RESTORATION ECOLOGY
The Journal of the Society for Ecological Restoration

RESEARCH ARTICLE

Effects of barrier island salt marsh restoration on marsh bird occurrence in the northern Gulf of Mexico

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In the northern Gulf of Mexico, salt marshes are threatened by sea level rise, erosion, and loss of protective barrier islands. These barrier islands provide critical habitat for wildlife, including globally significant populations of marsh and shorebirds. We investigated salt marsh restoration on two Louisiana barrier islands using presence of eight marsh bird species as an index to evaluate restoration success. Land loss was extensive for both islands prior to restoration, with submerged marsh restored by backfilling sediment into the marsh platform. Restoration methods were similar between the two islands, although Raccoon Island was built to a higher elevation (1.1 m) than Whiskey Island (0.8 m). Avian presence was estimated via passive acoustic monitoring and point counts. To evaluate restoration success, we modeled influence of habitat covariates on index species presence in restored and reference (intact) sites over three breeding seasons and modeled occupancy for six species. On Whiskey Island, index richness was higher in restored sites. Marsh specialists Seaside Sparrows (*Ammospiza maritima*) and Least Bitterns (*Ixobrychus exilis*) had higher occupancy in restored areas on Whiskey, while generalist species showed no response to site. These results are likely due to a strong association between habitat and vegetation type, with restored sites dominated by *Spartina alterniflora* and reference sites by *Avicennia germinans*. On Raccoon Island, species richness was low across all sites. Our results suggest that restoration efforts were successful in creating salt marsh habitat on Whiskey but not Raccoon as of the time of our study.

Key words: acoustic monitoring, *Avicennia germinans*, coastal wetlands, occupancy modeling, restoration monitoring, *Spartina alterniflora*

Implications for Practice

- In our study system, restoration efforts designed primarily to minimize erosion of marsh restoration sites were not conducive to the creation of salt marsh habitat.
- Spread of Black Mangrove (*Avicennia germinans*) into the northern Gulf of Mexico salt marsh may have negative consequences for avian communities, particularly for salt marsh specialists such as Seaside Sparrows (*Ammospiza maritima*). If reconstructing salt marsh habitat is a restoration goal, *A. germinans* should not be planted as part of restoration efforts in the northern Gulf of Mexico.
- Salt marsh restoration activities can be designed to balance both the needs of human communities (preventing land loss) and wildlife communities (creating quality habitat).

Introduction

Salt marshes are among the rarest ecosystems in the world due to their reliance on a narrow range of hydrological conditions along temperate and subtropical coastlines (Mcowen et al. 2017). High rates of habitat loss couple with these restrictive biophysical requirements to render this ecosystem type critically imperiled, with an estimated 25–50% of the world's coastal marshes lost in the twentieth century (Davidson 2014; Mcowen et al. 2017). Despite their relative rarity, salt marshes play a critical role in sheltering coastal zones from damage. As sea levels rise and storms intensify due to the effects of global climate change, coastlines are becoming increasingly vulnerable to damage from storm surges and flooding (Anderson et al. 2014; Dolan & Walker 2006). Salt marshes protect these coastal zones from storm surge effects by reducing wave energy and saltwater intrusion (Stone & McBride 1998; Bilskie et al. 2016), minimizing the amount of flood damage to coastal communities. With recognition of the ramifications of salt marsh loss, restoring coastal wetlands has become a major

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Author contributions: PAB, PLL conceived and designed the research; PAB, ARP collected the data; PAB analyzed the data; JHW provided modeling input; PAB, PLL wrote the manuscript, PAB, PLL, JHW edited the manuscript.

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© *2020 Society for Ecological Restoration*
doi: 10.1111/rec.13222
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.13222/suppinfo

conservation focus worldwide (Millennium Ecosystem Assessment 2005).

In the United States, over 50% of coastal marshes were lost in the twentieth century (Kennish 2001), with almost half of this loss occurring in the state of Louisiana (Couvillion et al. 2018). Louisiana has the most salt marsh in the United States, and it has also lost the most: over 4,800 km^2 of coastal marshland since the 1930s (Couvillion et al. 2018). Marsh loss in Louisiana is typically attributed to the combined effects of saltwater intrusion, land subsidence, and coastal development (Day et al. 2005), but fragmentation of the state's barrier islands also contributes to its salt marsh declines. Louisiana's network of fringing barrier islands help protect salt marshes by absorbing wave energy (Stone & McBride 1998; Deaton et al. 2014). Without these barrier islands, coastal marshes are directly exposed to wave energy, which can accelerate erosion (Stone & McBride 1998). Because of their role in mitigating land loss, there has been considerable investment of time and effort in restoration of Louisiana's barrier island chains (Penland et al. 2003; Byrnes et al. 2017). These barrier islands are characterized by their by Gulf-facing beaches and bayside (or back barrier) salt marsh, and barrier island restoration typically includes replenishment of both habitat types (Campbell et al. 2005; Byrnes et al. 2017). Both beach and salt marsh confer structural protections that stabilize the island and prolong its lifespan, with back-barrier salt marsh being particularly critical for its role in minimizing bayside erosion of the island (Campbell et al. 2005).

Louisiana's barrier islands provide ecosystem services beyond protecting mainland communities and reducing erosion. These barrier island beaches and salt marshes are important wildlife habitat, supporting globally significant populations of breeding, overwintering, and migrating birds each year (Curtiss & Pierce 2016; Remsen et al. 2019) and providing nursery habitat for many of the region's economically important fisheries species (Boesch & Turner 1984). Restoration of barrier island salt marsh has been largely successful in stalling land loss (Penland et al. 2003; Byrnes et al. 2017), but little follow-up monitoring has been conducted to evaluate the use of these restoration sites by wildlife. It is vital that wetland restoration efforts in the northern Gulf of Mexico work toward replacing lost wildlife habitat in addition to addressing the issue of coastal erosion.

We compared salt marsh restoration sites on two Louisiana barrier islands, Whiskey and Raccoon, over 3 years to evaluate the success of restoration efforts in creating wildlife habitat. Restoration techniques were comparable between the two islands, but with some key differences. Particularly, marsh elevation was built higher than is recommended for tidal marsh creation (Penland et al. 2003) on Raccoon to slow erosion at the restoration site, and protective levees were retained to minimize tidal inflow into the marsh platform (Byrnes et al. 2017). On Whiskey, the marsh platform was built to a recommended height, exposing it to tidal inflow, and canals were incorporated into the restoration planning to encourage natural hydrology (Byrnes et al. 2017).

For the purposes of our study, we defined as a site as successful if it provided high-quality salt marsh habitat. We defined a site as high quality if it was able to support the biotic interactions and ecological processes expected in a healthy salt marsh, which we evaluated via indicator species. Vegetative cover and species richness are commonly used metrics for investigating restoration success (Ruiz-Jaen & Aide 2005); however, in barrier island salt marsh, plant species richness is typically low, and late succession sites tend to be characterized by a few dominant species (Melvin & Webb Jr. 1998). Instead, we used an index of avian species presence to investigate restoration success. Presence–absence of marsh-dependent birds is a commonly used proxy for determining habitat quality due to the guild's well-characterized associations with specific habitat features (Melvin & Webb Jr. 1998;Seigel et al. 2006; Desrochers et al. 2008). For marsh birds, habitat can be considered high quality if it contains the prey resources (Rush et al. 2010) and specific microhabitat features required for breeding, such as presence of emergent vegetation and depth of standing water (Lor & Malecki 2006). If these conditions are not met, marsh bird presence is expected to be low compared to a high-functioning habitat (Seigel et al. 2006; Desrochers et al. 2008), particularly for specialist marsh species (Elphick et al. 2015). Our research addressed three primary questions: (1) Is index species presence comparable between restored and reference salt marsh sites? (2) Are restoration outcomes similar between the two islands, given their different treatment? (3) Do specialist marsh species show comparable occupancy between restored and reference salt marsh sites?

Methods

Study Site

This study took place in the Isles Dernieres Barrier Island Wildlife Refuge in Terrebonne Bay, Louisiana. These islands once formed a contiguous, 40-km-long barrier island called Last Island which was fragmented by a hurricane in 1856 (Byrnes et al. 2017); the remnants are now considered among the fastest-eroding coastal lands in the United States (Campbell et al. 2005; Byrnes et al. 2017). Efforts to preserve the islands have included beach nourishment, instillation of rock breakwaters, and back-barrier marsh restoration (Penland et al. 2003; Khalil & Lee 2006; Byrnes et al. 2017).

We evaluated habitat quality of restored back-barrier salt marsh sites on Whiskey and Raccoon islands, two of the three Isles Dernieres. Whiskey is approximately 6.5km long and is the central island (Fig. 1A), while Raccoon is approximately 2.4km long and forms the westernmost edge of the chain (Fig. 1B). Both islands are closed to public access, but Whiskey has an active oil platform on its easternmost point.

Approximately 129 ha of salt marsh were restored on Whiskey in 2009, along with creation of a network of tidal creeks and ponds (Byrnes et al. 2017). Land was created by dredging sediment from an offshore subaqueous sand bank and pumping it into the marsh platform, where it was temporarily held in place by a containment dike. *Spartina alterniflora* (Smooth Cordgrass;

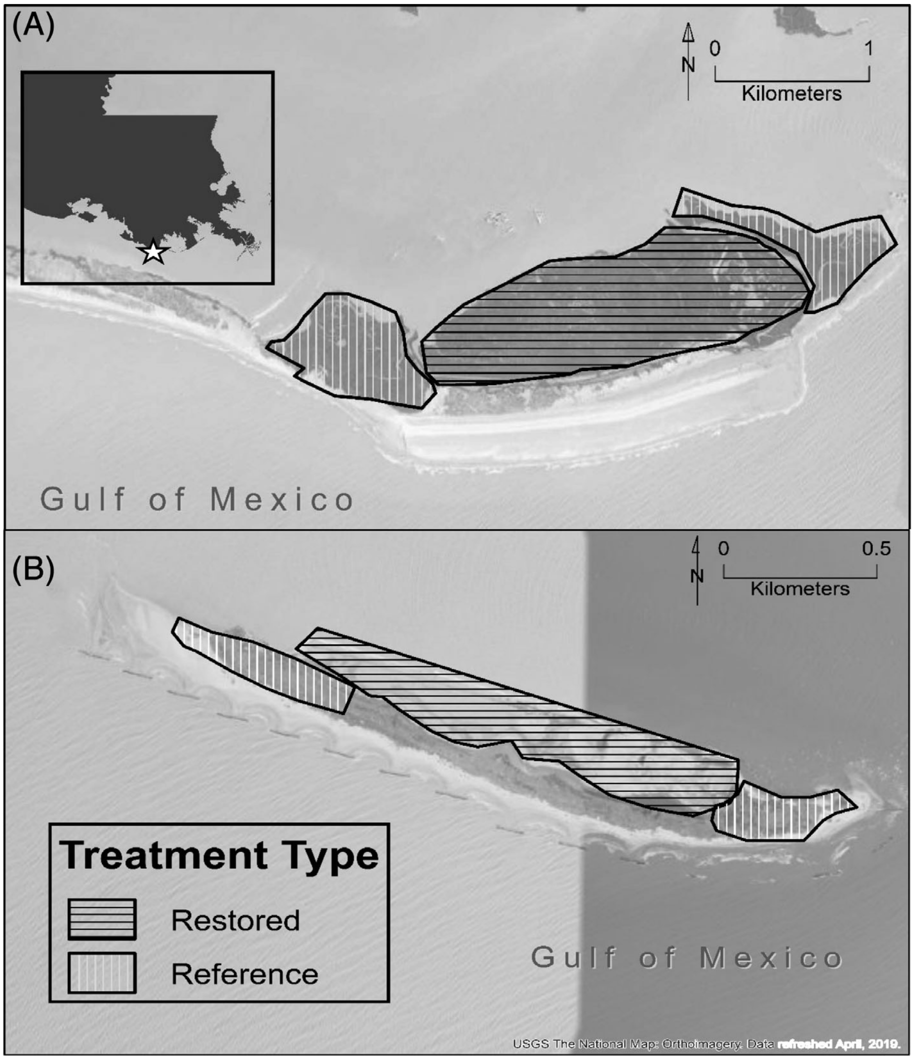

Figure 1 Locations of restoration and reference sites and select study plots on A) Whiskey (restored: 113.6 ha, reference: 75.58 ha) and B) Raccoon (restored: 21 ha, reference: 6.52 ha) islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018.

hereafter *S. alterniflora*), the foundational plant of Louisiana's salt marshes, was planted in the platform in 2010 (Faust 2010). The completed project is recorded by the state as having creating 110 ha of net emergent marsh (Byrnes et al. 2017). In 2013, 24 ha of marsh platform were created on Raccoon (Byrnes et al. 2017). Restoration methods were similar to Whiskey; however, plastic-encased containment dikes were left in place to hold in the marsh sediment, with channels built into the dikes to allow for tidal interchange. The marsh platform on Raccoon was also filled to a higher elevation than on Whiskey (1.1 and 0.8 m, respectively) with the expectation that erosion would bring the platform to an elevation consistent with tidal interchange within years 1–5 of the project (Lindquist 2007; Byrnes et al. 2017). *S. alterniflora* and *Avicennia germinans* (Black Mangrove; hereafter *A. germinans*) propagules were planted in the restoration platform in 2014, with supplemental plantings through 2019. As of 2015, the project was recorded as having created 29 ha of net marsh habitat (Byrnes et al. 2017).

Study Points

Habitat type (hereafter, treatment type: "restored" or "reference") was predefined using state records and satellite imagery. Evaluation of restoration success is often conducted through comparison of the restored area to an intact reference site to establish a target outcome (Moorhead 2013). Locating such comparison sites can be challenging for islands, as an effective reference site should be near the restored site, should be similar in size, and should experience a similar disturbance regime (White & Walker 1997; Ruiz-Jaen & Aide 2005), which can therefore lead to biases when comparing island to mainland sites. In this study, we have attempted to overcome this issue through use of on-island reference sites that had not been subject to significant restoration efforts and which approximated intact marsh (Byrnes et al. 2017).

Restored sites were defined as patches replenished with sediment and vegetation in the 2009 and 2013 restorations. Reference sites were either intact or had not experienced restoration activities since 2000. Restored sites were larger than reference sites on both Whiskey (restored: 113.6 ha, reference: 75.58 ha) and Raccoon (restored: 21 ha, reference: 6.52 ha). Total patch size has been found to affect distribution for some, although not all, of our target species (Benoit & Askins 2002). To account for any potential biases from differing patch sizes, we used index species presence and occupancy instead of density to evaluate our results, as density incorporates abundance and can therefore be more strongly influenced by area-size effects.

Study points were established with the goal of remaining 150–300m apart to ensure independence of detections while maximizing number of sites sampled. Plots were predefined using satellite imagery and then ground-truthed, with points then randomized within plots as much as possible given potential impediments to movement such as tidal canals.

Habitat Covariates

Habitat covariates were measured for 3 years on both islands, in mid-June of 2016 and mid-July of 2017–2018. Two 1-m^2 quadrats per season were selected using randomized coordinates within a 150 × 150–m^2 plot around each ARU (on Raccoon, the original ARU site was used as the central point for quadrat sampling). As sites were highly monotypic, two quadrats per plot per year for 3 years were considered adequate to characterize and capture variation across site covariates. Covariate measurements consisted of % ground cover of each cover type (plant, bare ground, water), % cover of each dominant plant type (succulent [primarily *Salicornia* spp.], *A. germinans*, *S. alterniflora*, woody shrub [primarily *Iva frutescens*]), standing water depth, and plant height (defined as height of tallest plant present in plot). Salinity was assumed to be consistent across the marsh platform and was not measured.

Acoustic Monitoring

Index species presence and occupancy were evaluated via passive acoustic monitoring using Wildlife Acoustics Song Meter SM2 acoustic recording units (hereafter, ARUs). ARUs were mounted to posts >1.5m aboveground and deployed within study plots through the breeding seasons of 2016–2018. Placement was altered as needed within plots across the study period due to habitat changes such as development of unstable substrate. For all years, ARUs were programmed to record 30 minutes a day in 10-minute increments at 15 minutes before sunrise, 15 minutes after sunset, and 2300 (to account for potential nocturnal bird presence).

In 2016 ARUs were deployed in restored and reference marsh plots on both Whiskey ($n = 22$) and Raccoon ($n = 13$). On Raccoon, ambient noise from breeding Laughing Gulls (*Leucophaeus atricilla*) was found to completely obscure recordings. Passive acoustic monitoring was therefore not used on Raccoon, and we added point counts on both islands to overcome this limitation while still facilitating comparison between islands. Acoustic monitoring results presented are therefore based on 3 years of sampling on Whiskey only.

Ambient noise in our system prevented the use of automated species detection, so acoustic monitoring data were manually analyzed. All species detections were made by a single trained observer to minimize identification biases associated with multiple observers. Wildlife Acoustics' Song Scope software was used to visualize recordings as spectrograms. Species identification was supplemented by region-specific recordings for Year 3 and spectrograms from the vocalization database Xeno-canto (https://www.xeno-canto.org/). For each one-day sampling period, species were recorded as present (1) if detected at least once or absent (0) if not detected. As most detections occurred during early morning recordings, time of observation was not considered to be a metric of interest and occurrence data was consolidated by day.

Point Counts

Point counts were conducted between roughly 0730 and 1300 at the same established points used for acoustic monitoring over two breeding seasons in 2017–2018. We conducted five point counts in 2017 and four point counts in 2018. When possible, two observers were dropped at opposite ends of the islands to maximize number of points surveyed at the earliest time possible. Presence/absence was determined via passive counts, with observer waiting 2 minutes and then conducting a 3-minute count of all species seen or heard within 100m of each point. Flushed birds were counted within 100m of each point, as cryptic species such as Least Bitterns (*Ixobrychus exilis*) are often only detectable when disturbed, and wading birds within plots often departed the site as the recorder approached. Flyovers were not counted.

Index Species Presence

Avian presence for both acoustic monitoring and point counts was evaluated through a community index comprised of eight breeding bird species (Table 1). Species used in the index were chosen due to their regular, distinguishable vocalizations during the breeding period and use of salt marsh habitat for breeding. ARUs have been found to have lower detection rates of birds than field observers under some conditions, given that human observers can identify calling birds at greater distances and have the added benefit of visual detection (Digby et al. 2013; Sidie-Slettedahl et al. 2015; Van Wilgenburg et al. 2017). However, in our study ARUs enabled greater temporal coverage, with higher frequency of surveys at times that birds were more likely to vocalize. Given the potential differences between the two methods, we did not quantify comparisons between data generated by acoustic and point count monitoring for Whiskey, instead modeling data from each monitoring method separately and comparing results qualitatively.

Analyses based on species presence and occupancy assume that the site is closed for the period of observation. We defined the breeding season as 1 April through 1 July based on state breeding records for target species. To investigate relationships between index species presence and habitat covariates we used generalized linear modeling with a Poisson distribution function

Table 1 Comparison of mean detections per point per day for an eight-species breeding bird community index in restored and reference salt marsh points on Whiskey ($n = 22$ points) and Raccoon ($n = 13$ points) islands in the Isles Dernieres Barrier Island Wildlife Refuge, Terrebonne Bay, Louisiana. Sampling was conducted via daily passive acoustic recording (ARU), 1 April to 1 July, 2016–2018 on Whiskey only and passive point counts on Whiskey (PC:W) and Raccoon (PC:R), 1 April to 1 July, 2017 (n = 5 sampling days) and 2018 (n = 4 sampling days).

Species	*Scientific Name*	*ARU*	*PC:W*	*PC:R*
Marsh Wren	*Cistothorus palustris*	0.59	0.49	0
Seaside Sparrow	*Ammodramus maritimus*	0.09	0.12	0
Orchard Oriole	*Icterus spurius*	0.33	0.12	0.01
Red-winged Blackbird	*Agelaius phoeniceus*	0.82	0.64	0.05
Willet	*Tringa semipalmata*	0.24	0.12	0.02
Clapper Rail	*Rallus crepitans*	0.76	0.46	0.01
Least Bittern	*Ixobrychus exilis*	0.02	0.11	0.08
Green Heron	*Butorides virescens*	0.02	0.08	0.05

for both monitoring types. Habitat covariates were averaged by year, but not across years, to account for potential site changes across the study period and were scaled for a mean of 0 to facilitate comparison of effect sizes. To check for multicollinearity, we first fit a full model of all covariates, then removed one of each pair of covariates with a variance inflation factor (VIF) greater than five. Covariates were then fit to models based on a priori hypotheses.

For all regression and occupancy models, model selection was made using Akaike's Information Criterion (AIC*c*; Burnham et al. 2011). We considered best supported models those with $\Delta AICc \leq 2$, indicating models have substantial support. If two or more models had $\Delta AICc \leq 2$ we selected the most parsimonious model ($\Delta AICc = 0$) as being the final model. All analyses were conducted in R v.2.15.0 (R Core Team 2018).

Site Occupancy

Differences in site occupancy on Whiskey were modeled using detection data from passive acoustic monitoring only. We focused on six species with varying degrees of reliance on salt marsh habitat: Seaside Sparrows (*Ammospiza maritima*), Red-winged Blackbirds (*Agelaius phoeniceus*), Orchard Orioles (*Icterus spurius*), Clapper Rails (*Rallus crepitans*), Marsh Wrens (*Cistothorus palustris*), and Least Bitterns. We omitted points < 200 m apart, ending with $n = 16$ points ($n = 8$ for each habitat type) and $n = 92$ survey days per year. Detection histories were assembled by compiling daily occupancy at each point per survey day.

We estimated detection probabilities and the effects of habitat covariates on individual species occupancy through dynamic single-species occupancy modeling (MacKenzie et al. 2003; Nichols et al. 2007) using the R package *unmarked* (Fiske & Chandler 2011). Dynamic occupancy modeling uses binomial detection data to estimate the proportion of sites occupied by the focal species (ψ), the probability of detecting the focal species if it is present at a site (p), and site colonization and extinction. These parameters can be estimated across multiple seasons, and habitat and temporal covariates can be modeled to investigate their influence on both ψ and p (MacKenzie et al. 2003). Detection data are stored in a three-dimensional detection history data frame y_{ijt}, where i represents site number, j denotes the replicate survey within a season, and t the season. If at least one individual is detected during a survey $y_{ijt} = 1$; if there is no detection, $y_{ijt} = 0$.

For each species, we first fit an intercept-only (constant) model to estimate ψ and p assuming no influence of covariates. To determine the effect of temporal covariates on p both within and among seasons over the 3 years of our study, we modeled each combination of year and Julian date as observational covariates with ψ, colonization, and extinction held constant. Covariates from the top model from this set were taken to influence p and were included as observational covariates in all subsequent models. Site colonization and extinction were then modeled as both varying by year and as constant to evaluate if site-wide colonization and extinction probability changed across the 3 years of our study. Given the short duration of our study and the homogeneity of our sites we did not expect within-site changes to our habitat covariates across the 3 years of our study, and therefore did not model the effects of habitat covariates on colonization or extinction likelihood. We then included extinction and colonization as either constant or varying by year for all subsequent models based on the top model from the set.

Habitat covariates were modeled for their influence on ψ by first fitting an intercept-only model, then sets of models with covariate combinations based on a priori hypotheses for each species. Dynamic occupancy models estimate the initial occupancy directly, and we derived occupancy and SE for the subsequent years with 1,000 bootstrapped replicates using the *smoothed* function in *unmarked* to estimate yearly ψ and SE among sampled sites (Weir et al. 2009). To evaluate ψ by treatment type and covariates, we derived predicted occupancy from the top model using the *predict* function in *unmarked*. Occupancy for years $t + 1$ were estimated using the formula

$$\psi_{t+1} = \psi_t \times (1 - extinction) + (1 - \psi_t) \times p_t$$

for Year 2 and

$$\psi_{t+2} = \psi_{t+1} \times (1 - extinction) + (1 - \psi_{t+1}) \times p_{t+1}$$

for Year 3.

For all species, we predicted that ψ and p would not vary across years and that p would decrease with ordinal date, as species vocalizations typically decline as the breeding season progresses. We predicted that Seaside Sparrows, Marsh Wrens, and Least Bitterns would have higher occupancy of restored sites and that Orchard Orioles, Red-winged Blackbirds, and Clapper Rails would have higher occupancy of reference sites, based on known species-specific habitat requirements (Dennis 1948; Leonard & Picman 1987; Rush et al. 2009, 2010; Elphick et al. 2015).

Results

Habitat Covariates

On Whiskey, primary cover type (plant, water, bare ground) did not differ by treatment type (Fig. S2a). Mean % covers of woody shrub (0, SD = 0) and succulent (0.05, SD = 0.15) plant types were low and were dropped from analyses. *S. alterniflora* and *A. germinans* were correlated to each other ($r = -0.7$, $p < 0.001$) and to treatment type (Fig. S1a), with restored sites having significantly higher *S. alterniflora* and lower *A. germinans* % cover than reference sites (see Fig. 2A & 2B). Treatment type was therefore used as a proxy covariate for these plant types in subsequent regression and occupancy models. Water cover was also removed due to its correlation to depth ($r = 0.6$, $p < 0.001$), leaving covariates treatment type (reference or restored), % plant cover, % bare cover, plant species diversity, standing water depth, and plant height.

On Raccoon, % cover of both plant and ground were correlated to each other ($r = -0.9$, $p < 0.001$) and to treatment type (Fig. S1b), with reference sites having significantly greater % plant cover and less bare ground than restored sites (see Fig. 2C & 2D). Only treatment type was used in subsequent regression and occupancy models. Treatment type and foundational plant types were not correlated (Fig. S1b). As on Whiskey, % water cover was removed due to its correlation to water depth ($r = 0.83$, $p < 0.001$), leaving covariates treatment type (reference or restored), % plant cover, % *S. alterniflora* cover, % *A. germinans* cover, plant species diversity, standing water depth, and plant height.

Acoustic Monitoring

Passive monitoring via ARUs on Whiskey resulted in recordings from $n = 3{,}513$ sample days (days * points) and $n = 105{,}390$ recording minutes (Table 1). Mean index species presence was 2.94 species/site (SD = 1.24) over the 3 years of our study (Table 2).

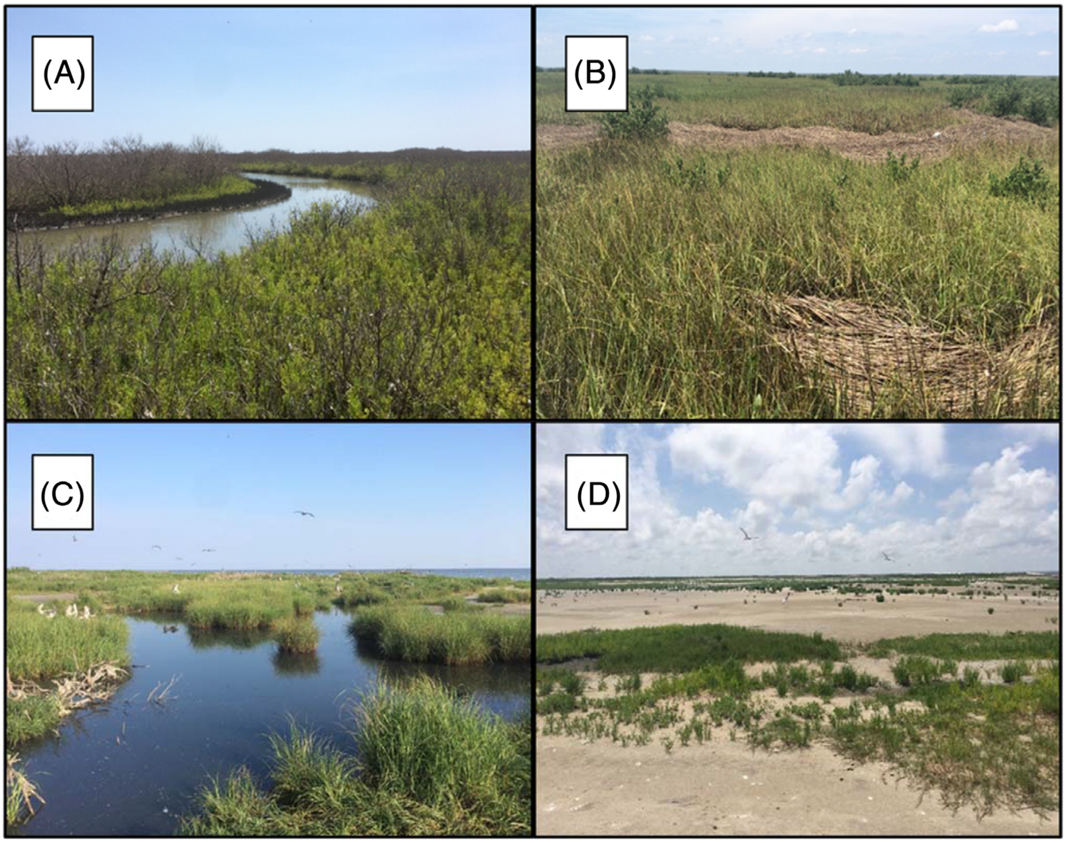

Figure 2 Comparison of treatment types on Whiskey and Raccoon islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018. A) Reference site, Whiskey; B) Restoration site, Whiskey; C) Reference site, Raccoon; D) Restoration site, Raccoon.

Table 2 Comparison of mean species richness and standard deviation among islands, sampling types (passive acoustic monitoring [ARU] and point counts), and treatment type (restoration [Res], reference [Ref], and mean [Av]) on Whiskey and Raccoon islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018.

	ARU			*Point Count*		
Island	*Res*	*Ref*	*Av*	*Res*	*Ref*	*Av*
Whiskey	3.11 ± 1.29	2.79 ± 1.18	2.94 ± 1.24	1.91 ± 1.40	2.38 ± 1.58	2.14 ± 1.51
Raccoon	—	—	—	0.31 ± 0.58	0.11 ± 0.32	0.22 ± 0.49

Table 3 Results from generalized linear models to evaluate effects of habitat covariates on avian species richness derived from passive acoustic sampling of breeding birds in restored and reference habitat on Whiskey Island in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018.

Model	k	*AICc*	Δ*AICc*	w_i
Treatment type + % plant cover + water depth + plant height	5	12,249.13	0.00	0.85
Treatment type + % plant cover + % bare ground + plant height + plant species diversity + water depth	7	12,252.77	3.64	0.14
Treatment type + % plant cover	3	12,258.31	9.17	0.01
Null	1	12,341.95	92.82	0.00

We fit $n = 12$ models with varying combinations of covariates to evaluate effects of habitat covariates on index species presence. One model was best supported (Table 3). Based on the top model ($\Delta \text{AIC}c = 0.00$), index species presence was higher in restored sites than treatment sites ($\beta = 0.16 \pm 0.01$; $p < 0.001$). Species presence decreased with % plant cover ($\beta = -0.08 \pm 0.01$; $p < 0.001$), increased with plant height ($\beta = 0.03 \pm 0.01$; $p = 0.03$), and showed a weak positive response to standing water depth ($\beta = 0.02 \pm 0.01$; $p = 0.13$).

Point Counts

Detection of index species and species presence was consistently lower on Raccoon than on Whiskey for point-count data (Tables 1 & 2). Because of the low number of count days, we fit fewer covariates per model than for the acoustic monitoring models, resulting in a set of $n = 9$ models tested for each island. For Whiskey, no model showed substantial support, with little difference between the intercept-only ($\Delta \text{AIC}c = 0.00$) and all other models (Table 4). For Raccoon, the best supported model (Table 4) indicated that index species presence was negatively associated with the restored site ($\Delta \text{AIC}c = 0.00$; $\beta = -1.02 \pm 0.51$; $p = 0.05$), but there was not a substantial difference between the best supported model and the null model ($\Delta \text{AIC}c = 2.16$).

Site Occupancy

Due to ARU malfunctioning we did not obtain recordings for all sites for each year, instead collecting recordings from 15 points in 2016, 14 points in 2017, and 12 points in 2018. Occupancy was modeled with $n = 3{,}772$ combined detection days (days * points) for years 1–3. Year 1 occupancies derived from the intercept-only models were too high to enable any informative comparisons of habitat association for Clapper Rails ($\psi = 1$; $p = 0.75$), Marsh Wrens ($\psi = 1$; $p = 0.55$), Orchard Orioles ($\psi = 0.94$; $p = 0.37$), and Red-winged Blackbirds ($\psi = 1$; $p = 0.78$). We removed these species from further occupancy analysis and report results only for Least Bitterns and Seaside Sparrows.

Seaside Sparrows were detected in 9 of 16 sites over the 3 years of our study, resulting in a naïve ψ of 0.56. The temporal covariate of year was found to influence p (Table S1), and we retained year in all subsequent models. The model for colonization and extinction held constant ($\Delta \text{AIC}c = 0.83$) versus allowed to vary by year ($\Delta \text{AIC}c = 0$) did not differ substantially and we

Table 4 Results from generalized linear models ($\Delta \text{AIC}c < 3$) to evaluate effects of habitat covariates on avian species richness derived from point counts of breeding birds in restored and reference habitat on Whiskey and Raccoon Islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 1 April to 1 July 2017 ($n = 5$ sampling days) and 2018 ($n = 4$ sampling days).

Island	*Model*	k	*AICc*	Δ*AICc*	w_i
Whiskey	Treatment type	2	379.74	0.0	0.35
	Intercept-only	1	380.42	0.68	0.25
	% plant cover	2	381.72	1.98	0.13
	Plant height	2	382.50	2.76	0.09
	Treatment type + water depth + % plant cover	4	381.89	2.16	0.12
Raccoon	Treatment type	2	113.33	0.00	0.45
	Treatment type + water depth	3	115.43	2.10	0.16
	Intercept-only	1	115.93	2.60	0.12

Table 5 Results (untransformed coefficient estimates [β], standard errors [SE], AIC*c*, ΔAIC*c*, and AIC*c* weights) from intercept-only and highest ranked occupancy models to evaluate effect of habitat covariates on Seaside Sparrow (*Ammospiza maritima*) and Least Bittern (*Ixobrychus exilis*) occupancy of restoration and reference sites on Whiskey Island, Louisiana, 2016–2018.

Species	*Model*	AIC_c	$\Delta\ AIC_c$	w_i
SESP	Treatment type + % plant cover + plant height	1,651.79	0.00	0.54
	Treatment type	1,653.39	1.60	0.24
	Treatment type + % plant cover + plant height + water depth	1,653.79	2.00	0.20
	Intercept-only	1,666.22	14.73	0.00
LEBI	Treatment type + % plant cover + plant height	760.43	0.00	0.83
	% plant cover	766.19	5.76	0.05
	Intercept-only	766.52	6.09	0.04
	% plant cover + plant height	767.07	6.64	0.03

held colonization and extinction as constant in all subsequent models to reduce model parameterization.

Three models showed substantial support for influence of habitat covariates on ψ ($\Delta AICc \le 2$; Table 5). Of these, the most parsimonious model ($\Delta AICc = 0$) indicated that Seaside Sparrow occupancy was highest in restored sites ($\beta = 27.21 \pm 69.9$), was positively associated with % plant cover ($\beta = 8.08 \pm 20.0$), and was negatively associated with plant height ($\beta = -6.40 \pm 19.7$). Detection was higher in 2016 ($p = 0.28 \pm 0.02$) than in 2017 ($p = 0.19 \pm 0.02$) or 2018 ($p = 0.19 \pm 0.02$). By year, ψ derived from the final model was 0.37 ± 0.12 for 2016, 0.50 ± 0.13 for 2017, and 0.37 ± 0.13 for 2018. At mean transformed % plant cover and height, predicted ψ differed for restored ($\psi = 0.99 \pm 0.1$) versus reference sites ($\psi < 0.01 \pm 0.001$).

Least Bitterns were detected at least once at 12 of 16 sites over the 3 years of our study, resulting in a naïve ψ of 0.81. Year and date were both included in the most parsimonious model of influence of temporal covariates on p (Table S1) and were retained in all subsequent models. The model for colonization and extinction held constant had higher support ($\Delta AICc = 0.00$) than the model that allowed these parameters to vary by year ($\Delta AICc = 2.20$) and we held colonization and extinction as constant in all subsequent models.

One model was best supported for influence of habitat covariates on ψ (Table 5), with Least Bittern ψ higher in restored sites ($\beta = 48.51 \pm 52.84$), negatively associated with % plant cover ($\beta = -20.88 \pm 23.38$), and positively associated with plant height ($\beta = 3.62 \pm 7.63$). Derived p was 0.03 ± 0.03 in 2016, 0.08 ± 0.01 in 2017, and 0.02 ± 0.01 in 2018. Derived ψ across the entire study area was 0.50 ± 0.23 in 2016, 0.57 ± 0.13 in 2017, and 0.63 ± 0.14 in 2018. At mean transformed % plant cover and height, predicted Least Bittern ψ by site was higher in restored ($\psi = 1.0 \pm 001$) than in reference ($\psi < 0.01 \pm 0.001$) sites.

Discussion

Restoration success as defined by avian species presence was mixed between the two islands. Avian species presence was greater in Whiskey's restored site than in the reference sites, and occupancy of both specialist and generalist breeding bird species was similarly greater across the restoration site. Habitat characteristics within the restored site were largely consistent with those of an intact salt marsh, with dense stands of Spartina interspersed with the ephemeral pools and tidal canals characteristic of an intact salt marsh. Results were different on Raccoon, with low index species presence, minimal vegetation, and minimal standing water within the restored site. Our results suggest that restoration efforts were largely successful in creating functional avian salt marsh habitat on Whiskey, but were not equally successful on Raccoon, and that the land created on Raccoon could not be characterized as emergent salt marsh within the time of our study.

Whiskey Island

Site was the most important predictor of species presence on Whiskey. Our findings of greater species presence in the restored site are inconsistent with comparable studies, which have found greater avian use of reference versus restored or disturbed marsh (Melvin & Webb Jr. 1998; Seigel et al. 2006; Desrochers et al. 2008; Rush et al. 2009). The results from our study were likely driven by the strong vegetative associations of the two treatment types on Whiskey, with *S. alterniflora* dominant in the restoration site and *A. germinans* in reference sites. These two foundational plants confer structural differences, including subsurface root density and understory shading (Yando et al. 2018), that have the potential to alter community composition across trophic levels (Scheffel et al. 2018; Loveless & Smee 2019). This variation in vegetative structural properties may have influenced marsh bird use of our sites, resulting in the observed difference in species presence between treatment types.

Index species presence on Whiskey also decreased with % plant cover and showed a weak association with standing water depth and plant height. Plots with lower % plant cover had higher interspersion of plant, water, and bare ground cover, while higher standing water depth was typically representative of ephemeral pools or tidal canals. These characteristics are all indicative of plots with greater habitat heterogeneity. Classic ecological theory on community structure states that species richness should increase with increasing habitat heterogeneity (MacArthur 1958; Levin & Paine 1974), and this has been empirically demonstrated in other marsh bird communities (Brawley et al. 1998; Benoit & Askins 2002; Rush et al. 2009). Restored marshes often fail to approximate natural marsh heterogeneity due to uniform, dense planting of *S. alterniflora* and other marsh grasses (Seigel et al. 2006; Desrochers et al. 2008). Our results suggest that the inclusion of tidal canals and ponds in the restored marsh on Whiskey contributed to habitat quality by breaking up monotypic vegetation stands and promoting habitat heterogeneity throughout the site.

Results from occupancy modeling varied by species. Occupancy of four index species did not differ with treatment, with Marsh Wrens, Red-winged Blackbirds, Orchard Orioles, and Clapper Rails common across all sites. The two species that did show variation in occupancy, Seaside Sparrows and Least Bitterns, were both strongly associated with the restored site. Both of these species specialize in grass-dominant marshes (Melvin & Webb Jr. 1998; Desrochers et al. 2008; Jobin et al. 2013), and our results suggest that they may be unable to diversify habitat use to include *A. germinans*-dominant marshes, as did the more generalist marsh birds in our study. These findings likely also contributed to the greater mean index species presence in the restoration site, as no species investigated displayed an opposite pattern of preference for the reference site.

Seaside Sparrows displayed a positive response to % plant cover and a negative response to plant height, possibly using dense stands of *S. alterniflora* to avoid nest predation. Conversely, Least Bittern occupancy was negatively associated with % plant cover and showed a weak positive association with plant height; these results are consistent with findings that Least Bittern select for nest sites with relatively low plant density and a lower ratio of plant cover to water (Lor & Malecki 2006). It should be noted that standard errors of the top models may have been inflated by the strong association of the two avian species with restoration treatment, a common phenomenon with maximum likelihood estimates of parameter error based on data with almost complete separation between levels of class variables. Accuracy of parameters derived from occupancy modeling is highly responsive to sample size (MacKenzie et al. 2009) and it is also possible that the large standard errors in our models were due to the low number of sites per treatment. Despite these potential limitations of our models, our results indicate that occupancy of both species was strongly dependent upon the presence of *S. alterniflora*.

Raccoon Island

The low index species presence and lack of breeding marsh species across both treatment types on Raccoon indicate that neither site can be considered quality salt marsh habitat based on avian presence. We did not see breeding passerines in any treatment type on Raccoon over the 2 years of counts. Although we regularly saw songbirds overwintering on the island, the presence of wading and seabirds in the breeding season likely limited the suitability of reference sites for breeding passerines. No marsh birds of any type were seen breeding in the restored site, and the habitat was largely unused for foraging by marsh birds on the island.

Survey Limitations

The different survey methods used on Whiskey Island resulted in different detection frequency and model results, likely a result of the limitations of both methods. While passive acoustic monitoring results in greater sampling effort, which can lead to greater model sensitivity to detect habitat effects, ARU monitoring is restricted to vocalizing species. Additionally, ARU detection is impeded by ambient noise, particularly wind (Bardeli et al. 2010; Digby et al. 2013). This may have reduced our ability to detect vocalizing birds at commonly obscured frequencies, such as Least Bitterns, which vocalize in the range of ambient wind noise. Point counts have been found to have higher detection rates than ARU data for cryptic species (Klingbeil & Willig 2015), but low sampling effort and inability to visit our sites during peak vocalizing times likely resulted in fewer detections and significantly lower estimates of species richness across our study. The differences in model results between the ARU and point count data likely stem from the limited ability of the point-count data to detect differences in the reduced dataset. Our results suggest that four visits per season may not be effective for evaluating associations between avian species presence and salt marsh habitat covariates when recommended point-count protocol cannot be followed.

Recommendations for Barrier Island Salt Marsh Restoration

Restored sites on these two islands did not perform equally as wildlife habitat, with greater index species presence on Whiskey than on Raccoon. The lack of both vegetation and standing water in the platform on Raccoon in 2016–2018 shows that the expectation that the platform would naturally decline in elevation within 1–5 years was not met during the course of our study, although our study was concluded within the period of 5 years postrestoration. Timing is an important consideration in marsh restoration projects, and it is possible that the restoration site would have met the necessary elevation change had our research extended past 2018. Earlier marsh restorations on Whiskey and neighboring Trinity Island were found to have similarly failed to create marsh habitat due to excessive marsh platform height (Penland et al. 2003). The higher elevation of the marsh platform and the retention of the postrestoration containment dikes on Raccoon likely prevented tidal interchange within the marsh platform, thereby inhibiting natural marsh hydrology (Penland et al. 2003; Campbell et al. 2005). Without tidal inflow, salt marsh vegetation was unable to colonize, and the restoration failed in terms of creation of marsh habitat. The disparity in results from Whiskey and Raccoon suggest that restoration planning focused primarily on the goals of minimizing erosion and creating land is not necessarily conducive to the creation of viable salt marsh habitat, and that target habitat characteristics should be taken into consideration during restoration planning. Following the conclusion of this study, the containment dikes were removed from Raccoon, increasing tidal interchange at the platform edge. Enhanced growth of marsh vegetation was observed within the restored site, but the implications for marsh avifauna has not been evaluated.

Index species presence and occupancy of marsh specialists were both higher in *S. alterniflora*-dominant restored marsh on Whiskey. In the Gulf of Mexico, *A. germinans* distribution is held in check by periodic freezes; however, it is expanding northward as average winter temperatures warm (Perry & Mendelssohn 2009; Osland et al. 2013). Where the two species coincide, *A. germinans* has the potential to outcompete *S. alterniflora* (Perry & Mendelssohn 2009), and *S. alterniflora* marshes are predicted to transition to forest across much of the Gulf of Mexico within the current century (Osland et al. 2013). *A. germinans* is commonly planted in restored salt marsh under the assumption that its extensive root network better facilitates sediment retention; however, *S. alterniflora* has been demonstrated to be superior for restoration purposes due to its rapid growth and higher capabilities for soil retention (Yando et al. 2019). Our finding that the spread of *A. germinans* within Gulf Coast salt marshes may have negative implications for some avian salt marsh obligate species provides further support for *S. alterniflora* as the optimal plant for Gulf of Mexico salt marsh restoration.

Acknowledgments

We would like to thank the many volunteers that helped with the field work component for this project, particularly C. Pavlik, W. Premeaux, and K. Hucks. K. Wakefield helped with first-year transportation and data collection. D. Leblanc and E. Johnson assisted with avian identification. J. Peacock and R. Dobbs helped with transportation and specialized knowledge of field sites. B. Ralston and M. Lamont provided comments on the manuscript. We would also like to thank the staff of Louisiana Universities Marine Consortium (LUMCON) for their logistical assistance throughout the project. This material is based upon work supported by the National Science Foundation Graduate Research Fellowship Program under Grant No. 1650114. Major funding for this research was provided by Louisiana Department of Wildlife and Fisheries State Wildlife Grant F15AF01267. Additionally, this project was paid for in part with federal funding from the Department of the Treasury through the Louisiana Coastal Protection and Restoration Authority's Center of Excellence Research Grants Program under the Resources and Ecosystems Sustainability, Tourist Opportunities, and Revived Economies of the Gulf Coast States Act of 2012 (RESTORE Act). The statements, findings, conclusions, and recommendations are those of the author(s) and do not necessarily reflect the views of the Department of the Treasury, CPRA, or The Water Institute of the Gulf. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Supporting Information

The following information may be found in the online version of this article:

Figure S1 Mean % cover (plant, bare ground, and standing water) in reference (ref) and restored (res) barrier island salt marsh sites on a. Whiskey and b. Raccoon Islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018.

Figure S2. Mean % cover of *Spartina alterniflora* and *Avicennia germinans* in restored and intact barrier island salt marsh sites on a. Whiskey and b. Raccoon Islands in the Isles Dernieres Wildlife Refuge, Terrebonne Bay, Louisiana, 2016–2018.

Table S1. Results from dynamic occupancy models evaluating effect of temporal covariates year and Julian date on detection probability of Seaside Sparrows (SESP; *Ammodramus maritimus*) and Least Bitterns (LEBI; *Ixobrychus exilis*) on Whiskey Island, Louisiana, 2016–2018.

Coordinating Editor: Robin Hale

Received: 18 February, 2020; First decision: 15 March, 2020; Revised: 10 June, 2020; Accepted: 12 June, 2020