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Research Article

Effects of Future Sea Level Rise on Coastal Habitat

BETSY VON HOLLE,[1] *University of Central Florida, Department of Biology, Orlando, FL 32816, USA*
JENNIFER L. IRISH, *Virginia Tech, Civil & Environmental Engineering, Blacksburg, VA 24061, USA*
ANNETTE SPIVY, *University of Maryland, Department of Geography, College Park, MD 20742, USA*
JOHN F. WEISHAMPEL, *University of Central Florida, Department of Biology, Orlando, FL 32816, USA*
ANNE MEYLAN, *Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL 33701, USA*
MATTHEW H. GODFREY, *North Carolina Wildlife Resources Commission, Beaufort, NC 28516, USA*
MARK DODD, *Georgia Department of Natural Resources, Brunswick, GA 31520, USA*
SARA H. SCHWEITZER, *North Carolina Wildlife Resources Commission, Raleigh, NC 27699, USA*
TIM KEYES, *Georgia Department of Natural Resources, Atlanta, GA 30334, USA*
FELICIA SANDERS, *South Carolina Department of Natural Resources, McClellanville, SC 29458, USA*
MELISSA K. CHAPLIN, *U.S. Fish and Wildlife Service, Charleston, SC 29407, USA*
NICK R. TAYLOR, *Virginia Tech, Civil and Environmental Engineering, Blacksburg, VA 24061, USA*

ABSTRACT Sea level rise (SLR) and disturbances from increased storm activity are expected to diminish coastal ecosystems available to nesting species by removing habitat and inundating nests during incubation. We updated the United States Geological Survey's (USGS) Coastal Vulnerability Index, which provides a qualitative and relative assessment of a coastal area's vulnerability to erosion and shoreline retreat as a function of SLR and other factors, for the South Atlantic Bight. We considered a eustatic SLR projection of 14 cm by 2030. We linked long-term survey data for 3 sea turtle species, 3 shorebird species, and 5 seabird species to future coastal erosion vulnerability to SLR to understand effects of future SLR on nesting habitats. Over 2,000 km (43%) of the South Atlantic Bight coastline is projected to have an increase in coastal erosion vulnerability by the 2030s, with respect to its present vulnerability. Future vulnerability of SLR-induced erosion along the South Atlantic Bight is spatially variable, and the 11 coastal study species also varied in their use of coastal habitats with high future coastal vulnerability to SLR. For example, only 23% of high-density nesting habitat for the brown pelican (*Pelecanus occidentalis*) is expected to be at increased vulnerability to SLR, whereas >70% of the high-nesting density habitat for 2 seabird species (gull-billed tern [*Gelochelidon nilotica*], sandwich tern [*Thalasseus sandvicensis*]) is predicted to have higher future coastal erosion vulnerability by 2030. We provide predictions for the level of susceptibility of the study species to erosion from future SLR, which is the first step in managing coastal species for the changing environmental conditions associated with climate change and SLR. © 2019 The Authors. *Journal of Wildlife Management* Published by Wiley Periodicals, Inc. on behalf of The Wildlife Society.

KEY WORDS coastal nesting, sea level rise, seabird, sea turtle, shorebird, South Atlantic Bight.

Sea level rise (SLR) varies regionally and is comprised of eustatic SLR and local SLR contributions due to processes including sediment compaction, anthropogenic activity, and tectonics. Global sea levels over the last century have risen 1.5–1.9 mm/year, with an acceleration to about 2.8–3.6 mm/year over the last 2 decades (Intergovernmental Panel on Climate Change [IPCC] 2013). Seventeen to 21 cm of SLR has been recorded from 1901–2010 (IPCC 2013). Global SLR projections demonstrate scientific consensus that sea levels will continue to rise over the coming decades at a rate that likely exceeds historical rates (National Research Council [NRC] 2012, Parris et al. 2012, IPCC 2013). The range of these projections, however, varies from 20 cm to 140 cm by 2100. In the shorter term, these global SLR projections range from 8 to 21 cm by 2030. The greatest unknown in predictions of SLR over the next century stem from the potential for rapid dynamic collapse of ice sheets (Allison et al. 2009, Bamber and Aspinall 2013).

Erosion and the land-cover changes projected to occur as a result of future SLR have the potential to increase coastal flooding (Lin et al. 2012, Woodruff et al. 2013, Resio and Irish 2015) and alter tidal dynamics (Hagen et al. 2013). For example, historical SLR in the New Orleans area of about

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[1]*E-mail: vonholle@mail.ucf.edu*

0.75 m since 1900 increased Hurricane Katrina's (2005) flood elevations by about 1.3 m (double the SLR amount), largely because of the SLR-induced loss of wetlands in the area (Irish et al. 2014). Low-lying narrow coastal and island beaches are particularly sensitive to SLR (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006, FitzGerald et al. 2008) and are especially vulnerable when coastal development prevents landward migration of beaches (Fish et al. 2008). Yet, prediction of these processes is difficult and the subject of ongoing research (FitzGerald et al. 2008). It is possible that coastal dunes can respond to SLR by migrating landward (Duran and Moore 2013); however, this can only take place in areas where there are no hard structures such as roads and buildings that inhibit this landward migration (Noss 2011). This coastal squeeze will be amplified with increasing human densities along the coasts (Small and Nicholls 2003).

Research investigating species response to climate change has primarily focused on the response of species to temperature and precipitation changes and variability (Noss 2011). However, rising sea levels will have effects on coastal species and ecosystems (Maschinski et al. 2011, Noss 2011, Saha et al. 2011, Reece and Noss 2014, Woodland et al. 2017). Several studies thus far have examined the potential effect of SLR on sea turtle nesting beaches using high precision digital elevation models (DEMs) coupled to simple passive inundation (bathtub) models (Baker et al. 2006). A Hawaiian study (Baker et al. 2006) predicted that up to 40% of low profile, green turtle (*Chelonia mydas*) nesting beaches could be flooded with 0.9 m of SLR. Other studies in Barbados (Fish et al. 2008) and Bonaire (Fish et al. 2005) suggested similar losses (~50%) of hawksbill turtle (*Eretmochelys imbricate*) nesting habitat with SLR. Inundation due to SLR will reduce foraging areas available to shorebirds in those areas where the tidal zone movement is inhibited by topography or human structures, such as seawalls (Galbraith et al. 2002). Seabirds may be the most vulnerable of all bird species to the effects of climate change, including SLR, because of their use of marine and terrestrial habitats and existence at the ecotone of the atmosphere and the ocean (Sydeman et al. 2012).

Coastal nesting ecosystems are dynamic systems subject to disturbance (i.e., erosion, accretion, overwash) as a result of regular tidal and storm events. Coastal nesting habitats already affected by anthropogenic activities are likely to be squeezed even more as sea levels rise. Seawater inundation leads to the failure of sea turtle nests to produce hatchlings when embryonic development is halted because of reduced oxygen exchange (Ragotzkie 1959, Martin 1996). Nests of shorebird and seabird species are regularly washed out with storm surges (George and Schweitzer 2004, Brooks et al. 2013, Denmon et al. 2013, Jodice et al. 2014, Schulte and Simons 2015). Spatial (location along a beach, elevation and distance from the high water line, nest depth) and temporal (time of year in relation to seasonal storm events, including hurricanes) nesting characteristics influence the susceptibility of different sea turtle (Pike and Stiner 2007), shorebird, and seabird species (Convertino et al. 2011). In addition to reducing the available nesting habitat, the frequency and extent of these disturbance events will be exacerbated as a result of SLR (Zhang et al. 2004); thus, storm surge events with a 1% annual exceedance frequency under SLR may become events with a 2–3% annual exceedance frequency (McInnes et al. 2003).

Past studies demonstrate SLR will affect populations of coastal nesting species (Maschinski et al. 2011, Noss 2011, Saha et al. 2011, Reece and Noss 2014, Woodland et al. 2017). Although there have been many landscape-scale studies of coastal vulnerability to SLR (Titus and Richman 2001, Titus et al. 2009, Noss 2011, Passeri et al. 2016), including human populations that will be affected by SLR (Hauer et al. 2015) and archaeological site vulnerability to SLR (Robinson et al. 2010), there have been relatively few spatially explicit landscape-scale analyses of coastal species vulnerability to SLR (Hunter et al. 2015, Woodland et al. 2017).

We integrated selected future SLR projections with long-term (2005–2013) field biological observations to predict vulnerable nesting ecosystems for 11 species. Additionally, we provide broad-scale maps of habitat vulnerability under a 2030 SLR projection, which will allow natural area managers to construct strategies for mitigating the effects of a changing climate on these coastal nesting species. Our objective of this predictive study, using observational data, was to predict which species will be most affected by SLR along the South Atlantic Bight and the ecosystems that will be the most affected by SLR.

STUDY AREA

We investigated the effect of local SLR and eustatic SLR on important sea turtle, seabird, and shorebird nesting habitat across the South Atlantic Bight (SAB; i.e., coastal USA between Cape Hatteras, NC and Sebastian Inlet, FL) using data collected between 2005 and 2013. The SAB is characterized by a broad, shallow, and flat coastal shelf bounded by the Gulf Stream, with warm, temperate waters, 4 seasons (winter, spring, summer, and fall) and extensive intertidal wetlands (Conley et al. 2017). The SAB is comprised of a mixture of agricultural, developed, and conservation lands, with Florida having the highest percentage of developed lands and North Carolina having the highest percentages of agricultural and conservation lands (Conley et al. 2017). For this study, we used all coastal areas immediately adjacent to the Atlantic Ocean (11,046 ha; Conley et al. 2017), except inlets and inter-coastal waterways. This included beaches, dunes, swales, and marshes. Eighty-five percent of our study area was characterized as low-lying barrier islands and 14% were beaches fronting low-lying inland areas. Beaches, dunes, swales, and marshes are breeding grounds for a diversity of seabirds and shorebirds, and several species of sea turtles (Conley et al. 2017). These land cover types also provide overwintering sites for migrant shorebirds (Harrington 1999).

Our study species included loggerhead sea turtle (*Caretta caretta*), green turtle, leatherback sea turtle (*Dermochelys coriacea*), American oystercatcher (*Haematopus palliatus*), piping plover (*Charadrius melodus*), Wilson's plover (*C. wilsonia*), black skimmer (*Rynchops niger*), gull-billed tern (*Gelochelidon nilotica*), royal tern (*Thalasseus maximus*), sandwich tern (*T. sandvicensis*), and brown pelican (*Pelecanus occidentalis*). Piping plovers (when nesting in the Great Lakes) and leatherback sea turtles are listed as Endangered. Loggerhead and green turtles (when nesting outside of Florida in our study area) are listed as Threatened under the Endangered Species Act (U.S. Fish and Wildlife Service [USFWS] 1973, 2016). In our study area, piping plover populations that breed along the Atlantic coast, outside of Florida, are classified as Threatened (USFWS 2016).

METHODS

Coastal Vulnerability Index Calculations

To broadly assess the area's vulnerability to physical change induced by SLR, we adopted the United States Geological Survey's (USGS) Coastal Vulnerability Index (CVI; Thieler and Hammar-Klose 1999). This approach provides a qualitative and relative assessment of vulnerability to physical change (i.e., vulnerability to coastal evolution as sea levels rise). Coastal evolution in our study area will present itself as accelerated coastal erosion and shoreline retreat because 85% of the study region is characterized as low-lying barrier islands and 14% are beaches fronting low-lying inland areas. Along the barrier islands, overall land loss is likely to occur in areas where human development hinders natural landward barrier island migration. We acknowledge, however, that there are small stretches of coastline where fringing vegetation is directly exposed to the open ocean (e.g., wetlands fronting relatively steep inland terrain along the open coast in Awendaw, SC); the direct loss of these areas due to oceanic processes combined with topographic constraints is expected to be the dominant influence of coastal evolution. We assumed that regions where higher CVI are projected would experience aggravated erosion and that this erosion could lead to increased inundation from SLR and from surge during episodic storm events (Passeri et al. 2015).

The qualitative CVI is based on quantitative estimates to characterize the physical setting: geomorphology (a classification of the shoreline's relative erodibility [G]), coastal slope (S), relative (local plus eustatic) SLR rate (RSLRR), shoreline erosion rate (ER), mean tide range (MTR), and mean wave height (MWH; Thieler and Hammar-Klose [1999] provide classification and methods details). For each of the 6 factors, we assigned a vulnerability rank from 1 to 5 where 1 indicates low vulnerability and 5 indicates high vulnerability. We determined an aggregate vulnerability by first computing the square root of the product of the vulnerability ranks for the 6 variables ($\widehat{\mathrm{CVI}}$):

$$\widehat{\mathrm{CVI}} = \sqrt{\mathrm{Rank(G)} \cdot \mathrm{Rank(ER)} \cdot \mathrm{Rank(S)} \cdot \mathrm{Rank(RSLRR)} \cdot \mathrm{Rank(MTR)} \cdot \mathrm{Rank(MWH)}/6} \quad (1)$$

Finally, we assigned a CVI rank from 1 (low) to 4 (high) based on the magnitude of the calculated $\widehat{\mathrm{CVI}}$.

The published CVI (Thieler and Hammar-Klose 1999) for the United States Atlantic coast has a resolution of 5 km. This published CVI was based on historical trends, and thus does not consider future climate change. We modified the CVI calculations to project future coastal erosion vulnerability for the SAB under various projections of future SLR. Of the 6 CVI factors, climate change and SLR potentially affect 4: mean tide range, mean wave height, relative SLR rate, and shoreline erosion rate. Of these factors, only relative SLR rate had a measurable effect on CVI ranking projections using the published CVI method (Thieler and Hammar-Klose 1999). To account for projected future change in relative SLR rate, we adopted the probabilistic eustatic SLR (ESLR) projection from Kopp et al. (2014) of 14 cm by 2030 (2000 base year), representing a median (50% probability) outcome for 3 Representative Concentration Pathways (RCPs): RCP 2.6, RCP 4.5, and RCP 8.5 (Meinshausen et al. 2011). We assumed future local SLR rates will continue at the historical rates. As such, we calculated future projected relative SLR rate for 2030 as:

$$\mathrm{RSLRR_{new}} = \left(\mathrm{RSLRR_{original}} - \mathrm{ESLRR_{original}}\right) + \mathrm{ESLRR_{new}}, \quad (2)$$

where the term in parentheses represents the local SLR rate contribution. The observed historical $\mathrm{ESLRR_{original}}$ was 1.8 mm/year over the last century (Church and Gregory 2001, White et al. 2005, Church and White 2006). We assumed the $\mathrm{ESLRR_{new}}$ equaled the above estimated mean ESLR value divided by the number of years since the base year of 2000; for example, $\mathrm{ESLRR_{new}} = (14\,\mathrm{cm})(10\,\mathrm{mm/cm})/(30\ \mathrm{year}) = 4.7\,\mathrm{mm/year}$. Finally, we took the $\mathrm{RSLRR_{original}}$ as the relative SLR rate given in the published CVI on a location-by-location basis (Thieler and Hammar-Klose 1999).

The CVI approach yields a qualitative estimate of relative vulnerability to coastal erosion based on quantitative empirical metrics. We adopted this approach to estimate relative change in shoreline vulnerability due to SLR by 2030. All shoreline vulnerability projections, from physics-based numerical simulation of coastal evolution that are computationally intensive and expensive (Ranasinghe et al. 2011), to rigorous empirically based models (Plant et al. 2016, Vitousek et al. 2017), to the more qualitatively based CVI, rely to some degree on upon approximations of multifaceted, multiscale processes, and thus are subject to many sources of error (Ranasinghe et al. 2011, Plant et al. 2016, Vitousek et al. 2017).

SLR Effects on Species

We compiled spatial location data for sea turtle, seabird, and shorebird species along the coastline abutting the SAB using historical nesting or occupancy spatial data for each species, collected by state agencies from 2005 to 2013 (Table 1). Within the SAB, 1,250 km were surveyed for sea turtle nests, 1,004 km for nesting shorebirds, 804 km for wintering piping plovers (*Charadrius melodus*), and 532 km for seabird nests. We produced spatially explicit nest density maps with global positioning system (GPS) data for our 11 study species across the coastline abutting the SAB using historical sea turtle, seabird, and shorebird nesting data. These provided primary locations along the shoreline and proximity to the high tide line. We integrated all geographic information system (GIS) and location-specific data of sea turtle, seabird, and shorebird nests and territorial pair locations collected across our study region into a data layer, at the scale of named beaches, for this study. We used data that had been previously collected by state agencies for monitoring each of the respective study taxa. The original data collection followed protocols and guidelines related to use of vertebrate animals in effect at the time the data were collected.

Sea turtles are iteroparous, making reproductive migrations once every few years. Female sea turtle nesting efforts leave conspicuous marks in the sand that can be used by trained surveyors to determine what species nested and whether eggs were deposited. Surveyors evaluated visual characteristics such as gait (alternating *vs.* synchronous flipper marks), track width, and nest size and shape during early-morning, daily surveys (Witherington et al. 2009). Surveyors received annual training from state agency biologists and followed standardized protocols for crawl evaluation. Trained personnel conducted annual surveys for turtle nests along the coastlines of Florida (GA border to Sebastian Inlet, FL), Georgia, and North Carolina from 2005–2011, and for South Carolina for 2010–2011.

Trained state agency personnel conducted winter and breeding season surveys for shorebirds and seabirds every 3 to 5 years within coastal estuarine areas of Florida, Georgia, North Carolina, and South Carolina (Table 1). Surveyors identified roost sites of wintering birds and nests of breeding birds by foot, boat, vehicle and aircraft, and recorded their locations with GPS. Surveyors counted nests and roost sites with transects or area counts depending on site conditions and type (e.g., marsh island, barrier island beach).

Species Densities

We ranked coastal areas based on relative density of nests, or nesting pairs, which determined the relative density for each species. For each species, we calculated an average density across all survey years along the SAB to reduce variance across states. We classified nest or species densities per beach in 1 of 5 categories: none, low, moderate, high, and very high for each species for our area of interest (Table SA1, available online in Supporting Information). We classified the species density data as none when the species was not present. We assigned quantile breaks in GIS (ArcGIS 10.1) for nest distributions and categories; this method classifies data into a certain number of categories with an equal number of units in each category (Environmental Systems Research Institute 2015).

To delineate the extent of occupied beach areas for sea turtles, we calculated nest densities (i.e., nests/linear shoreline distance [nests/km]) for each surveyed beach within the SAB. From the survey data, we calculated nest densities for each of the 3 species, for each of the survey years. We calculated the average nest density, across all survey years, for each species. We calculated these densities from the annual surveys that were conducted for the coastlines of Florida, Georgia, and North Carolina from 2005–2011, and for South Carolina for 2010–2011.

We calculated densities separately for colonial nesting seabirds and solitary shorebirds; seabird data sets contained the following fields: nest number, adult number, and chick number. We based nest density on average nest densities across the various surveyed years (Table 1). If a seabird or shorebird nest or wintering piping plover individual was recorded anywhere off the Atlantic-facing beach within the SAB, we assigned these nests or individuals to an associated beach, which was the closest Atlantic-facing beach within a 4.8-km radius of that point for seabirds and a 1.6-km radius for shorebirds. We based these distances on expert opinion of the distances that nesting seabirds and shorebirds will regularly travel to use our study areas directly adjacent to the Atlantic. We defined nest density as the number of nests/km of surveyed beach for seabirds and number of nesting pairs/beach for shorebirds. We used nesting density, or number of

Table 1. Data sources compiled for point location occurrences of seabird and shorebird nesting and wintering sites in the South Atlantic Bight.

		Seabird					Shorebird		Wintering shorebird
State	Sources[a]	Black skimmer	Gull-billed tern	Royal tern	Sandwich tern	Brown pelican	American oystercatcher	Wilson's plover	Piping plover
FL	FWC-Beach, FWC-Shorebird	2006, 2008–2011	2006, 2008–2011	2006, 2009–2011	2009, 2011	2011	2006–2011	2006–2011	2009–2012
GA	GADNR	2010–2011	2010–2011	2010–2011	2010–2011	2010–2011	2010–2011	2010–2011	2006–2013
SC	SCDNR, USFWS	2005–2011	2005–2011	2005–2011	2005–2011	2005–2011	2008	2009–2011	2006–2007 2007–2008
NC	NCWRC	2004, 2007, 2011	2004, 2007, 2011	2004, 2007, 2011	2004, 2007, 2011	2004, 2007, 2011	2007	2007	2007, 2010, 2011

[a] FWC-Beach = Florida Fish and Wildlife Conservation Commission (FWC) Nesting Birds website (2005–2010), FWC-Shorebird = FWC Florida Shorebird database (2011), GADNR = Georgia Department of Natural Resources, SCDNR = South Carolina Department of Natural Resources, USFWS = United States Fish and Wildlife Service, NCWRC = North Carolina Wildlife Resources Commission.

nests/linear shoreline distance (nests/km), and this approach did not account for external factors, such as prey resources. We calculated densities separately for colonial nesting seabirds and solitary shorebirds. We averaged the density of nests, or nesting pairs, across the survey years (Table 1).

For the solitary nesting shorebirds in our study region (American oystercatcher and Wilson's plover), we determined density calculations from the number of nesting pairs observed. We used the following calculations to select which parameter best delineated the density of occupied beach area. If the number of nesting pairs was >50% of the number of adults, then we used the number of nesting pairs. If 50% the number of adults was greater than the number of nesting pairs, then we used 50% the number of adults. If the number of nests was the only count taken, then we used that number.

For the colonial nesting seabirds in our study region (black skimmer, gull-billed tern, royal tern, sandwich tern, and brown pelican), we determined density from the number of nests observed. We used the following calculations to select which parameter best delineated density. If 50% of the number of adults was greater than the number of nests, then we used 50% the number of adults. If the number of nests was >50% the number of adults, then we used the number of nests. If the number of chicks was the only count taken, then we used that number.

Habitat Vulnerability

We combined the area's broad vulnerability to erosion and shoreline retreat under SLR, as represented by the calculated $\widehat{CVI}$ (Equation (1)), with coastal nesting density maps of our focal species to create habitat vulnerability maps for the 2000 baseline and the 2030 SLR projection. We viewed this as an initial effort to broadly identify those areas currently occupied by bird and sea turtle species that are vulnerable to future SLR and to quantify current nesting densities by these coastal species. Integrating these 2 measures allows coastal mangers, for example, to identify those areas with the highest current nesting densities that are also predicted to have high vulnerability to coastal change under SLR. In those areas where SLR causes increased erosion of breeding colonies and nesting sites, we expect reproductive output of these coastal nesting species to be reduced.

We appended the CVI for 2000 and 2030 to the density layer. We calculated the $\widehat{CVI}$ across our study area in 5-km segments, whereas the species survey data were observed at the level of named beaches. Analysis of the calculated $\widehat{CVI}$ and species density data together required integration of the 2 different data types (calculated $\widehat{CVI}$, species density) to the same spatial scale. To integrate the data, we performed a spatial join of the calculated $\widehat{CVI}$ (in 5-km segments) with the species density data (at the scale of named beaches). We conducted data integration to extract the closest beach name to the calculated $\widehat{CVI}$. We then used the summary statistic tool in ArcGIS 10.1 to calculate a weighted mean CVI ranking for each beach based on the length of each segment of the calculated $\widehat{CVI}$. For example, on Amelia Island in Nassau County, Florida, there were 3 named beaches: Amelia Island Beach (18.41 km), Amelia Island State Park (2.39 km), and Fort Clinch State Park (4.08 km). To calculate the weighted mean CVI for Amelia Island Beach, we spatially matched the respective 5-km CVI segments to create an overall weighted mean CVI for the entire 18.41-km beach. This calculation allowed for a more accurate interpretation of the average CVI ranking for each beach. Additionally, we assigned an overall CVI ranking of very high, high, moderate, and low to each beach within the study area, as described above.

To assess the broad vulnerability of current distributions of coastal nesting species to SLR, we presented the change in vulnerability to SLR from the 2000 baseline to the 2030 SLR projection for all beaches surveyed for that species, which includes those beaches where individuals of that species were not observed, and all habitat for that species, which includes only areas where that species was observed. We further subdivided all habitat for that species into high and low density categories. We attributed high density (for an individual species) to any area assigned a value of very high or high in the species density ranking calculations. We defined low density (for an individual species) as any species assigned a value of low or moderate in the species density ranking calculations.

RESULTS

There was a substantial increase in coastal erosion vulnerability under a modest increase in SLR by 2030 (Fig. 1). The projected number of coastal segments for which the refined CVI ranking was in the highest level of

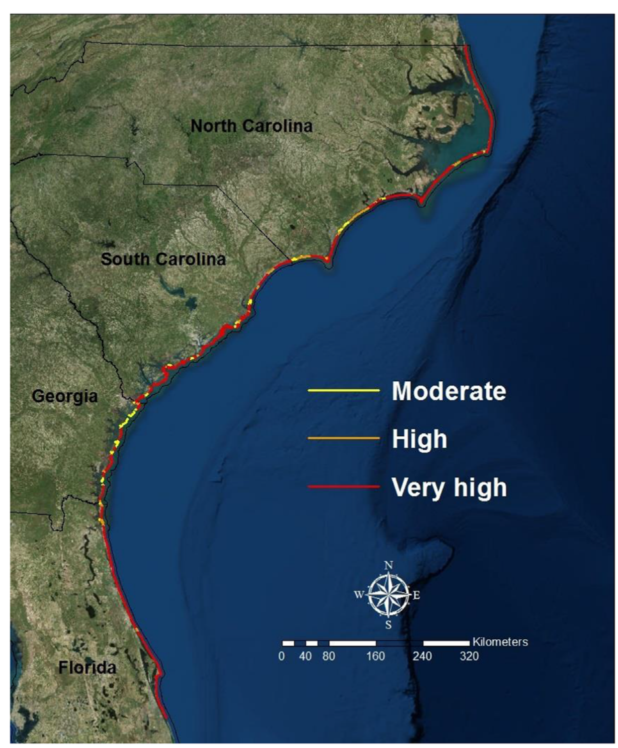

Figure 1. Coastal Vulnerability Index (CVI), created for the southeastern Atlantic coast of the United States for 2030, which provides an overall assessment of a coastal area's vulnerability to erosion and inundation.

vulnerability, very high, increases 30%, from 429 (45%) to 713 (75%). On the contrary, the lowest CVI ranking (low) dropped from 20 (2%) in 2000 to 0 (0%) coastal segments by 2030. Additionally, the projected number of coastal segments in the moderate category will drop from 265 (28%) to 115 (12%) and the projected number of coastal segments in the high category will drop from 235 (25%) to 121 (13%) by 2030. For the 2030 projection, the smallest relative SLR rate for any coastal segment in the SAB was projected to be 3.77 mm/year; thus, we projected that by 2030 the relative SLR rate at all locations will be categorized in the highest refined relative SLR rate vulnerability ranking (>3.35 mm/yr). Based on comparison of the baseline 2000 CVI rankings and the projected CVI for 2030, 2,035 km, or 43% of the SAB coastline, will have an increase in coastal erosion vulnerability (Fig. 1; Figs. SA1–SA11, available online in Supporting Information).

By 2030, 47% (586 km) of all beaches surveyed for sea turtle nesting across the study region (1,250 km) will have increased vulnerability to erosion, as compared to SLR rates in 2000 (Fig. 2). Similarly, 47% of loggerhead habitat was projected to be at increased vulnerability by 2030 (Table 2). Additionally, 48% (244 km) of the high nesting density habitat for this species was projected to be more vulnerable to erosion and inundation by 2030, as compared to historical levels. Forty-four percent (363 km) of the nesting habitat and 50% of the high-density nesting habitat for the green turtle is projected to be at increased vulnerability by 2030. Similarly, 39% (326 km) of all nesting habitat for leatherback sea turtles will have increased vulnerability by 2030, whereas 42% of the high-density nesting habitat for this species was projected to be at increased vulnerability to SLR. The percent increase in vulnerability to SLR of all beaches surveyed for sea turtles between 2000 and 2030 were similar to the increase in vulnerability for all nesting habitats and high-density habitats for the 3 sea turtle species.

Table 2. The length of all nesting habitat and high-density nesting habitat with increased vulnerability to coastal erosion due to sea level rise for each focal species in the South Atlantic Bight. The length is the beach area with increased vulnerability to sea level rise, relative to historical vulnerability (2000), for all nesting habitat for each species or for high-density (very high and high) nesting habitat for each species, by 2030.

	All nesting habitat		High nesting density	
	Length (km)	%	Length (km)	%
Sea turtles				
Loggerhead	586	47	244	48
Green turtle	395	44	268	50
Leatherback	326	39	98	42
Shorebirds				
Wilson's plover	363	51	217	53
Piping plover[a]	311	60	178	61
American oystercatcher	311	44	117	52
Seabirds				
Gull-billed tern	100	34	58	79
Brown pelican	228	42	45	23
Black skimmer	231	49	55	54
Royal tern	227	24	80	53
Sandwich tern	198	16	39	70

[a] For piping plover, these numbers represent wintering habitat, and not nesting habitat.

By 2030, 51% (363 km) of all coastal habitat used for nesting by Wilson's plovers and 44% of all habitat used by American oystercatchers for nests was projected to be more vulnerable to erosion due to SLR, as compared to its 2000 vulnerability (Table 2; Fig. 3). High-density nesting habitats for Wilson's plover and the American oystercatcher were projected to have 53% and 52% increased vulnerability to SLR, respectively, whereas 41% of all areas surveyed for these species will have increased vulnerability, as compared to 2000 (Fig. 3). This suggests that high nesting density and all nesting habitat for these 2 species are projected to be more vulnerable in the future, as compared to the overall area that was surveyed for these species. It is projected that, of the habitat used in winter by the piping plover, 50% (257 km) will have increased vulnerability by 2030, as compared to

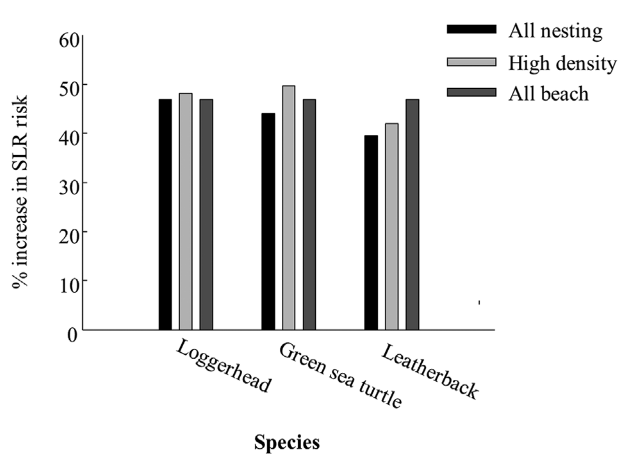

Figure 2. Nesting beach habitat in the South Atlantic Bight with increased vulnerability to sea level rise (SLR), for leatherback, loggerhead, and green turtles by 2030, as compared to historical vulnerability (in 2000) to SLR.

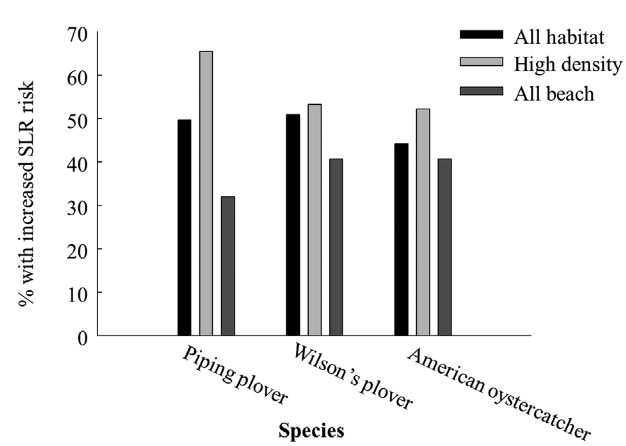

Figure 3. Wintering piping plover, and nesting Wilson's plover and American oystercatcher shorebird beach habitat with increased vulnerability to sea level rise (SLR) by 2030, as compared to historical vulnerability (in 2000) to SLR in the South Atlantic Bight.

2000 vulnerability levels. Sixty-six percent of the habitat with high wintering densities of this species was projected to be at increased vulnerability due to accelerating SLR, as compared to its vulnerability in 2000 (Table 2; Fig. 3).

Fifty percent (266 km) of coastal areas surveyed in the SAB for seabirds (539 km) was projected to be more vulnerable to erosion by 2030, as compared to 2000. Additionally, 42% of all nesting habitat for the brown pelican (228 km) was projected to be at increased vulnerability, whereas only 23% of high-density nesting habitat for this species (45 km) is predicted to be at increased vulnerability to SLR in 2030 (Fig. 4; Table 2), as compared to 2000 vulnerability levels. Two of the seabird species have high nesting densities in areas of greater future coastal erosion vulnerability (gull-billed tern 79% and sandwich tern 70%; Fig. 4), which is greater than the increased vulnerability of all beaches surveyed for these species (50%).

DISCUSSION

Our revised CVI analysis projects a significant change in erosion vulnerability under a moderate increase in SLR rate. By 2030, we project 75% of the SAB coastline will have a very high erosion vulnerability, an increase of 30% from 2000. Natural landward migration of beaches has occurred historically (e.g., overwash-induced barrier island inland migration with SLR; Dean and Dalrymple 2004); however, this migration is currently limited by the location of hard structures, such as buildings and roads, adjacent to beaches (Titus et al. 2009). In areas with sufficient sand and a lack of shoreline stabilization of beachfront hard structures, such as roads and buildings, beaches should be expected to maintain an equilibrium by shifting landward (Passeri et al. 2014), and provide nesting habitat for birds and sea turtles. We suggest the next step in predicting where habitats will be lost to SLR will be to align the CVI rankings from our study with those areas where human-made shoreline stabilization and other infrastructure inhibit natural migration of the landform and habitats alike. These areas will warrant more detailed analyses to more accurately project habitat loss with SLR. Fine-scaled modeling efforts, including an analysis of the ability of specific sections of coast to migrate and the ability of the coastal nesting species to shift their nesting locations is necessary to predict accurately the future of coastal nesting on specific sections of beach. We quantified densities of several coastal species; however, this is not necessarily correlated with habitat quality, which can be quantified using fecundity (van de Pol et al. 2010), or another factor as an index. Beaches between −1 m to 1 m in elevation are predicted to be more likely to respond dynamically to SLR through ecological or morphological responses, via erosion and accretion, than to simply become inundated in place (Lentz et al. 2016).

Developed coastal ecosystems are less likely to respond dynamically to SLR than natural areas (Lentz et al. 2016) and predicting which ecosystems these will be is integral for protecting future nesting habitat for coastal species. On the Caribbean island of Bonaire, for example, sea turtle nesting beaches that backed up to hotel and salt water lakes were least able to accommodate a 0.5-m rise in sea level, compared to those beaches with shrublands or roads behind them (Fish et al. 2005). In the Florida Keys, the likelihood of formation of marsh habitat suitable for the endangered Lower Keys marsh rabbit (*Sylvilagus palustris hefneri*) was inversely related to the proportion of development on the island (Schmidt et al. 2012). Coastal squeeze reduces the amount of beach habitat available for coastal-dependent species with SLR (Mazaris et al. 2009, Reece et al. 2013). Future research should focus on quantifying the relationship between increased vulnerability to erosion from SLR and the effects on species with anthropogenic degradation or loss of habitat. For example, the risk of extinction of the Florida Gulf Coast snowy plover (*Charadrius alexandrinus*) in the next 90 years increases from current levels of 7% to 9% with 1 m of SLR (Aiello-Lammens et al. 2011); this risk is exacerbated with increased anthropogenic effects.

The effect of increased erosion on sea turtle and bird nesting habitat is likely to differ depending on several factors. The response to climatic and ecosystem changes differ by species. Some species are more resilient and better able to adapt than others in response to accelerating SLR and SLR-induced erosion. These differences are often related to life-history characteristics, range, abundance, and diet (Sydeman et al. 2012). Species with high fertility and high dispersal rates may be able to adapt by expanding or contracting their range with current rates of SLR (Benscoter et al. 2013). Given that nesting densities vary by species across beaches with variable SLR-induced erosion vulnerabilities, management of coastal nesting species needs to be based at the species level when considering effects of SLR on future habitats. In those areas where SLR causes increased erosion of breeding colonies and nesting site habitats, we expect reproduction of these coastal nesting species will be reduced.

Beaches with the highest densities of sea turtle nests had similar vulnerabilities to SLR, as compared to all beaches where these species nested, or even all beaches within the SAB study area surveyed for these species (1,250 km). This is

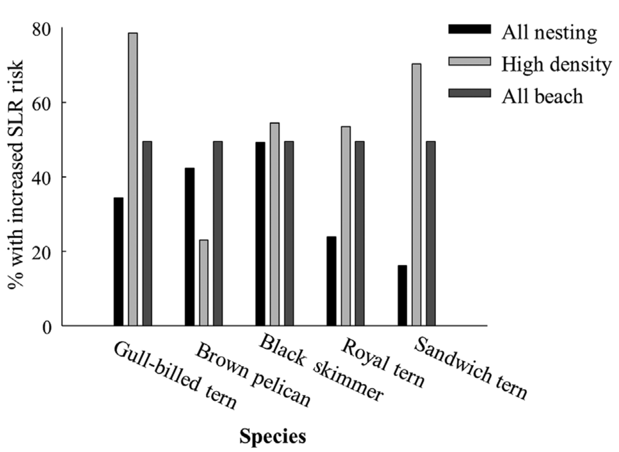

Figure 4. Nesting beach habitat with increased vulnerability to sea level rise (SLR), for gull-billed tern, brown pelican, black skimmer, royal tern, and sandwich tern seabirds by 2030, as compared to historical vulnerability (in 2000) to SLR in the South Atlantic Bight.

likely due to the wide-ranging use across the SAB for their nesting habitat (Figs. 1, SA1–SA3). The focus of this study is the SAB, and it should be recognized that significant sea turtle nesting occurs in Florida outside of the study area. The ability of sea turtle species to shift nesting habitat with changing climatic conditions, or that have a bet-hedging strategy such as that described for the leatherback, which places nests randomly across the width of the beach (Eckert 1987), will affect their resilience to climate change (Hawkes et al. 2007, 2009; Fuentes et al. 2010). It is unclear how nest site selection by sea turtles will change in response to thinner beaches because of predictions of coastal squeeze in some locations (Fish et al. 2008, Mazaris et al. 2009, Reece and Noss 2015).

High-density brown pelican habitat was less vulnerable to SLR than all brown pelican nesting habitats surveyed because by 2030, those beaches where brown pelicans nest in high densities will be less likely to experience severe erosion. For example, this species nests in high densities in high dunes or on artificial dredged-material islands, which are higher in elevation than natural beaches (Bailey 2005, Schweitzer and Abraham 2014). This lessened vulnerability of high-density brown pelican habitat to SLR suggests that SLR may not be as critical a factor to consider when managing for this species, as compared to species that preferentially nest on beaches with much greater SLR-induced erosion, such as the gull-billed tern and sandwich tern. High nesting density beaches for these 2 species were more vulnerable to future SLR than all beaches where these species nest in the study area, pointing to the need for consideration of SLR when managing for these shorebird species in the future. Likewise, high-density winter habitat of the piping plover are more vulnerable to SLR than all beaches where this species was found, which is an important management consideration for a species comprised of threatened and endangered populations.

We used the simplified CVI approach as a useful initial screening of shorelines most likely to experience increased erosion arising from SLR by 2030. We did not take into account the ability of the shoreline to migrate over time, and we acknowledge the CVI does not capture all relevant coastal processes—for example, information to reflect potential changes in alongshore sediment transport, the influence of changes in the rate of relative SLR from 2000 to 2030, annual or decadal sea level trends, the effect of dune elevation (Plant et al. 2016), or other anthropogenic stressors. Further, the CVI geomorphology measure only distinguishes between broad geomorphology categories; for example, although this classification distinguishes between rocky coasts and sandy beaches, it places all barrier islands and sandy beaches in the same category. We used the CVI approach as a qualitative, broad-brush approach, which can be followed up in focal areas of concern with additional quantitative, finer-scale analyses (e.g., spatially detailed shoreline change (Jackson et al. 2012)), including the spatial location of man-made structures adjacent to areas with high vulnerability to SLR, leading to an inability of the landform to shift (e.g., coastal squeeze). Areas we identified to have high nesting densities and high vulnerability warrant further investigation using more focused and fine-scale baseline observations and modeling efforts to more accurately predict future habitat vulnerability to SLR. A more rigorous and detailed analysis and projection of future coastal erosion and topographic change, using process-based models such as Delft3D (Hydro-Morphodynamics 2017) and XBeach (Roelvink et al. 2009) that directly simulate hydrodynamics and sediment transport, would provide a quantitative assessment of future coastal vulnerability. Such investigations should include an analysis of the ability of specific sections of coast to migrate and an analysis of the ability of the coastal nesting species to change their habitat preferences.

Although we evaluated direct effects of SLR and SLR-induced erosion on coastal nesting species, there are also indirect effects of SLR on coastal nesting species. The displacement of human populations from inundated areas have had major consequences for island biodiversity (Wetzel et al. 2012). Additionally, the predaceous invasive ant species, the red imported fire ant (*Solenopsis invicta*) infests sea turtle nests at significantly higher rates on narrower beaches, where sea turtle nests are located closer to the vegetated dunes where the nonnative ants nest (Wetterer et al. 2007). Future studies on the effects of climate change and SLR on biodiversity should incorporate land use changes projected to occur as people adapt in-place or migrate inland in response to increased SLR and altered biotic interactions as species shift inland.

Those species that are predicted to be affected by SLR can be managed in other ways to increase their reproductive success (Hamann et al. 2013), such as enhancing resilience through mitigation of other threats. For example, the sparsely vegetated habitat with little human disturbance that is preferred by American oystercatchers, Wilson's plovers, gull-billed terns, and black skimmers can be managed by erecting predator-excluding fencing, reducing human disturbance, or enhancing breeding sites (burning vegetation, adding dredged material; Rappole 1981, George and Schweitzer 2004). Combining management approaches has also been successful. For example, on a dredged-material island in Georgia, reducing vegetation cover and providing an electric fence to deter mammalian predators increased nesting success of least terns (*Sternula antillarum*; Spear et al. 2007), as compared to natural beaches without these management actions. We recommend an adaptive management approach that considers climate change as a source of environmental variation that may evolve over time, and that involves stakeholder input to define the goals and the actions to be considered at each decision point (Nichols et al. 2011).

MANAGEMENT IMPLICATIONS

The maps of habitat vulnerability under different projections of SLR will assist conservation managers to identify areas that need more in-depth study of future coastal evolution and inundation and of the potential for habitat restoration strategies. Coastal vulnerability to SLR, the likelihood of a dynamic response to SLR, nesting and occupancy densities by coastal species, species vulnerability to flooding, and land use type will influence management approaches to protect these coastal nesting species. We suggest tailoring restoration strategies for the specific location and focal species. For example, undeveloped areas that currently have high nesting densities of coastal species could deploy setback regulations or a minimum distance from mean high tide line for future development to allow for natural inland migration of these beaches with SLR. Additionally, protected areas that harbor high densities of coastal species with an inability to migrate inland could emphasize beach restoration of low-lying beaches or surrounding wetlands to counteract sand loss due to rising sea levels or storm erosion. Our approach is an important initial step in managing coastal species for the changing environmental conditions associated with climate change and SLR. It represents a broad-scale attempt at predicting the level of susceptibility of our study species to erosion and inundation from SLR.

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