Marsh terraces in coastal Louisiana increase marsh edge and densities of waterbirds

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ABSTRACT. We evaluated the influence of marsh terracing on waterbirds in Louisiana's Chenier Plain. Terracing is a novel technique used to slow coastal marsh loss. Terracing increases marsh edge and is assumed to slow erosion, decrease pond depth, and encourage vegetation production. From April to September 2005, we monitored waterbirds in paired terraced and unterraced ponds in three sites dominated by *Spartina patens*. We additionally sampled submerged aquatic vegetation (SAV) biomass, nekton density, and water quality. Waterbird density and species richness 3.8 and 1.4 times greater, respectively, in terraced ponds. By foraging guild, probers, aerial foragers, and dabbling foragers were more abundant in terraced ponds. Waders were frequently more abundant in terraced ponds. Diver density did not differ significantly between pond types. Terracing increased marsh edge in ponds 3.5 times. Nekton and SAV were more abundant in edge habitat than in open water, but water quality, water depth, SAV, and nekton did not differ significantly between pond types and did not influence bird density. Bird densities were higher in ponds with greater proportions of marsh edge, possibly because they are morphologically constrained to forage in shallow water or because of abundant food near edges.

Keywords: habitat interspersion; nekton; submerged-aquatic vegetation; waterbird communities; wetland restoration;

INTRODUCTION

Conservation of coastal marshes is important because they provide many unique ecosystem services and constitute valuable wildlife habitat. Louisiana contains 40% of the remaining coastal marsh in the continental United States (Field et al. 1991), and provides critical habitat for waterbirds (Michot 1996; Esslinger and Wilson 2001). However, many of these waterbirds are declining regionally from habitat loss. For example, 27 species classified as species of high or moderate concern by the Waterbird Conservation Council (Kushlan et al. 2002) are regularly seen in brackish marshes along the Gulf Coast. In 1984, coastal Louisiana contained more nesting colonies of seabirds and wading birds than any other state in the southeast US (Keller et al. 1984). However more recently, Louisiana colonies for most species have declined (Michot et al. 2003; Green et al. 2006). Wetlands in coastal Louisiana have been in rapid decline, accounting for 80% of US coastal wetland loss from 1950 to 1994 (Boesch et al. 1994), potentially imperiling dependent migratory birds.

Coastal marsh loss in Louisiana is considerable and results from conversion of marsh to shallow open water. These marshes can be divided into two major sub-regions, the Deltaic Plain of the Mississippi River, and the Chenier Plain, directly to the west. The two regions differ in process of formation, geomorphology, hydrology, and marsh loss. Marshes in the Chenier Plain, where this study was conducted, are formed over riverine sediments that have been reworked and deposited through marine action, creating new wetlands (Penland and Suter 1989). Unlike deltaic marshes, direct sediment deposition from riverine flow is unimportant in wetland creation in the Chenier Plain. Once created, emergent wetlands in the Chenier Plain depend upon vertical accumulation to offset sea-level rise and subsidence (Foret 2001). Vertical accretion proceeds via peat accumulation, which relies on nutrient supplements from sporadic over-wash events during large storms and autochthonous production (Foret 2001; Turner et al. 2006). Prior to 1956, these processes were sufficient to sustain uninterrupted expanses of emergent marsh throughout much of the Chenier Plain (Barras et al. 1994). Since then, dredging for navigation and petroleum exploration has directly or indirectly caused extensive interior marsh loss (Byrnes et al. 1995). Marsh loss rates in the Chenier Plain were 16.3 km^2/year from 1978–2000 (Barras et al. 2003). Interior loss is often attributed to salt-water intrusion or prolonged flooding that causes vegetation die-off in isolated hot spots. This creates open water areas approximately 1-m deep that rarely drain because of lengthy, convoluted hydraulic connections to the Gulf of Mexico. Once formed, these ponds increase in size even after vegetation die-off ceases, presumably because of soil erosion around pond margins. This phenomenon may be exacerbated by a variety of factors, including sea level rise and sediment starvation from channelization of the Mississippi and other rivers (Boesch et al. 1994; Turner 1997).

Pond terracing is a novel marsh restoration technique developed in response to open water conversion of interior marshes in the Chenier Plain (Underwood et al. 1991; Steyer 1993; Rozas and Minello 2001). Since 1990, 2,200 ha of Chenier Plain marsh have been restored using terraces (Stead and Hill 2004), and an evaluation of terraced marsh as habitat for dependent waterbirds is overdue. Terraces are discontinuous, narrow strips of created marsh. They are formed of dredge material stabilized by emergent vegetation such as Spartina alterniflora (Underwood et al. 1991; Steyer 1993; Rozas and Minello 2001). Sediment for terrace building usually is taken from pond bottoms and is piled using a backhoe, creating borrow pits within ponds. Terraces are thought to reduce wave energy and dampen the erosive force of water, potentially slowing lateral erosion of surrounding marsh edges (Underwood et al. 1991; Boesch et al. 1994). Additionally, reduction in wave energy may encourage sediment settling and increase water clarity, resulting in increased production of submerged aquatic vegetation (SAV). Increased sediment settling additionally may decrease pond depths, increase soil fertility, and provide a more hospitable environment for the expansion of emergent vegetation. Terraces do not return an area to a close approximation of its condition prior to marsh loss and thus are classified as management by some (NRC 1992) but as restoration by others (Boesch et al. 1994; Stead and Hill 2004; Feagin and Wu 2006).

Terracing is thought to improve wildlife habitat by increasing the amount of edge (the boundary between emergent vegetation and open water) within a pond (Rozas and Minello 2001). Shallow marsh edge frequently has been noted as a highly productive zone for waterbird forage such as plants, nekton, and invertebrates (Gosselink 1979; Peterson and Turner 1994; Chesney et al.

2000; Minello and Rozas 2002). Adding terraces to open water ponds can also increase habitat interspersion (mixing of open water and vegetated marsh habitats) by increasing the amount of emergent vegetated edge in open water ponds. A relationship between interspersion and wetland use by waterbirds has been noted for marshes elsewhere (Weller and Spatcher 1965; Mack and Flake 1980; Kaminski and Prince 1981; Murkin et al. 1982; Fairbairn and Dinsmore 2001). Thus, increasing the proportion of marsh edge around the perimeter and interior portions of marsh ponds could improve habitat quality for waterbirds. However, experimental tests of this prediction are few, and we are unaware of any in coastal marshes.

Previous studies have evaluated some effects of terracing. Steyer (1993) showed that terracing at Sabine National Wildlife Refuge increased primary productivity through the creation and expansion of emergent marsh. Terracing also increased the amount of emergent marsh to water edge, although terrace fields do not mimic natural marsh in shape or habitat complexity (Feagin and Wu 2006). Four studies suggested that terraced edge had more nekton biomass than open water controls and changed nekton community composition (Rozas and Minello 2001; Thom et al. 2004; Rozas et al. 2005; La Peyre et al. 2007). Cannaday (2006) concluded that terracing increased SAV abundance at both marsh edge and whole-pond scales. The efficacy of terraces at improving habitat quality for waterbirds, which depend heavily on coastal marshes, has not been evaluated.

Efficacy of terraces at improving marsh functions can be measured at two scales. First, effects can be compared between areas directly adjacent to terraces edges (restoration condition) and open water habitat far from any edge (unrestored condition). Additionally, effects could be evaluated at the whole-pond scale. Thus far, only Cannaday (2006) has conducted a whole-pond analysis of terrace effects (on SAV communities) using multiple sites. We evaluated the effect of pond terracing on waterbird habitat by comparing waterbird density and species richness at microhabitat and whole-pond scales between ponds restored with terraces and unrestored ponds. We also evaluated whether bird density varied by foraging guild between habitats. Finally, we compared SAV, nekton, and water quality at the microhabitat and whole-pond scales in restored and unrestored ponds, and evaluated whether terracing effects on nekton and SAV were influencing waterbird densities.

METHODS

Study Sites

Study sites were in coastal southwestern Louisiana within the Chenier Plain (Fig. 1), which extends from west of Vermilion Bay, Louisiana, to High Island, Texas. It consists of shore-parallel, stranded inland beach ridges separated by broad areas of low-elevation marsh. At three sites, we monitored one terraced (treatment) and one nearby unterraced pond (control). Each pair was hydrologically distinct from other pairs. However, all pairs were within the same drainage basin and had similar hydrologic regimes.

Study ponds were limited to those in marsh dominated by Spartina patens, an intermediate or brackish marsh species (Chabreck 1970). Ponds were limited to those known to have been emergent marsh prior to 1956, based on land change maps (Barras et al. 1994). Terraces were also limited to those sufficiently mature to have established emergent vegetation.

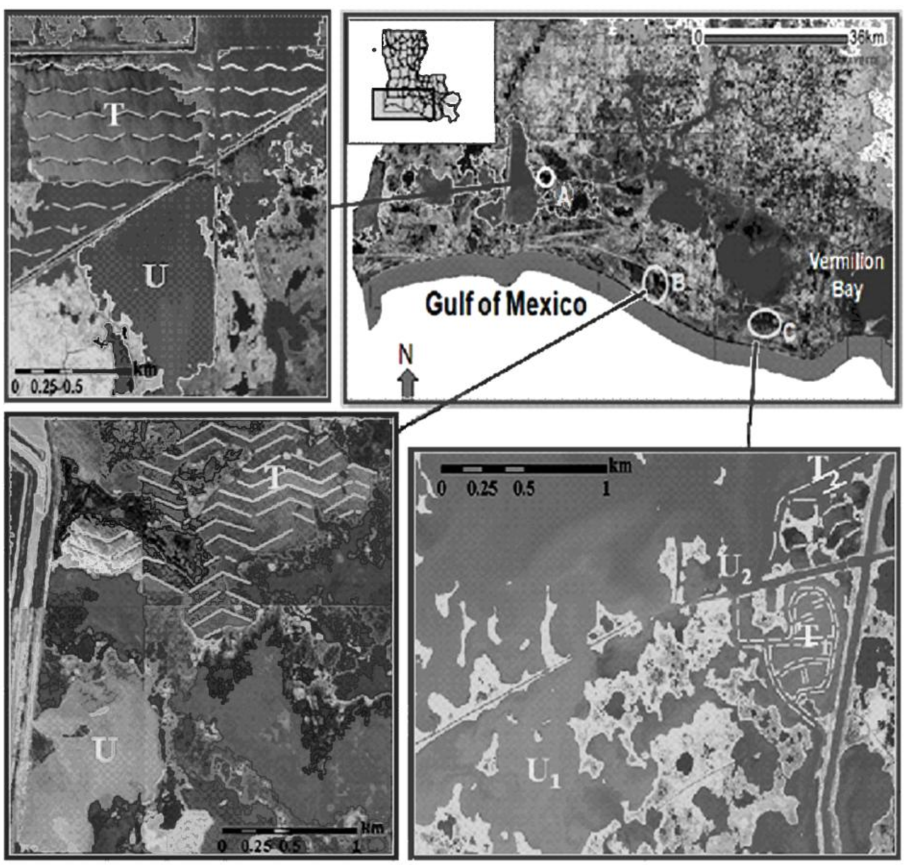


Fig. 1 Study sites locations in Louisiana's Chenier Plain, spring and summer 2005. A: Sweet Lake; B: Rockefeller SWR; C: Vermilion. Terraced pond = T; Unterraced pond= U

Only three sites in the Chenier Plain met all criteria. They were 1) Sweet Lake (UTM 1984 Zone 15N, coordinates: 480933 East, 3312269 North, owned by Sweet Lake Gas and Oil Co. and Miami Corporation), 2) an impounded area in Rockefeller State Wildlife Refuge (SWR) (UTM 1984 Zone 15N, coordinates: 523597 East, 3284941 North, Unit 4, owned by Louisiana Department of Wildlife and Fisheries), and 3) Vermilion (UTM 1984 Zone 15N, coordinates: 551870 East, 3275345 North, owned by Vermilion Corporation) (Fig. 1). The terraces were constructed in 2001 at Sweet Lake, in 2002 at Rockefeller, and in 2003 at Vermilion.

In the initial stages of our research, we sampled a second pond pair within the Vermilion site. We subsequently decided these ponds were not hydrologically distinct from the first pair. Data collected from the second Vermilion pond pair were analyzed as replicates within the Vermilion site (see statistical methods). These two pond pairs are called Vermilion 1 and Vermilion 2.

Survey Methods

Surveys were conducted once a month from 29 April 2005 through 3 September 2005. After the passage of Hurricane Rita (24 September 2005), methods had to be modified, and post-hurricane results are not included here. Each pond contained multiple survey plots. Prior to the first survey on a site, locations for plots were randomly selected, and boundaries were marked with PVC pipe. Plots were originally designed to be 12 ha, but some plots were smaller because of geographical constraints. Thus, the 18 plots ranged from 4 ha to 12 ha (mean plot size = 9.6, SD = 3.3 ha). One plot per pond was sampled during each survey session; different plots were sampled each session. Ultimately, sampling effort was even among plots. Each survey session included environmental, SAV, nekton, and bird sampling.

Environmental Sampling. Wind speed and air temperature were recorded prior to each bird survey, using an EA-3010TWC anemometer (La Crosse Technology, 1116 South Oak Street, La Crescent, MN). The other environmental variables were measured following each bird survey in two microhabitat types: within 5 m of a vegetated marsh edge, and open water > 25 m from edge. Edge sampling was of terraced edge in terraced ponds and natural edge in unterraced ponds.

Two water depth measurements were taken inside a 1-m^2 throw trap each time deployed (described below). This minimized variation in depth due to wave action. Borrow pits from terrace building were present, but were rarely encountered and were not measured for water depth. Salinity, conductivity, and water temperature were measured outside the throw trap, using a YSI model 63 meter (Yellow Springs Instruments Inc., Yellow Springs, OH). Turbidity samples were collected in undisturbed water and measured using a calibrated Oakton Instruments T100 Turbidity Meter Kit (model WD-35635-00, Oakton Instruments, Vernon Hills, IL).

Nekton and SAV Sampling. SAV and nekton also were sampled within 5 m of emergent vegetation edges (natural edge or terraced edge) and in open water > 25 m from an edge. Samples were collected using a 1-m^2 by 0.66-m high throw trap, a device commonly used for sampling decapods, small adult fish, and juveniles of large fish (Kushlan 1981; Raposa and Roman 2001) or SAV (Kanouse 2003; Cannaday 2006; La Peyre et al. 2007). The trap was constructed of a welded aluminum frame (1 x 1 m) covered by mesh cloth (mesh size = 1.6 mm). Mesh extended an additional 0.25 m beyond the metal frame and was supported by a buoyant 1 x 1 m PVC pipe frame, which lengthened the height of the trap if thrown into deep water. A bar seine (size 1.0 by 0.5 m, 1.6-mm mesh) was used to remove nekton from the trap. Bar seining was conducted from two sides of the trap until five consecutive passes yielded no nekton. Nekton were put on ice, counted, identified, and weighed in the lab. Additionally, all SAV within the trap was collected by hand. SAV was sorted by genus, oven dried, and weighed in the lab.

Bird Surveys. Surveys began at dawn and continued for 90 minutes. Observers arrived via boat and allowed a 15-minute settling period before beginning observations. During surveys, observers sat hidden in emergent vegetation, using camouflage netting for additional cover. Observations of terraced and unterraced ponds were conducted simultaneously by separate observers. Observers rotated equally between pond types on subsequent surveys to spread observer bias between treatments.

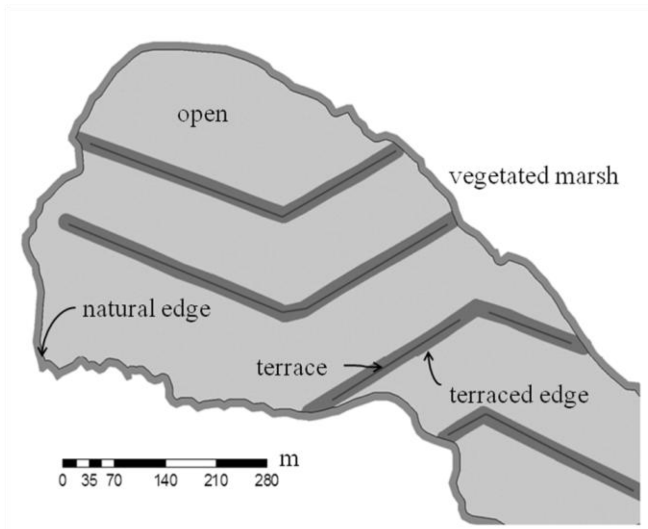
Observers recorded birds by species every 15 minutes, generating seven bird counts per plot each survey. Birds also were recorded as "near" (< 5 m) or "far" (> 5 m) from a pond edge. For flocks of > 10 individuals, total flock size per species was recorded but microhabitat details were taken for only a random subset of 10 individuals. Data for aerial foragers flying over the pond were recorded only after foraging behavior was exhibited (e.g., diving on the pond and subsequently circling over it).

Statistical Methods

All statistical analyses were conducted using SAS 9.1.2 (SAS Institute Inc., Cary, NC). Multiple days were required to sample all sites. However, for analysis, we assigned each survey a single date (the average of the dates over which the survey took place). Those average dates were 31 April, 19 May, 22 June, 30 July, 11 Aug, and 5 September 2005. For analysis, we classified the second Vermilion pond pair as plots within the Vermilion site because it was not hydrologically distinct from the other ponds sampled at that site. Thus, on surveys where both pairs of Vermilion ponds were sampled, they were analyzed as day-site replicates of each other.

Pond Characterization. To compare the proportion of different microhabitat types in each pond type, we analyzed 2004 DOQQ aerial photographs of all sites using ArcGIS 9.1 (ESRI Corporation, Redlands, CA). To do this, a pond was defined as consisting of only water. Any areas of emergent vegetation, whether natural or on terraces, were excluded from pond area (Fig. 2).

Fig. 2 The classification of water in ponds into edge and open habitat, showing the terraced pond at Rockefeller SWR as an example. Natural edge extends 10 m into the pond from vegetated marsh. Terraced edge extends 10 m into the pond on either side of a terrace



Each portion of the pond was classified as either edge (water <10 m of vegetation) or open (water >10 m from vegetation). ArcMap was used to generate the area of each microhabitat type. Areas were then converted into percent of total pond area, yeilding an edge metric that reflected habitat interspersion and could be expected to linearly relate to bird density. As noted by Rehm and Baldassarre (2007), a linear measurement of edge should better predict waterbird density than a cover to open water ratio (sensu Weller and Spatcher 1965); bird density does not vary linearly with a cover:open water ratio, but is maximized at a 50:50 ratio, and 100% open water or 100% vegetation covers have low bird densities. Further, a 50:50 ratio does not always represent high habitat interspersion, but can instead represent highly clumped habitat or simply vary with the amount of adjacent emergent marsh vegetation included in the analysis. However, the percent of water within 10 m of edge is always maximized when interspersion between water and vegetation is maximized.

We used a 10-m edge width for several reasons. Baltz et al. (1993) classified edge habitat as being 7 m wide based on their sampling of nekton from 0 m to 18 m from marsh edge. Minello and Rozas (2002) found most nekton species classified as edge species within 10 m of emergent vegetation. Irlandi and Crawford (1997) classified edge as being 10 m wide based on their sampling of pinfish. Cannaday (2006) assumed an edge width of 10 m for SAV. Weller and Spatcher (1965) concluded that ideal water areas for waterbirds were 18 m to 25 m across. Finally, Rehm and Baldassarre (2007) showed that an edge width of 5 m predicted rail, bittern, and grebe density in ponds. We therefore concluded that 10 m was a logical width estimate for the effect of vegetated edge on most factors influencing waterbird edge use in Louisiana coastal marsh ponds.

Because percent edge was not normally distributed, we used a nonparametric Wilcoxon Rank Sum test to compare the proportion of available edge habitat between pond types (terraced and unterraced). Water quality data also were not normally distributed. To reduce statistical error rates, we analyzed all the water quality variables together with logistic regression, comparing water quality from each microhabitat between pond types and also comparing microhabitats within a pond. Logistic regression is a robust statistical tool for non-normal data. Pond type was the response variable and water quality variables were dependent factors.

Nekton and SAV Analyses. We performed backwards stepwise regression to determine if any measured variables could explain nekton density without including pond type in the model. Potential explanatory variables were date, SAV biomass, water temperature, turbidity, conductivity, water depth, and proportion of microhabitat types (edge or open) in ponds. Average nekton and SAV for the whole pond were used in this analysis by multiplying edge and open sample means by the proportion of pond that was edge or open habitat, respectively. Data were log-transformed to achieve normality of residuals and equalize variances.

Bird Analyses. To avoid double-counting individuals, we used the greatest number of birds per species seen during a single count-interval as our estimate of abundance for that species. We converted bird abundance to bird density (individuals/ha) by dividing abundance by plot area.

Species richness was defined as the number of species observed during a survey. We used a repeated-measures ANOVA with blocking on site to compare bird density and species richness between terraced (treatment) and unterraced (control) ponds. We log-transformed bird density to achieve normality of residuals and equalize variances.

We classified birds into guilds based on foraging method as proposed by De Graaf et al. (*1985*), except that we classified common moorhens as dabblers based on our behavioral observations. The resulting guilds were as follows: 1) Divers (grebes, diving ducks, and cormorants); 2) Waders (herons, egrets, and Roseate Spoonbill); 3) Dabblers (dabbling ducks, Common Moorhen, and Purple Gallinule); 4) Probers (shorebirds and other probers or surface arthropod gleaners such as sandpipers, plovers, Black-Necked Stilt, and rails); and 5) Aerial foragers (terns, gulls, and Belted Kingfisher). Guild density was compared between pond types, using a repeated-measures ANOVA blocked on site. For most guilds (except waders), log-transformations were necessary to achieve normality of residuals and equalize variances. When significant pond type by time interactions existed, individual tests on each survey date were used to compare pond types.

We used a backwards stepwise regression to determine whether measured variables explained bird density, without including pond type in the model. Variables included nekton density, SAV biomass, water temperature, turbidity, salinity, conductivity, water depth, air temperature, wind speed, and proportion of microhabitat types (edge or open) in ponds. To assess whether individual species were observed more often in edge or open water habitats, we used one bird count from each survey. We usually opted for counts from the middle of surveys because they were probably most representative of natural behavior; birds were acclimated to our arrival but it was still close to dawn when bird activity levels are high. We used, in order of preference, data from count 4, 5, 3, 6, 7, 2, or 1, until we found an interval in which birds were observed. If an individual bird was within 5 m of emergent vegetation, we classified it as using edge habitat; otherwise, we classified it as using open water. We assessed habitat bias of birds for edge or open habitats (Anderson and Gutzwiller *2005*) using a chi-square test.

RESULTS

Pond Characteristics

Water quality variables measured (turbidity, water depth, salinity, conductivity, water temperature) did not differ between terraced and unterraced ponds (Table 1). Within ponds, only water depth differed between edge and open habitats ($\chi^2 = 15.96$, df = 1, $P < 0.0001$).

Table 1. Mean water quality (± SE) for marsh edge and open water habitats in terraced (n = 4) and unterraced ponds (n = 4), spring and summer of 2005, Chenier Plain, LA. Water quality did not differ significantly between pond types.

Variable	Microhabitat	Terraced Pond	Unterraced Pond
Turbidity (NTU)	open water	26.4 ± 4.5	52.6 ± 19.7
	marsh edge	55.8 ± 20.6	41.6 ± 7.5
Water depth (cm)	open water	40.3 ± 4.7	44.4 ± 3.9
	marsh edge	27.2 ± 3.8	26.7 ± 3
Salinity (ppt)	open water	6.33 ± 1.27	7.82 ± 1.65
	marsh edge	6.5 ± 1.2	8.2 ± 1.7
Conductivity (mS)	open water	11.6 ± 2.3	14.7 ± 2.7
	marsh edge	11.8 ± 2	13 ± 2.4
Water temperature (ºC)	open water	26.2 ± 1.4	25.7 ± 1.3
	marsh edge	26.2 ± 1.5	25.9 ± 1.2

Terraced ponds averaged 81 ha ($n = 4$, SE = 47, range 9–216 ha) and unterraced ponds averaged 65 ha ($n = 4$, SE = 19, range 10–101 ha). When terraced edge and natural edge were combined into one marsh edge category, terraced ponds had significantly more edge habitat (edge was 34% of pond area, SE = 12%) than did unterraced ponds (edge was 9% of pond area, SE = 2%) ($P < 0.0001$, Table 2).

Table 2. Area (ha) of water classified as open water (OW), natural edge (NE), terraced edge (TE), total edge (Total E), and the ratio of TE:NE in ponds at all sites.

Site	Pond Type	OW	TE	NE	Total E	TE:NE Ratio
Vermilion 1	Terraced	6.5	19.1	2.5	21.6	7.5
	Unterraced	64.3	-	7.2	7.2	-
Vermilion 2	Terraced	6.0	1.7	1.7	3.4	1
	Unterraced	8.5	-	1.6	1.6	-
Sweet Lake	Terraced	155.5	50.5	10.3	60.8	4.9
	Unterraced	95.5	-	5.4	5.4	-
Rockefeller	Terraced	59.3	7.3	5.1	12.3	1.4
	Unterraced	72.3	-	4.8	4.8	-

Nekton and SAV

Whole-pond nekton density in terraced ponds (53.9 nekton/m^2, SE = 25.6) and unterraced ponds (44.8 nekton/m^2, SE = 15.3) was similar. Whole-pond SAV biomass in terraced ponds (14.4 g/m^2, SE = 5.7) and unterraced ponds (12.6 g/m^2, SE = 4.2) was also similar. Log-transformations were necessary for most analyses, but untransformed means are presented for ease of interpretation. Nekton density was influenced by date, SAV biomass, and conductivity in ponds ($F_{3, 23} = 49.3$, $P < 0.0001$). The regression which best explained nekton density was: log (nekton density) = 0.59 log (SAV biomass) − 0.045 conductivity + 0.016 day − 0.41 ($R^2 = 0.84$).

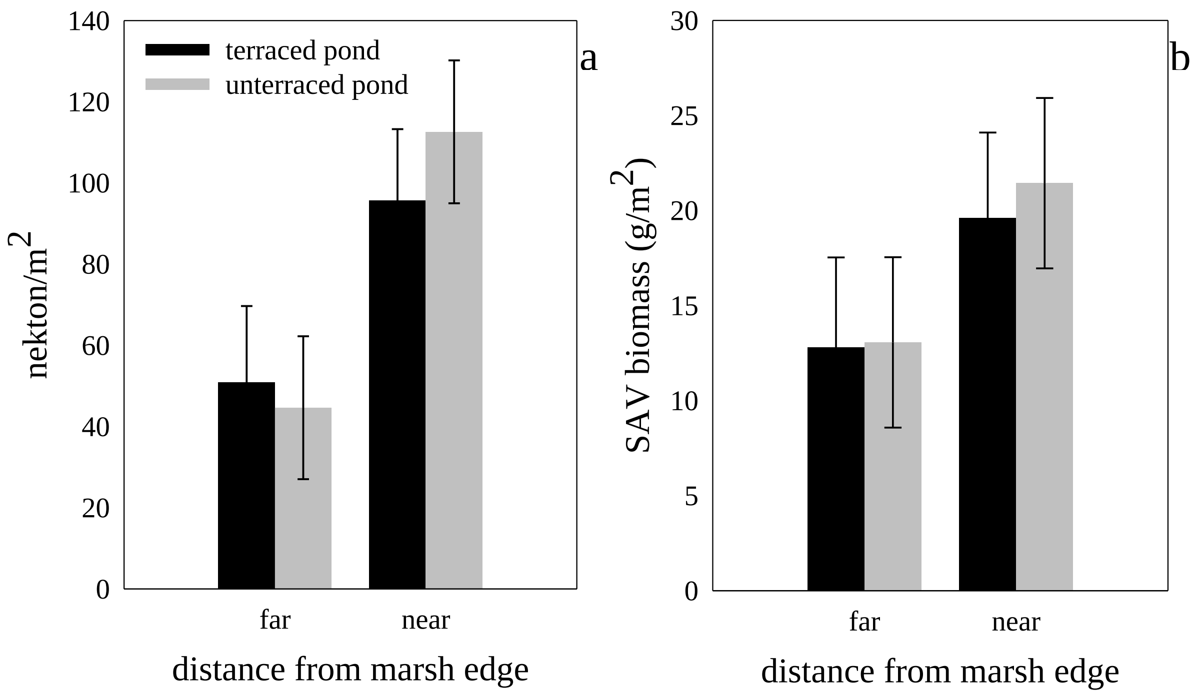


Fig. 3 Untransformed nekton density (a) and SAV biomass (b) in terraced and unterraced ponds at two habitats (mean±SE) in spring and summer of 2005, Chenier Plain, LA. Terraced and unterraced pond estimates did not differ significantly (nekton: $F_{1, 14} = 0.21$ $P = 0.65$; SAV: $F_{1, 14} = 0.41$, $P = 0.53$), and nekton and SAV microhabitat estimates were similar between pond types (nekton in edge: $F_{1, 32} = 0.63$, $P = 0.43$; SAV in edge: $F_{1, 29} = 0.01$, $P = 0.94$; nekton in open water: $F_{1, 32} = 0$, P = 0.95; SAV in open water: $F_{1, 29} = 0.41$, $P = 0.53$). Marsh edge has greater nekton and SAV density than open water (nekton: $F_{1, 29} = 15.21$, $P = 0.0005$; SAV: $F_{1, 39} = 5.7$, $P = 0.022$)

Terraced ponds and unterraced ponds had similar nekton density and SAV biomass at both marsh edge and open water (Fig. 3). However, when data from terraced and unterraced ponds were combined, nekton density at marsh edges (104.1 nekton/m^2, SE = 11.8) was greater than in open water (44.5 nekton/m^2, SE = 12.4), and similarly SAV biomass at edges (20.8 g/m^2, SE = 3) was greater than in open water (11.3 g/m^2, SE = 3.1).

Birds

Whole-pond bird density was 3.8 times higher in terraced than unterraced ponds (Fig. 4). Terraced ponds had greater species richness than unterraced ponds (Fig. 5).

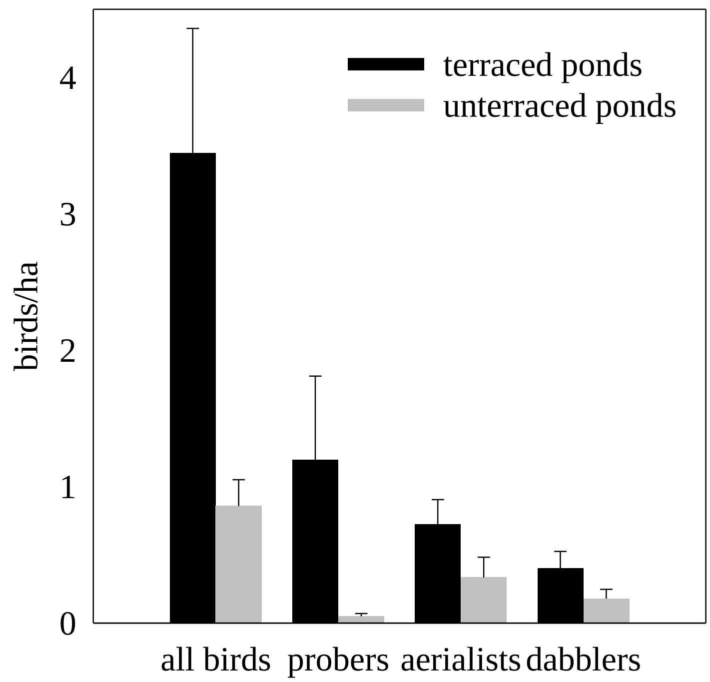


Fig. 4 Untransformed density for all birds, aerialists, dabblers, and probers in terraced and unterraced ponds (mean ± SE), in spring and summer of 2005, Chenier Plain, LA. All differences are significant following log-transformation (All birds: F1, 22 =22.95, *P* < 0.0001; probers: F1, 22 = 7.53 *P* = 0.01, aerialists: F1, 22 = 7.14 *P* = 0.01; dabblers: F1, 22 = 4.55 *P* = 0.04)

When pond types were combined, guilds were, in order of highest average density to lowest: probers (0.6 birds/ha, SE = 0.3), aerialists (0.5 birds/ha, SE = 0.1), dabblers (0.3 birds/ha, SE = 0.07), waders (0.3 birds/ha, SE = 0.05), and divers (0.07 birds/ha, SE = 0.02). Different foraging guilds differed in response to pond type. Three guilds, probers, aerialists and dabblers, had greater densities in terraced than unterraced ponds (Fig. 5).

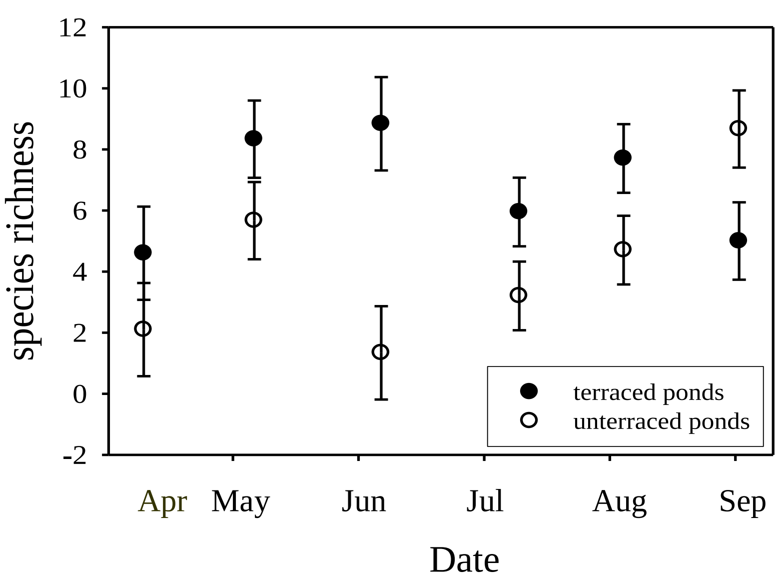


Fig. 5 Bird species richness through time in terraced and unterraced ponds (mean ± SE), in spring and summer of 2005, Chenier Plain, LA. ($F_{5, 22}$ = 12.09, *P* = 0.0021)

There was a significant pond type by time interaction for waders, such that waders were densest in terraced ponds in May, June, and July (Fig. 6). Divers were not abundant and density did not vary significantly between pond types.

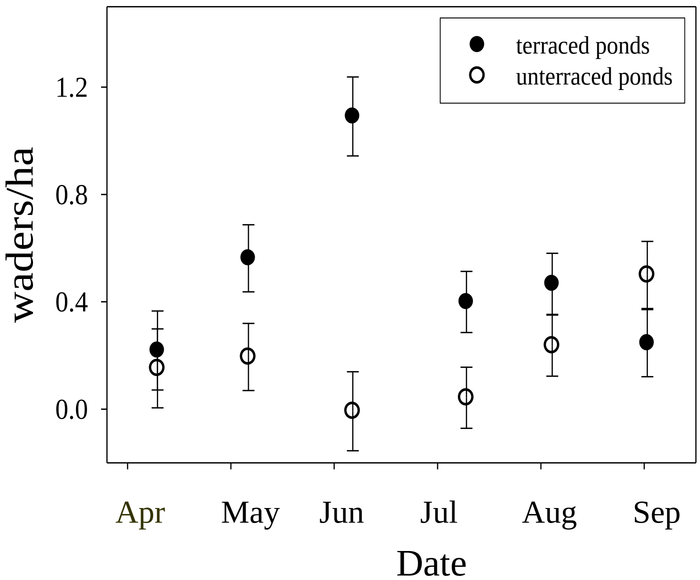


Fig. 6 Wader density through time in terraced and unterraced ponds (mean ± SE), in spring and summer of 2005, Chenier Plain, Louisiana. ($F_{5, 22}$ = 7.02, *P* = 0.0005)

Edge habitat accounted for only 26% (SE = 0.03) of total habitat, but 74% of birds were observed in edge habitat rather than open water (χ^2 = 7.33, df = 1, *P* = 0.0068). Across all habitats, bird densities did not vary significantly with any measured water quality variable, air temperature, wind speed, SAV biomass, or nekton density, but bird density was positively associated with the proportion of edge habitat (Fig. 7).

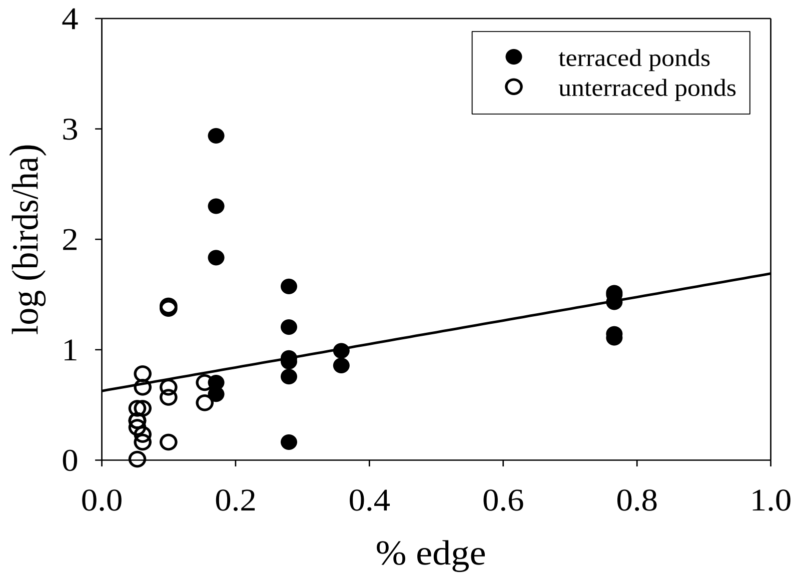


Fig. 7 Relationship between log of mean bird density and percent available marsh edge habitat in ponds, in spring and summer of 2005, Chenier Plain, LA ($F_{1, 34}$ = 6.17, P = 0.018, log (birds/ha) = 1.06 (% edge) + 0.63, R^2= 0.153)

DISCUSSION

Birds responded to pond terracing at both the microhabitat and whole-pond scales. Birds preferred edge habitat, and whole-pond density and species richness were greater in terraced ponds. Foraging guilds varied in their response to pond terracing. Probers, aerialists, and dabblers were consistently denser in terraced ponds than in unterraced ponds. Waders were significantly denser in terraced ponds only during May, June, and July. This corresponds to peak nesting for wading foragers when foraging activity was likely to be highest. Dabblers probably benefited from pond terracing because of increases in SAV adjacent to terrace edges. Dabblers forage for SAV, SAV seeds, and aquatic invertebrates among SAV (Ehrlich et al. 1988). SAV also provides structure, refuge, and forage for nekton (Rozas and Odum 1988; Castellanos and Rozas 2001), and thus may provide a profitable foraging area for omnivorous and piscivorous birds (such as waders). Probers probably respond to pond terracing because they prefer to use edge habitat where water depths are shallower. Divers were rare and did not seem to respond to pond terracing. All ponds were shallow (<1 m deep), and likely did not constitute important habitat for divers.

The best model explaining bird density in ponds included only the amount of edge habitat. Fairbairn and Dinsmore (2001) also found that densities of breeding waterbirds were related to percent edge in prairie pothole wetlands. Additionally, in other northern marshes, maximum bird density and species richness has been found to occur where the proportion of marsh edge is high and water:cover interspersion is high (Weller and Spatcher 1965; Mack and Flake 1980; Kaminski and Prince 1981; Rehm and Baldassarre 2007). Waterbird densities often vary with water quality (Velasquez 1992; Halse et al. 1993; Nagarajan and Thiyagesan 1996). However water quality did not differ between pond types in our study.

Terracing increased the amount of nekton and SAV directly adjacent to terrace edges. As seen in other Gulf Coast studies (Castellanos and Rozas 2001; Minello and Rozas 2002; La Peyre et al. 2007), natural marsh edge and terraced edge did not differ in nekton and SAV density. Nekton density was greatest in areas where SAV was most abundant. However, over whole-ponds, nekton and SAV did not differ between terraced and unterraced ponds. This may be because too few terraces were built in our study ponds. Rozas and Minello (2001) concluded that pond-level nekton biomass was not increased unless terraces occupied >25% of ponds.

Although bird density was not related to nekton or SAV density, edge may be optimal foraging areas for waterbirds because morphological constraints (bill, leg, and neck length) can limit prey accessibility in deep water (Baker 1979, Pöysä 1983). Additionally, habitat complexity may have influenced bird density. Pond terracing increases shape complexity of ponds (Feagin and Wu 2006), and birds may be attracted to habitats that provide desired structure, edge, and shape complexity where food is likely to be abundant. However, terrace edges may not provide foraging equivalence to natural edges.

It is important to note that the efficacy of terraces at slowing marsh erosion and encouraging emergent vegetation expansion has never been fully evaluated. Contrary to predictions about terrace effects, turbidity was not altered from unrestored conditions in our study. Turbidity reduction is believed to be a reason SAV is enhanced by pond terracing (Underwood et al. 1991; Steyer 1993). Terracing is also believed to increase out-of-basin sediment deposition in ponds, decreasing water depths and slowing lateral marsh erosion (Steyer 1993). However, water depths were similar between pond types in our study. Most terracing projects in our study were less than five years old (Stead and Hill 2004), and given time, hurricanes and storm fronts may bring sufficient out-of-basin sediments to reduce water depths. Terrace restoration also is predicted to encourage emergent vegetation expansion, but this has been only rarely demonstrated (see Steyer 1993). Although we did not measure lateral expansion directly, no obvious expansion of emergents into open water was observed adjacent to terrace edges. Peat accumulation also is important to coastal marshes because marsh vertical accretion often depends upon peat rather than mineral sediment accumulation (Bricker-Urso et al. 1989; Nyman et al. 2006). Unfortunately, the effect of terracing on peat accumulation processes has never been studied. Subsequent to this study, Hurricane Rita hit the Chenier Plain. Our personal observations following this storm suggest that terrace fields may be eroding as a result of background wave action and hurricanes forces. Many wetland functions depend on a well-developed soil organic matter layer and may take decades to return to undisturbed levels after a new disturbance. If constant repair of eroding terraces is needed, then such functions may never return to pre-disturbance conditions. Further study is necessary to determine whether terraces benefits are sustainable or expanding.

While terracing has obvious benefits to birds, combining it with other restoration techniques, such as water depth manipulation, could further improve waterbird habitat. Variation in water depth promotes use of wetlands by a diversity of waterbirds (Parsons 2002; Taft et al. 2002; Bolduc and Afton 2004; Darnell and Smith 2004), and management of water levels can promote waterfowl, shorebirds, and desirable vegetation (Kadlec 1962; Harris and Marshall 1963; van der Valk 1981; Twedt et al. 1998; Merino et al. 2005). Shorebirds only were seen in our study ponds when exposed shallow margins were available. However, most sites within the Chenier Plain lack water control structures needed to vary water depths seasonally. Terracing is a passive means of providing a range of water depths, especially if shallow slopes and broad shoulders are used.

We only addressed spring and summer waterbird communities and many winter migrants were absent. The effect terracing on wintering waterbirds also should be determined.

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