Review of the Benefits of Marsh Terraces in the Northern Gulf of Mexico

Michael G. Brasher

Ducks Unlimited, Inc., Gulf Coast Joint Venture

700 Cajundome Blvd, Lafayette, LA 70506

mbrasher@ducks.org

DUCKS UNLIMITED

Gulf Coast Joint Venture

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Abstract

The extent and rate of coastal wetlands loss in the northern Gulf of Mexico are the highest observed throughout the conterminous U.S. In response, numerous techniques have been developed to help slow these losses and restore productive coastal wetlands. Marsh terracing is a relatively new technique and has become a common feature of coastal restoration efforts in the northern Gulf of Mexico. Marsh terraces are segmented ridges of bare soil and emergent marsh constructed from excavated subtidal substrates in shallow, open water areas. They function by reducing fetch and wave energy, which is believed to help create emergent marsh, reduce shoreline erosion, increase growth of submerged aquatic vegetation, and ultimately increase habitat quality for marsh-dependent organisms. However, the efficacy of marsh terraces in achieving these objectives remains uncertain, largely due to a lack of rigorous evaluations, which has led to their de-emphasis in some coastal restoration programs and projects. I conducted a literature review to ascertain and summarize current knowledge and remaining gaps in our understanding of the benefits of marsh terraces. Available information provided general support for the effectiveness of marsh terraces, although the magnitude and consistency of benefits varied greatly among objectives. Benefits were most evident for improving nekton habitat, but were more variable for improving waterbird habitat, reducing shoreline erosion, and creating emergent marsh outside the terrace footprints. Recommendations will be provided for additional scientific investigations that are needed to definitively assess the benefits of marsh terraces and appropriately inform decisions about their design and application.

Introduction

Wetlands are a major feature of the U.S. Gulf of Mexico coast and are responsible for myriad ecological and economic services at regional and national scales (Chabreck 1988, Engle 2011). Areal coverage of wetlands in the coastal watersheds of this region is extensive, totaling nearly 15.5 million acres and accounting for over 47% of all wetlands within coastal watersheds of the conterminous U.S., excluding coastal wetlands of the Great Lakes (Dahl and Stedman 2013). The Gulf of Mexico region contains 52% of all intertidal wetlands in the conterminous U.S., yet this region is responsible for a disproportionately high percentage of U.S. intertidal wetland losses (Dahl and Stedman 2013). From 2004 to 2009, intertidal wetlands in the U.S. Gulf of Mexico region decreased by >95,000 acres, which accounted for >99% of all intertidal wetland losses in the conterminous U.S. during that same time frame (Dahl and Stedman 2013). Losses of intertidal wetlands along the U.S. Gulf Coast are neither new nor expected to abate. In Louisiana alone, >1.2 million acres of coastal emergent marsh has converted to open water since 1932 (Couvillion 2011), and an additional 1.1 million acres may be converted by 2060 (Coastal Protection and Restoration Authority of Louisiana 2012).

Causes of intertidal wetland loss along the Gulf of Mexico coast are many and varied, including the direct and indirect effects of natural processes and anthropogenic activities (Craig et al. 1979, Turner and Cahoon 1987). Among the most important causes are geologic subsidence; sea level rise; reduced sediment loads in major river systems (e.g., Mississippi River); isolation of major rivers from adjacent marshes by flood-control levees; creation of canals and waterways for oil and gas extraction, pipelines, and navigation; elevated salinity resulting from hydrologic alterations; and tropical storms (Craig et al. 1979, Gosselink et al. 1998). Numerous coastal restoration programs and plans have been developed to provide resources and guidance for actions to slow or reverse coastal wetlands loss within the Gulf region. These programs have enabled implementation of a variety of restoration techniques, which are typically designed to address a specific causal agent of wetland loss or otherwise contribute to the creation of new coastal emergent wetlands once the principle agent has been eliminated or lessened. Common restoration techniques include sediment diversions, freshwater diversions, beneficial use of dredge material, shoreline protection structures, vegetative plantings, saltwater barriers, and structural marsh management (Chabreck et al. 1989, Nyman and Chabreck 2012). These and other techniques have been implemented with varying levels of success for decades, yet growing challenges, unique environmental circumstances, and limited resources constantly force coastal conservationists to seek new and more efficient coastal restoration strategies and techniques (Chapman and Reed 2006).

Recently, another restoration technique, referred to as marsh terracing, has become increasingly common in coastal Louisiana and Texas. Marsh terraces are segmented ridges of bare soil and emergent marsh constructed in shallow, open water areas from *in situ* excavated subtidal substrates (Figure 1) (Turner and Streever 2002). Terraces are most often constructed in large water bodies that were once emergent marsh but have converted to open water as a result of exposure to a variety of marsh loss factors. Terraces are usually designed and constructed with a crown height equal to surrounding marsh elevation to enable periodic tidal inundation of the terraces and associated vegetation (Turner and Streever 2002). Vegetation plantings, including *Spartina alterniflora* (smooth cordgrass) in brackish and saline marsh, and *Schoenoplectus californicus* (California bullwhip) in fresh and intermediate marsh, are established on the perimeter of the terraces following construction to accelerate colonization by emergent vegetation and provide immediate protection from erosion. Marsh terraces function similarly to other restoration techniques such as breakwater and sediment retention structures (e.g., Christmas tree fences; Boumans et al. 1997), by reducing fetch and wave energy, thereby increasing the potential for sediment deposition in the leeward side of the structure (Steyer 1993, Turner and Streever 2002). Marsh terraces were conceived as a potentially effective restoration technique partly because of their theoretical ability to interrupt the negative feedback cycle of marsh erosion that is initiated once interior marshes begin to fragment. In this cycle, as intact marshes begin to fragment and convert to open water, fetch increases and enables production of greater wave energy, which in turn increases marsh erosion rates, ultimately accelerating conversion to an ever-expanding body of open water.

Terraces may be implemented to achieve one or multiple coastal restoration objectives. The primary restoration objectives include, 1) creating emergent marsh, 2) increasing marsh edge, 3) reducing turbidity to increase light penetration into the water column and promote growth of submerged aquatic vegetation (SAV), 4) reducing erosion of adjacent marsh by reducing fetch and wave energy, and 5) promoting the growth of emergent marsh within the terrace field as facilitated by increased bay bottom elevations resulting from reduced wave action and accretion of mineral sediments and organic material (e.g., SAV) (Steyer 1993, Turner and Streever 2002, Nyman and Chabreck 2012), and 6) enhancing primary productivity and habitat quality for marsh-dependent organisms (e.g., fish, crustaceans, birds) .

Use of marsh terraces as a coastal restoration technique has been most prevalent in Louisiana, but is becoming more common in Texas. Since 1990, marsh terraces have been constructed at over 80 sites in Louisiana and Texas, encompassing > 4,320 individual terrace ridges, whose combined length exceeds 673,000 m (Gulf Coast Joint Venture and Ducks Unlimited, Inc., unpublished data). Over the years, marsh terrace projects have been promoted and funded through numerous wetland restoration programs and conservation plans, including the Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA); North American Wetlands Conservation Act; North American Waterfowl Management Plan; Coastal Impact Assistance Program; Louisiana Coastal Protection and Restoration Authority; other state and federal programs; wetland mitigation programs; and private landowner interests. Although adaptive resource management is widely embraced within the coastal wetlands conservation community, its application to evaluate and improve the effectiveness of marsh terraces as a coastal restoration technique has been limited, despite calls for such by Turner and Streever (2002). Evaluation efforts have thus far been diffuse, focusing on individual projects or otherwise consisting of short-term, isolated research investigations (e.g., Steyer 1993, O'Connell and Nyman 2010, Miller and Aucoin 2011). The conceptual functionality of marsh terraces for achieving marsh restoration objectives is sound, and significant work has been conducted to demonstrate the value of marsh terraces at improving nekton habitat (Zimmerman and Minello 1984, Rozas and Minello 2001, Bush Thom et al. 2004, Rozas et al. 2005, La Peyre et al. 2007). However, there remains uncertainty about the ability of marsh terraces to achieve other objectives, largely because of the lack of rigorous (i.e., incorporating spatial and temporal replication) studies to evaluate their performance in this regard. This uncertainty has lead to reduced emphasis on marsh terracing projects within some coastal wetland conservation programs. Thus, there is an imminent need for clear understanding of the effectiveness of marsh terraces as a conservation technique to inform current and future resource allocation decisions.

I conducted a literature review to determine current knowledge and remaining gaps in our understanding of the benefits of marsh terraces in the northern Gulf of Mexico. The findings are summarized in this report, which should help identify future evaluation priorities and aid decisions regarding application of this technique. This review incorporated all known peer-reviewed literature, graduate theses and dissertations, and unpublished agency reports that evaluated and documented terrace benefits either through scientific studies or systematic monitoring of project performance (e.g., throughout the 20-year life of projects funded through the CWPPRA). Although the primary interest in this review was documenting benefits of marsh terraces as independent restoration techniques, I also included results from terrace projects used in combination with other techniques [e.g., beneficial use of dredge material, river crevasses (Hymel and Breaux 2012)] to ensure that the potential synergistic benefits of terraces were examined and acknowledged. Information is presented in sections corresponding to the suite of coastal restoration objectives targeted by marsh terraces.

Literature Review

Creation of emergent marsh

Terraces may lead to marsh creation either by establishment of vegetation on the footprint of terraces or retention of suspended sediments and emergence of vegetation in areas adjacent to terraces (Turner and Streever 2002, Nyman and Chabreck 2012). Successful establishment of vegetation on terrace footprints has been documented at numerous sites. Vegetation establishment is usually expedited via direct plantings within the intertidal area of the terraces, which expand and colonize the remaining terrace area during subsequent years. Steyer (1993) evaluated different methods for establishing *S. alterniflora* on terraces by comparing survivorship between gallon plug specimens and single stem sprigs at 2 terraced sites on Sabine National Wildlife Refuge (NWR). Survivorship was marginally greater for gallon plugs, although this difference was not statistically significant. Survivorship exceeded 82.9% across all treatment and site combinations. Areal coverage of individual plants after 1 year was greater for gallon plugs than sprigs (1.68 m vs. 1.07 m), although terraces were entirely vegetated within 2 years following plantings and there was no apparent difference in the spread between gallon plugs and sprigs (Steyer 1993). Because of lower mobilization costs, planting of sprigs was recommended instead of gallon plugs. Rapid establishment of vegetation on terrace footprints has been corroborated by Castellanos and Aucoin (2004), Thibodeaux and Guidry (2009), Miller and Aucoin (2011), Hymel and Breaux (2012), and McGinnis et al. (2012).

Steyer (1993) documented a total gain of 1.47 ha of new emergent marsh at 2 terrace sites within the year following construction, chiefly occurring on the terrace footprints. This increased net annual primary productivity by 3.67×10^4 kg dry weight/yr. Within 2 years following construction, emergent vegetation had spread beyond the terrace footprints to produce 6.8 ha of new marsh. Based on these observations, Steyer (1993) concluded that vegetation would eventually cover the entire terraced site. However, feldspar marker horizons and bathymetric surveys provided variable evidence of vertical accretion within the terraced site, ranging 0.05 – 8.97 cm during the first year following construction. Rozas and Minello (2001) working at the same site nine years later provided limited evidence of sediment accretion or further marsh expansion across the terraced site. Good et al. (2005) used aerial photography and GIS analyses to measure change in marsh area of the Sabine terraces 10 years post construction. Within a pre-defined area surrounding and inclusive of the terraced sites, they documented an 18% (4.71 ha) increase in total emergent vegetation, which included lateral expansion of marsh associated with terrace footprints, non-terrace islands, and adjacent shorelines. Good et al. (2005) reported differences in marsh growth between the 2 terraced sites, and suggested this was likely caused by disparate amounts of suspended sediment entering the sites. Visual observation of 2013 aerial imagery suggested continued expansion of emergent vegetation within the terraced sites, although this has not been quantified with formal analyses (MGB, personal observation).

Post-construction monitoring by Thibodeaux and Guidry (2009) of the Pecan Island restoration project in southwest Louisiana revealed that 67% of 81 ha of terraces were vegetated within three years after construction and planting. Within that same time frame, emergent vegetation outside the terrace footprints (i.e., that which existed prior to terrace construction) decreased by 16% at the terraced site, compared with only a 9% decrease in emergent vegetation on a nearby reference site. Yet at both sites the absolute magnitude of emergent marsh decline was small, accounting for only 3 and 1 ha of losses at the terrace and reference site, respectively (Thibodeaux and Guidry 2009).

Land change analyses suggested that construction of 51 ha of terraces on a site in the Calcasieu/Sabine basin helped reverse a long term (1985 – 2004) trend in land loss, changing from a loss rate of -0.04 %/yr to a slight gain of 0.04 %/yr (1985 – 2010) (McGinnis et al. 2012). However, the effects of Hurricanes Rita and Ike in 2005 and 2008 negated these gains and produced a land loss rate of -1.92 %/yr from 2005 – 2010. Land change analyses for eastern Little Vermilion Bay and Little White Lake terrace sites on the south-central Louisiana coast (i.e., Four Mile Canal Terracing and Sediment Trapping Project) from 2004 – 2007 revealed gains in vegetated marsh at both areas (Miller and Aucoin 2011). Total land gain in the eastern Little Vermilion Bay and Little White Lake sites was 7.3 and 2.8 ha, respectively. The majority of land gain occurred along terrace edges and was attributed to deposition of sediment in areas behind the terraces, some of which was likely transported by Hurricanes Lily, Katrina, Rita, and Gustav (Miller and Aucoin 2011).

Castellanos and Aucoin (2004) compared aerial photography collected immediately after construction with that from 2 years later and documented a loss of 2.1 ha of land in the Little Vermilion Bay Sediment Trapping project site, including losses from both terraces and pre-existing natural marsh. This represented a 7% decrease in land area and contrasts with the 0.2 ha loss (0.1% decrease) in a nearby reference site. However, land loss in the terraces site was at least partly attributed to erosion of a series of sacrificial terraces that were constructed at the southern extent of the Little Vermilion Bay Sediment Trapping project area (Castellanos and Aucoin 2004). Aerial photography also revealed the appearance of 80.7 and 45.1 ha of mudflats in the project and reference sites, respectively. Within the project site, mudflats were concentrated around the terraces, while those in the reference site were adjacent to natural features. Castellanos and Aucoin (2004) viewed this as evidence that terraces were likely contributing to sediment deposition and build-up of the bay bottom, which in the future may become colonized by emergent vegetation. Examination of recent (2014) aerial photography (MGB, personal observation) and site visits (J. D. Foret, NOAA National Marine Fisheries Service) indicate modest emergence of vegetation on the previously documented mudflats. Similarly, Wood et al. (2012)

reported anecdotal observations of extensive (approximately 325 ha) mudflat formation at a terrace site in West Cote Blanche Bay (i.e., The Jaws), although they were not yet colonized by vegetation.

Occasionally, marsh terraces have been combined with other restoration techniques in the hopes of enhancing project performance through synergistic effects. For example, marsh terraces were incorporated into the receiving embayment of a river crevasse project in Plaquemines Parish, Louisiana, to encourage settling and retention of suspended sediments carried by water diverted from the Mississippi River (Hymel and Breaux 2012). Five years after construction, 8 acres of vegetated land had been created within the project boundaries, representing a 1% gain. By way of comparison, nearby project sites affected by other crevasses but lacking terraces experienced land losses or gains ranging from -3% to +12%. This further contrasts with 2 reference sites that experienced vegetated land gains of 6 and 13% (Hymel and Breaux 2012). Although proportional vegetated land gain in the receiving bay containing terraces was similar to or lower than that at nearby crevasse receiving bays without terraces, aerial photography revealed significant mudflats throughout and around the terraces which may become vegetated in subsequent years (Hymel and Breaux 2012). Unfortunately, the time required for project benefits to accrue, and thus the appropriate timeframe for evaluating project performance, remains poorly understood, yet has a potentially profound effect on conclusions drawn from terrace evaluation efforts.

Collectively, these findings provide clear evidence that constructing marsh terraces and promoting establishment of vegetation on them can increase the amount of emergent marsh within the area affected by terraces, either when used as an independent technique or paired with others. However, various environmental factors affect the extent and speed of vegetation establishment. It has also been demonstrated that marsh terraces can promote sediment deposition and vegetation establishment outside the terrace footprints, although the magnitude and consistency with which this occurs is variable within and among sites. Myriad factors likely affect rates of emergent marsh creation on and around marsh terraces, including water depth, suspended sediment load, salinity, physical properties of marsh substrate, and density of terraces, among others (Rozas and Minello 2001, Turner and Streever 2002), yet the relative importance of these remains poorly studied.

Increases in marsh edge

Marsh terraces are constructed in areas of open water. Upon subsequent establishment by emergent vegetation, usually from plantings, increases in marsh edge are readily apparent. Steyer

(1993) documented a total increase of 11,756 m of marsh edge in 2 terraces sites at Sabine NWR within 1 year after construction, increasing to 16,000 m of edge 2.5 years after construction. However, the diversity of marsh habitat types (e.g., high marsh, low marsh) and the configuration of marsh edge may differ between terraced sites and natural marsh sites, with greater percentage of high marsh area and greater aggregation (i.e., larger percentage occurring as interior marsh) occurring in natural marsh sites (Feagin and Wu 2006). Terrace construction designs are highly linear, which may result in less marsh-water edge in terraced sites than in similar-sized sections of unmanaged marsh edge (Bush Thom et al. 2004), although these findings have not been universal (Feagin and Wu 2006). Rozas and Minello (2007) documented a 3-fold increase in marsh edge following construction of terraces in Galveston Island State Park. Similar results of greater amounts of edge in terraced than reference sites were documented by O'Connell and Nyman (2010). Thus, there is strong, empirical evidence that marsh terraces are effective at increasing the amount of marsh edge habitat within the area affected by terraces.

Reduction of shoreline erosion

Steyer (1993) measured shoreline retreat at 2 terraced sites and 1 reference site at Sabine NWR before and after terrace construction. Pre-construction erosion rates, based on imagery from 1933 – 1989, were -4.22 m/yr, -4.35 m/yr, and -2.03 m/yr at the 2 terraced sites and reference site, respectively, indicating a retreating shoreline at all locations. One year post construction, shoreline erosion reversed with shoreline expansion occurring at rates of +12.69 m/yr, +2.00 m/yr, and +4.51 m/yr, at the terraced sites and reference site, respectively. Average pre-construction erosion rate was -3.53 m/yr and post-construction erosion rate was +6.4 m/yr. Miller and Aucoin (2011) reported the results of monitoring efforts for the eastern Little Vermilion Bay and Little White Lake terrace sites 6 years post-construction, although without companion reference sites. Shoreline change rate in the Little White Lake site averaged 0.0 m/yr, but measurements ranged from -2.41 m/yr to +5.4 m/yr across individual sampling stations. Average shoreline change rate in the eastern Little Vermilion Bay site was +0.58 m/yr, ranging from -0.14 m/yr to +1.73 m/yr. Lack of nearby reference sites precludes meaningful assessment of the relative effectiveness of marsh terraces for reducing shoreline erosion at these sites.

Segmented terraces were constructed adjacent to the northwest shoreline of Callicon Lake in Cameron Parish, Louisiana to reduce erosion rates of the land bridge separating it from Grand Lake. Measurements collected four years after construction revealed minor shoreline erosion behind the terraces, but the rate was significantly less than that observed at the reference site. Average erosion rate behind the terraces was -0.06 m/yr, while that at the reference site was -1.26 m/yr (McGinnis and Guidry 2011). Efforts thus far suggest positive benefits of marsh terraces in reducing, and in some cases reversing, erosion of adjacent marsh shorelines. However, the extent and magnitude of these effects across the larger population of terrace sites remains unclear, largely because of the limited spatial replication and lack of rigorous study designs incorporated into existing evaluations of shoreline change.

Turbidity reduction and enhanced growth of submerged aquatic vegetation

The value of terraces in providing habitat for and encouraging the growth of SAV has been measured indirectly through assessment of how terraces affect turbidity, as well as directly through estimation of SAV resources. Findings from both types of studies are presented in this review.

The benefit of terraces for reducing fetch and dampening wave action has been documented by several studies. Steyer (1993) reported average wave heights within 2 terraced sites that were 37% and 48% lower than those observed at the reference site, yet there was no strong evidence for differences in turbidity among the terraced and reference sites. Anecdotally, Thibodeaux and Guidry (2004) noted decreases in fetch and erosive wave energy immediately following construction of a terrace site in southwest Louisiana. The ability of terraces to reduce fetch has the additional benefit of reducing the variability of water levels in open marsh systems, as demonstrated by water levels within a terrace field in southwest Louisiana that were 50% less variable (i.e., range in water levels was 0.12 m lower) following terrace construction (McGinnis et al. 2012).

Rozas and Minello (2001) reported turbidity levels in a terrace site during May were approximately half those in a reference site, but the differences were not statistically significant. Mean turbidity levels were similar between terraced and reference sites during September. Further, Rozas and Minello (2001) documented small and seemingly similar amounts of *Ruppia maritima* in reference and terraced sites, but limited occurrence of SAV in samples precluded formal analyses. At a terrace site in Texas, Rozas and Minello (2007) documented an approximately 45 – 56% reduction in turbidity and noted the appearance of seagrass in the years following terrace construction.

To assess effects of terraces on growth of SAV, Caldwell (2003) sampled SAV biomass and occurrence on transects at locations proximal to terraces and natural marsh within terraced sites on Sabine NWR. Data were collected during a period 2-3 years following construction of the terraces. Biomass of SAV was greater in plots near natural marsh than terraces (16 g/m^2 vs. 0 g/m^2), but the distribution of SAV was patchy, occurring in only 3 of 90 samples collected. Occurrence of SAV varied throughout the year, but differed between natural marsh and terraced sites only during September, occurring more frequently near natural marsh than terraced sites (42% vs. 9%). However, Caldwell (2003) employed a sample design whereby samples collected from natural marsh sites were within the same body of water where terraces were located. Thus, the natural marsh sites may not have represented a truly independent reference site. The study also lacked a reference site of purely open water, thus precluding comparison of SAV at a terraced site to one where terrace construction might feasibly be targeted.

Cannaday (2006) was the first to use a paired design with replication to compare SAV growth in terraced and reference sites (n = 3 pairs of sites). Turbidity was lower in terraced sites during 2 of 7 sampling periods, but was otherwise similar between terraced and reference sites during the study. All 3 sampling techniques that were used to measure SAV (i.e., rake sample, core sample, throw trap) revealed an inverse relationship between turbidity and SAV, although this relationship was inconsistent among sample periods. As measured from rake samples, SAV occurred 2x more frequently in terraced than reference sites (20% vs. 9%, respectively). Both core samples and throw trap samples revealed greater SAV biomass in terraced than reference sites (5.6 g/m^2 vs. 1.6 g/m^2), with *Ruppia maritima* (widgeongrass), *Myriophyllum spicatum* (Eurasian water-milfoil), *Potamogeton pusillus* (thin-leaf pondweed), *Najas guadalupensis* (southern naiad), and filamentous algae the most abundant species (Cannaday 2006).

La Peyre et al. (2007) also employed a paired design with three replicates to study terrace benefits in southwest Louisiana; 2 of the sites were the same studied by Cannaday (2006). Mean turbidity was similar between terraced and reference sites. Submerged aquatic vegetation was detected in 68% of samples at both terraced and reference sites, but mean biomass of SAV was greater at terraced sites. However, this difference was driven heavily by 1 site [i.e., the one not also studied by Cannaday (2006)]. When this site was excluded from analysis, there was no detectable difference in SAV between terraced and reference sites.

O'Connell and Nyman (2010) reported no significant difference in mean turbidity measured across 3 pairs of terrace and reference sites in southwest Louisiana, although point estimates of mean turbidity in open water habitats were lower at terraced than reference sites (26.4 vs. 52.6 NTU). Biomass of SAV was greater near marsh edge than in open water areas, yet when combined across edge and open water sampling sites [i.e., defined as "pond level" by O'Connell and Nyman (2010)], estimates of biomass were similar between terraced and reference sites, despite greater amounts of edge in terraced sites.

Thibodeaux and Guidry (2009) reported the appearance of several species of SAV [*Ceratophyllum demersum* (coontail), *M. spicatum, N. guadalupensis*] within a terrace field in southwest Louisiana 4 years post-construction. Surveys prior to construction detected only an unidentified algae present with the project area. Percent coverage of SAV, excluding algae, increased from approximately 0% pre-construction to 37 – 58% 4 years post-construction. In a nearby reference site, the only SAV detected during the pre- and post-construction sampling period was an unidentified algae. Occurrence of SAV was low at the Little Vermilion Bay Sediment Trapping project and reference sites, and there was no discernible increase in SAV following terrace construction (Castellanos and Aucoin 2004). However, SAV sampling for the post-construction period occurred during spring, which may have underestimated the presence of SAV because this does not correspond to the window of peak SAV abundance.

Steyer (1993) evaluated the effectiveness of SAV restoration through plantings of *R. maritima*, *Halodule wrightii* (shoalgrass), and *Thalassia testudinum* (turtlegrass) in terraced sites at Sabine NWR. Of the three species, only *H. wrightii* showed appreciable evidence of survival within 2 months after planting date, ranging from 22% to 54% among terraced sites. This was considered evidence of the potential merits of SAV plantings and speculated that low survival may have been caused by unsuitable environmental conditions following planting (e.g., high turbidity, series of low tides that may have enable desiccation of transplanted SAV).

There is convincing evidence that marsh terraces reduce fetch and resulting wave energy. Terraces have also been demonstrated to reduce turbidity, but this pattern is not consistent across sites and may vary temporally. Although not all studies have detected positive effects of terraces in reducing turbidity, rarely (if ever) was turbidity found to be greater in terraced than reference sites. Direct measures of changes in SAV growth in terraced sites have generally been positive, with occurrence and biomass of SAV in terraced sites equal to or greater than reference sites. However, the magnitude of this effect may vary significantly among sites and throughout the year.

Enhancement of habitat for estuarine nekton

Researchers have consistently documented positive benefits of terraces for estuarine nekton as indicated by higher fish and crustacean use of terraced than reference sites. Greater use of terraced sites is primarily attributed to increases in habitat structure and the amount of marsh edge, which has been demonstrated to positively influence habitat quality for various estuarine species (Zimmerman and Minello 1984). Rozas and Minello (2001) documented greater nekton use of a terraced site than a nearby unterraced, reference site. Specifically, mean densities and biomass of most shrimp and crabs were greater at terraced marsh sites than reference open water areas, which were considered representative of pre-restoration conditions. Densities and biomass of most crustaceans were similar between open water areas in the terraced and reference site, while patterns of fish density varied between the terraced and reference sites, with several species more abundant at reference marsh sites. Patterns of species composition and abundance differed between terraced and reference sites, leading Rozas and Minello (2001) to conclude that marsh established through terracing activities was not functionally equivalent to natural marsh, even after 9 years post-construction. Nevertheless, when the total amount of edge and open water was taken into account between a representative 1 ha of the terraced and reference sites, abundance and total biomass of most nekton species were 1.5 – 55x greater in the terraced than reference site.

Bush Thom et al. (2004) reported that nekton use (i.e., density, catch per use effort) of terrace edges was similar to use of marsh edge in a nearby reference site (i.e., unmanaged marsh edge) and at least 2x greater than that observed in open water areas, which were considered to represent pre-restoration conditions. Similar to reports of Rozas and Minello (2001), nekton community composition differed between terrace edge and unmanaged marsh edge, with terrace edge having a higher percentage of pelagic fish and lower percentage of benthic fish and crustaceans. These differences may have been caused by lower organic matter in the benthos at terrace edges, which likely negatively impacted the availability of benthic prey items upon which benthic fish species depend (Bush Thom et al. 2004).

Studying terraces of 3 different cell sizes in Galveston Island State Park, Rozas and Minello (2007) compared nekton use of terraced sites to a nearby reference site. Based on data collected pre-construction, abundance and biomass of nekton in shallow, non-vegetated bottom habitat was similarly low both before and after terrace construction. However, when measured over the entire project site, nekton abundance and density increased significantly following terrace construction. Densities of crustaceans and the 7 most abundant fish species were significantly greater in terrace and reference edge habitats than shallow, non-vegetated bottom (i.e., the habitat replaced by terraces), although certain species (e.g., gulf menhaden, *Brevoortia patronus*) were more abundant in non-vegetated bottom habitats. There were no consistent patterns of differences in nekton density between terraced and reference marsh sites (i.e., samples collected within 1-2 m of terrace/marsh edges).

La Peyre et al. (2007) documented similar nekton density and richness in nearshore (i.e., edge) habitats between terraced and reference sites in southwest Louisiana. However, nekton density and richness were greater at all (terrace and reference) nearshore habitats than open water sites, which were considered representative of pre-terrace conditions. Similar to previous studies, La Peyre et al. (2007) reported that terraced and reference sites were not functionally equivalent, because nekton communities differed between them. Reference sites contained a greater percentage of benthic species, while terraced sites were characterized by a greater percentage of pelagic species. Further, crustaceans were more abundant at nearshore habitats in reference than terraced sites, while marsh-SAV associated species made up a greater percentage of nekton in terraced open water than reference open water sites. Measures of fish condition indicated that *Microgobius gulosus* (clown goby) and *Poecilia latipinna* (sailfin molly) sampled in open water habitats were in poorer condition in terraced than reference sites. La Peyre et al. (2007) concluded that marsh terraces increased habitat values for estuarine nekton, because nekton density, biomass, and diversity were all greater in terraced nearshore than open water habitats.

Consistent with other studies, O'Connell and Nyman (2010) documented similar nekton density in terraced and reference sites when measured at the "pond level," but greater nekton density in edge than open water habitats. Lastly, Rozas et al. (2005) compared the costs and effectiveness of multiple marsh restoration techniques for providing nekton habitat. Although other project designs supported greater overall fishery values, marsh terraces were more efficient, because of their higher fishery-benefit:cost structure. Among all the objectives for which terraces are constructed, the evidence is strongest for the significant positive benefits that marsh terraces have on increasing habitat for estuarine nekton.

Enhancement of habitat for waterbirds

In contrast to studies examining the potential for marsh terraces to improve nekton habitat, only 1 study has thus far examined the benefits of terraces for improving waterbird habitat. O'Connell and Nyman (2010, 2011) used a paired design with 3 replicates to measure waterbird density and species richness on terraced and reference sites in southwest Louisiana. Originally designed as a study involving year-round, monthly sampling, researchers were forced to modify sampling methods and study objectives following landfall of Hurricane Rita in southwest Louisiana midway through the study. O'Connell and Nyman (2010) measured waterbird, SAV, and nekton abundance between terraced and reference sites, but restricted their analyses to pre-Hurricane data to avoid confounding effects of storm-related environmental changes. Waterbird density and species richness were 3.8 and 1.4x greater in terraced than reference sites, when measured at the "pond level." Bird response varied among foraging guilds, with probers, aerial foragers, and dabblers being more abundant in terraced sites across all time periods; waders were more abundant in terraced sites only during May, June and July. Greater abundance of waterbirds in terraced sites was presumably driven by the greater amount of edge habitat, because edge accounted for only 26% of total habitat area yet was responsible for 74% of bird counts. Moreover, waterbird density was positively related to proportion of edge habitat in individual study ponds. No relationship was detected between waterbird density and either SAV biomass or nekton density.

O'Connell and Nyman (2011) compared waterbird use of terraced and reference sites before and after Hurricane Rita; 2 of 3 study sites were the same as those used by O'Connell and Nyman (2010). Waterbird abundance was 75% greater in terraced than reference sites, with no difference in this pattern before and after Hurricane Rita. Response was greatest for dabblers and waders, being 82% and 77% more numerous in terraced than reference sites. Total number of species detected was greater on terraced than reference sites, but when adjusted for number of individuals observed, richness was statistically similar between terraced and reference sites.

Although O'Connell and Nyman (2010, 2011) reported consistent positive effects of marsh terraces on waterbird use, this objective remains poorly studied. Anecdotal observations of variable waterbird use of terraced sites throughout coastal Louisiana stimulate continued uncertainty of the effectiveness of terraces at enhancing waterbird habitat. Robust determination of waterbird benefits of terraces, or lack thereof, will require additional studies that capture greater spatial and temporal variation.

Discussion

Limited conservation resources and persistent threats to natural resources demand that conservation actions be evaluated to determine their effectiveness and refine or expand them as needed (Nichols and Williams 2006). Marsh terraces have become a prominent feature of coastal restoration in the northern Gulf of Mexico, yet their effectiveness continues to be debated among conservation professionals. This review, which included a variety of peer-reviewed articles and unpublished technical reports, revealed general support for the effectiveness of marsh terraces, although the magnitude of benefits and strength of empirical evidence varied greatly among objectives and studies. Evidence was strongest for benefits to improving nekton habitat, but was relatively sparse for benefits to improving waterbird habitat, reducing shoreline erosion rates, and creating emergent marsh outside the terrace footprints. Most existing studies relied on fewer than 3 study sites, which is unlikely to produce a representative sample of, and limits their applicability to, the larger universe of terrace sites. Indeed, results at individual sites may be influenced by numerous factors, including soil characteristics, hydrology, salinity, suspended sediment load, and various aspects of terrace construction design (Turner and Streever 2002; La Peyre et al. 2007, Rozas and Minello 2007). While study designs that pair terrace sites with nearby reference sites are able to control for some of these factors, few studies have yet attempted to identify which ones have the greatest impact (e.g., Rozas and Minello 2007, La Peyre et al. 2007). Such information will be essential for developing design recommendations to maximize the benefits of marsh terrace projects. Because of their relatively low cost, high probability of successful implementation, low probability of unintended negative consequences, and at least occasionally favorable results, marsh terraces are likely to remain a common feature in coastal restoration plans and strategies for the northern Gulf of Mexico. However, some coastal restoration programs are de-emphasizing marsh terracing projects because of uncertainty about the consistency and magnitude of marsh terrace benefits. Clearly, better science is needed to appropriately inform these decisions and ensure efficient allocation of coastal restoration resources.

Priorities for future research will vary among conservation organizations and funding sources, but the results of this review suggest emphasis should be placed on studies addressing effectiveness for marsh creation, shoreline erosion reduction, SAV production, and waterbird habitat enhancement. Future studies should incorporate a paired-site design (i.e., terrace sites paired with nearby reference sites) to help control for effects of spatially variable environmental factors. Studies should also employ greater replication across larger spatial and temporal scales, as this will increase the inferential scope of study results. Effective study designs will need to be tailored to evaluations of specific terrace objectives. For example, efforts to rigorously quantify benefits to SAV production and waterbird habitat enhancement will require logistically demanding *in situ* data collection of metrics such as vegetation presence/absence, vegetation biomass, waterbird abundance, and waterbird food abundance, among other habitat characteristics. In contrast, quantifying effects on marsh creation and shoreline erosion reduction may be accomplished using geospatial analysis of remotely sensed imagery, and preliminary work by conservation partners in the Gulf region (i.e., Ducks Unlimited and Gulf Coast Joint Venture [GCJV]) and efforts of earlier investigations (Castellanos and Aucoin 2004, Thibodeaux and Guidry 2009) have yielded promising results in this regard. Strongest inference for documenting the benefits to SAV production will likely be obtained through well-planned Before-After-Control-Impact designs (Stewart-Oaten et al. 1986), which are simple in concept but difficult in practice because of the need to identify and initiate data collection from terrace and reference sites months or years prior to construction.

Study designs capable of isolating and quantifying environmental and construction factors having the greatest impact on terrace project performance would be particularly valuable for identifying conditions and designs under which they are likely to be most successful. Design construction factors worthy of investigation may include water depth, soil conditions, suspended sediment load, terrace elevation and width, terrace configuration, and terrace density. For example, Steyer (1993) speculated that differences in observed marsh creation were attributable to variation in suspended sediment loads between sites. Miller and Guidry (2011) documented a failed terrace project in southwest Louisiana in which the terraces largely eroded and disappeared within 1 year after construction. Factors believed to have contributed to the ineffectiveness of the project included elevated water levels, unconsolidated soils resulting in poor structural integrity of the terraces, insufficient terrace height and width, and excessive spacing among terraces. These observations led Miller and Guidry (2011) to recommend incorporating a row of sacrificial terraces nearest the open water to afford interior terraces protection from wave energy long enough for them to become established by vegetation and thus more resistant to the erosive forces of waves. Interestingly, lessons learned from failed projects may be equally, if not more, valuable than those gleaned from successful projects.

Consideration of cost differentials among alternative terrace construction designs is also needed to ensure resources are allocated responsibly relative to expected benefits (Rozas and Minello 2007). Age since construction is another factor that undoubtedly affects the degree to which certain benefits may accrue (Steyer 1993), yet is probably the most poorly studied aspect of marsh terrace effectiveness. Finally, in addition to short-term research projects designed to address specific questions or hypotheses, agencies and programs responsible for funding and implementing marsh terraces should promote and adopt an adaptive management framework to ensure active learning and continual improvement occurs for this and other coastal restoration techniques (e.g., Coastal Protection and Restoration Authority 2012). The requirement for a 20-year monitoring plan for projects funded through CWPPRA has generated valuable information for evaluating the effectiveness of restoration techniques and represents an important stride towards adaptive management. The conservation community would be well-served if all funding entities required and provided resources to support similar monitoring efforts, albeit with an explicit mechanism by which monitoring results would be used to refine future restoration projects.

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Figure 1. Representative marsh terrace site in coastal Louisiana.