Responses of Brown Pelican (*Pelecanus occidentalis)* Nesting Ecology and Habitat to a Major Hurricane

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Abstract

Barrier islands in coastal Louisiana provide breeding habitat for colonial nesting waterbirds such as the Eastern Brown Pelican (*Pelecanus occidentalis carolinensis*). This study examined the nesting success of Brown Pelicans before and after Category 4 Hurricane Ida across three barrier islands in coastal Louisiana. Two islands have been restored (Raccoon and Queen Bess) and Philo Brice remains unrestored. I collected data from 387 monitored nests over three years (2021–2023). Apparent Nest Success (ANS), Daily Survival Rate (DSR), and Fledging Success (FS) were all significantly higher after the hurricane than before it. Nest success varied by island and year, showing the strongest gains on Philo Brice, despite its proximity closest to the storm's path and unrestored status. Nest success was overall higher on all islands after the storm. Causes of nest failure varied but were most often linked to flooding, storm impacts, and human disturbance, especially on Queen Bess. Philo Brice, despite being closest to the storm, had the lowest nest failure rate and no failures due to human activity. To investigate habitat suitability and fluctuations, I also examined vegetation type and cover on each island pre- and post-Ida from surveys in multiple habitat types. Remote sensing data showed dynamic shifts in island areal extent and fluctuations in plant community structure. Overall reproductive success of Brown Pelicans increased after Hurricane Ida in Louisiana revealing ecological resilience and efficacy of restoration efforts. These results suggest that the storm did not have a long-term negative effect on Brown Pelican nesting success.

Contrary to my original predictions, Hurricane Ida may have, in some areas, indirectly improved conditions through vegetation changes. However, continued monitoring on these remote islands is needed to understand longer-term trends.

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CPRA	Coastal Protection and Restoration Authority
DSR	Daily Survival Rate
ha	hectares
km	kilometers
LDWF	Louisiana Department of Wildlife and Fisheries
NDVI	Normalized Difference Vegetation Index
NOAA	National Oceanic and Atmospheric Administration
RIFA	Red Imported Fire Ant (*Solenopsis invicta* Buren)
RRSLR	relative sea level rise

General Introduction

Global sea levels are rising as a result of climate change, natural land subsidence, and anthropogenic-induced subsidence (Michener *et al.* 1997; Kirwan & Megonigal 2013; Mentaschi *et al.* 2018; Nicholls *et al.* 2021). Rising atmospheric temperatures due to industrial process have resulted in increased global sea level rise (RSLR) and extreme weather events (Bevacqua *et al.* 2020; IPCC 2023). Sea level rise results in erosion of soils, saltwater intrusion into freshwater systems, coastal flooding, plant stress, and ultimately decreased elevation (Nyman & DeLaune 1999; White & Kaplan 2017; Mentaschi *et al.* 2018; Taherkhani *et al.* 2020). The increased variability and frequency of extreme weather events have negatively impacted both human and natural coastal communities making protection and conservation efforts challenging (Weinstein *et al.* 2007; Feagin *et al.* 2010; Kirwan & Megonigal 2013; Maxwell *et al.* 2019).

Coastal ecosystems are dynamic and modified by relative RSLR, climate change, fires, wind, waves, human development, and storms (Sleeter *et al.* 2018; Reimann *et al.* 2023; Brown *et al.* 2024). Coastal wetlands provide services such as protection to coastal communities during tropical cyclones, mitigation against RSLR, commercial and recreational hunting and fishing, carbon sequestration, and habitat for flora and fauna (Kathiresan & Bingham 2001; McLeod *et al.* 2011; Seitz *et al.* 2014; Sheng *et al.* 2022). In addition to these ecosystem services, nearly 45% of the global human population lives within 150 km of the coast subjecting these communities to the hazards of RSLR and damage from extreme weather events (e.g. hurricanes, tsunamis, or flooding) (Cosby *et al.* 2024). The rate of coastal land loss and frequency of meteorological disasters is increasing and predicted to increase in the future from anthropogenic forcing (Bevacqua *et al.* 2020; Lima *et al.* 2021; Bhatia *et al.* 2022). Therefore, it is essential to implement conservation and management actions to protect and restore coastal areas.

Coastal landscapes (here defined as saltwater or brackish-water marshes with tidal influence) are often managed or restored for protection against storms, flooding, wildlife habitat restoration/creation, and carbon sequestration (Barbier *et al.* 2011; McLeod *et al.* 2011). Beach nourishment involves the placement of sediment to expand or restore a desired land size and elevation (Brown *et al.* 2024) and is a widely accepted method for coastal protection in the United States (Elko *et al.* 2021; Miselis *et al.* 2021), Europe (Hamm *et al.* 2002), Australia (Cooke *et al.* 2012), and China (Luo *et al.* 2016). Beach nourishment is a dynamic method involving the periodic placement of sediment, either by dredge or dump truck (Willson *et al.* 2017). Sediment is borrowed from a nearby source to increase the width of beaches undergoing erosion (Elko *et al.* 2021). Because these sediments will eventually erode over time due to wave action, storms and RSLR (Miner *et al.* 2009), in some sites renourishment projects are often required and may be useful only as an interim strategy as opposed to a long-term management solution (Willson *et al.* 2017; Parkinson & Oguracak 2018).

Dune management consists of enhancing existing dunes or creating artificial dunes with the primary purpose of defense against increased storm-driven water levels (Miselis *et al.* 2021). In order to create or restore dunes, sediment can be scraped with machinery from the beach to the dune line or vegetative plantings and sand fencing can be used to increase sand accumulation (Psuty & Silveira 2013). To mimic natural coastal ecosystems, native vegetation plantings are conducted alone or after sediment placement takes place to benefit target fauna species, stabilize shorelines, and minimize erosion (Kodikara *et al.* 2017; CPRA 2023).

Mangrove forests protect coasts against erosion and have declined globally due to deforestation and degradation (Lovelock *et al.* 2022). Restoration can include mangrove plantings, however success rates are site-specific and require post-restoration maintenance (Kodikara *et al.* 2017; Lovelock *et al.* 2022). Large-scale mangrove restoration projects (>1,000 ha) have been used after large natural disasters, such as the deadly Indian Ocean tsunami in 2004, but do not always prove effective. Small-scale projects (<1,000 ha) can have local benefits for storm protection and wildlife habitat, although fragmentation of wetland habitats may not deliver the desired landscape-scale benefits (Lovelock *et al.* 2022). Shoreline stabilization is critically important during tropical storms and hurricanes, and within mangrove stands, the vegetation structure, i.e., salt marshes with grasses and succulents or mangroves dominated by woody vegetation, influences the amount of wave attenuation (Kathiresan & Bingham 2001; Bao 2011; Doughty *et al.* 2017; Cahoon *et al.* 2020; Osland *et al.* 2022). For example, after Hurricane Harvey hit coastal Texas in 2017, Pennings *et al.* (2021) found areas dominated by mangroves had higher soil strength post-storm and prevented erosion better than areas dominated by marsh plants or non-vegetated areas. The relationship was nonlinear, with low mangrove cover providing most of the benefit of wave attenuation (Pennings *et al.* 2021). In the specific case of restoring land in deltaic regions, marsh nourishment can take place by the diversion of sediment-rich river water (riverine reintroduction) with the goal to increase elevation and create or maintain land that is suitable for vegetative growth (Miselis *et al.* 2021). For instance, to combat saltwater intrusion caused by agriculture and the construction of dikes in the Danube delta, sediment diversions successfully restored wetlands, fisheries, and raised elevation (Ebert *et al.* 2009).

Geomorphological features such as shoals or oyster and/or coral reefs naturally attenuate wave energy, but in regions where these structures do not provide adequate storm protection or have lost protection altogether, structural engineering methods are implemented (Koch *et al.* 2009; Wiberg *et al.* 2019). Methods to break wave-induced erosion include the placement of hard physical barriers, e.g., groins, jetties, riprap or submerged breakwaters (Stauble *et al.* 2003). Jetties are implemented to reduce wave action and improve navigation safety, however erosion control is directional. Jetties can increase beach accretion on the updrift side of the jetties; which, alternatively, can cause erosion in downdrift areas due to inadequate sediment transport (Saengsupavanich *et al.* 2024). When natural barriers are lost, multiple hard engineering methods can be combined. For instance, the loss of sand bars in low-lying coastal Florida resulted in the subsequent loss of seagrasses (Koch *et al.* 2009). This loss of natural coastal protection, necessitated implementation of seawalls, large boulders (riprap), breakwaters, jetties, dredged sediment placement, and restoration of natural vegetation (Sealey *et al.* 2021).

Globally coastal marsh loss occurs at an alarming rate (Nicholls *et al.* 2021), with high rates land loss and shoreline retreat in the northern Gulf of America (formerly named Gulf of Mexico, hereafter referred to as the Gulf), specifically the state of Louisiana (Yuill *et al.* 2009; Couvillion *et al.* 2017; Dietz *et al.* 2018). Marsh conversion to open water has been documented in the Mississippi River Delta, with Louisiana experiencing some of the highest levels of sea level rise in the Northern Gulf (Day *et al.* 2007). From 1932 to 2010, 4,877 km^2 of land eroded or subsided and it is projected to lose as much as 10,670 km^2 (nearly two-thirds of remaining coastal wetlands, including freshwater forested wetlands) over the next 50 years (Couvillion *et al.* 2017; CPRA 2023).

Rapid land loss in Louisiana is attributed to RSLR, subsidence, and hurricanes (Yuill *et al.* 2009; Morton & Barras 2011). For instance, major Hurricanes Katrina and Rita struck the Louisiana coast in 2005, causing around 500 km^2 of land loss (Killebrew & Khalil 2018). In response to accelerated land loss and frequent storm activity, restoration efforts via beach nourishment and shoreline protection have been made to protect and stabilize these rapidly disappearing habitats, specifically barrier islands (Mendelssohn *et al.* 1991; Khalil *et al.* 2013; Killebrew & Khalil 2018). These methods aim to restore island area and elevation to withstand annual tropical storms and hurricanes, land loss due to erosion, subsidence and sea level rise (CPRA 2023). Stochastic climatic events such as tropical cyclones bring high winds and storm surges in addition to RSLR (Frederiksen *et al.* 2008; Croxall *et al.* 2012). In regions where storms are frequent, such as the Gulf, coastal landscapes are impacted and changes in morphology and vegetation may occur (Doyle *et al.* 1995; Dietz *et al.* 2018; Velasquez-Montoya 2021). Storms can result in rapid beach erosion, overwash and barrier island breaching which can lead to repeated restoration of salt marshes and barrier islands (Morton & Sallenger 2003; Morton 2010).

Barrier islands provide critical breeding habitat for a number of bird species in addition to protecting the coastline, thus barrier island management is a central component of seabird conservation (Erwin *et al.* 2001; Behrouzi-Rad 2014; Craik *et al.* 2015; Remsen *et al.* 2019; Valle *et al.* 2023). Seabirds species are some of the most threatened of all bird groups with nearly half of seabird species facing decreasing populations (Croxall *et al.* 2012; IUCN 2022). Invasive alien species, bycatch and climate change are the top three threats affecting seabirds globally (Dias *et al.* 2019). Long-lived seabirds can show lower survival rates in times of high precipitation and strong winds (Frederiksen *et al.* 2008). Because colonial nesting seabirds in Louisiana congregate in large numbers on remote coastal and barrier islands, they are extremely vulnerable to habitat loss (Spendelow & Patton 1988; Martin & Lester 1990; Remsen *et al.* 2019). The management of coastal and barrier islands is a key focus of waterbird conservation in Louisiana (Leberg *et al.* 1995;

Raynor *et al.* 2013; Selman *et al.* 2016; Byerly *et al.* 2020; LeBlanc *et al.* 2023). These crucial nesting habitats face elevation loss due to subsidence and erosion, sea level rise, increased rates of storm activity, and consequential vegetation degradation (Visser *et al.* 2005; Sallenger *et al.* 2009; Walter *et al.* 2013; Remsen *et al.* 2019).

Natural ecological perturbations like extreme weather or hurricanes can affect bird populations by altering their habitats and consequentially impacting their breeding success and demographics (Leberg *et al.* 2007; Bonter *et al.* 2014; Descamps *et al.* 2015; Selman *et al.* 2016). The increased frequency of weather events such as heat waves, snowstorms, or rainstorms can impact terrestrial and waterbirds at all stages of life cycles (Şekercioĝlu *et al.* 2012; Descamps *et al.* 2015). Prolonged exposure to extreme heat can cause hyperthermia, dehydration, starvation, susceptibility to pathogens and influence body size (Delfino 2024). For example, the frequency of extremely hot days had a negative effect on the fledgling body mass and tarsus length of house sparrows in central Europe (Pipoly *et al.* 2013). Increases in annual precipitation can cause decreases in species richness, such is the case for forest birds in southwestern South America (Quilodrán *et al.* 2022). Additionally, extreme precipitation can be an important driver of avian breeding success. For instance, during the breeding season, snowstorms cause chick mortality in Antarctic petrels, negatively impacting nest survival and overall productivity (Descamps *et al.* 2015). Heavy and persistent rainfall can lead to not only chick mortality, but the loss of entire broods of Great Tits (*Parus major*) in Austria from hypothermia, starvation, and predation (Schöll & Hille 2020).

Hurricanes impact bird populations through direct mortality (Lavers *et al.* 2024), displacement (Tossas 2006; Dionne *et al.* 2008) and loss of nests (Debata 2019). Lavers *et al.* (2024) documented the direct mortality of 80-90% of five species of seabirds nesting on Bedout Island following Cyclone Ilsa in 2023. The cyclone hit when boobies were incubating, and frigatebirds were in the early stages of egg laying causing severe mortality of adults, chicks, and eggs due to sediment erosion which completely buried nests (Lavers *et al.* 2024). Hurricane Georges hit Puerto Rico in 1998 causing canopy loss and displaced tropical forest birds forcing higher elevation species into lower forest strata (Tossas 2006). Hurricane Wilma in 2005 caused a fallout of Chimeny Swifts (*Chaetura pelagica*) along the North Atlantic coast and as far east as western Europe. This storm not only caused displacement of birds into colder regions and chimneys, but direct mortality to a portion of the breeding population leading to lower numbers of breeding birds the following year (Dionne *et al.* 2008)

Indirect impacts in the aftermath of hurricanes include loss of foraging substrates, food availability (Askins & Ewert 1991; Dobbs *et al.* 2009), and overall habitat suitability (Torres & Leberg 1996; Wiley & Wunderle 1993; Şekercioĝlu *et al.* 2012). Hurricanes can also indirectly impact populations of breeding birds. For example, high-intensity winter cyclones in the North Atlantic impact breeding seabirds by reducing prey availability resulting in starvation (Clairbaux *et al.* 2021). However, exposure to cyclones is not uniform at all life history stages. For instance, in the Indian Ocean, Trindade/Round Island Petrels (*Pterodroma arminjoniana*) are exposed to cyclones at their breeding colony, during migration, and at sea (Nicoll *et al.* 2016). Adult petrels were mostly unaffected, while the survival of juveniles during their first year at sea was reduced from exposure to cyclones (Nicoll *et al.* 2016). Climatic conditions and hurricanes can have short-term and long-term impacts (Wiley & Wunderle 1993; Raynor *et al.* 2013; Dietz *et al.* 2018). Extreme meteorological events can also lead to changes in habitat with some species more resistant to disturbance than others (Brown *et al.* 2011; Debata 2019). For instance, Category 5 (Saffir-Simpson index) Hurricane Katrina hit Louisiana in 2005 and opened the canopy of a coastal forest. The bird community exhibited varied responses; canopy breeders showed no change, while species favoring understory vegetation increased in density (Brown *et al.* 2011).

Seabirds rely on coastal environments during the breeding season and the majority of seabird species are colonially nesting show high natal site fidelity (Croxall *et al.* 2012; Şekercioĝlu *et al.* 2012). Colonial breeding can benefit seabirds through information sharing and predator defense (Rolland *et al.* 1998; Erwin *et al.* 2001). Coloniality can also expose populations to a particular catastrophic ecological disturbance (Lavers *et al.* 2024) or repetitive disturbances which can impact reproductive success (Darrah 2024).

Many coastal waterbird species are of conservation concern in North America due to habitat loss and fragmentation (Walter *et al.* 2013; Langham *et al.* 2015; Pickens *et al.* 2017; Ringelman *et al.* 2021; Santos-Tovar *et al.* 2024). Habitat loss due to coastal erosion induced by hurricanes and RSLR have been shown to cause sharp declines in nesting numbers of Brown Pelicans in Louisiana the year following a major storm (Selman *et al.* 2016). Rapid shoreline retreat caused the conversion of vegetated areas to open water on Raccoon and Wine Islands, leading to the loss of pelican nests and the eventual loss of all vegetated areas on Wine by 2010, and subsequent loss of Wine Island (Walter *et al.* 2013).

The impact of natural stochastic climatic events and habitat restoration efforts made by the state and federal governments on the reproductive success of Brown Pelicans is not fully understood and my research aims to fill in some of the current data gaps.

In addition to the threat of rapid land loss and habitat damage from hurricanes, seabirds face predation pressure (Croxall *et al.* 2012; Borrelle *et al.* 2018; Byerly *et al.* 2021). Hurricanes can have varying impacts on mammal populations. For instance, island-dwelling mammalian species can suffer population declines as direct impacts of severe cyclonic storms (Pierson *et al.* 1996). Hurricanes can cause changes to island habitat structure. For instance, Pries *et al.* (2009) found reductions in dune height and vegetation caused declines in beach mice (*Peromyscus polionotus*) on (Pries *et al.* 2009). Additionally, hurricanes and flooding can facilitate biotic invasions (Diez *et al.* 2012). Invasive mammals such as rats (*Rattus sp.*) can establish on islands following hurricanes threatening native island species in the Caribbean (Shiels *et al.* 2020). Invasive ants have the potential to disrupt ecosystems and can favor open and disturbed habitats (e.g. hurricane damage) (Holway *et al.* 2002). The invasive red imported fire ants (RIFA) (*Solenopsis invicta* Buren) are widely distributed geographically dispersing by wind and ability to raft and float (Wylie *et al.* 2024). In response to hurricanes and flooding events, colonies of ants have the ability to raft in fresh, brackish, and saltwater (Hooper-Bùi *et al.* 2020; Wylie *et al.* 2024). RIFA are known predators of ground-nesting seabirds in the northern Gulf (Drees 1994; Lockley 1995). In populations from Louisiana, this ant species can be more aggressive in tidally-influenced areas than inland areas (Hooper-Bùi *et al.* 2020). One study hypothesized overwash from hurricanes depresses invasive RIFA in coastal Louisiana, but no comparative assessments have quantified the net impacts on these populations immediately after storms (Walter *et al.* 2013).

Hurricanes bring rain, high winds, wave action, and a rapid rise in sea level leading to storm surge and island breaching (Pielke & Pielke 1997; Fritz *et al.* 2007). In the Northern Hemisphere when hurricanes make landfall, onshore winds are strongest in the front right quadrant of a hurricane’s path and where ocean bathymetry focuses wave energy (e.g., narrow embayments) (Pielke & Pielke 1997; Keim *et al.* 2007). Storm conditions extend farther to the right than the left of the eye and Category 3-5 hurricanes can have a swath up to 240 km wide with tropical storm conditions an additional 80 km to the right and 40 km to the left (Keim *et al.* 2007). Storm surge diminishes as a hurricane moves inland meaning the most destructive effects are on beaches and offshore islands (Pielke & Pielke 1997). Hurricane Ida made landfall on August 29, 2021, at Port Fourchon, Louisiana as a Category 4 hurricane with maximum sustained winds of 241 km/h (Category 5 begins with >252 km/h) (NOAA). In the Northern Gulf, the path of Hurricane Ida traveled directly through Barataria and Terrebonne Basins and continued north. In order to investigate the possible impacts of Hurricane Ida on wildlife on barrier islands, I compare island characteristics on both temporal (pre-and post-hurricane) and spatial (west-east orientation) scales.

This study focuses on the Eastern Brown Pelican (*Pelecanus occidentalis carolinensis)* populations breeding on coastal islands in southeast Louisiana. This species has been identified as a high-priority species for monitoring and nesting habitat restoration due to its conservation history in the United States (Jodice *et al.* 2019). The population in Louisiana experienced a major decline in abundance in the 1950s and 1960s (Nesbitt *et al.* 1978) during the height of DDT (dichlorodiphenyltrichloroethane) pesticide use in North America. Population declines are generally considered to be associated with the use of DDT and pesticides which caused the thinning of eggshells and subsequent low nest success rates (King *et al.* 1977; Blus 1982). These declines eventually led to the extirpation of breeding populations in Louisiana by 1963, causing the Northern Gulf population to be considered endangered by the federal government (United States Fish and

Wildlife Service [USFWS] 1970; King *et al.* 1977). After DDT was banned in 1972, Brown Pelican populations were able to recover and stabilize across the Northern Gulf (Holm *et al.* 2003). To reestablish pelicans in Louisiana, translocations of juvenile pelicans from Florida to Louisiana coastal islands, such as Queen Bess Island, helped re-establish breeding populations in Louisiana (Nesbitt *et al.* 1978; McNease *et al.* 1984). In 2009, the Brown Pelican was removed from the Federal List of Endangered and Threatened Wildlife (USFWS 2009). In 2024, this species is listed globally as ‘apparently secure’ according to NatureServe but is listed as ‘vulnerable’ in several states, including Louisiana (LDWF 2024; NatureServe 2024). The primary goal of my research is to better inform the understanding of important criteria for successful island restoration efforts regarding seabird reproductive success, seabird predator population dynamics, and vegetation communities in this complex environment. My thesis focuses on three Brown Pelican breeding colonies in southeast Louisiana including Raccoon, Philo Brice, and Queen Bess Islands (Figs. 1 & 2). I analyzed how Brown Pelican reproductive success varies before and after Hurricane Ida on three islands using distance and orientation of island to the storm path as predictors. Queen Bess and Raccoon Islands have undergone restoration in the form of dredge material placement and shoreline protection using breakwaters, followed by vegetative plantings. Philo Brice has never undergone restoration.

Hurricanes disturb coastal ecosystems and may alter habitat suitability and breeding success for wildlife (Wang *et al.* 2016). I hypothesized stochastic weather events, such as Hurricane Ida, drive trends of reproductive success, with islands to the east of the storm path experiencing reduced reproductive success and islands to the west showing no change in success. I predicted breeding colonies in and to the east of Hurricane Ida’s path experienced higher reduction in vegetation cover, lower average height, and less plant species diversity in comparison to islands to the west of the storm path. In addition, I predicted post-storm effects on reductions of vegetation density will provide a more suitable nesting habitat for ground-nesting pelicans compared to shrub-nesting pelicans. Islands with vegetation reductions may provide less suitable habitat for successful pelican nests, and therefore indirectly relate to changes in reproductive success in the years following the storm (Walter *et al.* 2013). I also predicted coastal islands in and to the east of Hurricane Ida’s path will experience higher rates of erosion and reduced island size.

Chapter: Eastern Brown Pelican (*Pelecanus occidentalis*) Reproductive Success and Vegetation Cover Following Hurricane Ida

Introduction

Weather extremes such as heat waves and tropical cyclones are important components of global climate change (Şekercioĝlu *et al.* 2012; Stott 2016). Tropical cyclones, hereafter called hurricanes, are disruptive meteorological events in coastal and marine zones that cover large spatial areas and can alter the geomorphology of coasts (Stone *et al.* 2004), vegetation (Doyle *et al.* 1995; Uriarte *et al.* 2019), and directly or indirectly cause mortality of wildlife populations (Nicoll *et al.* 2016; Huang *et al.* 2017; Velasquez-Montoya *et al.* 2021). As hurricanes increase in frequency and intensity (Lima *et al.* 2021; Bhatia *et al.* 2022) there is an urgent need to study the ecological consequences on avian populations and habitats (Descamps *et al.* 2015; Maxwell *et al.* 2019).

Seabirds, particularly colonial nesting seabirds, face a variety of threats including loss or alteration of habitat, human disturbance, extreme weather events, and increasing predator populations (Burger 1984; Visser & Peterson 1994; Goodrich & Buskirk 1995; Erwin *et al.* 2006). Weather events including hurricanes and extreme heat or cold can impact bird populations throughout the annual cycle and influence movement and population dynamics (Wilkinson *et al.* 2019). For instance, snowstorms negatively impacted both breeding success and population dynamics of Antarctic Petrels (*Thalassoica antarctica*) (Descamps *et al.* 2014). Direct impacts from heat waves and storms caused significant chick mortality in breeding Bank

Cormorants (*Phalacrocorax neglectus*) on created islands in southern Africa (Sherley *et al.* 2012). Storms can induce shifts in optimal nesting sites that disproportionately impact reproductive success in waterbirds (Bonter *et al.* 2014; Shepherd *et al.* 1991). For example, in 1989, coastal destruction and saltwater intrusion into freshwater marshes due to Category 4 Hurricane Hugo resulted in variable reproductive responses in a mixed-species colony of Great Egrets (*Ardea alba*), Snowy Egrets (*Egretta thula*), Tricolored Herons (*Egretta tricolor*), Glossy Ibis (*Plegadis falcinellus*), and White Ibis (*Eudocimus albus*) (Shepherd *et al.* 1991). Hugo made landfall 80 km from Pumpkinseed Island, South Carolina, causing declines the following year in nesting Great Egrets, Tricolored Herons, and White Ibises, but not Snowy Egrets and Glossy Ibises. White Ibises plummeted from over 10,000 pairs to 0 in just one year. These declines were due to changes in the availability of optimal nest substrate (shrubs or grassy vegetation) and prey availability (Shepherd *et al.* 1991). Storms can also induce shifts in optimal nesting sites with nests at lower elevations and shorter distances to water at an increased risk of mortality (Bonter *et al.* 2014; Darrah 2024).

Increased tropical cyclone activity in the Atlantic Ocean, and particularly the northern Gulf (Lima *et al.* 2021), chronic sea-level rise (Kirwan & Megonigal 2013), and the compounding effects of coastal retreat due to land subsidence (Yuill *et al.* 2009) and erosion (Mentaschi *et al.* 2018) are major threats to the islands that support colonially nesting seabirds (Remsen *et al.* 2019; Brown *et al.* 2024). In addition, seabirds are specifically impacted by an increase in storm activity (Lima *et al.* 2021) and sea level rise due to their high fidelity to natal colonies and location of nesting sites on remote low-lying islands (Spendelow *et al.* 2002; Hatfield *et al.* 2012). Hurricanes can impact colonial nesting birds directly during the breeding season by causing adult and chick mortality from high winds, displacement, or nest failure (Debata 2019; Vasseur *et al.* 2023). For instance, from 2017-2021 three tropical cyclones made landfall during the peak breeding season of Least Terns (*Sternula antillarum*) on Mississippi's Gulf coast (Darrah 2024). Each storm surge caused complete overwash of all colonies leading to near complete loss of productivity despite renesting attempts (Darrah 2024). Hurricanes can also have indirect impacts on reproductive success, such as damage to optimal nesting vegetation from high wind and flooding (Shepherd *et al.* 1991), or subsequent erosion of islands leading to a decrease in the availability of nesting sites compared to years prior to the event (Marsh & Wilkinson 1991).

Barrier islands form a protective barrier between continental shorelines and offshore wave energy (Feagin *et al.* 2010). Additionally, barrier islands are extremely dynamic systems and changes in geomorphology and hydrology can take place over a short period of time, i.e., hours or days, in response to tropical cyclones and hurricanes (Feagin *et al.* 2010). Sea level rise and storms can cause the landward migration of barrier islands, decreases in width and height (Khalil *et al.* 2013), increases in frequency of overwash (Courtemanche *et al.* 1999) and breaching (Morton 2010). Hurricane Katrina was a Category 5 (Saffir-Simpson index) storm that passed over Louisiana's coast as a Category 3 storm in 2005 followed by Hurricane Rita one month later, causing massive destruction to coastal Northern Gulf states (Fritz *et al.* 2007; Steyer *et al.* 2010; Morton & Barras 2011). Three years later in 2008, two intense storms, Hurricane Gustav Category 3 and Hurricane Ike Category 2 made landfall causing vegetation scouring and erosion on Louisiana barrier islands (Morton & Barras 2011). These consecutive storms caused barrier island breaching, beach erosion and deposition of washover terraces (Morton & Barras 2011). Subsequent increases in open water coverage and decreases in land cover over much of Barataria Basin in southeastern Louisiana are attributed to the impacts of these repetitive hurricanes (Morton & Barras 2011; Yao *et al.* 2022; Potter & Sukanna 2024). Although some Louisiana barrier island vegetation communities can recover from storm impacts (Courtemanche *et al.* 1999) others, such as Wine Island, have been completely lost to erosion and RSLR (ACRE 2019).

Coastal Louisiana experiences frequent tropical storms and hurricanes (Morton & Barras 2011). From 2013-2023, 10 tropical cyclones (including hurricanes and tropical storms) occurred within 200 km of Barataria-Terrebonne Basins (Suppl. Table 1, Figure 2) (NOAA 2024). Hurricane Ida, a Category 4 with maximum sustained winds reaching 241 km/h, passed through the Barataria-Terrebonne region on August 29, 2021. Just 10 months prior, Hurricane Zeta Category 3 made landfall in the same basin (Yao *et al.* 2022; NOAA 2024). These consecutive storms caused shoreline retreat and up to 40% loss of beaches in nearby restored areas such as Bay Champagne (Yao *et al.* 2022). The rapid land loss in coastal Louisiana has led to an integrated approach combining hurricane protection, conservation, and the restoration of coastal areas through large-scale ecological restoration projects (Khalil *et al.* 2011).

Artificial wetlands cannot replace naturally occurring habitats, but can provide alternative habitat for wildlife during multiple life stages (Ma *et al.* 2010). Indirect methods have been shown to provide habitat for nesting birds such as dredge islands created from sediment removed from shipping canals (Leberg *et al.* 1995). The creation and enhancement of marshes and beaches within coastal landscapes are implemented in order to restore ecosystem functions (Ma *et al.* 2010; Armitage *et al.* 2014). One trajectory of ecosystem restoration is to create suitable wildlife habitat that is resilient to periodic stresses, such as hurricanes (SER 2004; Armitage *et al.* 2014).

Louisiana barrier islands have undergone area loss due to complex interactions of subsidence, erosion, RSLR, inadequate sediment supply causing fragmentation (Khalil *et al.* 2013) and in some cases island submergence (e.g. Wine Island) (ACRE 2019; Enwright *et al.* 2020). To mitigate coastal land loss in Louisiana, barrier and coastal island restoration aims to reinforce these island ecosystems that serve as the first line of defense against storms (Khalil *et al.* 2011; Khalil *et al.* 2013). Management practices have been implemented, including adding breakwaters around island for shoreline protection (Khalil *et al.* 2011; CPRA 2019), placing dredged sediment from nearby sources into dunes and back bay marshes (Lindquist 2007; Khalil *et al.* 2011) and planting native vegetation (Mendelssohn *et al.* 1991;

Sylvest *et al.* 2022). These efforts aim to manage habitat characteristics including water depth and fluctuation, island elevation, island area, vegetation cover, therefore providing storm protection and suitable seabird nesting habitat (Visser *et al.* 2005; Lindquist 2007; Lopez 2009).

Many restoration projects have occurred to stabilize coastal and barrier islands in Louisiana and almost all seabird breeding colonies in the region depend on restoration (Visser & Peterson 1994; Khalil *et al.* 2013; DWH LA TIG 2023). To mitigate seabird habitat loss, direct management efforts can create new artificial islands (Leyva-García *et al.* 2023), expand existing breeding colonies, restore historical populations, and protect colonies from ongoing and future threats such as RSLR and hurricanes (DWH LA TIG 2023). Dredge material can be used beneficially to fill eroded islands and increase elevation for seabird nesting sites (Leberg *et al.* 1995; Lindquist 2007; DWH LA TIG 2023). For the restoration of islands in Louisiana, the desired elevation has been determined by balancing island inundation exceedance probability for vegetation and nest survival with the cost of construction (DWH LA TIG 2023). Island inundation exceedance probability is the likelihood that the nesting areas/elevations will be flooded during the nesting season (DWH LA TIG 2023). To restore sediment-starved islands, vegetation plantings and sand fencing can be used to stimulate sand accumulation (Mendelssohn *et al.* 1991). To mimic natural coastal ecosystems and jumpstart land gain, native vegetation plantings of salt marsh and/or mangrove foundation species (e.g. salt marsh grass (*Spartina alterniflora*)), black mangrove (*Avicennia germinans*) are conducted to benefit breeding seabirds, stabilize shorelines against wave action, and minimize erosion (Yando *et al.* 2019; CPRA 2023).

Hurricanes can alter the condition of available habitat for breeding seabirds which can indirectly impact reproductive success (Walter *et al.* 2013; Darrah 2024). The amount of available nesting habitat is a limiting factor for Brown Pelican populations in Louisiana (Holm *et al.* 2003; Selman *et al.* 2016). Of the ten documented attempts to create colonial waterbird breeding islands (primarily for Brown Pelicans), only 4 have persisted (DWH LA TIG 2023). The most successful restoration efforts have occurred at historical Brown Pelican colonies including Raccoon, Rabbit, and Queen Bess Islands (Visser *et al.* 2005; DWH LA TIG 2023). A study conducted in 2013 on Louisiana barrier islands found Brown Pelican reproductive success trends before and after hurricanes can be impacted by the condition of available breeding habitat (Walter *et al.* 2013). Brown Pelicans exhibit non-random nest site selection and their probability of nesting in shrubs or on the ground is site dependent (Robinson & Dindo 2011). In Louisiana, Brown Pelicans exhibit a strong preference for nesting platforms in shrubs, e.g. black mangrove, marsh elder (*Iva frutescens*), or Christmasberry (also commonly known as matrimony vine, *Lycium carolinianum*), that are higher and sturdier than ground nests on bare sand or grass (e.g. salt grass or smooth cordgrass) (Blus *et al.* 1979; Walter *et al.* 2013).

Pre-storm investigations on coastal islands in southern Louisiana provided an opportunity to assess how pelican colony success responded to the effects of

Hurricane Ida. Brown Pelicans have received specific focus as it is the Louisiana state bird, and the rebound of this species in the United States is considered a conservation success. I used Brown Pelicans to understand the potential indirect effects of a large weather event on habitat suitability for colonial nesting seabird populations in southeast Louisiana.

Hurricane Ida passed over the three islands, Raccoon, Philo Brice and Queen Bess (Figure 2) that had been the subject of nest success investigations prior to the storm. Two of these three sites experienced extensive restoration. I analyzed pre-storm and post-storm nest observation data to quantify the effect of the storm on the breeding ecology of Brown Pelicans. These relationships provided insight into the suitability of restored and unrestored sites that have been impacted by a major hurricane for successful Brown Pelican breeding colonies. I hypothesized that the storm damage would reduce reproductive success.

Category 3-5 hurricanes can have a swath up to 240 km wide with tropical storm conditions an additional 80 km to the right and 40 km to the left (Pielke & Pielke 1997; Keim *et al.* 2007). Therefore, I predicted that these reductions would be greater for islands in or to the east of the storm path relative to the island west of the storm track. This prediction is based on the tendency for storm conditions to extend farther to the right (east of Ida) than the left (west of Ida) of the eye (Pielke & Pielke 1997; Beven *et al.* 2021).

Islands with reduced shrub vegetation may provide less suitable habitat for successful pelican nests, and therefore pelicans nesting in those sites might experience reduced reproductive success in the years following the storm. Therefore, I predicted breeding colonies in and to the east of Hurricane Ida's path experienced higher reduction in vegetation cover, lower average height, and less plant species diversity in comparison to islands to the west of the storm path. In addition, I predicted post-storm effects on reductions of vegetation density result in a shift from shrub-nesting to ground-nesting pelicans.

Methods

Study Area

I examined Brown Pelican nesting behavior on 2 restored islands and 1 unrestored island in southeastern Louisiana located in Barataria and Terrebonne basins (Figure 1). The Isles Dernieres barrier system is composed of multiple islets in Terrebonne Basin. The easternmost island in this chain, Raccoon Island (29.05276, -90.93368), is situated 53.1 km to the west of Hurricane Ida's path (Figure 2). Raccoon Island hosts a large breeding colony with an area of approximately 0.6 km^2 that is part of the Isle Dernieres Barrier Islands Refuge and managed by the Louisiana Department of Wildlife and Fisheries (LDWF). In 1997, breakwaters were positioned along the shoreline for protection and were able to increase sediment behind the breakwaters (Curole & Babin 2020). This island underwent beach nourishment and was filled with sediment that was dredged from an offshore borrow area in 2007, 2013, and 2018 in order to increase island size and elevation (Lindquist 2007).

Containment dikes were constructed to enclose open water on the bayside of Raccoon Island (Curole & Babin 2020). In both 2003 and 2008 the sand spit on the western end of the island detached from the island due to the passage of Hurricanes Gustav and Ike (Curole & Babin 2020). After the hurricane season in 2008, extensive overwash and erosion caused the spit was relocated further into the bay (Curole & Babin 2020). Native plantings of herbaceous and woody plant species were conducted in 2020 on dikes and marsh platform to restore Gulf coast barrier island communities (CPRA 2023). This island contains *Spartina*-dominated saline marsh on the north side and short-stature black mangrove stands with a lagoon in the center of the island.

Philo Brice Island (29.18597, -90.34215), which was previously referred to as Philo Brice Islands, in Terrebonne Basin, has undergone substantial land loss since 2010 (Couvillion *et al.* 2017; Capitolo *et al.* 2023). Philo Brice Island is a low elevation inundated island comprised almost exclusively of black mangrove stands. The remainder of the island includes two shell hash areas, and a flooded saltmarsh mangrove-dominated habitat, with no dune habitat. The eye of Hurricane Ida passed 1.9 km to the west of Philo Brice Island in 2021 (Figure 3). This island was not an active colony from 2010-2015, but since 2018 has hosted a large Brown Pelican colony with around 5,700 nests counted in 2021 (Capitolo *et al.* 2023), and an area of around 0.13 km^2 in 2023 (Figure 3). Philo Brice Island is owned by a private entity, Conoco Phillips, and has never undergone restoration.

Queen Bess Island Wildlife Refuge (29.30606, -89.95770) is in Barataria Basin 38.4 km to the east of Hurricane Ida is a breeding colony managed by LDWF (Figure 3). Projects to protect Queen Bess in 1992 and 1996 installed a rock ring around the island that had become partially open water (CPRA 2021). A recent restoration in 2020 increased island size from 2 ha to 14.6 ha and increased elevation for more suitable breeding habitat for colonially nesting waterbirds and seabirds such as Brown Pelicans, Royal Terns (*Thalasseus maximus*), Sandwich Terns (*Thalasseus sandvicensis*), and Laughing Gulls (*Leucophaeus atricilla*) (CPRA 2023). The project included installation of rock dikes, breakwaters, placement of sand fill, limestone, bird ramps, placement of hay bales, plantings of marsh elder, groundsel bush/eastern baccharis (*Baccharis halimifolia*), Christmasberry, and post-construction planting of black mangrove and native shrub species (CPRA 2021). This island has similar habitat structure to Raccoon Island with less unvegetated dune. Given its proximity to the only inhabited barrier island (3.7 km to Grand Isle), Queen Bess encounters the highest level of human interaction or disturbance (pers. observation), including boating, angling, construction, and biological sampling.

Nest Success Surveys

Field crews and I placed motion-detecting nest cameras (Moultrie M-999i and Spypoint Force-20) on all three colonies during the Brown Pelican breeding season from March to August in 2021, 2022, and 2023. Cameras were placed in different locations each year and the number of cameras on each island determined by the relative size and accessibility of each island. In 2021, 26 total cameras recorded pelican nesting activity on Raccoon (n = 10), Queen Bess (n = 11), and Philo Brice Island (n = 5). In 2022, 32 total cameras monitored Raccoon Island (n = 15), Queen Bess Island (n = 12), and Philo Brice Island (n = 5). In 2023, 51 total cameras were placed on nest sites on Raccoon Island (n = 20), Queen Bess Island (n = 16), and Philo Brice Island (n =15). All camera locations were recorded on a Garmin handheld GPS unit. To reduce stress on the birds, we restricted colony visits to the cooler time of day, ending activity on the colony at 11:00 AM. Camera sites aimed at both shrub-nesting and ground-nesting pelicans, primarily on the exterior of colonies to reduce human disturbance to nesting birds. Philo Brice Island is composed of primarily inundated thick black mangrove stands and therefore does not support ground-nesting pelicans. We visited islands once every 2-4 weeks to replace batteries and SD memory cards.

From 2021-2023 we collected over 4.8 million photographs of Brown Pelican nesting behavior, and I assessed the daily survival of 363 nests. The fate of 306 nests was determined as either successful or failed. The remaining 57 nests were not monitored for the full duration of their nesting period due to camera malfunction or battery failure between site visits. Camera locations were different each year with an average of 121 nests monitored yearly. Nests were defined as active if they had at least 1 egg or chick visible in the nest and an incubating adult present.
Images of active nests were checked every 48 hours beginning at the time of first light (approximately 30 minutes before sunrise). If no change in nest status was

detected, observations continued every 48 hours until all chicks in the nest reached a fledgling age of at least 21 days or perished (Walter *et al.* 2013).

Chicks were considered fledged if observed from hatch date to 21 days old (Sachs & Jodice 2009; Walter *et al.* 2013). If the exact hatch date was unknown, fledging status was assessed based on relative size, behaviors and plumage traits typical of 3 to 4.5-week-old chicks (Schreiber 1976; Sachs & Jodice 2009). This developmental stage is the threshold for when Brown Pelican chicks begin sitting or standing upright as opposed to lying prone, begin naturally abandoning nests, are no longer associated with a particular nest site, are not closely brooded by one or both parents (Schreiber 1976; Walter *et al.* 2013), and have developed thermoregulatory abilities (Sachs & Jodice 2009). Plumage characteristics allowed for approximate aging, with chicks considered fledged when thick white down covered a bird, light tan down covered the head, and/or scapular feather stubs were visible through the down (Schreiber 1976). If a chick remained in the nest after death, I determined the last image where movement was detected and recorded this as a nest reduction event.

At the end of each observation date, I determined a binomial outcome of a successful nest day or a failed nest day. When change in chick or egg number occurred during a 48-hour interval, each photograph in that interval was reviewed to determine the time and possible cause of change. Nests reduction events (not necessarily failed nests) were considered to have occurred if the reduced egg and/or chick number remained the same in images for a period of 24 hours following the reduction event. If the number of eggs or chicks in imagery returned to the number in the nest prior to the putative reduction, I assumed that the reduction had not occurred. Daily nest stages were recorded for each observation including: ‘empty’ (stage 0), ‘at least 1 egg present’ (stage 1), ‘at least 1 chick present’ (stage 2), and ‘at least 1 chick fledged’ (stage 3).

Nests were categorized successful if at least 1 chick reached the age of 21 days or were considered fledged based on behavior and plumage traits (Schreiber 1976). Nests were considered abandoned and given a fate of failed if: adults left nest unattended and never returned to incubate; adults deconstructed nests or covered eggs/dead chicks with straw; adults were present at nest but no visible evidence of eggs, chicks, or feeding for 14 days; or if previously active nests were empty with no signs of nesting activity during field visits.

Nest failure cause was classified as one of six categories: flooding/storm event, RIFA predation, human disturbance, intraspecific competition, mammalian disturbance, or unknown. Failure was attributed to flooding/storms if abandonment occurred after one or more flooding events or rainstorms lasting overnight were documented on images. RIFA predation was confirmed when chicks were observed alive and covered in crawling insects, bites, and perished within 24 hours followed by the parent never returning. Failure was attributed to human disturbance when nests were abandoned within 12 hours of humans visiting the site or of nocturnal banding activities that caused adults to flush. Intraspecific competition was considered to cause failure when nearby adults and/or chicks attacked the monitored adults and/or chicks causing them to perish or to be displaced out of the nest followed by abandonment. Nest failure was attributed to mammalian disturbance if mammals were observed in or near the nest when adults left the nest unattended, followed by nest abandonment. In addition to behavioral observations, the substrate under each nest was designated as: grasses, forbs, marsh elder, black mangrove, or dead woody vegetation.

Vegetation Surveys

A crew of trained technicians, other graduate students, and I surveyed herbaceous and woody vegetation in August 2018, 2019, 2022, and 2023 on Raccoon Island and in 2022 on Queen Bess (restoration construction occurred in 2020 and 2021 preventing access for these surveys). Philo Brice Island was not included in these surveys due to the impenetrable terrain composed almost exclusively of black mangrove. Survey plots within three microhabitat types were sampled at Raccoon Island (restored dune, restored marsh, and unrestored). Each habitat type contained 2 linear transects of 3 survey points spaced 200 m apart on Raccoon Island in 2022 with a total of 18 points surveyed. Plots on Queen Bess were placed 75 m apart due to the smaller total area of the island. We surveyed 2 transects each with 3 points in 2022 for a total of 6 transect points. Most of the landscape on Queen Bess was extensively restored in 2020, apart from the northeastern portion of the island, which was treated as ‘unrestored’. We surveyed woody vegetation in each 10 x 10 m plot and visually estimated percentage vegetation cover (PVC) of each plant species/genera, water and bare ground. Maximum vegetation height and average vegetation height were also collected at each point.

Remote Sensing Using Satellite and Aerial Imagery

Major storm paths in southeast Louisiana over the last decade were visualized using hurricane and tropical storm vector files obtained from the NOAA Historical Hurricane Tracks portal (https://coast.noaa.gov/hurricanes). In ArcGIS Pro, a polygon encompassing all study sites was created, and tropical cyclone tracks (including tropical storms and hurricanes) from 2013 to 2023.

I employed remote sensing techniques to analyze change in island area over time, calculate Normalized Difference Vegetation Index (NDVI), and classify land cover types. A review of 13 different imagery sources was conducted to identify datasets available for the required time periods and locations. I originally targeted imagery before and after Hurricane Ida (August 29, 2021) during the breeding season (Feb-August) during the study years (2021-2023), but due to limited availability and coverage, target periods were expanded to all months from 2018-2023. However, none were completely suitable due to excessive cloud cover, coarse resolution, lack of data for the timeframe, or restricted access (Suppl. Table 2). As a result, a combination of aerial imagery and satellite imagery was selected to meet the goal of comparing pre- and post-Hurricane Ida land cover. All imagery analyzed for the year 2021 was taken before landfall of Hurricane Ida (August 29, 2021) with the single exception of Google Earth Pro imagery used to calculate the total area of Philo Brice Island which was acquired by the satellite on September 10, 2021.

Island areas were calculated to detect change before and after Hurricane Ida on Raccoon, Philo Brice, and Queen Bess Islands in available years from 2017 to 2023. A combination of imagery from Google Earth Pro (1-30 m) and Esri World Imagery Wayback (< 1 m) was analyzed to supplement years not covered by NAIP or Sentinel-2 imagery (Suppl. Table 2 and 3). To quantify changes in island shorelines and area I manually digitized polygons around each island in ArcGIS Pro for available fine resolution images (0.3 – 10 m), and in Google Earth Pro (1 – 30 m) for images not freely accessible for processing in ArcGIS Pro.

I downloaded high resolution (0.3 - 0.6 m) multispectral imagery from the National Agriculture Imagery Program (NAIP) on the USGS EarthExplorer Portal targeting island study sites Raccoon, Philo Brice and Queen Bess Islands. NAIP imagery has a spatial resolution of 0.3m or 0.6m with 3-band RGB color and infrared (NIR) imagery which is collected every 2-3 years in the summer in the United States. Since recent pre-hurricane imagery of Philo Brice and Queen Bess Islands was not available through NAIP, I accessed satellite imagery from Copernicus Sentinel-2 data processed by Sentinel Hub with 4-band 10m resolution (Suppl. Table 3). All imagery was geoprocessed in ArcGIS Pro in the North American Datum 1983 (NAD 1983) geographic coordinate system and NAD 1983 UTM Zone 15N or 16N projected coordinate system. To compare multispectral imagery of varying resolutions on each island, I resampled Raccoon Island imagery to 0.6 m resolution, Philo Brice Island imagery to 0.6 m resolution, and Queen Bess Island imagery to 1 m resolution using bilinear resampling. Band composites were created for red and infrared bands and rasters were analyzed for Normalized Difference Vegetation Index (NDVI), which is derived from the red: near-infrared reflectance ratio [NDVI=(NIR-RED)/ (NIR+RED)]. NIR and RED are the amounts of near-infrared and red light reflected by the vegetation and captured by the sensor of the satellite ranging from -1 (no vegetation/water) to 1 (high vegetation) (Myneni *et al.* 1995; Pettorelli *et al.* 2005). Since NAIP imagery originally had higher spatial resolution (0.3 m or 0.6 m), finer-scale patterns were visible when compared to Sentinel-2 imagery (10 m).

Land cover types were classified based on NDVI values (-1 to 1) using a manual thresholding approach applied to both NAIP and Sentinel-2 imagery (Suppl. Table 4). The manually assigned thresholds were verified through visual inspection by overlaying classification results with high-resolution reference imagery and prior field knowledge of spatiotemporal vegetation distribution in the study areas. The land cover classification schemes were manually set (Suppl. Table 4) and all raster cells were reclassified to one of six habitats including: water, sand/bare, dead/sparse (dead vegetation and/or low-density of herbaceous plants), low vegetation (mixed herbaceous and low-density shrubs), medium (mixed herbaceous, shrubs, and mangrove stands), and high (high-density healthy vegetation and dense mangrove stands). The areas (ha) of each vegetation class within manually digitized island polygons were calculated from pixel count and resolution using the equation

[Area(ha) = ((Number of pixels) * (Resolution in meters)^2)/10,000] (Suppl. Tables 5-7).

Statistical Analysis

Reproductive success was evaluated on three islands over three years based on three measures: apparent nest success (ANS), daily survival rate (DSR), and fledging success (FS).

Apparent Nest Success. Apparent nest success is the proportion of surviving nests in a population, with surviving nests defined as nests with at least one chick reaching fledging age of greater than 21 days (Johnson 2007). I generated a generalized linear model with a binomial family, where the dependent variable of ANS was modeled as a binary outcome 1 (survived) or 0 (failed). Although this proportion is informative, it can be biased and may exclude nests with short survival periods or nests that are not monitored for the entire nesting period (Mayfield 1961; Mayfield 1975; Shaffer 2004). It does, however, provide a measure of overall success of a nest in producing at least one chick.

Daily Survival Rate. A more informative estimate of reproductive success is the daily survival rate (DSR) at each nest during the exposure period, which spans the duration the nest was actively monitored. Exposure period may cover the entire nesting cycle (incubation to fledging of youngest chick) or a subset of it, depending on when monitoring began and ended. This method reduces the biases associated with estimation of ANS.

I used a generalized linear mixed model with a custom logit link function that links variation in exposure period to the standard logistic regression model and constrains DSR to 0 or 1 (Shaffer 2004). I only included nests that had an exposure period ≥ 1 day. Because survival was determined for exposure periods, this analysis could use nests that were not monitored all the way to survival or fledging.

Fledging Success. Fledging success was defined as the number of chicks per nest that were monitored to fledging age (21 days post-hatch). I used a generalized linear model with a Poisson distribution to fit the model. The response variable is the mean number of fledglings per nest.

I examined the fixed effects of year, island, and nest location (on ground or in shrub) on ANS, DSR, and FS. Camera was a random variable in all analyses, controlling for possible non-independence in the fates of nearby nests. Because there will no pelican nests located on the ground on Philo Brice Island, two analyses were conducted, one limited to nests in shrubs on all three islands (Shrub Nests), and the other including both ground and shrub nests but limited to Raccoon and Queen Bess Islands (All Nests).

I modeled all three reproductive success measures (ANS, DSR, and FS) using a form of backward selection. Because I typically had insufficient data to model three-way interactions (island * year * nest location), I first fit a model with all main effects and two-way interactions. I then eliminated any two-way interaction with a P-value > 0.10 and refit the model. I would also remove two-way interactions that the GLM could not estimate, based on implausibly large estimates of SE. If after removing the two way interactions, there were main effects with a P > 0.10, I would remove those non-significant effects and refit the model. This approach eliminated nuisance variables that contributed little to model fit. Although I used a criterion of P > 0.10 to retain a term in a model, I only discuss interactions and main effects when P < 0.05. For visualization, DSR models were refit with a standard logit link instead of the custom exposure link to generate predicted DSR using ggpredict. All analyses were conducted in R, version 4.3.2 (R Core Team 2024).

Nest Failure. Contingency table analysis with a Fisher's exact Test was conducted to examine the independence of nest fate (successful/failed) and island. This analysis included 96 total nests on all three islands. I tested for possible association between nest stage at time of failure and island, including nests that failed in stage 1 (at least 1 chick present) and nests that failed in stage 2 (at least 1 chick present).

A Fisher's Exact Test was conducted to examine the independence of the reason for failure and island. The analysis included 53 failed nests classified by cause (e.g. flooding/storm, RIFA, human disturbance, intraspecific competition, and mammalian disturbance). Nests that had an unknown failure classification were removed from this analysis (n=43). In an attempt to understand the causes of nest failure during different parts of the nesting cycle on each island, I examined causes of nests that failed during stage 1 and during stage 2.

Pre- and Post-Storm Vegetation on Raccoon Island. A linear mixed-model and planned contrasts in a generalzedi linear model (GLM) was created to compare within island conditions on Raccoon Island pre-storm (2018-2019) and post-storm (2022-2023) percent vegetation coverage (PVC) on the island overall, bare ground cover, and vegetation height. Time period was treated as a fixed effect with two levels combining 2018 and 2019 data as 'pre-storm' and 2022 and 2023 data as 'post-storm'. A linear contrast was used to compare pre-storm and post-storm measures of vegetation structure and composition.

Using the same methodology, I examined temporal changes in vegetation composition within three habitat types on Raccoon Island (restored marsh, unrestored marsh, and restored dune) between pre-storm (2018-2019) and post-storm (2022-2023). Vegetation cover classes included black mangrove, marsh elder, total shrubs (including black mangrove, marsh elder, saltwort (*Batis maritima*), and coastal goldenbush (*Isocoma menziesii*)), forb species, grasses, and bare areas. A linear mixed-effects model was used to examine the effect of time (pre- vs. post-Hurricane Ida) on each cover class with transect site included as a random intercept to account for repeated sampling.

Raccoon and Queen Bess Islands 2022 Vegetation. Hurricane effects were predicted to be more severe on Queen Bess than on Raccoon Island. A generalized linear model with island as a fixed effect was created to compare overall PVC, bare ground cover, and vegetation height between Raccoon and Queen Bess Islands in 2022, the year following Hurricane Ida. Additionally, a linear mixed-effects model was used with island and habitat type as fixed effects to compare PVC of vegetation classes (black mangrove, marsh elder, total shrubs, forbs, and grasses) overall between islands, and within marsh habitat types between islands. The dune habitat was excluded from the analysis due to the lack of a dune habitat on Queen Bess. For all models the interaction of island and habitat type estimated marginal means were plotted for significant effects.

Remote Sensing Habitat Classification. A contingency table analysis with Pearson's Chi-square test of independence was used to test if proportions of manually classified vegetation habitats (high, medium, low, sparse, bare, and water) on three islands changed significantly over time based on aerial imagery. Proportion of area was calculated and measured by total cell count of each vegetation class in ArcGIS Pro. Classified vegetated habitats were analyzed on Raccoon Island in 2019, 2022, and 2023; on Philo Brice Island in 2019, 2021, and 2023; and on Queen Bess Island in 2021 and 2023. Post-hoc analysis used standardized residuals from Chi-square to compare habitat composition over years. All analyses were conducted in R, version 4.4.1 (R Core Team 2024).

Results

Nest Success

Apparent Nest Success. Data from 306 of 387 total monitored nests was used to calculate the apparent nest success (ANS) (Table 1). The fates of 81 nests were unknown and not included in the analysis due to camera malfunction, battery life, obstructed views from dense vegetation growth or obstruction by bird behavior. Following backward elimination, there was a strong interaction between year and island on ANS (Figure 4). For nests on shrubs, nest success was lowest on all three islands prior to Hurricane Ida than after the storm (Figure 4), although this difference across years was not significant for nests on Philo Brice Island. Before the hurricane, in 2021, the success of nests on shrubs was higher on Philo Brice Island, the unrestored site, than on either of the two restored sites. In 2022, immediately after the storm, the success of nests on shrubs was relatively the same across sites. In 2023, nest success remained high for Philo Brice and Raccoon Islands but was reduced for Queen Bess Island, although it was still above pre-storm levels (Figure 4).

When considering ANS for both nests on shrubs and on the ground, only Raccoon and Queen Bess Islands were included. The only significant interaction was between year and island on ANS. Nest success was 2-3 times higher the year after the hurricane than before it (Figure 5). In 2023, ANS was not different from 2022 on Raccoon Island, but declined on Queen Bess Island.

Daily Survival Rate. Daily Survival Rate (DSR) was determined for 370 of the original 387 total monitored nests because 17 nests had an exposure period of less than 1 day. When examining only nests in shrubs on three islands, there were significant main effects of year and island on DSR. DSR for nests in shrubs was significantly higher after the hurricane (2022, 2023) than before (Figure 6). DSR in shrubs was significantly higher on Philo Brice and Queen Bess Islands than on Raccoon Island (Figure 7). This result suggests that nests on islands located in or to the east of Ida’s track, had a positive effect on DSR. There were no significant interactions between the fixed effects when considering only nests located in shrubs. When modeling DSR of nests on Raccoon and Queen Bess Islands located on shrubs and on the ground, there were significant interactions between island and year and between island and nest vegetation location. Prior to the hurricane, DSR was lower on Raccoon Island than on Queen Bess Island; but DSR on both islands was higher after the hurricane (Figure 8). Nest survival was higher on the ground than in shrubs on Raccoon, while the opposite pattern was observed on Queen Bess island (Figure 9).

Fledging Success. Data from 306 of 387 total nests monitored were used to calculate fledging success (FS) because 81 nests had an unknown nest fate due to camera malfunction preventing monitoring to fledging date. The number of chicks that fledged each nest (reached at least 21 days) ranged from 0 to 3 with an average of 1.28 (Figure 10). FS was lowest on all three islands in 2021 before Ida (Figure 10). The only significant effects for FS across three islands for birds nesting in shrubs, and across two islands (Raccoon and Queen Bess Islands) with birds nesting both in shrubs and on the ground, was year (Figs Figure **11** & Figure **12**, respectively). For both comparisons, FS was much lower before the hurricane than in the subsequent years. Island and location of nests on shrubs or ground, and their interactions had no significant effect on these relationships.

Nest Failure

Of the 306 nests followed until there was a known fate, 96 were observed to have failed (Table 2). A Fisher's exact Test showed a significant association between island and proportion of failures ($p = 0.0011$). As expected from the analysis of nest success, there was no clear association between the proportion of failed nests and the proximity to the hurricane track. Philo Brice Island which was closest to the hurricane impact, had the lowest rates of nest failure (0.15 ± 0.05) (Figure 13). Pairwise tests showed significant differences between islands. The highest rate of nest failure occurred on Queen Bess Island (0.41 ± 0.04) compared to both Raccoon Island (0.28 ± 0.04, $p = 0.0430$) and Philo Brice Island (0.15 ± 0.05, $p = 0.0011$) (Figure 13). Although Raccoon Island had a higher rate of failure than Philo Brice Island, the difference was not significant ($p = 0.1018$).

The reason for abandonment was unknown for 43 nests. Loss during flooding and storm events was the most frequent cause of failures and the only identified cause of failure on Philo Brice Island (Figure 14).

Human disturbance was the next greatest cause of failure, resulting in loss of several nests on Queen Bess Island (n = 10) and on Raccoon Island (n = 2), but none on Philo Brice Island. Of the twelve nests apparently abandoned due to human disturbance (Figure 15), three of these nests were abandoned immediately following our placement of cameras, and another three were lost after subsequent visits to check sites (i.e., recover images or change camera batteries). Four nests were lost in association with nocturnal banding activities by LWDF on Queen Bess Island, and two nests failed due to diurnal disturbance by unidentified visitors such as anglers or other trespassing visitors (Figure 15).

Intraspecific competition for nest sites also occurred on Queen Bess Island, and I observed three nests that abandoned due to aggression between adults from different nests and/or aggression between adults and chicks from neighboring nests. I did not observe this pattern of behavior on Raccoon or Philo Brice Islands. Fire ants (RIFA) were associated with the loss of a single nest on both Queen Bess and Raccoon Islands, although this may be underreported due to the quality of images. The least frequent cause of failure was mammalian disturbance due to nutria (*Myocastor coypus*), which led to loss of one nest on Raccoon Island (Figure 14). Nests failed at different stages depending on the island (*p* = 0.0053). All the nests on Philo Brice Island failed during the egg incubation period (n = 8); nests with chicks were all successful (Figs Figure **16** & Figure **17**). Over 6 times the number of nests failed in the egg stage (n = 49, 86%) than in the chick stage (n = 8, 14%) on Queen Bess Island; however, on Raccoon Island the number of failed nests with eggs (n =

18, 58%) was only around 1.4 times the number with chicks (n = 13, 42%). Queen Bess Island nests failed more often in stage 1 compared to Raccoon Island (*p* = 0.0076) which had similar failure rates for stage 1 and stage 2 failures. Raccoon Island also differed significantly from Philo Brice Island which had all nests fail in stage 1 (*p* = 0.0352).

In an attempt to understand the causes of nest failure during different parts of the nesting cycle, I examined causes of failure for nests in stage 1 (Figure 16) and during stage 2 (Figure 17). Of the nests with known causes for failure (n = 53), the causes of nest failure did not significantly vary among islands for either stage 1 or stage 2 nests, potentially due to the small number of nests in each comparison. Flooding was a major cause of lost nests with either eggs or chicks. Human disturbance played a larger role in nest loss during egg incubation than during chick care (Figs Figure **16** & Figure **17**). Fire ants appeared to play more of a role in nest abandonment after the chicks had hatched, but the numbers of lost nests to ants was small (Figs Figure **16** & Figure **17**).

Vegetation on Raccoon Island Pre- and Post-Hurricane Ida

Raccoon overall PVC increased each year sampled from 2018-2023. The model detected a significant effect of time on overall percent vegetation cover (PVC) on Raccoon Island with 29.3% higher PVC post-storm than pre-storm ($F_{1,48} = 15.46$, $p = 0.0003$) (Figure 18a). Conversely, bare ground cover decreased by 28% on

Raccoon Island after Hurricane Ida indicating a substantial replacement of bare ground with vegetation ($F_{1,48}$ = 14.77, p = 0.0004) (Figure 18b).

Average vegetation height on Raccoon Island decreased following the storm ($F_{1, 44}$ = 4.32, p = 0.0435) and was lowest in 2022. Post-storm vegetation height was 16.4 cm lower than pre-storm levels, suggesting the storm may have impacted plant structure (Figure 18c). The model suggests maximum vegetation height differed significantly on Raccoon Island before and after the storm ($F_{1,44}$ = 2.43, p = 0.127) (Figure 18d). Plant cover showed temporal variation across habitats on Raccoon Island, but only for some functional groups. For example, marsh elder was only present in the unrestored transect pre-storm, however post-storm, it was recorded in both restored and unrestored marsh. Despite these differences, there were no significant temporal or habitat trends for marsh elder cover. Shrubs were only found in marsh habitats and total shrub cover varied from 7-18%, however, the model provided no significant evidence for temporal changes in coverage. Within habitats, mangrove cover was highest in the unrestored marsh and this habitat declined significantly after the storm (p = 0.0015) (Figure 19). Grass cover was highest in unrestored marsh before the storm (63%) and highest in the dune (77%) after the storm. Grass cover increased post-storm by 60% in restored dune (p = < 0.0001) and no significant changes were observed in marsh habitats (Figure 20). Forb cover was highest in unrestored marsh (5-15%). There was no evidence forb cover differed before and after the storm. Bare cover was highest in restored dune and restored marsh habitats before the storm and both underwent significant decreases in bare areas after the disturbance, experiencing increased vegetative cover (Figure 21).

Vegetation on Raccoon and Queen Bess Islands in 2022

In 2022, the island east of the storm track, Queen Bess Island, had 25% higher PVC than the island west of the storm, Raccoon ($F_{1,\ 28}$ = 8.82, p = 0.0061 (Figure 22a). Bare ground cover was 17% higher on Raccoon Island than Queen Bess Island ($F_{1,\ 28}$ = 4.59, p = 0.0410) (Figure 22b). Maximum vegetation height was 74.5 cm higher on Queen Bess Island (122.75 ± 18.61 cm) compared to Raccoon Island (48.22 ± 15.20 cm) ($F_{1,\ 28}$ = 9.63, p = 0.0044) (Figure 22c). The average vegetation height was 32 cm higher on Queen Bess Island (55.2 ± 9.38 cm) compared to Raccoon Island (23.5 ± 7.66 cm) ($F_{1,\ 28}$ = 6.87, p = 0.0140) (Figure 22d).

Each island was composed of woody and herbaceous plants with grass species making up the majority of both Queen Bess and Raccoon Island plots. Three woody species are important for Brown Pelican nesting including black mangrove, marsh elder, and Christmasberry. Other shrubs on the islands include saltwort and coastal goldenbush. Queen Bess had 19% average cover of Christmasberry and a higher cover of black mangrove than Raccoon, while Raccoon had a higher cover of marsh elder and an absence of Christmasberry. Raccoon and Queen Bess Islands had no significant differences in cover of black mangrove or marsh elder, however, total shrub cover was 21% higher on Queen Bess Island than Raccoon Island (p = 0.0089) (Figure 23). The interaction between island and habitat type was significant for shrub cover ($p = 0.0453$). The unrestored portion on Queen Bess Island had twice the shrub cover as the restored marsh. Conversely, shrub cover was more than two times higher in restored marsh compared to unrestored marsh on Raccoon Island (Figure 24).

Analysis of non-shrub vegetation categories showed forb cover did not differ significantly between the two islands. The interaction between island and habitat type was significant for grass cover ($p = 0.0017$). The restored marsh on Queen Bess Island had almost three times the cover of grass than the unrestored marsh. Raccoon Island had the opposite trend, with 16% more grass cover in the unrestored than restored marsh (Figure 25).

Remote Sensing and Island Change

Aerial and satellite raster analysis showed that the area of Raccoon Island increased by 41.3% in 2022, the year following Hurricane Ida, and decreased 34% from 2022 to 2023 (Suppl. Table 3). The unrestored island, Philo Brice Island, decreased in size by 10.7% from September 2020 to September 2021 (a few days after Ida made landfall in LA) and increased by 9.4% the following year in 2022 (Suppl. Table 3). In April 2023, Philo Brice Island area decreased by 14.5%, and by July 2023 decreased an additional 5.9%. Queen Bess Island, restored in 2021, decreased by 4.2% the year following the storm and decreased an additional 1.3% from 2022 to 2023 (Suppl. Table 3). However, observations of island size may have been affected by seasonal tidal and water levels, so examination of vegetative portions of the islands might be more informative.

Island Habitat Classification. Remote sensing techniques were used to calculate area (ha) of each of the 6 vegetation classifications assigned based on NDVI values per pixel, including water, sand/bare, dead/sparse, low, medium, and high-density vegetation on the three breeding islands before and after Hurricane Ida (Table 3, Suppl. Table 4). Although standardization and resampling of imagery were conducted, results should be taken with some degree of caution, however, due to seasonal effects, potential effects of atmospheric reflectance, and the differences in sensors and spatial resolution between available imagery sources for analysis between years (Suppl. Table 2 and 4).

There was a significant association between year and classified vegetated habitats on Raccoon Island ($\chi^2 = 1,444,909$, $p < 0.0001$) and all chi-square residuals were significant (Table 3, Suppl. Table 5, Figure 26). The area of dead/sparse habitat exhibited significant shifts with higher coverage in 2022 than either 2019 or 2023 (Figure 26). Low-density vegetation increased slightly in 2023 compared to 2019 and 2022, and medium-density area decreased each year sampled (Figure 26). High-density vegetation area showed significant differences with around 13% coverage of the island in 2019 decreasing to nearly 0% in 2022 and 2023 (Suppl. Table 5, Figure 26).

The habitat composition of Philo Brice Island changed significantly between 2019, 2021 and 2023 ($\chi^2 = 230{,}221.60$, $p < 0.001$) (Table 3, Suppl. Table 6, Figure 27). Dead/sparse and sand/bare areas declined over the study period, while medium and high-density vegetation area increased in 2021 then declined in 2023 (Figure 27). Queen Bess Island experienced significant differences in habitat composition between 2021 and 2023 ($\chi^2 = 33461.072$, $p < 0.001$) (Table 3, Suppl. Table 7, Figure 28). High-density areas decreased after the storm. Conversely, low-density, medium-density vegetation and sparse/dead areas increased from pre-storm levels (Figure 28).

Discussion

Reproductive Success

I found the productivity of nesting pelicans in the 3 Louisiana colonies ranged from 0.76 – 2.1 chicks per nest, which was similar to or higher than other Brown Pelican colonies in the northern Gulf. Schreiber (1979) studied a pelican colony on the west coast of Florida from 1969-1970 and found the number of fledglings per successful nest ranged from 1.3 - 1.5 and the percent of successful nests ranged from 43-65%. Holm *et al.* (2003) looked at Louisiana populations of nesting pelicans and determined the overall mean nestling per successful nest from 1971-2001 was 1.66. Walter *et al.* (2013) found the number of fledglings prior to a hurricane to be 0.7 - 1.6 on Raccoon Island and nearby Wine Island. On Gaillard and Cat Islands in Alabama, 1.02 – 1.29 chicks fledged per nest with DSR of nests ranging from 91-99% during incubation and brood-rearing (Streker *et al.* 2021).

The different measures of reproductive productivity all showed a strong increasing trend following Hurricane Ida compared to observations before the storm. When there were differences among the islands in ANS, post-hurricane increases were strongest on Philo Brice Island, which was closest to the storm's impact. Although the patterns I observed across years could be due to other factors than the effects of Hurricane Ida, and it would be optimal to have more data over a longer period of time, none of the measures of nest success show any evidence of a negative impact of the storm.

My observation of increased nest success following Hurricane Ida contrasts with the results of Walter *et al.* (2013). They found that FS decreased on Raccoon Island in the years following Hurricane Gustav in 2008, which they attributed to a reduction in shrubs. Based on remote sensing and vegetation surveys, I did not detect a large decrease in overall shrub vegetation, which may explain why I did not see a decrease in nest success, but it does not explain why I saw the increases in DSR, ANS, and FS following Hurricane Ida.

Walter *et al.* (2013) did observe an increase in FS on Wine Island the year following Gustov, despite an almost total loss of shrub habitat, which forced pelicans to nest on the ground. The increase they observed on Wine Island was similar to the post-storm increase observed in my study. They speculated that the increase in pelican productivity might have been due to the overwash from Gustov eliminating fire ants on Wine Island during the breeding season after the storm. While such an effect would be possible on both Raccoon and Queen Bess Islands in my study, it would be unlikely on Philo Brice Island, because the habitat was too consistently inundated and wet to support populations of fire ants.

I found that nest success was influenced by whether pelicans nested in shrubs or on the ground. This effect was strongly significant in DSR on Queen Bess Island, where nests on the ground had much lower daily survival than nests in shrubs. This effect was also strongly significant in DSR on Raccoon Island, where nests on the ground had a much higher DSR than shrub nests. Nests on Queen Bess Island were more prone to loss during flood events than on Raccoon Island (see below). This might have occurred because Queen Bess Island tends to flood and retain water in pelican nesting areas more commonly than Raccoon Island (pers. observation). Furthermore, shrub habitat is rarer on Queen Bess Island than on Raccoon Island (see below), forcing more nesting on the ground.

While nests on the ground had higher productivity than nests in shrubs on Raccoon Island, the same was not true for Queen Bess Island which had higher productivity in shrubs. Establishing more shrubs on Queen Bess Island would probably help decrease losses of nests due to high water events.

Robinson & Dindo (2011) reported higher reproductive success for Brown Pelican nests in lower vegetation densities and highest success in areas where ground nests were most frequent. In contrast, Walter *et al.* (2013) found that pelicans on Raccoon Island had higher FS when nesting on shrubs than on the ground, but that there was little difference in FS due to nest substrate on Wine Island. In the years since the Walter *et al.* (2013) study, Raccoon Island elevation increased due to a restoration project, which may have decreased the negative consequences of pelicans nesting on the ground. Interestingly, the site examined by Robinson & Dindo (2011) with high success for nests on the ground, was a man-made and had a high elevation for a coastal island.

Nest Failure

Flooding due to high tides and storm events were identified as the major cause of nest abandonment on all three islands. High water often made it difficult to incubate eggs. The portion of lost nests was lowest on Philo Brice Island, which was unrestored, even though it had the lowest elevation and throughout the season remained mostly inundated. The lack of dry land forced all the birds to nest in elevated mangroves and these nests were generally resistant to flooding. Compared to Philo Brice Island, more nests were lost on Raccoon and Queen Bess Islands, both of which had undergone recent restoration. Promotion of more black mangrove availability, as suggested by Walter *et al.* (2013), on those sites may decrease nest losses to flooding.

Wind, rain, and wave action due to hurricanes can cause direct mortality to nesting waterbirds (Wiley & Wunderle 1993). Wind damage can negatively impact the number of breeding pairs as was recorded after 2005 hurricanes in southern Louisiana waterbirds (Leberg *et al.* 2007). Wind driven flooding was shown to cause up around 50% of the losses of two waterbird species and that percentage increased gradually over a 50 year period (Koivula *et al.* 2025). Hurricanes can also lead to colony abandonment by colonial waterbirds, but not necessarily decrease overall numbers of birds (Leberg *et al.* 2007). An indirect result is beach erosion, which can make the remaining parts of the island more vulnerable to high tides and nest flooding. Cely (1991) noted the hurricane directly killed up to 400 Brown Pelicans, and following erosional impacts from Hurricane Hugo, flooding from high tides decreased then number of Brown Pelican nests by 15% in coastal South Carolina.

Human disturbance, including recreation and research, can disrupt seabird colonies and lead to decreased nesting success (Anderson & Keith 1980; Boellstorff *et al.* 1988). Disturbing nesting activities is always a concern when researchers enter a dense seabird colony and can cause negative impacts such as loss of eggs, predation and abandonment (Anderson & Keith 1980; Bolduc & Guillemette 2003). A study of breeding American White Pelicans (*Pelecanus erythrorhynchos*) in North Dakota reported 99% failure of over 200 nests in 4 colonies that were visited just once by researchers, though they noted this was not necessarily from human disturbance and could have also been attributed to inadequate food supplies (Johnson & Sloan 1978). Boellstorff *et al.* (1988) reported only 0.5 fledglings per nest on colonies of American White Pelicans that were visited by researchers, compared to 1.2 fledged per nest on an undisturbed colony.

Brown Pelicans have been shown to have a negative relationship between reproductive success and the frequency of human disturbance (Schreiber & Risebrough 1972; Schreiber 1979; Anderson & Keith 1980). For example, Anderson & Keith (1980) found researcher-disturbed California Brown Pelican nests fledged only 0.16 chicks per nest compared to undisturbed areas, which fledged an average of 1.5 chicks. Brown Pelicans incubate with their feet and when an intruder flushes adults, sometimes eggs can break and eggs and naked hatchlings are left to endure potential temperature stress (Schreiber & Risebrough 1972). Furthermore, Lamb *et al.* (2020) found that the longer a captured Brown Pelican was handled for tagging (transmitter attachment), the lower the breeding success of that bird, with a significant decrease in nesting success if handled for greater than 20 min. When humans approach active nests, the general reaction of Brown Pelicans is to flap their wings, flush the nest, circle above the nest, or land in the water nearby. In our work setting up and maintaining nest cameras, we did not observe any direct egg breakage and most adult pelicans returned to their nests within as little as 1 minute (as recorded on images).

While the probable loss of 6 nests to my activities is undesirable, it is not clear how future researchers could further reduce nest losses. One solution might be to use cameras with solar panels to recharge batteries and remote image downloads. This would reduce the number of times an investigator would need to visit a nest. The remoteness of some of the islands, such as Raccoon and Philo Brice Islands makes remote downloads over the cellular network unpractical, but this solution could help reduce visits to Queen Bess Island. Larger storage capacity might also reduce the need for visits to download images, but would increase the risk of images being lost to storms between visits.

If it is early in the breeding season, Brown Pelicans may abandon nests and pelicans captured in Louisiana show a higher abandonment rate than in nearby states (Walter *et al.* 2014). I also found abandonment was more common during egg incubation than after the eggs hatch. This is not surprising as birds become more committed to their eggs later in the nesting cycle, and this aligns with previous findings where the DSR of Brown Pelicans during incubation (91-99%) was lower than DSR during brood-rearing (> 99%) (Streker *et al.* 2021). Therefore, another solution to reduce investigator-induced nest loss would be to limit visits to periods of time when egg incubation is largely complete, as most losses to disturbance occurred during the early phases of nesting. However, such a strategy would result in considerable loss of information about the success and failure of nests during incubation.

I attempted to keep my disturbance of nests to a minimum only visiting during the morning and keeping my visits short. LDWF banded pelicans at night to decrease disturbance of nesting birds, and focused activities after most chicks are hatched, but several nests were still abandoned on nights when banding occurred. Ultimately, investigators and management agencies need to determine if the small but real risk of nest loss (3% of monitored nests in this study) is worth the information gained by monitoring nests with cameras or banding chicks and/or adults.

Losses due to other humans visiting the islands might also be unpreventable. Both Raccoon and Queen Bess Islands are managed wildlife refuges that are posted to prevent intrusion, but humans not apparently approved for visitation are occasionally observed (as seen on images). However, the remoteness of these sites would make greater enforcement of access restrictions difficult and would probably not be justified given the small numbers of nests lost.

I have observed little evidence that mammalian predation had any effect on nest failure. There is no evidence of mammalian carnivores such as raccoons (*Procyon lotor*) or coyotes (*Canis latrans*) on any of these islands, so this result is not surprising. Nutria have been observed disturbing other seabirds nesting in this system (Windhoffer 2017), but the levels of disturbance I observed were low with only one pelican nest being abandoned after disturbance by nutria.

Intraspecific competition due to crowded conditions in colonies can result in death of unattended young caused by other adults (Rickfels 1969). This behavior was observed to cause nest abandonment on Queen Bess Island, the most densely occupied island, most likely due to crowded conditions and increased competition for nesting sites and resources.

Invasive ant species are known predators of seabirds nesting on islands (Plentovich *et al.* 2009; DeFisher & Bonter 2013; Boieiro *et al.* 2018). Fire ants are a known predator on other ground-nesting birds (Giuliano *et al.* 1996; Seymour 2007) and specifically colonially-nesting waterbirds (Drees 1994; Lockley 1995) and have been suggested as potential predators on Brown Pelican chicks (Walter *et al.* 2013) with some evidence for negative effects on pelican nesting success (Rosamond & Geary 2017, B. Geary unpubl. data). Rosamond & Geary (2017) note RIFA can inflict physical harm by stinging adults, leading to stress on parents, with RIFA only explaining some of the variance in nest success. However, to my knowledge, mine is the first study to record and demonstrate direct fire ant predation on pelican chicks. In general, fire ant predation was uncommon although it was possible that these small insectivorous predators went undetected in some of my camera images due to camera resolution capabilities. At the present time, fire ant management is probably not justified to protect pelican chicks, but it might be appropriate for other species of concern, as it could also promote fledging success of terns and skimmers on the same islands (Santariello 2025).

It was not possible for me to determine the cause of nest failure for about 30% of the failed nests. Cameras may have missed the cause of nest abandonment, or these nests were abandoned due to flooding, mortality of a parent (Robinson & Dindo 2011), age of the parent (Blus & Keahey 1978), or low food supplies leading to starvation (Rickfels 1969). These causes of nest abandonment could not have been detected using cameras and would require following the fates of individual nesting birds.

Vegetation

My results indicate substantial changes in vegetation composition and land cover over time, likely reflecting both natural variability and/or disturbance events between 2019 and 2023. Hurricane Ida likely contributed to these changes.

From 2018 to 2023, overall vegetation cover on Raccoon Island increased by about 30% post-storm with marsh elder expanding from the unrestored marsh transect to all marsh habitats post-storm. Marsh elder is a salt-tolerant shrub that has been shown to regenerate after severe damage (Michener *et al.* 1997), and it showed consistent cover before and after the storm on Raccoon Island. Black mangrove, also salt-tolerant, cover decreased on Raccoon Island, but only in the unrestored marsh transect. This suggests higher stability and resilience in the restored portion of the island. Grass cover also increased after the storm, specifically in the dune. Additionally, forb cover remained consistent, suggesting an overall resilient vegetation composition after a major hurricane.

Walter *et al.* (2013) found different trends than I observed after Hurricane Gustav, with declines recorded primarily in marsh elder (85% loss) and black mangrove (25% loss). Furthermore, they found grass and forb cover decreased the year after the storm, although it eventually returned to pre-hurricane levels on Raccoon Island, reflecting patterns of early successional plants adapted to colonize rapidly after a disturbance (Hester & Willis 2015). Wine Island, however, underwent further vegetation declines, and eventually retreated below sea level. These declines were attributed to shoreline retreat due to land loss post-hurricane (Walter *et al.* 2013).

Higher measures of PVC, vegetation height, and lower bare ground cover on Queen Bess Island, than on Raccoon Island, after Hurricane Ida may be due to post-restoration (including vegetative plantings) and post-hurricane vegetation succession (Hester & Willis 2015). Lower vegetation cover and higher bare ground cover on Raccoon Island, compared to Queen Bess Island, after the storm could be attributed to lower post-hurricane vegetation recovery rates, higher rates of herbivory (e.g., nutria) or longer time since restoration (Hester & Willis 2015; Nyman *et al.* 2019). Additionally, previous analysis of the flooding impacts in these sites after Hurricanes Katrina (2005) and Rita (2005) showed that Terrebonne Basin (Raccoon Island location) saw six times the conversion of land to open water as did Barataria Basin (Queen Bess Island location) showing differential spatial response to the same event (Barras 2007). Each storm has variable effects, but this could be one possible explanation for the differences in vegetative cover on the two islands.

The decrease in PVC, vegetation height, and increase in bare cover on Raccoon Island after Hurricane Ida could be partially attributed to disturbance of the storm, but the effects of the storm on vegetation were expected to be worse on Queen Bess Island, located in the right quadrant and closer to the path of Ida (Doyle *et al.*

1995; Keim *et al.* 2007), and this was not the case. It is also possible that the changes in vegetation on Raccoon Island were partly due to herbivory pressure from invasive mammals leading to an increase in dead or sparse areas and mostly lower vegetated areas overall post-Ida in 2022. Nutria were observed in large numbers each year on Raccoon Island and these non-native invasive herbivorous mammals have been shown to predate other ground-nesting seabirds on this island (Windhoffer 2017), directly graze on vegetation in some cases causing ‘eatouts’ (Nyman *et al.* 2019), and cause disturbance through burrowing (Baroch *et al.* 2002). In other systems, nutria caused disturbance through predation of eggs (Windhoffer 2017), and destruction of eggs by standing on nests (Bertolino *et al.* 2011). All of these factors could cause disturbance and stress to both plants and nesting birds by destabilizing roots, and therefore subsequently increased erosion rates (Baroch *et al.* 2002). Declines in PVC and average vegetation height on Raccoon Island could be attributed to plant species turnover with shifts from shrubs (e.g., black mangrove) towards shorter herbaceous vegetation (e.g., saltgrass).

Finally, climatic factors in the summer of 2022 may have had an impact on declines in vegetation. From March to July 2022, Terrebonne, Lafourche, and Jefferson Parishes in Louisiana underwent an extreme drought (D3 on a scale of 5 levels from D0 Abnormally Dry to D4 Exceptional Drought) (U.S. Drought Monitor, NOAA NIDIS). However, it is not clear why drought effects would be worse on Raccoon Island than on Queen Bess Island. Nesting pelicans may be negatively impacted by harsh winters and drought, for example, decreases in Brown Pelican nest numbers occurred in the spring of 1990 and 1996 corresponding to the breeding season following extended freezing temperatures in Louisiana. In 1999 and 2000, a 50–100-year local drought coincided with low Mississippi River levels resulted in high salinity across the Louisiana coast (Holm *et al.* 2003).

Island Change

Island areas before and after Hurricane Ida differed between restored and unrestored sites, suggesting variable recovery responses to storm disturbance. Raccoon Island, which is restored, experienced a substantial 41.3% increase in area in 2022 from 2021. This gain in area was most likely due to sediment deposition and redistribution due to seasonal and hurricane-driven processes. The next year the island dropped in area by 34%, indicating erosion and/or subsidence. This landscape undergoes constant fluctuations, which may at times benefit from overwashed sediment replenishing land, but can remain vulnerable to ongoing RSLR, erosion and subsidence.

In contrast, unrestored site, Philo Brice Island, exhibited yearly fluctuations in area each year over the study period, losing 1.9 ha of land alone from September 2020 to September 2021 (a few days after Ida). This trend of land loss reflects what I would expect for an unrestored island, and despite the decreasing availability of nesting habitat, pelican nesting success remained high and comparable to restored islands.

The area of Queen Bess Island was restored in 2020 and remained mostly consistent throughout the study, with minor declines in area that could be actual or due to water level anomalies or seasonal tidal variations. Analyzing total island area changes using remote sensing is valuable and offers insight into geomorphological dynamics and availability of nesting habitats. However, seasonal tidal variation, water level anomalies, and satellite sensor characteristics may bias some assumptions or calculations. The remote location of these islands, far from inhabited regions, meant the availability of imagery was greatly reduced compared to more populated regions of the globe. Therefore, ground-truthing of vegetation data and habitat classification is recommended in future studies on these islands.

Frequency, intensity, and timing of hurricanes can directly alter island vegetation (Doyle *et al.* 1995), which can consequently alter the locations and frequency of inundation (Michener *et al.* 1997). Stress to vegetation from wind damage and waterlogging has been shown to cause declines in mangrove stands, with damage to foliage higher with decreasing distance from the storm track (Doyle *et al.* 1995; Michener *et al.* 1997).

Although I documented minimal reductions of nesting habitat, and no reductions in reproductive success of pelicans, following a major hurricane, the loss of islands as nesting sites due to hurricanes can impact reproductive success. In 2018, Hurricane Walaka passed over East Island near Hawai'i causing complete island loss and consequently displacing over 2,000 breeding pairs of Black-footed Albatrosses

(*Phoebastria nigripes)* (Young *et al.* 2024). Jodice *et al.* (2007) found the counts of Brown Pelican nests at Deveaux Bank, South Carolina, increased between 1969-1979, but this nesting site was severely eroded by Hurricane David in 1979 undergoing complete submergence. This resulted in no pelicans nesting there until the 1990s, after substantial accretion enhanced the habitat suitability for nesting pelicans (Jodice *et al.* 2007). Walter *et al.* (2013) documented the loss of Wine Island as nesting habitat two years after Hurricane Gustov passed nearby. This loss was associated with the loss of marsh elder as a result of the hurricane. The loss of marsh elder, the dominate shrub on Wine Island, was followed by rapid erosion. Similarly, on a wading bird breeding colony in coastal South Carolina, following Hurricane Hugo, marsh elder stands were substantially damaged losing up to 75% of above-ground woody biomass (Shepherd *et al.* 1991).

Both marshes and mangroves offer shoreline protection against storms to varying degrees, with black mangroves found to provide more protection than marshes in nearby coastal Texas after Hurricane Harvey (Doughty *et al.* 2017; Pennings *et al.* 2021). But, in some cases, hurricanes can inflict great damage to mangrove forests (Doyle *et al.* 1995) with island mangrove stands undergoing significantly more damage than mainland forests after hurricanes in Florida systems (Smith *et al.* 2009).

The islands I studied may have fared better because black mangroves tended to be the dominant shrub vegetation. Both Walter *et al.* (2013) and my research suggests black mangrove is fairly resistant to mortality following hurricanes.

I was surprised that Philo Brice Island experienced little loss of habitat after the passage of Hurricane Ida. Nearby, many similar small, low elevation mangrove islands used by nesting pelicans have been lost as a result of erosion (Selman *et al.* 2018). It is not clear how long Philo Brice Island will continue to support nesting pelicans without restoration, but this island's continued persistence during this study suggests that another major storm event will not necessarily result in its loss as a nesting site for Brown Pelicans. Alleng (1990) noted that despite extensive mangrove habitat damage due to Hurricane Gilbert on Jamaica's coast, a population of waterbirds continued to use the site for nesting post-hurricane and no change in population size was observed. Furthermore, following Hurricane Hugo, Brown Pelican populations in the Virgin Islands had higher reproductive success than average and apparently adjusted nesting behaviors to utilize downed vegetation (Pierce 1990).

Management Implications

Certain species are able to adapt to habitat changes driven by hurricanes, and despite ongoing sea level rise and hurricane activity, my study shows Brown Pelicans have some degree of plasticity in nesting adaptations. My study emphasizes the importance of long-term monitoring of seabird nesting habitat to inform future restoration decisions. My results also show that an ecological approach is needed for monitoring and project development. Island availability, islands area, vegetation composition, and vegetation structure play vital roles in creating suitable nesting habitat for Brown Pelicans. These long-term monitoring projects are labor-intensive, time consuming and expensive, but the benefits outweigh the costs when limited dollars are at stake. Rapid rates of coastal land loss have led to the priorities of restoring barrier and coastal islands for storm protection inland. In the efforts to nourish and protect our quickly disappearing coastal habitats, we must prioritize impacts to barrier island systems.

The responses of islands and nesting pelicans to hurricanes vary considerably. Based on the work of Walter *et al.* (2013), Hurricane Gustov had a negative effect on island persistence, woody vegetation, and pelican nest success. My results paint a different picture after Hurricane Ida. Walter *et al.* (2013), and Selman *et al.* (2016) found that island loss was common following storms and other environmental disturbance, and that restored islands seemed more resilient to loss than islands with little or no recent restoration. All three islands I studied were resistant to loss of habitat due to Hurricane Ida and the unrestored island fared well compared to the two islands that were recently restored. Basing management decisions on the disparate results of these studies will provide challenges. Given the large number of hurricanes and tropical storms that track through and near southeastern Louisiana, a better understanding of the interactions between restoration, vegetation composition and island loss in the face of hurricanes, is critical to the long-term management of seabird nesting colonies in Louisiana.

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Tables and Figures

Table 1. The Brown Pelican daily survival rate (DSR), apparent nest success (ANS), and total number of nests included in the calculation from 2021-2023 on 3 islands.

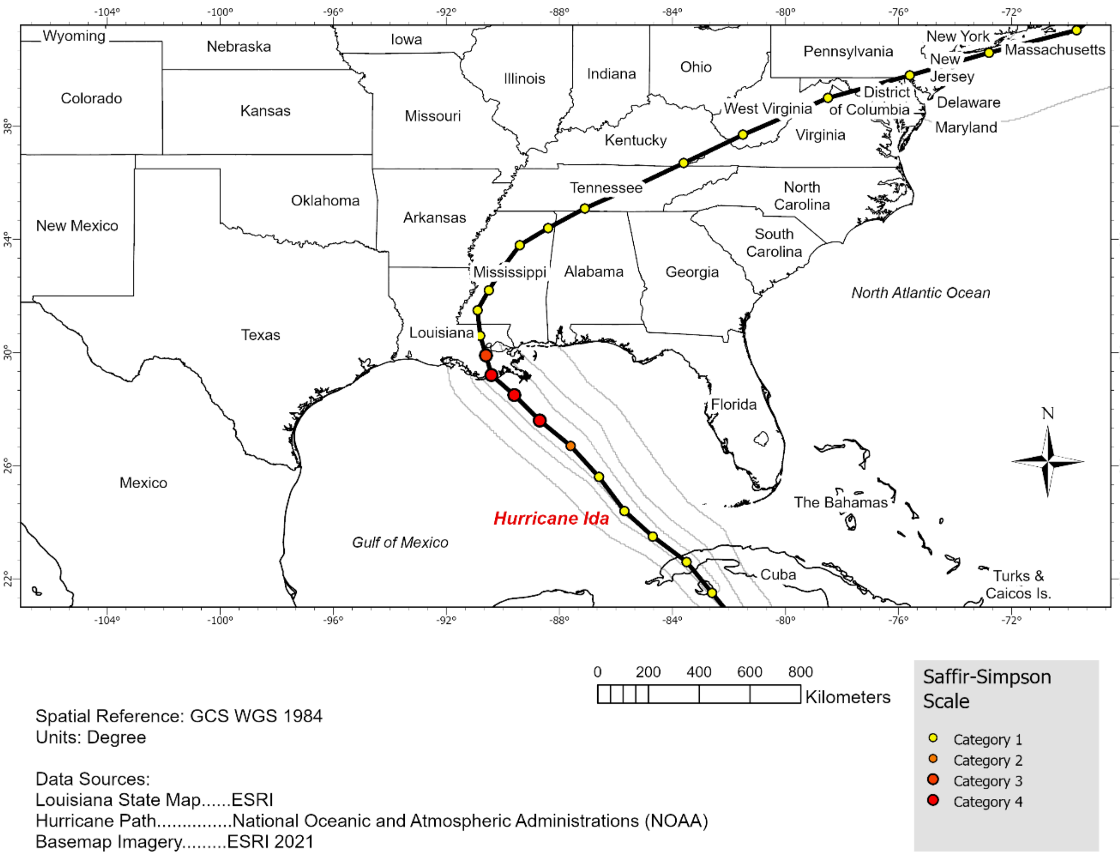
Island	Year(s)	DSR	No. of nests	SE	ANS	No. of nests	SE
Raccoon	2021-2023	0.991	141	0.0016	0.72	112	0.0265
Philo Brice	2021-2023	0.995	58	0.0018	0.85	54	0.0483
Queen Bess	2021-2023	0.987	171	0.0018	0.59	140	0.0415
Raccoon	2021	0.978	47	0.0044	0.43	44	0.0747
Raccoon	2022	0.997	48	0.0019	0.88	32	0.0585
Raccoon	2023	0.999	46	0.0011	0.94	36	0.0382
Philo Brice	2021	0.999	16	0.0039	0.64	14	0.1280
Philo Brice	2022	0.994	20	0.0062	0.94	18	0.5399
Philo Brice	2023	0.997	22	0.0018	0.91	22	0.0613
Queen Bess	2021	0.983	89	0.0027	0.45	75	0.0575
Queen Bess	2022	0.997	39	0.0020	0.85	27	0.0684
Queen Bess	2023	0.989	43	0.0032	0.68	38	0.0754
All	2021	0.983	152	0.0021	0.47	133	0.0432
All	2022	0.997	107	0.0013	0.88	77	0.0366
All	2023	0.995	111	0.0013	0.83	96	0.0380

Table 2. Numbers of failed and successful nests used to understand nest failures on three islands.

Island	Failed Nests	Successful Nests	Proportion Failed	SE Proportion
Raccoon	31	81	0.277	0.042
Philo Brice	8	46	0.148	0.048
Queen Bess	57	83	0.407	0.042

Table 3. Absolute changes in classified vegetation areas (ha) from raster analysis on three islands over time before and after Hurricane Ida (2019-2023).

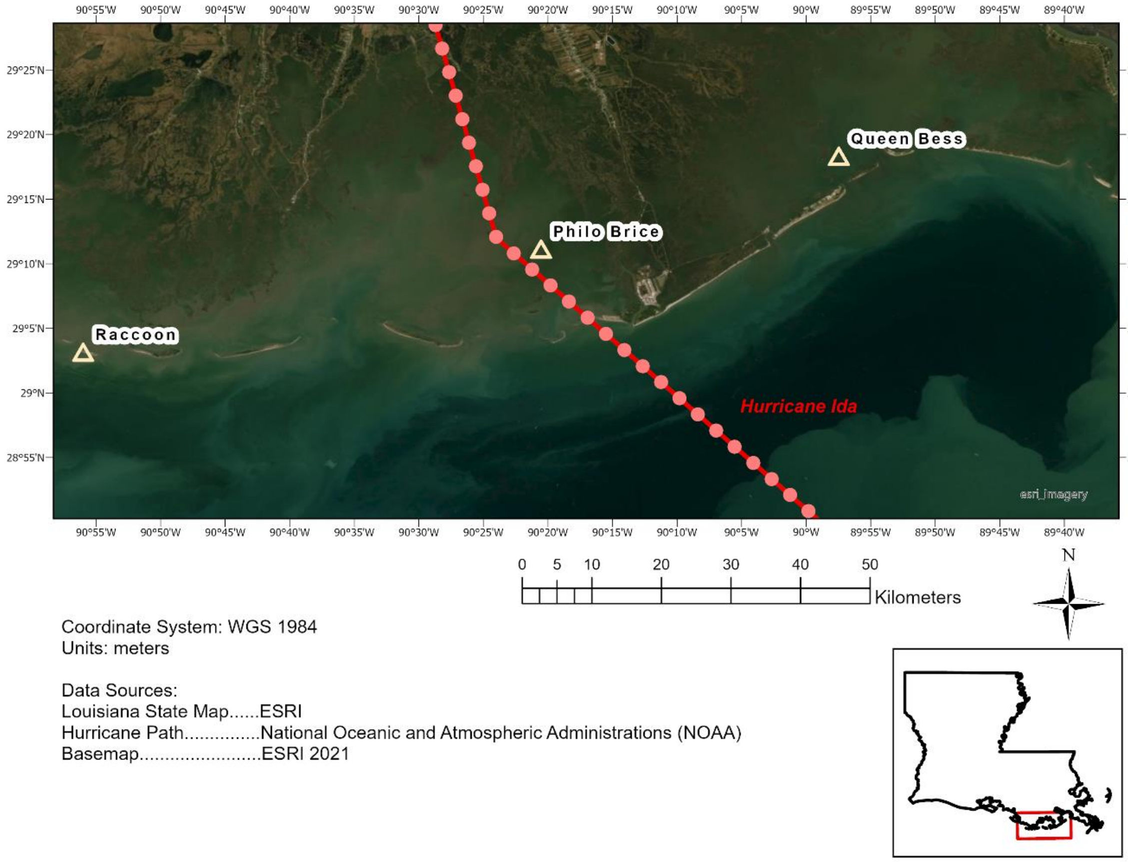
	Absolute Area (ha)				Area Change (ha)	Area Change (ha)	Area Change (%)
Island Habitat	**2019**	**2021**	**2022**	**2023**	**2019-2022**	**2019-2023**	**2019-2023**
Raccoon							
Water	10.17	-	1.88	4.07	-8.29	-6.11	-60
Sand	20.63	-	48.28	31.31	27.65	10.68	52
Dead/Sparse	10.01	-	16.22	5.32	6.21	-4.69	-47
Low	4.09	-	4.06	5.79	-0.03	1.70	41
Medium	6.91	-	4.63	3.41	-2.28	-3.50	-51
High	9.84	-	2.26	1.14	-7.58	-8.70	-88
Philo Brice					**2019-2021**	**2019-2023**	**2019-2023**
Water	4.63	0.02	-	0.05	-4.60	-4.58	-99
Sand	3.56	0.14	-	1.02	-3.42	-2.54	-71
Dead/Sparse	3.27	0.98	-	1.72	-2.29	-1.55	-47
Low	3.55	2.43	-	3.72	-1.12	0.17	5
Medium	5.16	8.45	-	4.36	3.30	-0.79	-15
High	0.29	3.48	-	2.79	3.19	2.50	866
Queen Bess						**2021-2023**	**2021-2023**
Water	-	1.05	-	0.55	-	-0.50	-48
Sand	-	1.78	-	2.23	-	0.45	25
Dead/Sparse	-	2.63	-	3.17	-	0.53	20
Low	-	2.73	-	2.92	-	0.19	7
Medium	-	3.71	-	5.57	-	1.86	50
High	-	3.18	-	0.14	-	-3.04	-95



Spatial Reference: GCS WGS 1984
Units: Degree

Data Sources:
Louisiana State Map......ESRI
Hurricane Path...............National Oceanic and Atmospheric Administrations (NOAA)
Basemap Imagery.........ESRI 2021

Figure 1. Map of path of Hurricane Ida ratings on the Saffir-Simpson Scale as it traveled through the Gulf of Mexico making impact near Port Fourchon, Louisiana on August 29, 2021.



Coordinate System: WGS 1984
Units: meters

Data Sources:
Louisiana State Map......ESRI
Hurricane Path...............National Oceanic and Atmospheric Administrations (NOAA)
Basemap.......................ESRI 2021

Figure 2. Map of three Brown Pelican breeding colonies in Barataria and Terrebonne Basins in reference to the path of Hurricane Ida in 2021.

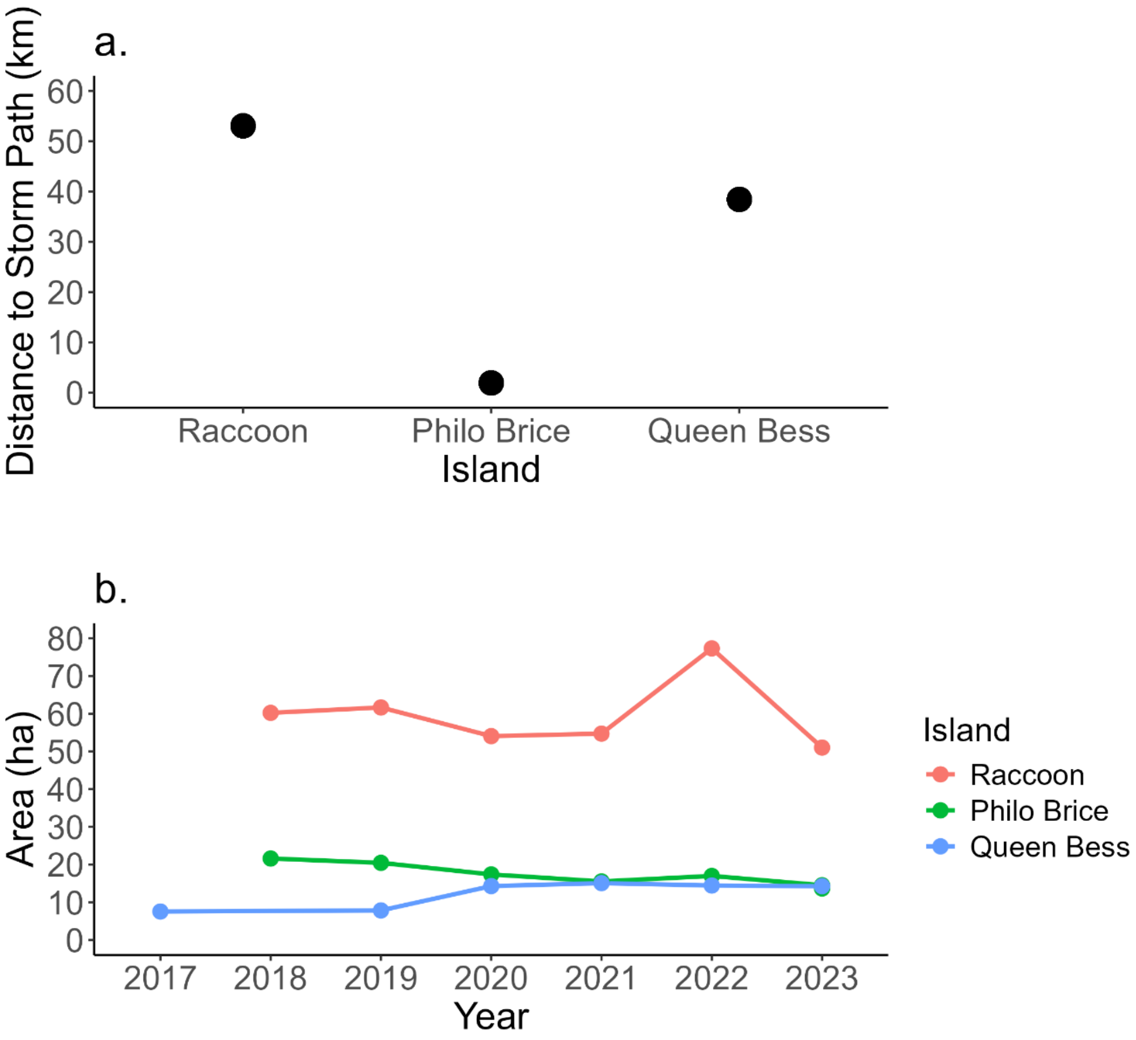


Figure 3. a) Distance of three study islands to storm path (km) of Hurricane Ida on August 29, 2021. b) Total area in hectares of islands in each year available from 2017-2023 calculated from remote sensing data.

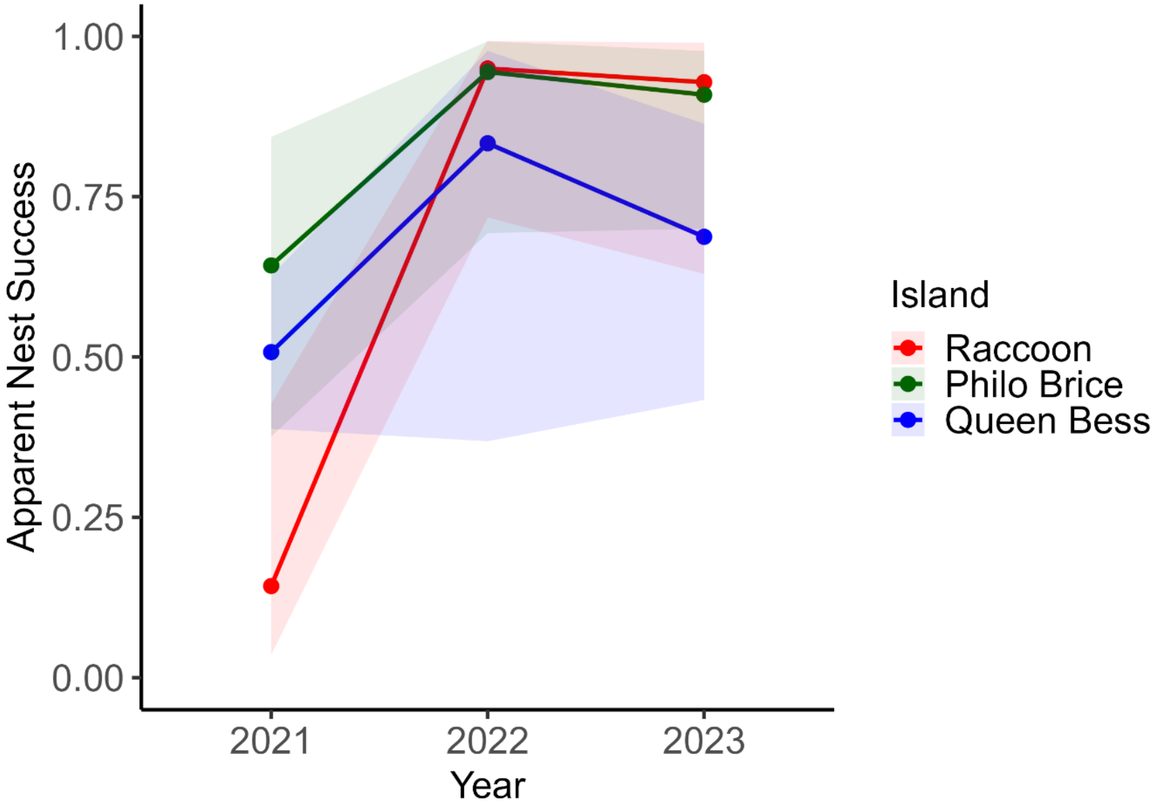


Figure 4. Interaction of year and island on apparent nest success (with 95% CI ribbons) of Brown Pelicans nesting in shrubs from 2021-2023 on Raccoon, Philo Brice, and Queen Bess islands. Hurricane Ida affected the area after the 2021 data were collected.

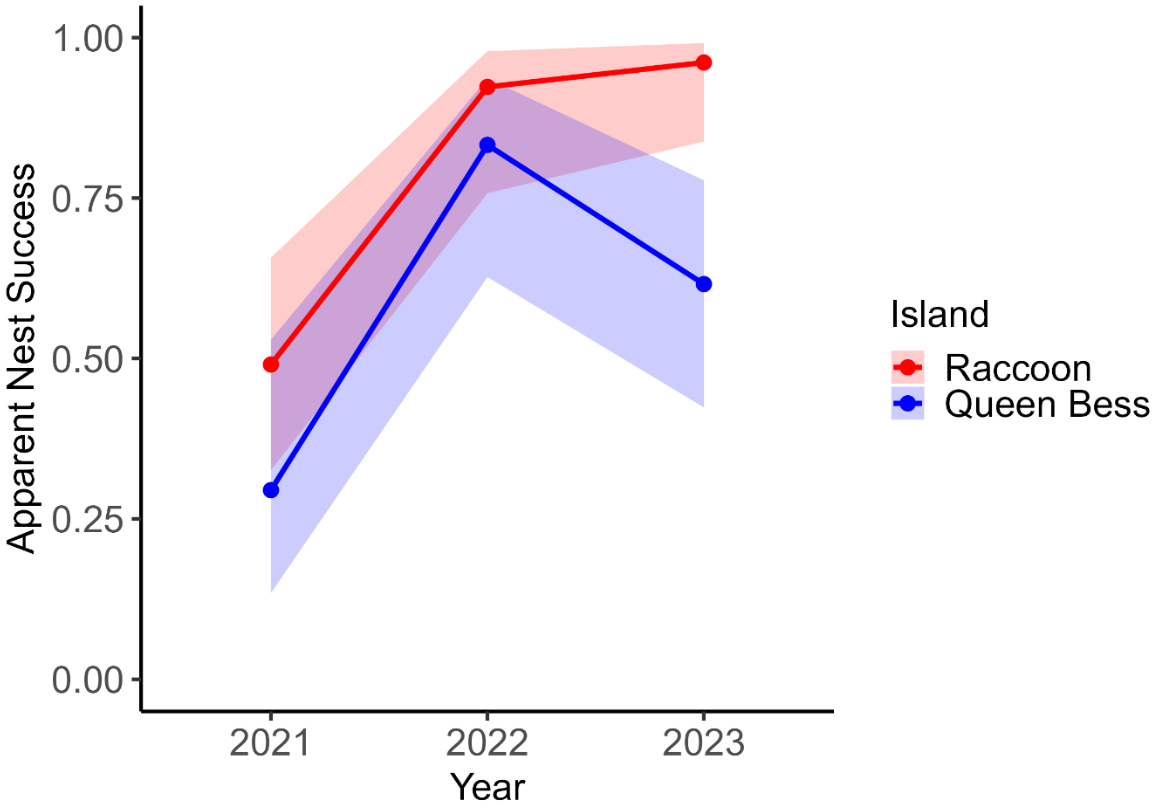


Figure 5. Apparent nest success (with 95% CI ribbons) of nesting Brown Pelicans (shrub and ground nests) on Raccoon and Queen Bess islands as a function of the interaction between island and year. Hurricane Ida affected the area after the 2021 data were collected.

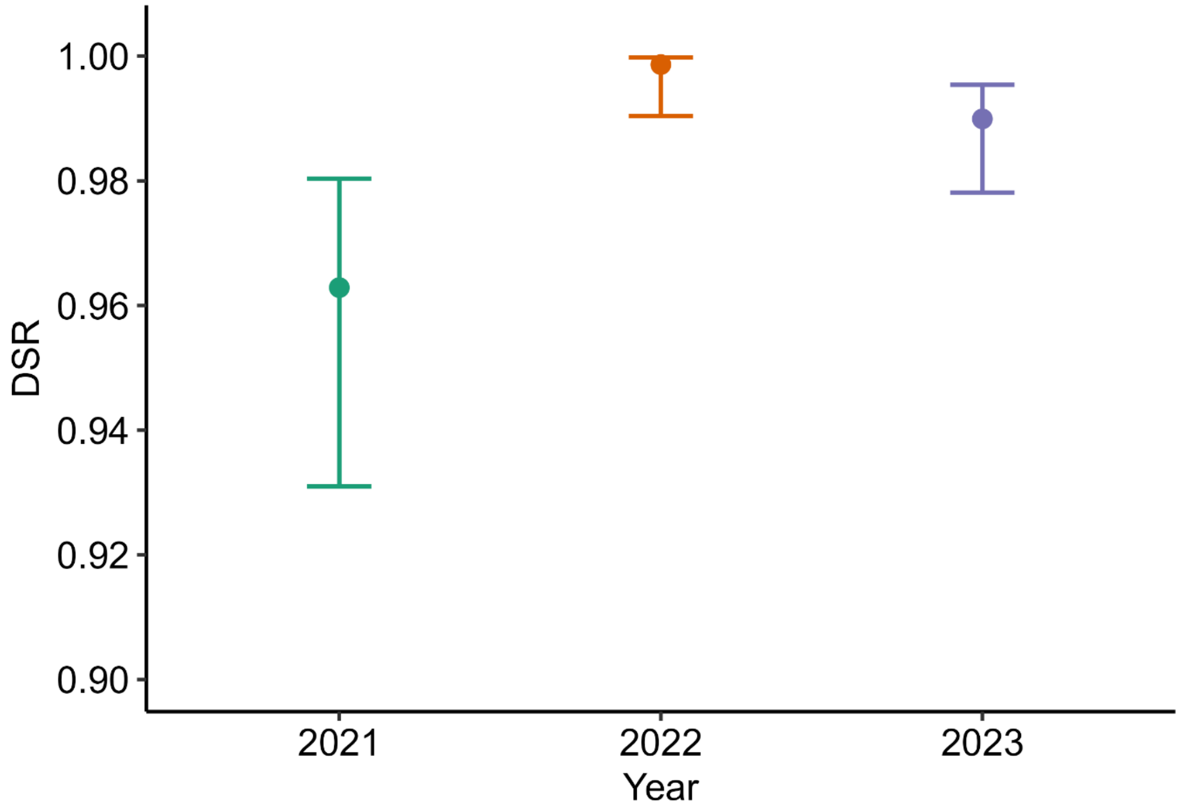


Figure 6. Brown Pelican predicted daily survival rate (DSR) (with 95% confidence intervals) of shrub nests as a function of year. Hurricane Ida affected the area after the 2021 data were collected.

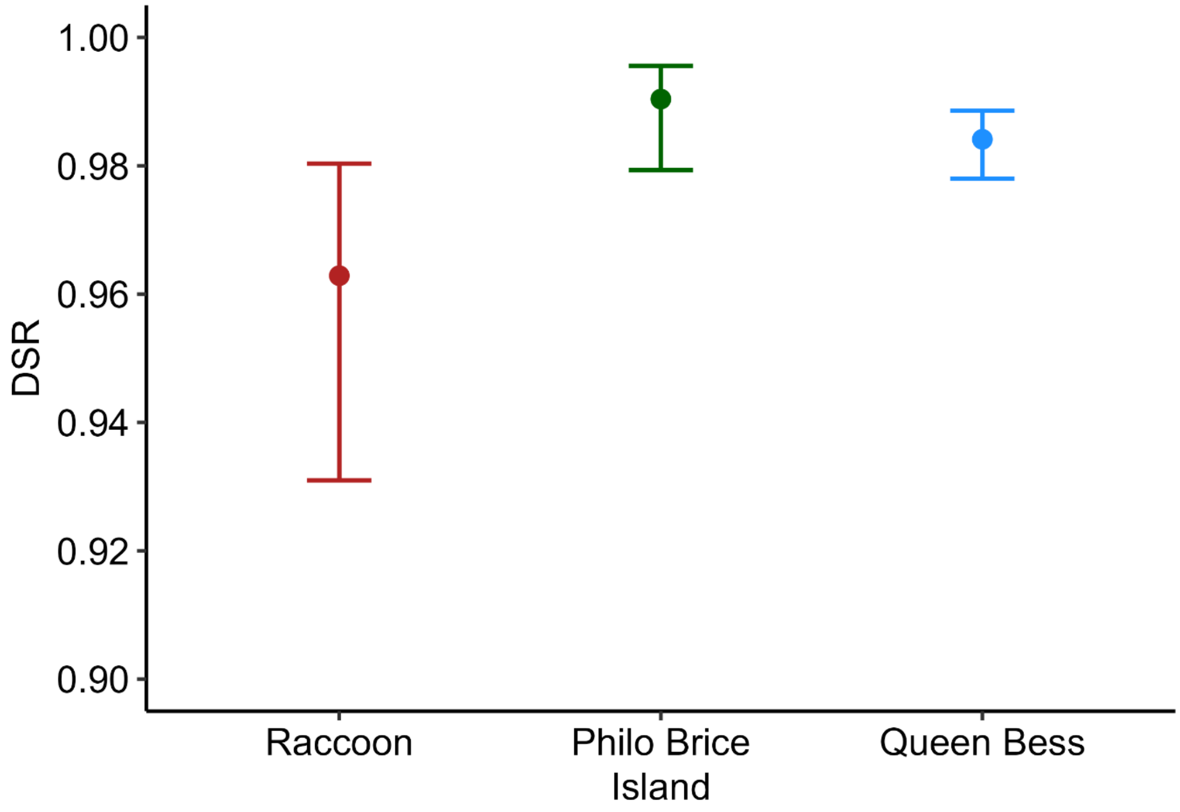


Figure 7. Brown Pelican daily survival rate (DSR) (with 95% confidence intervals) of shrub nests on Raccoon, Philo Brice, and Queen Bess Islands as a function of island.

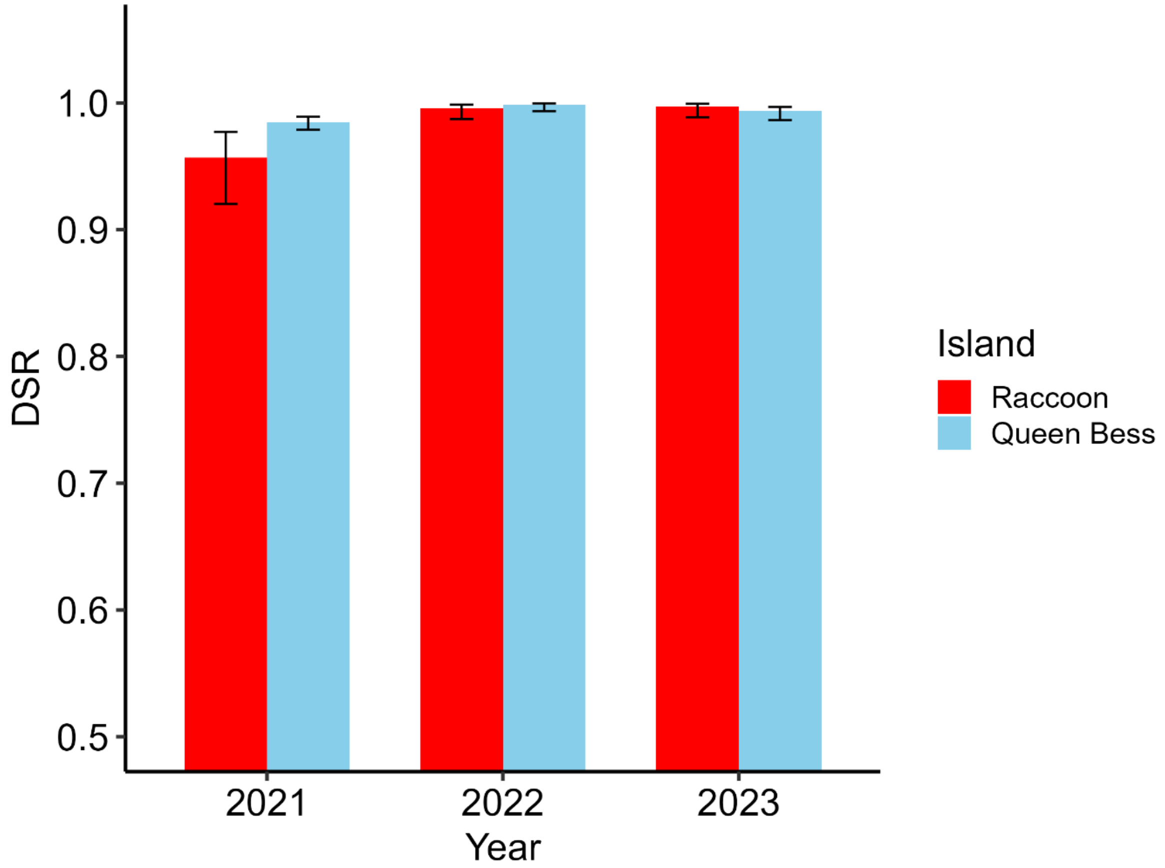


Figure 8. Brown Pelican predicted daily survival rate (DSR) (with 95% confidence intervals) on Raccoon and Queen Bess Islands (shrub and ground nests) as a function of the interaction between island and year. Hurricane Ida affected the area after the 2021 data were collected.

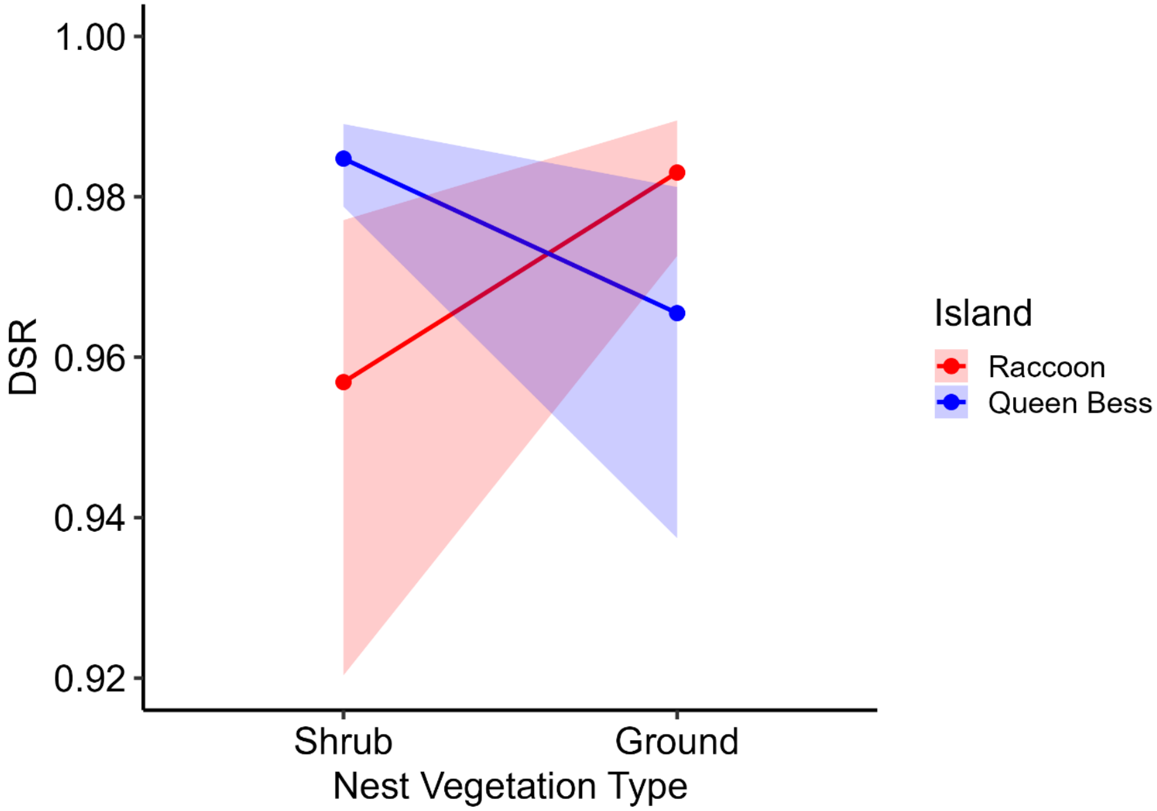


Figure 9. Brown Pelican predicted daily survival rate (DSR) (with 95% CI ribbons) on Raccoon and Queen Bess Islands (shrub and ground nests) as a function of the interaction between island and nest vegetation type.

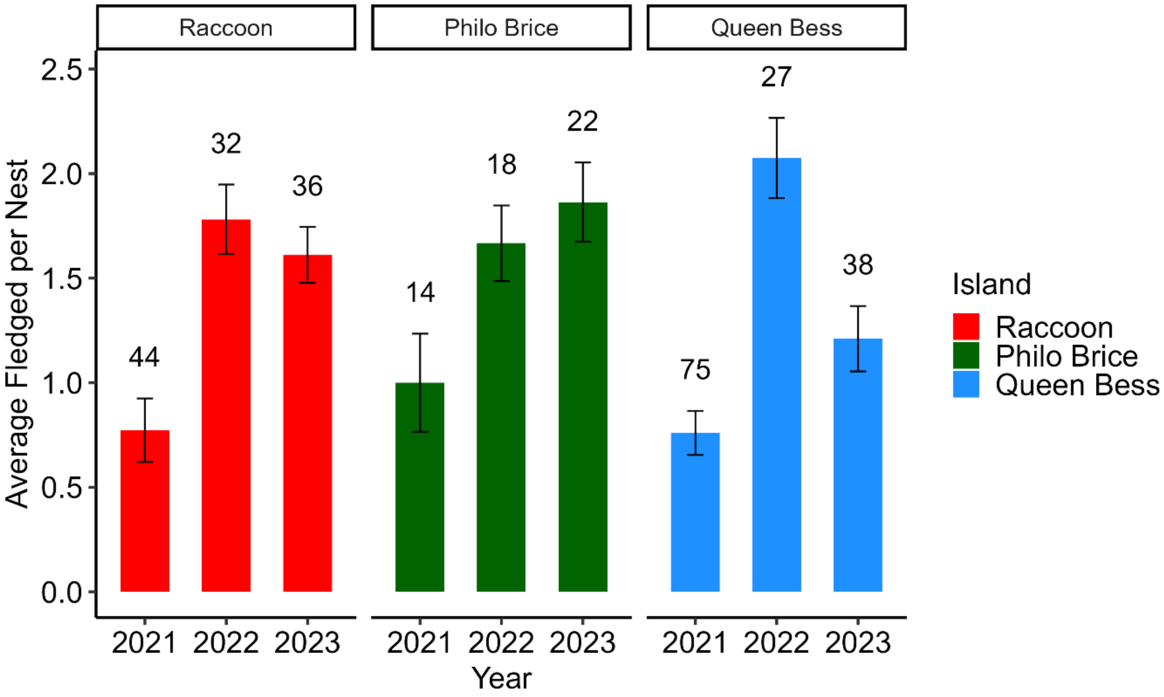


Figure 10. Average number of fledged chicks per nest from 2021-2023 on three islands (with 95% CI) with numbers above the bars indicating the number of nests included. Hurricane Ida affected the area after the 2021 data were collected.

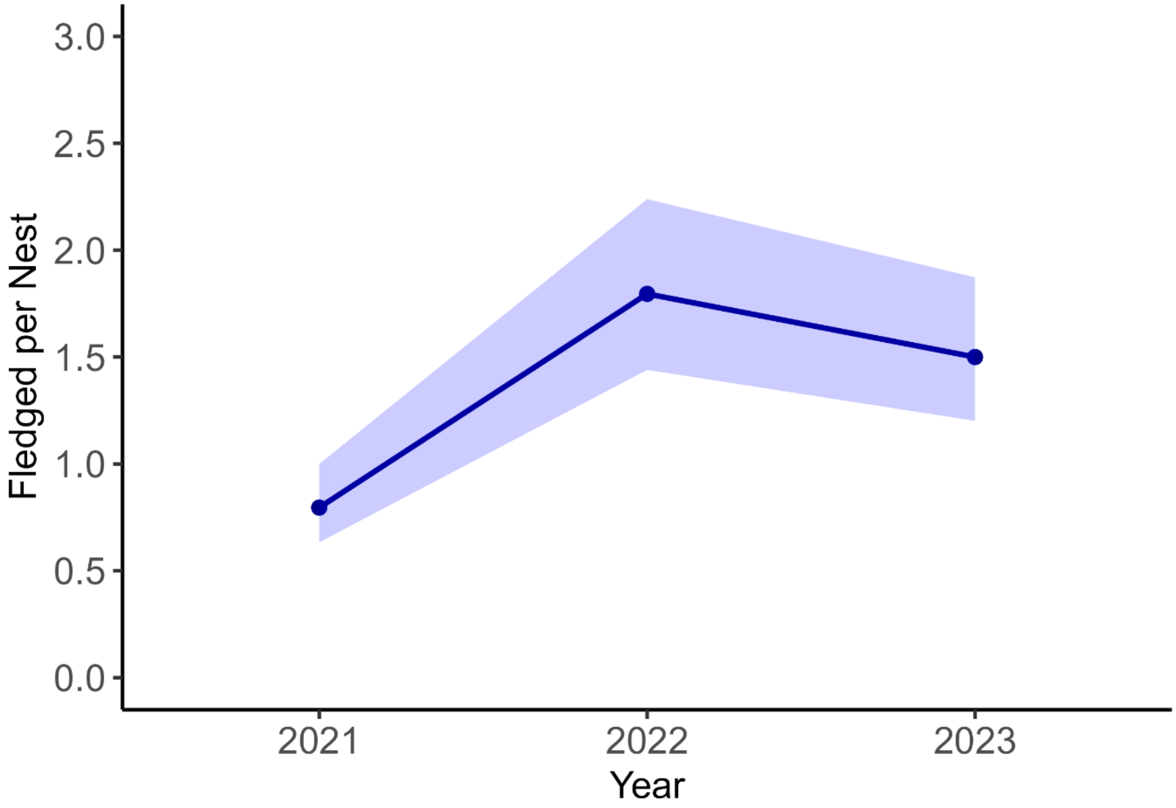


Figure 11. Annual number of fledglings per nest (with 95% CI ribbons) of Brown Pelicans located in shrubs from 2021-2023 on three islands. Hurricane Ida affected the area after the 2021 data were collected.

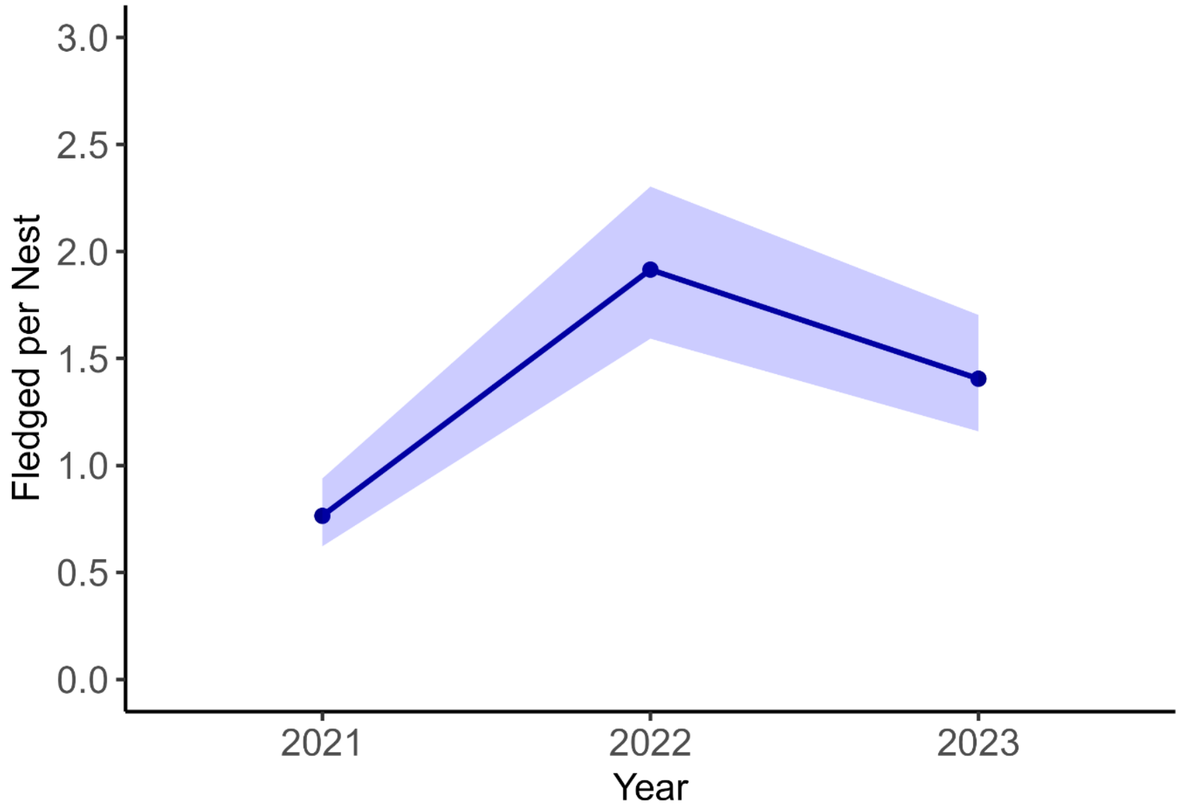


Figure 12. Annual variation in number of fledglings per nest (with 95% CI ribbons) of nests on Raccoon and Queen Bess (shrub and ground nests). Hurricane Ida affected the area after the 2021 data were collected.

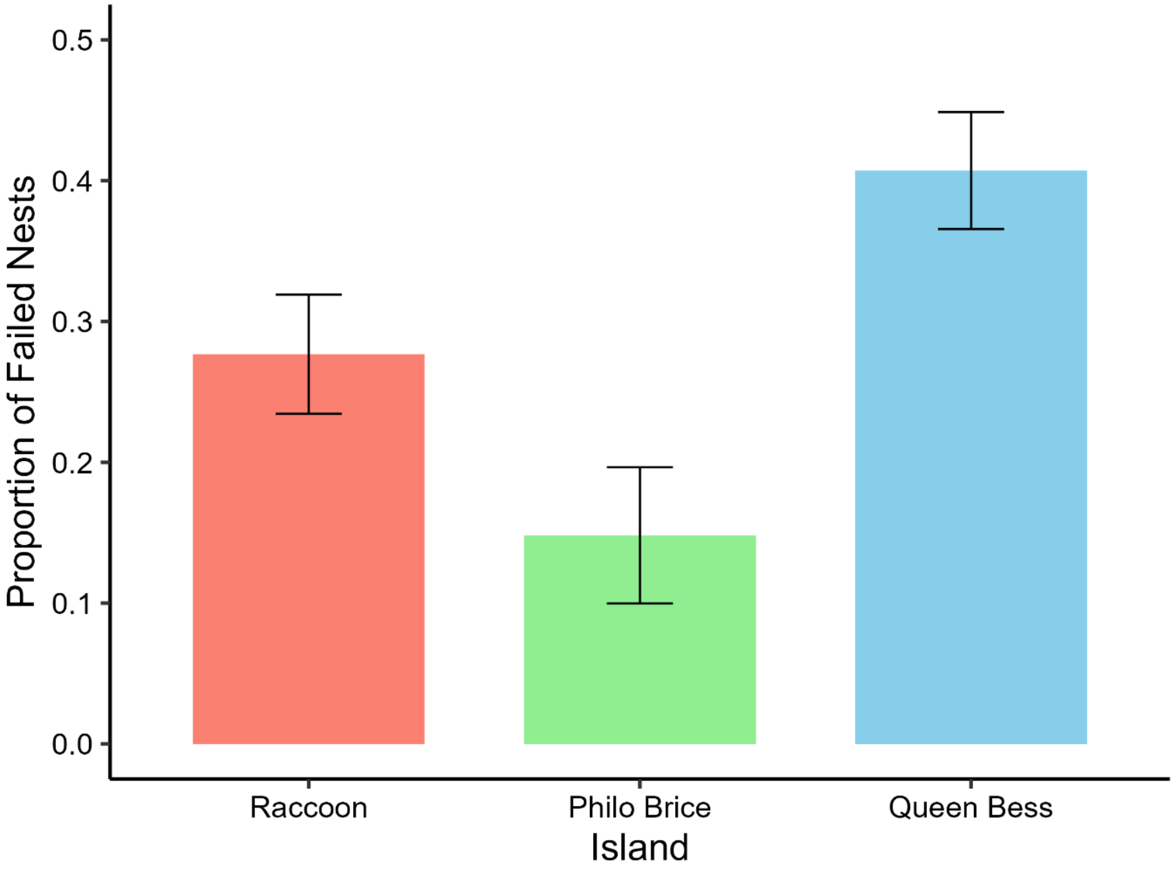


Figure 13. Proportion of failed nests on three islands with CI’s.

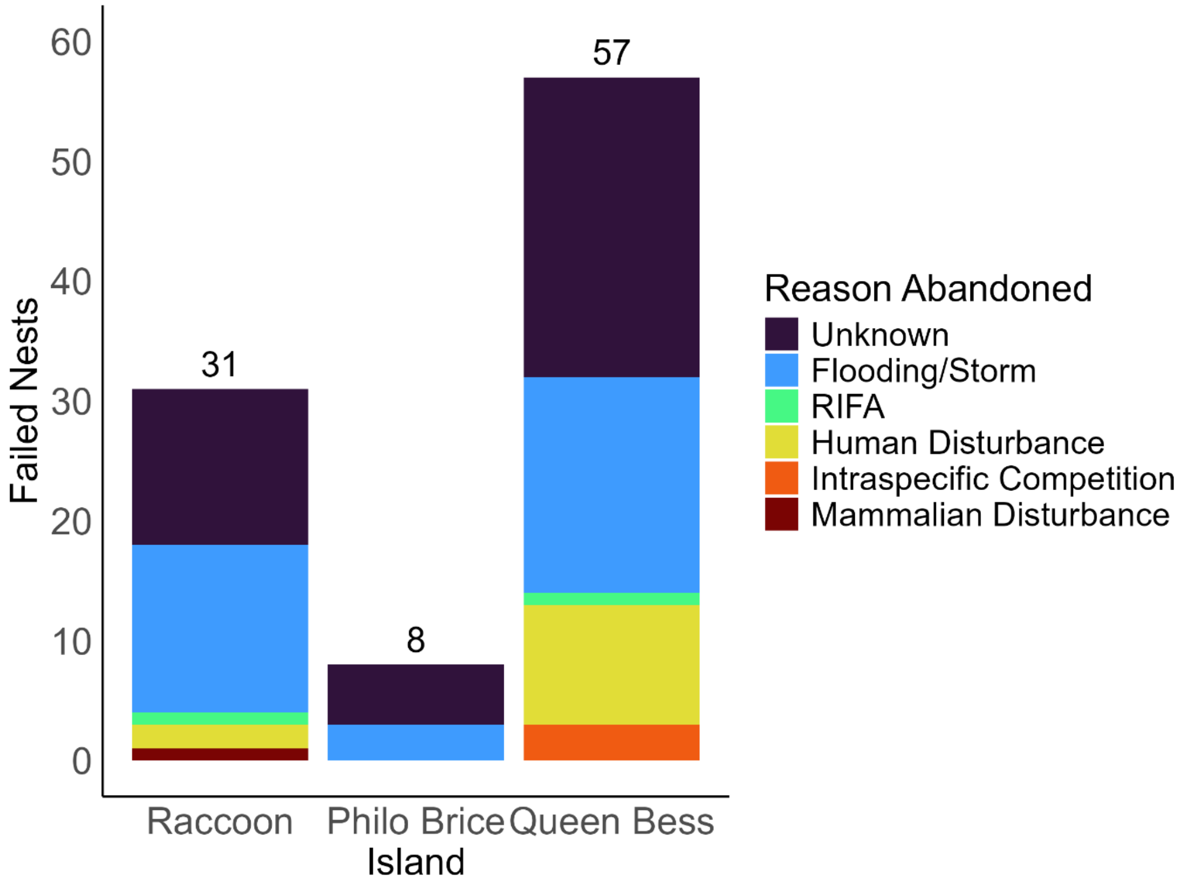


Figure 14. Total number of nest failures on Raccoon, Philo Brice, and Queen Bess islands categorized by reason for abandonment.

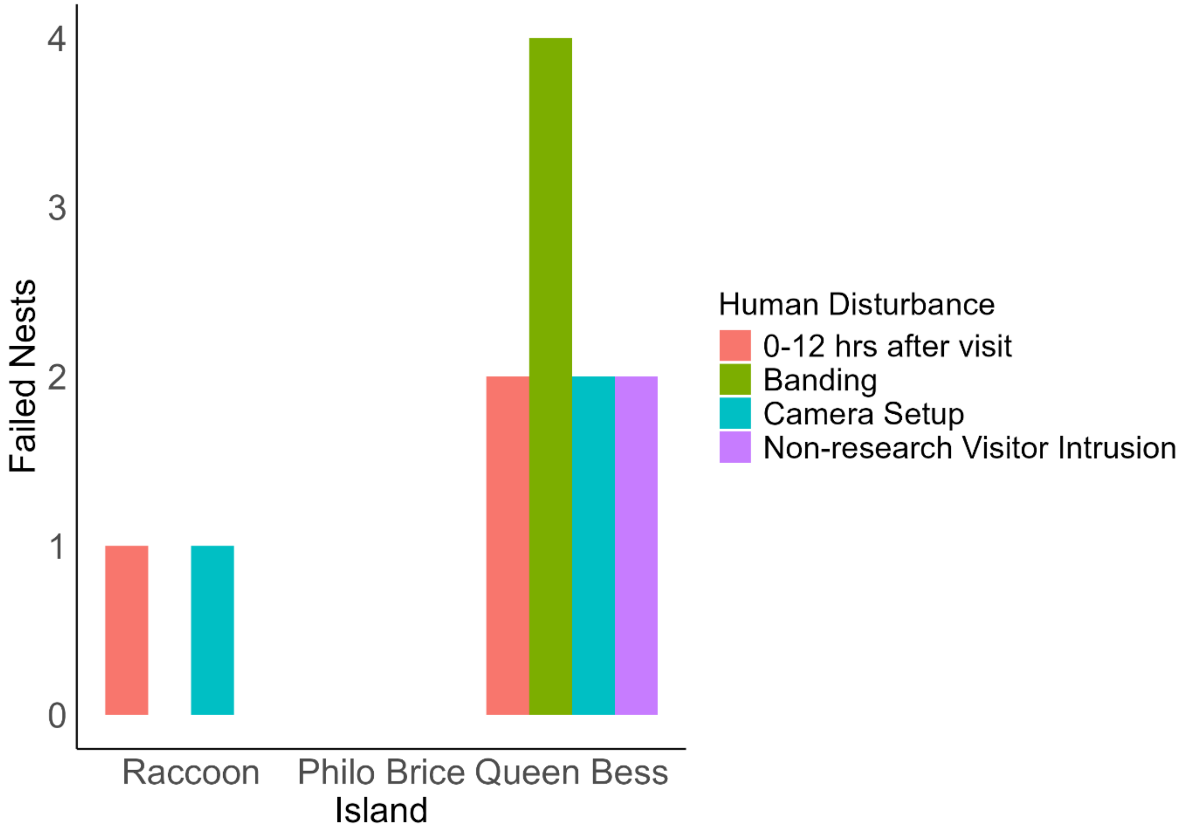


Figure 15. Number of failed nests on Raccoon, Philo Brice, and Queen Bess islands attributed to human disturbances.

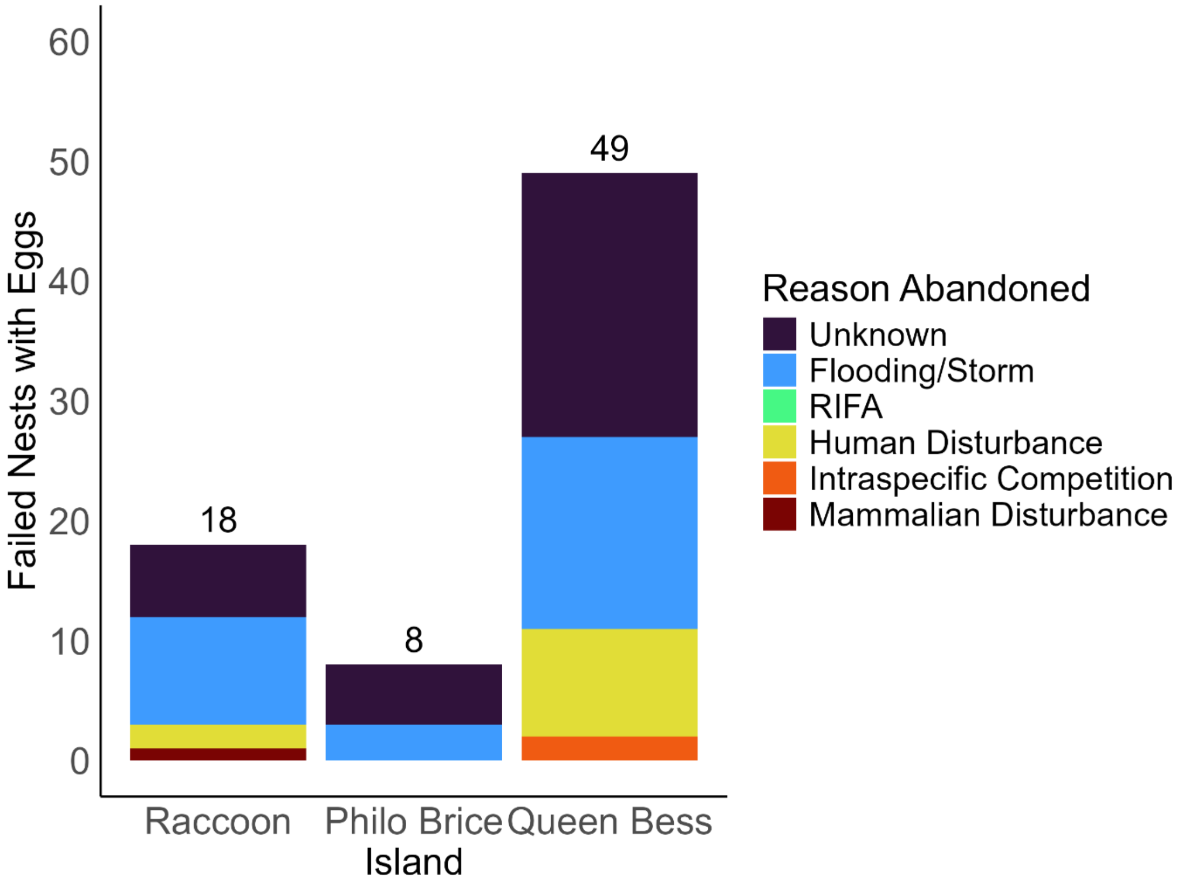


Figure 16. Total number of failed nests in stage 1 (containing at least 1 egg, but no chicks) on Raccoon, Philo Brice, and Queen Bess Islands categorized by reason for abandonment.

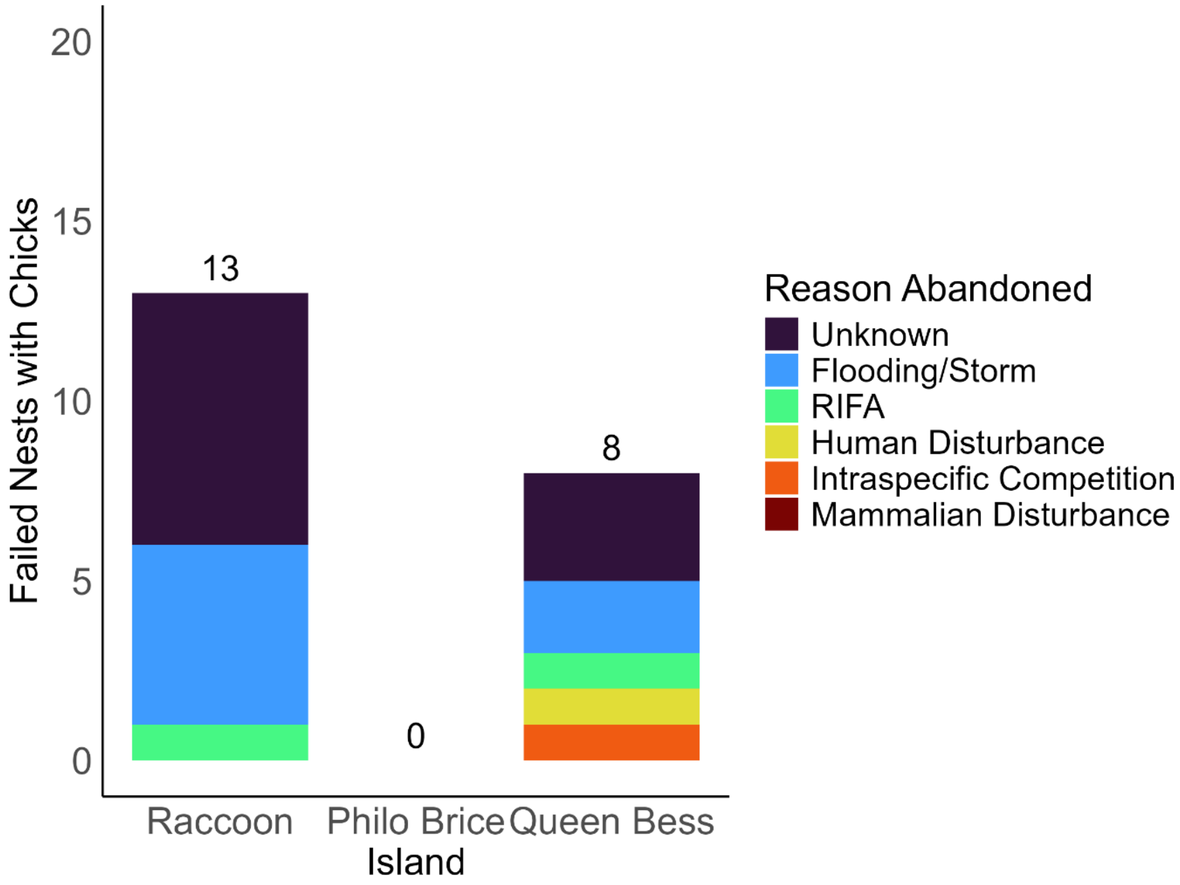


Figure 17. Total number of failed nests in nest stage 2 (at least 1 chick present) on Raccoon, Philo Brice, and Queen Bess Islands categorized by reason for abandonment.

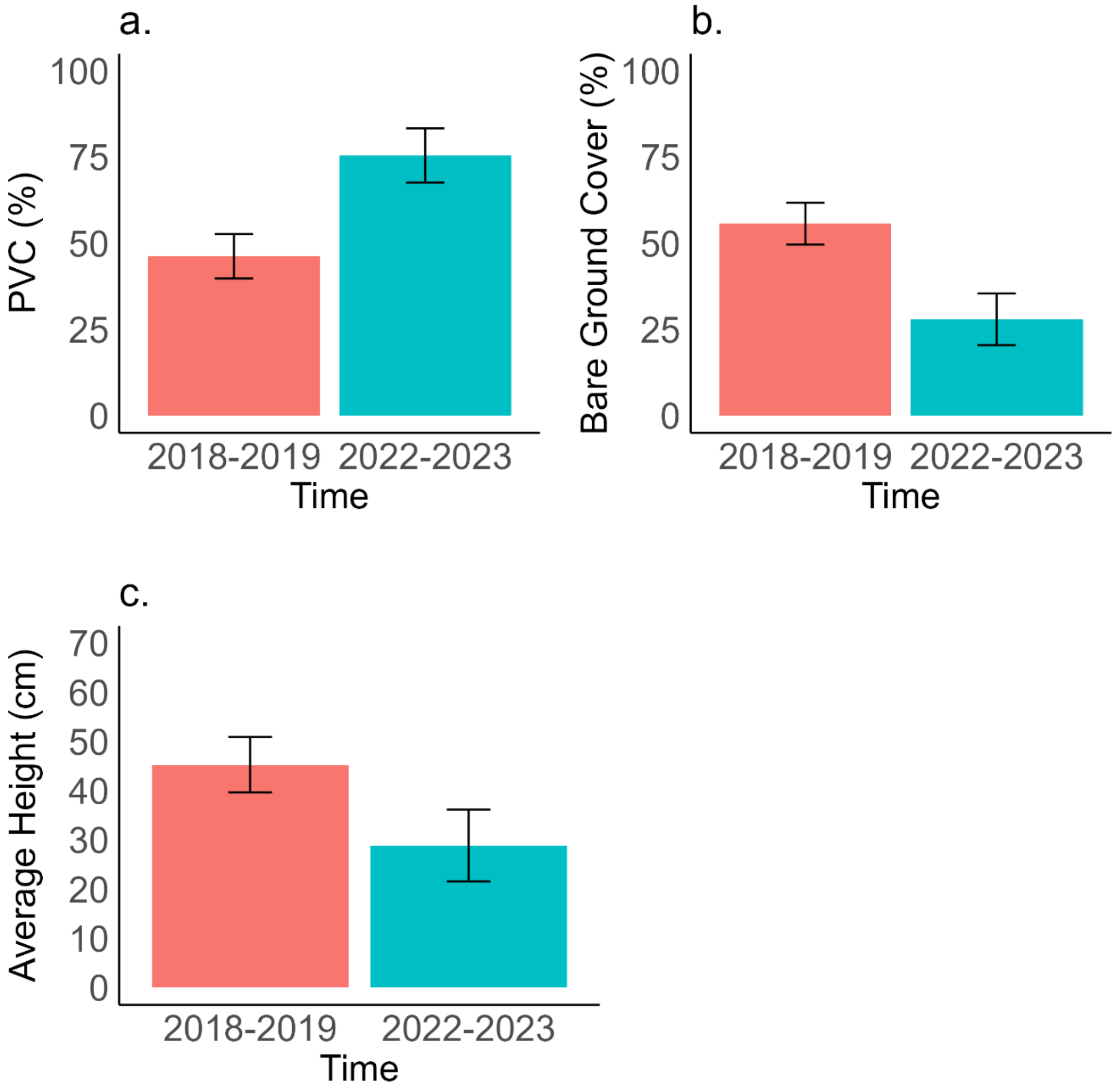


Figure 18. Model-estimated means with standard errors of a) percent vegetation cover (PVC) (*p* = 0.0003) b) bare ground cover (%) (*p* < 0.0001) and c) average vegetation height (cm) (*p* = 0.0435) across transect points before (2018-2019) and after (2022-2023) Hurricane Ida (2021) on Raccoon Island.

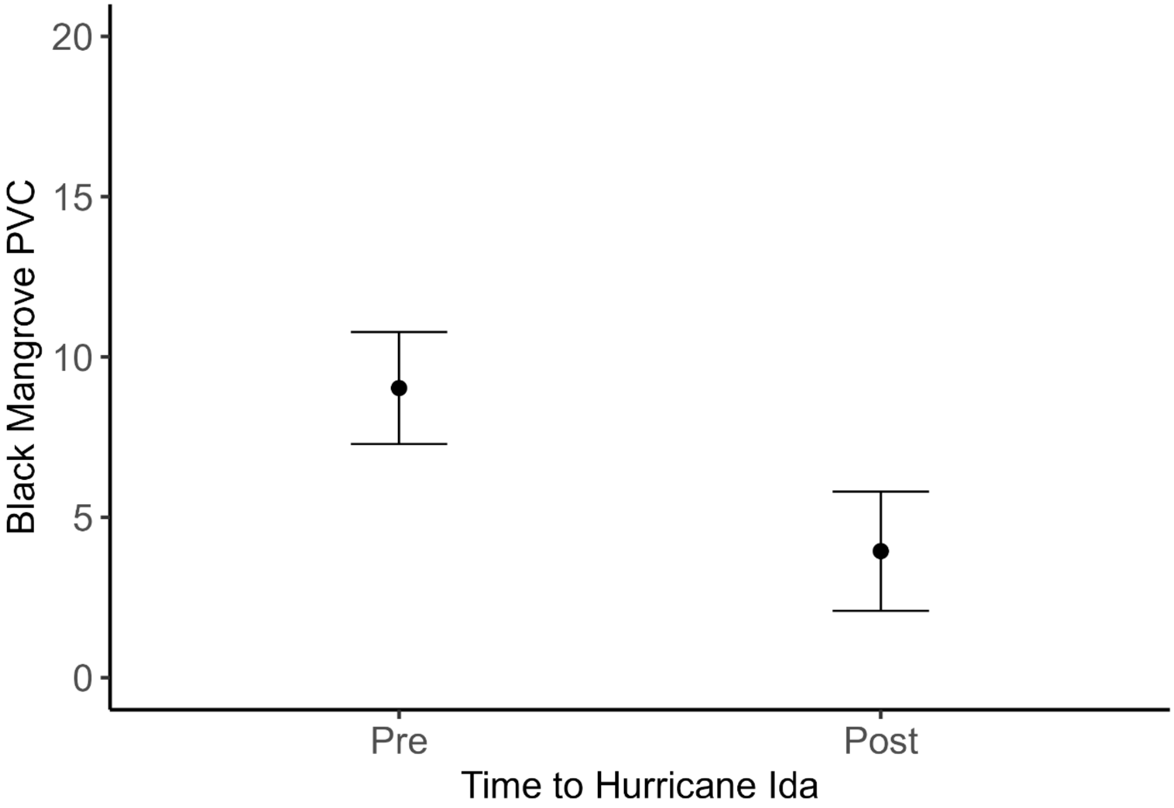


Figure 19. Model-estimated means with standard error of black mangrove percent vegetation cover (PVC) in unrestored marsh (*p* = 0.0015) on Raccoon island pre- (2018-2019) and post-storm (2022-2023).

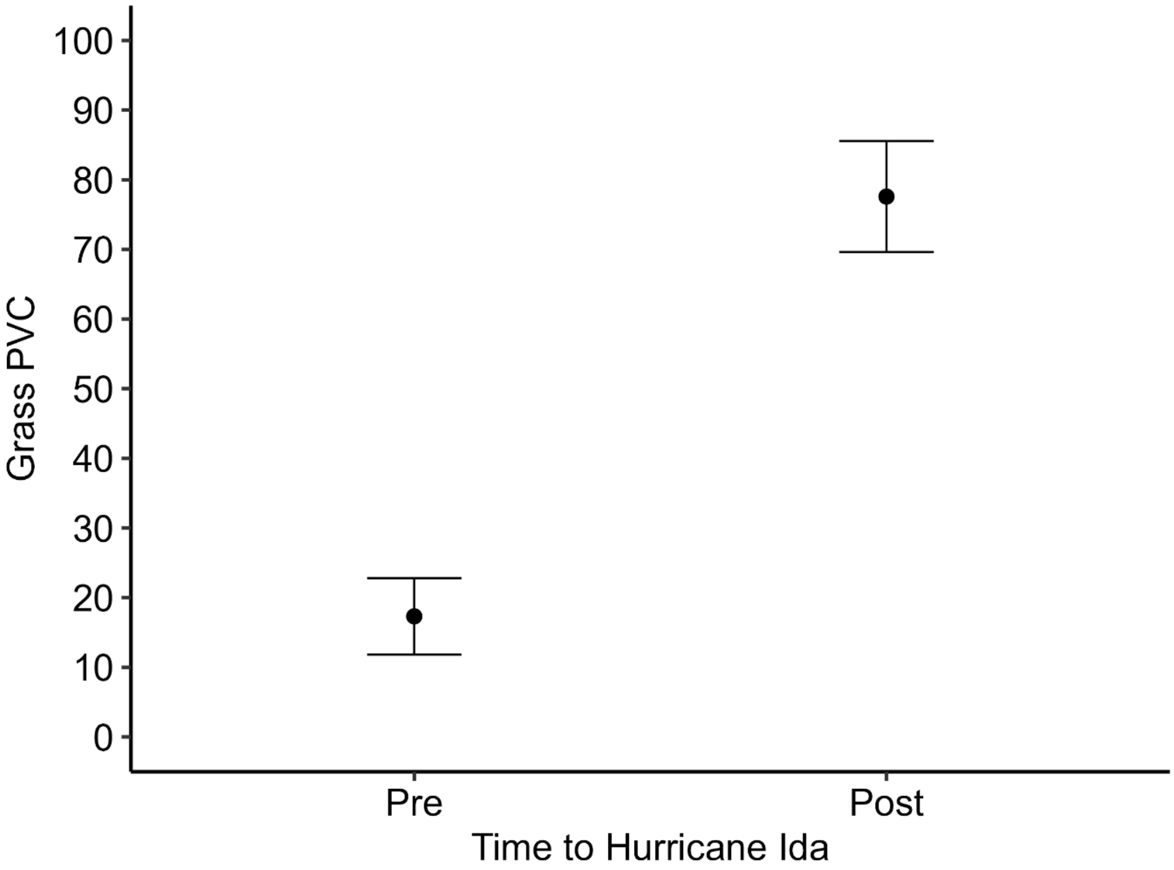


Figure 20. Model-estimated means with standard error of grass percent vegetation cover (PVC) in restored dune on Raccoon island pre-storm (2018-2019) and post-storm (2022-2023) (*p* < 0.0001).

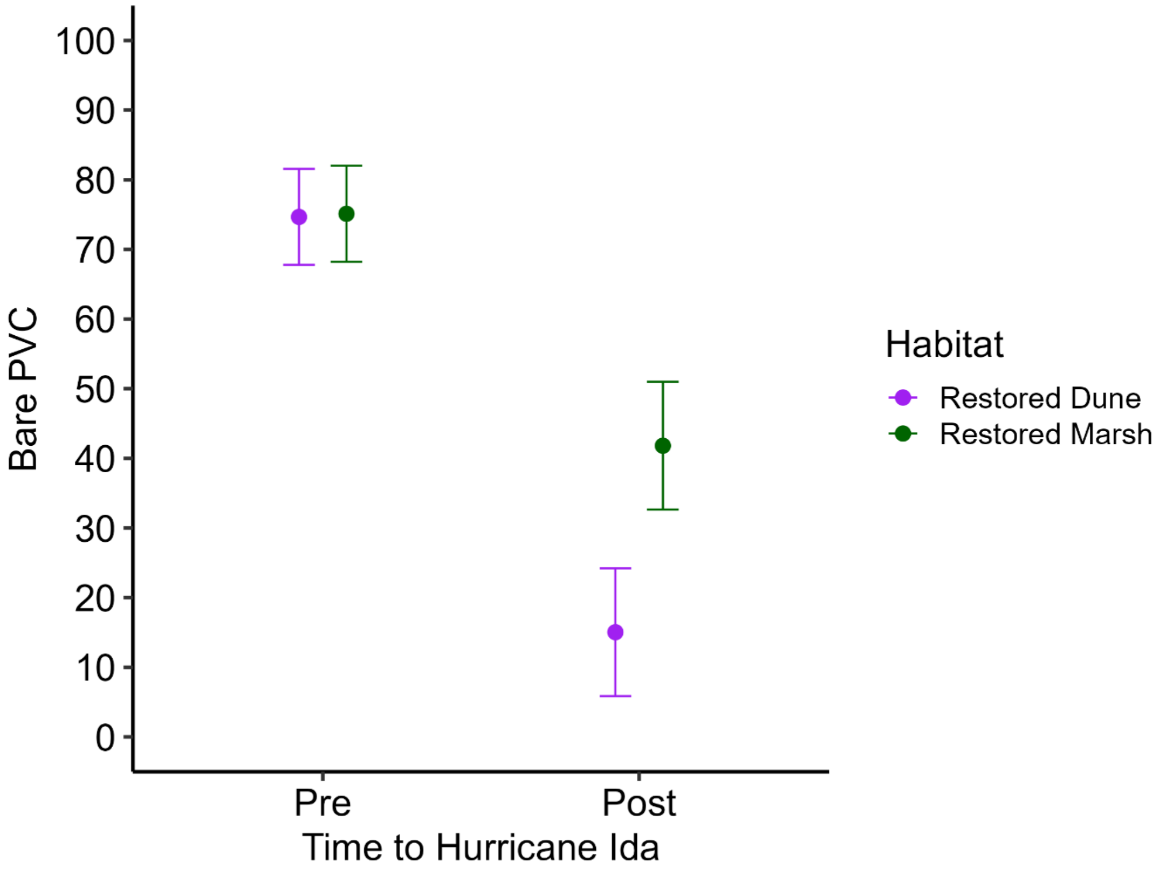


Figure 21. Model-estimated means with standard error of bare ground percent vegetation cover (PVC) in restored dune (*p* < 0.0001) and restored marsh (*p* = 0.0026) on Raccoon island pre-storm (2018-2019) and post-storm (2022-2023).

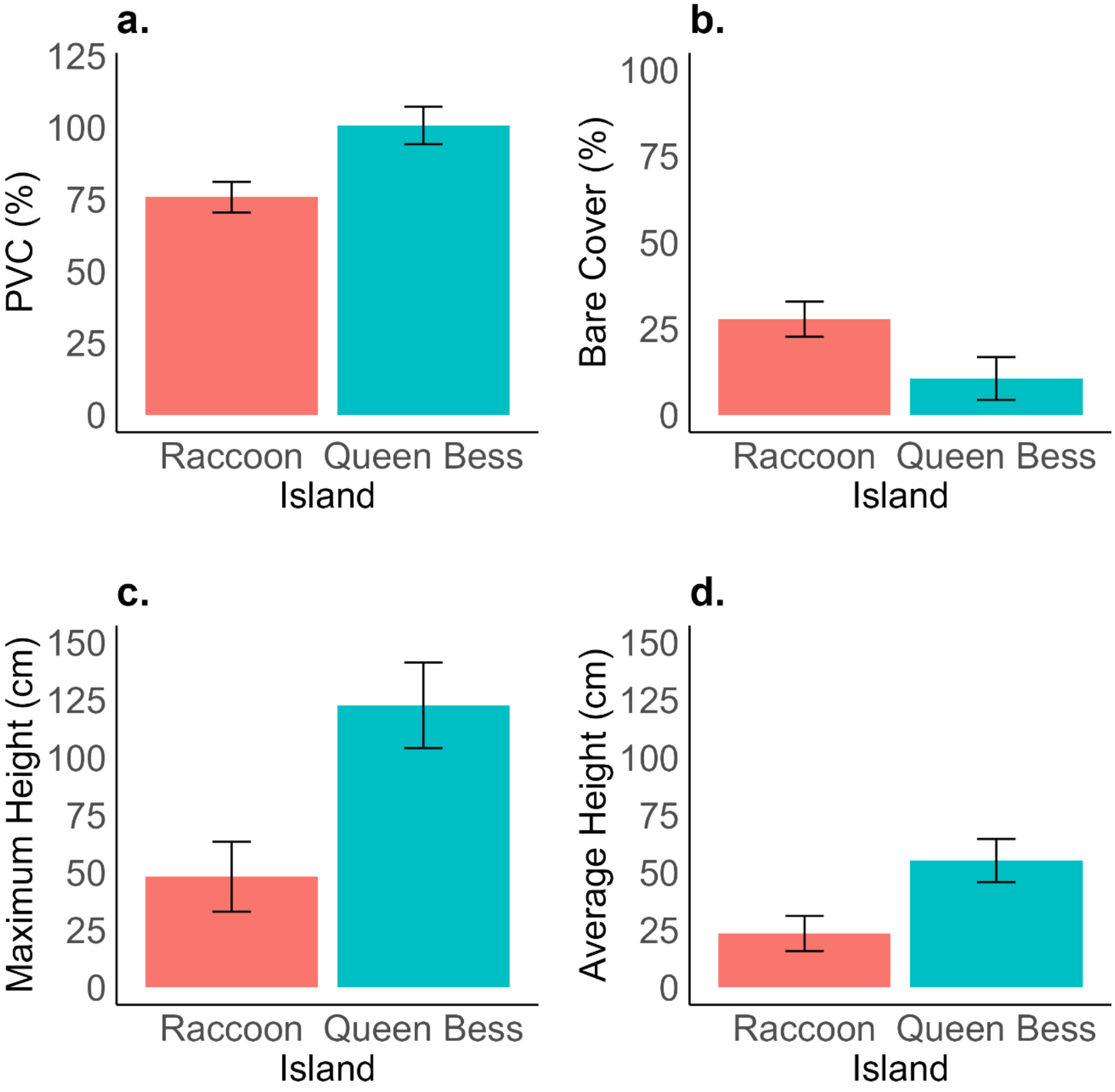


Figure 22. Model-estimated means with standard errors of a) percent vegetation cover (PVC) ($F_{1,28}$ = 8.822, *p* = 0.0061), b) bare ground cover ($F_{1,28}$ = 4.587, *p* = 0.0411), c) maximum vegetation height (cm) ($F_{1,28}$ = 9.626, *p* = 0.0044), and d) average vegetation height (cm) ($F_{1,28}$ = 6.874, *p* = 0.0140) on Raccoon and Queen Bess in 2022, the year following Hurricane Ida.

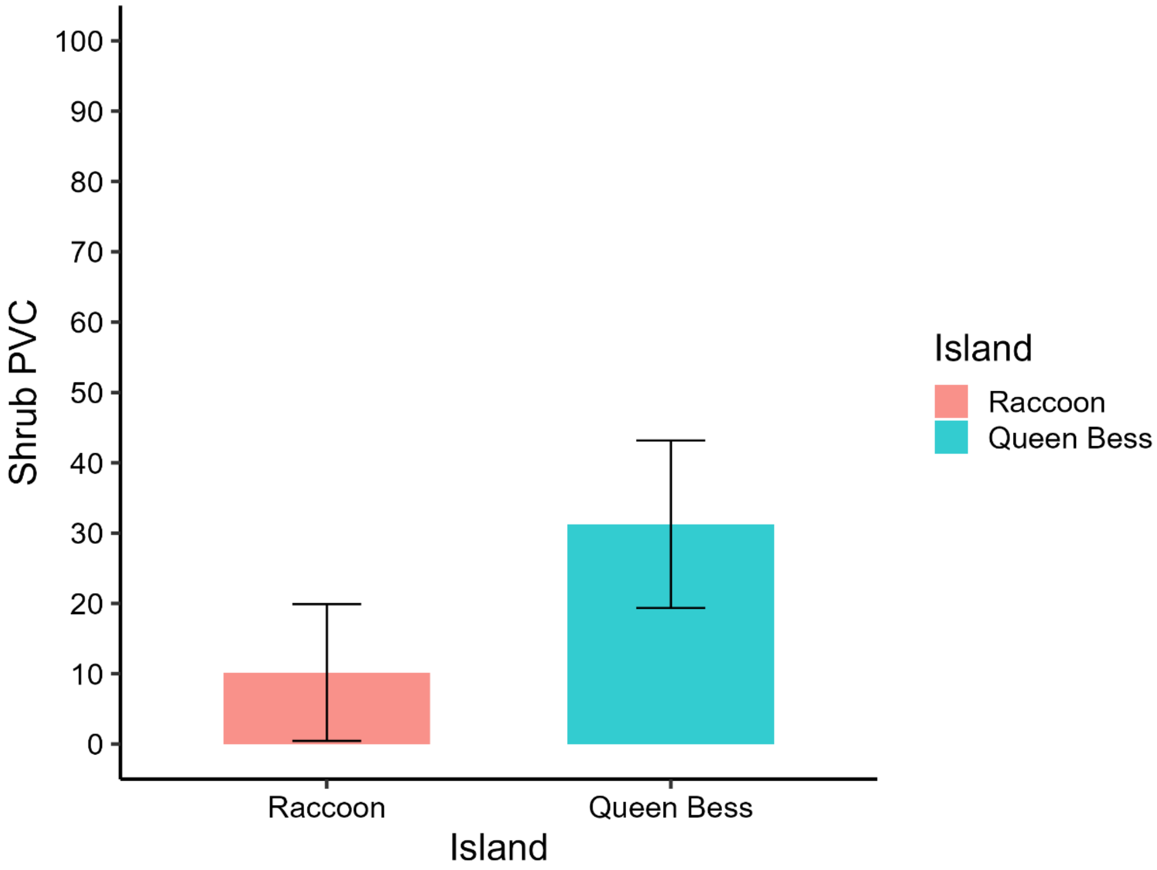


Figure 23. Model-estimated means with standard errors of percent vegetation cover (PVC) of shrubs on Raccoon and Queen Bess in 2022, the year following Hurricane Ida (*p* = 0.0089).

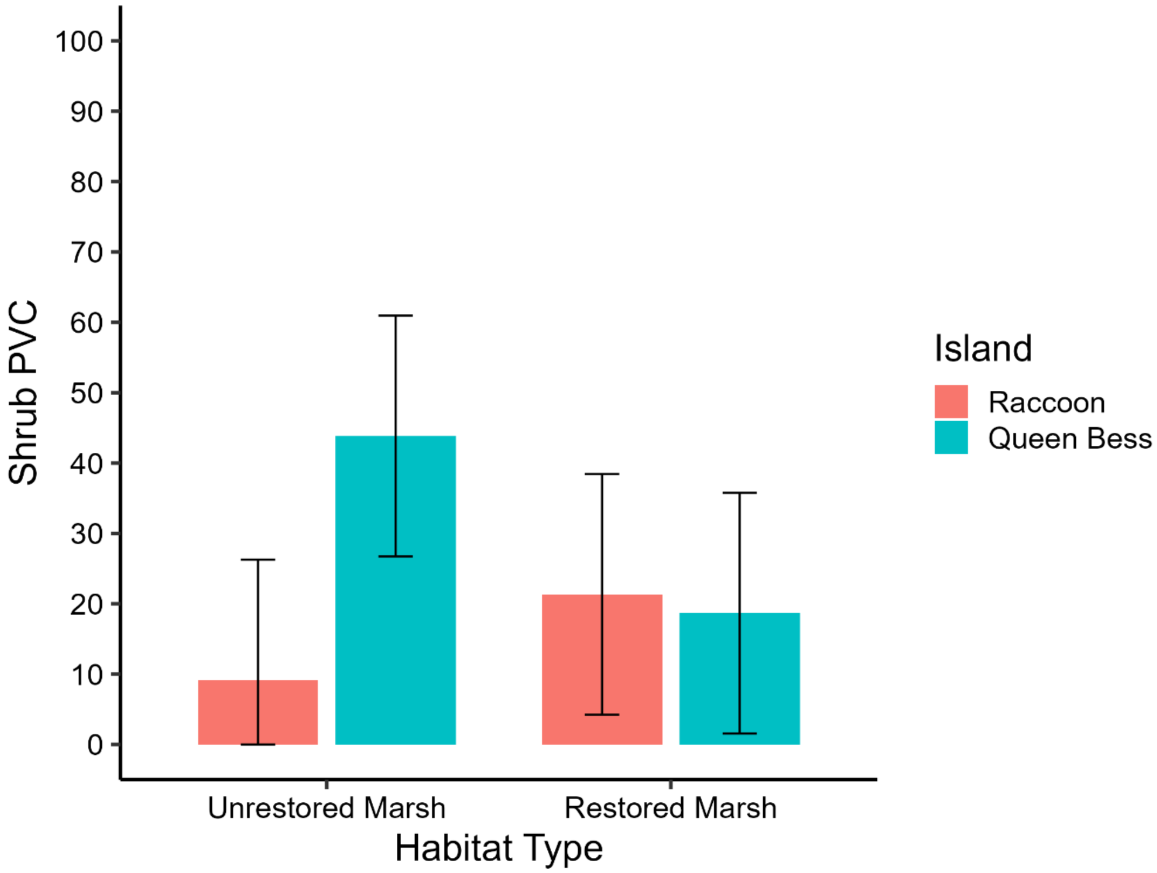


Figure 24. Model-estimated means with standard errors of percent vegetation cover (PVC) of shrubs in marsh habitats on Raccoon and Queen Bess 2022, the year following Hurricane Ida (*p* = 0.0340).

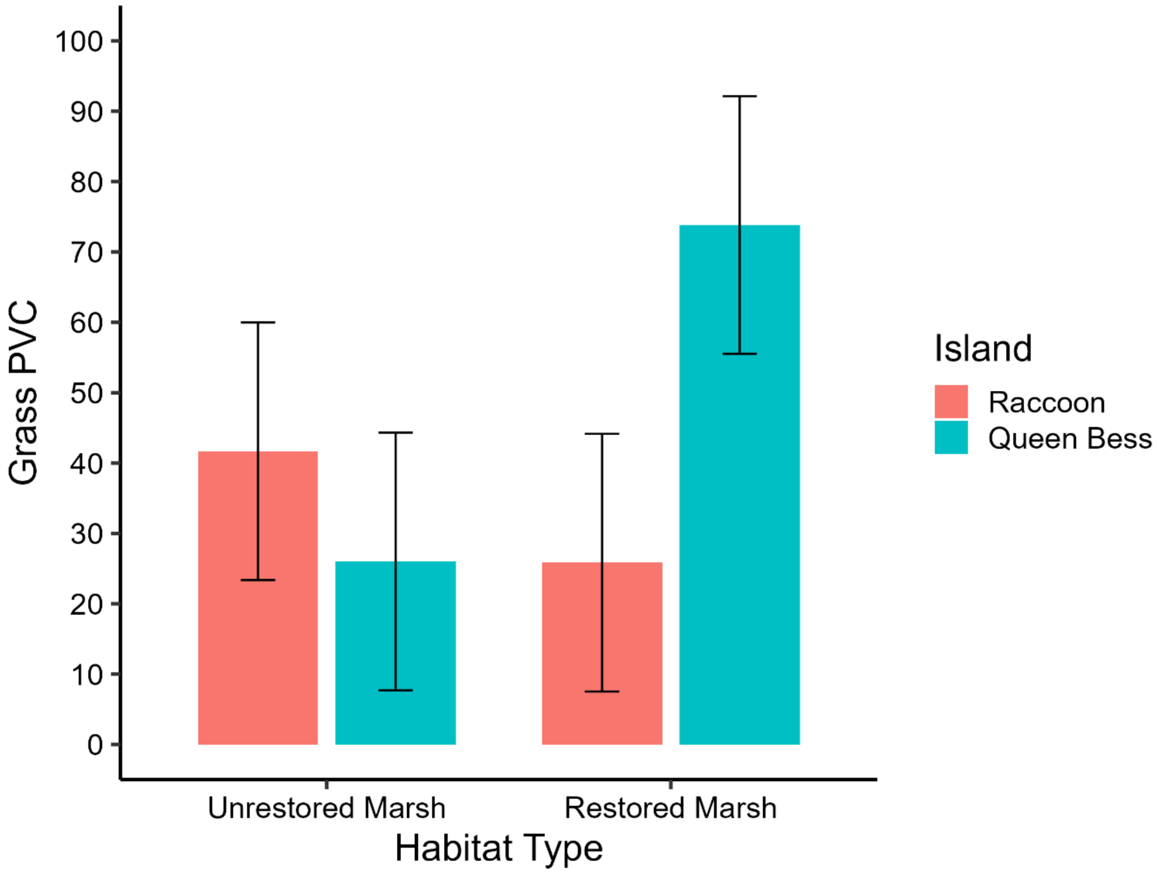


Figure 25. Model-estimated means with standard errors of percent vegetation cover (PVC) of grass in marsh habitats on Raccoon and Queen Bess 2022, the year following Hurricane Ida (*p* = 0.0017).

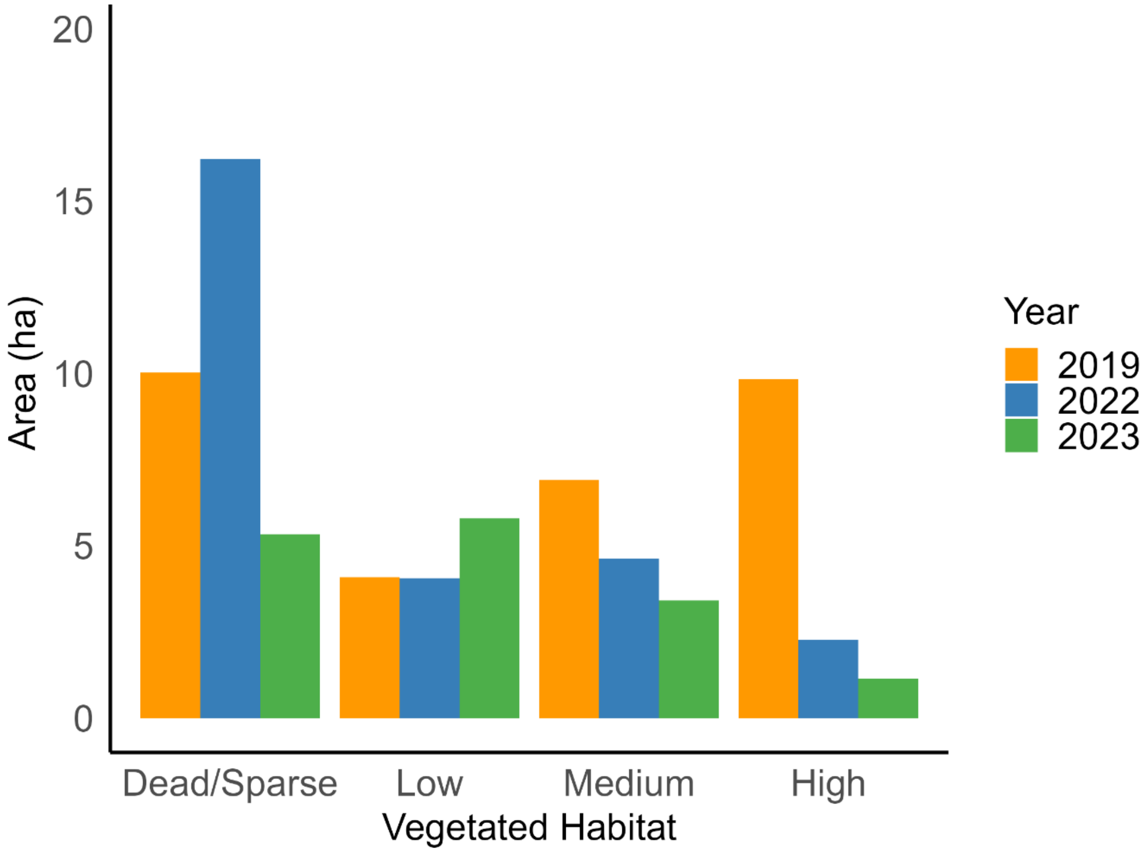


Figure 26. Area (ha) calculated from imagery cell size and count for vegetated habitats on Raccoon in 2019, 2022 and 2023. Hurricane Ida occurred in 2021.

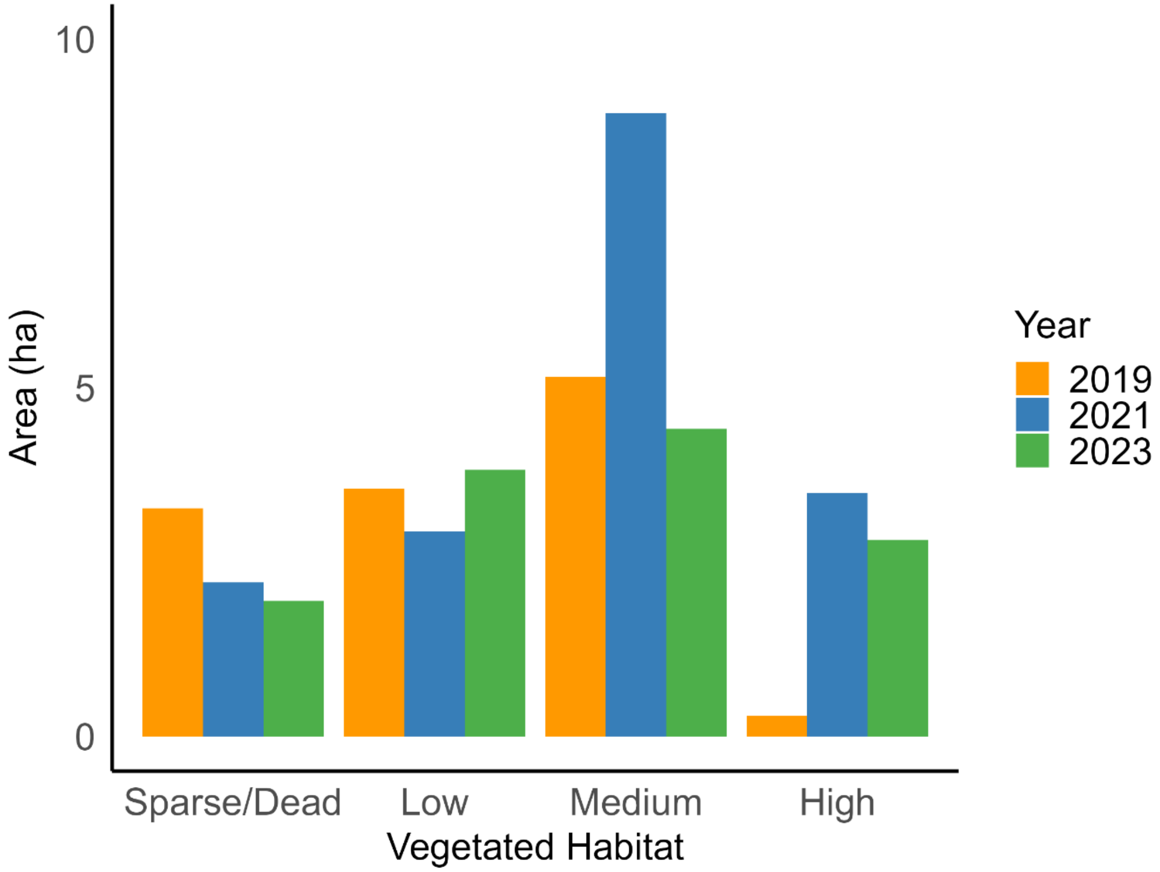


Figure 27. Area (ha) calculated from imagery cell size and count for vegetated habitats on Philo Brice in 2019, 2021, and 2023. Hurricane Ida occurred in 2021 before the imagery used for the 2021 habitat classification.

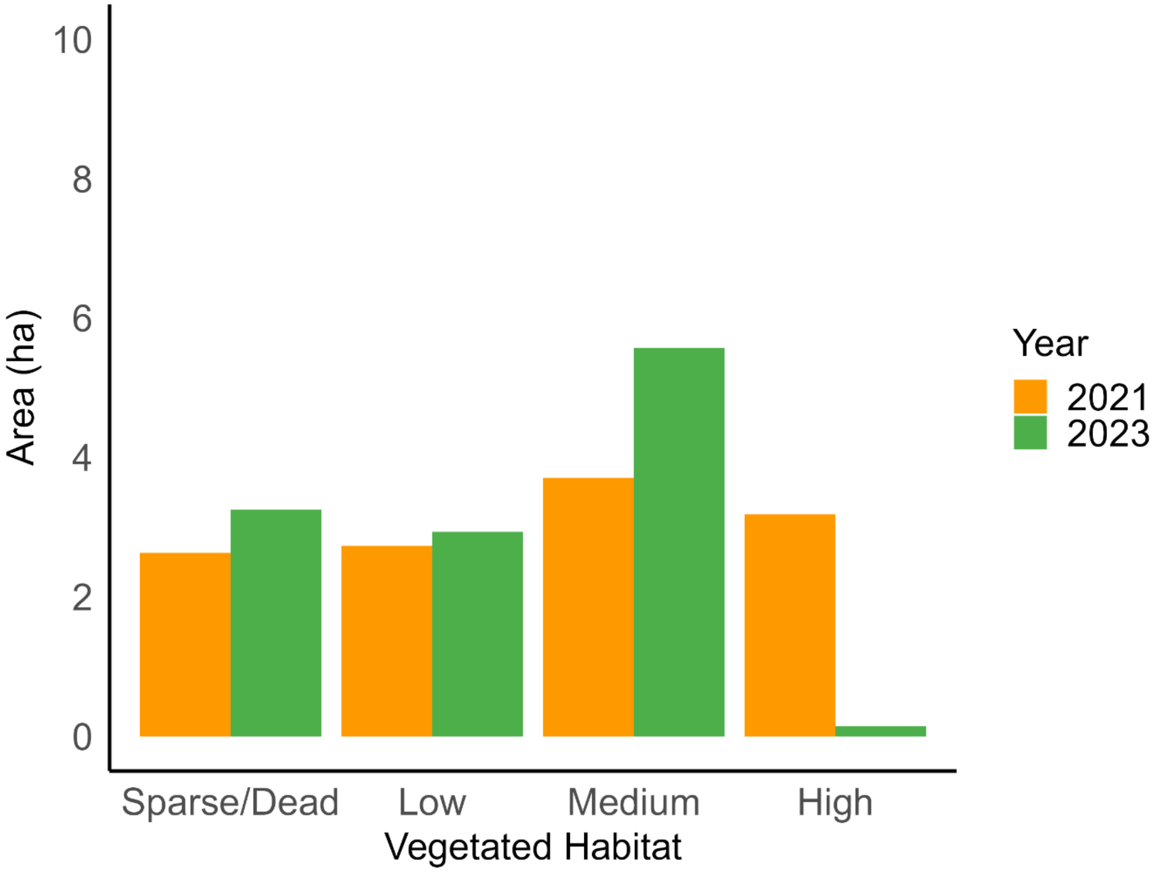


Figure 28. Area (ha) calculated from imagery cell size and count for vegetated habitats on Queen Bess in 2021 and 2023. Hurricane Ida occurred in 2021 after the imagery used for the 2021 habitat classification.

Appendix. Supplementary Information

Suppl. Table 1. Tropical cyclones (i.e., hurricanes and tropical storms) within 200 km buffer of study area polygon in southeast Louisiana from 2013-2023 (National Oceanic and Atmospheric Administration 2024).

Cyclone name	Year	Dates	Saffir-Simpson hurricane wind scale category at landfall
Tropical Storm Karen	2013	October 3-6	TS
Hurricane Nate	2017	October 3-11	1
Tropical Storm Gordon	2018	September 2-7	TS
Hurricane Barry	2019	July 10-16	1
Tropical Storm Cristobal	2020	June 1-12	TS
Hurricane Marco	2020	August 20-25	1
Hurricane Sally	2020	September 11-18	2
Hurricane Zeta	2020	October 24-30	3
Tropical Storm Claudette	2021	June 17-23	TS
Hurricane Ida	2021	August 26-September 4	4

Suppl. Table 2. Summary of imagery sources reviewed for availability at study site locations during and prior to the study period 2018-2023 evaluated based on spatial and temporal coverage, cloud cover, and accessibility.

Source	Availability	Limitations
Airbus	Available for purchase	Cost prohibitive
DOQQ, EarthData, USGS	Unavailable	No imagery found
ERMA, Digital Coast, NOAA	Limited	Coverage not suitable
Esri World Imagery Wayback	Limited	Resolution limited, no NIR band
GeoEye-1, Maxar, European Space Agency	Restricted	Only available to EU countries and Canada
Google Earth Pro	Limited	Resolution, coverage, and spectral bands not suitable
Hurricane Ida Imagery, Remote Sensing Division, NOAA	Unavailable	No imagery found
Landsat, EarthExplorer, USGS	Limited	Limited coverage, cloud cover, resolution too coarse
MODIS, Earth Scan Laboratory, LSU	Available	High cloud cover, resolution too coarse
NGS National Shoreline, NOAA	Unavailable	Not applicable for study timeframe, shorelines not accurate
Sentinel-2	Limited	Limited coverage
World View, EarthData, NASA	Available	Resolution and coverage not suitable

Suppl. Table 3. Pre- and post-Hurricane Ida island area (ha) on three islands analyzed from available imagery sources from 2017-2023

Island	Area (ha)	Year	Acquisition Date	Source
Raccoon	60.23	2018	April 23, 2018	Esri World Imagery Wayback
	61.66	2019	August 30, 2019	Sentinel-2
	54.06	2020	October 28, 2020	Esri World Imagery Wayback
	54.72	2021	July 24, 2021	Sentinel-2
	77.32	2022	January 3, 2022	Sentinel-2
	51.02	2023	April 30, 2023	Sentinel-2
Philo Brice	21.60	2018	April 23, 2018	Esri World Imagery Wayback
	20.45	2019	August 30, 2019	Sentinel-2
	17.38	2020	September 20, 2020	Esri World Imagery Wayback
	15.52	2021	September 10, 2021	Google Earth Pro
	16.97	2022	February 12, 2022	Google Earth Pro
	14.51	2023	April 30, 2023	Sentinel-2
	13.65	2023	July 27, 2023	Google Earth Pro
Queen Bess	7.54	2017	November 22, 2017	Esri World Imagery Wayback
	7.82	2019	August 30, 2019	NAIP
	14.29	2020	September 23, 2020	Sentinel-2
	15.07	2021	August 20, 2021	Sentinel-2
	14.44	2022	January 5, 2022	Esri World Imagery Wayback
	14.26	2023	April 30, 2023	Esri World Imagery Wayback

Suppl. Table 4. Manually assigned threshold-based land cover classification on three islands based on NDVI values of NAIP (0.3-0.6 m) and Sentinel-2 (10m) imagery before and after Hurricane Ida (August 29, 2021).

Island	Source	Imagery Acquisition Date(s)	Vegetation Class	NDVI Range
Raccoon	NAIP	August 30, 2019	Water	-0.974 - -0.2
			Sand	-0.199 - -0.08
			Dead/Sparse	-0.079 - 0
			Low	0.001 - 0.1
			Medium	0.101 - 0.5
			High	0.501 - 1.0
Raccoon	NAIP	January 3, 2022	Water	-0.768 - -0.28
			Sand	-0.279 - -0.06
			Dead/Sparse	-0.059 - 0.07
			Low	0.071 - 0.123
			Medium	0.124 - 0.2
			High	0.201 - 1.0
Raccoon	NAIP	April 22, 2023	Water	-0.669 - -0.404
			Sand	-0.403 - -0.171
			Dead/Sparse	-0.17 - -0.104
			Low	-0.103 - 0.003
			Medium	0.004 - 0.1
			High	0.101 - 1.0

Suppl. Table 4 cont.

Island	Source	Imagery Acquisition Date(s)	Vegetation Class	NDVI Range
Queen Bess	Sentinel-2	August 20, 2021	Water	-0.035 - 0.177
			Sand	0.178 - 0.31
			Dead/Sparse	0.311 - 0.42
			Low	0.421 - 0.515
			Medium	0.516 - 0.604
			High	0.605 - 1.0
Queen Bess	NAIP	April 30, 2023	Water	-0.549 - -0.231
			Sand	-0.233 - -0.107
			Dead/Sparse	-0.106 - 0.008
			Low	0.009 - 0.095
			Medium	0.096 - 0.268
			High	0.269 - 1.0

Suppl. Table 4 cont.

Island	Source	Imagery Acquisition Date(s)	Vegetation Class	NDVI Range
Philo Brice	Sentinel-2	August 20, 2021	Water	-0.121 - -0.068
			Sand	-0.067 - 0
			Dead/Sparse	0.001 - 0.127
			Low	0.128 - 0.212
			Medium	0.213 - 0.4
			High	0.401 - 1
Philo Brice	NAIP	August 30, 2019	Water	-0.96 - -0.227
			Sand	-0.266 - -0.053
			Dead/Sparse	-0.052 - -0.09
			Low	0.091 - 0.247
			Medium	0.248 - 0.47
			High	0.471 - 1
Philo Brice	NAIP	April 30, 2023	Water	-0.744 - -0.227
			Sand	-0.266 - -0.053
			Dead/Sparse	-0.052 - -0.09
			Low	0.091 - 0.247
			Medium	0.248 - 0.47
			High	0.471 - 1

Suppl. Table 5. Area (ha) and proportion of habitat classes relative to yearly island area and multi-year extent on Raccoon in 2019, 2022 and 2023.

Island	Year	Habitat Class	Area (ha) per Year	Proportion of Yearly Island Area	Area within Multi-Year Extent (ha)	Proportion of Multi-Year Extent
Raccoon	**2019**	Water	10.2	0.165	26.2	0.336
		Sand/Dead	20.6	0.335	20.8	0.267
		Sparse/Grass	10.0	0.162	10.0	0.129
		Low	4.1	0.066	4.1	0.053
		Medium	6.9	0.112	6.9	0.089
		High	9.8	0.160	9.8	0.126
		Total Area	**61.7**		**77.8**	
	2022	Water	1.9	0.024	2.3	0.030
		Sand/Dead	48.3	0.624	48.4	0.621
		Sparse/Grass	16.2	0.210	16.2	0.208
		Low	4.1	0.053	4.1	0.052
		Medium	4.6	0.060	4.6	0.059
		High	2.3	0.029	2.3	0.029
		Total Area	**77.3**		**77.8**	
	2023	Water	4.1	0.080	30.1	0.386
		Sand/Dead	31.3	0.614	32.1	0.412
		Sparse/Grass	5.3	0.104	5.3	0.068
		Low	5.8	0.113	5.8	0.074
		Medium	3.4	0.067	3.4	0.044
		High	1.1	0.022	1.1	0.015
		Total Area	**51.0**		**77.8**	

Suppl. Table 6. Area (ha) and proportion of habitat classes relative to yearly island area and multi-year extent on Philo Brice in 2019, 2021, and 2023.

Island	Year	Habitat Class	Area (ha) per Year	Proportion of Yearly Island Area	Area within Multi-Year Extent (ha)	Proportion of Multi-Year Extent
Philo Brice	**2019**	Water	4.63	0.226	4.63	0.226
		Sand	3.56	0.174	3.56	0.174
		Dead/Sparse	3.27	0.160	3.27	0.160
		Low	3.55	0.173	3.55	0.174
		Medium	5.16	0.252	5.15	0.252
		High	0.29	0.014	0.29	0.014
		Total Area	**20.45**		**20.45**	
	2021	Water	0.02	0.001	1.51	0.074
		Sand	0.14	0.009	1.35	0.066
		Dead/Sparse	0.98	0.063	2.21	0.108
		Low	2.43	0.157	2.94	0.144
		Medium	8.45	0.545	8.94	0.437
		High	3.48	0.224	3.49	0.170
		Total Area	**15.52**		**20.45**	
	2023	Water	0.05	0.003	5.37	0.263
		Sand	1.02	0.075	2.09	0.102
		Dead/Sparse	1.72	0.126	1.94	0.095
		Low	3.72	0.272	3.82	0.187
		Medium	4.36	0.320	4.41	0.216
		High	2.79	0.204	2.82	0.138
		Total Area	**13.65**		**20.45**	

Suppl. Table 7. Area (ha) and proportion of habitat classes relative to yearly island area and multi-year extent on Queen Bess in 2021 and 2023.

Island	Year	Habitat Class	Absolute Area (ha) per Year	Proportion of Yearly Island Area	Area within Multi-Year Extent (ha)	Proportion of Multi-Year Extent
Queen Bess	**2021**	Water	1.05	0.070	1.050	0.070
		Sand	1.78	0.118	1.780	0.118
		Dead/Sparse	2.63	0.175	2.630	0.174
		Low	2.73	0.181	2.730	0.181
		Medium	3.71	0.246	3.710	0.246
		High	3.18	0.211	3.180	0.211
		Total Area	**15.07**		**15.1**	
	2023	Water	0.55	0.039	0.800	0.053
		Sand	2.23	0.156	2.410	0.160
		Dead/Sparse	3.17	0.222	3.250	0.215
		Low	2.92	0.205	2.930	0.194
		Medium	5.57	0.390	5.570	0.369
		High	0.14	0.010	0.140	0.009
		Total Area	**14.26**		**15.1**	

Biographical Sketch

Bonnie Jean Slaton was born in Arizona to Beverly C. Duhon and James C. Slaton. She grew up in Alaska, Germany, and Louisiana and graduated with honors from Louisiana State University A&M Baton Rouge in 2013 with a Bachelor of Science degree in Natural Resources Ecology & Management with a concentration in Conservation Biology. She then worked as a marine fisheries biologist in New Orleans for the Louisiana Department of Wildlife and Fisheries until she entered the biology master's degree program at UL Lafayette in 2021. Her research has focused on the impacts of coastal land loss and hurricanes on the ecology of seabirds in Louisiana and was supported by the Coastal Protection and Restoration Authority's Coastal Science Assistantship Program. She graduated in Fall 2025 with a Master of Science degree in Biology.