Title: The impacts of mangrove range expansion on wetland ecosystem services in the southeastern United States: current understanding, knowledge gaps, and emerging research needs

Running Title: Impacts of mangrove range expansion

List of Authors: Michael J. Osland[1], A. Randall Hughes[2], Anna R. Armitage[3], Steven B. Scyphers[2], Just Cebrian[4], Savannah H. Swinea[2], Christine C. Shepard[5], Michael S. Allen[6], Laura C. Feher[1], James A. Nelson[7], Cherie L. O'Brien[8], Colt R. Sanspree[9], Delbert L. Smee[10], Caitlin M. Snyder[11], Andrew P. Stetter[9], Phil W. Stevens[12], Kathleen M. Swanson[13], Lauren H. Williams[14], Janell M. Brush[15], Joseph Marchionno[15], Rémi Bardou[2]

Institutional Affiliations: [1]U.S. Geological Survey, Wetland and Aquatic Research Center, Lafayette, LA USA, [2]Northeastern University Marine Science Center, Nahant, MA USA, [3] Department of Marine Biology, Texas A&M University at Galveston, Galveston, TX USA, [4] Northern Gulf Institute, Mississippi State University, Stennis Space Center, MS USA, [5]The Nature Conservancy, Gulf of Mexico Program, Key West, FL USA, [6]University of Florida, Cedar Key, FL USA, [7]University of Louisiana at Lafayette, Lafayette, LA USA, [8]Texas Parks and Wildlife Department, Dickinson, TX USA, [9]U.S. Fish and Wildlife Service, Austwell, TX USA, [10]Dauphin Island Sea Lab, Dauphin Island, AL USA, [11]Apalachicola National Estuarine Research Reserve, Eastpoint, FL USA, [12]Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, St. Petersburg, FL USA, [13]Mission-Aransas National Estuarine Research Reserve, Port Aransas, TX USA, [14]The Nature Conservancy, Corpus Christi, TX USA, [15]Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, Gainesville, FL USA

Contact info for corresponding author: Michael Osland; mosland@usgs.gov; 919-308-0539

Abstract

Climate change is transforming ecosystems and affecting ecosystem goods and services. Along the Gulf of Mexico and Atlantic coasts of the southeastern United States, the frequency and intensity of extreme freeze events greatly influences whether coastal wetlands are dominated by freeze-sensitive woody plants (mangrove forests) or freeze-tolerant grass-like plants (salt marshes). In response to warming winters, mangroves have been expanding and displacing salt marshes at varying degrees of severity in parts of north Florida, Louisiana, and Texas. As winter warming accelerates, mangrove range expansion is expected to increasingly modify wetland ecosystem structure and function. Because there are differences in the ecological and societal benefits that salt marshes and mangroves provide, coastal environmental managers are challenged to anticipate effects of mangrove expansion on critical wetland ecosystem services, including those related to carbon sequestration, wildlife habitat, storm protection, erosion reduction, water purification, fisheries support, and recreation. Mangrove range expansion may also affect wetland stability in the face of extreme climatic events and rising sea levels. Here, we review current understanding of the effects of mangrove range expansion and displacement of salt marshes on wetland ecosystem services in the southeastern United States. We also identify critical knowledge gaps and emerging research needs regarding the ecological and societal implications of salt marsh displacement by expanding mangrove forests. One consistent theme throughout our review is that there are ecological trade-offs for consideration by coastal managers. Mangrove expansion and marsh displacement can produce beneficial changes in some ecosystem services, while simultaneously producing detrimental changes in other services. Thus, there can be local-scale differences in perceptions of the impacts of mangrove expansion into salt marshes. For very specific local reasons, some individuals may see mangrove expansion as a positive change to be embraced, while others may see mangrove expansion as a negative change to be constrained.

Keywords: climate change, coastal wetland, ecosystem services, mangrove, range expansion, winter climate change

Introduction

In response to warming air and ocean temperatures, temperate ecosystems are being transformed by tropical organisms whose range limits are expanding poleward (Vergés *et al.* 2014; Osland *et al.* 2021). In North America, the transition between tropical and temperate ecosystems is greatly influenced by the frequency and intensity of winter temperature extremes, as the northern range limits of most tropical organisms are governed by extreme freezing temperatures (Boucek *et al.* 2016; Osland *et al.* 2021). Climate change is producing warmer winters with fewer extreme freeze events (USGCRP 2017; Carter *et al.* 2018), which allows tropical organisms to move north of their current range limits. Coastal environmental managers near tropical-temperate transitions are increasingly faced with making natural resource management decisions related to the range expansion of these tropical organisms. Thus, there is a need to advance understanding of the ecological implications. In this communication, we review the literature on the ecological and societal impacts of warming winters and tropical range expansion within coastal wetland ecosystems in the southeastern United States, where tropical mangrove forests are expected to invade and ultimately displace salt marshes as they continue to move north (Osland *et al.* 2013; Cavanaugh *et al.* 2014; Gabler *et al.* 2017).

Mangrove forests and salt marshes are both highly productive coastal wetland ecosystems that occupy very similar geomorphic positions within tidal saline environments (Cahoon *et al.* 2020). Thus, there are many similarities in the ecosystem goods and services provided by these tree- and grass-dominated coastal wetlands. Both ecosystems are frequently ranked among the most valuable ecosystems on the planet (Costanza *et al.* 2014). In addition to providing habitat for fish and wildlife species, mangrove forests and salt marshes sequester large quantities of CO_2, protect coastlines from storms, reduce erosion, improve water quality, support productive fisheries, and provide recreational opportunities (Barbier *et al.* 2011).

Despite the similarities in the ecosystem goods and services provided by mangrove forests and salt marshes (Table 1), there are trade-offs and differences in the magnitude, spatiotemporal scale, and characteristics of the good or service provided (Ewel *et al.* 1998; Barbier *et al.* 2011; Kelleway *et al.* 2017). Research on the ecological effects of mangrove expansion in the southeastern United States has accelerated rapidly in recent years (Figure 1). Here, we synthesize this recent knowledge to review current understanding of the effects of mangrove range expansion and displacement of salt marsh on the ecosystem services provided by coastal wetlands in the southeastern United States. We begin with a background section that describes the history and expected future of mangrove expansion within this region. Next, based on the current state of knowledge, we review changes in wetland ecosystem services that occur as salt marshes are replaced by mangroves. We also identify critical knowledge gaps and emerging research needs for improving our understanding and management of mangrove expansion and salt marsh displacement in the southeastern United States.

Background: mangrove range expansion in the southeastern United States

The Gulf of Mexico and Atlantic coasts of North America are global hotspots for mangrove range expansion (Osland *et al.* 2017b; Cavanaugh *et al.* 2018) (Figure 2). Coastal wetlands are abundant in the southeastern United States due to the region's expansive low-lying coastal plains (Deegan *et al.* 1986; Enwright *et al.* 2016). In warmer and more tropical southern coastal reaches, tidal saline wetlands are dominated by mangrove trees and shrubs (Odum *et al.* 1982), with the three most common mangrove species in the region being *Avicennia germinans* (black mangrove), *Rhizophora mangle* (red mangrove), and *Laguncularia racemosa* (white mangrove). The northern range limits of these mangrove species (Figure 2) are controlled by extreme freeze events, which can lead to physiological damage and/or mortality (Sherrod & McMillan 1985; Pickens & Hester 2011; Cavanaugh *et al.* 2014; Bardou *et al.* 2021). Thus, along cooler, northern coastal reaches, tidal saline wetlands are dominated by grass-like salt marsh plants (i.e., grasses, sedges, and rushes) (Gabler *et al.* 2017; Osland *et al.* 2019a), because salt marsh plants can tolerate freezing temperatures via winter dormancy. Conversely, mangroves outcompete salt marsh plants along southern coastal reaches where temperatures are suitable for mangrove growth and canopy development (Kangas & Lugo 1990; Feher *et al.* 2017).

The northernmost mangrove populations in the region are present in Texas (Sherrod & McMillan 1981; Armitage *et al.* 2015), Louisiana (Osland *et al.* 2017a; Day *et al.* 2020; Osland *et al.* 2020b), and the Gulf of Mexico and Atlantic coasts of northern Florida (Stevens *et al.* 2006; Simpson *et al.* 2017; Snyder *et al.* 2021). Isolated mangrove individuals have also been found on Mississippi's barrier islands (Scheffel *et al.* 2013; Scheffel *et al.* 2017; Macy *et al.* 2019). In the past century, mangrove range limits have expanded across the region during freeze-free years and contracted due to mass mortality during extreme freeze events (Sherrod & McMillan 1985; Brown *et al.* 2016; Osland *et al.* 2017a; Cavanaugh *et al.* 2019). The last major freeze event resulting in region-wide mass mangrove mortality and range contraction occurred in December 1989 (Lonard & Judd 1991; Stevens *et al.* 2006; Osland *et al.* 2017a). Since then, mangroves have been expanding in parts of northern Florida, Louisiana, and Texas. Although 1989 was the last cold event to affect mangroves across the entire northern Gulf of Mexico, there have been several smaller, less intense cold events (e.g., 1996, 2000, 2001, 2002, 2003, 2010, 2011, 2014, 2018, 2021), which have caused short-term mangrove damage, mortality, and/or local reductions in coverage (e.g., Osland *et al.* 2015; Osland *et al.* 2017a; Osland *et al.* 2019b; Osland *et al.* 2020a; Osland *et al.* 2020b; Snyder *et al.* 2021). The most recent event (February

2021) caused varying levels of mangrove damage and mortality across the Texas coast [Anna Armitage (Texas A&M Galveston) and Kathleen Swanson (Mission Aransas National Estuarine Research Reserve), oral communication, March 2021). In response to future warming, mangrove forests are expected to expand farther north and displace salt marshes in much of Texas, Louisiana, and northern Florida (Osland *et al.* 2013; Cavanaugh *et al.* 2015; Gabler *et al.* 2017; Cavanaugh *et al.* 2019).

Inundation and salinity regimes are critical abiotic factors that govern ecosystem structure and function in coastal wetlands (Ibáñez *et al.* 2012; Twilley & Day 2012). Thus, across the southeastern United States, there is some variation in the salt marsh plant communities that are being replaced by range-expanding mangrove forests (Yando *et al.* 2016; Gabler *et al.* 2017; Osland *et al.* 2019a) (Figure 3). In general, salt marshes are dominated by grass-like (graminoid) plants (e.g., *Spartina alterniflora*, *Juncus roemerianus*, *Spartina patens*, *Schoenoplectus americanus*) near mangrove range limits in northern Texas, Louisiana, and northern Florida (Yando *et al.* 2016; Brockmeyer *et al.* 2017; Gabler *et al.* 2017; Osland *et al.* 2019a; Chapman *et al.* 2021). Salt marshes in all three of these areas receive comparatively large freshwater inputs that maintain salinities below or near ocean waters (NOAA 1990; USEPA 1999; Osland *et al.* 2014). In contrast, along more arid coasts that receive less rainfall and smaller riverine freshwater inputs, such as the lower and central Texas coast, hypersaline conditions can develop when high evaporation concentrates oceanic salts (Longley 1995; Withers 2002b; Montagna *et al.* 2007; Osland *et al.* 2014). Halophytic succulent plant species (e.g., *Batis maritima*, *Salicornia depressa*, *Borrichia frutescens*, *Monanthochloe littoralis*) tend to dominate hypersaline salt marsh plant communities (Yando *et al.* 2016; Gabler *et al.* 2017; Osland *et al.* 2019a). Limited connectivity with the ocean can also produce hypersaline conditions and halophytic succulent plant-dominated salt marshes, as in parts of the Indian River Lagoon along the Atlantic coast of Florida (Brockmeyer *et al.* 2017; Simpson *et al.* 2019; Chapman *et al.* 2021). These examples show that regional variation in climate- and hydrology-controlled salt marsh plant community composition and structure is important to consider as it can influence how wetland ecosystem services change due to mangrove expansion and salt marsh displacement.

How do mangrove range limits in the southeastern United States compare with other mangrove range limits across the world? At the global scale, mangrove range limits are governed by many factors including winter air temperature regimes, aridity, ocean temperatures, and dispersal limitations (Duke *et al.* 1998; Saenger 2002; Osland *et al.* 2017b; Van der Stocken *et al.* 2019a). Winter air temperatures affect mangrove range limits to varying degrees in Australia, New Zealand, South Africa, Brazil, and China (Quisthoudt *et al.* 2012; Osland *et al.* 2017b). Due to the movement of cold air outbreaks from the arctic across continental land masses in the Northern Hemisphere, mangrove range limits in the Northern Hemisphere (i.e., North America and China) are affected by colder winter air temperature extremes than range limits in the Southern Hemisphere (i.e., Australia, New Zealand, South Africa, Brazil) (Osland *et al.* 2017b; Smith & Sheridan 2020; Osland *et al.* 2021). Thus, mangrove range expansion is more dynamic and more strongly influenced by cold temperature extremes in the southeastern United States (Osland *et al.* 2017a; Cavanaugh *et al.* 2018; Cavanaugh *et al.* 2019) and China (Chen *et al.* 2017) compared to Australia, New Zealand, South Africa, and Brazil (Osland *et al.* 2017b).

While mangrove expansion is occurring on several other continents, there is much variation in the environmental settings and the drivers responsible these changes (Saintilan & Williams 1999; Saintilan *et al.* 2014; Rogers & Krauss 2019). For example, changing rainfall regimes (Diop *et al.* 1997; Eslami-Andargoli *et al.* 2009), rapid sedimentation (Lovelock *et al.* 2010; Asbridge *et al.* 2015; Walcker *et al.* 2018), hydrologic alterations (Raabe *et al.* 2012), and rising sea levels (Krauss *et al.* 2011; Howard *et al.* 2020) are all common drivers of mangrove expansion. These drivers and the accompanying differences in environmental setting (e.g., geomorphology, climate, and the ecosystem properties of the interacting salt marshes and mangrove forests) greatly influence the ecological implications of mangrove expansion. Our review focuses primarily on the southeastern United States to avoid confounding interpretations due to such differences.

Local impacts and trade-offs

At global and regional scales, coastal wetlands are typically valued for their support of multiple ecosystems services (Barbier *et al.* 2011; Costanza *et al.* 2014). However, at local scales (e.g., a specific wetland within a refuge, park, or neighborhood), coastal management actions are sometimes motivated by an interest to maintain or enhance a specific ecosystem good or service. For example, if a wetland is valued for waterfowl hunting, management actions may be guided primarily by an interest to maintain or enhance waterfowl habitat (Mitchell *et al.* 2006). Conversely, if an urban municipal wetland provides valuable recreation opportunities, wetland management may be driven by an interest to optimize public recreation (Zedler & Leach 1998). Across the southeastern United States, there is much variation in the coastal wetland ecosystem services that are prioritized by coastal communities and managers (Feagin *et al.* 2010; Engle 2011; Yoskowitz *et al.* 2012). Thus, there can be local-scale differences in perceptions of the ecological and societal impacts of mangrove expansion into salt marshes. For very specific local reasons, some individuals may see mangrove expansion as a positive change to be embraced, while others may see mangrove expansion as a negative change to be constrained (Table 2; Figure 4). At several points in this communication, we have included specific examples that illustrate some of the variation in local perceptions of the impacts and tradeoffs associated with mangrove expansion.

Carbon sequestration

How will mangrove range expansion and encroachment into salt marshes affect carbon cycling and storage in coastal wetlands? The short answer is that mangrove expansion is expected to universally increase aboveground carbon storage, but the effects of mangrove expansion on soil carbon storage are variable and heavily influenced by site-specific abiotic and biotic conditions, as described below.

One of the most striking and consistent effects of mangrove expansion is an increase in canopy height and aboveground carbon stocks (Yando *et al.* 2016; Feher *et al.* 2017; Gabler *et al.* 2017; Simpson *et al.* 2017). Mangroves are woody plants that can rapidly accumulate aboveground carbon stocks during the early stages of forest development (Lovelock *et al.* 2010; Walcker *et al.* 2018; Osland *et al.* 2020d; Chapman *et al.* 2021). In contrast, salt marshes are dominated by herbaceous plants that lose their aboveground biomass each year during winter senescence (Macy *et al.* 2020). Thus, mangrove forests are typically taller and contain more aboveground biomass than their salt marsh counterparts. Studies conducted in mangrove-marsh ecotones across the southeastern United States have consistently shown that mangrove expansion into salt marsh results in taller plants, more aboveground biomass, and increases in aboveground carbon stocks (Perry & Mendelssohn 2009; Doughty *et al.* 2016; Hutchison 2016; Yando *et al.* 2016; Hutchison *et al.* 2018; Simpson *et al.* 2019; Charles *et al.* 2020; Macy *et al.* 2020).

However, the magnitude of those increases in aboveground carbon stocks and storage rates can be influenced by other factors, including geomorphic position and interactions between aridity, salinity, and plant productivity (Yando *et al.* 2016; Gabler *et al.* 2017; Osland *et al.* 2018b).

What are the implications of mangrove expansion for belowground carbon cycling and storage? Soil carbon burial represents the largest long-term carbon storage pathway within coastal wetlands (Chmura *et al.* 2003; Breithaupt *et al.* 2012). Thus, there has been much interest and debate regarding the soil carbon implications of mangrove expansion. Mangrove forests and salt marshes are both highly productive ecosystems that, on a per unit area basis, have the potential to support soil carbon burial rates that are among the highest on the planet (Donato *et al.* 2011; Mcleod *et al.* 2011). The factors that contribute to rapid soil organic matter development in both mangrove forests and salt marshes include high rates of primary productivity, prolific belowground root production, low rates of decomposition due to anaerobic conditions, and continued organic matter burial as wetlands trap sediments and build elevation to adjust to rising sea levels (Cahoon *et al.* 2020). As in terrestrial grasslands, where there is much variation in the edaphic effects of woody plant encroachment due to climate-plant trait interactions (Barger *et al.* 2011; Eldridge *et al.* 2011; Archer *et al.* 2017), studies in coastal wetlands have revealed divergent, nuanced results regarding the effects of mangrove expansion on soil carbon cycling and storage. While some studies have found no measurable effect of mangrove expansion on soil carbon stocks (Perry 2007; Perry & Mendelssohn 2009; Henry & Twilley 2013; Doughty *et al.* 2016; Yando *et al.* 2018; Charles *et al.* 2020; Macy *et al.* 2020), others have found increases in soil carbon associated with mangrove expansion (Bianchi *et al.* 2013; Simpson *et al.* 2019; Vaughn *et al.* 2020). When viewed collectively, these studies indicate that the edaphic and soil carbon effects of mangrove expansion are highly site dependent and greatly influenced by the plant traits and ecosystem properties of the interacting salt marshes and mangrove forests (Yando *et al.* 2016; Osland *et al.* 2018b; Charles *et al.* 2020) (Figure 3). In the subsequent paragraphs, we briefly examine several alternative hypotheses regarding the divergent soil carbon-focused findings in the literature.

Biomass-based hypotheses regarding changes in soil carbon cycling and storage due to mangrove expansion presume that increases in aboveground biomass will be accompanied by concomitant increases in belowground carbon storage. Across some coastal wetland abiotic gradients (e.g., nutrient limitation gradients or extreme salinity gradients), there can be strong positive relationships and positive feedbacks between coastal wetland plant biomass, plant productivity, and soil carbon stocks and cycling (McKee *et al.* 2007; Kauffman & Bhomia 2017; Osland *et al.* 2018b; Rovai *et al.* 2018). However, aboveground biomass increases due to mangrove expansion do not necessarily affect net soil carbon cycling and storage. Indeed, across the tropical-temperate transition zone in eastern North America, which spans productive mangrove forests in the warmer south and productive salt marshes in the colder north, coastal wetland aboveground biomass varies greatly but is not correlated to soil carbon stocks or soil carbon burial (Chmura *et al.* 2003; Feher *et al.* 2017; Holmquist *et al.* 2018; Osland *et al.* 2018b). Under comparable geomorphic conditions, mangrove forests and salt marshes have the potential to support similarly high soil carbon stocks and soil carbon accumulation rates. Thus, instead of being linked directly to aboveground biomass, mangrove-triggered changes in coastal wetland soil properties and soil carbon burial rates, where present, are more likely linked to changes in other processes (e.g., productivity, decomposition). Moreover, where mangrove expansion is occurring across a salinity or elevation gradient (e.g., Ross *et al.* 2000; Krauss *et al.* 2011; Lewis *et al.* 2021), the effects of mangrove expansion may be due to concomitantly changing abiotic conditions that are governed by other drivers (e.g., saltwater intrusion, rising sea levels).

In general, more productive coastal wetland ecosystems tend to support higher soil carbon burial rates and the rapid development of soil carbon stocks (McKee *et al.* 2007; Kauffman & Bhomia 2017; Osland *et al.* 2018b; Rovai *et al.* 2018). Thus, where decomposition rates are similar (e.g., Geoghegan *et al.* 2020), the soil carbon implications of mangrove expansion may be heavily influenced by the productivity of the interacting salt marsh and mangrove plant communities. Due in part to differences in leaf C:N ratios, aboveground leaf litter decay rates can be higher in *A. germinans* compared to *S. alterniflora* (Perry & Mendelssohn 2009; Smith *et al.* 2019; Simpson *et al.* 2020). However, the belowground abiotic conditions in mangroves and marshes along Florida's Atlantic coast produced similar belowground decomposition rates (Simpson *et al.* 2020). If a salt marsh and an expanding mangrove forest both support similarly high productivity rates and similar belowground decomposition rates, then there may be no effect of mangrove expansion on soil carbon burial rates. For example, studies conducted within highly productive salt marshes dominated by graminoid plants in Louisiana and Florida have found no changes in soil carbon storage due to mangrove expansion (Perry 2007; Perry & Mendelssohn 2009; Henry & Twilley 2013; Doughty *et al.* 2016; Yando *et al.* 2016; Yando *et al.* 2018; Macy *et al.* 2020). In contrast, soil carbon burial may increase if the expanding mangrove forest supports higher rates of productivity, especially belowground (root) productivity, compared to the incumbent salt marsh. For example, several studies conducted within salt marshes dominated by succulent plants along the central Texas coast (Bianchi *et al.* 2013; Yando *et al.* 2016), in Australia (Kelleway *et al.* 2016), and in Florida (Simpson *et al.* 2019) have noted soil carbon increases associated with mangrove expansion. Beyond just carbon quantity, soil carbon quality can be affected by mangrove expansion (Lewis *et al.* 2014; Breithaupt *et al.* 2020; Charles *et al.* 2020; Vaughn *et al.* 2020). For example, along the central Texas coast, succulent marsh plant (*B. maritima*) leaf and root breakdown rates were 1000% and 35% faster, respectively, than *A. germinans* (Charles et al. 2020). The effects of mangrove expansion on autochthonous detrital inputs and allochthonous carbon burial via changes in sedimentation (Guo *et al.* 2017; Chen *et al.* 2018; Charles *et al.* 2020; McKee *et al.* 2020; Kuhn *et al.* 2021) also warrant further investigation.

In summary, although mangrove expansion is universally expected to increase aboveground carbon storage, the effects on soil carbon storage are highly variable and warrant closer investigation via field- and greenhouse-based manipulative experiments as well as measurements from a larger number of mangrove-marsh ecotones across the region (Table 3). There is also a pressing need for studies that, instead of measuring carbon stocks, directly measure critical processes (e.g., root productivity, decomposition, sediment deposition) that directly affect soil carbon storage and cycling. Species-specific effects on soil carbon quality and quantity also warrant further investigation. For example, most studies have focused on the effects of *A. germinans* expansion, but due to differences in plant traits, the edaphic effects of *R. mangle* expansion may differ from *A. germinans* (McKee 1993). There is also a need for species-specific salt marsh research that moves beyond just *S. alterniflora*. Although there are 24 dominant foundation plant species in tidal saline wetlands across the conterminous United States, 45 percent of publications have been focused on *S. alterniflora* (Osland et al. 2019a).

Soil elevation change and vertical adjustments to sea-level rise

Mangrove forests and salt marshes have the potential to adjust to moderate rates of rising sea levels via soil elevation gains that are driven by positive feedbacks between inundation, plant growth, and sediment deposition (Morris *et al.* 2002; Krauss *et al.* 2014b). Given the vulnerability of coastal wetlands to accelerated sea-level rise, will mangrove expansion increase soil elevation building capacity and the ability of coastal wetlands to adjust to rising sea levels? This question has been investigated through the use of radioisotopes, sediment traps, marker horizons, and surface elevation tables. Many of the processes that affect soil carbon storage and cycling (e.g., root productivity, belowground decomposition, and sediment deposition) are also the primary processes that govern soil elevation change. Thus, given the variable and highly site-specific effects of mangrove expansion on soil carbon storage, we expect that the effects of mangrove expansion on soil elevation change dynamics are also highly context-dependent and governed by site- and species-specific conditions (e.g., geomorphology, abiotic conditions, and the ecosystem properties of the interacting salt marsh and mangrove forest).

The surface elevation table-marker horizon (SET-MH) approach (Cahoon *et al.* 2002; Lynch *et al.* 2015) has been used to directly measure shorter-term (e.g., 5 to 30 year) soil elevation change dynamics and quantify the contributions of critical above and belowground processes (e.g., accretion, erosion, subsurface root zone expansion, subsurface subsidence). There are only a few studies that have directly measured the soil elevation change implications of mangrove expansion using the SET-MH approach. An SET-MH study conducted along the Atlantic coast of Florida within *Distichlis spicata*-dominated marshes found higher elevation gains associated with warming temperatures and mangrove (*L. racemosa*) expansion; vertical accretion was about threefold higher in mangrove plots exposed to a warming temperature treatment compared to salt marshes exposed to the same treatment (Coldren *et al.* 2019). In contrast, an SET-MH study conducted in Louisiana found similar rates of soil elevation gain in creekbanks dominated by *A. germinans*, *S. alterniflora*, or a mixture of both species (McKee & Vervaeke 2018). A hurricane sediment-focused study conducted using soil cores from within the same Louisiana mangrove-marsh ecotone found no difference in sediment capture by stands dominated by *S. alterniflora* or *A. germinans* (McKee *et al.* 2020). A comparison of short-term accretion rates in the Louisiana mangrove-marsh ecotone, measured using sediment traps (biweekly accretion) and feldspar marker horizons (annual accretion), also found no difference in stands dominated by *S. alterniflora* or *A. germinans* (Perry & Mendelssohn 2009). These results suggest that replacement of marsh by mangroves would have no effect on sediment capture in the Louisiana mangrove-marsh ecotone. However, as discussed in these papers and others across the region, the lack of a difference may be unique to the sedimentary setting, the size and developmental stage of the mangroves, and/or to the specific species compared (Perry 2007; McKee & Vervaeke 2018; Charles *et al.* 2020; McKee *et al.* 2020; Kuhn *et al.* 2021).

These divergent findings indicate that there is a need to further investigate the effects of mangrove expansion on soil elevation change dynamics (Table 3). Most existing studies have compared adjacent mangrove and marsh stands. However, these is also a need for manipulative experiments in which marsh plots are experimentally planted with mangrove seedlings and compared to marsh plot controls. Such comparisons would enable direct measurements of changes in soil elevation dynamics that may occur when marsh is replaced by mangroves.

Although there have been several studies that have used radioisotope-based methods to compare vertical accretion rates in mangroves and salt marshes near poleward mangrove range limits in eastern North America (Perry & Mendelssohn 2009; Comeaux *et al.* 2012; Bianchi *et al.* 2013; Vaughn *et al.* 2020), century- and multi-decadal scale mangrove expansion patterns near range limits can be dynamic and include freeze-controlled mangrove-marsh expansion and contraction cycles (Osland *et al.* 2017a; Cavanaugh *et al.* 2019). Thus, radioisotope-based approaches should ideally be paired with historical analyses of imagery (e.g., Perry & Mendelssohn 2009) and climate data to identify freeze event-driven oscillations in mangrove and/or salt marsh coverage during the time periods represented by soil cores.

Water quality: nutrient and sediment retention

How does mangrove expansion and salt marsh displacement affect the water quality-based ecosystem services provided by coastal wetlands? Salt marshes and mangroves lie at the land-ocean interface, where they can improve water quality by filtering pollution inputs both from the ocean (e.g., tidal and wave-driven inputs) and from the inland watershed (e.g., surface and groundwater inputs) (Mitsch & Gosselink 2007; Mitsch *et al.* 2015). The main mechanisms through which nitrogen inputs are filtered in these coastal wetlands are plant uptake and denitrification (Sparks *et al.* 2015; Steinmuller *et al.* 2019). While denitrification represents a net loss of nitrogen from the system, and thus complete filtration, plant uptake only constitutes temporary filtration, because the nutrients bound to plant tissues can be delivered back into the coastal environment through decomposition. There have been many studies of nutrient cycling and storage in salt marshes and mangroves (e.g., Mozdzer *et al.* 2011; Simpson *et al.* 2013; Hunter *et al.* 2015; Weaver & Armitage 2018; Dangremond *et al.* 2020; Craig *et al.* 2021; Martin *et al.* 2021; and references cited therein).

Based on the current literature, it is difficult to predict how the replacement of salt marshes by mangroves may affect nutrient pollution filtration in coastal wetlands as the existing studies are spatially and temporally disjointed. Many environmental factors other than changes in dominant vegetation can affect the differences observed between salt marsh- and mangrove-focused studies. Long-term studies that directly record changes in nutrient cycling and storage as mangroves replace salt marshes would be informative; however, we are unaware of such studies. Another alternative is a ‘space-for-time substitution’ approach (Pickett 1989), where nutrient cycling and storage are compared between contiguous stands of salt marsh and mangroves present across ecotones where the expansion of mangroves and replacement of salt marshes is actually occurring. A few such ‘space-for-time substitution’ studies exist (e.g., Henry 2012; Macy *et al.* 2019; Steinmuller *et al.* 2019; Macy *et al.* 2020) and provide some indication of the changes that will occur with mangrove expansion. These studies show higher nitrogen content in the aboveground tissues of black mangroves (*A. germinans*) than in marsh smooth cordgrass (*S. alterniflora*) (McKee & Rooth 2008; Macy *et al.* 2020). Higher aboveground biomass and aboveground nitrogen stocks (Macy et al. 2020) suggest higher nitrogen uptake from the soil by plants in *A. germinans* than in *S. alterniflora* stands, which is consistent with the lower nitrogen concentrations observed in the soil porewater of *A. germinans* in comparison with *S. alterniflora* stands (Macy et al. 2020). All together, these results suggest that *A. germinans*, by removing larger quantities of nitrogen from the soil, may be larger filters of nitrogen pollution than *S. alterniflora*. However, more studies are needed for a clear and robust description of the effects of salt marsh replacement by mangroves on nitrogen cycling and storage, and ultimately nitrogen pollution filtration in coastal wetlands.

Sediment and sediment-bound phosphorus retention is another process through which salt marshes and mangroves can improve water quality in coastal ecosystems. Wetlands with more near-surface plant structural complexity (e.g., greater stem density, biomass, flexibility) can be more effective at retaining sediments by slowing down the flow of incoming and outgoing waters

(Feagin *et al.* 2015; Chen *et al.* 2018; McKee *et al.* 2020). A hurricane-focused study conducted in Louisiana found that despite differences in plant structure there was no difference in hurricane sediment capture within stands dominated by *S. alterniflora* or *A. germinans* (McKee *et al.* 2020). Conversely, a study conducted along the central Texas coast within a mangrove-marsh ecotone that included *A. germinans* and succulent plants (e.g., *B. maritima*) found that hurricane sediment accretion decreased with increasing mangrove cover (Kuhn *et al.* 2021). Despite these two hurricane-focused studies, there are no studies that fully examine how salt marsh replacement by mangroves in this region may alter wetland sediment retention and soil build-up under dynamic conditions ranging from daily tidal cycles to storms. More research is needed to better understand how salt marsh replacement by mangroves alters wetland sediment retention, nutrient retention, and associated water quality-based ecosystem services (Table 3).

Coastal protection

Given the high vulnerability of coastal communities to storms, what is the potential for mangrove expansion to enhance the coastal protection benefits provided by coastal wetlands? Mangroves and salt marsh ecosystems both can reduce the height and energy of waves passing through them (Gedan *et al.* 2011; Marois & Mitsch 2015), increase soil strength (Sasser *et al.* 2018; Jafari *et al.* 2019), and facilitate sediment deposition (Cahoon *et al.* 2020). These plant-controlled processes can lead to reductions in shoreline erosion, decreases in flooding extent, and avoided damages to infrastructure (Arkema *et al.* 2013; Barbier 2016; Narayan *et al.* 2019). The ability of wetlands to support these coastal protection ecosystem services is dependent upon multiple factors ranging from the characteristics of the individual wetland plants to landscape-scale ecosystem coverage and configuration. Coastal protection services are also influenced by geomorphology and the types and magnitudes of the hazard forces. The structural differences between mangroves and salt marshes, with mangroves being generally larger, more complex and rigid, have led to the hypothesis that coastal protection services may increase following mangrove expansion into salt marsh.

Existing data, from a range of modeled computer simulations, natural experiments, wave tank studies, and semi-controlled field experiments indicate that mangroves may provide more coastal protection value than salt marshes. For example, in a recent field study conducted within large experimental mangrove removal plots in Texas, Pennings et al. (2021) found that mangroves are more effective at preventing erosion than succulent plant-dominated salt marshes, presumably due to: (1) the enhanced wave-buffering effects of taller, stiffer mangrove stems; and (2) the soil-strengthening effects of greater mangrove root biomass. In a model-based assessment that was parametrized with local field-based measurements of vegetation structure along the Atlantic coast of Florida, Doughty et al. (2017) found that wave attenuation and erosion prevention were greater in mangroves than graminoid-dominated salt marshes, which was attributed to the larger canopy heights and stem diameters of mangrove trees. Similarly, in a model-based comparison of mangroves and graminoid-dominated marshes in Louisiana, Hijuelos et al. (2019) noted that wave attenuation was greater in mangroves than graminoid-dominated marshes. All three of these studies indicate that there may be a gain in coastal protection services associated with mangrove expansion, which is especially important given that climate change is increasing the frequency of major hurricanes (Kossin *et al.* 2017). However, there is a need for more empirical evaluations of how salt marsh versus mangrove shorelines influence wave attenuation and storm impacts on civic infrastructure.

Within the context of coastal erosion, there is interest in advancing understanding of mangrove expansion effects on the ability of wetlands to tolerate and recover from extreme events and other disturbances (e.g., droughts, hurricanes, floods, freezes, and oil spills). For example, in the Galveston region, frequent oil spills are a concern for resource managers because they can lead to vegetation dieback (Rozas *et al.* 2000; Williams *et al.* 2017). For many coastal wetland practitioners in the Galveston area, there is interest in maximizing the ability of wetlands to recover from potential future oil spills [Cherie O'Brien (Texas Parks and Wildlife Department), oral communication, 24 May 2021]. However, there is some local concern that expanding mangroves (*A. germinans*) may decrease wetland resilience to oil spills [Cherie O'Brien (Texas Parks and Wildlife Department), oral communication, 24 May 2021] (but see: Hughes *et al.* 2018). Thus, during coastal wetland restoration efforts in the Galveston area within the past decade, *S. alterniflora* has typically been the targeted species, and very few Texas Parks and Wildlife permits (i.e., the Permit to Introduce Fish, Shellfish or Aquatic Plants into Public Waters) have been granted to plant mangroves [Cherie O'Brien (Texas Parks and Wildlife Department), oral communication, 24 May 2021]. This example highlights the importance of considering the effects of extreme events on wetland stability and coastal protection benefits. Another example in this arena comes from Louisiana, where landscape-scale marsh and mangrove dieback can be triggered by extreme events. For example, drought can lead to marsh dieback (McKee *et al.* 2004; Alber *et al.* 2008) and extreme cold events can lead to mangrove dieback (Osland *et al.* 2017a). Mangroves (*A. germinans*) may be more drought tolerant than *S. alterniflora* marshes in Louisiana due to lower water use (Krauss et al. 2014a). Conversely, extreme freeze events similar to the 1989 freeze could lead to landscape-scale mangrove mass mortality, peat collapse, and accelerated erosion (McBride & Byrnes 1997; Penland *et al.* 2003;

Osland *et al.* 2020b), which would compromise the coastal protection benefits provided by expanding mangrove forests.

Coastal food webs and fisheries

Marshes and mangrove forests support critical coastal food webs and fisheries-based ecosystem services (Rönnbäck 1999; Beck *et al.* 2001; Minello *et al.* 2003; Nagelkerken *et al.* 2008; Engle 2011). However, due to the structural and functional differences between the two ecosystems, how will mangrove range expansion and marsh displacement affect coastal food webs, fish communities, and key ecosystem services (e.g., fisheries, fisheries-based recreation)?

Overall, the current evidence suggests that mangrove range expansion may lead to changes in associated fish and invertebrate community composition (Armitage *et al.* 2021), but there is less evidence that mangroves will substantially alter the major production sources to aquatic food webs. Stable isotope analyses indicate that microphytobenthos and phytoplankton are the dominant energy channels that directly support many aquatic consumer groups (e.g., fishery species, wetland specialists, zoobenthivores, phytodetritovores, and planktivores) (Baker *et al.* 2021). Contributions from both mangrove and marsh plants typically flow through the detrital pathway and play a lesser role in directly supporting food webs (Nelson *et al.* 2019; Baker *et al.* 2021). In Louisiana, no detrital inputs from mangroves contributed significantly to the dominant consumers in the food web (Nelson *et al.* 2019). It is uncertain if mangroves will ultimately replace marsh grass detritus as an energy source (Nelson *et al.* 2019; Harris *et al.* 2020; James *et al.* 2020). However, reductions in marsh habitat as a result of mangrove expansion may lead to a decline in energy supplied to some consumers, as mangrove detritus is not readily used as a food source by the current nekton species in Louisiana (Harris *et al.* 2020).

A study in Texas also noted that differences in leaf nutritive quality (e.g., higher tannin, phenolic, and lignin contents) may be linked to lower fitness for benthic consumers that consume mangrove detritus (Goeke & Armitage 2021).

Most comparative studies of mangroves and marshes have detected differences in food webs and community structure; however, those differences are not universal and can vary depending on the measured taxonomic group, community metric, or functional attribute (Diskin & Smee 2017; Loveless & Smee 2019; Walker *et al.* 2019; Armitage *et al.* 2021). For instance, the relative abundance and species richness of nektonic fish and invertebrates in Texas were similar between sites with and without mangroves, though individual species varied considerably across those same sites (Armitage *et al.* 2021). Differences among areas with and without mangroves can also be masked by abiotic conditions or disturbances that influence associated faunal abundance and diversity (Diskin & Smee 2017). For example, Smee et al. (2017) found significant differences in nekton and infaunal community structure in stands of smooth cordgrass (*S. alterniflora*) bordered by mangroves (*A. germinans*) versus stands without mangrove. Infaunal organisms, blue crabs (*Callinectes sapidus*), and shrimp (*Farfantepenaeus aztecus, Palaemonetes* spp.) were less abundant in marshes bordered by mangroves than in marshes without mangroves, whereas mud crabs (Xanthidae/Panopeidae) and fish were more abundant in marshes bordered by mangroves (Smee *et al.* 2017). Similarly, macrofaunal communities differed in marsh- versus mangrove-dominated vegetation along the Atlantic coast of Florida, due to the influence of fine-scale plant structural differences and stand-level habitat attributes (Johnston & Gruner 2018). Along the Atlantic coast, studies have noted differences in detrital-based epifaunal communities (Smith *et al.* 2019) and subtidal nekton communities (Kimball & Eash-Loucks 2021) associated with mangroves compared to marshes. For example, Smith et al.

(2019) found that crabs were 2-7 times more abundant in *S. alterniflora* detritus than *A. germinans* detritus. In an early study comparing nekton communities in marsh and mangroves in Louisiana, Caudill (2005) found that while fishes [e.g., gulf killifish (*Fundulus grandis*) and sheepshead minnow (*Cyprinodon variegatus*)] were more abundant in *S. alterniflora*-dominated marshes, white shrimp (*Litopenaeus setiferus*) were more abundant in *A. germinans*-dominated mangrove stands.

Beyond just the direct effects to animals, mangrove expansion can also affect microbial (Barreto *et al.* 2018) and rhizosphere communities (Chen *et al.* 2020), which can affect abiotic conditions, biotic interactions, and biogeochemical cycling. The effects of mangrove expansion on coastal food webs and fisheries may also increase with time due to the influence of forest developmental stage (Barimo & Serafy 2003; Scheffel *et al.* 2018). Linkages to adjacent ecosystems can also be important. For example, along the northwestern coast of Florida, mangrove expansion affects the seasonal delivery of organic matter to adjacent seagrass ecosystems, which can alter epifaunal communities, subtidal environmental conditions, gas fluxes, and seagrass standing biomass (Sullivan *et al.* 2021).

Mangroves can also impact coastal food webs via changes in habitat complexity that affect key species or predator-prey interactions (Scheffel *et al.* 2017; Johnston & Smith 2018). Habitat complexity influences feeding efficiency and predation rates in fishes (Gotceitas & Colgan 1989; Ahrens *et al.* 2012), and thus we expect that a shift from salt marsh to mangrove will influence fish foraging and community composition. From the existing studies, the consensus is that grass shrimps (*Palaemonetes* spp.), blue crabs (*C. sapidus*), and some marsh-associated fishes (e.g., *F. grandis*) are more abundant in marshes compared to mangrove habitats (Caudill 2005; Johnston & Caretti 2017; Smee *et al.* 2017; Armitage *et al.* 2021). However, there are some inconsistencies in findings for penaeid shrimp that may be attributed to the type of predators present at a given location. For example, penaeid shrimp preferred marsh grass over mangroves in experimental trials, likely due to increased protection from predatory gray snapper (*Lutjanus griseus*) (Scheffel *et al.* 2017). On the other hand, benthic invertebrates, such as penaeid shrimp and the marsh periwinkle (*Littoraria irrorata*), were less vulnerable to predation by blue crabs in the presence of mangroves in mesocosm and field studies, respectively (Glazner *et al.* 2020; Glazner *et al.* 2021). The aerial root structures of mangroves (i.e., pneumatophores and prop roots) can provide refuge from benthic predators, which could lead to shifts in predator-prey interactions and altered trophic dynamics (Glazner *et al.* 2020; Glazner *et al.* 2021). In Louisiana, penaeid shrimp foraging in mangroves had higher trophic levels than those foraging in marsh-dominated habitats (Nelson *et al.* 2019). Increased predation on small benthic infauna could feedback to other parts of the food web with uncertain consequences. Mangroves can have additional negative impacts on blue crabs beyond the crab-shrimp interaction, with juvenile crabs exhibiting a habitat preference for and higher survival in marsh grasses in experimental trials (Johnston & Caretti 2017).

Much of the existing literature has focused on the effects of *A. germinans* expansion; however, the impacts due to the range expansion of other common mangrove species may be different. For example, the habitat quality characteristics for fishes differ between red mangrove (*R. mangle*) and black mangrove (*A. germinans*), with the former having prop roots and the ability to grow at lower intertidal elevations, thus influencing fish habitat inundation time and depth. In areas where *R. mangle* occurs, overhanging limbs and prop roots can extend fringing shoreline habitat another 10 m into the water. The overhanging vegetation and prop roots support fishes by providing shade and cover (Ley *et al.* 1999; Ley & McIvor 2002). Because mangroves are almost always located in shallow water and often in sheltered environments, it can be difficult to isolate the effects of *R. mangle* prop roots as fish habitat (Ellis & Bell 2004). A study using an experimental design with artificial mangroves found that the structure provided by *R. mangle* prop roots can support unique juvenile fish assemblages (Nagelkerken & Faunce 2008). In estuaries of southwestern Florida, *R. mangle*-dominated shorelines provide habitat for larger-bodied species such as common snook (*Centropomus undecimalis*), sheepshead (*Archosargus probatocephalus*), striped mullet (*Mugil cephalus*), gray snapper (*L. griseus*), redfin needlefish (*Strongylura notata*), striped mojarra (*Eugerres plumieri*), and Atlantic spadefish (*Chaetodipterus faber*) (Greenwood *et al.* 2007). The large-bodied species that associate closely with *R. mangle* prop roots are often plastic in their habitat use and may not be fully dependent on mangroves (e.g., Stevens *et al.* 2018). However, there are other species that exhibit greater dependency on *R. mangle* habitat, such as juvenile smalltooth sawfish (*Pristis pectinata*) and goliath grouper (*Epinephelus itajara*). Studies of fine-scale habitat use show that juvenile smalltooth sawfish feed on shallow flats during the night and occupy protected *R. mangle*-dominated embayments during the day (Lear *et al.* 2014; Huston *et al.* 2017), and specific use of *R. mangle* prop roots to avoid predators has been observed (Poulakis *et al.* 2011). Juvenile goliath grouper are found under overhanging mangroves and associated *R. mangle* prop roots in areas where currents create deep undercuts in creek banks (Koenig *et al.* 2017). Adult goliath grouper move long distances to aggregate in south Florida during the spawning season, which occurs when mangroves are most flooded and thus most available as habitat for juveniles (Koenig *et al.* 2017).

In the southeastern United States, mangrove expansion is just one of many dynamic aspects of global change that challenge our ability to predict the future of coastal food webs and fisheries. For example, accelerated sea-level rise is expected to ultimately lead to wetland fragmentation, loss, and the transformation of coastal landscapes (Kirwan & Megonigal 2013; Enwright *et al.* 2016; Couvillion *et al.* 2017; Borchert *et al.* 2018; Stagg *et al.* 2020). In the short term, wetland loss and fragmentation could lead to ephemeral increases in fisheries production due to increases in the amount of valuable edge habitat (Baker *et al.* 2020; Harris *et al.* 2020). However, in the long-term and under higher rates of sea-level rise, the landscape-scale loss of coastal wetlands (Saintilan *et al.* 2020; Törnqvist *et al.* 2020) and the associated edge habitat are expected to have a negative impact on fisheries.

The northward range expansion of tropical fish and invertebrate species is another aspect of climate change that will interact with mangrove expansion to affect coastal food webs and fisheries (Vergés *et al.* 2014; Osland *et al.* 2021). Warming winter water temperatures could transform fish and invertebrate assemblages across the southeastern United States. Thus, in addition to shifts in vegetation structure due to mangrove encroachment and sea-level rise, coastal food webs and fisheries in the region may also change as new tropical species migrate northward in response to warming winter water temperatures (Vergés *et al.* 2014). In North America, the northern distributions of tropical, cold-sensitive fish species are governed by the frequency and intensity of extreme cold water temperatures (Martin & McEachron 1996; Boucek & Rehage 2014; Stevens *et al.* 2016). For example, the northern limits of common snook (*C. undecimalis*) (Howells *et al.* 1990), gray snapper (*L. griseus*) (Hare *et al.* 2012), and tarpon (*Megalops atlanticus*) (Mace *et al.* 2017) are governed by winter cold temperature extremes, which can lead to mass mortality events (i.e., fish kills). Conversely, warming winters can lead to population growth and range expansion. For example, the common snook expanded northward along Florida's Gulf of Mexico coast (Anderson *et al.* 2020; Purtlebaugh *et al.* 2020) after a 2010 cold event that resulted in mass mortality and range contraction (Stevens *et al.* 2016).

In contrast to mangroves, which rely on passive water transport of propagules for dispersal and migration (Van der Stocken *et al.* 2019a; Van der Stocken *et al.* 2019b), fish are actively mobile organisms. The high adaptive capacity of many coastal fish and invertebrate species to warming winter waters indicates that some fish and invertebrate species will move northward at rates that exceed the northward migration of mangroves (Riley *et al.* 2014; Cannizzo *et al.* 2020; Stevens *et al.* 2021). Anthropogenic habitat and microclimatic refugia can aid in the expansion of mangrove fauna. For example, the occupation of warm-water springs allows common snook to overwinter in a climate where coastal water temperatures can still drop below their lethal limits (Stevens *et al.* 2018). Along the Atlantic coast of North America, the northern range limit of mangrove tree crabs (*Aratus pisonii*) is currently in Georgia (Riley *et al.* 2014), which is north of mangrove forests' current range limit in Florida. Interestingly, in the absence of mangrove trees, mangrove tree crabs can adapt to suboptimal novel marsh environments by: (1) shifting behavioral strategies from predator evasion (i.e., tree climbing) to autotomy (i.e., leg dropping) to optimize survival (Johnston & Smith 2018); or (2) using anthropogenic structures like docks (Cannizzo *et al.* 2018; Cannizzo & Griffen 2018; Cannizzo & Griffen 2019; Cannizzo *et al.* 2019; Cannizzo *et al.* 2020). Anthropogenic structures such as bridges and docks may also provide habitat for juvenile goliath grouper in the northern Gulf of Mexico where adult spawning sites have been recently documented (Malinowski *et al.* 2019).

In addition to affecting the distribution and abundance of fish, mangrove expansion can affect the recreational fishing experience in positive and negative ways. For example, along the central Texas coast near San Jose Island, many anglers fly fish for redfish (*Sciaenops ocellatus*)

(Figure 4), but mangrove expansion has reduced the amount of viable fishing area. While fly fishing is possible within the region's grass and succulent plant-dominated salt marshes, fly fishing is not possible within dense meter-tall mangrove stands [Chuck Naiser, oral communication, 26 May 2021]. However, on windy days, the same mangroves can have a positive effect on the fishing experience by buffering winds (Guo *et al.* 2017), which can improve boat control, water clarity, and the ability to sight cast [Billy Trimble, oral communication, 24 May 2021]. Interestingly, by hindering airboat access into wetlands, mangroves may also prevent damage to wetland vegetation and sediments, which has the potential to reduce rates of erosion, hydrologic change, and wetland loss in areas with high boat and angler concentrations [Billy Trimble and Chuck Naiser, oral communication, 24 and 26 May 2021].

Avian communities

What are the effects of mangrove expansion on avian communities? The response of migratory and resident avifauna is arguably one of the least understood consequences of mangrove expansion in the southeastern United States. However, these responses are likely to have a broad range of consequences for ecosystem functions and services. Birds serve as vectors of connectivity between coastal wetland habitats and adjacent systems, and are thus critical and wide-ranging influencers of ecosystem connectivity, including fluxes of energy and nutrients (Buelow & Sheaves 2015). Mangroves in the Gulf of Mexico are important habitat for foraging, resting, nesting, and migrant avifauna (Burger 2017). However, many of these birds do not use mangroves year-round, instead relying on a species-dependent matrix of mudflat, marsh, and other coastal habitats during migration, nesting, and wintering. Therefore, the consequences of mangrove expansion into salt marshes for birds could be substantial yet variable across taxa. The following sections address known and hypothesized consequences of mangrove expansion for four major taxonomic groups of coastal birds: wading birds, shorebirds, marsh birds, and passerines.

Wading birds (hereafter waders) are large and conspicuous wetland-dependent birds, including members from Families Ardeidae, Threskiornithidae, and Gruidae. Many waders nest in trees or shrubs, including mangrove stands found across much of the southeastern United States (Portnoy 1977; Davis *et al.* 2005; Burger 2017). Thus, mangrove expansion may improve nesting habitat for some wading birds. However, the effects of expanding mangroves on foraging behaviors may be less beneficial, as ibis and herons select areas with lower mangrove cover to feed (Guo *et al.* 2017; see also the whooping crane section below). Wader species richness tends to be lower at encroached sites on the Texas coast, though relative abundances for most common species are generally similar between marsh and mangrove sites (Armitage *et al.* 2021).

Shorebirds comprise four major families (Scolopacidae, Charadriidae, Recurvirostridae, Haematopodidae) and dozens of genera. These mostly small and often gregarious birds frequently forage and roost in large groups within coastal estuaries. The northern Gulf of Mexico is a particularly important part of the North American central migratory flyway (Withers 2002a). Although some shorebirds will use mangrove shrubs for nesting or roosting (Zwarts 1988), mangroves produce extensive aerial root complexes that may reduce the accessibility of tidal flats or ponds, which are the preferred foraging habitat for many shorebird species (Withers 2002a; Darnell & Smith 2004). In addition, dense mangrove stands may conceal and attract predators, and nonbreeding shorebirds will actively avoid areas with mangroves in favor of salt marshes, tidal flats, and other coastal habitats (Straw & Saintilan 2006; Kelleway *et al.* 2017;

Vitale *et al.* 2021). For example, optimal high-tide shorebird roosts were located far from woody vegetation at a critical wintering and migratory stopover site in Florida's Big Bend (Brush *et al.* 2017). There could be survival and energy costs associated with antipredator behaviors and increased predation if alternative roosting and foraging habitats are not available (e.g., due to human disturbance or habitat loss) (Rogers *et al.* 2006). Because shorebirds are relatively long-lived, their populations are greatly affected by small changes to demographic parameters, such as adult mortality (Colwell 2010). One study concluded that mangrove expansion into roosting habitats likely contributed to local declines of wintering shorebird populations and biodiversity (Woodley 2004). Accordingly, surveys of wintering shorebirds on the Texas coast revealed that diversity and relative abundance are higher in marshes than at sites with high mangrove cover (Whitt 2016; Armitage *et al.* 2021).

Marsh birds are coastal marsh-dependent and live at the ecotone between aquatic and terrestrial ecosystems. These often cryptic colored and elusive species include members from the families Rallidae, Troglodytidae, and Passerellidae. Marsh habitat loss, regardless of environmental or anthropogenic drivers, can have detrimental impacts on marsh-dependent species. For example, nearly 50% of marsh birds found in the northern Gulf of Mexico are of conservation concern primarily due to loss of wetland habitats (Woodrey *et al.* 2019). Species that are dependent on graminoid marsh plants [e.g., seaside sparrows (*Ammospiza maritima*) or marsh wrens (*Cistothorus palustris*)] are unlikely to utilize mangrove-encroached wetlands (Schwarzer *et al.* 2020). There could be direct or indirect impacts of mangrove expansion into high marsh habitats that are critical for the eastern black rail (*Laterallus jamaicensis*) (Watts 2016). More information is also needed to understand the migratory ecology of marsh bird species to determine potential impacts of mangrove expansion in different parts of their geographic range.

The passerine group (Order Passeriformes) includes many species of conspicuous migratory warblers that are highly sought-after targets by birdwatchers and photographers. During migration stopovers, these birds roost on coastal shrubs and trees, sometimes including mangroves. Thus, it is often hypothesized that mangroves could increase migratory stopover habitat quality for warblers and other passerines (Kelleway *et al.* 2017). However, there is little data from the Gulf of Mexico or Atlantic coast to quantitatively support this hypothesis, largely due to the challenges of accurately censusing populations of these small, active birds. Analyses derived from eBird, a community science data repository, suggest that birdwatchers detect substantially more warblers in marsh than in mangrove habitats on the Texas coast (Whitt 2016). Additional species- and community-level analyses are needed for a more robust evaluation of whether mangrove expansion will be a net benefit or detriment to passerine assemblages.

Birds are visible and iconic users of coastal wetlands, attracting recreational users and supporting a vigorous ecotourism industry (Barbier *et al.* 2011) that brings millions of dollars to the southeastern United States' coastal economies each year (Kildow *et al.* 2008; NRDC 2010). Mangrove expansion into salt marshes in the southeastern United States may alter the perceived and actual value of coastal wetland habitats for avifauna. For example, in coastal Texas, birdwatchers spend more time visiting coastal marsh sites with fewer mangroves (Whitt 2016; Armitage *et al.* 2021). The reasons for that preference may be linked to accessibility or visibility. Identifying the drivers behind this pattern is an area ripe for future study that reaches across social and life science disciplines.

For some coastal parks and refuges, coastal resource managers' perspectives on mangrove expansion can be heavily influenced by their organizations' local avian priorities. For example, on Louisiana's barrier islands, dense mangrove stands provide valuable habitat for nesting colonies of brown pelicans (*Pelecanus occidentalis*) (Figure 4), which is the state bird and a species with a rich conservation history. Following pesticide-driven extirpation and reintroduction in the 1960s, brown pelican populations have rebounded to the extent that they were removed from the endangered species list in 2009 (Walter *et al.* 2013a). Nevertheless, barrier island habitat losses in Louisiana have been very high (Martinez *et al.* 2009), and pelican nesting habitat has become a priority during recent coastal restoration efforts. Pelicans prefer to place nests on mangroves, which provide strong nesting platforms that are tall enough to avoid flooding during island overwash events (Hintgen *et al.* 1985; Visser *et al.* 2005; Walter *et al.* 2013b). Thus, mangrove expansion and growth on these islands is viewed as a positive change for pelicans, and therefore is sometimes facilitated during barrier island restoration efforts [Darin Lee (Louisiana Coastal Protection and Restoration Authority), oral communication, 21 May 2021].

Another example of how avian habitat priorities can affect local perspectives on mangrove expansion can be found in the Aransas National Wildlife Refuge (ANWR) in Texas. The salt marshes within and near ANWR provide critical wintering grounds for the rare and endangered whooping crane (*Grus americana*). Whooping crane populations declined during the late 19th and early 20th century to the point where less than 25 individuals remained in the early 1940s (Allen 1952; Erickson & Derrickson 1981). The only self-sustaining population that persists today is the Aransas-Wood Buffalo population, which winters in coastal Texas in the ANWR region and migrates to its summer breeding grounds in north-central Canada. During the winter of 2019-2020, the Aransas-Wood Buffalo population was estimated to contain just 506 individuals (Butler *et al.* 2020). While wintering in coastal Texas, whooping cranes forage within salt marshes for blue crabs (*C. sapidus*), wolfberries (*Lycium carolinianum*), and other foods (Hunt & Slack 1989; Chavez-Ramirez 1996). However, the ANWR region is a recent hotspot for mangrove expansion in Texas (Armitage *et al.* 2015; Brown *et al.* 2016), and there is concern that mangrove replacement of salt marsh will negatively affect the habitat and food resources available for whooping cranes (Stehn & Prieto 2010; Chavez-Ramirez & Wehtje 2012). Blue crabs are an especially important food source for cranes (Hunt & Slack 1989; Chavez-Ramirez 1996), and blue crab abundance may have a positive relationship with whooping crane survival (Pugesek *et al.* 2013). Thus, mangrove expansion effects on blue crabs and other foods are a concern. Several studies have indicated that blue crab abundance may decrease with mangrove expansion (Johnston & Caretti 2017; Smee *et al.* 2017; Glazner *et al.* 2020). For coastal managers at ANWR, maintaining the region's whooping crane population is a high priority; however, managers are concerned that continued mangrove expansion could have a negative impact [Andrew Stetter and Colt Sanspree (U.S. Fish and Wildlife Service), oral communication, 30 March 2021]. In addition to research that better characterizes the effects of mangrove expansion on whooping cranes, coastal managers in the ANWR region are interested in management strategies for limiting mangrove expansion into the salt marshes used by whooping cranes [Andrew Stetter and Colt Sanspree (U.S. Fish and Wildlife Service), oral communication, 30 March 2021].

Insects, other terrestrial arthropods, and mangrove honey

How might mangrove expansion affect insects, other terrestrial arthropods, and associated ecosystem services (e.g., mangrove honey)? In a study conducted in Texas, insect abundance and biomass decreased with mangrove abundance, whereas insect richness and diversity increased (Loveless & Smee 2019). An arthropod-focused study conducted along Florida's Atlantic coasts also found distinct arthropod communities in mangrove- and marsh-dominated wetlands, indicating that mangrove expansion may increase habitat heterogeneity, which would produce an increase in arthropod diversity (Nathan 2020). In marsh-dominated coastal wetlands, mangroves may provide a novel resource for insects and other terrestrial arthropods in the form of nectar-producing flowers (Nathan 2020). However, more data are needed to evaluate the effects of mangrove encroachment on terrestrial arthropods and the terrestrial portion of coastal wetland food webs (see Loveless & Smee 2019; Nathan 2020).

In areas that historically supported salt marsh plants, expansion of mangroves with nectar-producing flowers (Nathan 2020) may provide a novel kind of honey for beekeepers along northern coastal reaches of the southeastern United States. Along many tropical coastlines, mangrove forests are popular honey-producing areas for beekeepers and honey hunters (Bradbear 2009). In Florida, which is one of the largest honey-producing states of the United States, mangroves are a popular source of nectar for honey production (Sanford 2016). For this reason, in some of the scientific literature and in herbarium records from the early 20th century, the common name used for *A. germinans* was honey mangrove (e.g., Penfound & Hathaway 1938). Currently, mangrove honey in the southeastern United States is produced primarily along the southern and central coasts of Florida. However, as mangroves expand further north, there is the potential for mangrove honey to be produced along Florida's northern coasts. For example, in the Apalachicola region, where freshwater swamps have historically supported tupelo honeyspecialty businesses for more than a century, the recent expansion of mangroves on the region's barrier island ecosystems (Snyder *et al.* 2021) has prompted interest to also produce mangrove honey [George Watkins, oral communication, 24 May 2021].

Recreation

Generally, the recreational value of coastal wetlands is underpinned by many of the ecological and biophysical characteristics described in previous sections of this review, as well as numerous other socioeconomic factors (Hamilton & Snedaker 1984). For instance, many of the recreational benefits of mangroves and marshes result from the diverse species and food webs that depend on these habitats (Kelleway *et al.* 2017). Rare, exotic, and charismatic species can be particularly important for recreation in wetlands. More broadly, the aesthetic appeal or scenery, uniqueness of a site, proximity to population centers, trail and water access, and overall public awareness are key landscape and socioeconomic factors that influence recreational use, value, and satisfaction of people (Hamilton & Snedaker 1984; Harty 2009; Lundquist *et al.* 2014).

Similar to other ecosystem services, mangrove expansion in marshes is likely to have variable and site-specific impacts on recreation (Kelleway *et al.* 2017). For instance, shifting assemblages of birds may impact, either positively or negatively, recreational demand and satisfaction among bird watchers (Whitt 2016; Armitage *et al.* 2021). Likewise, the same scenario would apply to recreational fishers if expanding mangroves support different fish communities than salt marshes. These impacts are likely to be greatest among highly specialized recreational users who may value specific places or prioritize certain species that could become more or less common (Oh *et al.* 2013). The expansion of mangroves may also impact water access and viewscapes (Kaplowitz 2001; Harty 2009; Lundquist *et al.* 2014; see also coastal food webs and fisheries section). Both mangroves and marshes are appreciated for their aesthetic or scenic values (Kaplowitz 2001), but water access is more difficult in areas with dense mangrove forests. In general, recreation-related attitudes, preferences, and behaviors represent a major gap in the literature on mangrove ecosystem services, especially within the context of mangrove expansion into salt marshes.

Where mangrove expansion is occurring near residential areas, mangrove replacement of marsh can have striking effects on residents' recreational activities and interactions with their coast (Harty 2009; Lundquist *et al.* 2014). For example, in the 1990s following the major freeze events of the 1980s, many residents in the Cedar Key area of Florida had wide-ranging views that looked out onto expansive wetlands dominated by salt marsh grasses (Stevens *et al.* 2006). However, expanding mangrove forests have grown to heights greater than 6 m (Yando *et al.* 2016), which is tall enough to block valuable landscape views of the salt marsh [Andrew Gude (U.S. Fish and Wildlife Service) and Michael Allen (University of Florida), oral communication, 19 May 2021]. Despite the potential gains in coastal protection supported by mangrove expansion (Doughty *et al.* 2017; Hijuelos *et al.* 2019; Pennings *et al.* 2021), the loss of salt marsh views is a critical and overarching concern for many Cedar Key residents, and some residents are interested in managing the expanding mangrove forests to regain the previously present salt marsh views and access [Andrew Gude (U.S. Fish and Wildlife Service) and Michael Allen (University of Florida), oral communication, 19 May 2021]. Further, by reducing wind speeds, the mangrove forests can also increase the number of mosquitos and other nuisance insects near residences [Andrew Gude (U.S. Fish and Wildlife Service) and Michael Allen (University of Florida), oral communication, 19 May 2021]. Conversely, from a practical landscape maintenance perspective, mangrove expansion can reduce the amount of storm debris (i.e., marsh wrack) that is transported and deposited onto waterfront properties [Andrew Gude (U.S. Fish and Wildlife Service) and Michael Allen (University of Florida), oral communication, 19 May 2021]. Thus, mangroves can be viewed as both positive (shoreline protection) and negative (impeding ocean views and thus reducing property values) by local homeowners.

Knowledge gaps and emerging research needs

The majority of research regarding the effects of mangrove expansion has focused on coastal food webs and fisheries (Figure 1e; 31 papers) and carbon sequestration (Figure 1a; 27 papers). Less research has examined soil elevation dynamics (Figure 1b; 12 papers), water quality and nutrient cycling (Figure 1c; 12 papers), avian communities (Figure 1f; 9 papers), coastal protection (Figure 1d; 4 papers), recreation (Figure 1h; 4 papers), and insects and terrestrial arthropods (Figure 1g; 2 papers). Within the ecosystem service-focused sections of this review, we have identified knowledge gaps and emerging research needs. In Table 3, we summarize this information to: (1) rank the level of knowledge regarding the impacts of mangrove expansion on specific ecosystem services; and (2) summarize the most critical knowledge gaps and research needs.

Interactions with other aspects of climate change

Our review focuses primarily on the ecological effects of mangrove expansion driven by warming winter temperatures. However, coastal wetlands in the southeastern United States are vulnerable to many different aspects of climate change. In addition to warming winter temperatures, coastal wetlands in the region are affected by rising sea levels, saltwater intrusion, and changes in the frequency and intensity of extreme climatic events (USGCRP 2017; Carter *et al.* 2018; Osland *et al.* 2018a). For example, rising sea levels can lead to wetland conversion to open water (Couvillion *et al.* 2017; Saintilan *et al.* 2020; Törnqvist *et al.* 2020) and the landward migration of wetlands (Enwright *et al.* 2016; Borchert *et al.* 2018). Landscape-scale wetland vegetation dieback events can be triggered by extreme drought (Alber *et al.* 2008; Duke *et al.* 2017; Lovelock *et al.* 2017; Sippo *et al.* 2018), flooding (Stagg *et al.* 2021), hurricanes (Cahoon *et al.* 2003; Osland *et al.* 2020c), and freeze events (Osland *et al.* 2017a). Macroclimate drivers govern foundation plant communities in the region, which means that changing temperature and precipitation regimes can transform coastal wetlands (Gabler *et al.* 2017; Osland *et al.* 2019a). Climate change is also expected to increase the frequency of major hurricanes, which affects the dispersal of mangroves beyond current range limits (Kennedy *et al.* 2016; Van der Stocken *et al.* 2019a; Kennedy *et al.* 2020). Collectively, these examples show that mangrove range expansion dynamics will be greatly influenced by interactions between many different aspects of climate change.

Conclusions

In the past two decades, there has been a rapid increase in the number of studies investigating the effects of mangrove range expansion on ecosystem goods and services in the southeastern United States (Figure 1). This review synthesizes that information with an emphasis on impacts to carbon sequestration, wildlife habitat, storm protection, erosion prevention, water purification, fisheries support, and recreation. One consistent theme throughout this work is that there are ecological trade-offs for consideration by coastal scientists and natural resource managers (e.g., Table 2; Figure 4). Mangrove expansion and marsh displacement can produce beneficial changes in some ecosystem services, while producing detrimental changes in other services. As a result, there is much variation in opinions regarding the social and ecological consequences of mangrove expansion. For the same location, some individuals may view mangrove expansion as a positive transformation to be fostered, while others may see it as a negative change to be prevented. Such variation in perceptions and impacts highlights the benefits of careful planning and discussion preceding management efforts (e.g., mangrove planting or removal) that could accelerate or constrain the pace of mangrove expansion.

Acknowledgements

We thank the many individuals across the region that took the time to speak to us about this topic and provide local examples from their coasts. This research was funded by the U.S. Geological Survey Southeast Climate Adaptation Science Center, which is managed by the USGS National Climate Adaptation Science Center. This work was also supported by a National Academy of Sciences, Engineering, and Medicine Gulf Research Program Healthy Ecosystems Grant to ARH, SBS, CCS, and MJO. Additional support for MJO was provided by the USGS Climate R&D Program and the USGS Greater Everglades Priority Ecosystem Science Program. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Ahrens RN, Walters CJ, Christensen V (2012) Foraging arena theory. *Fish and fisheries,* **13**, 41-59.

Alber M, Swenson EM, Adamowicz SC, Mendelssohn IA (2008) Salt marsh dieback: an overview of recent events in the US. *Estuarine, Coastal and Shelf Science,* **80**, 1-11.

Allen RP (1952) *The whooping crane. National Audobon Society Research Report 3,* National Audobon Society, New York, New York, USA.

Anderson J, Williford D, González-Barnes A, Chapa C, Martinez-Andrade F, Overath RD (2020) Demographic, taxonomic and genetic characterization of the snook species complex (*Centropomus* sp.) along the leading edge of its range in the northwestern Gulf of Mexico. *North American Journal of Fisheries Management*, **40**, 190-208.

Archer SR, Andersen EM, Predick KI, Schwinning S, Steidl RJ, Woods SR (2017) Woody plant encroachment: causes and consequences. In: *Rangeland systems: processes, management, and challenges* (ed Briske DD), pp 25-84. Springer, Cham, Switzerland.

Arkema KK, Guannel G, Verutes G *et al.* (2013) Coastal habitats shield people and property from sea-level rise and storms. *Nature Climate Change,* **3**, 913-918.

Armitage AR, Highfield WE, Brody SD, Louchouarn P (2015) The contribution of mangrove expansion to salt marsh loss on the Texas Gulf Coast. *PLOS ONE,* **10**, e0125404.

Armitage AR, Weaver CA, Whitt AA, Pennings SC (2021) Effects of mangrove encroachment on tidal wetland plant, nekton, and bird communities in the Western Gulf of Mexico. *Estuarine, Coastal & Shelf Science,* **248**, 106767.

Asbridge E, Lucas R, Accad A, Dowling R (2015) Mangrove response to environmental changes predicted under varying climates: case studies from Australia. *Current Forestry Reports,* **1**, 178-194.

Baker R, Abrantes K, Feller IC (2021) Stable isotopes suggest limited role of wetland macrophyte production supporting aquatic food webs across a mangrove-salt marsh ecotone. *Estuaries and Coasts,* **44**, 1619-1627.

Baker R, Taylor MD, Able KW *et al.* (2020) Fisheries rely on threatened salt marshes. *Science,* **370**, 670-671.

Barbier EB (2016) The protective service of mangrove ecosystems: A review of valuation methods. *Marine Pollution Bulletin,* **109**, 676-681.

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR (2011) The value of estuarine and coastal ecosystem services. *Ecological Monographs,* **81**, 169-193.

Bardou R, Parker JD, Feller IC, Cavanaugh KC (2021) Variability in the fundamental versus realized niches of North American mangroves. *Journal of Biogeography,* **48**, 160-175.

Barger NN, Archer SR, Campbell JL, Huang Cy, Morton JA, Knapp AK (2011) Woody plant proliferation in North American drylands: a synthesis of impacts on ecosystem carbon balance. *Journal of Geophysical Research: Biogeosciences,* **116**, G00K07.

Barimo JF, Serafy JE (2003) Fishes of a restored mangrove habitat on Key Biscayne, Florida. *Florida Scientist,* **66**, 12-22.

Barreto CR, Morrissey EM, Wykoff DD, Chapman SK (2018) Co-occurring mangroves and salt marshes differ in microbial community composition. *Wetlands,* **38**, 497-508.

Beck MW, Heck Jr KL, Able KW *et al.* (2001) The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates: A better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. *BioScience,* **51**, 633-641.

Bianchi TS, Allison MA, Zhao J, Li X, Comeaux RS, Feagin RA, Kulawardhana RW (2013) Historical reconstruction of mangrove expansion in the Gulf of Mexico: linking climate change with carbon sequestration in coastal wetlands. *Estuarine, Coastal and Shelf Science,* **119**, 7-16.

Borchert SM, Osland MJ, Enwright NM, Griffith KT (2018) Coastal wetland adaptation to sea-level rise: quantifying the potential for landward migration and coastal squeeze in northern Gulf of Mexico estuaries. *Journal of Applied Ecology*, **55**, 2876-2887.

Boucek RE, Gaiser EE, Liu H, Rehage JS (2016) A review of subtropical community resistance and resilience to extreme cold spells. *Ecosphere*, **7**, Article e01455.

Boucek RE, Rehage JS (2014) Climate extremes drive changes in functional community structure. *Global Change Biology*, **20**, 1821-1831.

Bradbear N (2009) *Bees and their role in forest livelihoods: a guide to the services provided by bees and the sustainable harvesting, processing and marketing of their products,* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Breithaupt JL, Smoak JM, Bianchi TS *et al.* (2020) Increasing rates of carbon burial in southwest Florida coastal wetlands. *Journal of Geophysical Research: Biogeosciences,* **125**, e2019JG005349.

Breithaupt JL, Smoak JM, Smith TJ, Sanders CJ, Hoare A (2012) Organic carbon burial rates in mangrove sediments: Strengthening the global budget. *Global Biogeochemical Cycles,* **26**, GB3011.

Brockmeyer R, Beal J, Sharpe B *et al.* (2017) Indian River Lagoon. In: *Coastal Habitat Integrated Mapping and Monitoring Program Report for the State of Florida. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute Technical Report No. 21.* (eds Radabaugh KR, Powell CE, Moyer RP), pp 134-143. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, St. Petersburg, Florida, USA.

Brown R, Paine JG, Saylam K, Tremblay TA, Andrews JR, Averett A (2016) *Mangrove monitoring using airborne VNIR in the Espiritu Santo Bay area, central Texas coast,* Bureau of Economic Geology, University of Texas at Austin, Austin, Texas, USA.

Brush JM, Schwarzer AC, Frederick PC (2017) Importance and function of foraging and roost habitat for wintering American Oystercatchers. *Estuaries and Coasts,* **40**, 286-295.

Buelow C, Sheaves M (2015) A birds-eye view of biological connectivity in mangrove systems. *Estuarine, Coastal and Shelf Science*, **152**, 33-43.

Burger J (2017) Avian resources of the northern Gulf of Mexico. In: *Habitats and biota of the Gulf of Mexico: before the Deepwater Horizon oil spill. Volume 2: fish resources, fisheries, sea turtles, avian resources, marine mammals, diseases and mortalities* (ed Ward CH), pp 1353-1488. Springer, New York, New York, USA.

Butler MJ, Sanspree CR, Harrell W (2020) *Whooping crane survey results: Winter 2019–2020,* U.S. Fish and Wildlife Service, Austwell, Texas, USA.

Cahoon DR, Hensel P, Rybczyk J, McKee KL, Proffitt CE, Perez BC (2003) Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology,* **91**, 1093-1105.

Cahoon DR, Lynch JC, Perez BC *et al.* (2002) High-precision measurements of wetland sediment elevation: II. The rod surface elevation table. *Journal of Sedimentary Research,* **72**, 734-739.

Cahoon DR, McKee KL, Morris JT (2020) How plants influence resilience of salt marsh and mangrove wetlands to sea-level rise. *Estuaries and Coasts,* **44**, 883-898.

Cannizzo ZJ, Dixon SR, Griffen BD (2018) An anthropogenic habitat within a suboptimal colonized ecosystem provides improved conditions for a range-shifting species. *Ecology and Evolution,* **8**, 1521-1533.

Cannizzo ZJ, Griffen BD (2018) Habitat-specific impacts of Hurricane Matthew on a range-expanding species. *Hydrobiologia*, **809**, 79-89.

Cannizzo ZJ, Griffen BD (2019) An artificial habitat facilitates a climate-mediated range expansion into a suboptimal novel ecosystem. *PLOS ONE*, **14**, e0211638.

Cannizzo ZJ, Lang SQ, Benitez-Nelson B, Griffen BD (2020) An artificial habitat increases the reproductive fitness of a range-shifting species within a newly colonized ecosystem. *Scientific Reports*, **10**, 554.

Cannizzo ZJ, Nix SK, Whaling IC, Griffen BD (2019) Individual morphology and habitat structure alter social interactions in a range-shifting species. *Diversity*, **11**, 6.

Carter L, Terando A, Dow K *et al.* (2018) Southeast. In: *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* (eds Reidmiller DR, Avery CW, Easterling DR, Kunkel KE, Lewis KLM, Maycock TK, Stewart BC), pp 743-808. U.S. Global Change Research Program, Washington, DC, USA.

Caudill MC (2005) Nekton utilization of black mangrove (*Avicennia germinans*) and smooth cordgrass (*Spartina alterniflora*) sites in southwestern Caminada Bay, Louisiana. MS Thesis. Louisiana State University, Baton Rouge, Louisiana, USA.

Cavanaugh KC, Dangremond EM, Doughty CL *et al.* (2019) Climate-driven regime shifts in a mangrove–salt marsh ecotone over the past 250 years. *Proceedings of the National Academy of Sciences*, **116**, 21602-21608.

Cavanaugh KC, Kellner JR, Forde AJ, Gruner DS, Parker JD, Rodriguez W, Feller IC (2014) Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences*, **111**, 723-727.

Cavanaugh KC, Osland MJ, Bardou R, Hinijosa-Arango G, López-Vivas JM, Parker JD, Rovai AS (2018) Sensitivity of mangrove range limits to climate variability. *Global Ecology and Biogeography*, **27**, 925-935.

Cavanaugh KC, Parker JD, Cook-Patton SC, Feller IC, Williams AP, Kellner JR (2015) Integrating physiological threshold experiments with climate modeling to project mangrove species' range expansion. *Global Change Biology*, **21**, 1928-1938.

Chapman SK, Feller IC, Canas G *et al.* (2021) Mangrove growth response to experimental warming is greatest near the range limit in northeast Florida. *Ecology*, **102**, e03320.

Charles SP, Kominoski JS, Armitage AR, Guo H, Weaver CA, Pennings SC (2020) Quantifying how changing mangrove cover affects ecosystem carbon storage in coastal wetlands. *Ecology*, **101**, e02916.

Chavez-Ramirez F (1996) Food availability, foraging ecology, and energetics of whooping cranes wintering in Texas. PhD Thesis. Texas A&M University, College Station, Texas, USA.

Chavez-Ramirez F, Wehtje W (2012) Potential impact of climate change scenarios on whooping crane life history. *Wetlands*, **32**, 11-20.

Chen E, Blaze JA, Smith RS, Peng S, Byers JE (2020) Freeze tolerance of poleward-spreading mangrove species weakened by soil properties of resident salt marsh competitor. *Journal of Ecology*, **108**, 1725-1737.

Chen L, Wang W, Li QQ *et al.* (2017) Mangrove species' responses to winter air temperature extremes in China. *Ecosphere*, **8**, e01865.

Chen Y, Li Y, Thompson C, Wang X, Cai T, Chang Y (2018) Differential sediment trapping abilities of mangrove and saltmarsh vegetation in a subtropical estuary. *Geomorphology*, **318**, 270-282.

Chmura GL, Anisfeld SC, Cahoon DR, Lynch JC (2003) Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, **17**, 1111.

Coldren GA, Langley JA, Feller IC, Chapman SK (2019) Warming accelerates mangrove expansion and surface elevation gain in a subtropical wetland. *Journal of Ecology*, **107**, 79-90.

Colwell MA (2010) *Shorebird ecology, conservation, and management*, University of California Press, Berkeley, California, USA.

Comeaux RS, Allison MA, Bianchi TS (2012) Mangrove expansion in the Gulf of Mexico with climate change: Implications for wetland health and resistance to rising sea levels. *Estuarine, Coastal and Shelf Science*, **96**, 81-95.

CONABIO (2016) *Distribución de los Manglares en México en 2015*, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Sistema de Monitoreo de los Manglares de México (SMMM), Ciudad de México, México.

Costanza R, de Groot R, Sutton P *et al.* (2014) Changes in the global value of ecosystem services. *Global Environmental Change*, **26**, 152-158.

Couvillion BR, Beck H, Schoolmaster D, Fischer M (2017) *Land area change in coastal Louisiana 1932 to 2016: U.S. Geological Survey Scientific Investigations Map 3381, 16 p. pamphlet*, https://doi.org/10.3133/sim3381.

Craig H, Antwis RE, Cordero I *et al.* (2021) Nitrogen addition alters composition, diversity, and functioning of microbial communities in mangrove soils: an incubation experiment. *Soil Biology and Biochemistry*, **153**, 108076.

Dangremond EM, Simpson LT, Osborne TZ, Feller IC (2020) Nitrogen enrichment accelerates mangrove range expansion in the temperate–tropical ecotone. *Ecosystems*, **23**, 703-714.

Darnell TM, Smith EH (2004) Avian use of natural and created salt marsh in Texas, USA. *Waterbirds*, **27**, 355-361.

Davis SM, Childers DL, Lorenz JJ, Wanless HR, Hopkins TE (2005) A conceptual model of ecological interactions in the mangrove estuaries of the Florida Everglades. *Wetlands*, **25**, 832-842.

Day RH, Michot TC, Twilley RM, From AS (2020) *Geographic distribution of black mangrove (Avicennia germinans) in coastal Louisiana in 2009: U.S. Geoglogical Survey data release*, https://doi.org/10.5066/P9RC8EIE.

Deegan LA, Day Jr. JW, Gosselink JG, Yañez-Arancibia A, Soberón-Chàvez G, Sànchez-Gil P (1986) Relationships among physical characteristics, vegetation distribution and fisheries yield in Gulf of Mexico estuaries. In: *Estuarine Variability* (ed Wolfe DA), pp 83-100. Academic Press, Orlando, Florida USA.

Diop ES, Soumare A, Diallo N, Guisse A (1997) Recent changes of the mangroves of the Saloum River Estuary, Senegal. *Mangroves and Salt Marshes*, **1**, 163-172.

Diskin MS, Smee DL (2017) Effects of black mangrove *Avicennia germinans* expansion on salt marsh nekton assemblages before and after a flood. *Hydrobiologia*, **803**, 283-294.

Donato DC, Kauffman JB, Murdiyarso D, Kurnianto S, Stidham M, Kanninen M (2011) Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience*, **4**, 293-297.

Doughty CL, Cavanaugh KC, Hall CR, Feller IC, Chapman SK (2017) Impacts of mangrove encroachment and mosquito impoundment management on coastal protection services. *Hydrobiologia*, **803**, 105-120.

Doughty CL, Langley JA, Walker WS, Feller IC, Schaub R, Chapman SK (2016) Mangrove range expansion rapidly increases coastal wetland carbon storage. *Estuaries and Coasts,* **39**, 385-396.

Duke NC, Ball MC, Ellison JC (1998) Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecology and Biogeography Letters,* **7**, 27-47.

Duke NC, Kovacs JM, Griffiths AD *et al.* (2017) Large-scale dieback of mangroves in Australia's Gulf of Carpentaria: a severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research,* **68**, 1816-1829.

Eldridge DJ, Bowker MA, Maestre FT, Roger E, Reynolds JF, Whitford WG (2011) Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecology Letters,* **14**, 709-722.

Ellis WL, Bell SS (2004) Conditional use of mangrove habitats by fishes: depth as a cue to avoid predators. *Estuaries,* **27**, 966-976.

Engle VD (2011) Estimating the provision of ecosystem services by Gulf of Mexico coastal wetlands. *Wetlands,* **31**, 179-193.

Enwright NM, Griffith KT, Osland MJ (2016) Barriers to and opportunities for landward migration of coastal wetlands with sea-level rise. *Frontiers in Ecology and the Environment,* **14**, 307-316.

Erickson RC, Derrickson SR (1981) The whooping crane. In: *Crane research around the world: Proceedings of the International Crane Symposium at Sapporo, Japan* (eds Lewis JC, Masatomi H), pp 104-118. International Crane Foundation, Baraboo, Wisconsin, USA.

Eslami-Andargoli L, Dale P, Sipe N, Chaseling J (2009) Mangrove expansion and rainfall patterns in Moreton Bay, southeast Queensland, Australia. *Estuarine, Coastal and Shelf Science,* **85**, 292-298.

Ewel KC, Twilley RR, Ong JE (1998) Different kinds of mangrove forests provide different goods and services. *Global Ecology and Biogeography Letters,* **7**, 83-94.

FDEP (2016) *Statewide Land Use Land Cover,* Florida Department of Environmental Protection, Tallahassee, Florida USA.

Feagin RA, Figlus J, Zinnert JC *et al.* (2015) Going with the flow or against the grain? The promise of vegetation for protecting beaches, dunes, and barrier islands from erosion. *Frontiers in Ecology and the Environment,* **13**, 203-210.

Feagin RA, Martinez ML, Mendoza-Gonzalez G, Costanza R (2010) Salt marsh zonal migration and ecosystem service change in response to global sea level rise: a case study from an urban region. *Ecology and Society,* **15**, article 14.

Feher LC, Osland MJ, Griffith KT *et al.* (2017) Linear and nonlinear effects of temperature and precipitation on ecosystem properties in tidal saline wetlands. *Ecosphere,* **8**, e01956.

Gabler CA, Osland MJ, Grace JB *et al.* (2017) Macroclimatic change expected to transform coastal wetland ecosystems this century. *Nature Climate Change,* **7**, 142-147.

Gedan KB, Kirwan ML, Wolanski E, Barbier EB, Silliman BR (2011) The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. *Climatic Change,* **106**, 7-29.

Geoghegan EK, Langley JA, Chapman SK (2020) A comparison of mangrove and marsh influences on soil respiration rates: A mesocosm study. *Estuarine, Coastal & Shelf Science,* **248**, 106877.

Giri C, Ochieng E, Tieszen LL *et al.* (2011) Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography,* **20**, 154-159.

Glazner R, Ballard J, Armitage AR (2021) Predation refuge values of marsh and mangrove vegetation for the marsh periwinkle *Littoraria irrorata*. *Marine Ecology Progress Series,* **672**, 153-162.

Glazner R, Blennau J, Armitage AR (2020) Mangroves alter predator-prey interactions by enhancing prey refuge value in a mangrove-marsh ecotone. *Journal of Experimental Marine Biology and Ecology,* **526**, 151336.

Goeke JA, Armitage AR (2021) Coastal woody encroachment reduces food quality for basal consumers. *Ecosphere,* **12**, e03511.

Gotceitas V, Colgan P (1989) Predator foraging success and habitat complexity: quantitative test of the threshold hypothesis. *Oecologia,* **80**, 158-166.

Greenwood MF, Idelberger CF, Stevens PW (2007) Habitat associations of large-bodied mangrove-shoreline fishes in a southwest Florida estuary and the effects of hurricane damage. *Bulletin of Marine Science,* **80**, 805-821.

Guo H, Weaver C, Charles SP *et al.* (2017) Coastal regime shifts: Rapid responses of coastal wetlands to changes in mangrove cover. *Ecology,* **98**, 762-772.

Hamilton LS, Snedaker SC (1984) *Handbook for mangrove area management,* United Nations Environment Program and East-West Environment and Policy Institute, Honolulu, Hawaii, USA.

Hare JA, Wuenschel MJ, Kimball ME (2012) Projecting range limits with coupled thermal tolerance-climate change models: an example based on gray snapper (*Lutjanus griseus*) along the US east coast. *PLOS ONE,* **7**, e52294.

Harris JM, James WR, Lesser JS, Doerr JC, Nelson JA (2020) Foundation species shift alters the energetic landscape of marsh nekton. *Estuaries and Coasts,* **44**, 1671-1680.

Harty C (2009) Mangrove planning and management in New Zealand and South East Australia–A reflection on approaches. *Ocean & Coastal Management,* **52**, 278-286.

Henry KM (2012) Linking nitrogen biogeochemistry to different stages of wetland soil development in the Mississippi River Delta, Louisiana. PhD Thesis, Louisiana State University, Baton Rouge, Louisiana, USA.

Henry KM, Twilley RR (2013) Soil development in a coastal Louisiana wetland during a climate-induced vegetation shift from salt marsh to mangrove. *Journal of Coastal Research,* **29**, 1273-1283.

Hijuelos AC, Dijkstra JT, Carruthers TJB, Heynert K, Reed DJ, van Wesenbeeck BK (2019) Linking management planning for coastal wetlands to potential future wave attenuation under a range of relative sea-level rise scenarios. *PLOS ONE,* **14**, e0216695.

Hintgen TM, Mulholland R, Zale AV (1985) Habitat suitability index models: eastern brown pelican. U.S. Fish and Wildlife Service Biological Report 82(10.90). U.S. Fish and Wildlife Service, Slidell, Louisiana, USA.

Holmquist JR, Windham-Myers L, Bliss N *et al.* (2018) Accuracy and precision of tidal wetland soil carbon mapping in the conterminous United States. *Scientific Reports,* **8**, 9478.

Howard RJ, From AS, Krauss KW, Andres KD, Cormier N, Allain L, Savarese M (2020) Soil surface elevation dynamics in a mangrove-to-marsh ecotone characterized by vegetation shifts. *Hydrobiologia,* **847**, 1087-1106.

Howells RG, Sonski AJ, Shafland PL, Hilton BD (1990) Lower temperature tolerance of snook, *Centropomus undecimalis*. *Northeast Gulf Science*, **11**, 155-158.

Hughes A, Cebrian J, Heck K, Goff J, Hanley T, Scheffel W, Zerebecki R (2018) Effects of oil exposure, plant species composition, and plant genotypic diversity on salt marsh and mangrove assemblages. *Ecosphere,* **9**, e02207.

Hunt HE, Slack RD (1989) Winter diets of whooping and sandhill cranes in south Texas. *The Journal of Wildlife Management*, **53**, 1150-1154.

Hunter A, Cebrian J, Stutes JP, Patterson D, Christiaen B, Lafabrie C, Goff J (2015) Magnitude and trophic fate of black needlerush (*Juncus roemerianus*) productivity: does nutrient addition matter? *Wetlands*, **35**, 401-417.

Huston CA, Stevens PW, Blaxton RM, Tolley SG, Scharer RM, Tornwall BM, Poulakis GR (2017) Diel movements of juvenile smalltooth sawfish: implications for defining the size of a nursery hotspot. *Endangered Species Research*, **34**, 311-322.

Hutchison LM (2016) *Operationalizing coastal wetland ecosystem services for enhanced decision making and resilience. PhD Dissertation.* , Texas A&M University Corpus Christi, Corpus Christi, Texas, USA.

Hutchison LM, Pollack JB, Swanson K, Yoskowitz D (2018) Operationalizing Blue Carbon in the Mission-Aransas National Estuarine Research Reserve, Texas. *Coastal Management*, **46**, 278-296.

Ibáñez C, Morris JT, Mendelssohn IA, Day JW (2012) Coastal marshes. In: *Estuarine Ecology, Second Edition* (eds Day JW, Crump BC, Kemp MW, Yáñez-Arancibia A), pp 129-163. John Wiley and Sons, Hoboken, New Jersey, USA.

Jafari NH, Harris BD, Cadigan JA *et al.* (2019) Wetland shear strength with emphasis on the impact of nutrients, sediments, and sea level rise. *Estuarine, Coastal and Shelf Science*, **229**, 106394.

James WR, Santos RO, Rehage JS, Doerr JC, Nelson JA (2020) E-scape: consumer specific landscapes of energetic resources derived from stable isotope analysis and remote sensing. *bioRxiv*, **https://doi.org/10.1101/2020.08.03.234781**.

Johnston CA, Caretti ON (2017) Mangrove expansion into temperate marshes alters habitat quality for recruiting *Callinectes* spp. *Marine Ecology Progress Series*, **573**, 1-14.

Johnston CA, Gruner DS (2018) Marine fauna sort at fine resolution in an ecotone of shifting wetland foundation species. *Ecology*, **99**, 2546-2557.

Johnston CA, Smith RS (2018) Vegetation structure mediates a shift in predator avoidance behavior in a range-edge population. *Behavioral Ecology*, **29**, 1124-1131.

Kangas PC, Lugo AE (1990) The distribution of mangroves and saltmarsh in Florida. *Tropical Ecology*, **31**, 32-39.

Kaplowitz MD (2001) Assessing mangrove products and services at the local level: the use of focus groups and individual interviews. *Landscape and Urban Planning*, **56**, 53-60.

Kauffman JB, Bhomia RK (2017) Ecosystem carbon stocks of mangroves across broad environmental gradients in West-Central Africa: Global and regional comparisons. *PLOS ONE*, **12**, article e0187749.

Kelleway JJ, Cavanaugh K, Rogers K, Feller IC, Ens E, Doughty C, Saintilan N (2017) Review of the ecosystem service implications of mangrove encroachment into salt marshes. *Global Change Biology*, **23**, 3967-3983.

Kelleway JJ, Saintilan N, Macreadie PI, Skilbeck CG, Zawadzki A, Ralph PJ (2016) Seventy years of continuous encroachment substantially increases 'blue carbon' capacity as mangroves replace intertidal salt marshes. *Global Change Biology*, **22**, 1097-1109.

Kennedy JP, Dangremond EM, Hayes MA, Preziosi RF, Rowntree JK, Feller IC (2020) Hurricanes overcome migration lag and shape intraspecific genetic variation beyond a poleward mangrove range limit. *Molecular Ecology,* **29**, 2583-2597.

Kennedy JP, Pil MW, Proffitt CE, Boeger WA, Stanford AM, Devlin DJ (2016) Postglacial expansion pathways of red mangrove, *Rhizophora mangle*, in the Caribbean Basin and Florida. *American Journal of Botany,* **103**, 260-276.

Kildow JT, Colgan CS, Pendleton L (2008) The changing coastal and ocean economies of the United States Gulf of Mexico. In: *Gulf of Mexico Origin, Waters, and Biota: Volume 2, Ocean and Coastal Economy* (ed Cato JC), pp 47. Harte Research Institute, Corpus Christi, Texas, USA.

Kimball ME, Eash-Loucks WE (2021) Estuarine nekton assemblages along a marsh-mangrove ecotone. *Estuaries and Coasts,* **44**, 1508-1520

Kirwan ML, Megonigal JP (2013) Tidal wetland stability in the face of human impacts and sea-level rise. *Nature,* **504**, 53-60.

Koenig CC, Bueno L, Coleman F *et al.* (2017) Diel, lunar, and seasonal spawning patterns of the Atlantic goliath grouper, *Epinephelus itajara*, off Florida, United States. *Bulletin of Marine Science,* **93**, 391-406.

Kossin JP, Hall T, Knutson T, Kunkel KE, Trapp RJ, Waliser DE, Wehner MF (2017) Extreme storms. In: *Climate Science Special Report: Fourth National Climate Assessment, Volume I* (eds Wuebbles DJ, Fahey DW, Hibbard KA, Dokken DJ, Stewart BC, Maycock TK), pp 257-276. U.S. Global Change Research Program, Washington, DC, USA.

Krauss KW, From AS, Doyle TW, Doyle TJ, Barry MJ (2011) Sea-level rise and landscape change influence mangrove encroachment onto marsh in the Ten Thousand Islands region of Florida, USA. *Journal of Coastal Conservation,* **15**, 629-638.

Krauss KW, McKee KL, Hester MW (2014a) Water use characteristics of black mangrove (*Avicennia germinans*) communities along an ecotone with marsh at a northern geographical limit. *Ecohydrology,* **7**, 354-365.

Krauss KW, McKee KL, Lovelock CE, Cahoon DR, Saintilan N, Reef R, Chen L (2014b) How mangrove forests adjust to rising sea level. *New Phytologist,* **202**, 19-34.

Kuhn AL, Kominoski JS, Armitage AR, Charles SP, Pennings SC, Weaver CA, Maddox TR (2021) Buried hurricane legacies: increased nutrient limitation and decreased root biomass in coastal wetlands. *Ecosphere,* **12**, e03674.

Lear G, Bellamy J, Case BS, Lee JE, Buckley HL (2014) Fine-scale spatial patterns in bacterial community composition and function within freshwater ponds. *The ISME journal,* **8**, 1715-1726.

Lewis DB, Brown JA, Jimenez KL (2014) Effects of flooding and warming on soil organic matter mineralization in *Avicennia germinans* mangrove forests and *Juncus roemerianus* salt marshes. *Estuarine, Coastal and Shelf Science,* **139**, 11-19.

Lewis DB, Jimenez KL, Abd-Elrahman A *et al.* (2021) Carbon and nitrogen pools and mobile fractions in surface soils across a mangrove saltmarsh ecotone. *Science of the Total Environment,* **798**, 149328.

Ley J, McIvor C, Montague C (1999) Fishes in mangrove prop-root habitats of northeastern Florida Bay: distinct assemblages across an estuarine gradient. *Estuarine, Coastal and Shelf Science*, **48**, 701-723.

Ley JA, McIvor C (2002) Linkages between estuarine and reef fish assemblages: enhancement by the presence of well-developed mangrove shorelines. In: *The Everglades, Florida Bay and coral reefs of the Florida Keys: an ecosystem sourcebook* (eds Porter JW, Porter KG), pp 539-562. CRC Press, Boca Raton, Florida, USA.

Lonard RI, Judd FW (1991) Comparison of the effects of the severe freezes of 1983 and 1989 on native woody plants in the Lower Rio Grande Valley, Texas. *The Southwestern Naturalist*, **36**, 213-217.

Longley WL (1995) Estuaries. In: *The Impact of Global Warming on Texas: A Report to the Task Force on Climate Change in Texas* (eds North GR, Schmandt J, Clarkson J), pp 88-118. The University of Texas, Austin, Texas, USA.

Loveless JB, Smee DL (2019) Changes in arthropod communities as black mangroves *Avicennia germinans* expand into Gulf of Mexico salt marshes. *Arthropod-Plant Interactions,* **13**, 465-475.

Lovelock CE, Feller IC, Reef R, Hickey S, Ball MC (2017) Mangrove dieback during fluctuating sea levels. *Scientific Reports,* **7**, article 1680.

Lovelock CE, Sorrell BK, Hancock N, Hua Q, Swales A (2010) Mangrove forest and soil development on a rapidly accreting shore in New Zealand. *Ecosystems,* **13**, 437-451.

Lundquist CJ, Morrisey DJ, Gladstone-Gallagher RV, Swales A (2014) Managing mangrove habitat expansion in New Zealand. In: *Mangrove ecosystems of Asia* (eds Faridah-Hanum I, Latiff A, Rehman Hakeem K, Ozturk M), pp 415-438. Springer, New York, New York, USA.

Lynch JC, Hensel P, Cahoon DR (2015) *The surface elevation table and marker horizon technique: A protocol for monitoring wetland elevation dynamics. Natural Resources Report NPS/NCBN/NRR-2015/1078,* U.S. National Park Service, Fort Collins, Colorado, USA.

Mace MM, Haffey ER, Kimball ME (2017) Low-temperature tolerance of juvenile tarpon *Megalops atlanticus*. *Environmental Biology of Fishes,* **100**, 913-922.

Macy A, Osland MJ, Cherry JA, Cebrian J (2020) Changes in ecosystem nitrogen and carbon allocation with black mangrove (*Avicennia germinans*) encroachment into *Spartina alterniflora* salt marsh. *Ecosystems,* **24**, 1007–1023.

Macy A, Sharma S, Sparks E *et al.* (2019) Tropicalization of the barrier islands of the northern Gulf of Mexico: A comparison of herbivory and decomposition rates between smooth cordgrass (*Spartina alterniflora*) and black mangrove (*Avicennia germinans*). *PLOS ONE*, **14**, e0210144.

Malinowski C, Coleman F, Koenig C, Locascio J, Murie D (2019) Are Atlantic goliath grouper, *Epinephelus itajara*, establishing more northerly spawning sites? Evidence from the northeast Gulf of Mexico. *Bulletin of Marine Science,* **95**, 371-391.

Marois DE, Mitsch WJ (2015) Coastal protection from tsunamis and cyclones provided by mangrove wetlands–a review. *International Journal of Biodiversity Science, Ecosystem Services & Management,* **11**, 71-83.

Martin JH, McEachron LW (1996) *Historical annotated review of winter kills of marine organisms in Texas bays. Management Data Series No. 118,* Texas Parks and Wildlife Department, Coastal Fisheries Division, Austin, Texas.

Martin S, Sparks EL, Constantin AJ, Cebrian J, Cherry JA (2021) Restoring fringing tidal marshes for ecological function and ecosystem resilience to moderate sea-level rise in the rorthern Gulf of Mexico. *Environmental Management*, **67**, 384-397.

Martinez L, O'Brien S, Bethel M, Penland S, Kulp M (2009) *Louisiana Barrier Island Comprehensive Monitoring Program (BICM) Volume 2: shoreline changes and barrier island land loss 1800's-2005,* University of New Orleans, Pontchartrain Institute for Environmental Sciences, New Orleans, Louisiana, USA.

McBride RA, Byrnes MR (1997) Regional variations in shore response along Barrier Island systems of the Mississippi River Delta plain: historical change and future prediction. *Journal of Coastal Research*, **13**, 628-655.

McKee KL (1993) Soil physicochemical patterns and mangrove species distribution - reciprocal effects? *Journal of Ecology*, **84**, 477-487.

McKee KL, Cahoon DR, Feller IC (2007) Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography*, **16**, 545-556.

McKee KL, Mendelssohn IA, Hester MW (2020) Hurricane sedimentation in a subtropical salt marsh-mangrove community is unaffected by vegetation type. *Estuarine, Coastal, and Shelf Science*, **239**, 106733.

McKee KL, Mendelssohn IA, Materne MD (2004) Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? *Global Ecology and Biogeography*, **13**, 65-73.

McKee KL, Rooth JE (2008) Where temperate meets tropical: multi-factorial effects of elevated CO_2, nitrogen enrichment, and competition on a mangrove-salt marsh community. *Global Change Biology*, **14**, 971-984.

McKee KL, Vervaeke WC (2018) Will fluctuations in salt marsh-mangrove dominance alter vulnerability of a subtropical wetland to sea-level rise? *Global Change Biology*, **24**, 1224-1238.

Mcleod E, Chmura GL, Bouillon S *et al.* (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment*, **9**, 552-560.

Minello TJ, Able KW, Weinstein MP, Hays CG (2003) Salt marshes as nurseries for nekton: testing hypotheses on density, growth and survival through meta-analysis. *Marine Ecology Progress Series*, **246**, 39-59.

Mitchell LR, Gabrey S, Marra PP, Erwin RM (2006) Impacts of marsh management on coastal-marsh bird habitats. *Terrestrial Vertebrates of Tidal Marshes: Evolution, Ecology, and Conservation.*

Mitsch WJ, Bernal B, Hernandez ME (2015) Ecosystem services of wetlands. *International Journal of Biodiversity Science, Ecosystems Services & Management*, **11**, 1-4.

Mitsch WJ, Gosselink JG (2007) *Wetlands,* John Wiley & Sons, New York, New York, USA.

Montagna PA, Gibeaut JC, Tunnell Jr JW (2007) South Texas climate 2100: coastal impacts. In: *The Changing Climate of South Texas 1900-2100: Problems and Prospects, Impacts and Implications* (eds Norwine J, John K), pp 57-77. CREST-RESSACA. Texas A & M University, Kingsville, Texas, USA.

Morris JT, Sundareshwar PV, Nietch CT, Kjerfve B, Cahoon DR (2002) Responses of coastal wetlands to rising sea level. *Ecology*, **83**, 2869-2877.

Mozdzer TJ, Kirwan M, McGlathery KJ, Zieman JC (2011) Nitrogen uptake by the shoots of smooth cordgrass *Spartina alterniflora*. *Marine Ecology Progress Series,* **433**, 43-52.

Nagelkerken I, Blaber SJM, Bouillon S *et al.* (2008) The habitat function of mangroves for terrestrial and marine fauna: a review. *Aquatic Botany,* **89**, 155-185.

Nagelkerken I, Faunce C (2008) What makes mangroves attractive to fish? Use of artificial units to test the influence of water depth, cross-shelf location, and presence of root structure. *Estuarine, Coastal and Shelf Science,* **79**, 559-565.

Narayan S, Thomas C, Matthewman J, Shepard CC, Geselbracht L, Nzerem K, Beck MW (2019) *Valuing the flood risk reduction benefits of Florida's mangroves,* The Nature Conservancy Gulf of Mexico Program, Florida, USA.

Nathan M (2020) *Plant-insect interactions in a shifting coastal ecosystem: Avicennia germinans and its associated arthropods. Ph.D. Dissertation,* University of Maryland, College Park, Maryland, USA.

Nelson JA, Lesser J, James WR, Behringer DP, Furka V, Doerr JC (2019) Food web response to foundation species change in a coastal ecosystem. *Food Webs,* **21**, e00125.

NOAA (1990) *Estuaries of the United States: vital statistics of a national resource base. A special NOAA 20th anniversary report.,* National Oceanic and Atmospheric Administration, National Ocean Service, Office of Oceanography and Marine Assessment, Rockville, MD.

NRDC (2010) *What's at stake: the economic value of the Gulf of Mexico's ocean resources.* http://docs.nrdc.org/water/files/wat_10051101a.pdf, Natural Reource Defense Council, Washington, D.C.

Odum WE, McIvor CC, Smith III TJ (1982) *The ecology of mangroves of south Florida: a community profile,* U.S. Fish and Wildlife Service, Office of Biological Services, Washington, D.C., USA FWS/OBS-81/24.

Oh C-O, Sutton SG, Sorice MG (2013) Assessing the role of recreation specialization in fishing site substitution. *Leisure Sciences,* **35**, 256-272.

Osland MJ, Day RH, From AS, McCoy ML, McLeod JL, Kelleway JJ (2015) Life stage influences the resistance and resilience of black mangrove forests to winter climate extremes. *Ecosphere,* **6**, Article 160.

Osland MJ, Day RH, Hall CT, Brumfield MD, Dugas JL, Jones WR (2017a) Mangrove expansion and contraction at a poleward range limit: climate extremes and land-ocean temperature gradients. *Ecology,* **98**, 125-137.

Osland MJ, Day RH, Hall CT *et al.* (2020a) Temperature thresholds for black mangrove (*Avicennia germinans*) freeze damage, mortality, and recovery in North America: refining tipping points for range expansion in a warming climate. *Journal of Ecology,* **108**, 654-665.

Osland MJ, Day RH, Michot TC (2020b) Frequency of extreme freeze events controls the distribution and structure of black mangroves (*Avicennia germinans*) near their northern range limit in coastal Louisiana. *Diversity and Distributions,* **26**, 1366-1382.

Osland MJ, Enwright N, Day RH, Doyle TW (2013) Winter climate change and coastal wetland foundation species: salt marshes vs. mangrove forests in the southeastern United States. *Global Change Biology,* **19**, 1482-1494.

Osland MJ, Enwright N, Stagg CL (2014) Freshwater availability and coastal wetland foundation species: ecological transitions along a rainfall gradient. *Ecology,* **95**, 2789-2802.

Osland MJ, Feher LC, Anderson GH *et al.* (2020c) A tropical cyclone-induced ecological regime shift: mangrove conversion to mudflat in Florida's Everglades National Park (Florida, USA). *Wetlands*, **40**, 1445-1458.

Osland MJ, Feher LC, Griffith KT *et al.* (2017b) Climatic controls on the global distribution, abundance, and species richness of mangrove forests. *Ecological Monographs,* **87**, 341-359.

Osland MJ, Feher LC, López-Portillo J, Day RH, Suman DO, Guzmán Menéndez JM, Rivera-Monroy VH (2018a) Mangrove forests in a rapidly changing world: global change impacts and conservation opportunities along the Gulf of Mexico coast. *Estuarine, Coastal and Shelf Science,* **214**, 120-140.

Osland MJ, Feher LC, Spivak AC *et al.* (2020d) Rapid peat development beneath created, maturing mangrove forests: ecosystem changes across a 25-year chronosequence. *Ecological Applications,* **30**, e02085.

Osland MJ, Gabler CA, Grace JB *et al.* (2018b) Climate and plant controls on soil organic matter in coastal wetlands. *Global Change Biology*, **24**, 5361-5379.

Osland MJ, Grace JB, Guntenspergen GR, Thorne KM, Carr JA, Feher LC (2019a) Climatic controls on the distribution of foundation plant species in coastal wetlands of the conterminous United States: knowledge gaps and emerging research needs. *Estuaries and Coasts,* **42**, 1991-2003.

Osland MJ, Hartmann AM, Day RH, Ross MH, Hall CT, Feher LC, Vervaeke WC (2019b) Microclimate influences mangrove freeze damage: implications for range expansion in response to changing macroclimate *Estuaries and Coasts*, **42**, 1084-1096.

Osland MJ, Stevens PW, Lamont MM *et al.* (2021) Tropicalization of temperate ecosystems in North America: The northward range expansion of tropical organisms in response to warming winter temperatures. *Global Change Biology*, **27**, 3009-3034.

Penfound WMT, Hathaway ES (1938) Plant communities in the marshlands of southeastern Louisiana. *Ecological Monographs,* **8**, 1-56.

Penland S, Zganjar C, Westphal KA, Connor P, List J, Williams SJ (2003) *Shoreline changes in the Timbalier Barrier Islands Arc- 1887-1996 Terrebonne Parish, Louisiana. In: U.S. Geological Survey Open-File Report 03-398.* https://pubs.usgs.gov/of/2003/of03-398/posters/pdf/cont_pdf/ti_atlas.pdf.

Pennings SC, Glazner RM, Hughes ZJ, Kominoski JS, Armitage AR (2021) Effects of mangrove cover on coastal erosion during a hurricane in Texas, USA. *Ecology*, **102**, e03309.

Perry CL (2007) Ecosystem effects of expanding populations of *Avicennia germinans* in a southeastern Louisiana *Spartina alterniflora* saltmarsh. M.S. Thesis. Lousiana State University. Department of Oceanography and Coastal Sciences, Baton Rouge.

Perry CL, Mendelssohn IA (2009) Ecosystem effects of expanding populations of *Avicennia germinans* in a Louisiana salt marsh. *Wetlands,* **29**, 396-406.

Pickens CN, Hester MW (2011) Temperature tolerance of early life history stages of black mangrove *Avicennia germinans*: implications for range expansion. *Estuaries and Coasts,* **34**, 824-830.

Pickett STA (1989) Space-for-time substitution as an alternative to long-term studies. In: *Long-term studies in ecology: approaches and alternatives* (ed Likens GE), pp 110–135. Springer-Verlag, New York, New York, USA.

Portnoy JW (1977) *Nesting colonies of seabirds and wading birds, coastal Louisiana, Mississippi, and Alabama. FWS/OBS-77/07,* U.S. Fish and Wildlife Service, Biological Services Program, NSTL Station, Mississippi, USA.

Poulakis GR, Stevens PW, Timmers AA, Wiley TR, Simpfendorfer CA (2011) Abiotic affinities and spatiotemporal distribution of the endangered smalltooth sawfish, *Pristis pectinata*, in a south-western Florida nursery. *Marine and Freshwater Research,* **62**, 1165-1177.

Pugesek BH, Baldwin MJ, Stehn T (2013) The relationship of blue crab abundance to winter mortality of Whooping Cranes. *The Wilson Journal of Ornithology,* **125**, 658-661.

Purtlebaugh CH, Martin CW, Allen MS (2020) Poleward expansion of common snook *Centropomus undecimalis* in the northeastern Gulf of Mexico and future research needs. *PLOS ONE,* **15**, e0234083.

Quisthoudt K, Schmitz N, Randin CF, Dahdouh-Guebas F, Robert EMR, Koedam N (2012) Temperature variation among mangrove latitudinal range limits worldwide. *Trees,* **26**, 1919-1931.

Raabe EA, Roy LC, McIvor CC (2012) Tampa Bay coastal wetlands: nineteenth to twentieth century tidal marsh-to-mangrove conversion. *Estuaries and Coasts,* **35**, 1145-1162.

Riley ME, Johnston CA, Feller IC, Griffen BD (2014) Range expansion of *Aratus pisonii* (Mangrove Tree Crab) into novel vegetative habitats. *Southeastern Naturalist,* **13**, N43-N48.

Rogers DI, Piersma T, Hassell CJ (2006) Roost availability may constrain shorebird distribution: exploring the energetic costs of roosting and disturbance around a tropical bay. *Biological Conservation,* **133**, 225-235.

Rogers K, Krauss KW (2019) Moving from generalisations to specificity about mangrove–saltmarsh dynamics. *Wetlands,* **39**, 1155-1178.

Rönnbäck P (1999) The ecological basis for economic value of seafood production supported by mangrove ecosystems. *Ecological Economics,* **29**, 235-252.

Ross MS, Meeder JF, Sah JP, Ruiz PL, Telesnicki GJ (2000) The southeast saline Everglades revisited: 50 years of coastal vegetation change. *Journal of Vegetation Science,* **11**, 101-112.

Rovai AS, Twilley RR, Castañeda-Moya E *et al.* (2018) Global controls on carbon storage in mangrove soils. *Nature Climate Change,* **8**, 534-538.

Rozas LP, Minello TJ, Henry CB (2000) An assessment of potential oil spill damage to salt marsh habitats and fishery resources in Galveston Bay, Texas. *Marine Pollution Bulletin,* **40**, 1148-1160.

Saenger P (2002) *Mangrove ecology, silviculture and conservation,* Springer, Dodrecht, Netherlands.

Saintilan N, Khan N, Ashe E, Kelleway J, Rogers K, Woodroffe CD, Horton B (2020) Thresholds of mangrove survival under rapid sea level rise. *Science,* **368**, 1118-1121.

Saintilan N, Williams RJ (1999) Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography,* **8**, 117-124.

Saintilan N, Wilson NC, Rogers K, Rajkaran A, Krauss KW (2014) Mangrove expansion and salt marsh decline at mangrove poleward limits. *Global Change Biology,* **20**, 147-157.

Sanford MT (2016) *Honey and its uses,* UF/IFAS Extension, Gainesville, Florida, USA.

Sasser CE, Evers-Hebert E, Holm GO, Milan B, Sasser JB, Peterson EF, DeLaune RD (2018) Relationships of marsh soil strength to belowground vegetation biomass in Louisiana coastal marshes. *Wetlands,* **38**, 401-409.

Scheffel W, Heck Jr KL, Cebrian J, Johnson M, Byron D (2013) Range expansion of black mangroves (*Avicennia germinans*) to the Mississippi barrier islands. *Gulf of Mexico Science*, **31**, 79-82.

Scheffel WA, Heck Jr KL, Rozas LP (2017) Effect of habitat complexity on predator—prey relationships: implications for black mangrove range expansion into northern Gulf of Mexico salt marshes. *Journal of Shellfish Research*, **36**, 181-188.

Scheffel WA, Heck KL, Johnson MW (2018) Tropicalization of the northern Gulf of Mexico: impacts of salt marsh transition to black mangrove dominance on faunal communities. *Estuaries and Coasts*, **41**, 1193-1205.

Schwarzer AC, Cox WA, Tornwall B (2020) Factors influencing occupancy and density of salt marsh songbirds in northeast Florida. *Journal of Field Ornithology*, **91**, 156-169.

Sherrod CL, McMillan C (1981) Black mangrove, *Avicennia germinans*, in Texas: past and present distribution. *Contributions in Marine Science*, **24**, 115-131.

Sherrod CL, McMillan C (1985) The distributional history and ecology of mangrove vegetation along the northern Gulf of Mexico coastal region. *Contributions in Marine Science*, **28**, 129-140.

Simpson L, Osborne T, Duckett L, Feller I (2017) Carbon storages along a climate induced coastal wetland gradient. *Wetlands*, **37**, 1023-1035.

Simpson L, Stein C, Osborne T, Feller I (2019) Mangroves dramatically increase carbon storage after 3 years of encroachment. *Hydrobiologia*, **834**, 13-26.

Simpson LT, Cherry JA, Smith RS, Feller IC (2020) Mangrove encroachment alters decomposition rate in saltmarsh through changes in litter quality. *Ecosystems*, **24**, 840-854.

Simpson LT, Feller IC, Chapman SK (2013) Effects of competition and nutrient enrichemnt on *Avicennia germinans* in the salt marsh-mangrove ecotone. *Aquatic Botany*, **104**, 55-59.

Sippo JZ, Lovelock CE, Santos IR, Sanders CJ, Maher DT (2018) Mangrove mortality in a changing climate: An overview. *Estuarine, Coastal and Shelf Science*, **215**, 241-249.

Smee DL, Sanchez JA, Diskin M, Trettin C (2017) Mangrove expansion into salt marshes alters associated faunal communities. *Estuarine, Coastal and Shelf Science*, **187**, 306-313.

Smith ET, Sheridan SC (2020) Where do cold air outbreaks occur and how have they changed over time? *Geophysical Research Letters*, **47**, e2020GL086983.

Smith RS, Osborne TZ, Feller IC, Byers JE (2019) Detrital traits affect substitutability of a range-expanding foundation species across latitude. *Oikos*, **128**, 1367-1380.

Snyder CM, Feher LC, Osland MJ, Miller CM, Hughes AR, Cummins KL (2021) The distribution and structure of mangroves (*Avicennia germinans* and *Rhizophora mangle*) near a rapidly changing range limit in the northeastern Gulf of Mexico. *Estuaries and Coasts*, **https://doi.org/10.1007/s12237-021-00951-0**.

Sparks EL, Cebrian J, Tobias CR, May CA (2015) Groundwater nitrogen processing in Northern Gulf of Mexico restored marshes. *journal of Environmental Management*, **150**, 206-215.

Stagg CL, Osland MJ, Moon JA *et al.* (2021) Extreme precipitation and flooding contribute to sudden vegetation dieback in a coastal salt marsh. *Plants*, **10**, 1841.

Stagg CL, Osland MJ, Moon JA *et al.* (2020) Quantifying hydrologic controls on local- and landscape-scale indicators of coastal wetland loss. *Annals of Botany*, **125**, 365-376.

Stehn TV, Prieto F (2010) Changes in winter whooping crane territories and range 1950-2006. In: *Proceedings of the Eleventh North American Crane Workshop, Sep 23-27, 2008* (ed

Hartup BK), pp 40-56. North American Crane Working Group, Baraboo, Wisconsin, USA.

Steinmuller HE, Foster TE, Boudreau P, Hinkle CR, Chambers LG (2019) Tipping points in the mangrove march: characterization of biogeochemical cycling along the mangrove–salt marsh ecotone. *Ecosystems,* **23**, 417-434.

Stevens PW, Blewett DA, Boucek RE *et al.* (2016) Resilience of a tropical sport fish population to a severe cold event varies across five estuaries in southern Florida. *Ecosphere,* **7**, Article e01400.

Stevens PW, Boucek RE, Trotter AA *et al.* (2018) Illustrating the value of cross-site comparisons: Habitat use by a large, euryhaline fish differs along a latitudinal gradient. *Fisheries Research,* **208**, 42-48.

Stevens PW, Fox SL, Montague CL (2006) The interplay between mangroves and saltmarshes at the transition between temperate and subtropical climate in Florida. *Wetlands Ecology and Management*, **14**, 435-444.

Stevens PW, Kimball ME, Elmo GM, Williams KL, Ritch JL, Crane DP (2021) Investigation into the occurrence of juvenile common snook Centropomus undecimalis, a subtropical estuarine sport fish, in saltmarshes beyond their established range. *Estuaries and Coasts,* **44**, 1477-1483.

Straw P, Saintilan N (2006) Loss of shorebird habitat as a result of mangrove incursion due to sea-level rise and urbanization. In: *Waterbirds around the world* (eds Boere GC, Galbraith CA, Stroud DA), pp 717-720. TSO Scotland, Edinburgh, UK.

Sullivan CR, Smyth AR, Martin CW, Reynolds LK (2021) How does mangrove expansion affect structure and function of adjacent seagrass meadows? *Estuaries and Coasts*, **101**, e02916.

Törnqvist TE, Jankowski KL, Li Y-X, González JL (2020) Tipping points of Mississippi Delta marshes due to accelerated sea-level rise. *Science Advances,* **6**, eaaz5512.

Twilley RR, Day JW (2012) Mangrove wetlands. In: *Estuarine Ecology* (eds Day JW, Crump BC, Kemp MW, Yáñez-Arancibia A), pp 165-202. John Wiley & Sons, Hoboken, New Jersey, USA.

USEPA (1999) *The ecological condition of estuaries in the Gulf of Mexico. EPA 620-R-98-004,* U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, Gulf Breeze, FL.

USGCRP (2017) *Climate Science Special Report: Fourth National Climate Assessment, Volume I,* U.S. Global Change Research Program, Washington, DC, USA.

Van der Stocken T, Carroll D, Menemenlis D, Simard M, Koedam N (2019a) Global-scale dispersal and connectivity in mangroves. *Proceedings of the National Academy of Sciences,* **116**, 915-922.

Van der Stocken T, Wee AK, De Ryck DJ *et al.* (2019b) A general framework for propagule dispersal in mangroves. *Biological Reviews,* **94**, 1547-1575.

Vaughn DR, Bianchi TS, Shields MR, Kenney WF, Osborne TZ (2020) Increased organic carbon burial in northern Florida mangrove-salt marsh transition sones. *Global Biogeochemical Cycles,* **34**, e2019GB006334.

Vergés A, Steinberg PD, Hay ME *et al.* (2014) The tropicalization of temperate marine ecosystems: climate-mediated changes in herbivory and community phase shifts. *Proceedings of the Royal Society B: Biological Sciences,* **281**, 20140846.

Visser JM, Vermillion WG, Evers DE, Linscombe RG, Sasser CE (2005) Nesting habitat requirements of the Brown Pelican and their management implications. *Journal of Coastal Research,* **21**, e27-e35.

Vitale N, Brush J, Powell A (2021) Loss of coastal islands along Florida's Big Bend Region: implications for breeding American oystercatchers. *Estuaries and Coasts,* **44**, 1173-1182.

Walcker R, Gandois L, Proisy C *et al.* (2018) Control of "blue carbon" storage by mangrove ageing: Evidence from a 66-year chronosequence in French Guiana. *Global Change Biology,* **24**, 2325-2338.

Walker JE, Angelini C, Safak I, Altieri AH, Osborne TZ (2019) Effects of changing vegetation composition on community structure, ecosystem functioning, and predator–prey interactions at the saltmarsh-mangrove ecotone. *Diversity,* **11**, 208.

Walter ST, Carloss MR, Hess TJ, Athrey G, Leberg PL (2013a) Movement patterns and population structure of the brown pelican. *The Condor,* **115**, 788-799.

Walter ST, Carloss MR, Hess TJ, Leberg PL (2013b) Hurricane, habitat degradation, and land loss effects on Brown Pelican nesting colonies. *Journal of Coastal Research,* **29**, 187-195.

Watts B (2016) *Status and distribution of the eastern black rail along the Atlantic and Gulf Coasts of North America. The Center for Conservation Biology Technical Report Series, CCBTR-16-09,* College of William and Mary/Virginia Commonwealth University, Williamsburg, Virginia, USA.

Weaver CA, Armitage AR (2018) Nutrient enrichment shifts mangrove height distribution: Implications for coastal woody encroachment. *PLOS ONE,* **13**, article e0193617.

Whitt AA (2016) *Utilizing citizen science to assess bird community composition within a changing marsh-mangrove ecotone in Texas. Master's Thesis,* Texas A&M University, Galveston, Texas, USA.

Williams AK, Bacosa HP, Quigg A (2017) The impact of dissolved inorganic nitrogen and phosphorous on responses of microbial plankton to the Texas City "Y" oil spill in Galveston Bay, Texas (USA). *Marine Pollution Bulletin,* **121**, 32-44.

Withers K (2002a) Shorebird use of coastal wetland and barrier island habitat in the Gulf of Mexico. *The Scientific World Journal,* **2**, 514-536.

Withers K (2002b) Wind-tidal flats. In: *The Laguna Madre of Texas and Tamaulipas* (eds Tunnell JW, Judd FW), pp 114-126. Texas A&M University Press, College Station, Texas, USA.

Woodley K (2004) Threats to shorebirds: Managing mangrove expansion in the Firth of Thames, New Zealand. In: *Status and conservation of shorebirds in the East Asian-Australasian Flyway: Proceedings of the Fourth Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia* (ed Straw P), pp 92-93. Wetlands International Global Series 18, International Wader Studies 17, Sydney, Australia.

Woodrey MS, Fournier AMV, Cooper RJ (2019) GoMAMN Strategic Bird Monitoring Guidelines: marsh birds. In: *Strategic bird monitoring guidelines for the northern Gulf of Mexico. Mississippi Agricultural and Forestry Experiment Station Research Bulletin 1228* (eds Wilson RR, Fournier AMV, Gleason JS, Lyons JE, Woodrey MS), pp 71-96. Mississippi State University, Starkville, Mississippi, USA.

Yando ES, Osland MJ, Hester MW (2018) Microspatial ecotone dynamics at a shifting range limit: plant-soil variation across salt marsh-mangrove interfaces. *Oecologia,* **187**, 319-331.

Yando ES, Osland MJ, Willis JM, Day RH, Krauss KW, Hester MW (2016) Salt marsh-mangrove ecotones: using structural gradients to investigate the effects of woody plant encroachment on plant-soil interactions and ecosystem carbon pools. *Journal of Ecology*, **104**, 1020-1031.

Yoskowitz D, Corollo C, Beseres-Pollack J, Welder K, Santos C, Francis J (2012) *Assessment of changing ecosystem services provided by marsh habitat in the Galveston Bay Region. Final Report to the Gulf of Mexico Foundation and the Habitat Conservation and Restoration Team of the Gulf of Mexico Alliance,* Texas A&M University. Harte Reseach Institute for Gulf of Mexico Studies, Corpus Christi.

Zedler JB, Leach MK (1998) Managing urban wetlands for multiple use: research, restoration, and recreation. *Urban Ecosystems,* **2**, 189-204.

Zwarts L (1988) Numbers and distribution of coastal waders in Guinea-Bissau. *Ardea,* **76**, 42-55.

Table 1. Mangrove forests and salt marshes are both frequently ranked among the most valuable ecosystems on the planet (Costanza et al. 2014). There are many similarities in the ecosystem goods and services provided by these tree- and grass-dominated coastal wetlands. The left column shows broad ecosystem service categories supported by mangrove forests and salt marshes (partially adapted from Barbier et al. 2011), and the right column provides more specific examples within those categories.

Ecosystem services provided by mangrove forests and salt marshes	**Examples**
Carbon sequestration	Plant carbon storage, soil carbon storage
Land loss avoidance	Elevation gains to offset relative sea-level rise
Erosion control	Shoreline erosion reduction
Water purification	Nutrient and sediment removal
Coastal protection	Wave and wind attenuation
Maintenance of fisheries	Fish, shrimp, and crabs
Maintenance of avifauna	Wading birds, shorebirds, marsh birds, and passerines
Recreation and tourism	Fishing, birdwatching, kayaking
Raw materials and food	Mangrove honey

Table 2. Mangrove expansion can produce beneficial changes in some ecosystem services while producing detrimental changes in other ecosystem services. Cedar Key (Florida) and Aransas (Texas) are two prominent areas where such trade-offs are observed. The table below shows some of the positive and negative changes associated with mangrove expansion in these two locations.

Location	**Positive change associated with mangrove expansion**	**Negative change associated with mangrove expansion**
Cedar Key (Florida)	*Improved coastal protection *Storm debris reduction *Enhanced pelican habitat *Increased aboveground carbon storage *Altered food webs and fisheries	*Loss of coastal views *Increased nuisance insects *Increased freeze vulnerability *Altered food webs and fisheries
Aransas (Texas)	*Improved coastal protection *Improved wind protection for fishing *Improved erosion control *Enhanced pelican habitat *Increased aboveground carbon storage *Increased soil carbon storage *Altered food webs and fisheries	*Reduced access to fishing grounds *Reduced whooping crane habitat and food resources *Increased freeze vulnerability *Altered food webs and fisheries

Table 3. Level of knowledge and critical research gaps regarding the impacts of mangrove expansion on ecosystem goods and services.

Ecosystem good and service	**Level of Knowledge**	**Critical Research Gaps**
Carbon sequestration	Better understood	Why is there so much variability in the effects of mangrove expansion on soil carbon storage and cycling? What is the role of site- and species-specific conditions (e.g., geomorphology, environmental variables, and the ecosystem properties of the interacting salt marsh and mangrove forest)?
Soil elevation change and vertical adjustments to sea-level rise	Somewhat understood	What are the effects of mangrove expansion on soil elevation dynamics and the ability of coastal wetlands to adjust to sea-level rise? How are these effects influenced by geomorphology, environmental conditions, and the ecosystem properties of the interacting salt marsh and mangrove forest?
Water quality: nutrient and sediment retention	Somewhat understood	What are the effects of salt marsh replacement by mangroves on nitrogen cycling and storage, and ultimately nitrogen pollution filtration by coastal wetlands?
Coastal protection	Poorly understood	How does mangrove expansion affect coastal protection against storms, including wave and wind attenuation? What are the effects of salt marsh replacement by mangroves on soil erosion rates, shear strength, and sediment deposition?
Coastal food webs and fisheries	Better understood	How are the effects of mangrove expansion on coastal food webs and fisheries influenced by site- and species-specific conditions (e.g., geomorphology, environmental conditions, and the ecosystem properties of the interacting salt marsh and mangrove forest)?
Avian communities	Poorly understood	How does mangrove expansion affect communities of wading birds, shore birds, marsh birds, and passerines? How are the perspectives of coastal resource managers on mangrove expansion influenced by avian conservation priorities of local organizations?
Insects, other terrestrial arthropods, and honey	Poorly understood	Can mangrove expansion provide a novel kind of honey for beekeepers? How does mangrove expansion affect terrestrial arthropod communities?
Recreation	Poorly understood	How does mangrove expansion affect recreation and tourism through changes in water access, viewscapes, birdwatching, and fishing?

Figure 1. The cumulative number of publications investigating the effects of mangrove expansion in the southeastern United States on (a) carbon sequestration; (b) soil elevation change and vertical adjustment to sea-level rise; (c) water quality, specifically nutrient and sediment retention; (d) coastal protection; (e) coastal food webs and fisheries; (f) avian communities; (g) insects and other terrestrial arthropods; and (h) recreation.

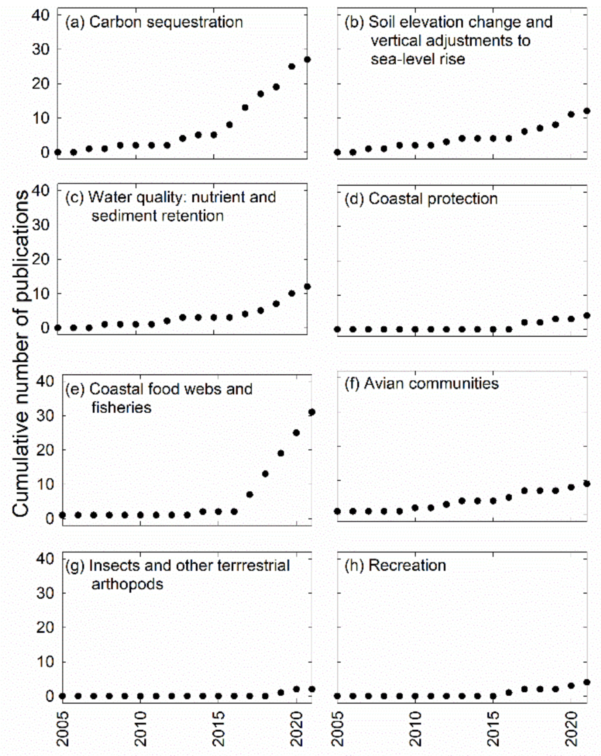

Figure 2. Map showing the distribution of mangroves in the southeastern United States. Mangrove expansion hotspots in Texas, Louisiana, and north Florida are shown with the four black ovals. Mangrove species' poleward range limits are shown with the black, red, and grey lines. Mangrove distribution data are from: (1) Mexico: CONABIO (2016); (2) Cuba: Giri et al. (2011); (3) south and central Florida: FDEP (2016); (4) northwestern Florida: Snyder et al. (2021); (5) Louisiana: Day et al. (2020); (6) Texas: Sherrod and McMillan (1981) and Armitage et al. (2015). Map updated and adapted from Osland et al. 2018.

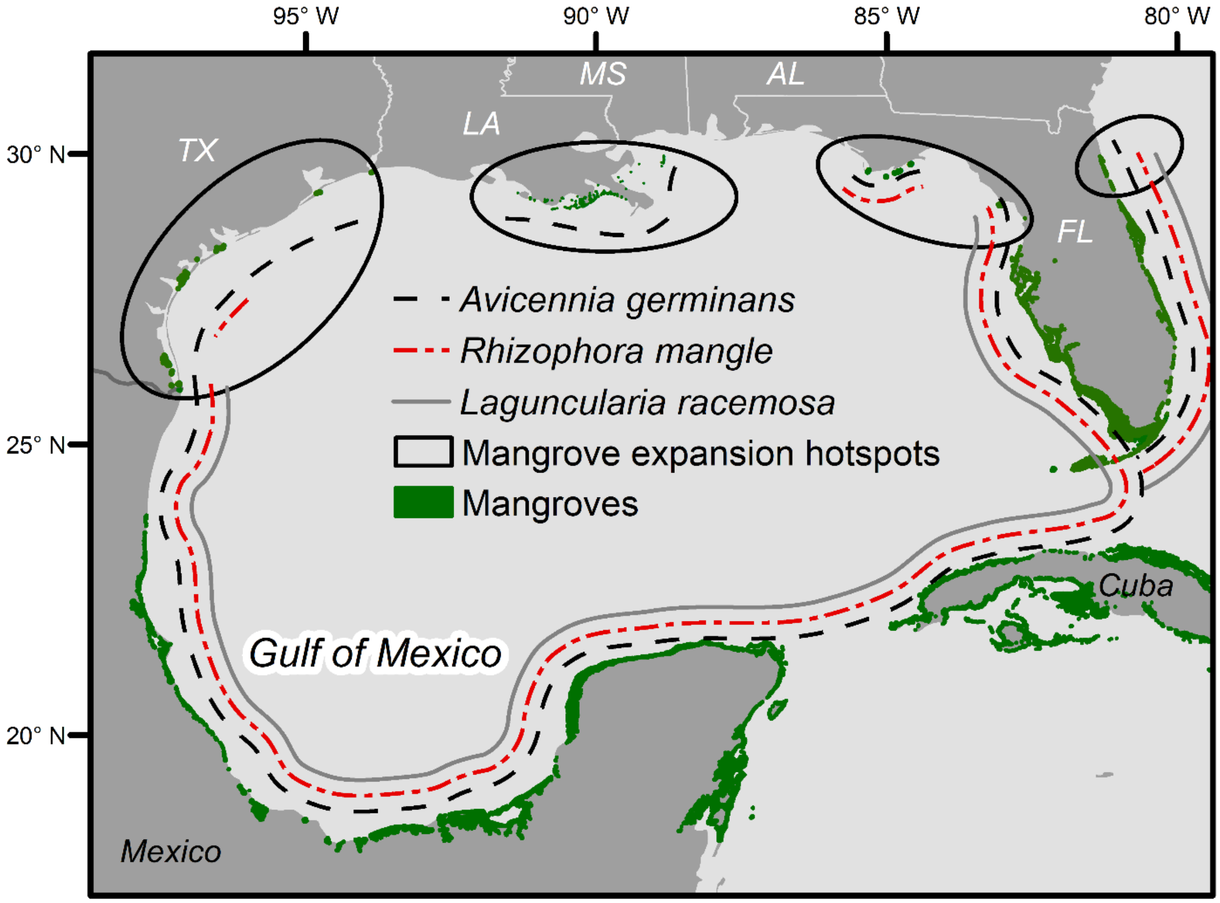

Figure 3. The effects of mangrove expansion on ecosystem goods and services are highly dependent upon the structure and composition of: (1) the expanding mangrove forests; and (2) the salt marsh that is being replaced. These photos illustrate some of the variation in marsh and mangrove plant communities near mangrove range limits in the southeastern United States. Upper photo: A mangrove-marsh ecotone in north Florida (near Cedar Key), which contains highly productive grass- and succulent plant-dominated marshes and comparatively tall (>6 m height) mangrove plants. Lower photo: A mangrove-marsh ecotone in south Texas (within Laguna Atascosa National Wildlife Refuge), which contains less productive succulent plant-dominated marshes, comparatively short (< 2 m height) mangrove plants, and hypersaline salt flats that lack vascular plants. Photo credits: Michael Osland

Figure 4. Across the southeastern United States, there is considerable variation in perceptions of the ecological and societal impacts of mangrove expansion into salt marsh. For very specific local reasons, some individuals may see mangrove expansion as a positive change to be embraced, while others may see mangrove expansion as a negative change to be constrained. These two photos provide an example of a negative and positive impact of mangrove expansion and salt marsh displacement. Left photo (negative impact): Along Texas' central coast, mangrove expansion reduces viable fishing areas. While fly fishing is possible within the region's grass and succulent plant-dominated salt marshes, fly fishing for redfish (*Sciaenops ocellatus*) is not possible within dense meter-tall mangrove stands. Right photo (positive impact): On Louisiana's barrier islands, expanding mangroves provide valuable habitat for brown pelicans (*Pelecanus occidentalis*). Photo credits: Chuck Naiser (left photo) and Louisiana Coastal Protection and Restoration Authority (right photo).