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Mammal-Exclusion Fencing and the Reproductive Success of an Endangered Native Waterbird

By
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Abstract/Motivation

Novel relationships in ecological communities are forming faster than historical rates due to globalization and the resulting increase in species introductions. In the Hawaiian Islands, which prior to humans had no terrestrial reptiles or amphibians and only one terrestrial mammal, the introduction of invasive predators dramatically impacted island food webs. Wetlands, as ecosystems where terrestrial, aquatic, and marine species intersect, were particularly impacted by introduced species. The Hawaiian Stilt (*Himantopus mexicanus knudseni*) or Ae‘o, is one of five Hawaiian waterbirds listed under the Endangered Species Act. Currently estimated to range from ~1,300 to ~1,800 individuals, the Hawaiian Stilt must reach a self-sustaining population of 2,000 birds in order to be delisted. One factor hindering recovery may be the predation of Stilt eggs and chicks by invasive predators. To address this threat, the U.S. Fish and Wildlife Service recently constructed a fence that excludes mammalian predators around a wetland on O‘ahu. Although most island avian communities respond positively to invasive mammalian predator removal, mammal-exclusion fencing is an expensive tool, and only controls for one type of predator – mammals. Avian, aquatic, and amphibious predators may still impact birds inside the conservation fencing. In this study, I compared the reproductive success of Stilts nesting inside and outside the newly built mammal-exclusion fence to test hypotheses regarding the impact of invasive predators and estimate the effectiveness of mammal-exclusion fencing as a management action. The results of the proposed research should help to inform management decisions regarding which predator control tools will be most cost-effective in a given scenario, by identifying the improvement in reproductive success of the Hawaiian stilts nesting inside versus outside of a mammalian predator exclusion fence.

Introduction

Fencing that excludes mammalian predators, often referred to as predator-proof fencing or pest-proof fencing, prevents the predation of eggs, chicks, and adult waterbirds by excluding invasive mammalian predators (Burns, Innes, and Day 2012). However, avian, aquatic, and amphibious predators may still access potential prey inside fences (Figure 1). Thus, there remains a knowledge gap regarding the potential gains in reproductive success from excluding mammalian predators, in comparison to nearby locations where mammalian predators are controlled via trapping and removal – an approach in which cats, rats, and mongooses may still impact nesting success before removal. In this study we quantify the reproductive response for one of Hawai‘i's Endangered waterbird species inside and outside of mammal-exclusion fencing. Monitoring ecological communities and research into how they respond to such management actions is essential in quantifying the cost effectiveness and measuring any reduction in extinction risk.

Island species dependent upon wetlands have been impacted by the introduction of invasive predators and competitors, following increased globalization and a corresponding increase in the rate of nonnative species introductions (Salo et al. 2007). Thus, recovery programs for endangered species in island systems often include invasive predator removal, particularly during reproductive phases of the threatened species (Holt et al. 2008). Native bird responses to predator removal vary (Meckstroth and Miles 2005), but ground nesters tend to benefit most from predator removal (Lavers, Wilcox, and Donlan 2010). However, targeted removal of a single predator type, such as cats, may lead to predator release of rats or other mesopredators, which may increase in number and increase predation on eggs or chicks, further impacting native bird populations (Rayner et al. 2007). Thus, predator control approaches that target multiple species or exclude entire taxonomic groups of invasive predators, such as mammal-exclusion fencing, may be warranted in some cases, despite a high cost (Smith et al. 2010; Young et al. 2013).

With the increased use of mammal-exclusion fencing worldwide, there is a need to understand when it should be utilized (Côté and Sutherland 1997; Scofield, Cullen, and Wang 2011; Innes et al. 2012). Depending on the life stage, major threats to a species may differ and thus management actions meant to mitigate those threats can also differ. Multiple methods for invasive predator removal are available, including traps, fences, and poison baiting, each with varying costs, secondary impacts, and varying effectiveness at reducing mortality in the protected species. For highly endangered species in which predation significantly contributes to extinction risk, fencing may be the most effective option for species recovery, despite its high cost. In contrast, in species nearing population sizes that may warrant delisting, the question remains whether mammal-exclusion fencing is a cost-efficient option for recovery. Resource managers use changes in population through time to measure the effectiveness of management actions (Reed et al. 2007); however, there may be a lag in population growth immediately following conservation actions. Measures of reproductive success and mortality provide more immediate measures of potential changes in population demographics. Studies of target species following the implementation of a management action allows resource managers to evaluate its cost effectiveness in mitigating extinction risk.

In the Hawaiian Islands, the introduction of nonnative predators, loss of habitat, and hunting of waterfowl by humans, led to the extinction of many native waterbird species (Groombridge 1992; O'Donnell, Clapperton, and Monks 2014). Waterfowl hunting was banned in 1939 and by 1970 the federal government had listed four of Hawaiʻi's native waterbirds as Endangered (USFWS 2011). Recovery efforts have focused on habitat acquisition and restoration as well as invasive predator control. Eradication of invasive mammalian predators on the main Hawaiian islands is not yet feasible (Reed et al. 2012). Thus, Hawaiian waterbirds are considered conservation-reliant and perhaps would not persist in the absence of management actions such as invasive predator removal (Underwood et al. 2013).

In June 2018 the U.S. Fish and Wildlife Service (USFWS) completed a mammal-exclusion fence around the 1,006 m perimeter of the Honouliuli wetland unit within the Pearl Harbor National Wildlife Refuge (PHNWR) complex on Oʻahu. The fence cost $500,000 to construct, not including maintenance, and has a life expectancy of about 20-25years depending on the proximity to the coast. This mammal-excluding fence offers the ability to compare reproductive success with the nearby Waiawa wetland unit, also within the PHNWR, in which trapping, and removal of mammalian predators is the main form of invasive predator removal. In both locations, avian, aquatic, and amphibious predator types are not controlled.

The USFWS Recovery Plan for the Aeʻo or Hawaiian Stilt (*Himantopus mexicanus knudseni*) requires that a minimum population of 2,000 individuals must be reached before delisting (USFWS 2011). Available wetland habitat has increased substantially since the 1970's and the population was estimated around 1,500 individuals in 2011 with a slowly increasing trend (Reed et al. 2015; Reed et al. 2011; USFWS 2011). However, because of the extensive list of potential predators, both native and non-native (Table 1), wetland sites may not be reaching full recruiting potential. More effective invasive predator control may further increase reproductive success, helping the Hawaiian Stilt to achieve recovery goals.

Hawaiian Stilts have the capacity to re-nest, combine egg clutches, and aggressively respond to different predator types, suggesting they may be able to withstand considerable predation pressure, similar to the Black-necked Stilts (*Himantopus mexicanus*; (Coleman 1981; Sordahl 2004). Hawaiian Stilts typically lay four eggs over the course of four days (one egg per day), begin their incubation period after the third or fourth egg is laid, and hatching is said to be ‘fairly synchronized’ within ~24 hours (Coleman 1981). During the hatching period, adult Stilts do not provision hatchlings and chicks are often foraging nearby as adults continue to protect the nest. Thus, the hatching period is one of the most vulnerable phases of life for ground nesting birds. In this study we compared nesting success during the hatching period between a location with a mammal-exclusion fence, and a location nearby without a mammal-exclusion fence, where trapping alone is used to control mammalian predators.

Hypotheses

- H_{null}: There will be no difference in reproductive success between Hawaiian Stilts inside and outside of mammal-exclusion fencing.
- H_A: Reproductive success will be greater for Stilts inside the mammal-exclusion fence.
- H_{null}: Mean time spent by the chicks in the nesting area will not differ between Hawaiian Stilts inside and outside of the mammal-exclusion fence.
- H_A: Stilt chicks inside the mammal-exclusion fence will spend more time in the nesting area when compared to Stilt chicks outside the mammal exclusion fence.

Methods/Approach

Study Sites

This study was conducted in two wetlands on the island of Oʻahu, Hawaiʻi, USA (Figure 2). Honouliuli and Waiawa wetlands are both located within the Pearl Harbor National Wildlife Refuge (PHNWR) complex and were chosen because of their shared region on the island, similar rainfall regimes, similar proximity to urban development, and shared management under USFWS. Water levels are managed in both wetlands and trapping of mammalian predators occurs regularly in a similar manner by the same employee(s). Construction for a 1,006 m (3,300 ft) mammal-exclusion fence around the perimeter of Honouliuli was finished in June 2018 and mammalian predators are almost entirely eradicated from the wetland with only a few remaining mice.

The Honouliuli unit is nearly 14.7 hectare (~36.3 acres) and is typically the fresher of the two wetlands (0-10 ppt) while the Waiawa unit is about 9.9 hectares (~24.5 acres) and is typically more brackish (40-117 ppt) (Coleman 1981). Plant community composition differs between sites with ʻAeʻae or water hyssop (*Bacopa monnieri*), non-native pickleweed (*Batis maritima*), and cattails (*Typha spp.*) dominating Honouliuli, while Waiawa is dominated almost exclusively by non-native pickleweed (*B. maritima*) with California grass (*Brachiaria mutica*) along the perimeter. The avian community between the two sites is similar in species richness year-round.

Data Collection

During the Hawaiian Stilt breeding season of 2019, which runs from March until August (Coleman 1981), active nests were identified by surveying for incubating adults. A survey consisted of driving around a managed area and spotting incubating adults with binoculars or field scopes. Once incubating adults were identified, foot searches were used to confirm nests. For each nest GPS coordinates were recorded and a Bushnell HD No-Glow Trophy Camera was placed approximately three meters (10 ft) from the nest, mounted on a pole roughly 1.2 meters (4 ft) off the ground, and pointed downward to get a direct view of the nest and a broader view of the nesting area (Figure 3). The *nesting area* was defined by the camera's field of view. If the view was too wide than the nesting area was defined by the area around the nest within reason. Cameras were used to monitor nests for the number of eggs laid, the number of eggs hatched, the time spent by chicks in the nesting area, predation events, predator types that visited the nesting area, and the frequency of predator visits. Cameras were checked one or two times per week to maintain memory and battery power. Cameras were programmed to take one photo every five minutes, one photo after being triggered and another photo after a two-second delay following being triggered. Preliminary data from 2018 were used to define the *hatching period* as the mean time spent in the hatching area plus two standard deviations giving a conservative estimate of ~87 hours after the first chick hatched. All nests that had at least a single egg hatch were monitored for a minimum of 10 days after hatch. If adults continued to incubate a nest past 10 days, data were collected until the nest reached a fate or the camera was needed on a newer nest. All photos were processed using VirtualDub open source software.

Statistical Analysis

All analyses were performed in the program R version 3.5.1. (R core team, 2013). R packages included “readr,” “plyr,” “dplyr,” “Rmisc,” “ggplot2,” and “car”. Pearson’s t-tests were run to compare the number of eggs laid per nest, the number of eggs hatched per nest, and the proportion of eggs hatched per nest between sites. For each site, paired t-tests were used to compare the number of eggs laid to the number of eggs hatched for each nest within that site.

Results

As predicted, there were a significantly greater number of eggs hatched per nest ($t = 4.71$, $p < 0.001$, $df = 25$) as well as a greater proportion of hatched eggs per nest ($t = 4.37$, $p < 0.001$, $df = 26$; Figure 4) inside the mammal-exclusion fence than at the site without fencing. Surprisingly, there were significantly greater number of eggs laid per nest inside the fence ($t = 3.16$, $p = 0.005$, $df = 20$; Figure 5). These values are summarized in Table 2. Intuitively, the number of eggs that hatched were significantly less than the number of eggs laid per nest for both inside the fence ($t = 2.31$, $p = 0.050$, $df = 8$) and at the site without fencing ($t = 7.63$, $p < 0.001$, $df = 20$). Inside the fence, 44.4% of nests had at least one egg hatch and 55.6% of nests had the full clutch hatch. At the site without mammal exclusion fencing, only 14.3% were full clutch hatches, 33.3% had at least one egg hatch, 28.6% were depredated and the other 23.8% were abandoned, flooded or had unknown fates (Figure 6). Unexpectedly, the time spent by the Stilt chicks in the nesting area did not differ significantly between sites ($t = 0.59$, $p = 0.56$; Figure 7).

Discussion & Conclusions

Our results suggest that mammal-exclusion fencing results in greater nesting success than trapping mammalian predators alone. Although trapping is continuous, there may be some delay before mammalian predators are removed allowing opportunity for predation prior to removal. The trapping and removal of waterbird predators from sites may allow for predator-release (Rayner et al. 2007) if predators from adjacent habitats readily recolonize the site. The loss in trapping effectiveness may be greater still in systems near urban development that have a ready source of rats, feral cats, and dogs to fill the void. Reed et al. (1998) conducted a sensitivity analysis and found parameters such as catastrophic stochastic events, maximum age and density dependent reproduction had “little effect” on the Hawaiian Stilt projections. However, the likelihood of Stilts persisting more than 200 years dropped notably when clutch failure rate or first-year mortality rate increased above 70%. Surprisingly, clutch sizes laid by breeding pairs within the mammal-exclusion fencing were significantly greater than those without an enclosure ($p=0.005$). Perceived predation risk has been shown to invoke a physiological response that can negatively impact clutch size and adult fitness (Zanette et al. 2011; Thomson et al. 2010).

Having colonized the Hawaiian islands around 750,000 years ago (Price and Clague 2002; Fleischer and McIntosh 2001), Hawaiian Stilts would have had several millennia without exposure to mammalian predators. Stilts have been documented defending against avian predators such as the native ‘Auku‘u or Black-crowned Night Heron (*Nycticorax nycticorax*) or the Hawaiian Short-eared Owl (*Asio flammeus sandwichensis*) also known as the Pueo. In our study, we observed defensive behaviors by nesting adult Stilts. Potential avian and amphibious predators were confronted and typically met with mobbing (camera data) while perceived mammalian predators, including humans, received weaker responses such as circling and dive bombing. Our research was unable to quantify a varied response to predator types but would have likely been consistent with North American literature (Sordahl 2004). However, in the absence of mammalian predators, non-native avian, aquatic and amphibious predators may experience predator-release inside mammal-exclusion fences and could be present in higher numbers inside of mammal-exclusion fences. Regardless, we would still expect some nest failures from avian, aquatic or amphibious predators, nest abandonment, and/or flooding. Thus, we expected the number of eggs laid versus hatched within a site to differ greatly. Without mammal exclusion fencing, paired t-tests indicated fewer hatchlings than eggs as expected. Surprisingly, our results for inside the fence indicated a *barely* significant difference between the number of eggs laid vs hatched ($p = 0.050$, $df = 8$). This may be due to the Stilt’s potential to respond to avian and amphibious predators.

At the site level, nest detection within the wetland was high but it is possible that some nests were not discovered. At the nest level, camera traps missed predation events and the field of view varied. Detection of predators at both sites were too low for any comparative analysis. Nest densities differed between sites. The sites were nearby, but differences between sites may occur in food availability or other factors that could influence nesting success.

Over the last four decades, National Wildlife Refuges and the State of Hawaiʻi have expanded protected lowland wetland areas to over 6,000 ha in which the endangered waterbirds commonly use. State and federal managers continue to restore degraded wetlands in hopes to increase suitable nesting habitat. Reed et al. (1998) modeled Hawaiian Stilts as both a single population and a metapopulation of six subpopulations both growing to fill the available habitat but were unable to reach the recovery goal of 2,000 individuals. If carrying capacity is indeed the limiting factor, this may warrant future funds being utilized towards the acquisition, restoration and/or the creation of additional wetland habitats instead of the construction of additional fencing.

Scofield, Cullen, and Wang (2011) criticized New Zealand's expenditure on mammal exclusion fencing emphasizing that they are 'little more than expensive zoos.' Innes et al. (2012) responded by detailing the role of mammal exclusion fencing in acting as short-term interventions when threatened taxa face extinction. The authors also argue that fencing can be important for capturing public interest and support. It is impossible for mammal exclusion fencing to restore ecosystems to their original, pristine state because many taxa that existed in island systems before human contact have since become extinct and their ecological niche or function lost. Instances do exist where mammal exclusion fencing is ineffective in the conservation of waterbirds (Sanders, Brown, and Keedwell 2007)

Managers must consider the *cost-effectiveness* or the impact per dollar spent. The scale of the protected area, the life expectancy of the fence, initial construction costs and the continuous maintenance costs are important factors. However, the number of species protected, the number of individuals per species protected and any potential gains in reproductive success must also be considered in evaluating the cost effectiveness. Clapperton and Day (2001) estimated the payoff for fences to be between 4-6 years and did not recommend fencing areas smaller than 5000 hectares. The authors did not discuss the number of species protected in any particular case study nor any gains in reproductive success. Young et al. (2013) estimated the payoff for a fence protecting shorebirds to be nearly 16 years and the life expectancy of fence to be 20-25 years.

The Honouliuli unit is nearly 14.7 hectare (~36.3 acres) and near the coast making the raw materials susceptible to degradation, potentially diminishing returns. After considering the many factors and opportunity costs, decision makers can decide when the gain in reproductive success per species per dollar spent is “worth it.”

This study suggests mammal-exclusion fencing may be a useful tool to help recover endangered waterbirds, by increasing nesting success. For other Endangered waterbird species that breed year-round inside the enclosure, the recruitment potential could be substantial. The study results, though limited in sample size inside the fence, also suggest stress may be lower in birds inside the mammal-exclusion fence, potentially increasing clutch size. A banded population is essential for estimating recruitment inside and outside the mammal-exclusion fences. Future research into chick mortality and causes of death in and out of the fence would lend another metric to bring context to invasive biology in the Hawaiian Islands. As complete island-wide eradication of mammalian predators is not feasible at this time, Hawai‘i's Endangered waterbirds are conservation reliant and predator management must be sustained for waterbirds to persist. Continuous monitoring of waterbird populations over time is necessary for adaptive management.

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Tables & Figures

Table 1: A list of all potential predators of Hawaiian Stilts.

Potential Predators				
Name		Predation Phase		Cited Reference
Common	Scientific	Eggs	Chicks	
*Domestic Dog	*Canis familiaris*	x	x	Coleman 1981
*Domestic Cat	*Felis catus*	x	x	Coleman 1981
*Indian Mongoose	*Herpestes auropunctatus*	x	x	Coleman 1981
*Rat	*Rattus spp.*	x		Coleman 1981
*Bullfrog	*Rana catesbeiana*		x	USFWS 2009
*Cattle Egret	*Bubulcus ibis*	x	x	Coleman 1981
Black-crowned Night Heron	*Nycticorax nycticorax*	x	x	Coleman 1981
Hawaiian Short-eared Owl	*Asio flammeus sandwichensis*	x	x	Coleman 1981

*Barn Owl	*Tydo alba*		x	Robinson et al. 1999
*Common Myna	*Acridotheres tristis*	x	x	Coleman 1981
Ruddy Turnstone	*Arenaria interpres*	x		Coleman 1981
Laughing Gull	*Larus atricilla*	x		Coleman 1981
Catfish	*Clarias spp.*		x	USFWS 2011

* Indicates a non-native introduced species.

Table 2: Means and standard error (±SE) for the eggs laid, eggs hatched and the proportion of eggs that hatched are summarized by site. Significant differences between the two sites is indicated by the p-value on the bottom row.

Site	$\bar{X}$ Eggs laid (± SE)	$\bar{X}$ Eggs Hatched (± SE)	Prop. of Eggs Hatched (± SE)
Honouliuli (mammal-exclusion fence), n=9	4.00 (± 0.00)	3.33 (±0.29)	0.83 (± 0.07)
Waiawa (no mammal-exclusion fence), n=21	3.67(± 0.11)	1.29 (± 0.33)	0.34 (± 0.09)
	$p = 0.005$**	$p < 0.001$**	$p < 0.001$**

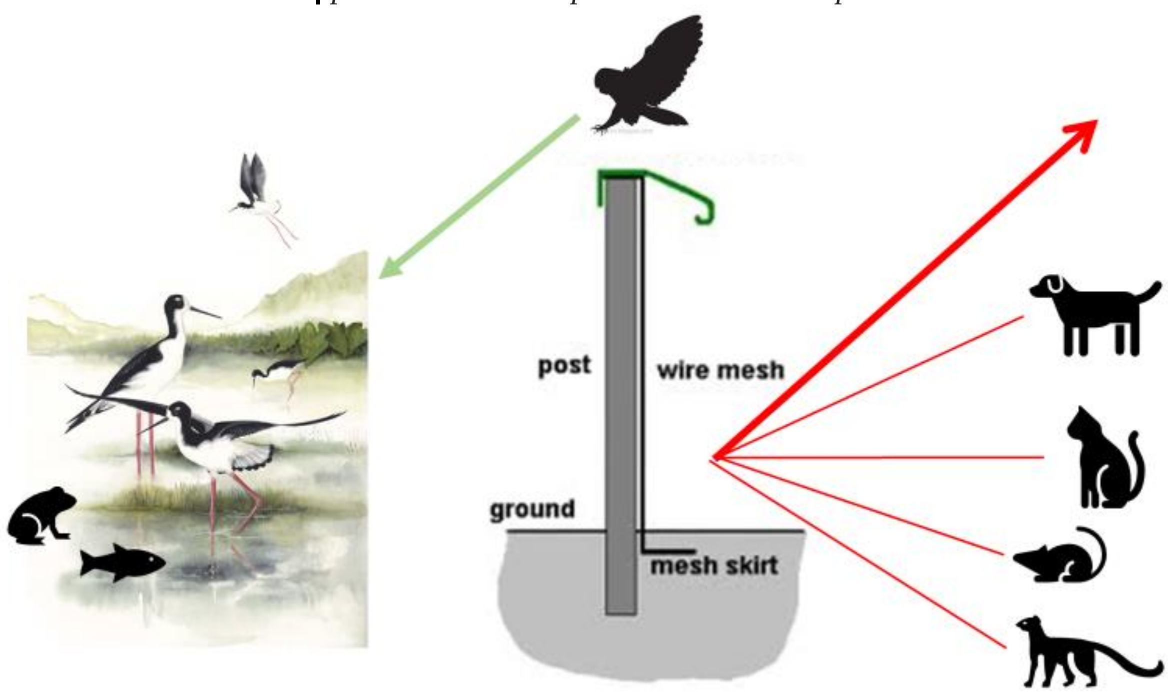

Figure 1: A conceptual picture of threat dynamics. Painting by Marian Berger.

Figure 2: An image of O'ahu and the study sites within the Pearl Harbor National Wildlife Refuge (PHNWR). Waiawa does not have a mammal-exclusion fence and Honouliuli does.

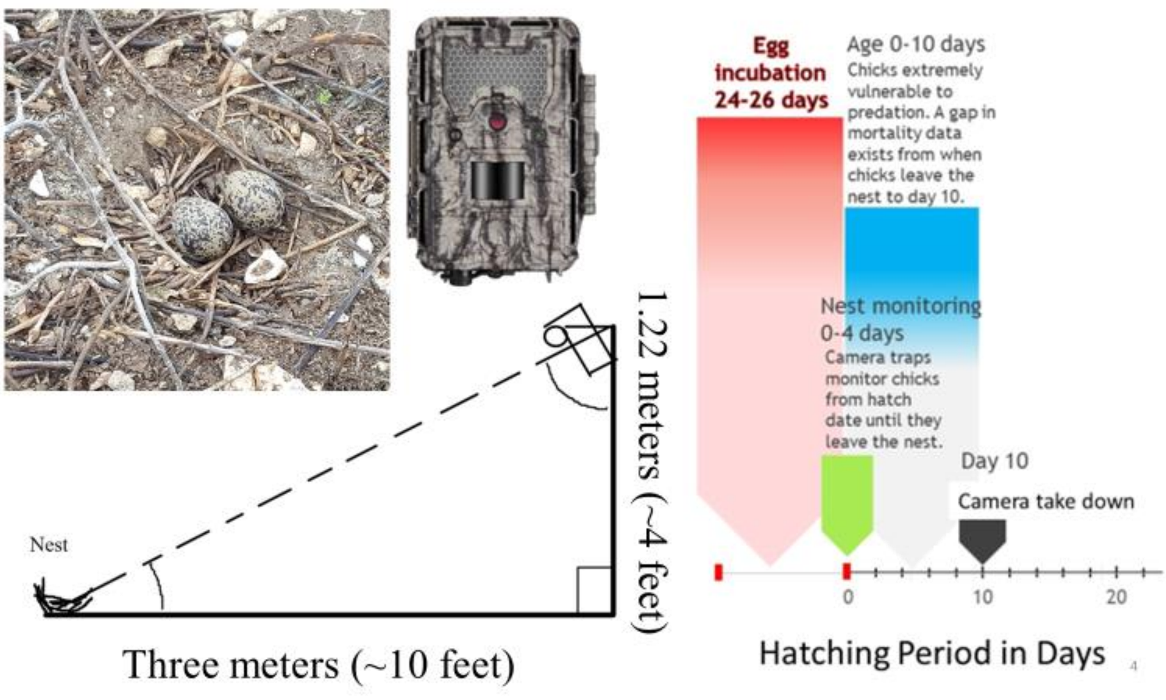

Figure 3: A photograph of a Hawaiian Stilt nest with two eggs, a picture of a Bushnell game camera, a conceptual drawing of a game camera mounted on a wood post, and a life history timeline highlighting the hatching period.

2019 Eggs laid & Hatched

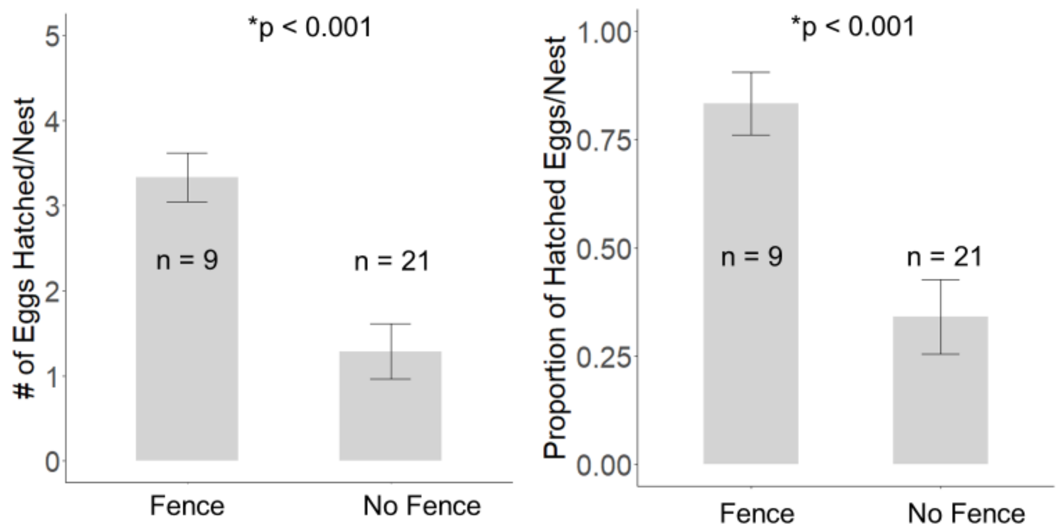

Figure 4: Bar plots depicting the numbers of eggs hatched and the proportion of eggs hatched per nest at each site for 2019 with standard error whiskers.

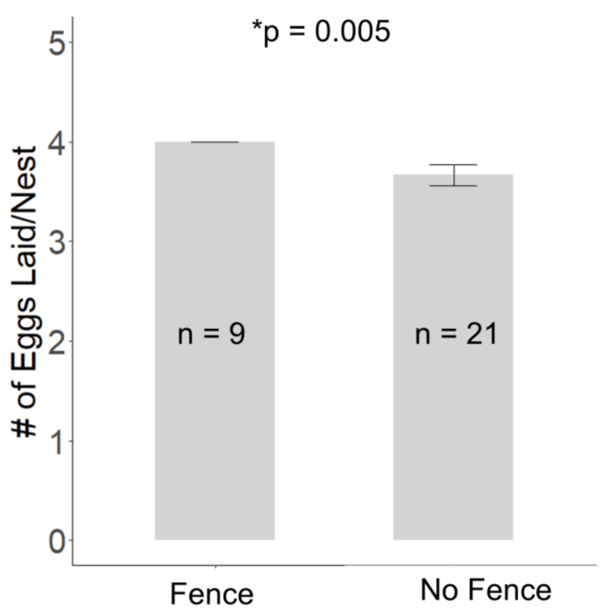

Figure 5: Bar plot of the number of eggs laid per nest by site with standard error whiskers.

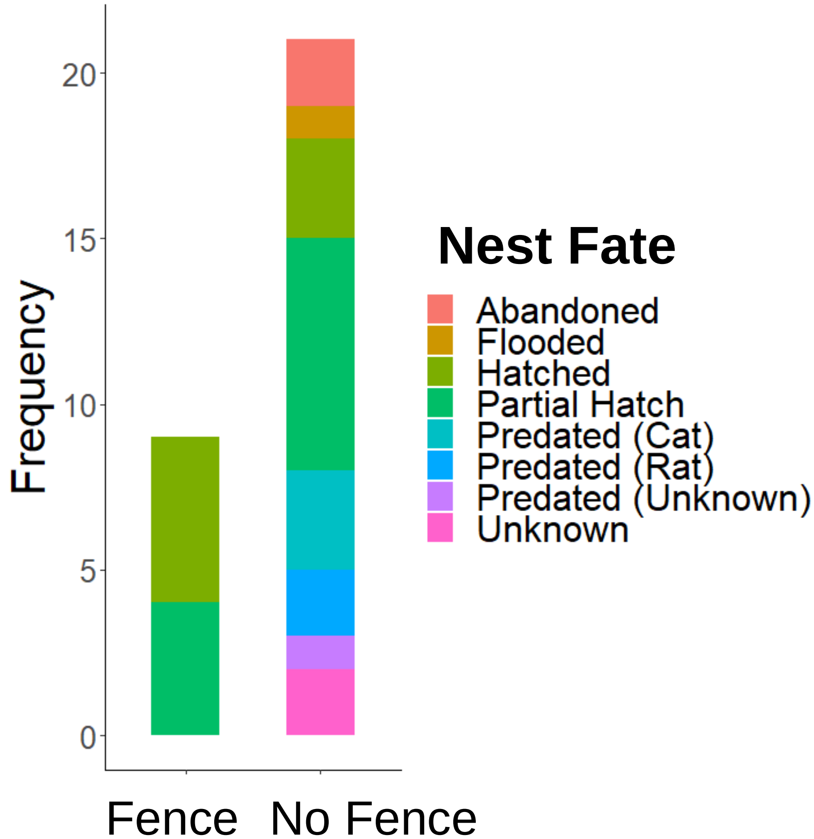

Figure 6: A stacked bar chart indicating the fates of the nest at each site (n = 30).

Time spent in the nesting area (2019)

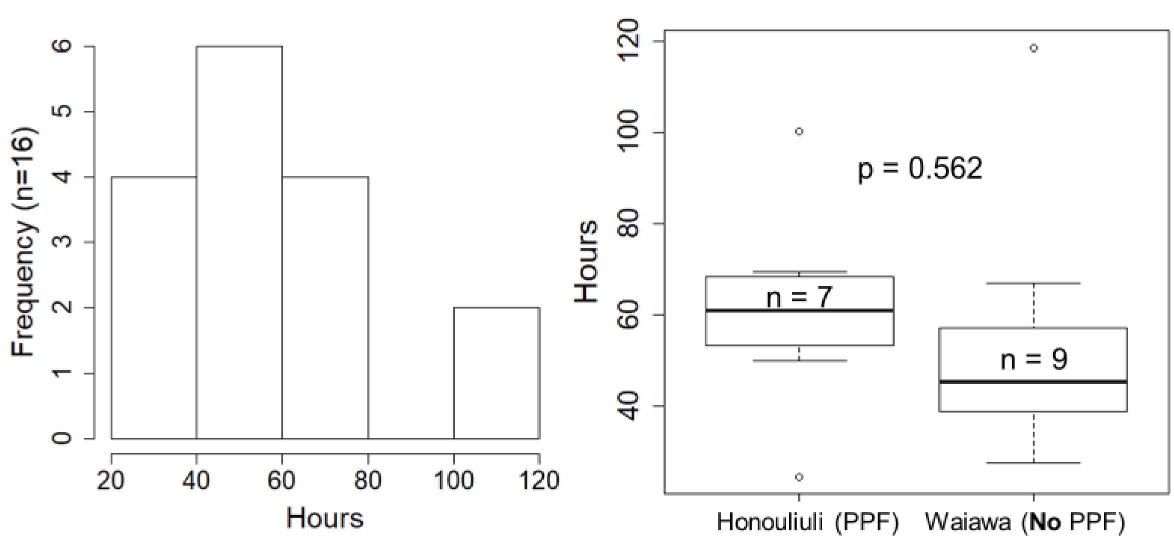

Figure 7: A histogram and bar plots comparing the hours spent in the nesting area for each site. ($\bar{X}$ = 57.16; range =24.5-118.66)