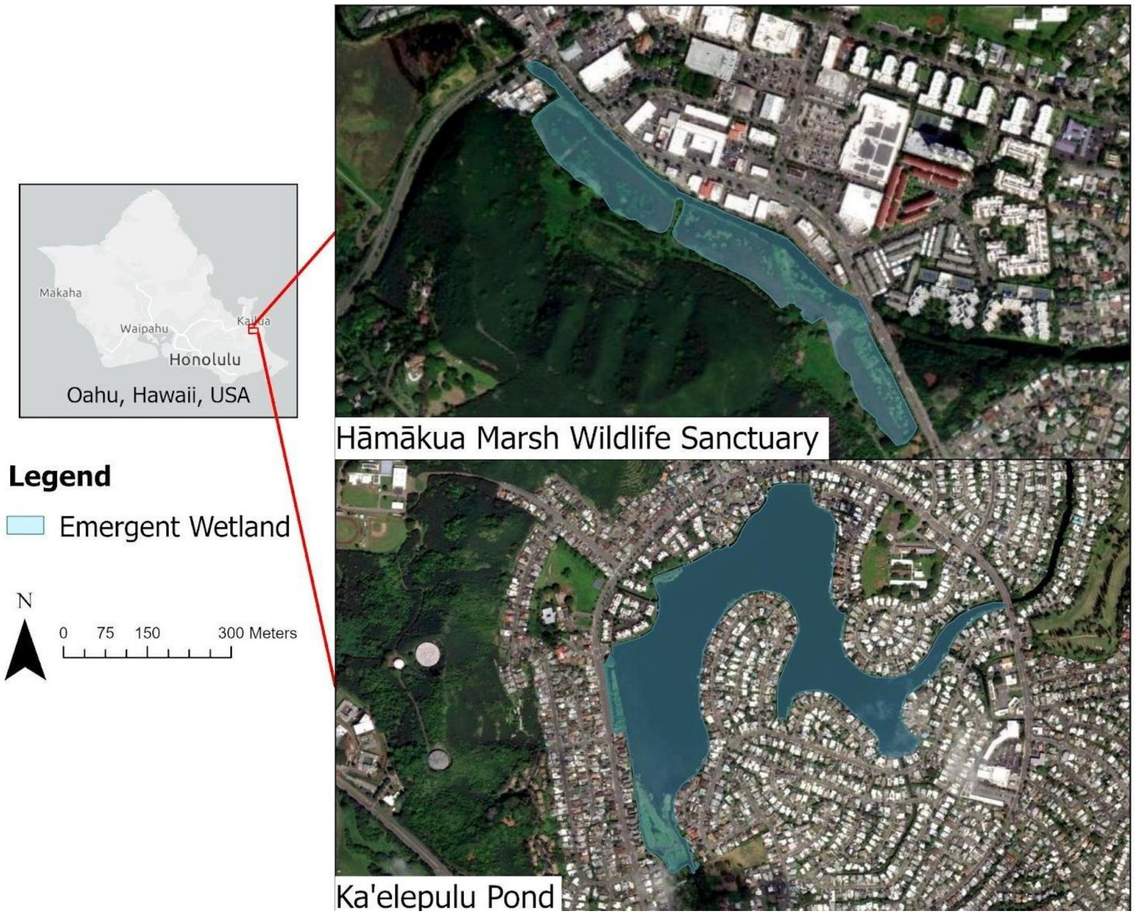

Figure 2: Map of Hāmākua Marsh Wildlife Refuge and Ka‘elepulu Wetland where we conducted additional waterbird surveys within the Kailua wetland complex. Hāmākua was a secondary site for tagging Hawaiian coots.

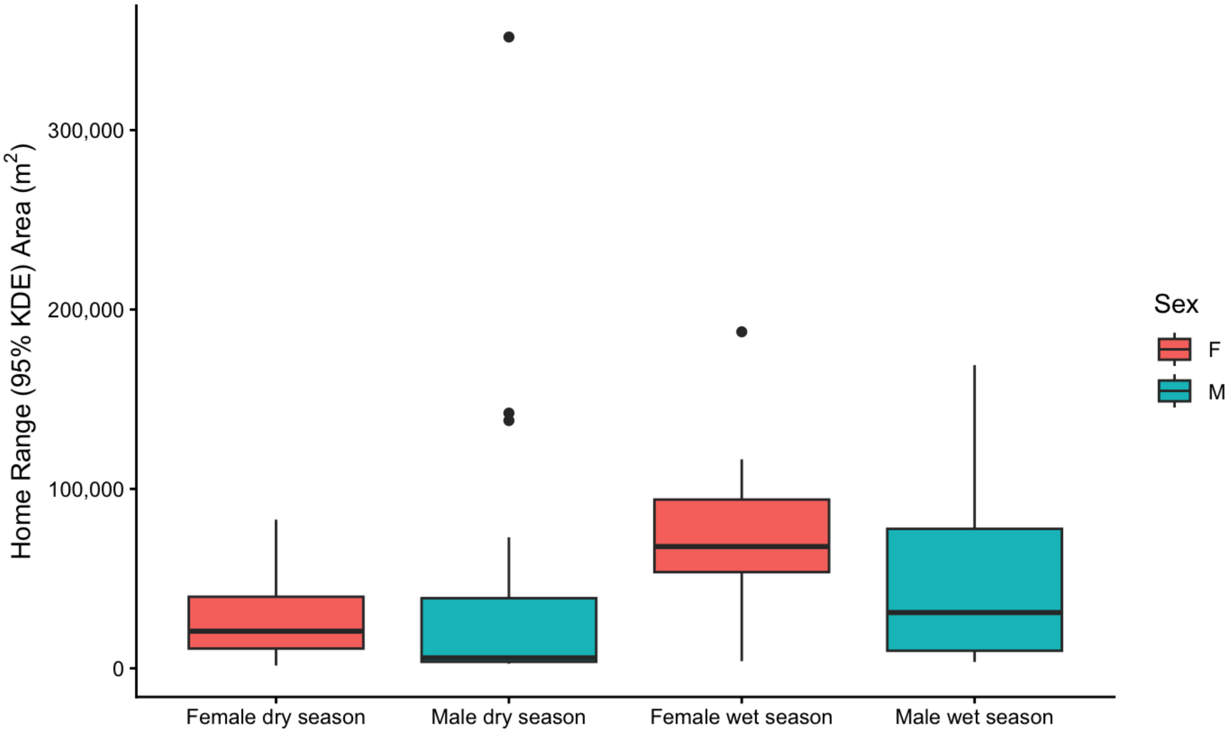

Figure 3: Summary boxplot of Hawaiian coot home ranges (95% KDE) in m^2 across seasons (wet and dry season) and sex. Hawaiian coot home range sites include Pa'akai (MCBH), Kaluapuhi (MCBH), and Hāmākua. The horizontal line within the box indicates the median, dots indicate outliers, and the bars represent the upper and lower quartiles within the box.

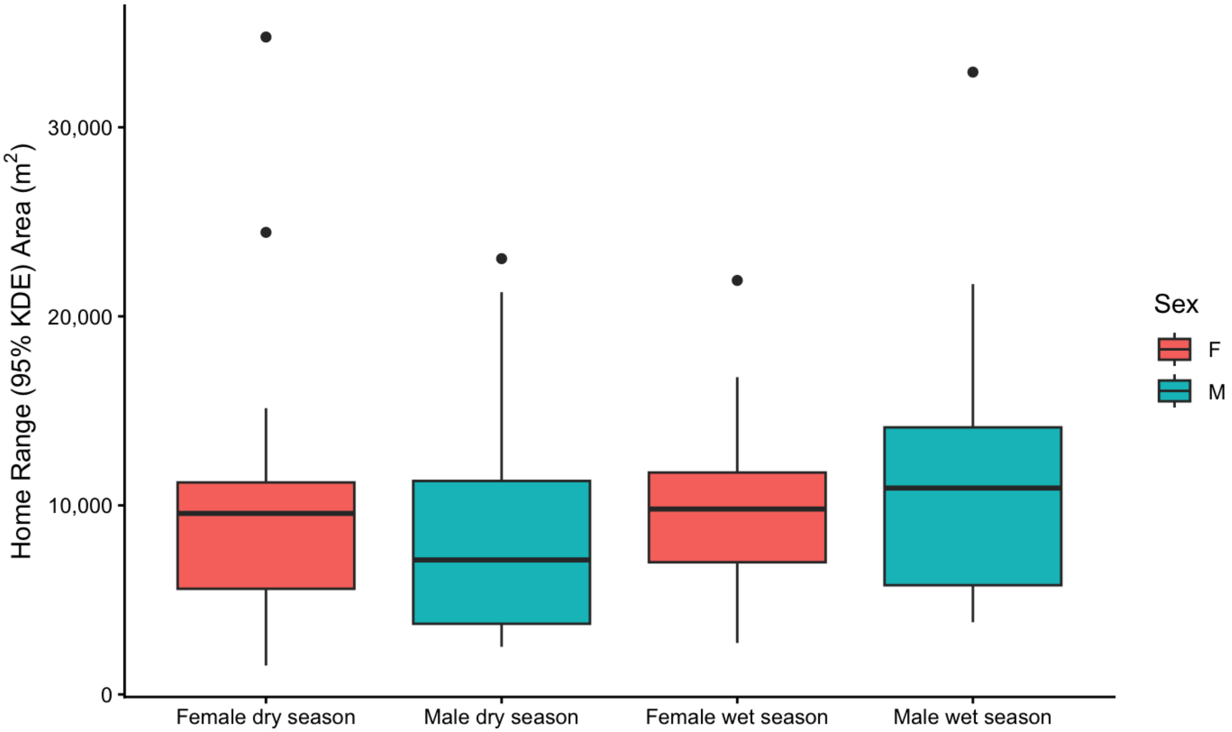

Figure 4: Summary of Hawaiian gallinule home ranges (95% KDE) in m^2 across seasons and sex. Hawaiian gallinule home ranges were all within MCBH, wetland sites include Pa'akai, Waipuna, Klipper, Sag Harbor, TLF, and Perc Ditch. The horizontal line within the box indicates the median, dots indicate outliers, and the bars represent the upper and lower quartiles within the box

Figure 5: Comparison of home ranges (95% KDE) between Hawaiian coots (n=3, solid border) and Hawaiian gallinule (n=1, dashed border) at Pa'akai (MCBH). Home range was estimated using GPS data from 03/23/2025-4/22/2025. Hawaiian gallinule home range is shown in pink with dashed border (#261778, 14253.7 m^2), distinctly using emergent wetland area. Whereas Hawaiian coot home ranges are overlapped across individuals, utilizing open water and emergent wetland, and significantly larger (#261788 (purple)=94,947.5 m^2, #261791 (green)=85,444.17 m^2, #261774 (blue)=47,587.8 m^2).

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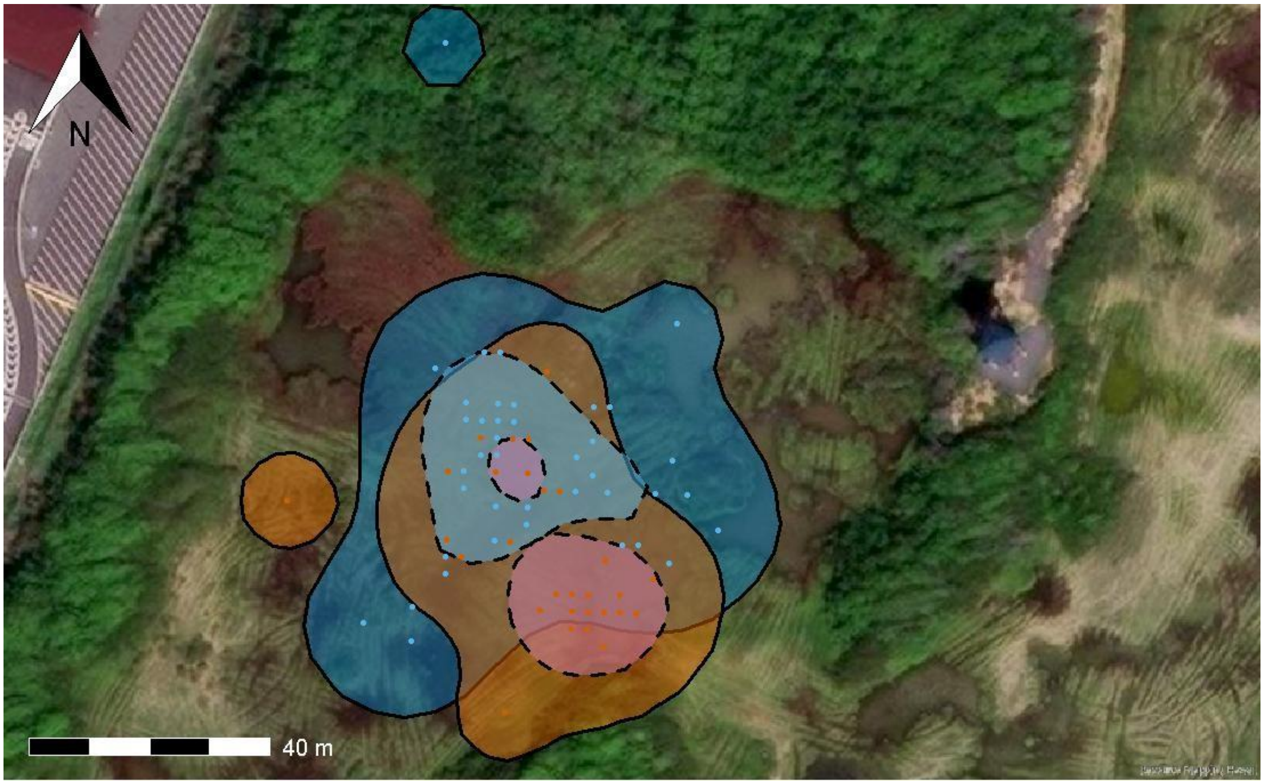


Figure 6: Monthly home range (95% KDE) and core range (50% KDE) of Waipuna coot #261780 in the wet and dry season. February wet season home range (4,071.4 m^2) is shown in blue, core range is shown in light blue with dashed border (890.6 m^2) with light blue GPS points. May dry season home range (2,868.2 m^2) is shown in orange, core range is shown in pink with dashed border (589.1 m^2) with orange GPS points.

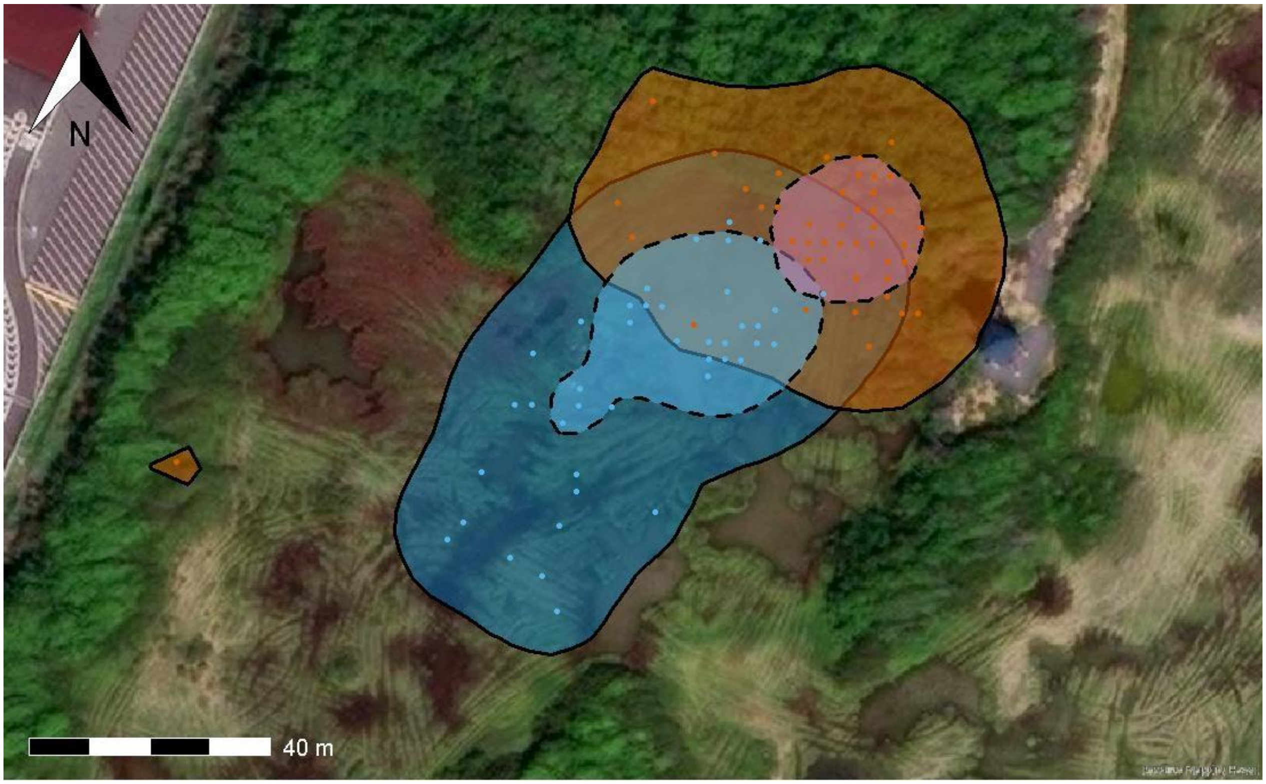


Figure 7: Monthly home range (95% KDE) and core range (50% KDE) of Waipuna gallinule #261771 in the wet and dry season. December wet season home range (4 ,154.5 m^2) is shown in blue, core range is shown in light blue with dashed border (1,012.6 m^2) with light blue GPS points. August dry season home range (3,131.3m^2) is shown in orange, core range is shown in pink with dashed border (499.2 m ^{2}) with orange GPS points.

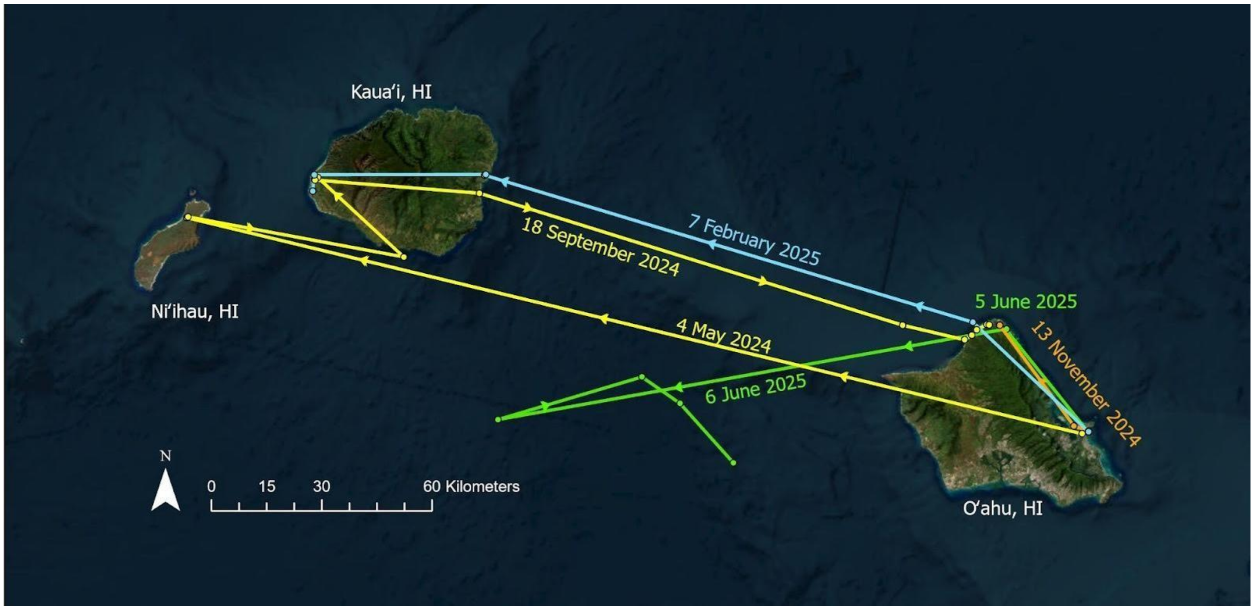


Figure 8: Map of long-distance movements of all Hawaiian coots that moved off their wetland boundary including dates of departure, color coded by individual. All individuals that moved were tagged either at Pa'akai or Kaluapuhi and were all female. Hawaiian coots ZA:GY (yellow), ZA:BB (orange), ZA:GO (light blue), and GG:ZA (green) flight paths are shown. Arrows indicate direction of movement. See Appendix A1 for flight distances of each individual.

Figure 9: Large scale movement of Hawaiian gallinule BG:ZA from TLF wetland at MCBH. First movement recorded off base was 5/6/2023 and she returned to TLF (green polygon) 5/10/2023. The total linear distance moved (i.e. sum of distances for each trip segment) was 13.13 km.

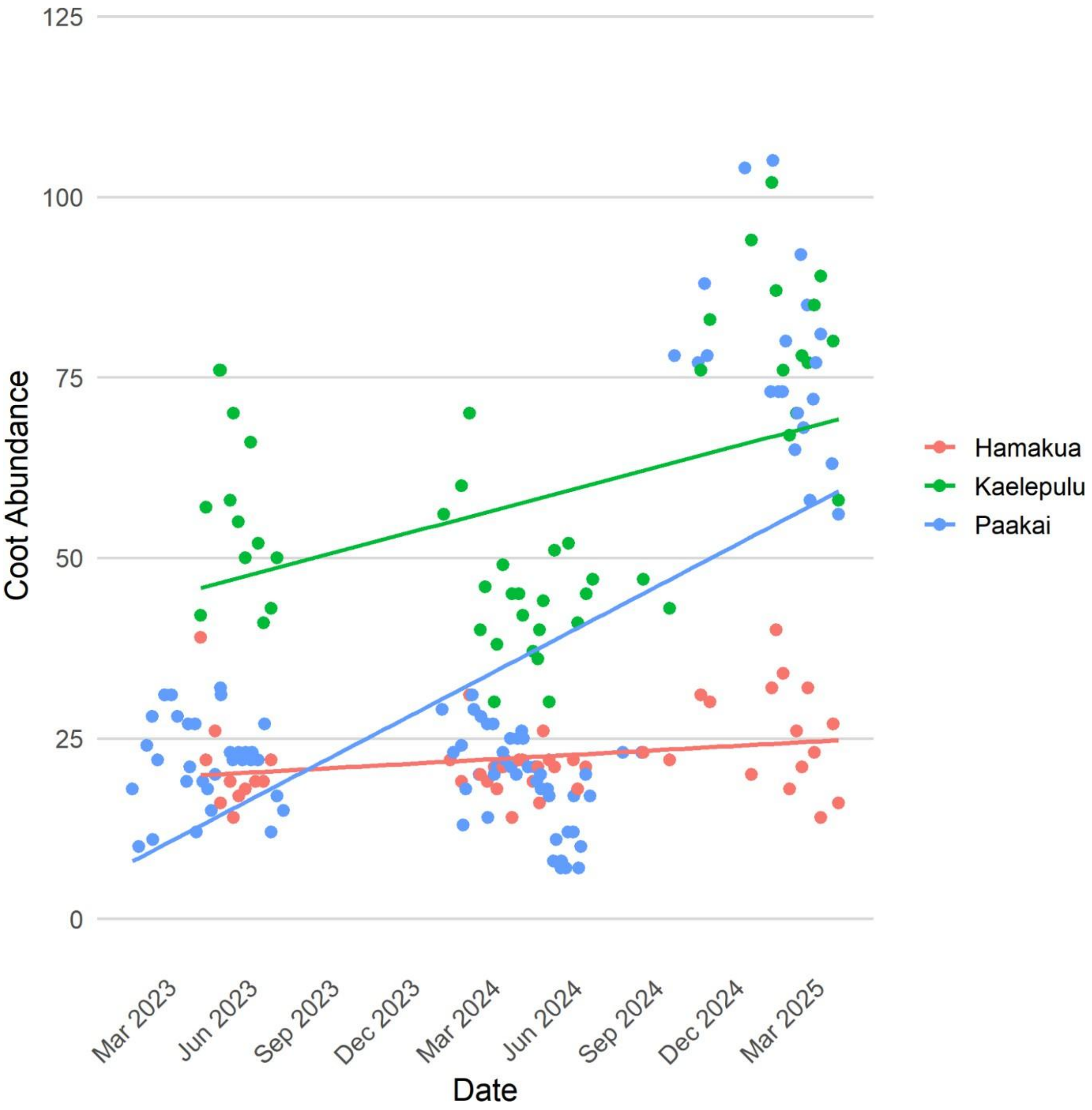


Figure 10: Hawaiian coot abundance over time at Hāmākua, Kaʻelepulu, and Pa'akai/Kaluapuhi. Surveys were conducted weekly from January-July of 2023-2024 and January-March of 2025. Off season surveys were conducted monthly from August-December of 2024.

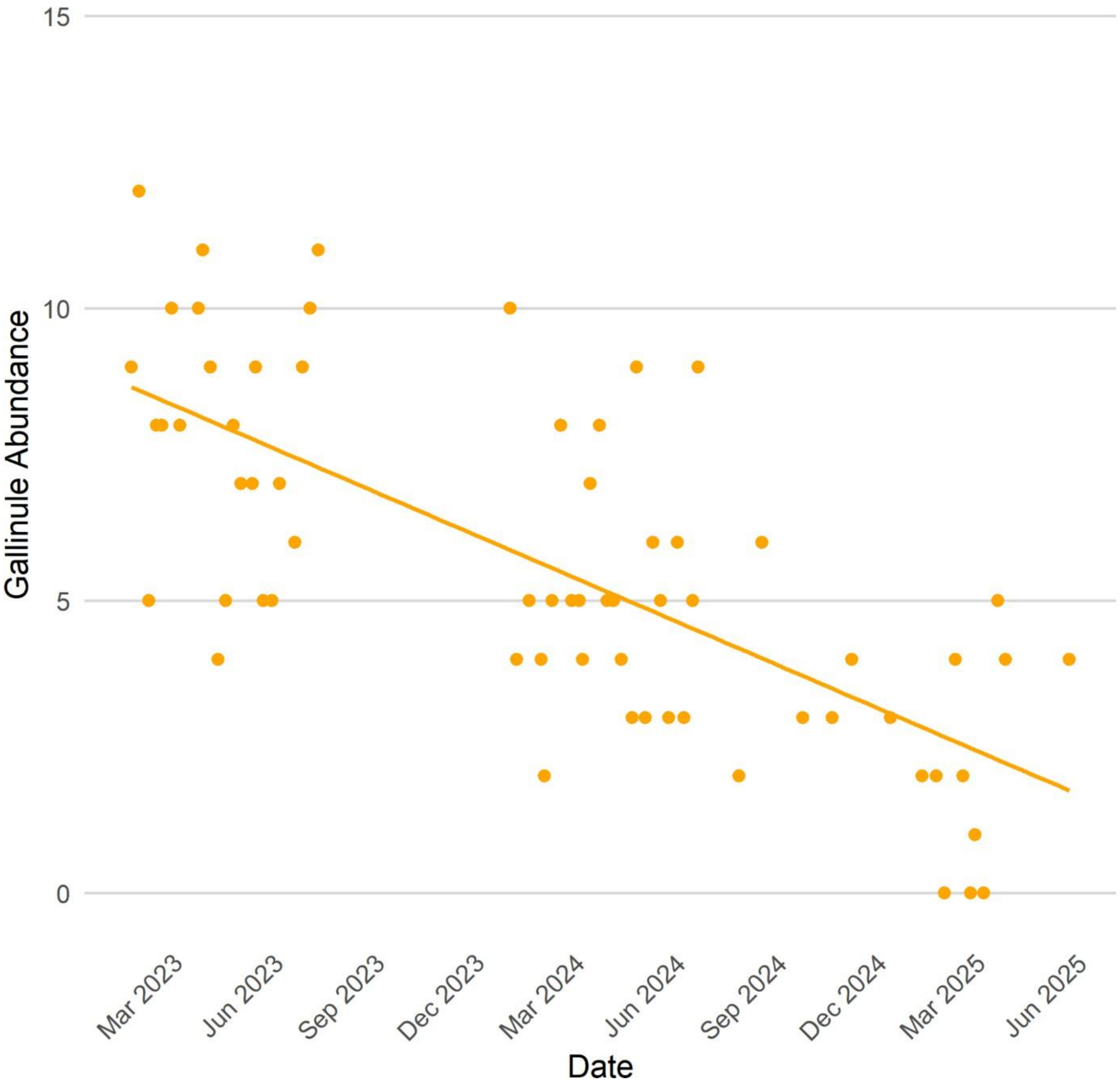


Figure 11: Hawaiian gallinule abundance over time at Klipper. Surveys were conducted weekly from January-July of 2023-2024 and January-March of 2025. Off season surveys were conducted monthly from August-December of 2024.

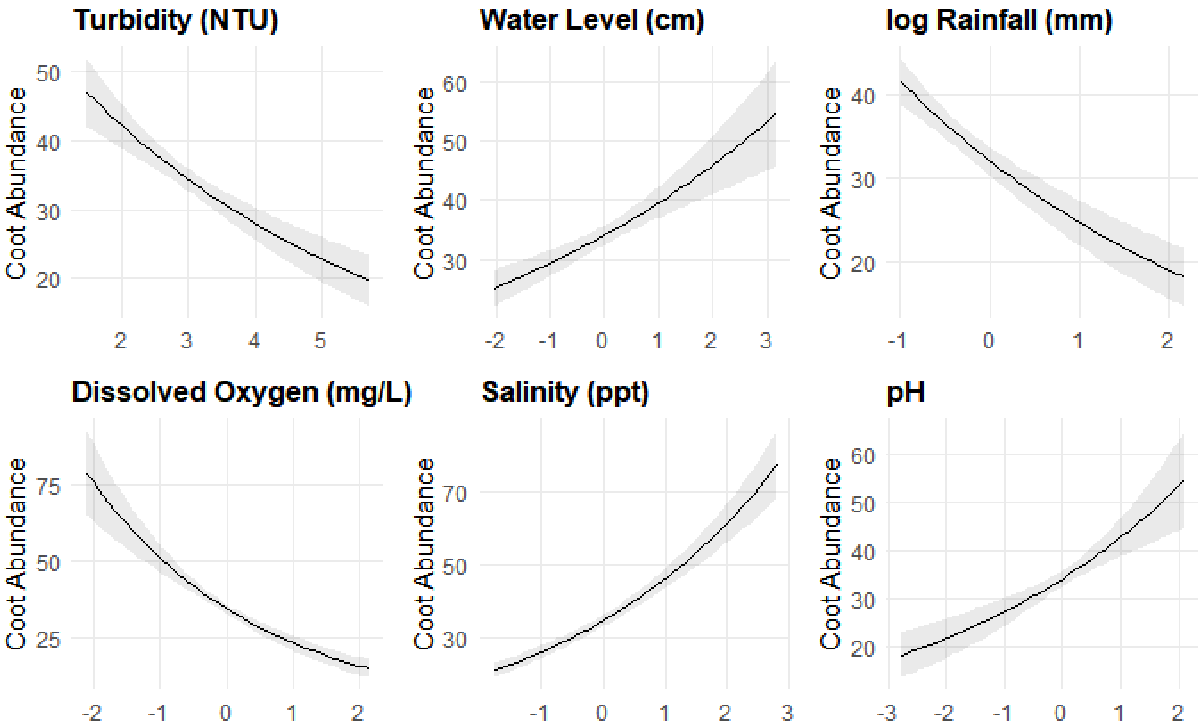


Figure 12: Prediction plots of the effect of turbidity, water level, log-rainfall, dissolved oxygen, salinity, and pH on coot abundance at Pa'akai from 2024-2025. Predictions were made from a generalized linear model, and all covariates were scaled.

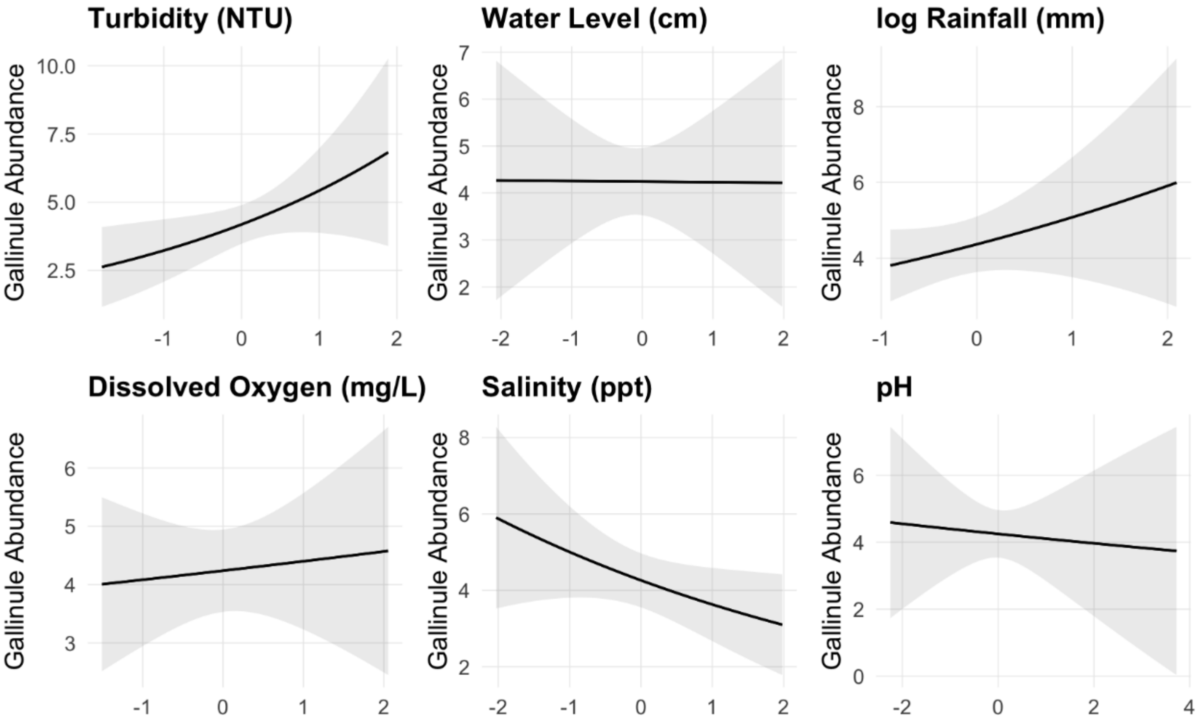


Figure 13: Prediction plots of the effect of turbidity, water level, log-rainfall, dissolved oxygen, salinity, and pH on gallinule abundance at Klipper from 2024-2025. Predictions were made from a generalized linear model, and all covariates were scaled.

References

Bluso-Demers, J., M. A. Colwell, J. Y. Takekawa, and J. T. Ackerman (2008). Space use by Forster's Terns breeding in south San Francisco Bay. *Waterbirds* 31:357–369.

Byrd, G. V., and C. F. Zeillemaker (1981). Ecology of nesting Hawaiian Common Gallinules at Hanalei, Hawaii. *Western Birds* 12:1–12.

Byrd, G. V., R. J. Shallenberger, and C. S. Arume (1985). Notes on the breeding biology of the Hawaiian race of the American Coot. *Elepaio* 45:1–12.

Chen, W., C. Cao, D. Liu, R. Tian, C. Wu, Y. Wang, Y. Qian, G. Ma, and D. Bao (2019). An evaluating system for wetland ecological health: Case study on nineteen major wetlands in Beijing-Tianjin-Hebei region, China. *Science of The Total Environment* 666:1080–1088.

Conway, C. J., W. R. Eddleman, S. H. Anderson, and L. R. Hanebury (1993). Seasonal changes in Yuma Clapper Rail vocalization rate and habitat Use. *The Journal of Wildlife Management* 57:282–290.

Conway, C. J., and P. Pyle (2024). Virginia Rail (*Rallus limicola*), version 1.1. In Birds of the World (A. F. Poole, F. B. Gill, and M. G. Smith, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi-org.ezproxy2.library.colostate.edu/10.2173/bow.virrai.01.1

Cox, E. F., and P. L. Jokiel (1997). Environmental study of Nu'upia Ponds Wildlife Management Area, Marine Corps Base Hawaii, Kaneohe Bay. Environmental Compliance and Protection Department, Environmental Affairs Division, Kaneohe Bay, HI, USA.

Dai, A. (2011). Drought under global warming: a review. *WIREs Climate Change* 2:45–65.

Darrah, A. J., and K. G. Smith (2014). Ecological and behavioral correlates of individual flocking propensity of a tropical songbird. *Behavioral Ecology* 25:1064–1072.

Davidson, N. C. (2014). How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research* 65:934-941.

Demers, S. A., M. A. Colwell, J. Y. Takekawa, and J. T. Ackerman (2008). Breeding stage influences space use of female American Avocets in San Francisco Bay, California. *Waterbirds* 31:365–371.

DesRochers, D. W., S. R. McWilliams, M. D. Silbernagle, and J. M. Reed (2009). Macronutrient profiles of wetland plants consumed by the Hawaiian Moorhen (*Gallinula chloropus sandvicensis*). *Wetlands* 29:845–853.

DesRochers, D. W., S. R. Mcwilliams, and J. M. Reed (2010). Evaluating if energy and protein limit abundance of Hawaiian Moorhen. *The Journal of Wildlife Management* 74:788–795.

Donnelly, J. P., J. N. Moore, M. L. Casazza, and S. P. Coons (2022). Functional wetland loss drives emerging risks to waterbird migration networks. *Frontiers in Ecology and Evolution* 10:844278.

Doyle, S., D. Cabot, L. Griffin, A. Kane, K. Colhoun, C. Redmond, A. Walsh, and B. J. McMahon (2023). Home range of a long-distance migrant, the Greenland Barnacle Goose *Branta leucopsis* , throughout the annual cycle. *Bird Study* 70:37–46.

Dwyer, T. J. (1972). An adjustable radio-package for ducks. *Bird-Banding* 43:282–284. EA Engineering, Science, and Technology, Inc (2024). Site assessment report processes and challenges of implementing nature-based solutions to ensure mission resilience and preserve habitat Marine Corps Base Hawaii – Nuʻupia Ponds. U.S. Army Engineer Research and Development Center Progressing Engineering with Nature EA Project No. 6395901.

Eddleman, W. R., F. L. Knopf, and C. T. Patterson (1985). Chronology of migration by American Coots in Oklahoma. *The Journal of Wildlife Management* 49:241–246.

Engilis, A. Jr., and T. K. Pratt (1993). Status and population trends of Hawaii's native waterbirds, 1977 - 1987. *Wilson Ornithological Society* 105:142–158.

Fournier, A. M. (2017). Phenology, habitat use, and the impacts of wetland management on autumn migrating rails in Missouri. Ph.D. dissertation, University of Arkansas, Fayetteville, Arkansas, USA.

Frazier, A. G., T. W. Giambelluca, H. F. Diaz, and H. L. Needham (2016). Comparison of geostatistical approaches to spatially interpolate month-year rainfall for the Hawaiian Islands. *International Journal of Climatology* 36:1459–1470.

Gardner, R. C., and M. C. Finlayson (2018). Ramsar Convention on Wetlands. (2018). Global Wetland Outlook: State of the World's Wetlands and their Services to People. Ramsar Convention Secretariat.

Gong, P., Z. Niu, X. Cheng, K. Zhao, D. Zhou, J. Guo, L. Liang, X. Wang, D. Li, H. Huang, Y. Wang, et al. (2010). China's wetland change (1990–2000) determined by remote sensing. *Science China Earth Sciences* 53:1036–1042.

Haig, S. M., S. P. Murphy, J. H. Matthews, I. Arismendi, and M. Safeeq (2019). Climate-altered wetlands challenge waterbird use and migratory connectivity in arid landscapes. *Scientific Reports* 9:4666.

Hawaiʻi Climate Change Mitigation and Adaptation Commission (2021). *State of Hawaiʻi Sea Level Rise Viewer | PacIOOS*. [Online.] Available at https://www.pacioos.hawaii.edu/shoreline/slr hawaii/.

Henry, A. R. (2017). Hawaii biannual waterbird survey 2017 protocol update. Wetland Ecologist, Scaup & Willet LLC.

Hohman, T. R., R. W. Howe, D. C. Tozer, E. E. Gnass Giese, A. T. Wolf, G. J. Niemi, T. M. Gehring, G. P. Grabas, and C. J. Norment (2021). Influence of lake levels on water extent, interspersion, and marsh birds in Great Lakes coastal wetlands. *Journal of Great Lakes Research* 47:534–545.

IUCN (2025). Movement patterns classification scheme (Version 2.1). *IUCN Red List of Threatened Species*. [Online.] Available at https://www.iucnredlist.org/resources/movement-patterns-classification-scheme.

Kolts, J. R., and S. B. McRae (2017). Seasonal home range dynamics and sex differences in habitat use in a threatened, coastal marsh bird. *Ecology and Evolution* 7:1101–1111.

Levin, L. A., C. Neira, and E. D. Grosholz (2006). Invasive cordgrass modifies wetland trophic function. *Ecology* 87:419–432.

Longcore, J. R., D. G. McAuley, G. W. Pendelton, C. R. Bennatti, T. M. Mingo, and K. L. Stromborg (2006). Macroinvertebrate abundance, water chemistry, and wetland characteristics affect use of wetlands by avian species in Maine. *Hydrobiologia* 567:143–167.

MacDonald, G. M. (2010). Water, climate change, and sustainability in the Southwest. *Proceedings of the National Academy of Sciences* 107:21256–21262.

Mayer, T. D., and S. L. Pilson (2019). Interactions of water levels with water quality, endemic waterbirds, and invasive species in a shallow, tropical pond. *Hydrobiologia* 829:77–93.

McGinness, H. M., M. V. Jackson, L. Lloyd-Jones, F. Robinson, A. Langston, L. G. O'Neill, S. Rapley, M. Piper, M. Davies, J. Hodgson, J. M. Martin, et al. (2024). Extensive tracking of nomadic waterbird movements reveals an inland flyway. *Ecology and Evolution* 14:e70668.

Mojica Vélez, J. M., S. Barrasa García, and A. Espinoza Tenorio (2018). Policies in coastal wetlands: Key challenges. *Environmental Science & Policy* 88:72–82.

Mwangi, J., R. H. G. Klaassen, M. Muchai, and B. I. Tieleman (2020). Home-ranges of tropical Red- capped Larks are influenced by breeding rather than vegetation, rainfall or invertebrate availability. *Ibis* 162:492–504.

Nielsen, D. L., and M. A. Brock (2009). Modified water regime and salinity as a consequence of climate change: prospects for wetlands of Southern Australia. *Climatic Change* 95:523–533.

Ofstad, E. G., I. Herfindal, E. J. Solberg, and B.-E. Sæther (2016). Home ranges, habitat and body mass: simple correlates of home range size in ungulates. *Proceedings of the Royal Society B: Biological Sciences* 283:20161234.

PacIOOS (2025). Weather Observations: Moku o Loʻe, Oʻahu | PacIOOS. *Pacific Islands Ocean Observing System (PacIOOS)*. [Online.] Available at https://www.pacioos.hawaii.edu/weather/obs- mokuoloe/.

Paxton, E. H., K. L. Paxton, M. T. Kawasaki, P. M. Gorresen, C. B. Van Rees, and J. G. Underwood (2022). Hawaiian waterbird movement across a developed landscape. *The Journal of Wildlife Management* 87:e22336.

Prosser, D. J., C. S. Teitelbaum, S. Yin, N. J. Hill, and X. Xiao (2023). Climate change impacts on bird migration and highly pathogenic avian influenza. *Nature Microbiology* 8:2223–2225.

Reed, J. M., D. W. DesRochers, E. A. VanderWerf, and J. M. Scott (2012). Long-term persistence of Hawaii's endangered avifauna through conservation-reliant management. *BioScience* 62:881–892.

Riggs, R. (2016). Movement and sexual dimorphism of the endangered Hawaiian Coot, (*Fulica alai*), on Oahu. Master's Thesis, University of Hawaii at Manoa, Honolulu, HI, USA.

Riggs, R. A., A. D. Taylor, R. C. Thomson, and R. H. Cowie (2019). Sexual dimorphism and seasonal variability of shield size in the endangered Hawaiian Coot (*Fulica alai*). *Waterbirds* 42:70.

Rush, S. A., K. F. Gaines, W. R. Eddleman, and C. J. Conway (2020). Clapper Rail (*Rallus crepitans*), version 1.0. In Birds of the World (P. G. Rodewald, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi-org.ezproxy2.library.colostate.edu/10.2173/bow.clarai11.01

Saintilan, N., B. Horton, T. E. Törnqvist, E. L. Ashe, N. S. Khan, M. Schuerch, C. Perry, R. E. Kopp, G. G. Garner, N. Murray, K. Rogers, et al. (2023). Widespread retreat of coastal habitat is likely at warming levels above 1.5 °C. *Nature* 621:112–119.

Shizuka, D., and B. E. Lyon (2008). Improving the reliability of molecular sexing of birds using a W-specific marker. *Molecular Ecology Resources* 8:1249–1253.

Stott, P. (2016). How climate change affects extreme weather events. *Science* 352:1517–1518.

Strong, C., B. Zuckerberg, J. L. Betancourt, and W. D. Koenig (2015). Climatic dipoles drive two principal modes of North American boreal bird irruption. *Proceedings of the National Academy of Sciences* 112.

Takano, L. L., and S. M. Haig (2004). Seasonal movement and home range of the Mariana Common Moorhen. *The Condor* 106:652–663.

Takekawa, J. Y., J. T. Ackerman, L. A. Brand, T. R. Graham, C. A. Eagles-Smith, M. P. Herzog, B. R. Topping, G. G. Shellenbarger, J. S. Kuwabara, E. Mruz, S. L. Piotter, and N. D. Athearn (2015a). Unintended consequences of management actions in salt pond restoration: Cascading effects in trophic interactions. *PLOS ONE* 10:e0119345.

Takekawa, J. Y., S. R. Heath, S. A. Iverson, N. Gaidet, J. Cappelle, T. Dodman, W. Hagemeijer, W. D. Eldridge, S. A. Petrie, G. S. Yarris, S. Manu, et al. (2015b). Movement ecology of five Afrotropical waterfowl species from Malawi, Mali and Nigeria. *Ostrich* 86:155–168.

Underwood, J. G., M. Silbernagle, M. Nishimoto, and K. Uyehara (2013). Managing conservation reliant species: Hawai'i's endangered endemic waterbirds. *PLoS ONE* 8:e67872.

USFWS (2011). Recovery plan for Hawaiian waterbirds, Second Revision. USFWS.

Van De Pol, M., L. D. Bailey, M. Frauendorf, A. M. Allen, M. Van Der Sluijs, N. Hijner, L. Brouwer, H. De Kroon, E. Jongejans, and B. J. Ens (2024). Sea-level rise causes shorebird population collapse before habitats drown. *Nature Climate Change* 14:839–844.

Van Der Kolk, H.-J., B. J. Ens, K. Oosterbeek, W. Bouten, A. M. Allen, M. Frauendorf, T. K. Lameris, T. Oosterbeek, S. Deuzeman, K. De Vries, E. Jongejans, and M. Van De Pol (2020). Shorebird feeding specialists differ in how environmental conditions alter their foraging time. *Behavioral Ecology* 31:371–382.

VanderWerf, E. A. (2024). Feasibility assessment of ‘Alae ‘Ula (Hawaiian Common Gallinule; *Gallinula galeata sandvicensis*) reintroduction to a third island. Hawaii Division of Forestry and Wildlife.

Van Rees, C. B., and J. M. Reed (2014). Wetland loss in Hawai'i since human settlement. *Wetlands* 34:335–350.

Van Rees, C. B., and J. M. Reed (2018). Predicted effects of landscape change, sea level rise, and habitat management on the extirpation risk of the Hawaiian common gallinule (*Gallinula galeata sandvicensis*) on the island of O‘ahu. *PeerJ* 6:e4990.

Van Rees, C. B., J. M. Reed, R. E. Wilson, J. G. Underwood, and S. A. Sonsthagen (2018a). Landscape genetics identifies streams and drainage infrastructure as dispersal corridors for an endangered wetland bird. *Ecology and Evolution* 8:8328–8343.

Van Rees, C. B., P. R. Chang, J. Cosgrove, D. W. DesRochers, H. K. W. Gee, J. L. Gutscher-Chutz, A. Nadig, S. E. Nagata, M. Silbernagle, J. G. Underwood, K. Uyehara, and J. M. Reed (2018b). Estimation of vital rates for the Hawaiian Gallinule, a cryptic, endangered waterbird. *Journal of Fish and Wildlife Management* 9:117–131.

Van Rees, C. B., M. A. Muñoz, S. C. Cooke, and J. M. Reed (2021). Morphological differences in the island-endemic Hawaiian subspecies of the Common Gallinule *Gallinula galeata*. *Pacific Science* 74.

Walter, W. D., and J. W. Fischer (2016). Manual of applied spatial ecology. *Walter Applied Spatial Ecology Lab, Pennsylvania State University, University Park.* [Online.] Available at https://ecosystems.psu.edu/research/labs/walter-lab/manual.

Walther, G.-R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J.-M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein (2002). Ecological responses to recent climate change. *Nature* 416:389–395.

Works, A., L. Nietmann, T. Shimabukuro, K. Harmon, J. Botet Rodriguez, and M. Price (2024). Nest predation and daily survival rates of three Hawaiian endemic species. *Journal of Field Ornithology* 95:art5.

Wu, D., T. Chen, W. Hu, Z. Wang, B. Liu, C. Lu, and H. Zhang (2024). Impact of active vegetation restoration and native vegetation on breeding bird communities in coastal wetlands under disturbance by reintroduced Père David’s deer. *Global Ecology and Conservation* 56:e03345

CHAPTER 2

ABIOTIC AND BIOTIC FACTORS INFLUENCING NEST SITE SELECTION AND HATCHING SUCCESS OF AN ENDANGERED COASTAL WATERBIRD

Introduction

Wetlands are one of the most productive ecosystems, hosting a large diversity of waterbirds, but they are also disproportionately vulnerable to climate change-related habitat alteration and loss (Steen et al. 2016, Chen et al. 2019, Saintilan et al. 2023). Climate change-driven sea level rise (SLR) threatens coastal wetlands by reducing their area and impairing their ecological functions through tidal encroachment, causing habitat shifts (He and Silliman, 2019). However, SLR effects can be detrimental to coastal waterbird populations well before habitat is completely lost from coastal inundation (Van de Pol et al. 2024). Incremental changes to plant communities, water quality, and freshwater availability can be affecting waterbird reproduction and population productivity presently (Warren and Niehrig 1993, Haig et al. 2019). Overall, waterbird populations have increased 18% since 1970 due to waterbird conservation policies enacted, like the North American Wetlands Conservation Act and the Federal Duck Stamp Program (Brown et al. 2002, Vrtiska et al. 2013, Hostetler and Sauer 2025). But some groups of waterbirds, such as secretive marsh birds like rails, have been declining due to habitat loss and other anthropogenic threats, given their limited movements (Tozer 2016, Hostetler and Sauer 2025). We are unsure if and how waterbirds can adapt in the face of climate change accelerating wetland habitat loss (Donnelly et al. 2022, Gaget et al. 2025). Conserving habitats and maintaining their habitat quality are vital for reproduction and recovery of Rallid species (Underwood et al. 2013, Arcilla et al. 2015).

At the microhabitat scale, both biotic and abiotic factors shape the quality of rail nesting habitat; anthropogenic changes can threaten productive breeding (Hartman et al. 2016). Essential rail nesting habitats need density of aquatic invertebrates, shallow water depths for foraging, and emergent vegetation structure for cover from predators (Malone et al. 2021). Climate change is projected to intensify precipitation variability, with more frequent heavy rains and droughts altering water availability and, in turn, affecting water quality for waterbirds (Dai 2011, Haig et al. 2019). For example, in tropical coastal wetlands, water levels affected water quality, which in turn influenced midge abundance and emergence timing, thereby influencing native rail species' food abundance (Mayer and Pilson 2019). Nest site selection of American coots (*Fulica americana),* common gallinules (*Gallinula galeata),* and soras (*Porzana carolina)* can be influenced by wetland plant diversity and structural complexity as well (Fournier et al. 2021). Native plant communities can provide preferred habitat and food resources for native waterbirds over disturbed areas (Wu et al. 2024). However, the introduction of non-native species has negatively altered native habitats. Invasion by common reed (*Phragmites australis*) in freshwater and brackish wetlands in New York, USA reduced avian diversity and significantly reduced nesting by wetland-obligate species such as the Virginia rail (*Rallus limicola*) (Wells et al. 2008). Collectively, these pressures highlight how the microhabitat features on which waterbirds rely for food, cover, and safety are increasingly vulnerable to anthropogenic change. Island environments can see accelerated effects of climate change (Weigelt et al. 2016, Klöck and Nunn 2019). The island of Oahu in Hawaii has lost an estimated 65% of its wetland extent post-settlement (van Rees and Reed 2014). Endemic Hawaiian rail species may be extirpated from their range due to habitat loss alone, but much is unknown about the effects of climate change on nesting habitat preceding habitat loss (van Rees and Reed 2018).

Coastal wetlands in Hawai'i are at risk of SLR-driven inundation, as Hawai'i's sea level is projected to rise 0.7-1.5 feet by 2050 (Hawaiʻi Climate Change Mitigation and Adaptation Commission 2021). In Hawai'i, five endemic waterbirds remain, all listed as federally endangered (USFWS 2011). Hawai'i is home to two endemic Rallid species, the Hawaiian gallinule (ʻalae ʻula, *Gallinula galeata sandvicensis)* and the Hawaiian coot (ʻalae keʻokeʻo, *Fulica alai*); both are endangered from the introduction of non-native species, habitat loss, disease, and more (USFWS 2011, Reed et al. 2012, Underwood et al. 2013). The Hawaiian gallinule is a unique and cryptic rail with an estimated population of 737 inhabiting only Oʻahu and Kauaʻi (Gorresen et al. 2024). Gallinules can inhabit fresh and brackish wetlands and typically occur in shallow wetland habitats with dense emergent vegetation (USFWS 2011). They are ground - nesting waterbirds that nest over water, in dense wetland vegetation, and in agroecosystems (lo'i kalo) (Byrd and Zeillemaker 1981, Webber 2022). Across their range, nest failures are typically caused by flooding and predation by terrestrial mammals (e.g., mongoose (*Herpestes auropunctatus*), cats (*Felis catus)*, pigs (*Sus scrofa)* (Webber 2022, Works et al. 2024, van Rees et al. 2024). Denser vegetation cover for nesting gallinules can provide safety from aerial and terrestrial predators, but much is unknown about their nesting vegetation preferences or factors affecting hatching success. Hawaiian waterbird management has prioritized invasive plant removal of pickleweed (*Batis maritima*), California grass (*Urochloa mutica*), and red mangrove (*Rhizophora mangle*); yet some of these species may be more resilient in future climate conditions than native species (Marcone 2003, Reed et al. 2012, Underwood et al. 2013). As climate change persists with more variable weather (Frazier and Giambelluca 2016), Hawaii's wetlands are prone to increased flooding and drying, leading to changes in water availability and likely quality. Recent estimates are that >40% of O‘ahu's gallinule population will be extirpated from habitat loss associated with SLR (van Rees and Reed 2018). However, SLR and variable rainfall can affect areas before inundation and habitat loss, including increased salinity, vegetation changes, and changes in aquatic insect diversity. Information on the Hawaiian gallinule's current salinity tolerance, preferred vegetation, and water quality conditions for nesting will enhance projections of this species' ability to adapt to changing conditions in coastal Hawaiian wetlands.

To fill that knowledge gap, we explored the abiotic and biotic factors associated with Hawaiian gallinule nests at Marine Corps Base Hawai‘i (MCBH). First, we quantified nesting and hatching success at a variety of wetland sites within MCBH and assessed whether nest failures were predominantly due to predation, flooding, or both. Next, we evaluated the influence of nest-site characteristics on nest-site selection. Lastly, we assessed the effect of abiotic and biotic nest site factors on Hawaiian gallinule hatching success. To achieve these objectives, we conducted weekly nesting surveys at 7 MCBH wetland sites from January-June of 2023-2024, and January-March of 2025, and monitored nests using cameras. We collected abiotic and biotic microhabitat data at Hawaiian gallinule nests and in the surrounding habitat to determine whether vegetation composition, water depth, water proximity, and water quality affected nesting site selection and hatching success.

Methods

Study Area

We studied Hawaiian gallinules at Marine Corps Base Hawaiʻi (MCBH) on the Mōkapu peninsula in Hawaiʻi, USA (21.4398° N, 157.7552° W) on the east side of Oʻahu. There was a total of eight brackish and freshwater wetlands where Hawaiian gallinules occurred, ranging in elevation from 0-4.9 m. (Figure 14). Paʻakai and Kaluapuhi are tidal wetlands influenced by the incoming tide from Kāneʻohe Bay, with rainwater and groundwater influence along the edges of the main pond and at Waipuna (Cox and Jokiel 1997). Sag Harbor is a brackish pond that is not tidally influenced: its proximity to the ocean makes conditions more saline due to groundwater intrusion, and the pond closest to the ocean floods during the wet season and dries during the dry season. Freshwater habitats included Percolation Ditch (hereafter Perc Ditch), Klipper Golf Course (hereafter Klipper; ʻAlae, ʻIwa, and Koloa pond), Elua, and Temporary Lodging Facility (hereafter TLF) stream. The dominant plants in brackish sites were non-native pickleweed (*Batis maritima*), followed by native sedge kaluhā (*Bolboschoenus maritimus*), and non-native woody herbaceous plants, Indian camphorweed (*Pluchea indica*). Freshwater site flora included the plants in brackish wetlands, native ʻahuʻawa (*Cyperus javanicus*), native aeʻaeʻ (*Bacopa monnieri*), non-native umbrella sedge (*Cyperus alternifolius*), non-native red mangrove (*Rhizophora mangle),* non-native California grass (*Urochloa mutica*), and non-native milo (*Thespesia populnea*).

Nest Searching and Habitat Assessment

We searched for active gallinule nests each week at all 7 study sites from January to mid-June in 2023 and 2024, and from January to March in 2025. Hawaiian gallinules are ground-nesting birds that form a cup-shaped nest from nearby vegetation, within emergent wetland or flooded taro agroecosystems (USFWS 2011). We conducted systematic nest searches on foot, during which 2-4 people walked in parallel, each surveying 10-20m swaths of habitat until the entire wetland was searched. Nest searches at Perc Ditch were conducted similarly, but by a single surveyor on a stand-up paddleboard, due to the water depth.

Upon finding a gallinule nest, we recorded location (latitude and longitude), number of eggs, and/or number of chicks present. Additionally, we recorded plant species used for nest construction and water depth (cm) at the nearest water body. We marked each nest with flagging 10 m from the nest. Initial visits were limited to 10 minutes to reduce disturbance at the nest, and nests were not checked more than twice per week. We visited nests a second time, 1-4 days after discovery, to collect additional nest and habitat data. On our second visit to the nest, we again recorded clutch size. If the number of eggs did not change, we floated two eggs to estimate nest initiation date and hatch date. Eggs were aged based on the angle of the floating egg from Rush et al. (2007) methods for aging clapper rail (*Rallus crepitans*) eggs, which have a similar incubation period. Hawaiian gallinules will incubate for approximately 19-22 days (Shallenberger 1977, Byrd and Zeillemaker 1981). Habitat measurements included distance to water (cm), water depth at the nearest water body (cm), percent vegetation cover, and horizontal visual obstruction reading (VOR) (Webber 2024). Distance to water was considered 0cm if the nest was constructed over water, and land nests were measured from the edge of the nest to the nearest pond edge. We placed a 1 m^2 quadrat over the nest and estimated percent cover of each plant species present (to the nearest 5%), as well as percent cover occupied by water or bare ground. Horizontal VOR measurements were taken by holding a board (0.5 m tall by 0.3 m wide) over the nest, and recording the percentage of vegetation cover (to the nearest 5%) observed from 1m away. VOR was recorded for each cardinal direction and then averaged. Additionally, two nests were found by USDA predator control staff outside of the main study sites at ‘Elua (Figure 14) and monitored until hatch; however, only one nest was accessible for habitat measurements.

We monitored each nest with a Reconyx Hyperfire 2.0 Professional Covert IR motion detection camera (Holmen, WI, USA) to document eggs laid, chicks hatched, predation and flooding events, and other causes to nest failure, unless the nest failed between the first and second visit (n=2). Cameras were deployed on the second visit and placed 1m from the nest on a swivel mount, with a camouflaged stake driven into the substrate or vegetation. Cameras were set to take 3 images in rapid succession after motion detection on medium-high sensitivity, with a 5-second delay between pictures, as well as an hourly photo. Nests were monitored with cameras three weeks after hatch, and if a camera was available, we mounted it on the nearest shoreline to the nest to monitor broods. We checked cameras weekly to replace batteries and SD cards when needed.

After the second visit, we checked nests twice a week, 3-4 days apart. We recorded egg and/or chick count, water depth, and water quality once a week (only in 2024 and 2025), and noted evidence of predation, nest failures, and other observations (e.g., flooding, egg displacement). Each nest had up to 10 visits, limited to 10 minutes per visit. Nest searching and monitoring was authorized under ES Permit Number: ES67157D.

Almost all nests were monitored with cameras (n=30), and images were used to determine hatching date, maximum number of eggs laid, maximum number of chicks hatched, and impediments to nest success. Nest success was defined as at least one egg hatching in a clutch, and hatching success was defined as the number of chicks hatched out of the total eggs laid. We recorded the cause of nest failure (e.g., predation, abandonment, flooding) if it could be determined. We recorded the species of any predators observed visiting the nest and whether the visit involved direct predation of eggs, chicks, or adults. Photo review and nest data entry were completed manually after the breeding season was over.

Water Quality

Water quality at the nearest water body was collected at the nest upon the second visit using a handheld YSI ProDSS multi-parameter water sampling instrument (Yellow Springs, OH, USA). Approximately 4 oz of water was collected with the calibration cup from the middle of the water column, if possible; samples were then run for 30 seconds. The YSI collected date, time, water temperature (F°), pressure (mmHg), dissolved oxygen (mg/L), specific conductance (µS/cm), salinity (ppt), pH, turbidity (NTU), and location (latitude and longitude). Water quality measures were only collected in 2024 and 2025.

Habitat Assessment at Non-nest Locations

To compare nesting sites to surrounding habitats, we collected the same habitat data at non-nesting locations as we did for nesting locations. We generated random GPS points within the likely home range of the nesting adults (estimated at ~27m radius from the nest, (Byrd et al. 1985, Chapter 1)), one for each of four quadrants (four locations total per nest). Randomized points of non-nesting locations that landed in unsuitable nesting habitat (e.g., golf course green, paved road, wooded forest) were regenerated within the sampling area until they landed on a suitable nesting area. If no suitable nesting was found within a quadrant, it was excluded from sampling, resulting in some nests having 2 -3 matching non-nesting locations. To minimize disturbance near active nests, we collected data at non-nesting locations in July of 2023 and 2024. Distance to water was measured from the center of the quadrat to the nearest water body. Water quality was measured in the 2024 season at the nearest water body to each location. We were unable to collect habitat data at non-nesting locations in 2025.

Data Analysis

We developed a set of models to evaluate the effects of abiotic and biotic factors on nest-site selection and Hawaiian gallinule hatching success. We modelled the effects of (a) water quality and (b) vegetation characteristics and water-related habitat variables around the nest on (1) nesting site selection, comparing nesting and non-nesting locations, and (2) hatching success using four generalized linear models (GLM) with a binomial distribution in R (version 4.5.0). Initially, we examined clustering in the data by fitting nest ID and year as individual random effects in generalized linear mixed effects models (GLMER) with binomial error structure and no fixed effects; given no variance in either of these random effects and small sample sizes (n=105 locations associated with 35 nests), we proceeded with GLMs that included only fixed effects.

Model 1b and 2b's local habitat variables fixed effects were percent native vegetation within a m^2 quadrat (native), average vegetation visual obstruction reading (VOR) from all cardinal directions (VOR), water depth (depth), and distance to water (distance). Percent vegetation cover was categorized into native, non-native, and abiotic features (e.g., bare ground, open water, rock), with all categories summing to 100. All variables were scaled, and depth and distance were log-transformed due to skewness. In terms of nesting site selection, we predicted Hawaiian gallinules would select nesting areas with higher native vegetation density and higher VOR, since the dominant native plant species *(Bolboschoneus maritimus)* at our sites provides high cover (thus, concealment from predators (Works et al. 2024)), food resources (DesRochers et al. 2009), and continental gallinules prefer robust emergent vegetation at nest sites (Weller and Fredrickson 1974). Hawaiian gallinules will nest in areas with standing water less than 60 cm deep (USFWS 2011). Therefore, we expected gallinules to select nest sites closer to water, and with low water depths supporting emergent vegetation and increased aquatic invertebrates for foraging during incubation and chick rearing (USFWS 2011). Predictions of the effects of local habitat variables on hatching success follow a similar rationale to those for nest site selection. Native vegetation and VOR predictions followed nest-site selection: increased cover can increase hatching success. There are tradeoffs between foraging off-bouts and nest defense during the nesting period, but nest site habitat quality can affect the frequency and duration of nesting off-bouts, thus affecting hatching success (Bruant et al. 2020, McGinness et al. 2025). Low water levels in coastal wetlands on the island of Maui increased invertebrate midge abundance (Mayer and Pilson 2019); therefore, we predicted that low water depths closest to the nest would increase hatching success, if foraging on higher-abundance invertebrates there decreased nest off-bout frequency and duration. Decreased distance to water could also positively affect hatching success by providing parents closer access to aquatic invertebrate food resources, thus limiting off-bout duration and frequency, which can increase nest protection.

We evaluated the water quality variables in a separate set of models (model 1a, 2a) since water quality data were collected only for 2024 and 2025 nests. Our global model's fixed effects were pH, salinity (ppt), turbidity (NTU), and dissolved oxygen (DO, mg/L). Specific conductance (µS/cm) was not used due to high collinearity with salinity, and salinity was the preferred water quality variable to evaluate. All variables were scaled, and salinity and turbidity were log-transformed to counteract skewness. We predicted that fresher water quality would have a positive effect on nesting site selection and hatching success due to the limited salinity tolerances of the Hawaiian gallinule (USFWS 2011, C. B. van Rees, personal communication) and increased invertebrate abundance associated with freshwater influence in wetlands (Muresan et al. 2020, Hohman et al. 2021). Therefore, we hypothesized that increased dissolved oxygen, decreased salinity and turbidity, and stable pH would be preferential for gallinule nesting sites and increase hatching success.

We created four global GLMs for each objective with respective fixed effects. Using the *MuMIn* (v1.48.11) package, our global models underwent model selection and ranked by AIC. We model-averaged the top models within ≤2 ΔAIC, and interpreted the estimates for each variable from the model-averaged coefficients.

Results

Nesting Summary

From 2023-2025, 36 Hawaiian gallinule nests were found across six sites, of which 35 had known fates (Table 1); 30 nests were monitored with cameras. Twenty-nine nests (83%) were successful by the criterion of at least one egg hatching; there were a total of 86:157 (55% hatching success) chicks to eggs that were confirmed through camera trap data or nest visits. The average clutch size of a successful nest was 4.5±1.3, similar to failed nests (4.5±1.8), and the average chicks hatched in successful nests was 3.0±1.3. Of the 35 nests, 6 failed (17%), totaling 27 eggs. Two nests failed when an adult Hawaiian gallinule rolled all the eggs of the clutch out of the nest; two nests were abandoned, and two had unknown causes of failure. Nests that had unknown causes of failure failed before camera deployment: both nests had eggs rolled out in the water, and one nest had pinhole to dime-sized holes in the eggs. Rats (*Rattus sp.)* and a feral cat (*Felis catus)* were seen at the nest, but no eggs or chicks were taken.

Biotic and abiotic data were collected from 35 of the 36 nests (Table A1). The majority of nests (69%) were constructed of native vegetation, specifically kaluhā (n=20*)* and ʻahuʻawa (n=5*)*. Eleven nests were constructed out of non-native vegetation (30%), consisting of 10 pickleweed nests and one milo nest. The majority of nests were constructed on land (81%), and 19% were constructed over water. Most nests were pedestal nests on emergent vegetation (61%), followed by bank nests (33%), and floating anchored nests last (6%). Percent vegetation cover around the nest included categories of native, non - native, and abiotic cover. Native vegetation was the dominant plant type at nest locations (x̄=44.1±37.2, range:0-100) (See Table B1, Figure 15). Nesting sites included fair amount of cover (VOR), low water depths, and had a buffer from the nearest water source (Figure 15). Additionally, nesting sites had higher dissolved oxygen, lower specific conductance, lower salinity, basic pH, and higher turbidity than non - nesting sites (Figure 15, Table B1). Salinity averaged 12.52±8.49 ppt (range: 3.86–36.59).

A total of 104 non-nesting habitat locations were surveyed with the same variables collected as nesting habitat (Table B2, Figure 15). The dominant vegetation at non-nest locations was non-native (65.1±32.2, range: 0-100), followed by native, then abiotic cover (Table B2, Figure 15). The average VOR and water depth were lower than at nesting sites, and the distance to water increased at non-nesting sites (Figure 15, Table B2). Water quality at non-nesting sites was comparable to nesting sites; salinity (ppt) increased and averaged 13.86±7.58 ppt with a range of 0.51-35.08 ppt (Figure 15, Table B2).

Nesting Site Selection

Thirty-five Hawaiian gallinule nest locations and 104 non-nesting habitat locations were considered in our local habitat global model for nest site selection, and four models within 2 ΔAIC were averaged (Table 2). Percent native vegetation within a m^2 quadrat of the nest had a positive effect on nest site selection (β̂=0.786±0.206, z=3.811, p=<0.001) (Figure 16). By contrast, distance to water (log) (β̂=-0.280±0.221, z=1.155, p=0.248), water depth (log) (β̂=-0.025±0.104, z=0.240, p=0.811), and VOR average (β̂=0.020±0.104, z=0.195, p=0.846) had no effect on nest selection (Figure 16). To evaluate the effect of water quality (salinity, turbidity, pH, and dissolved oxygen) on nesting site selection, 16 nests were compared with 57 paired non-nesting locations. Six candidate nest selection models were well supported by the data (Table 3). Salinity was dropped in model selection, and turbidity (log) (β̂=-0.202±0.351, z=0.575, p=0.565), pH (β̂=-0.427±0.632, z= 0.676, p=0.499), and dissolved oxygen (β̂=0.648±0.688, z=0.942, p=0.346) did not significantly influence gallinule nest site selection (Figure 17).

Hatching Success

Thirty-four nests were considered in our local habitat global hatching success model. Three models within 2 ΔAIC were averaged; VOR was dropped as a variable within the top models (Table 5). Depth had a positive effect on hatching success and was the only variable to be significant (β̂=0.394±0.140, z=2.704, p=0.007) (Figure 18). By contrast, distance to water (log) (β̂=0.066±0.138, z=0.467, p=0.641) and percent native vegetation around the nest ($\hat{\beta}=-0.280\pm0.221$, z=1.155, p=0.248) had no effect on hatching success (Figure 18). To evaluate the effect of water quality near the nest on hatching success, we evaluated water quality data from 16 nests. Three candidate models within 2 ΔAIC were averaged, and turbidity was excluded from model selection (Table 6). Salinity ($\hat{\beta}=0.278\pm0.423$, z=0.438, p=0.527), dissolved oxygen ($\hat{\beta}=-0.367\pm0.778$, z=0.459, p=0.646), and pH ($\hat{\beta}=0.4006\pm0.8580$, z=0.454, p=0.650) had no significant effect on hatching success (Figure 19).

Discussion

We observed Hawaiian gallinule nesting preferences at the microhabitat scale, as plant communities, water quality, and water availability can be the first factors to change as sea levels rise. Most gallinule nests were constructed from native kaluhā sedges, and the proportion of native vegetation at the nest had a strong positive effect on nesting site selection. However, native vegetation was not found to be associated with hatching success. Instead, hatching success was higher for nests adjacent to deeper water, perhaps due to invertebrate abundance associated with higher water levels. Nesting success at MCBH was high (83%), without failures from flooding and mammalian predation common at other sites (Webber 2022, Botet Rodriguez 2024, Works et al. 2024, van Rees et al. 2024). Instead, failures resulted from conspecific predations, abandonment, and unknown causes.

Hawaiian gallinules at MCBH exhibited high nesting success (83%); pressures of mammalian predators and flooding did not affect nest success in our study, unlike similar studies (Webber 2022, Botet Rodriguez 2024, Works et al. 2024). Hawaiian gallinule nests at our sites experienced some fluctuations in water level, as observed through camera monitoring, but these did not prevent nesting success. MCBH partners with USDA to manage predators, and the lack of mammalian predators at nesting sites demonstrates its effectiveness. Although predator control is labor-intensive and therefore expensive, it is consistently identified as effective and necessary for Hawaiian waterbird conservation (Underwood et al. 2013, Roerk et al. 2022). Six instances of rats (*Rattus sp.)* visiting a nest at night were captured on camera, but the visits occurred after hatching. One nest had a dead chick in it, with no other individuals present, and a rat was recorded eating the dead chick. We are unsure whether nest sites are selected with predator threats in mind, but predator exclusion is important for protecting waterbird nesting habitat. Though nest success was higher, hatching success was comparable to studies of this species at other locations (55%) (Botet Rodriguez 2024, Works et al. 2024). As no partial predation of clutches was observed, food availability and body condition of incubating adults may have instead influenced hatching success (Gajdošová et al. 2023).

Native vegetation cover around the nest was the most important factor in nesting habitat selection for Hawaiian gallinules; non-native vegetation cover, water depth, water proximity, and water quality had no effect on nest site selection. The majority of the nests were constructed from and/or surrounded by kaluhā, a tall sedge that provides aerial and terrestrial cover, which may be advantageous for protection from predators. Non-native pickleweed was the dominant vegetation in the surrounding area. Elsewhere, invasion of non-native plants has reduced diversity, abundance, and nesting of wetland-dependent birds (Wells et al. 2008). Although pickleweed can provide suitable nesting habitat - gallinules in our study did construct and hatch nests in pickleweed - it may also take over wetland sites and reduce native sedge biomass that gallinules apparently prefer. Although vertical vegetation density at the nest (VOR) did not have an independent effect on nesting site selection, greater cover is a likely consequence of selection for native kaluhā. Due to reduced predation pressure from predator control and the abundance of low-cover pickleweed, vegetation cover may not have influenced nest-site selection.

We originally expected that closer proximity to water and low water depths would be advantageous for accessing aquatic invertebrate food sources for incubating adults and growing chicks, but neither water depth nor proximity affected nest site selection. Because gallinule often relocate broods soon after hatching (Byrd and Zeillemaker 1981), proximity to open water and water depth may be less critical to nest site selection, provided adults remain within an effective commuting distance of suitable foraging habitat to feed precocial chicks during the first few days post-hatching. We are unsure what their diet consists of, and perhaps it does not actually consist mainly of aquatic invertebrates during nesting bouts, but rather of wetland plants (Desrochers et al. 2010). Water quality factors of salinity, turbidity, dissolved oxygen, and pH had no effect on nest site selection; despite predictions that salinity would have a strong influence, it was not included in the best-fit models. The salinity tolerance of Hawaiian gallinules is poorly understood, and the previously hypothesized upper limit was 10 ppt (VanderWerf 2024, C. B. van Rees, personal communication). Gallinules at MCBH seem somewhat resilient to salinity, as the average nest-site salinity was 12.5±8.5 ppt, with an outlier successfully hatching in hypersaline conditions at 36.59 ppt. Note that though nests did hatch near these conditions, they may not have been suitable for survival of chicks. Anecdotally, we observed some chicks dead near nests and overall low juvenile recruitment at more saline (Pa‘akai) versus more freshwater (Klipper) sites. Future work should follow chick survival in the weeks after hatching to learn more about potential sources of mortality (Webber 2022), and if lower in higher-salinity conditions, prioritize the protection or expansion of fresher-water ponds along the edges of Pa‘akai. Additionally, our ability to detect nest site selection based on water quality may have been hampered by both sample size of nests with associated water quality measurements and the timing of our sample collection: water quality at each nest was sampled at the time the nest was active, whereas water quality at paired non-nesting locations within the home range was not sampled until the end of each field season. More information on diet and food availability influences on nest site selection may reveal patterns in Hawaiian gallinule nesting ecology.

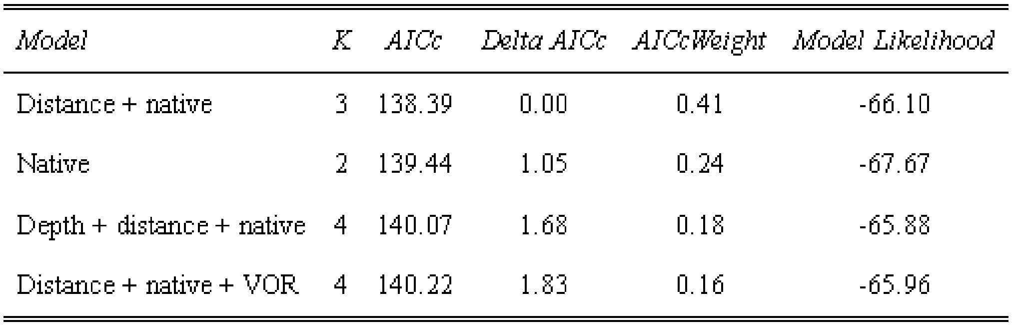
When evaluating Hawaiian gallinule hatching success at the microhabitat scale, water depth had a significant positive effect, whereas all other abiotic and biotic factors had no effect. Hawaiian gallinules are shallow-water feeders; observationally, they are mostly seen walking on aquatic vegetation, preying upon plant rhizomes, aquatic invertebrates, and small fish in shallow depths (USFWS 2011). Higher water levels in shallow ponds may provide more habitat for invertebrates and efficient foraging depths for gallinules. In the Great Lakes coastal wetlands, increased vegetation interspersion and higher macroinvertebrate abundance associated with higher lake levels positively affected the population abundance of marsh birds, suggesting that higher water levels could increase the available nesting habitat (Hohman et al. 2021). Body condition of nesting females affects egg size and hatching success; increased body condition can reduce time spent off the nest to forage, thereby reducing susceptibility to predators and increasing egg viability through continuous incubation (Gajdošová et al. 2023). Therefore, water depths near the nest that increase aquatic invertebrate abundance may increase egg hatchability and help protect nests during incubation by reducing the time spent foraging. We expected lower water levels to increase invertebrate abundance, similar to Mayer and Pilson's (2019) study, but we did not see any association with hatching success and lower water levels, perhaps because of the lack of midges at MCBH. Additionally, though predation levels were low at our site, deeper water may also protect waterbird nesting from terrestrial predators such as mongoose (Works et al. 2024). Surprisingly, if nest location is important for foraging efficiency and incubation success, distance to water had no effect on hatching success; however, the small home ranges of gallinules may limit distances necessary to find suitable foraging grounds (Chapter 1). Unlike for nest-site selection, native vegetation density around nests did not affect hatching success, likely because low predator abundance reduced the importance of native plant cover during incubation. Similarly, nesting success of Hawaiian gallinules was not impacted by vegetation cover at Hanalei NWR, Kaua'i, HI, where mongoose is absent (Webber 2022). Continuous tracking of nests year-round and consistent water quality monitoring may reveal other factors associated with hatching success.

Before wetland habitat is lost from climate change and SLR, increased temperature fluctuations will affect wetland conditions, altering the availability of coastal wetland nesting sites for waterbirds (Dai 2011, He and Silliman 2019, Van De Pol et al. 2024). We are unsure whether the Hawaiian gallinule can adapt to drastic environmental changes, but we observed nest site flexibility within native sedges and non- native pickleweed, along with higher tolerances to salinity than previously documented. Information on current nesting-site selection and hatching success can aid managers in preventing local extirpation. At the microhabitat scale, native vegetation density around the nest and water depth are the only significant factors in nesting site selection and hatching success, respectively. Indirect effects of climate change will alter plant and invertebrate communities as coastal wetlands become more saline. Native sedges preferred by Hawaiian gallinules may not be adaptable to increases in salinity, and non-native pickleweed may become even more invasive due to its climate resilience (Kane et al. 2014). Identifying climate-resilient plants, similar to the native sedges' gallinules preferred, will be important in conserving coastal wetland habitat structure for nesting Hawaiian waterbirds. Climate change effects will increase precipitation and temperature variability, which can affect water availability and depth, thereby affecting hatch success in the future and potentially constraining gallinules to more seasonal nesting patterns. Long-term conservation goals should consider changes in coastal wetland water availability; implementing freshwater pumping into wetlands can preserve water availability in shallow wetland habitats and their plant communities, providing gallinules with adequate habitat for nesting. Future studies of Hawaiian gallinule diet, the effects of adult body condition on incubation behavior and hatching success, and chick survival can aid in the conservation of the Hawaiian gallinule during a vital life history stage. Future conservation and translocation efforts for the Hawaiian gallinule should prioritize native vegetation characteristics and deeper water to enhance invertebrate abundance at host sites and support population persistence.

Table 1: Summary of Hawaiian gallinule nests at all nesting sites at MCBH, Oʻahu, Hawaiʻi from weekly nest searches from January-June of 2023-2024, and from January-March in 2025. Nesting success is defined as at least one egg hatching, hatching success is the total number of chicks divided by eggs laid.

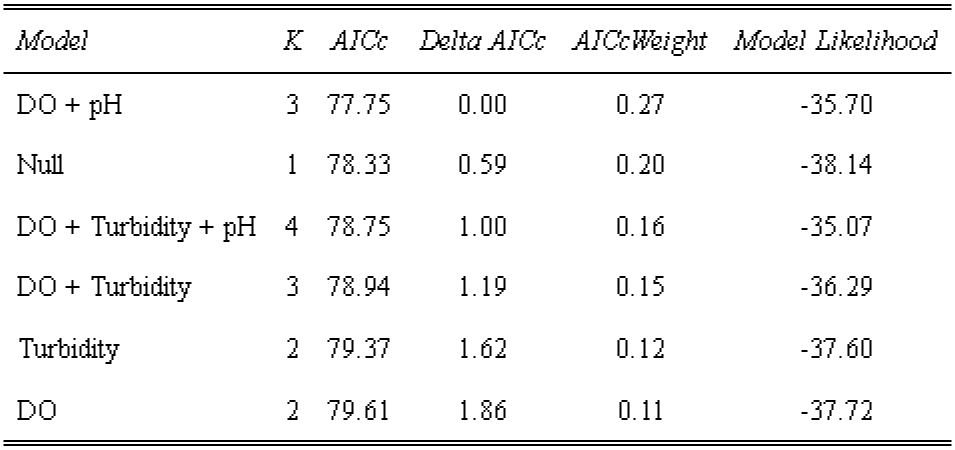
	Site	Nests	Eggs	Chicks	Nesting Success	Hatching Success
Year 2023	Paakai	6	28	11	83.3%	39%
	Waipuna	3	19	10	66.7%	53%
	Klipper	3	10	4	66.7%	40%
	TLF	2	8	6	100%	75%
Year 2024	Paakai	10	41	16	70%	39%
	Waipuna	3	15	10	100%	67%
	Klipper	2	6	6	100%	100%
	Sag Harbor	2	11	7	100%	64%
	Elua	2	9	9	100%	100%
Year 2025	TLF	1	5	3	100%	60%
	Sag Harbor	1	5	4	100%	80%
Total		35	157	86	82.90%	54.70%

Table 2: Candidate models, number of parameters (K), AICc, ΔAICc, Akaike weight (wi), and log-likelihood (LL) for Hawaiian gallinule nest site selection at MCBH from 2023-2025 based on biotic and abiotic parameters distance to water (distance), native vegetation density around the nest (native), water depth (depth), and average VOR (VOR) in generalized linear model selection. All models within 2 AICc units from the model with lowest AICc value are shown.



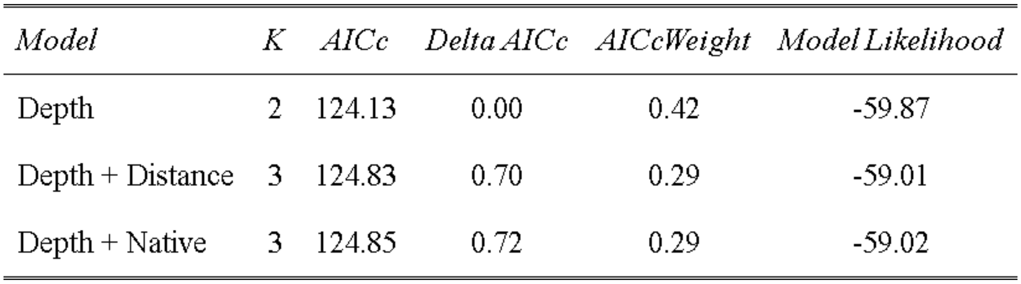
Model	*K*	*AICc*	*Delta AICc*	*AICcWeight*	*Model Likelihood*
Distance + native	3	138.39	0.00	0.41	-66.10
Native	2	139.44	1.05	0.24	-67.67
Depth + distance + native	4	140.07	1.68	0.18	-65.88
Distance + native + VOR	4	140.22	1.83	0.16	-65.96

Table 3: Candidate models, number of parameters (K), AICc, ΔAICc, Akaike weight (wi), and log-likelihood (LL) for Hawaiian gallinule nest site selection at MCBH from 2024-2025 based on water quality parameters dissolved oxygen (DO), turbidity, pH, and salinity in generalized linear model selection. All models within 2 AICc units from the model with lowest AICc value are shown.



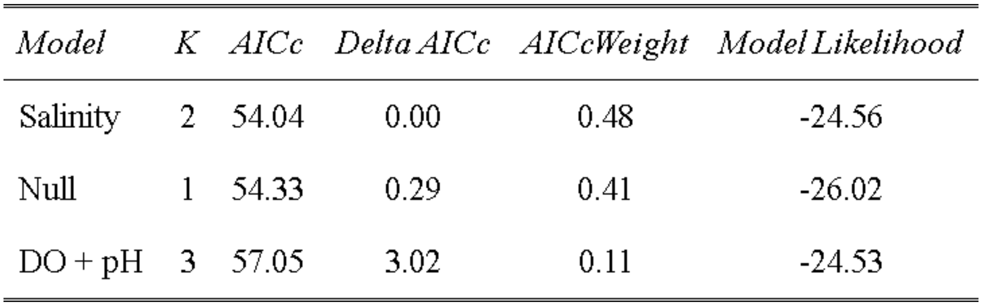
Model	*K*	*AICc*	*Delta AICc*	*AICcWeight*	*Model Likelihood*
DO + pH	3	77.75	0.00	0.27	-35.70
Null	1	78.33	0.59	0.20	-38.14
DO + Turbidity + pH	4	78.75	1.00	0.16	-35.07
DO + Turbidity	3	78.94	1.19	0.15	-36.29
Turbidity	2	79.37	1.62	0.12	-37.60
DO	2	79.61	1.86	0.11	-37.72

Table 4: Candidate models, number of parameters (K), AICc, ΔAICc, Akaike weight (wi), and log-likelihood (LL) for Hawaiian gallinule hatching success at MCBH from 2023-2025 based on biotic and abiotic parameters distance to water (distance), native vegetation density around the nest (native), and water depth (depth) in generalized linear model selection. Variable average VOR (VOR) was dropped within the top candidate models. All models within 2 AICc units from the model with lowest AICc value are shown.



Model	*K*	*AICc*	*Delta AICc*	*AICcWeight*	*Model Likelihood*
Depth	2	124.13	0.00	0.42	-59.87
Depth + Distance	3	124.83	0.70	0.29	-59.01
Depth + Native	3	124.85	0.72	0.29	-59.02

Table 5: Candidate models, number of parameters (K), AICc, ΔAICc, Akaike weight (wi), and log-likelihood (LL) for Hawaiian gallinule hatching success at MCBH from 2024-2025 based on water quality parameters dissolved oxygen (DO), pH, and salinity in generalized linear model selection. Turbidity was dropped within the top candidate models. All models within 2 AICc units from the model with lowest AICc value are shown.



Model	*K*	*AICc*	*Delta AICc*	*AICcWeight*	*Model Likelihood*
Salinity	2	54.04	0.00	0.48	-24.56
Null	1	54.33	0.29	0.41	-26.02
DO + pH	3	57.05	3.02	0.11	-24.53

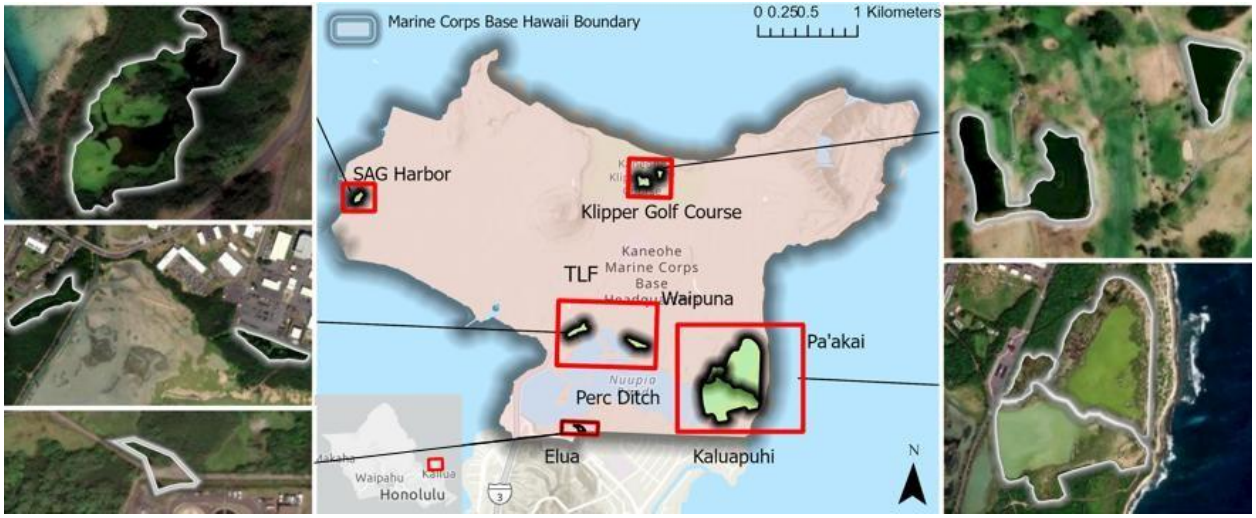


Figure 14. Map of Marine Corps Base Hawai‘i (MCBH), Kāne‘ohe Bay, O‘ahu, Hawai‘i, USA. Study area boundary is shown in gray, and black boundaries highlighted by red boxes indicate locations of 7 wetland study sites with weekly nest surveys of Hawaiian waterbirds from January-June 2023-2024, and January-March 2025. Wetlands include (clockwise from left) Sag Harbor; Klipper Golf Course; Pa'akai, Waipuna, and Kaluapuhi ponds; Percolation Ditch (“Perc Ditch”), Temporary Lodging Facility (TLF) stream, and Elua.

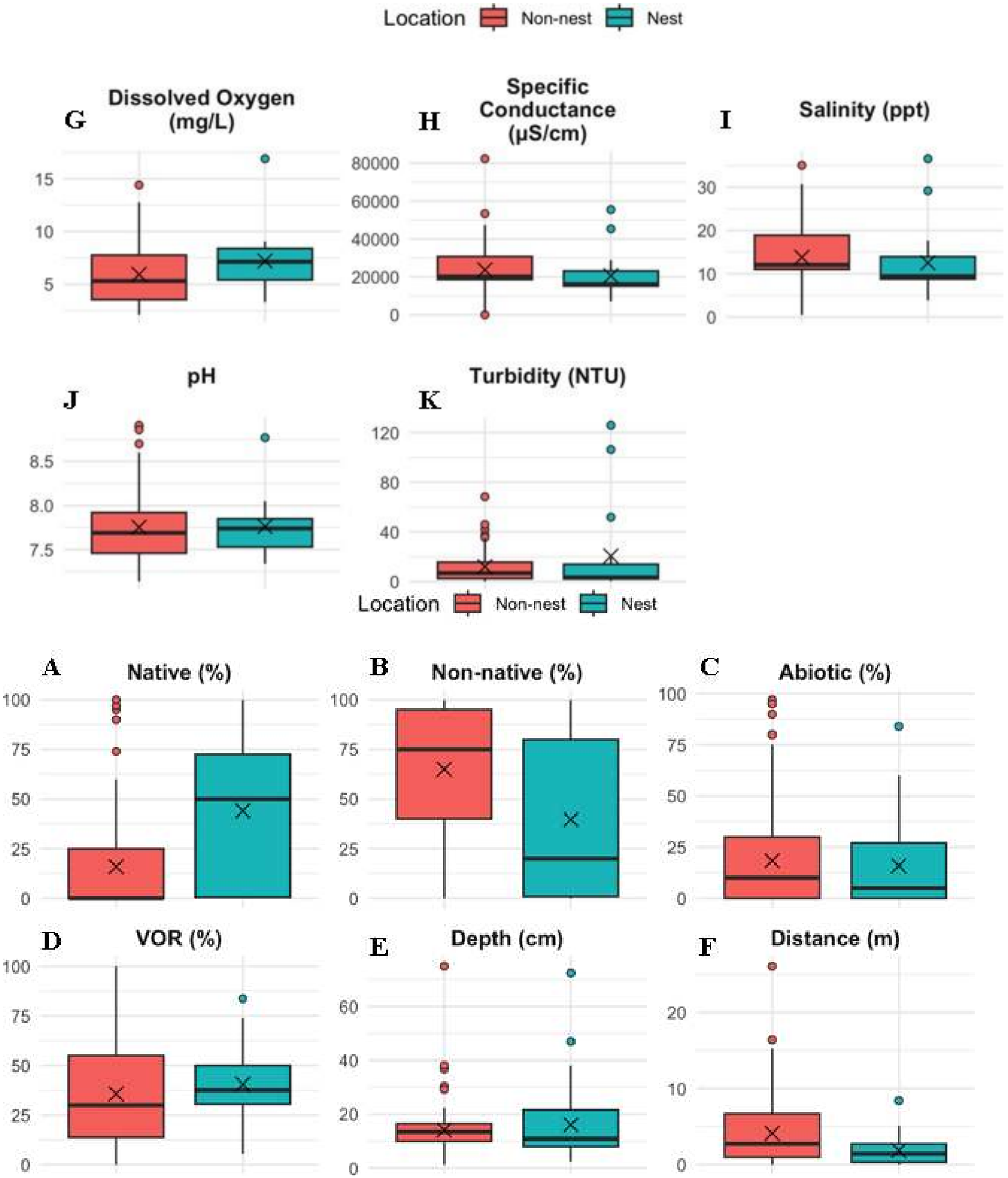


Figure 15: Boxplots comparing abiotic and biotic factors at nest (blue) vs. non-nest locations (red). Plots A-D were collected at the nesting site or within a 1m^2 quadrat of available habitat. Plots E-K was collected at the nearest waterbody. The horizontal line within the box indicates the median, stars indicate the mean, dots indicate outliers, and the bars represent the upper and lower quartiles within the box

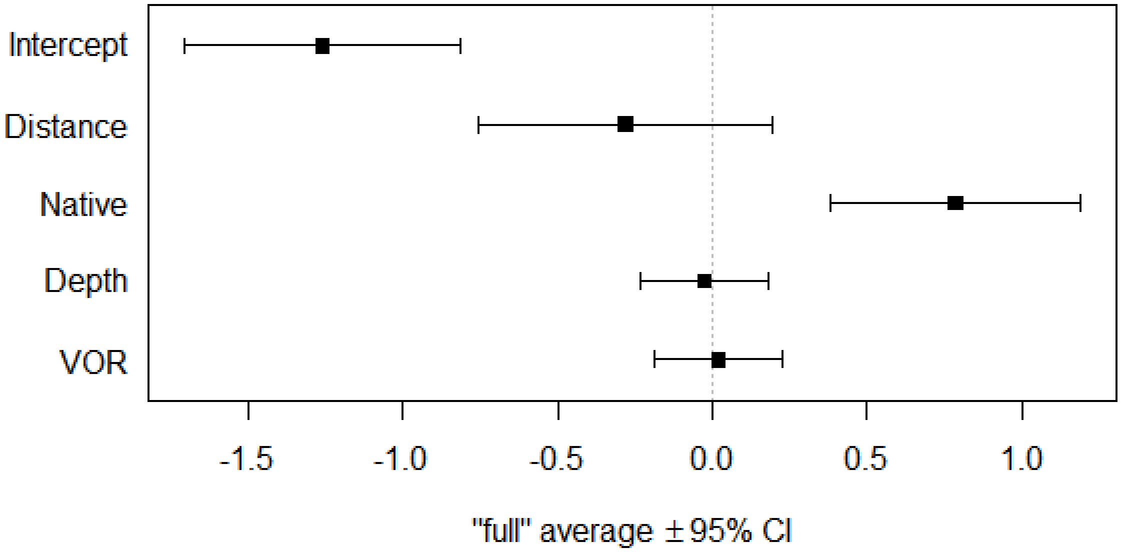


Figure 16: Coefficient plots of model averaged coefficients and standard errors (SE) of the effect of local habitat variables (water proximity (distance), native vegetation density around the nest (native), water depth nearest to the nest (depth), and vegetation cover at the nest (VOR)) on Hawaiian gallinule nest site selection.

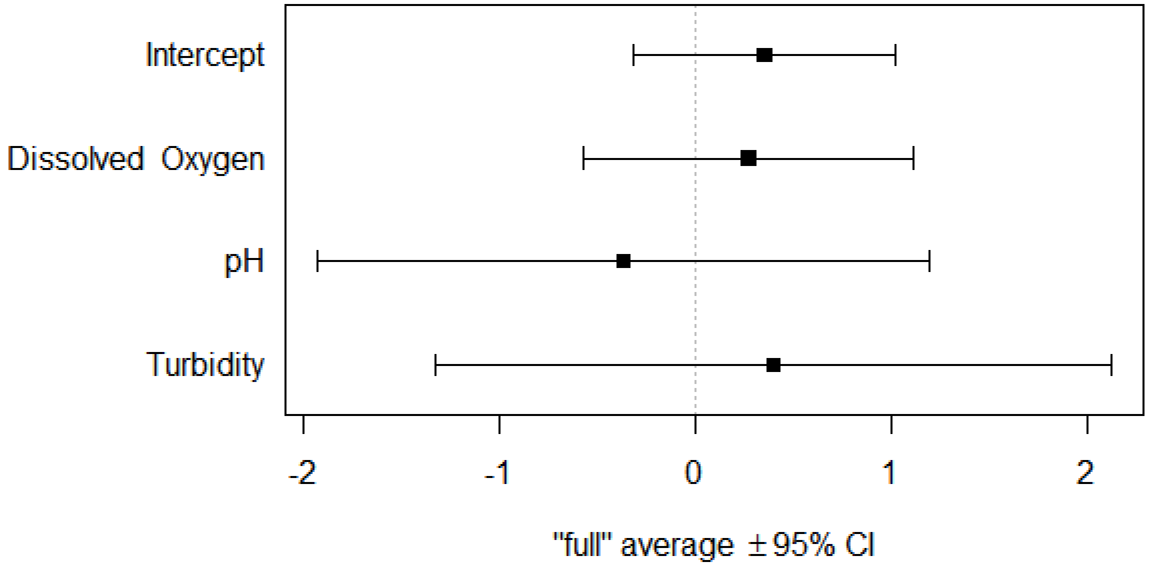


Figure 17: Coefficient plots of model averaged coefficients and standard errors (SE) of the effect of water quality (dissolved oxygen (DO), pH, and turbidity) on Hawaiian gallinule nest site selection.

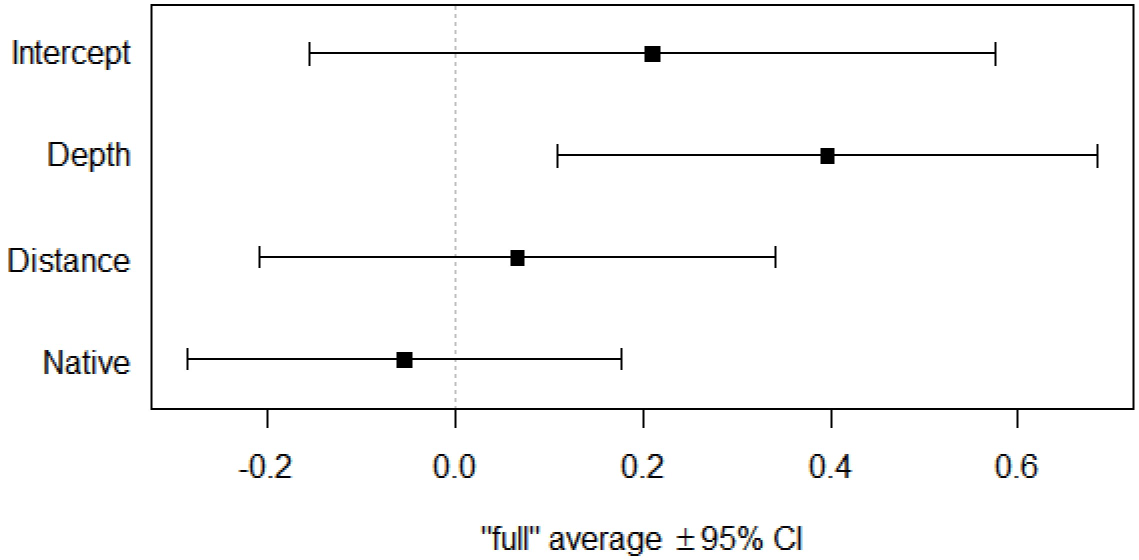


Figure 18: Coefficient plots of model averaged coefficients and standard errors (SE) of the effect of local habitat variables (water proximity (distance), native vegetation density around the nest (native), water depth nearest to the nest (depth), and vegetation cover at the nest (VOR)) on Hawaiian gallinule hatching success.

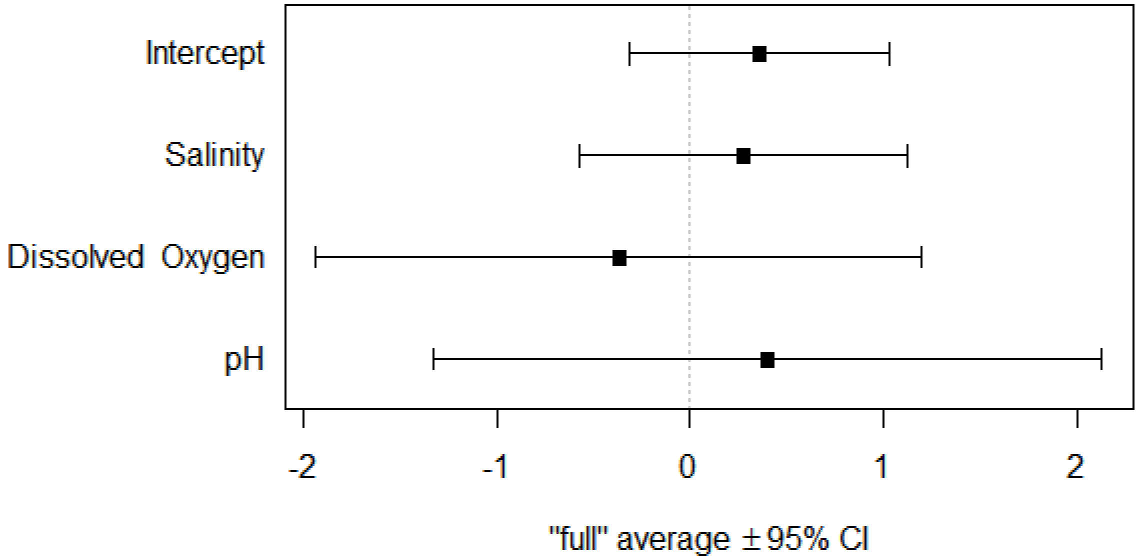


Figure 19: Coefficient plots of model averaged coefficients and standard errors (SE) of the effect of water quality salinity, (dissolved oxygen (DO), and pH) on Hawaiian gallinule hatching success.

References

Botet Rodriguez, J. A. (2024). Nest site selection and predator impacts on the reproductive success of three endangered waterbirds in Hawaii. Master's Thesis, University of Hawaii Manoa, Honolulu, HI, USA.

Brown, S. C., B. A. Harrington, K. C. Parsons, and E. P. Mallory (2002). Waterbird use of Northern Atlantic Wetlands protected under the North American Wetlands Conservation Act. *Waterbirds: The International Journal of Waterbird Biology* 25:106–114.

Bruant, A., S. Picardi, P. Frederick, and M. Basille (2020). Effect of foraging and nest defense tradeoffs on the reproductive success of Wood Storks (*Mycteria americana*). *Waterbirds* 43:134–146.

Byrd, G. V., and C. F. Zeillemaker (1981). Ecology of nesting Hawaiian Common Gallinules at Hanalei, Hawaii. *Western Birds* 12:1–12.

Byrd, G. V., R. J. Shallenberger, and C. S. Arume (1985). Notes on the breeding biology of the Hawaiian race of the American Coot. *Elepaio* 45:57–68.

Chen, W., C. Cao, D. Liu, R. Tian, C. Wu, Y. Wang, Y. Qian, G. Ma, and D. Bao (2019). An evaluating system for wetland ecological health: Case study on nineteen major wetlands in Beijing-Tianjin-Hebei region, China. *Science of The Total Environment* 666:1080–1088.

Cox, E. F., and P. L. Jokiel (1997). Environmental study of Nu'upia Ponds Wildlife Management Area, Marine Corps Base Hawaii, Kaneohe Bay. Environmental Compliance and Protection Department, Environmental Affairs Division, Kaneohe Bay, HI, USA.

Dai, A. (2011). Drought under global warming: a review. *WIREs Climate Change* 2:45–65.

Desrochers, D. W., S. R. Mcwilliams, and J. M. Reed (2010). Evaluating if energy and protein limit abundance of Hawaiian Moorhen. *The Journal of Wildlife Management* 74:788–795.

DesRochers, D. W., S. R. McWilliams, M. D. Silbernagle, and J. M. Reed (2009). Macronutrient profiles of wetland plants consumed by the Hawaiian Moorhen (*Gallinula chloropus sandvicensis*). *Wetlands* 29:845–853.

Donnelly, J. P., J. N. Moore, M. L. Casazza, and S. P. Coons (2022). Functional wetland loss drives emerging risks to waterbird migration networks. *Frontiers in Ecology and Evolution* 10:844278.

Fournier, A., J. Lancaster, A. Yetter, C. Hine, T. Beckerman, J. Figge, A. Gioe, M. Greider-Wagner, D. Jen, C. Johnson, M. Larreur, et al. (2021). Nest success and nest site selection of wetland birds in a restored wetland system. *Avian Conservation and Ecology* 16(1):6

Frazier, A. G., and T. W. Giambelluca (2017). Spatial trend analysis of Hawaiian rainfall from 1920 to 2012. *International Journal of Climatology* 37:2522–2531.

Gaget, E., O. Ovaskainen, U. Bradter, F. Haas, L. Jonas, A. Johnston, T. Langendoen, A. S. Lehikoinen, T. Pärt, D. Pavón-Jordán, B. K. Sandercock, et al. (2025). Changes in waterbird occurrence and abundance at their northern range boundaries in response to climate warming: Importance of site area and protection status. *Animal Conservation* 28:490–500.

Gajdošová, D., P. Musil, J. Zouhar, Z. Musilová, Š. Neužilová, and D. Pavón-Jordán (2023). Long-term increase in female body condition and its effect on reproduction in two European red-listed species, Common Pochard (*Aythya ferina*) and Tufted Duck (*Aythya fuligula*). *Ibis* 165:1217–1234.

Gorresen, M. P., R. J. Camp, and E. H. Paxton (2024). Distribution and trends of endemic Hawaiian waterbirds, HCSU-113. Hawai‘i Cooperative Studies Unit, University of Hawai‘i at Hilo, Hawaii National Park, HI, USA. U.S. Geological Survey, Pacific Island Ecosystems Research Center Hawaii National Park, HI, USA.

Guha, H., and S. Panday (2012). Impact of sea level rise on groundwater salinity in a coastal community of south Florida. *JAWRA Journal of the American Water Resources Association* 48:510–529.

Haig, S. M., S. P. Murphy, J. H. Matthews, I. Arismendi, and M. Safeeq (2019). Climate-altered wetlands challenge waterbird use and migratory connectivity in arid landscapes. *Scientific Reports* 9:4666.

Hartman, C. A., J. T. Ackerman, J. Y. Takekawa, and M. P. Herzog (2016). Waterbird nest-site selection is influenced by neighboring nests and island topography. *The Journal of Wildlife Management* 80:1267–1279.

Havens, K. E. (2008). Cyanobacteria blooms: effects on aquatic ecosystems. In *Cyanobacterial Harmful Algal Blooms: State of the Science and Research Needs* (H. K. Hudnell, Editor). Springer, New York, NY, pp. 733–747.

Hawai‘i Climate Change Mitigation and Adaptation Commission (2021). *State of Hawai‘i Sea Level Rise Viewer | PacIOOS*. [Online.] Available at https://www.pacioos.hawaii.edu/shoreline/slr-hawaii/.

He, Q., and B. R. Silliman (2019). Climate change, human impacts, and coastal ecosystems in the Anthropocene. *Current Biology* 29:1021–1035.

Hohman, T. R., R. W. Howe, D. C. Tozer, E. E. Gnass Giese, A. T. Wolf, G. J. Niemi, T. M. Gehring, G. P. Grabas, and C. J. Norment (2021). Influence of lake levels on water extent, interspersion, and marsh birds in Great Lakes coastal wetlands. *Journal of Great Lakes Research* 47:534–545.

Hostetler, J.A., and Sauer, J.R. (2025). Estimates of United States bird trends and bird group relative abundance trajectories, 1970 – 2022. *U.S. Geological Survey*. https://doi.org/10.5066/P1VEAIHO.

Kane, H. H., C. H. Fletcher, L. N. Frazer, and M. M. Barbee (2015). Critical elevation levels for flooding due to sea-level rise in Hawai‘i. *Regional Environmental Change* 15:1679–1687.

Klöck, C., and P. D. Nunn (2019). Adaptation to climate change in small island developing states: A systematic literature review of academic research. *The Journal of Environment & Development* 28:196–218.

Malone, K., E. Webb, D. Mengel, L. Kearns, S. Matteson, and A. McKellar (2021). Secretive marsh bird habitat associations in the Mississippi Flyway: A meta-analysis. *Avian Conservation and Ecology* 16(2):12.

Marcone, M. F. (2003). *Batis maritima* (Saltwort/Beachwort): a nutritious, halophytic, seed bearings, perennial shrub for cultivation and recovery of otherwise unproductive agricultural land affected by salinity. *Food Research International* 36:123–130.

Mayer, T. D., and S. L. Pilson (2019). Interactions of water levels with water quality, endemic waterbirds, and invasive species in a shallow, tropical pond. *Hydrobiologia* 829:77–93.

McGinness, H. M., L. R. Lloyd-Jones, F. Robinson, M. Hawken, D. Cook, L. G. O'Neill, S. Rapley, M.V. Jackson, M. Piper, M. Davies, J. Martin, et al. (2025). Satellite telemetry informs nesting ecology and management of nomadic ibis and spoonbills (Threskiornithidae) in remote breeding sites. *Ornithological Applications* 127:1-17.

Muresan, A. N., M. Gaglio, V. Aschonitis, G. Nobili, G. Castaldelli, and E. A. Fano (2020). Structural and functional responses of macroinvertebrate communities in small wetlands of the Po delta with different and variable salinity levels. *Estuarine, Coastal and Shelf Science* 238:106726.

Reed, J. M., D. W. DesRochers, E. A. VanderWerf, and J. M. Scott (2012). Long-term persistence of Hawaii's endangered avifauna through conservation-reliant management. *BioScience* 62:881–892.

Roerk, L. S., L. Nietmann, and A. J. Works (2022). Efficiency and efficacy of DOC-200 versus Tomahawk Traps for controlling Small Indian Mongoose, *Herpestes auropunctatus* (Carnivora: Herpestidae) in wetland wildlife sanctuaries. *Pacific Science* 76(2):201-207.

Rush, S. A., R. J. Cooper, and M. S. Woodrey (2007). A nondestructive method for estimating the age of Clapper Rail eggs. *Journal of Field Ornithology* 78:407–410.

Saintilan, N., B. Horton, T. E. Törnqvist, E. L. Ashe, N. S. Khan, M. Schuerch, C. Perry, R. E. Kopp, G. G. Garner, N. Murray, K. Rogers, et al. (2023). Widespread retreat of coastal habitat is likely at warming levels above 1.5 °C. *Nature* 621:112–119.

Steen, V. A., S. K. Skagen, and C. P. Melcher (2016). Implications of climate change for wetland-dependent birds in the Prairie Pothole Region. *Wetlands* 36: 445–459.

Tozer, D. C. (2016). Marsh bird occupancy dynamics, trends, and conservation in the southern Great Lakes basin: 1996 to 2013. *Journal of Great Lakes Research* 42:136–145.

Underwood, J. G., M. Silbernagle, M. Nishimoto, and K. Uyehara (2013). Managing conservation reliant species: Hawai'i's endangered endemic waterbirds. *PLoS ONE* (6): e67872

USFWS (2011). Recovery plan for Hawaiian waterbirds, Second Revision. USFWS.

Van De Pol, M., L. D. Bailey, M. Frauendorf, A. M. Allen, M. Van Der Sluijs, N. Hijner, L. Brouwer, H. De Kroon, E. Jongejans, and B. J. Ens (2024). Sea-level rise causes shorebird population collapse before habitats drown. *Nature Climate Change* 14:839–844.

van Rees, C. B., and J. M. Reed (2014). Wetland loss in Hawai'i since human settlement. *Wetlands* 34:335–350.

van Rees, C. B., and J. M. Reed (2018). Predicted effects of landscape change, sea level rise, and habitat management on the extirpation risk of the Hawaiian common gallinule (*Gallinula galeata sandvicensis*) on the island of O'ahu. *PeerJ* 6:e4990.

van Rees, C. B. van, B. Webber, L. Nietmann, A. J. Works, B. Dugger, and J. M. Reed (2024). Updated vital rate estimates for the endangered Hawaiian Common Gallinule (*Gallinula galeata sandvicensis*). *The Wilson Journal of Ornithology* 136:106–113.

VanderWerf, E. A. (2024). Feasibility assessment of ʻAlae ʻUla (Hawaiian Common Gallinule; *Gallinula galeata sandvicensis*) reintroduction to a third island. Hawaii Division of Forestry and Wildlife.

Vrtiska, M. P., J. H. Gammonley, L. W. Naylor, and A. H. Raedeke (2013). Economic and conservation ramifications from the decline of waterfowl hunters. *Wildlife Society Bulletin* 37:380–388.

Warren, R. S., and W. A. Niering (1993). Vegetation change on a northeast tidal marsh: Interaction of sea-level rise and marsh accretion. *Ecology* 74:96–103.

Webber, B. M. (2022). Reproductive success of the Hawaiian common gallinule (*Gallinula galeata sandvicensis*) nesting in taro and managed wetlands on Kauaʻi, Hawaiʻi. Master's Thesis, Oregon State University, Corvallis, OR, USA.

Weigelt, P., M. J. Steinbauer, J. S. Cabral, and H. Kreft (2016). Late Quaternary climate change shapes island biodiversity. *Nature* 532:99–102.

Weller, M.W. & L.H. Fredrickson (1973). Avian ecology of a managed glacial marsh. *Living Bird* 12:269-291.

Wells, A. W., W. C. Nieder, B. L. Swift, K. A. O'Connor, and C. A. Weiss (2008). Temporal changes in the breeding bird community at four Hudson River tidal marshes. *Journal of Coastal Research* 10055:221–235.

Works, A. J., L. Nietmann, T. Shimabukuro, K. Harmon, J. A. B. Rodriguez, and M. R. Price (2024). Nest predation and daily survival rates of three Hawaiian endemic species. *Journal of Field Ornithology* 95.

Wu, D., T. Chen, W. Hu, Z. Wang, B. Liu, C. Lu, and H. Zhang (2024). Impact of active vegetation restoration and native vegetation on breeding bird communities in coastal wetlands under disturbance by reintroduced Père David's deer. *Global Ecology and Conservation* 56:e03345

APPENDIX 1

Table A1: Hawaiian coot monthly seasonal KDE summary over wet (November 1 -April 30) and dry (May 1- October 31) season. Total birds are not the sum of wet and dry season birds evaluated, but the total number of birds analyzed.

	Individuals	Months
Wet Season	14	52
Male	7	30
Female	7	22
Dry Season	6	28
Male	4	18
Female	2	10
Total	16	80

Table A2: Hawaiian gallinule monthly seasonal KDE summary over wet (November 1 -April 30) and dry (May 1- October 31) season. Total birds is not the sum of wet and dry season birds evaluated, but the total number of birds analyzed.

	Individuals	Months
Wet Season	8	39
Male	4	20
Female	4	19
Dry Season	10	41
Male	6	16
Female	4	25
Total	10	80

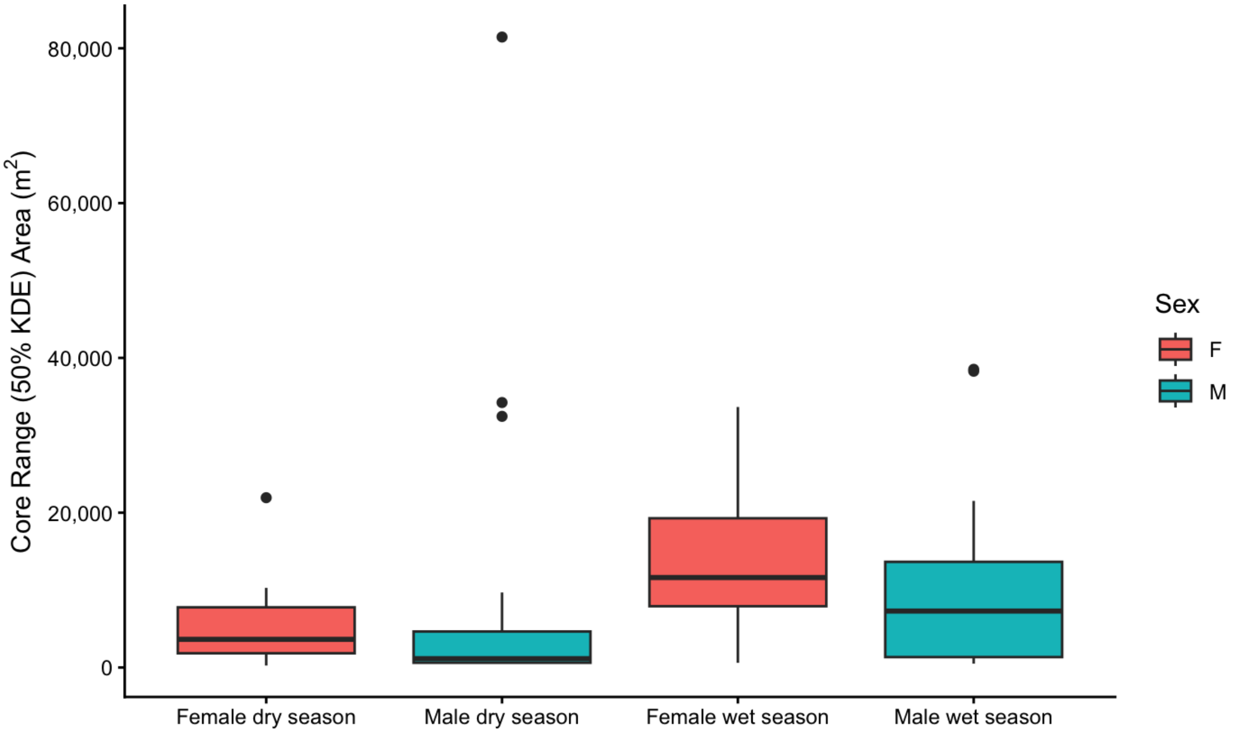


Figure A1: Summary boxplot of Hawaiian coot core ranges (50% KDE) in m2 across seasons (wet and dry season) and sex. Hawaiian coot home range sites include Pa'akai (MCBH), Kaluapuhi (MCBH), and Hāmākua. The horizontal line within the box indicates the median, dots indicate outliers, and the bars represent the upper and lower quartiles within the box.

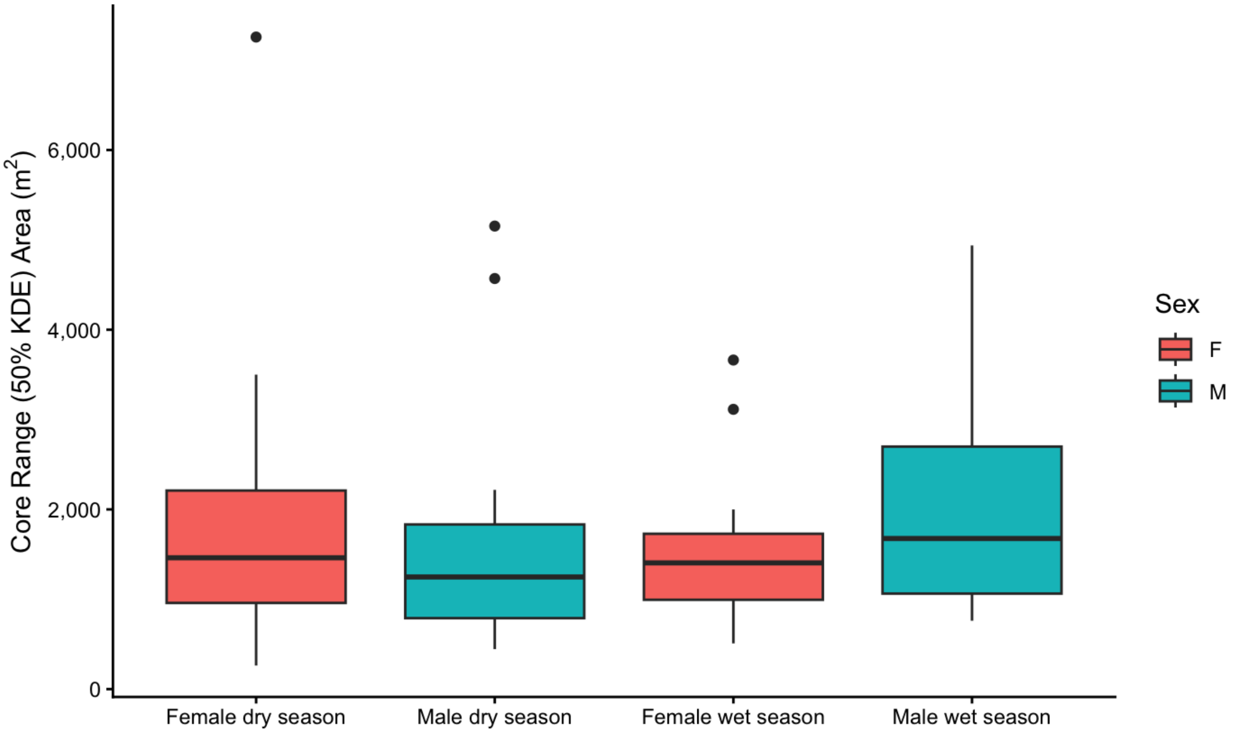


Figure A2: Summary of Hawaiian gallinule core ranges (50% KDE) in m^2 across seasons and sex. Hawaiian gallinule home ranges were all within MCBH, wetland sites include Pa'akai, Waipuna, Klipper, Sag Harbor, TLF, and Perc Ditch. The horizontal line within the box indicates the median, dots indicate outliers, and the bars represent the upper and lower quartiles within the box.

Figure A3: Home ranges (95% KDE) of two Klipper Hawaiian gallinules in April, 2023. Gallinule #237247 home range is shown in blue with black border with blue GPS points, gallinule #237249 home range is shown in orange with black border with orange GPS points. Gallinules exhibit high territoriality and have little overlap in home range despite limited wetland extent of two golf course ponds.

Appendix A1:

Long Distance Movements Coots

Coot ZA:GY (transmitter #261781) was tagged 4/09/2024. She spent a month in Pa‘akai, before heading towards Ni‘ihau (Figure 8). She left Nu‘upia ‘Ekahi, west of Pa'akai on MCBH on 5/4/24 before arriving to the northwest side of Ni‘ihau on the 5th. The total Euclidean distance of this foray was 251 km with no stopovers detected by GPS locations, only two transmissions were received 24 hours apart during her journey. She remained at a coastal wetland on Ni‘ihau until September 2024 when she flew back to O‘ahu. She landed on the west side of Kaua‘i on 9/16/24 and landed on the North Shore of O‘ahu on 9/20/24. The total Euclidean distance between GPS fixes was 273.81 km. On September 20th she was spotted at a vacation home at Sunset Beach on the North shore reported by a tourist to the Hawai‘i Wildlife Center. She remained at a pond in the Turtle Bay Golf Course from the 21st to the end of her transmissions on 9/28/24. We could not determine if this was due to a failed transmitter or mortality.

Another Hawaiian coot (ZA:BB, transmitter #261785) from Pa‘akai had a long distance movement off of MCBH (Figure 8). She first moved to Sag Harbor (MCBH) on 11/13/2024 and had two last points transmitted from the North Shore on 11/17/2024. The last two points were on James Campbell National Wildlife Refuge. The total Euclidean distance traveled was 38.26 km. Hawaiian coot GG:ZA (transmitter #261788) was captured at Kaluapuhi, as a hatch year female on 1/16/2025. She was not fully fledged yet and had pin feathers growing in on her wings when captured. But she flew to one of the wetland units at James Campbell National Wildlife Refuge on 6/5/2025, flying 35.93km between an 8 hour span (Figure 8). She was spotted by James Campbell staff utilizing managed wetland areas, before flying northwest off O‘ahu toward Kaua‘i. Her last four points of transmission were in the ocean 141.2 km west of O‘ahu.

Hawaiian coot ZA:GO (transmitter #261787) was captured and transmitted 1/16/25 during a large population influx. She flew from Pa'akai to the east side of Kaua‘i on 2/7/2025 (Figure 8). The next day she flew across the island to the west side of Kaua‘i near the Pacific Missile Range Facility at Barking Sands, then settled at Mānā Plains which is a wetland restoration site managed by DLNR DOFAW. She remained at Mānā Plains until her transmissions ended on 6/23/2025. The total Euclidean distance traveled from Pa'akai to Mānā Plains was 232.91 km.

Appendix A2:

Long Distance Movements Gallinule

Hawaiian gallinule BG:ZA (transmitter # 239070) left TLF wetland and made her first stop in a residential area near the Friendship Garden trailhead on 5/6/2023 (Figure 9). Then she made her next flight to Kawainui Marsh Wildlife Sanctuary on 5/7/2023 and stayed for an extra day. Afterward, she flew to Waipuna wetland on MCBH on 5/8/2023-5/9/2023. She made a stop at Perc Ditch on 5/9/2023, before returning to TLF on 5/10/2023, where she stayed until at least 04/14/2025. She moved a total of 13.13 km (Euclidean distance) in a five day span.

APPENDIX 2

Table B1: Summary statistics of abiotic and biotic variables taken at Hawaiian gallinule nests at MCBH from January 2023-March of 2025 during weekly nest searches from January -June.

Variable	Minimum	Maximum	Median	Mean	Standard Deviation
Native(% 1m^2)	0	100	50	44.06	37.25
Non-native(% 1m^2)	0	100	20	39.63	41.63
Abiotic(% 1m^2)	0	84	5	15.89	20.72
Water Depth(cm)	2.54	72.39	10.92	16.06	13.83
Distance(m)	0	8.43	1.45	1.85	1.95
VOR average	5.5	83.75	37.5	40.49	16.91
Dissolved oxygen (mg/L)	3.31	16.9	7.13	7.207	3.12
Specific conductance(µS/cm)	7089	55501	16197	20640	12,558.43
Salinity(ppt)	3.86	36.59	9.44	12.52	8.49
pH	7.34	8.77	7.74	7.762	0.33
Turbidity (NTU)	0.34	125.87	3.26	20.53	38.20

Table B2: Summary statistics of abiotic and biotic variables of non-nesting locations. Four habitat points were collected per nest within a 27 m radius of the nest in available nesting habitat, quadrants were divided by cardinal direction, and locationss were randomly sampled. A total of 104 non-nesting locations were sampled.

Variable	Minimum	Maximum	Median	Mean	Standard Deviation
Native(% 1m^2)	0	100	0	15.63	25.96
Non-native(% 1m^2)	0	100	75	65.10	32.17
Abiotic(% 1m^2)	0	97	10	18.55	25.50
Water Depth(cm)	1.27	74.93	13.4366	14.14	8.67
Distance(m)	0	26.0604	2.7178	4.07	4.49
VOR average	0	100	30	35.41	27.01
Dissolved oxygen (mg/L)	2.1	14.41	5.395	5.98	3.00
Specific conductance(µS/cm)	10.7	82326	20320	23,818.24	14,094.23
Salinity(ppt)	0.51	35.08	12.095	13.86	7.58
pH	7.14	8.91	7.695	7.76	0.43
Turbidity (NTU)	0.1	68.35	6.92	12.07	13.58