

Hurricane, Habitat Degradation, and Land Loss Effects on Brown Pelican Nesting Colonies

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ABSTRACT

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Nesting colonies of coastal avifauna are perennially threatened by hurricanes, land loss, and environmental contamination. To understand how nest substrate and habitat availability influence reproductive success of Brown Pelicans (*Pelecanus occidentalis*), we monitored 802 nests and quantified vegetation cover on two barrier islands in Louisiana from 2008 to 2010. In 2008, Hurricanes Gustav and Ike caused habitat degradation and land loss at our study sites and thus allowed comparison of pelican productivity in pre- and post-hurricane conditions. As habitat availability changed across years and islands, pelicans shifted from nesting in woody vegetation, to grasses, forbs, and bare ground. More chicks that survived until the age of 3 to 4.5 weeks old were from higher nests, and the loss of woody vegetation might have elicited colony abandonment. Habitat reduction was attributed to hurricane-induced erosion, and shoreline retreat was an average 5.5 times (range 3.3 to 11) greater than regional rates from 1887 to 2002. Furthermore, land loss (16% to 99% of vegetated regions) was restricted to areas without protective breakwaters. In addition to the effects of habitat decline on pelican reproduction, contamination by the Deepwater Horizon oil spill might have further decreased nest success. Large and productive seabird colonies can be rapidly degraded by both human and natural disturbance, making amelioration of such threats a management priority.

ADDITIONAL INDEX WORDS: *Pelecanus occidentalis*, reproductive success, shoreline retreat, barrier islands, vegetation change, oil spill, coastal avifauna, conservation.

INTRODUCTION

In the context of rapidly changing ecosystems, the population viability of wildlife subjected to disturbance and habitat degradation (Michener *et al.*, 1997) often depends on the efficacy of conservation management to meet species' needs adequately. Avifauna breeding in coastal ecosystems, and especially on barrier islands that are particularly vulnerable to hurricanes (Georgiou, FitzGerald, and Stone, 2005; Miner *et al.*, 2009) and environmental contamination (Piatt *et al.*, 1990), are increasingly reliant on restoration and conservation to provide productive breeding sites (Parnell *et al.*, 1988). Because hurricanes might be increasing in intensity and frequency (Bender *et al.*, 2010; Webster *et al.*, 2005), understanding the effects of storms on habitat (Sallenger *et al.*, 2009) and how best to ameliorate their detrimental effects on breeding waterbirds is of paramount importance to sustaining threatened habitat and nesting colonies. Furthermore, the implications of oil spills on reproductive success presents yet another management priority (Parnell *et al.*, 1988).

Across the northern Gulf of Mexico the conservation of Brown Pelicans (*Pelecanus occidentalis*) has been of interest for several decades, particularly in Louisiana where extensive management of the species has occurred since its 1963 temporary extirpation (McNease *et al.*, 1984; Nesbitt *et al.*, 1978). Because Brown Pelicans in Louisiana predominately rely on deteriorating barrier islands for reproduction (Visser *et al.*, 2005), their viability has been threatened by recent hurricane-induced habitat degradation and oil contamination at colony sites (Sallenger *et al.*, 2009; USFWS 2011). Brown Pelicans are representative of other coastal waterbird species that exhibit natal island fidelity (Shields, 2002; Visser and Peterson, 1994), and their reproductive success might be affected over several years when traditional colony sites are affected by storms or contamination.

Despite the long-standing interest to conserve Brown Pelicans in Louisiana, only coarse-scale data have been collected in regard to reproductive success (McNease, Richard, and Joanen, 1992; Nesbitt *et al.*, 1978). Furthermore, limited information is available in regard to Brown Pelican population dynamics, such as rates of colony abandonment or establishment, within the context of changing environmental conditions (Visser and Peterson, 1994; Visser *et al.*, 2005). Consequently, to assess how environmental factors influence Brown Pelican nest and colony success, we monitored reproductive efforts across two islands in Louisiana from 2008 to 2010. We also quantified vegetative cover and nesting densities to assess nest

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substrate selection and brood sizes in relation to colony site characteristics. To determine whether habitat degradation and land loss that resulted from Hurricanes Gustav and Ike in 2008 decreased productivity, we compared recruitment before and after the storms. Our research suggests several factors contributed to the decline of one colony and the success of another. Insights into factors that support population success of Brown Pelicans might also contribute to conservation of other waterbird species that rely on similar habitat used by pelicans.

METHODS

Study Sites

We conducted our research on Raccoon and Wine Islands in the Isles Dernieres, Louisiana (Figure 1). Both islands are less than 1 m in elevation (Visser *et al.*, 2005), and vegetation consists of a variety of species growing on sandy and silt substrate. Raccoon Island supports black mangrove (*Avicennia germinans*) and marsh elder (*Iva frutescens*), whereas Wine Island only has marsh elder. For a thorough overview of plant community assemblages typical of our study sites, see Hester, Spalding, and Franze (2005).

Nest Success Surveys

The success of 802 nests was assessed from 2008 to 2010 on Raccoon and Wine Islands. Nests were observed from the periphery of the colony at five observation locations per island and year. Different observation locations were used over years, and an average of 29 nests were identified per survey location (range 10 to 48). We identified nests in the first week of their construction and surveyed them with binoculars on average every 6 days from 2 April to 27 August across years, for an average of 16 surveys per nest per year, across years and islands. During surveys we recorded the date, number of chicks, and approximate chick age on the basis of chick size and plumage development. Chicks were assigned to one of six age categories on the basis of age in weeks: 0 to 1.5, 1.5 to 3, 3 to 4.5, 4.5 to 6, 6 to 7.5, and 7.5 to 9. These categories were developed from our 13,819 nest observations over the course of our study that tracked chick appearance since time of hatching. No chicks were associated with nests after 9 weeks of age. We monitored

nests until we were unable to view any chicks for two consecutive surveys, indicating that chicks either perished or naturally abandoned nests.

We also determined the number of chicks per nest that survived to the age category 3 to 4.5 weeks old. We selected this developmental stage as the threshold for counting chicks per nest because, after this period, chicks began to abandon their nests naturally, and we could not associate them with a particular nest. Conversely, younger chicks rarely left their nests.

Nest Site Characteristics Surveys

With binoculars we surveyed nest site characteristics for 729 of our 802 nests within a month of when chicks began to leave their nests (73 nests were lost to flooding or shoreline erosion before surveys could be conducted); surveys were conducted from the periphery of the colony to avoid disturbance. The substrate on which nests were placed was designated as: bare ground (no vegetation), grasses, forbs, marsh elder, black mangrove, and dead woody vegetation (either black mangrove or marsh elder). We also estimated nest height, as well as the distance to the nearest Brown Pelican nest as a surrogate measure of nest density. Finally, to quantify plant assemblages associated with nesting, we visually established an approximately 4-m² circular plot around each nest. In each plot the percent vegetation cover was estimated using the above habitat categories, and total plot coverage equaled 100%. Finally, within each plot, the dominant cover was determined as the habitat category with the greatest percent cover.

Habitat Availability Surveys

We quantified available vegetation across all vegetated portions of Raccoon and Wine Islands from 2008 to 2010. On Raccoon Island we established transects perpendicular to the island at 50-m intervals. On each transect, three habitat plots were positioned across the island at distances 25%, 50%, and 75% of the width of the vegetated area ($n = 141$ plots total). Because Wine Island is smaller and circular in shape, plots were placed on a grid, with plots located at the intersections of grid lines located 15 m apart in both dimensions ($n = 148$ plots total). To maintain consistency with nest site measurements, we used the same plot size and methods to estimate percent vegetation cover. We conducted surveys from atop a 3.5-m stepladder that afforded a better view and used a global positioning system, compass, and laser range finder to document habitat plot locations that allowed yearly remeasurements.

Habitat Visual Estimation Error Surveys

To evaluate error associated with measurements of percent vegetation cover at nest sites and habitat availability plots that were estimated from outside of the colony, we conducted habitat visual estimation error surveys in July 2009. We randomly established 93 (Raccoon Island = 51, Wine Island = 42) approximately 4-m² plots that were 3.1 to 145.9 m from the visual survey location. In each plot an observer (who also performed all estimates throughout the study) visually estimated percent vegetation cover from the periphery of the colony (*i.e.* estimated data). A second person was then directed to the plot to estimate percent cover at close proximity (*i.e.* actual data). Furthermore, 20 random pelican nests were

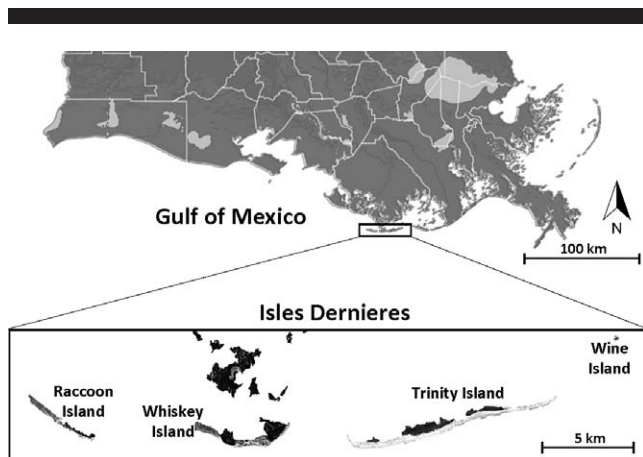


Figure 1. Map of the Louisiana coast and the Isles Dernieres archipelago.

Table 1. Average differences between data from 289 approximately 4-m² plots measured either at the plot (i.e. actual values) or visually estimated by an observer 3 to 146 m away (i.e. estimated values). Simple linear regression was used to assess precision (r^2) and bias (slope) between the two measurement techniques. Assessment of marsh elder percent cover was not possible because of insufficient data from a limited number of plots with this cover over time.

Variable	Avg. Difference of Actual – Estimated Values	r^2	Slope	SE
Bare % cover	6.72	0.88	1.01	0.04
Grass % cover	0.91	0.88	0.93	0.04
Forb % cover	-3.82	0.73	0.69 ^a	0.04
Marsh elder % cover	0.05	na	na	na
Black mangrove % cover	-3.28	0.90	0.69 ^a	0.02
Dead woody veg. % cover	-0.59	0.85	0.78 ^a	0.03
Distance to nearest nest (m)	0.77	0.93	1.14	0.07

Avg. = average, na = not assessed, veg. = vegetation.

^a Estimated data were considered biased when slopes were >2 standard errors (SE) from 1.

selected at 11.3 to 97.2 m from the survey point for estimating distances from each nest to the nearest neighboring nest. A second person then measured the actual distance on the ground with a tape measure.

To compare actual (measured on the ground) and estimated (estimated visually from outside the colony) data, we performed separate simple linear regressions (SAS Institute Inc., 2008; PROC REG) for percent vegetation cover categories and distance to nearest nest. The precision of estimated data was relatively high on the basis of r^2 values (0.73 to 0.93). However, the slopes of three regressions, with zero as the intercept, were >2 standard errors from 1 (Table 1). This indicates that percent forb, mangrove, and dead woody vegetation cover were underestimated relative to actual values, and bias in estimates was worse at higher values of percent cover. In these cases where the degree of bias changes with the variable being estimated, bias in our estimates could possibly affect interpretation of our analyses. Therefore, when appropriate for statistical analyses of forb, black mangrove, and dead wood cover, as noted below, we adjusted estimates of percent cover by multiplying each value by its slope.

Shoreline Retreat Surveys

To evaluate shoreline loss on Wine Island, which appeared to have markedly higher erosion rates compared with other study areas, we quantified the magnitude and rate of shoreline retreat from 14 May 2009 to 13 July 2009. Along the south (i.e. Gulf of Mexico) shore where erosion was most pronounced, we placed five stakes spaced at 15-m intervals at inland locations. From each stake we established a bearing to the closest point where shoreline erosion began. Every 5 days we measured the distance, to the nearest centimeter, from each stake to the nearest point of erosion along the respective bearings ($n = 12$ surveys).

Statistical Analyses

Nest Success

To assess nest success, defined as the number of chicks per nest that reached the age category 3 to 4.5 weeks old, we used the following candidate explanatory variables: island (Raccoon, Wine), year (2008, 2009, 2010), nest substrate, dominant cover, nest height, and distance to nearest neighboring pelican nest. Plot cover data were adjusted for visual bias (Table 1) before determining dominant cover. The potential for multicollinearity of the explanatory variables that were

not multinomial (island, year, nest height, and distance to nearest pelican nest) were assessed with Pearson's correlations. Correlations were less than 0.62, suggesting that these variables contained at least some independent information. To help evaluate whether relationships among variables had a strong effect on our analyses, we evaluated models containing each variable both separately and with other variables. We used data from 729 of our original 802 monitored nests because 73 nests were lost to tidal flooding or shoreline erosion before habitat data were gathered.

To evaluate the influence of our explanatory variables on nest success we performed Akaike's Information Criterion (AIC) analyses, with the response variable as the number of chicks per nest that reached 3 to 4.5 weeks old. Three individual AIC analyses were conducted: one for Raccoon and Wine data evaluated together and one each for islands considered separately. Each of these three assessments used 29 biologically plausible models that included one, two, and three main effect models and their interactions, as well as the fully saturated global model. We also ran the null intercept model for comparison. For analyses of islands considered separately, we used the variables year, nest substrate, nest height, nearest nest, and dominant cover. When islands were evaluated together we used the variables island, year, nest substrate, nest height, and nearest nest. We included "island" to assess its possible influence, and removed "dominant cover," which was not influential when islands were analyzed together. To obtain AIC_c values, adjusted for small sample size (Burnham and Anderson, 2002), we used Poisson regression (SAS Institute Inc., 2008; PROC GENMOD). Models with $\Delta AIC_c < 2$ units from the model with the smallest AIC_c value were considered to provide reasonable support for the data (Burnham and Anderson, 2002). No competing models contained overdispersed data (deviance/df ≤ 1.28). Model weights (w_i) were calculated to provide an additional measure of model support; models with larger weights provide more support for the data (Burnham and Anderson, 2002). Finally, we evaluated the relationship between the number of chicks per nest that reached 3 to 4.5 weeks old and nest height, and distance to nearest neighbor with separate Poisson regressions.

Nest Site Characteristics

We used Goodness of Fit (GoF) tests, with 10,000 Monte Carlo data resamplings to determine exact p values (SAS

Institute Inc., 2008; PROC FREQ), to compare numbers of pelican nests on each of the nest substrate categories (bare ground, grasses, forbs, marsh elder, black mangrove, and dead woody vegetation) with their proportionate availability. Substrate use was based on 729 nests across years, and substrate availability was based on 289 habitat plots surveyed each year. Because habitat conditions differed considerably between islands (Raccoon, Wine) and among years (2008, 2009, 2010), we performed six separate GoF tests for each year and island combination. Because this analysis is based on numbers of nests on each type of substrate category, there was no adjustment possible for biases in our estimates of percent cover.

Habitat Availability

To assess changes in habitat availability from 2008 to 2010, we used separate multivariate analyses of variance (MANOVAs; SAS Institute Inc., 2008; PROC GLM) for Raccoon and Wine Islands. We combined proportional habitat cover plot data into the vegetation categories: bare ground, grasses and forbs, woody vegetation (mangrove and marsh elder), and dead woody vegetation. We adjusted data for visual estimation error and used an arcsine square root transformation for proportional data. Year was used as the main effect, and individual plots served as a blocking factor to reduce interplot variation on each island.

RESULTS

Nest Success

Across islands and years the average number of chicks that reached the age class 3 to 4.5 weeks old ranged from 0.0 to 1.6 chicks per nest (Figure 2). When Raccoon and Wine Island data were analyzed together, only the fully saturated global model best explained our chick data (Table 2). Inclusion of the island variable in the model and its interactions with other variables indicates the importance of analyzing island data separately to investigate factors that could have led to generally fewer chicks per nest reaching the age of 3 to 4.5 weeks old on Wine Island compared with Raccoon Island (Figure 2).

When islands were assessed independently, models that included variables year, nest height, and nearest nest, along with interactions, best explained both Raccoon and Wine Island data (Table 2). Raccoon Island had an overall trend of more chicks in higher nests (Figure 3; 2008: $\chi^2_1 = 11.62$, $p = 0.0007$; 2009: $\chi^2_1 = 1.92$, $p = 0.17$; 2010: $\chi^2_1 = 25.39$, $p < 0.0001$). Wine Island had comparable numbers of chicks per nest across nest heights (Figure 3; 2008: $\chi^2_1 = 0.63$, $p = 0.43$; 2009: $\chi^2_1 = 0.04$, $p = 0.87$; 2010: no chicks of any age were observed on Wine Island).

In regard to distances from our survey nests to the nearest pelican nest, on Raccoon Island there were no significant trends during our study (Figure 3; 2008: $\chi^2_1 = 0.86$, $p = 0.35$; 2009: $\chi^2_1 = 1.70$, $p = 0.19$; 2010: $\chi^2_1 = 0.38$, $p = 0.54$). On Wine Island, increasing distances between survey nests and neighbors resulted in fewer chicks in 2008, but not 2009 (Figure 3; 2008: $\chi^2_1 = 9.26$, $p = 0.002$; 2009: $\chi^2_1 = 0.74$, $p = 0.39$). No chicks were hatched in 2010 on Wine Island.

Nest Site Characteristics

Pelican's use of nesting substrates (bare ground, grasses, forbs, marsh elder, black mangrove, dead woody vegetation)

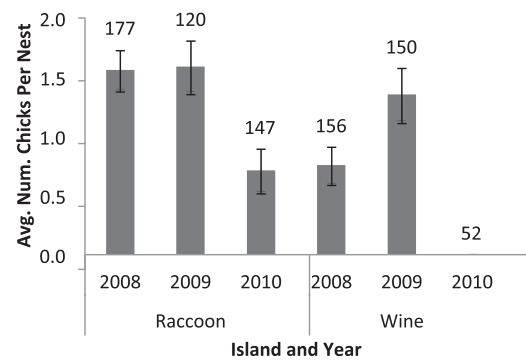


Figure 2. Average number of chicks that reached the age class 3 to 4.5 weeks old per nest on 802 nests monitored from 2008 to 2010 on two islands in Louisiana. Numbers of nests surveyed each year are listed above each bar, and error bars are 95% confidence intervals.

differed from their availability on both Wine and Raccoon Islands in all three years of the study ($\chi^2_{1-5} = 88.96-1330.91$, $p < 0.0001$). In general, pelicans predominately selected woody vegetation as nest substrates on both islands when available (Figure 4). When woody vegetation (marsh elder) was absent in 2009 and 2010 on Wine Island, pelicans increasingly nested on bare ground and grasses (Figure 4).

Habitat Availability

Year influenced proportional vegetation cover from 2008 to 2010 on both Raccoon ($F_{8,556} = 15.92$, Pillai's trace $p < 0.0001$) and Wine Islands ($F_{8,584} = 91.13$, Pillai's trace $p < 0.0001$). On the basis of Tukey's multiple comparisons of least square means, on both Raccoon and Wine Islands, woody vegetation declined from 2008 to 2009 but did not change in cover from 2009 to 2010 (Figure 5). Forb and grass cover declined during the first 2 years of our study on both islands. Coverage then increased to 2008 levels in the final year on Raccoon Island but continued to decrease on Wine Island. The proportional cover of bare ground increased on both islands from 2008 to 2009. This unvegetated ground then returned to 2008 levels in 2010 on Raccoon Island, whereas on Wine Island, levels dropped to the lowest of 3 years in 2010. The proportion of cover that was dead woody vegetation increased from 2008 to 2009 on both islands. This cover then declined to earlier levels in 2010 on both islands. The considerable decrease in all habitat covers on Wine Island over years was largely due to shoreline retreat that converted vegetated regions to open water. Incidentally, the individual habitat plot variable used as a blocking factor to reduce interplot variation was significant on Raccoon ($F_{560,1120} = 2.09$, Pillai's trace $p < 0.0001$) and Wine ($F_{588,1176} = 1.24$, Pillai's trace $p = 0.0009$) Islands.

Shoreline Retreat

Of the 141 habitat availability plots established in 2008 on Raccoon Island, 15% (21 plots) were lost to erosion by 2009, and an additional 1% (2 plots) by 2010. Of the 148 habitat plots on Wine Island, 51% (76 plots) were lost from 2008 to 2009, and an additional 48% (70 plots) by 2010.

Table 2. Models used to assess the influence of different on the number of chicks per nest that reached 3 to 4.5 weeks old from 2008 to 2010 on Raccoon and Wine Islands, Louisiana. Separate analyses assessed data for islands combined or individually (Raccoon Island = 444 nests; Wine Island = 285 nests). Variables assessed for analyses of islands combined were: island, year, nest substrate, nearest neighboring pelican nest, and nest height. Variables used when islands were evaluated separately were the same, except “dominant habitat cover” at each nest site was used instead of “island” (see “Statistical Analyses”). Models with the notation “FACTORIAL” indicate all combinations of variable interactions are included. AIC_c values were used to compare models, and models with $\Delta AIC_c \leq 2$ units from the model with the smallest AIC_c value are reported. Null and global models are included for comparison.

Island, Response Variable, and Model	K	AIC_c	ΔAIC_c	w_i
Raccoon and Wine: No. of chicks				
Fully saturated global model	65	1942.80	0.00	1.00
Null intercept	1	2216.18	273.38	0.00
Raccoon: No. of chicks				
Year + nest ht. + nearest nest FACTORIAL	12	1269.44	0.00	0.53
Year + nest ht. + year \times nest ht.	6	1269.74	0.30	0.45
Fully saturated global model	73	1312.95	43.51	0.00
Null intercept	1	1360.47	91.03	0.00
Wine: No. of chicks				
Year + nearest nest	4	647.09	0.00	0.48
Year + nearest nest + year \times nearest nest	5	648.46	1.37	0.24
Year + nest ht. + nearest nest	5	648.51	1.42	0.23
Fully saturated global model	49	720.93	73.84	0.00
Null intercept	1	853.08	205.99	0.00

K = No. of parameters estimated + 1, $AIC = -2 \times \log \text{likelihood} + 2 \times K$, $AIC_c = AIC$ adjusted for small sample size, $\Delta AIC_c = AIC_c$ difference from the model with the lowest AIC_c , w_i = model weight—higher values provided increased model support, ht. = height.

During summer 2009 an average of 1.1 m of shoreline retreat occurred per week on the south (Gulf) side of Wine Island (9.57 m total from 14 May to 13 July). As erosion reached the nesting regions on the island in 2009, approximately 20% of all nests (80 of 400 total nests) were lost to shoreline retreat. Furthermore, in 2010 15% of all nests were lost to the Gulf because of continued shoreline retreat (8 of 52 total nesting attempts).

DISCUSSION

Across years and colonies we observed an average range of 0 to 1.6 chicks per nest that reached the age category of 3 to 4.5 weeks old and that nest success was influenced by multiple factors and their interactions. Given that we monitored colonies only 30 km apart over only 3 years, we did not anticipate the large variance in nest success or the breeding failure on Wine Island in 2010. To explain differential nest and colony success, it is useful to consider reproductive success within the context of changing habitat conditions because of the 2008 Hurricanes Gustav and Ike.

In 2009 on Raccoon Island, we recorded a reduction in woody vegetation cover compared with the previous year before the hurricanes, as has been reported on other islands after hurricanes (Green *et al.*, 2011; Leberg *et al.*, 2007; Michener *et al.*, 1997). Most of the loss was attributed to the reduction of marsh elder (85% loss) compared with black mangroves (25% loss). In Fall 2008 we observed that mangroves on Raccoon Island had already begun to regenerate new leaves 1 month after the hurricanes, as was also observed in other research on black mangroves after disturbance (Smith *et al.*, 1994). This rapid growth response allowed for the persistence of the Brown Pelican's preferred nesting substrate over subsequent years of our study.

On Wine Island, marsh elder cover was reduced 99% by 2009. Although marsh elder has the capacity to regenerate after

severe damage (Michener *et al.*, 1997), it was not able to recover before shoreline retreat resulted in the complete loss of the vegetated portion of the island by 2010. Although our 2-year assessment suggests that black mangroves appear to be more resilient than marsh elder to storm events in the short term, other research indicates that damaged black mangroves might experience delayed mortality more than 2 years after disturbance (Smith *et al.*, 1994). Further research is required to evaluate hurricane-induced change in woody vegetation cover, in relation to land loss, over time at our study sites.

Grass and forb cover also declined at our study sites the year after the hurricanes. On Raccoon Island, the return of coverage to roughly prehurricane levels by 2010 was likely due to the rapid growth and expansion patterns of these early successional plants (Hester, Spalding, and Franze, 2005; Sharitz and Pennings, 2006). However, on Wine Island, grass and forb cover further declined from 2009 to 2010. This loss was largely due to continued shoreline retreat and exemplifies the ability of hurricane-induced land loss to diminish cover of vegetation with robust growth patterns.

The hurricanes' routes could have influenced differences in plant responses over years between Raccoon and Wine Islands. Hurricane Ike made landfall in Texas >350 km away from our study sites (Stormpulse, 2008), so its effects to our study sites were likely similar. However, Hurricane Gustav passed over Trinity Island (Stormpulse, 2008), which resulted in the right side of the hurricane passing over Wine Island. In the Northern Hemisphere, wind speeds and wave heights are typically greatest in the upper right quadrant of a hurricane's path. Consequently, Wine Island could have experienced greater hurricane forces and damage compared with Raccoon Island.

Nest success varied widely across islands and years. One of the most influential factors of fledging success was nest height, which corresponds to heights of available woody vegetation. Our results show that pelicans exhibited a strong preference

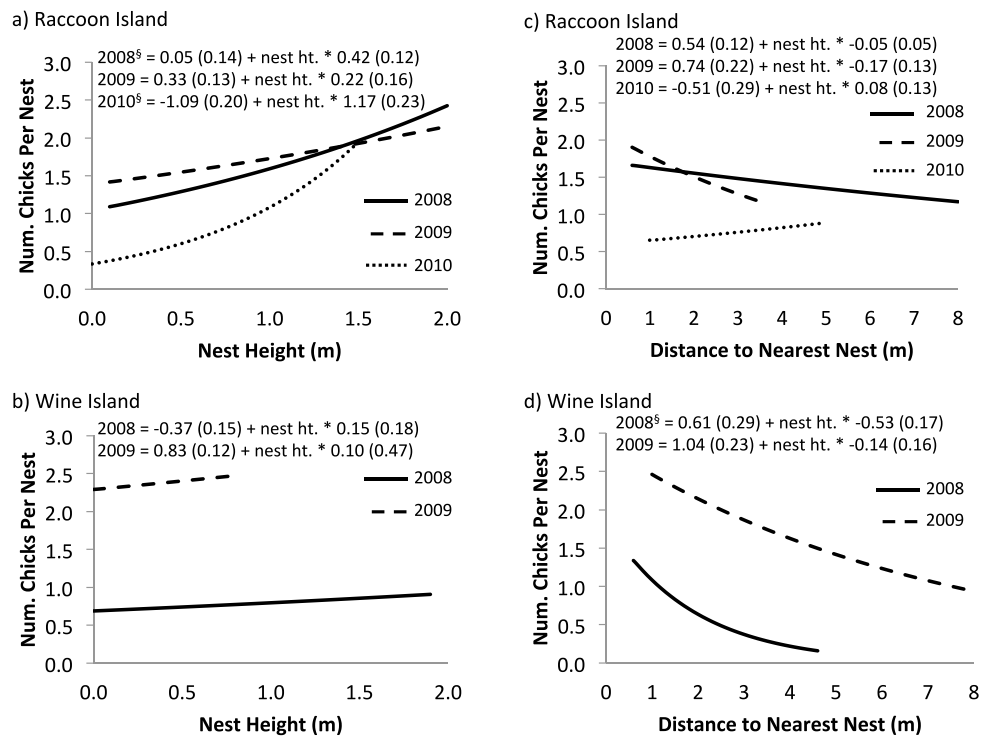


Figure 3. Number of Brown Pelican chicks that reached 3 to 4.5 weeks old from nests at different nest heights (a, b) and at different distances from the nearest pelican nest (c, d) in Louisiana from 2008 to 2010. We surveyed 444 and 285 total nests on Raccoon and Wine Islands, respectively. No chicks hatched from nests on Wine Island in 2010. Regression equations are provided as the intercept (SE) + variable \times slope (SE). Years with $p < 0.05$ are indicated by §.

for shrub species as nesting platforms, which is likely due to the ability of black mangroves and marsh elder to provide both the sturdiest and highest nest sites. The low elevation of Louisiana barrier islands subjects nesting colonies to the perennial threat of storm overwash and reduced nest success. For example, McNease, Richard, and Joanen (1992) documented the loss of nearly 400 Brown Pelican nestlings and numerous eggs because of tidal flooding; nests closest to the ground experienced the greatest losses. Our observation of more chicks reaching 3 to 4.5 weeks old at greater nest heights in some years provides support for the value of taller nesting platforms.

The relatively uniform nesting densities we found across years and islands demonstrate the Brown Pelican's tendency for close, communal nesting (Shields, 2002). Colonial waterbird nesting offers the benefits of "public information" in regard to optimal nest and foraging site selection (Danchin, Boulinier, and Massot, 1998; Forbes and Kaiser, 1994). For instance, greater chick growth and survival of gulls has been documented in "dense" compared with "loose" nesting subcolonies (Savoca *et al.*, 2011). However, individuals in dense colonies are also subjected to increased susceptibility to disease, intraspecific aggression, and competition for nest sites (Hunt, Eppley, and Schneider, 1986; Pius and Leberg, 1997). We found that slightly fewer pelican chicks reached the age of 3 to 4.5 weeks old in nests at greater distances from neighboring nests. Because adult Brown Pelicans aggressively protect nest sites from other pelicans (Shields, 2002), the small average distances

to neighboring nests we observed are not likely a factor of beneficial dense nesting but, rather, reflect attempts to occupy the best quality nest sites. Low numbers of fledglings in isolated nests in our study suggest that suboptimal nest locations might have resulted in lower nest success during 1 year of our study.

Trends in nest success before and after the hurricanes exemplify the potential importance of breeding associations with habitat condition. In 2008 and 2009 on Raccoon Island, we observed an average of 1.6 chicks per nest reaching 3 to 4.5 weeks old per year, which is similar to estimates of 1.2 to 1.7 chicks fledged per nest across other Louisiana islands from 1971 to 2001 (McNease *et al.*, 1984; McNease, Richard, and Joanen, 1992). In 2010 we observed an average of 0.7 chicks per nest on Raccoon Island, which is roughly half of prehurricane nesting recruitment on the same island and of historic fledging averages. An average of 1.3 young per breeding female per year can maintain a sustainable colony (McNease, Richard, and Joanen, 1992), so prehurricane conditions on Raccoon Island appeared to support nesting productivity that would promote population growth. However, if the reproductive success observed in 2010 continues over subsequent years, the colony would not be able to support itself without outside recruitment.

On Wine Island during prehurricane conditions in 2008, we observed relatively low productivity at 0.8 chicks per nest, despite considerable availability of preferred marsh elder nesting habitat. But in 2009, among marginal habitat

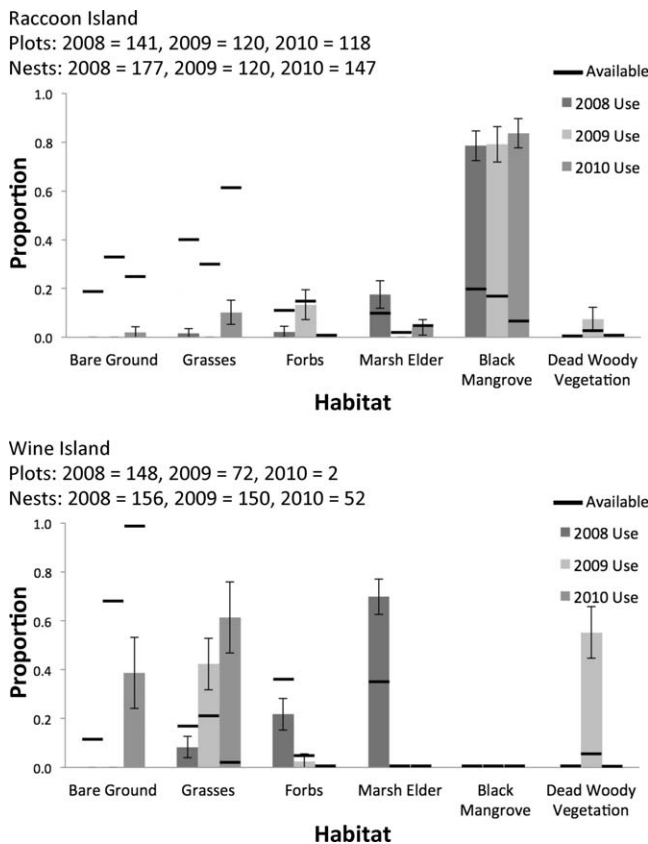


Figure 4. Habitat availability and use by Brown Pelicans as nesting substrate on Raccoon and Wine Islands in the Isles Dernieres archipelago, Louisiana, from 2008 to 2010. Habitat use data are from nests surveyed for success, and proportionate habitat availability data are from random, approximately 4-m² plots surveyed each year. During the study, plots were lost to shoreline erosion, and graphed habitat values represent proportions only from remaining plots each year. Error bars are 95% confidence intervals for proportions.

conditions, we observed an increased average of 1.4 chicks per nest. This increase in productivity from 2008 to 2009 is contrary to the typical decline of other waterbird nesting numbers after hurricane-induced habitat degradation (Green *et al.*, 2011; Leberg *et al.*, 2007). However, waterbird nest success might increase after hurricanes because of a reduction in nest predators, such as raccoons (*Procyon lotor*). Although Wine Island does not support mammalian predators, a less conspicuous predator could have contributed to chick recruitment trends from 2008 to 2009: the red imported fire ant (*Solenopsis invicta*). Although we did not conduct fire ant surveys, we observed their presence on Wine Island in 2008, but not in 2009 or 2010. Because fire ant predation on chicks is known to reduce coastal waterbird nest success in the northern Gulf of Mexico (Drees, 1994; Lockley, 1995), the potential relationship between fire ants and Brown Pelican nest success in our study area warrants further investigation.

The severe habitat degradation on Wine Island likely contributed to the failure of Brown Pelican nesting attempts

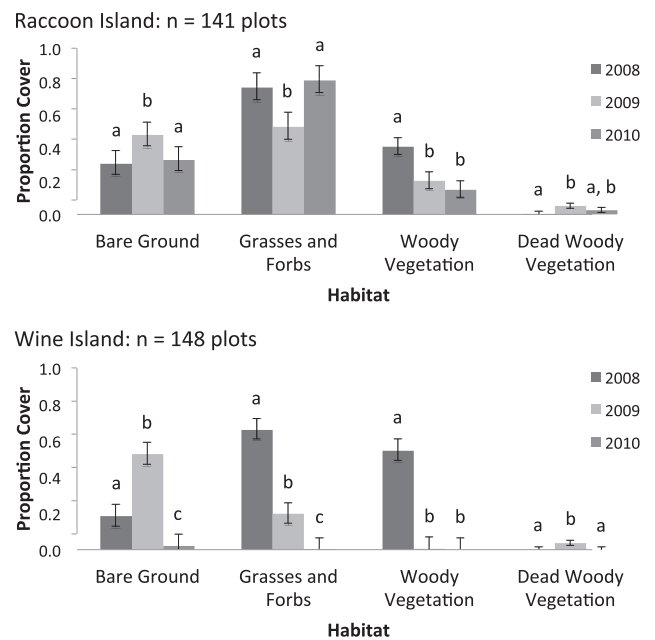


Figure 5. Proportion of habitat cover in approximately 4-m² plots on Raccoon and Wine Islands, Louisiana, from 2008 to 2010. Proportion of cover was visually estimated in June and July to correspond to when Brown Pelican chicks naturally abandon their nests in our study site. Data were arcsine transformed, and all data except those for bare ground were adjusted for visual estimation error. Least squares means were used to assess within-habitat category differences across years, as indicated by different letters, on the basis of Tukey's multiple comparisons and an alpha of 0.05. Least squares means of transformed and adjusted data are presented.

in 2010. By this year, no marsh elder was present, and grass and forb cover that constituted 20% cover in 2009 had dropped to <1%. The lack of preferred nesting habitat relegated pelicans to nest on patches of grass and bare ground, and all 52 nests in 2010 were abandoned before any eggs hatched. Given the low average nest height (0.18 m) and the decreasing size of Wine Island, nests were extremely susceptible to tidal overwash that has been attributed to the decrease in nest success in various waterbird species (Marsh and Wilkinson, 1991; McNease, Richard, and Joanen, 1992). A possible overwash event in 2010, in conjunction with the island's severely degraded habitat and loss of 15% of nests to shoreline erosion, might have provided sufficient environmental cues to elicit nest desertion.

Although land loss on barrier islands is a common result of hurricanes in Louisiana (Sallenger *et al.*, 2009), the average rate of 1.1 m of weekly shoreline retreat on Wine Island during 2009 was 5.5 times higher (range 3.3 to 11 times) than yearly averages from 1887 to 2002 within the Isles Dernieres (Penland *et al.*, 2005). This rapid rate of erosion was particularly unexpected given its occurrence during summer months that have calmer weather relative to winter (Georgiou, FitzGerald, and Stone, 2005). We suggest that the influence of the 2008 hurricanes on Wine Island's vegetation contributed to the hastened shoreline retreat.

As Wine Island's vegetation declined following Hurricanes Gustav and Ike, its structural ability to stabilize the substrate, both above- and belowground (Hester, Spalding, and Franze, 2005), was diminished. Consequently, the island's sand and silt substrate was more prone to erosion in post- *vs.* prehurricane conditions. In addition to vegetative cover, differences exist in shoreline protection between Raccoon and Wine Islands. On the eastern half of Raccoon Island, breakwaters constructed from boulders are parallel to the Gulf-side shoreline. These barriers are designed to attenuate wave energy and thereby mitigate shoreline erosion (Georgiou, FitzGerald, and Stone, 2005). During our study, none of the habitat plots behind breakwaters were lost, whereas all plots located at the western end of Raccoon Island that lacks breakwaters were converted to open water. Although no breakwaters are present at Wine Island, a dike constructed of boulders is on the Gulf side. However, the height of the dike is at or below sea level. We did not evaluate wave energy associated with the dike or breakwaters and thus cannot associate land loss trends with these structures. However, the land loss on Wine Island (99% of habitat plots) and Raccoon Island in areas without breakwaters (100% of habitat plots)—yet no loss of plots behind breakwaters—suggests their potential value in at least short-term habitat preservation.

In summer 2010, oil released from the failed Deepwater Horizon wellhead located 80 km SE of the mouth of the Mississippi River (Camilli *et al.*, 2010; Crone and Tolstoy, 2010) might also have adversely influenced Brown Pelican reproduction. Much of the Louisiana coastline received oil (Chen and Denison, 2011), including our study sites, and oil contamination has previously imparted various lethal and sublethal effects on waterbird populations (Belanger *et al.*, 2010; Briggs, Gershwin, and Anderson, 1997; Piatt *et al.*, 1990). Furthermore, our study sites experienced extended periods of human disturbance related to oil response and cleanup efforts. Although we lack data for a direct evaluation, contaminants from the Deepwater Horizon incident might have reduced nest success on Raccoon Island from 2009 to 2010 and also prompted the 2010 colony abandonment on Wine Island. However, despite the plausibility of this scenario, we emphasize that additional evidence is needed to determine whether observed declines in nest success are adequately explained by this hypothesis.

CONCLUSIONS

Our research has quantified various deleterious effects of hurricane-induced degradation of Brown Pelican nesting habitat. We suggest that protective breakwaters might have mitigated the magnitude of the hurricane forces and the consequential loss of habitat in some of our study area. Although we acknowledge potential drawbacks of breakwaters (*e.g.* a down-shore erosional shadow), we encourage their consideration as a restoration and protection practice within the context of the rapid decline of Louisiana pelican colony sites. We have observed the maintenance of five additional Brown Pelican colonies by use of boulder-reinforced shorelines in Texas, Louisiana, Alabama, and Florida. These islands are contrasted against the continued erosion of

numerous nonprotected colony sites. Beach nourishment and dune construction are additional measures that have been successfully employed to conserve barrier islands (Penland *et al.*, 2005). Depending on the bathymetrical context of particular islands, one or more of the above practices might benefit Brown Pelicans, as well as a suite of co-occurring waterbird species.

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