Localized Scenarios and Latitudinal Patterns of Vertical and Lateral Resilience of Tidal Marshes to Sea-Level Rise in the Contiguous United States

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Key points:

- We quantified capacity of tidal wetlands to accrete relative to sea-level rise or migrate inland, using localized sea-level rise scenarios.
- More northerly wetlands have more capacity for relative vertical accretion, and more southerly wetlands have more lateral migration space.
- Between 43% and 48% of current coastal wetland area is vulnerable to vertical accretion deficit and lack of capacity for inland migration.
Abstract

Coastal wetlands have two dimensions of vulnerability to sea-level rise (SLR), a vertical one, in cases where SLR outpaces their capacity to vertically accrete, and a lateral one, in cases where they are restricted from migrating inland by topography and land use. We conducted a meta-analysis of accretion rates, standardized our analysis by using only $^{137}$Cs based estimates, and used model intercomparison to generate a vertical resilience index, a function of local SLR, tidal range, and tidal elevation category for the tidal wetlands of the contiguous United States. We paired the vertical resilience index with a lateral resilience index made up of elevation, water level, and land cover maps, then projected them both into the future using localized SLR scenarios. At the regional scale, the vertical resilience index predicts changes from marsh aggradation to submergence for the coastal US Mid-Atlantic, Southeast, and portions of the Northeast by 2100. At the sub-regional scale, there is a geographic tradeoff between vertical and lateral resilience with more northerly wetlands vulnerable to the lack of suitable proportional area to migrate into, and more southerly wetlands vulnerable to accretion deficit. We estimate between 43% to 48% of existing contiguous US wetland area, almost entirely located in watersheds along the Gulf of Mexico and Mid-Atlantic coasts, is subject to both vertical and lateral limitations. These vertical and lateral resilience indices could help direct future research, planning and mitigation efforts at a national scale, as well as supplement more processed informed approaches by local planners and practitioners.
Plain Language Summary: Wetlands that occur within the tidal zone are potentially vulnerable to sea-level rise but have some capacity to adapt by forming new soil and elevation or moving inland. Their ability to gain elevation can be limited if sea-level rise is too rapid, and their ability to migrate can be limited by a lack of available adjacent land. Our analysis combined a synthesis of soil formation rates from previous studies with sea-level rise rates from tide gauges measured over the same time periods, as well as water level, elevation and landcover maps in order to compare vertical and lateral resilience potential across the Contiguous United States. We found north to south tradeoffs in the strengths and availability of the two resilience pathways that tracked patterns in current and projected sea-level rise and the range of tides. More northerly wetlands had more capacity to gain elevation, and more southerly wetlands had more capacity to move inland. We estimate that a high proportion of current wetland areas--43% to 48% depending on the sea-level rise scenario used--are vulnerable to sea-level rise because they occur in areas where projected sea-level rise is high and where upland areas available for migration are limited.
1. Introduction

Coastal marshes have two major paths to resilience to sea-level rise (SLR). They can gain elevation because of positive feedback between inundation and sediment accretion (Kirwan et al., 2016; Kirwan & Megonigal, 2013; Morris et al., 2002), and they can migrate inland (Anisfeld et al., 2017; Lentz et al., 2016). However, both paths to marsh resilience are limited by geomorphic, topographic and land-use factors that vary regionally and locally (Kirwan & Guntenspergen, 2010; Torio & Chmura, 2013; Weston, 2014) among the vast and diverse coastal marshes of the contiguous United States (CONUS) (Anisfeld et al., 1999; Callaway et al., 2012; DeLaune et al., 1992; Thom, 1992).

During the 20th century, global sea-levels have risen 13.8 ± 1.5 cm with anthropogenic climate change explaining 51% of the rise (Kopp et al., 2016). SLR is accelerating (Church & White, 2006; Watson et al., 2015) and is projected to further accelerate by 2100 (Rahmstorf, 2007; Slangen et al., 2014). Furthermore, local geological conditions interact with eustatic SLR to produce differing rates of relative sea-level rise (RSLR) (Kopp et al., 2014). Areas that are subsiding, such as the Louisiana Coast (Ingebritsen & Galloway, 2014; Jones et al., 2016), have higher rates of RSLR than areas that are tectonically uplifting or isostatically rebounding, such as the Pacific Northwest (Komar et al., 2011). Regional SLR projections can also vary locally because of the effect of oceanic forces (Carson et al., 2016; Slangen et al., 2014). For example, the US Mid-Atlantic coast has been experiencing RSLR approximately 3 to 4 times that of global averages because of a weakening Atlantic Meridional Overturning Circulation (AMOC) (Sallenger et al., 2012).

Marshes can gain elevation at rates keeping pace with modest RSLR through increased plant productivity (Morris et al., 2002; Mudd et al., 2009; Nyman et al., 2006) and increased inorganic sediment supply and trapping (Morris et al., 2002, 2016; Mudd et al., 2009; Swanson et al., 2014). However for cases in which RSLR is too rapid (Cronin et al., 2007) or accretion is
physically limited (Weston et al., 2014), marshes can collapse, converting to open water or unvegetated tidal flats (Day et al., 2011; Kirwan & Guntenspergen, 2010). Inland migration onto previously un-inundated land allows for rapid adjustment to RSLR but may be limited by current land use in addition to other geomorphic factors such as topography and connectivity. Marshes in many developed coastal areas that could otherwise maintain their area by advancing inland, may be halted by development including hardened shorelines, tidal restrictions, structures and impermeable surfaces (Linhoss et al., 2015; Torio & Chmura, 2013).

A number of studies have conducted large-scale comparisons of observed sediment accretion rates or inland migration potential of coastal marshes to projected SLR. For example, Kirwan et al. (2016) compiled a meta-analysis of observed accretion rates data for CONUS and observed that Gulf Coast marshes on the higher end of the RSLR spectrum were keeping pace with historical levels of RSLR (~9 mm yr⁻¹). Their results indicate that frequently inundated marshes are responding dynamically to SLR and should be considered separately from higher elevation, less frequently inundated marshes. Thorne et al. (2018) performed data-intensive site-specific parameterization to forecast tidal marsh vertical resilience to sea-level along the Pacific Coast using the WARMER model paired with a migration. They concluded that Pacific Coast marshes were vulnerable RSLR under higher scenarios but noted a general north to south trend in vertical vulnerability with northern latitudes of the Pacific Northwest less vulnerable to RSLR than marshes at lower latitudes in Southern California. Enwright et al. (2016) analyzed the capacity for wetlands to migrate inland by quantifying the availability of adjacent land using digital elevation maps, land use, and population. However, this effort was limited to the Gulf Coast of the U.S. and was not extended to the Pacific and Atlantic Coasts. Neither Thorne et al. (2018) nor Enwright et al. (2016) used localized SLR scenarios to create their lateral resilience indices; Enwright et al. (2016) tested increases of 0.5, 1.2, and 2 m uniformly across the Gulf Coast while Thorne et al. (2018) used a uniform increase of 1.66 for California and 1.42 m for Oregon and Washington. Such works have also relied on a variety of
accretion estimation techniques (Chmura et al., 2003; Jarvis, 2010; Ouyang & Lee, 2013) which can provide different vertical accretion estimates dependent upon the technique used (Nolte et al., 2013; Turner et al., 2006). For example, Turner et al. (2006) found that accretion estimates based on $^{210}$Pb dated core sections were about 25% less than estimates from the more core sections dated by $^{137}$Cs.

The goal of this study is to map patterns of vertical and lateral resilience to RSLR around the contiguous U.S, including closely adjacent portions of Canada, using localized RSLR scenarios, standardized methodology based on $^{137}$Cs dating for estimates of vertical resilience, and indexed assessment of lateral migration potential. This paper greatly expands previous meta-analyses’ datasets by using a database with more studies and a wider geographic expanse (https://figshare.com/s/96f4554ff62aea7005b5; Supplemental Tab. 1). We improve upon previous estimates of salt marsh vertical resilience by additionally controlling for method of estimation of sediment accretion rates, computing RSLR over the same time frames as accretion rate records, and selecting ecosystem parameters to include in a vertical resilience index based on linear modeling. We apply the results of accretion fit using linear models and lateral resilience metrics by using localized SLR scenarios to quantify and compare CONUS-scale trends. By extrapolating our results spatially, we compare patterns in vertical and lateral resilience looking at the proportional land available within the tidal zone given localized RSLR scenarios.

2. Materials and Methods

In this paper we built then analyzed indices of tidal wetland vertical and lateral resilience to RSLR for CONUS. Then we examined the implication of geographic patterns in environmental drivers by calculating those indices using localized RSLR scenarios. First we designated an area of interest for the study. Second, we designed the vertical resilience index by using a linear modeling framework with vertical accretion rate as the dependent variable. We
built the calibration dataset for vertical accretion by performing a meta-analysis of dated soil cores, focusing in the end on \(^{137}\text{Cs}\)-dated cores. We build the environmental covariate dataset by querying databases and parsing the original sources for additional information. Third, for lateral resilience index we performed a geographic information system (GIS) analysis summarizing the relative area and types of land available for tidal wetland transgression by combining publically available elevation, water-level, and land cover data. Fourth, we compare both indices run into 2100 using RSLR scenarios by Kopp et al. (2014).

2.1. Defining an Area of Interest

To define our area of interest we mapped all CONUS coastal watersheds containing tidal wetlands based on the National Wetlands Inventory (NWI) (U.S. Fish and Wildlife Service, 2014). We downloaded NWI data for each oceanic coastal state including: Washington, Oregon, California, Texas, Louisiana, Mississippi, Alabama, Florida, Georgia, South Carolina, North Carolina, Virginia, Maryland, Delaware, New Jersey, Pennsylvania, New York, Rhode Island, Connecticut, Massachusetts, New Hampshire, and Maine. We included any NWI polygons with a tidal modifiers ‘E’ or ‘M’, indicating estuarine or marine systems, or with both a ‘P’, indicating palustrine system, and ‘R’, ‘S’, ‘T’, or ‘V’, which indicate palustrine tidal systems. For watershed units we used intermediate watershed boundaries (Hydrologic Unit Code Level 8 [HUC8]; United states Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS), the United States Geological Survey (USGS), and the Environmental Protection Agency (EPA), 2015). We selected HUC8 units that overlapped NWI tidal wetlands, and manually removed watershed units that were selected because of obvious outliers caused by mistakes in NWI coding.

2.2. Vertical Resilience Index
2.2.1. Vertical Resilience Index Formulation

We calculated vertical resilience (VR) as modeled net accretion (Eq. 1), RSLR subtracted from modeled accretion. Values greater than 0 indicate aggradation, while values less than 0 indicate submergence.

\[ VR = \text{Modeled Accretion} - \text{RSLR} \quad \text{Eq 1.} \]

We used a series of linear models to determine which environmental variables to include in the vertical resilience index. We also investigated the effect of time itself on accretion since longer periods of time could potentially decrease observed accretion because of sediment compaction and belowground carbon loss. RSLR, greater diurnal tidal range (GT), watershed level flow weighted average suspended sediment concentration (FWA-SSC), inundation class, and time were all treated as independent variables and accretion rate as the dependent variable. RSLR, GT, FWA-SSC, and time were all treated as additive, with inundation status interacting with them. We used linear modeling functionality in base R (R Development Core Team, 2010). In a series of iterative steps we eliminated the least significant parameter until all parameters were significant (p<0.05). We used Akaike’s Information Criterion for small sample sizes (AICc) to intercompare models, quantifying tradeoffs between performance and parsimony (Barton, 2015). Since frequently inundated marshes respond to SLR more dynamically than infrequently inundated marshes (Kirwan et al., 2016), we used the linear model parameters for frequently inundated marshes to create a vertical resilience index.

2.2.2. Vertical Resilience Calibration Dataset

Our analysis uses a dataset of 1,004 accretion rates from 116 different studies in emergent tidal marshes of North America (Anisfeld et al., 1999; Armentano & Woodwell, 1975; Baumann et al., 1984; Baustian et al., 2012; Beckett, 2012; Bergquist, 1978; Brevik & Homburg, 1984; Brevik & Homburg,
2004; Bricker-Urso et al., 1989; Bryant & Chabreck, 1998; Cahoon et al., 1995, 1996; Cahoon & Turner, 1989; J. C. Callaway et al., 1997; Callaway, 2008, 2010; Callaway, Borgnis, Eugene Turner, et al., 2012; Callaway, Borgnis, Turner, et al., 2012; Carey, 1997; Chague-Gofp et al., 2001; Childers et al., 1993; Chmura et al., 2001; Chmura & Hung, 2004; T. M. Church et al., 1986; Clague et al., 1994; Clarke et al., 2014; Clark & Patterson, 1984; Cochran et al., 1998; Cole & Liu, 1994; Cole & Wahl, 2000; Connor et al., 2001; Cornu & Sadro, 2002; Craft et al., 1993; Craft, 2007; DeLaune et al., 1978; DeLaune et al., 1981, 1983; DeLaune et al., 1986; DeLaune et al., 1987, 1989; Delaune et al., 1992; DeLaune et al., 2003; DeLaune & Pezeshki, 2003; Donnelly et al., 2004; Elington, 2012; Elsey-Quirk et al., 2011; Feagin & Yeager, 2008; Flessa et al., 1977; Ford, 1997; Gallagher, 1996; Gehrels et al., 2005; Goldberg et al., 1979; Goman & Wells, 2000; Greiner & Hershner, 1998; Hales, 2000; Harrison & Bloom, 1977; Hartig et al., 2002; Hatton et al., 1983; Johnson et al., 2007; Kastler & Wiberg, 1996; Kearney et al., 1994; Kearney & Ward, 1986; Kemp et al., 2012; Kim et al., 2004; M. L. Kirwan et al., 2011; Knaus & van Gent, 1989; Kolker et al., 2009; Kostaschuk et al., 2007; Kraft et al., 1992; Lagomasino et al., 2013; Lane et al., 2006; Letzsch & Frey, 1980; Mattheus et al., 2010; B. A. McKee et al., 1994; K. L. McKee & Cherry, 2009; Milan et al., 1995; Mudie & Byrne, 1980; Nikitina et al., 2000; Nydick et al., 1995; J. A. Nyman et al., 1993; Nyman et al., 2006; Orson et al., 1998; Orson et al., 1987; Orson & Howes, 1992; Palacios-Fest et al., 2006; Patrick & DeLaune, 1990; Piazza et al., 2011; Redfield, 1972; Richard, 1978; Roman et al., 1997; Rybczyk & Cahoon, 2002; Sharma et al., 1987; Smith et al., 2013; Smith, 2009; Stevenson et al., 1985; Stuart, 2010; Sturdevant et al., 2002; Thom, 1992; Turner et al., 2002; Turner et al., 2006; Unger, 2014; Vogel et al., 1996; Wallace et al., 2005; Ward et al., 1998, 2008; Watson, 2004; Watson & Byrne, 2013; Weis et al., 2001; Weis et al., 2005; Weston et al., 2014; White et al., 2002; White & Calnan, 1990; Williams, 2003; Williams & Roberts, 1989; Williams & Hamilton, 1995; Wilson & Allison, 2008; Zeppie, 1977) (Supplemental Table 1: https://figshare.com/s/96f4554ff62aea7005b5). The initial meta-analysis included data from

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artificial markers, $^{137}$Cs-dated cores, $^{210}$Pb-dated cores, historical sediment horizons tied to agricultural or pollution events, and radiocarbon ($^{14}$C) dates. An analysis of variance (ANOVA) indicated that accretion rates varied significantly across methods (p<0.001).

Because there were significant differences between methods, and the 1963 to core-collection-date time-scale overlaps with wide scale water-level gauging, we screened the sediment accretion estimates to only include those based on $^{137}$Cs dated sediment cores. In addition, such $^{137}$Cs accretion spans a long enough timeframe to provide average rates for decadal periods rather than short records which may be highly influenced by an unusual annual event. We excluded data from mangroves and other forested wetlands, submerged aquatic vegetation beds, and bay bottoms. If a source included freshwater wetland data as well as estuarine data, we included freshwater data points, provided that they were identified as ‘coastal’ or ‘tidal’ according to the source or NWI maps (Section 2.1.).

We investigated the potential effect of differing regional suspended sediment concentrations (SSC), hypothesizing that regional SSC should correlate positively with accretion. We replicated the FWA-SSC calculations outlined by Weston (2014) and extended the analysis to all USGS gauges within watersheds that contained tidal wetlands. In order to calculate watershed average FWA-SSC (P80154), we downloaded SSC from the USGS web portal (USGS, 2016), specifying gauges with more than 50 discrete records in all states that have coastal wetlands. For each gauge we also downloaded average daily discharge (P00060) data from the same tide gauge over the entire gauge period. We used the ‘R’ Package ‘hydrostats’ (Bond, 2015) to calculate base flow and base flow index for the entirety of the daily discharge record and calculated FWA-SSC as the average of all SSC taken on days when baseflow was greater than 20, meaning baseflow made up greater than 20% of the discharge (Weston, 2014) (Supplemental Table 2; https://figshare.com/s/96f4554ff62aea7005b5). Despite slight differences in methodology between Weston’s 2014 analysis and our own, the two analyses correlate nearly 1:1 ($R^2 = 0.97$) for those records which they have in common.
Because of this correlation, we integrated FWA-SSC values for gauges included in Weston’s 2014 analysis but not in ours. We calculated the average FWA-SSC for all watersheds that had 1 or more gauges falling within their borders using Spatial Join in ArcGIS (Esri Inc., 2017).

For each $^{137}$Cs dated core in the accretion dataset, we first determined the nearest NOAA tide gauge (Permanent Service for Mean Sea Level, 2016) with an overlapping time frame as the core, with a five-year tolerance, using MapTools in R (Bivand & Lewin-Koh, 2016). We then downloaded monthly mean water level data from NOAA servers (https://tidesandcurrents.noaa.gov/) using the package downloader (Chang, 2015). We calculated RSLR as the slope of a linear model with time as the independent variable and tidal elevation as the dependent variable using linear model in R (R Development Core Team, 2010). For all cores, we also assigned the GT range from the nearest NOAA tide gauge (NOAA, 2017) with a reported GT range and complete data over the most recent datum epoch using a spatial join in ArcGIS Pro (Esri Inc., 2017; Supplemental Table 3; https://figshare.com/s/96f4554ff62aea7005b5).

As Kirwan et al. (2016) did, we attempted to control for differential responses of accretion to RSLR between more and less frequently inundated marshes. However we encountered complexity and lack of consistency in terminology in our expanded literature survey. Therefore we wrote a decision tree attempting to classify as much of the meta-analysis dataset as possible into two categories, higher energy, more mineral dominated systems, as frequently inundated marshes, and lower energy, less mineral dominated systems, as infrequently inundated marshes. According to the decision tree we classified coring locations based on the following criteria, in order: explicit descriptions as relatively frequently or infrequently inundated; if not that then further parsing of the relative tidal elevation and plant community descriptions high, mid, and low marsh; if not that then gleaning additional context from coring location descriptors; if not that then the presence of indicator plant species described as present and dominant at the coring locations.
First, if the study described sites as ‘frequently’ or ‘infrequently inundated’ we utilized those definitions. Second, an initial ANOVA showed that high and mid marshes had significantly lower accretion rates than low marshes, but not from each other (p < 0.001). So, for studies which compared ‘high’, ‘mid’ and ‘low’ elevation marshes we categorized both high and mid marsh elevations as ‘infrequently inundated’, and low elevations as ‘frequently inundated’. Third, for other studies that compared marsh position or inundation regime in other ways we classified marshes that were described as ‘shoreward’ (Lagomasino et al. 2013), ‘levee’ (Hatton et al., 1983; Craft, 2007), or ‘streamside’ (Delaune et al. 1978; Craft et al., 1993) as ‘frequently inundated’ and opposed to marshes described as ‘inland’ (Delaune et al. 1978), ‘back marsh’ (Craft et al., 1993; Sturdevant et al., 2003; Lagomasino et al. 2013), or ‘plains’ (Craft, 2007) as ‘infrequently inundated’. For studies that compared natural to impounded sites we always classified impounded marshes as ‘infrequently inundated’ (Bryant et al., 1998; Sturdevant et al., 2003). We classified ‘natural’ sites according to additional vegetation descriptors.

Fourth and finally, for studies that reported vegetation we classified marshes that contained *Spartina alterniflora* long-form (Valiela et al. 1978; Bertness 1988), *Spartina alterniflora* with no growth type specified (Bertness, 1991), and *Schenolplectus* (previously known as Scirpus) species (Kirwan and Guntenspergen, 2015; Langley et al., 2013; Holmquist et al., In Press) as frequently inundated, and those that contained *Spartina alterniflora* short-form (Valiela et al. 1978; Bertness 1988), *Spartina patens* (Bertness, 1991), *Distichlis spicata* (Langley et al., 2013; Holmquist et al., In Press), *Salicornia* spp. (Zedler et al., 1999; Janousek et al., 2016), and *Juncus roemerianus* (Watson et al., 2015) as infrequently inundated. We deviated from this rubric only in the case that the original source (Elsey-Quirk et al. 2011) described *Spartina alterniflora*-short form as representative low marsh elevation zone. We omitted cases from further analysis in which we could not categorize according to this scheme including, freshwater tidal areas and marshes dominated by *Phragmites australis* (Holmquist et al., In Press).
Inundation classifications and the criteria we used to determine them are recorded in Supplemental Table 3 (https://figshare.com/s/96f4554ff62aea7005b5). We recognize that there is a degree of subjectivity which is unavoidable in this schema, and further discuss implications in our discussion section under section 4.2., Limitations and Caveats. Because the assignment of inundation regimes was based partially on context gleaned from original papers which had a subjective element, we tested the effect of eliminating the inundation term.

2.3. Lateral Resilience Index

2.3.1. Lateral Resilience Index Formulation

We calculated lateral resilience (LR) index as the ratio of the land cover amenable to marsh migration in the SLR zone, to modern day coastal wetland cover below modern day mean higher high water spring (MHHWS).

\[
LR = \frac{\text{Amenable Migration Area}}{\text{Tidal Wetland Area}} \quad \text{Eq. 2}
\]

Values 1 or higher indicate amenable areas are at or above a one to one replacement level given lateral wetland migration. Values between zero and one indicate amenable area is less than replacement level for given lateral wetland migration.

We define the SLR zone as land below MHHWS at 2100 given a RSLR scenario. We assumed that amenable landcover types in the SLR zone included currently non-tidal wetlands, uplands, and bare land. We excluded developed land, cultivated land and open water. For tidal wetland area, we included all brackish and saline wetlands (referred to throughout as estuarine wetlands) and any freshwater (referred to throughout as palustrine wetlands) below the modern day MHHWS elevation.

2.3.2. Lateral Resilience Index Input Datasets
To calculate LR index, we used three types of datasets, one for land cover, one for land elevation, and one for MHHWS elevation. For landcover, we used the 1996-2010 Coastal Change Analysis Program (C-CAP) (NOAA, 2013) to map major coastal landcover types to parse freshwater (a.k.a. palustrine) wetlands. We downloaded and mosaiced 1996 to 2010 C-CAP data for all states with coastal wetlands. We simplified the 2010 C-CAP’s 22 land classes into tidal wetlands, non-tidal wetlands, water, developed, cultivated, upland, and bare land (Tab. 1).

To compare broad regional trends in future modeled accretion to the potential future inland marsh migration we utilized publicly available digital elevation models (DEMs), spatially extrapolated high spring tide levels from a previous study (Holmquist, Windham-Myers, Bernal, et al., 2018), and localized SLR scenarios from Kopp et al. (2014). For elevation, we downloaded DEMs associated with the NOAA SLR viewer as well as Louisiana, Maryland State, and San Joaquin Delta data sources. The DEMs are all the same as those used to map coastal lands by Holmquist et al. (2018) and are listed in supplemental information therein. All DEMs were converted to meters relative to North American Vertical Datum of 1988 (NAVD88), if not already in those units, and had variable pixel size ranging from 1 to 10 m. DEMs were resampled to 30 x 30 m resolution using a ‘maximum area’ rasterization in ArcGIS in order to spatially match the resolution of C-CAP.

DEMs went through two transformations, from meters relative to current day NAVD88 to meters relative to MHHWS. We used a NAVD88 to MHHWS conversion layer used to determine the extent of ‘coastal lands’ for a U.S.-wide coastal wetland carbon inventory (Holmquist, Windham-Myers, Bernal, et al., 2018), and assumed an average 17.3 cm of positive elevation bias for wetland surfaces, introduced by dense vegetation interfering with Light Detection and Ranging penetration (Holmquist, Windham-Myers, Bernal, et al., 2018). The MHHWS line was also used to classify Palustrine Wetlands into finer tidal and non-tidal classifications using the elevation of MHHWS as a cutoff point, Palustrine wetlands below are classified as tidal, above
are classified as non-tidal. We shifted the datum a second time, converting meters relative to modern MHHWS to meters relative to MHHWS 100 years in the future.

\[ \text{MHHWS}_{t=100} = \text{NAVD88}_{t=0} + \text{MHHWS}_{t=0} + \text{SLR}_{0:100} \]  
Eq. 3

2.3. Sea-Level Rise Scenarios

We applied both the vertical and lateral resilience metrics by extrapolating tide gauge data, or predictions using GIS summarized by both state boundaries and intermediate watershed units. We calculated vertical resilience index at the location of tide gauges (Supplemental Tab. 4; https://figshare.com/s/96f4554ff62aea7005b5), then extrapolated them spatially using GIS. Similar to our comparison of accretion to RSLR for the core dataset, we queried the NOAA Permanent Service for Mean Sea Level dataset, and calculated RSLR over a deliberate and consistent time-frame. The linear modeling exercise resulted in RSLR, GT and inundation regime as being significantly correlated with accretion. We calculated RSLR from 1996-2010, the same time frame as the national-scale coastal land cover change data used in the LR index. We selected gauges within a 1 year tolerance of that time frame, and had at least 66% data completeness. This resulted in 75 different tide gauges to model net accretion in recent decades. We also modeled accretion rate using likely 2000-2100 RSLR data for the same gauges (Kopp et al., 2014). We used median estimates from probabilistic local SLR models based on low, medium, and high impact climate change scenarios known as representative concentration pathways (RCPs). We used RCPs 2.6, RCP 4.5 and RCP 8.5, which assumed 2.6, 4.5, and 8.5 watts m\(^{-2}\) of radiative forcing by 2100 (Meinhausen et al., 2011). We assumed an average linear increase in sea-level.

In addition to using localized RSLR scenarios to estimate vertical resilience ratios, we also calculated LR index by extrapolating mean total RSLR predicted under RCP’s 2.6, 4.5, and 8.5, raising the current MHHWS boundary by those amounts, and summarizing the land cover
types in the SLR zone (Eq. 3). We define the SLR zone as the land between the current MHHWS boundary and the future MHHWS boundary given a RSLR scenario.

We extrapolated prediction surfaces from gauges using Empirical Bayesian Kriging in ArcGIS Pro (Esri Inc., 2017). Empirical Bayesian Kriging extrapolates point data to a surface using by modeling spatial autocorrelation between points, and accounts for uncertainty in the semivariogram used to model spatial autocorrelation (Gribov and Krivoruchko, 2020). Settings that are needed to replicate the calculation are unmodified from generic inputs and include: an output cell size of 300 m, no data transformation, a power semivariogram model, 100 maximum local point in each model, a local model area overall factor of 1, and 100 simulated semivariograms. For the search neighborhood we used a standard circular neighborhood, 15 max neighbors and 10 min, 1 sector, and angle of 0 and a radius of 15.

We report VR, LR and wetland area for intermediate-scale watershed units (Supplemental Tab. 5; https://figshare.com/s/96f4554ff62aeab7d05b5) and aggregate metrics at three scales CONUS-wide, by political/geographic zones, and intermediate watershed units. The six major political/geographic zones in the US included: the Northwest (Washington and Oregon), the Southwest (California), South Central (Texas, Louisiana, Mississippi, Alabama, and the Florida Gulf Coast), Southeast (the Florida Atlantic Coast, Georgia, South Carolina, and North Carolina), the mid-Atlantic (Virginia, District of Columbia, Maryland, and Delaware), and the Northeast (New Jersey, Pennsylvania, New York, Connecticut, New Hampshire, Massachusetts, Rhode Island, and Maine). Since our meta-analysis contained no data on mangroves we excluded six HUC8-level watersheds in which Estuarine Forested Wetlands (mangroves) were the dominant estuarine land cover type in 2010.

We calculated regional and national summaries from watershed level statistics for both vertical and lateral resilience in two ways, by wetland area, and by wetland percentage. For each watershed, we classified vertical resilience as a binary; wetlands and watersheds were classified as either vertically vulnerable if VR was greater than or equal to 0. For LR we
summarized wetland area and watersheds using a weighted sum. If the LR was less than 1, then the total wetland area was multiplied by LR to estimate the area of resilient wetlands, and (1-LR) to estimate the area of vulnerable wetlands. If LR index was ≥1, then all the wetland area in the watershed was classified as laterally resilient.

3. Results

3.1. Linear Model Results

We fit five linear models to two subsets of data. First, we tested models 1 - 3 (Tab. 2) using a dataset which included all data which could be categorized by inundation frequency and had watershed-level FWA-SSC data (n=94). Neither FWA-SSC (Tab. 2; model 1) nor time (Tab. 2; model 2) were significantly correlated with accretion rate. In model 3, which had only inundation class, RSLR, GT, and interactive effects between inundation class, both RSLR and GT were each significant (p<0.01). Model 3 had a lower AICc score than Models 1 or 2 (Tab. 2) and had an overall p-value of <0.001 and an R² of 0.54 (Tab. 2). For this model RSLR (in mm yr⁻¹) and GT (in meters) were both positively correlated with accretion for frequently inundated marshes, and RSLR was positively correlated with accretion in infrequently inundated marshes. Accretion was significantly higher for infrequently inundated marshes.

Because model 1 indicated that FWA-SSC did not have a significant impact on accretion, we created models 4 and 5 using a larger subset of data in which cores could be categorized as frequently or infrequently inundated (n = 269). Once again time was a non-significant input variable (model 4) and removing it reduced the AICc score (model 5; Tab. 2). Overall model 5 had an R² of 0.33 and a p-value <0.001 (Tab. 2) and indicated that inundation class, RSLR, GT, and interactive effects between inundation class contributed to variability in vertical accretion rates (p<0.001).

For the subset of data without FWA-SSC and with inundation regimes (n = 269), eliminating the inundation term still resulted in an overall significant model (p < 2.2e-16),
although the model explained less of the variance in the data (R² value decreased from 0.33 to 0.23), and resulted in a less parsimonious model (AICc increased from 1204 to 1222; Tab. 2). When the inundation term was removed, GT was no longer a significant parameter (p = 0.92). For the larger total dataset (n = 478), the same model form was significant (p < 2.2e-16) and R² increased from 0.23 to 0.30; GT was still not a significant parameter without the inundation term given the larger dataset.

3.2. Vertical Resilience Index Results

We predicted net accretion for frequently inundated marshes as a metric with which to broadly compare resilience to RSLR across major geographic gradients. VR is described by equation 4, in which RSLR is in mm yr⁻¹ and GT is in m.

\[ VR = (0.61 \text{ RSLR} + 1.4 \text{ GT} + 0.39) - \text{RSLR} \quad \text{Eq. 4} \]

At 75 tide gauges, predicted net accretion varied regionally between 1996 and 2010 based on GT and RSLR (Fig. 3). The most vulnerable marshes would be those associated with the Grand Isle, Louisiana gauges where RSLR was 7.4 mm yr⁻¹ and GT was 0.323 m (Supplemental Tab. 4; https://figshare.com/s/96f4554ff62aea7005b5). Nine out ten gauges in Louisiana and Texas had modeled negative net accretion, meaning that they are submerging at present. At one gauge in the mid-Atlantic (Solomon’s Island, Maryland) and three in Northeast (Montauk, New York; Woods Hole, Massachusetts, Nantucket Island, Massachusetts) predicated net accretion was also negative. In the US Northwest and Southwest all net accretion predictions were greater than 2 mm yr⁻¹.

When median local 2100 projections of RSLR based on RCP 2.6 (Kopp et al., 2014) were applied in place of 1996-2010 values, and GT was assumed to be static, many locations switched from aggrading to submerging (Fig. 3). Nine out of ten gauges from the mid-Atlantic,
and six of the more southern Northeast sites switch from aggrading to submerging under RCP 2.6. Under the more extreme RCP 8.5 values, portions of the Southwest and the Southeast, as well as more of the Northeast, switch from aggrading to submerging (Fig. 4A, 5A, 6A). In the northernmost portion of the Atlantic Coast, as well as much of the Northwest, macrotidal conditions and projected RSLR is low, and the Eq. 3 predicts net accretion will remain positive.

In our analysis we see that threshold beyond which marshes fail to accrete relative to RSLR is regionally variable, depending on GT, and in many cases is already crossed, or will likely be crossed by 2100. If we set VR to zero, and rearrange equation 4, we get Eq. 5, in which we solve for RSLR for a specific GT in meters. This is a RSLR threshold for when net accretion would from positive to negative.

\[ \text{RSLR} = 3.59 \text{GT} + 1 \quad \text{Eq. 5} \]

3.3. Lateral Resilience Index Results

In addition to regional vulnerability due to RSLR and GT gradients there is also substantial regional variation in the area available for marsh migration. Within the 198 watersheds for which there was adequate DEM data, ‘palustrine’ wetlands were the dominant low-elevation land cover class, dominating 100 watersheds in total (Fig. 3). ‘Palustrine’ wetlands are a prominent feature along the CONUS South Central coast, as well as the Southeast and the mid-Atlantic (Fig. 4B, 4C, 5B, 5C, 6B, 6C). Marsh migration potential is highest along the coast of North Carolina (Fig. 5B) because low elevation ‘palustrine’ areas are extensive in proportion to current tidal wetland areas.

Watersheds with relatively low migration potential values can broadly be classified into three different conditions. First, they can represent areas where the SLR zone is dominated by ‘cultivated’ or ‘developed’ lands. Southern California and much of the North Eastern Seaboard between New Jersey and New Hampshire are examples of the 'developed' dominated classes
(Fig. 4B, 4C, 5B, 5C, 6B, 6C). Second, they can represent watersheds that do have adjacent undeveloped inland migration space, but that space is limited by the relief of adjacent slopes, such as the North Eastern Coast of Maine (Fig. 6B). Third, they can represent watersheds in which inland migration space exists but in relatively low proportion to the area of tidal wetlands. This occurs in Louisiana where tidal wetland area within some watersheds is very high (Fig. 5B).

Regions varied generally from north to south in the proportion of land that was available to accommodate lateral wetland migration. The northwest (Oregon and Washington) had the least, 21% under RCP 4.5. This was followed by the Southwest (CA) with 21% and Northeast (from Maine in the North to New Jersey in the South) with 23% (Tab. 3). The South Central (Louisiana and Texas), Southeast (the rest of the Gulf Coast and Atlantic Florida south north to North Carolina), and the mid-Atlantic all had much higher proportions of land available for wetland migration. The mid-Atlantic zone had 43%, the Southeast had 44%, and the South Central zone had the most with 50%, under RCP 4.5.

3.4. Paired Vertical and Lateral Resilience Metrics at Watershed Scale

At a watershed scale the trends in vertical and lateral resilience indices were negatively correlated (Fig. 7), meaning that watersheds that were more likely to be vulnerable to losing elevation to SLR, were also more likely to have adjacent areas available to accommodate marsh migration. Likewise marshes that are more likely to have inland migration space available are less likely to be able to vertically accrete at pace with SLR.

For individual watersheds both indices follow latitudinal and regional trends. The Pacific Coast is generally below a lateral replacement level of one to one, and above a mean net accretion rate of zero. In the Northwest wetlands are the most resilient vertically, and have the least available migration space. The coast of California is similarly restricted in terms of lateral migration, but also loses VR from north to south (Fig. 7). The California coast is near a mean net accretion rate of 0, meaning vertically vulnerable in RCP 8.5, with mean accretion rates in
some watersheds lower than the rate of RSLR. The Northeast similarly is below replacement
level laterally, and vertically resilient in the north, which declines to the south. For the Southeast,
mid-Atlantic and south central zones, vertical resilience declines southward, but LR increases.
Regionally the highest proportions of doubly vulnerable wetlands are in the Mid-Atlantic under
RCP 2.6 (59%), and in the Northeast (58%) under RCP 4.5 (Tab. 3).

The wetland area classified by these indices as doubly vulnerable made up a substantial
portion of extant wetlands; 43% (RCP 2.4) to 48% (RCP 8.5) of existing contiguous US wetland
area are in watersheds with conditions indicating that they have both vertical and lateral
limitations to resilience. The majority of this vulnerable marsh area is located in watersheds
along the Gulf, Southeastern and Mid-Atlantic coasts (Tab. 3). There are 15% (RCP 2.4) to 4%
(RCP 8.5) of wetlands in watersheds that are classified as vertically resilient but laterally
vulnerable. There are 38% (RCP 2.4) to 47% (RCP 8.5) of wetland areas classified as laterally
resilient but vertically vulnerable. Only 4% (RCP 2.4) to 1% (RCP 8.5) of wetlands are both
laterally and vertically resilient.

Vertical and lateral resilience shifts depending on the severity of the RCP. With RCP
severity, the proportion of laterally resilient and doubly resilient wetland area increases for all
regions. However increasing LR does not fully make up for decreased VR. The proportion of
doubly vulnerable wetlands increases in total with RCP, although this is not the trajectory region
to region. Resilience in the South Central region is unchanged between RCP’s 2.5 and 4.5 and
decreases slightly under RCP 8.5. The proportion of doubly vulnerable wetland in the Southeast
and Northeast increased substantially from RCP 2.6 to 8.5; lateral and vertical vulnerability in
the mid-Atlantic drops of slightly from RCP 4.5 to 8.5.

4. Discussion

The goal of this paper was to synthesize existing data to extract spatial trends of vertical
and lateral tidal marsh resilience to SLR in the contiguous U.S. and in closely adjacent Canada.
We use the most extensive synthesis of accretion rates compiled thus far allowing us to test the inter-comparability of sediment accretion using a standardized method ($^{137}$Cs dated sediment cores) for developing the VR index. This meta-analysis adds new insight into the relationship between observed accretion, RSLR and tidal range along the U.S. coast and reveals some striking regional and latitudinal differences. Paired with the GIS analysis of lateral resilience, our results suggest that at a broad spatial scale both current and future tidal wetland vulnerability is, and will be, spatially variable, with accretion response falling off southward, because RSLR, tidal range, and the space available for potential marsh migration are spatially correlated with latitude.

The fact that accretion was not sensitive to FWA-SSC was contrary to our hypothesis and we think necessitates some additional examination of the modeling literature, as well as the dataset itself. First, modeling work by Vincent et al (2019) posits that for micro-tidal estuaries, accretion in interior marshes is insensitive to FWA-SSC, and that organic matter accretion is the dominant driver of total accretion. In our core dataset, the higher range FWA-SSC values we record come from the Mississippi river delta region of the Gulf of Mexico, where micro-tidal conditions make accretion relatively insensitive to the potential input. Second, modeling work by Morris et al (2016) argue that low SSC, high productivity, and resulting organic dominated soil accretion are typical of Atlantic Coast conditions. Holmquist et al (2018) showed that the majority of sampled marsh soil volume in the U.S. is dominated by organic matter.

On closer examination, two cores in our review dataset had inconclusive accretion rates, in which actual accretion rate was higher than could be captured by the core depth. One core came from the Mississippi delta, in Louisiana (DeLaune et al., 1992), a region which has high SSC and a prograding delta (NOAA, 2013). The other came from a mineral dominated low marsh setting in San Diego Bay (Weis et al., 2001). We hypothesize that the we did not see a significant relationship between FWA-SSC and accretion because of a combination of geomorphic, spatial, and sampling phenomenon: that there are negative spatial correlations...
between FWA-SSC and GT, that wetlands dominated by mineral accretion are less frequent than those dominated by organic accretion, and that it is harder to measure accretion using $^{137}$Cs dating in faster accumulating mineral dominated systems because the 1963 peak is more likely to be below the maximum depth of the barrel-style corer, typically used for this type of sampling.

4.1. Regional Trends in Resilience and Vulnerability

The two metrics we use to compare marsh resilience or vulnerability across CONUS-scale gradients, modeled net accretion and marsh migration potential, indicate a nuanced and potentially complex future for marsh vulnerability. DeLaune and White (2012) posited that the high rates of RSLR experienced by the Louisiana Gulf Coast, the highest historically observed in the CONUS, may be the closest analogues we have to 2100 projected SLR. For the Gulf coast, our model (Eq. 1) predicts continued vertical deficits under both RCP 2.6 and 8.5. RSLR scenarios. But, as Kopp et al. (2014) point out, these RCP scenarios are based on trends in past anomalies may be too pessimistic because fluid extraction, and associated submergence, may decline in the future (Intergovernmental Panel On Climate Change, 2014). Although we find that the South Central zone is most vulnerable to elevation loss, the area has lowlands primarily dominated by land cover class types favorable to migration (‘palustrine’, ‘upland’) and few watersheds dominated by ‘cultivated’ or ‘developed’ land. Nevertheless, because some ocean facing watersheds in Louisiana contain extensive existing wetlands, inland migration potential is low -- that is the limited area of non-wetland land available to migration is not proportional to existing coastal wetland area. A few of these ocean-facing watersheds with low inland migration potential could be underestimating migration potential if migration can occur beyond the watershed boundaries that make up the fundamental unit of this analysis.

The reason for the southward decline in vertical accretion capacity remains unknown, but may have to do with several intercorrelated geographic patterns including higher rates of
historic and projected RSLR from north to south, and decreasing tidal range from north to south. Future RSLR is projected to approach and exceed the rates (7.4 at Grand Isle, Louisiana; 1996-2010) in other micro-tidal regions, such as the Chesapeake Bay (7.4 mm yr\(^{-1}\) at Sewells Point, Virginia; 2000-2100 RCP 2.6; Supplemental Tab. 4; https://figshare.com/s/96f4554ff62aea7005b5). The Mid-Atlantic Coast is projected to have higher rates of RSLR compared to global averages because the loss of mass in ice sheets result in an overall weakening of their gravitational fingerprint (Engelhart & Horton, 2012; Kopp et al., 2014; Tamisiea & Mitrovica, 2011). The continued weakening of the AMOC will also contribute to RSLR (Sallenger et al., 2012). Many of these future potential vulnerable areas have low elevation land cover that is dominated by ‘palustrine’ or ‘upland’, and two mapped watersheds being dominated by ‘developed’ land. Migration potential is relatively low, likely because ‘cultivated’ land is often secondarily dominant in these watersheds, or because of topographic constraints. Compared to the Mid-Atlantic the Southeast is both less vulnerable to RSLR and to coastal squeeze. Tidal ranges are wider and projected RSLR is lower south of Cape Hatteras. There are also extensive low lying ‘palustrine’ dominated areas in high proportion to the current marsh area, exceeding 100% in some cases.

Southwest and Northeast projections differ from those of the mid-Atlantic and Southeast in that they are less vulnerable to losing their capacity to accrete, but more vulnerable to coastal squeeze (Fig. 7). These zones have both lower projected RSLR and wider tidal frames than other regions. This was observed by the recently published MARS index (Raposa et al., 2016). The Southwest and Northeast each have extensive low-lying areas predominantly dominated by development (Fig. 4C, 6C), which would presumably halt any potential marsh inland migration (Linhoss et al., 2015). Some areas, such as Southern California, not only have small tidal wetland area but are also the most potentially limited by lateral migration barriers (Fig. 4B, 7). The Southwest contains a high proportion of low lying agricultural land, especially around the Sacramento Delta, and in Northern California (Fig. 4C). If RSLR is higher than the median likely
scenarios predicted by RCP 8.5, then many Southwestern marshes, assumed resilient by other metrics, could be particularly vulnerable to coastal squeeze, as seen in the recently published WARMER model which predicts extreme habitat losses for the Pacific Coast under high SLR scenarios (Thorne et al., 2018). The CONUS Northwest is the least vulnerable to losing accretion capacity of all regions; migration potential is also high for the Northwest compared to the CONUS Southwest and Northeast.

The median GT of our calibration dataset (1.6 m) results in a RSLR threshold of 6.7 mm/yr (Eq. 5). For the sake of comparison, this threshold is only slightly higher than and estimates by Morris et al. (2016). They used a first principles soil formation modeling approach and estimate that total steady state accretion should not be greater than 5 mm yr$^{-1}$ given typical Gulf Coast and Atlantic conditions. Different regions likely have variable RSLR thresholds because of latitudinal and other spatial variability in tidal properties.

We hypothesize that the trends that we see in both vertical and lateral vulnerability are ultimately driven by latitudinal geographic patterns in postglacial geology, tidal ranges, and human settlement patterns. In more southerly regions, there is more ongoing isostatic rebound from the last glacial period, meaning that there are higher projected rates of RSLR. Southerly latitudes also have smaller tidal ranges, thus more vertically vulnerability. More northerly regions have less isostatic rebound, and some regions are isostatically uplifting, meaning slower projected RSLR. More northerly regions also have larger tidal ranges, leading to more VR.

Similarly, lateral vulnerability also follows latitudinal patterns. We hypothesize that in the further South Central region, lower topographic relief leads to more of a gradation between tidal and non-tidal wetlands meaning that there is more amenable land to facilitate inland marsh migration. In the more Northern and Western regions, we hypothesize that higher relief land leads to less of a gradation. Also in more Northerly Latitudes, we hypothesize much more low elevation land is devoted to development and agriculture.
4.2. Limitations and Caveats

There are limitations to this analysis to be addressed in future research using process-based modeling and vetted site-appropriate data (Kirwan et al., 2016; Swanson et al., 2014; Thorne et al., 2015). First, the net accretion model presented here was calibrated using modern data, and extrapolated out into the future using projected SLR. In other words, future conditions may be out of the boundaries of our calibration dataset. For example the coast of Maine has a combination of high projected SLR and wide tidal frames; most of our historical high SLR data is from areas with small tidal ranges. Although the linear modeling here and assumptions we have made may be viewed as simplistic, it allows the use of current rates of accretion for a first-cut resilience and vulnerability assessment that serves as a comparative metric, or an a priori hypothesis for future work. This could be approached next by using a targeted study applying process-based modeling over local to continental scale-gradients.

Similar to our analysis of the vulnerability of wetland accretion, our analysis of marsh migration is based on some key assumptions, and is meant to present a comparative metric based on current conditions. The analysis does not take into account artificially built tidal restrictions present now, or in the future. It also assumes a direct one-to-one replacement of non-cultivated, non-developed land cover types with tidal wetlands as sea-level rises. However, salinity intrusion into freshwater systems, and wetland plant competition with upland species are both biogeochemically and ecologically complex; more research is needed to determine whether or not marsh migration into adjacent uplands and palustrine wetlands can keep pace with edge erosion and pool expansion (Kirwan et al., 2016).

Our meta-analysis found that $^{137}$Cs dating is by far the most common dating technique available in the literature, with 480 out of 1,004 observations (Supplemental Tab. 1; https://figshare.com/s/96f4554ff62aea7005b5). Drexler et al (2018) point out that the utility of $^{137}$Cs dating will have a limited life time, that quality of dating is likely geographically variable because of differing rates of fallout in different regions, and that peaks are prone to error.
because of erosion, redeposition, and sediment mobility. The 1963 $^{137}$Cs peak quality can be highly uncertain because of the lack of published raw profiles and lab errors, and will become less reliable in the future as the peak decays exponentially with a half-life of 30.17 years. Our study reanalyzes data collected from 1978 to 2011 (Supplemental Tab. 2), so the fact that the quality of $^{137}$Cs is predicted to decline in the future is not as much as an issue as it would be if we were producing new data. As far as depositional issues and lab errors, one of the advantages of synthesis studies is that combining data from multiple sources and depositional environments allows us to extract trends despite lab or site-specific effects. The regression modeling accretion as a function of RSLR, GT, and relative flooding frequency had an $R^2$ of 0.33, which leaves room for future studies exploring other factors affecting accretion. We recommend future studies apply hierarchical models to disentangle the effects of environmental drivers from data quality and site-level variability (Lebauer et al., 2013). Our study did not synthesize full profiles of $^{137}$Cs but could have benefited from them if they were widely available. Future studies should use best practices in archiving $^{137}$Cs information including full radioactivity profiles and lab errors (Drexler et al., 2019). Finally, we propose that we researchers need a way of propagating uncertainty in $^{137}$Cs dating, similar to what we have $^{14}$C (Blaauw and Christen, 2011) and $^{210}$Pb dating (Aquino-López et al., 2018). Doing so could help us disentangle environmental signals from data quality issues as $^{137}$Cs dating using the 1963 peak inevitably becomes less reliable into the future.

Despite the caveats of using the present and recent past to infer the future, such scenario development has value. Our analysis, based upon the largest such dataset of linked marsh accretion, RSLR, tidal range, and FWA-SSC data, provides evidence that tidal wetlands of the CONUS, and particularly the South Central zone are vulnerable to current rates of SLR. Many more in the CONUS Southeast, and mid-Atlantic, such as the Chesapeake Bay, will likely be vulnerable to even conservative estimates of future RSLR. Although tidal wetlands in the CONUS Southwest and Northeast are comparatively less vulnerable to losing their capacity to

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accrete relative to SLR, they are more vulnerable to coastal squeeze because of topographic constraints, and the dominance of developed and/or cultivated land in many watersheds.

4.3. Future Research Directions

Tidal wetlands are vital ecosystems that serve as nurseries for fishery species (Boesh and Turner, 1984), carbon sinks (Mcleod et al 2011), and are home to endangered and endemic species (Rosencranz et al., 2019). Previous efforts analyzing their resiliency to RSLR (Enwrite et al., 2016; Thorne et al., 2018), as well as current monitoring programs (Steyer et al., 2003), have largely been regionally focused. However national-scale action such as funding, guidance, and goal setting, will likely be needed to ensure wetland persistence and function into the future as sea-levels rise.

In our linear modeling of vertical vulnerability, we learned that inundation regime is important for two reasons. Without it them model explains 10% less of the variance in the data and GT is not a significant parameter. The fact that GT’s role in accretion is conditional on separate consideration of inundation regime Points to a potentially important physical link between the two that more advanced process-model could potentially elucidate. Inundation regime based on a list of criteria that is somewhat subjective. The problem we had was that there is not a consistent definition of relative inundation across the broad array of studies analyzed. Future studies could have a more nuanced integration by using a separate data-model for inundation class, having multiple researchers code inundation status, and integrating more uncertainty for classifications that are less straightforward. Futures studies could also attempt at a more unified concept of tidal inundation status such as dimensionless tidal elevation (Z*; Janousek et al., 2016) or flooding time (Hickey, 2019). A continuous driver as opposed to a binary definition would not be as affected by regional definitions or judgement calls made by data interpreters.
What we cannot learn from this dataset and linear models are what marsh collapses resulting from accretion deficit will occur, whether they will be gradual or sudden, instigated by disturbance events such as hurricanes, and what the early stages look like. We hypothesize that monitoring with sediment elevation tables (SETs; Cahoon et al., 2002) could catch early stages of collapse by observing persistent accretion deficit in more frequently flooded, more mineral dominated sections of marsh. However, conclusive evidence supporting this would take an expanded database, additional monitoring, and possibly a more process rich modeling approach. We should also anticipate places and times where thresholds are passed or as it could mean marsh collapse events could spatially and temporally cluster such as following El Niño events (Goodman et al. 2018), or correlated to the lunar nodal cycle. (Peng et al., 2019)

Throughout the text we have referred to vertical and lateral resilience indices, because we think the metrics presented here are far from mature enough to be thought of as a forecast, used for actionable decision making and land use planning. For future research we recommend more extensive data-model integration and the development towards a forecasting system (Dietze et al., 2018) by integrating process- and data-models, using the statistical modeling framework here for model intercomparison or model averaging, and further developing the indices presented here to inform management goals.

First, we recommend a forecasting system which integrates environmental drivers, process models and data models. Drivers from this study that could be integrated include considerations of regional topography, low elevation land cover class, and localized RSLR scenarios. Process models could include members of the Marsh Equilibrium Model (MEM; Morris et al., 2002) family and WARMER (Thorne et al., 2018). A more sophisticated data model could account for differences in data-quality across time and between studies (Drexler et al. 2018). Bayesian hierarchical models have the capacity to propagate uncertainties in environmental drivers, parameters, and models themselves (Hobbs and Hooten, 2015; Dietze et al., 2018)
Second, near-term forecasting (Dietze et al., 2018), which is increasingly being used as a tool for promoting actionable decision support and collaboration between scientists and decision makers, benefits from Bayesian models and from model selection (Conn et al., 2018) and model averaging techniques. The vertical vulnerability index, which is a simple linear model could be used to performance test more complex models against.

Third and finally the indices here could be used to inform goals for restoration planning. VR index, which is simply a modeled accretion deficit, could be used to set goals and both sediment and financial budgets for sediment diversion (Snedden et al., 2007) or sediment augmentation projects (Ulibarri et al., 2020). The two components of LR index – topography and low elevation land cover -- could help planners determine acreage goals for securing marsh migration space through negotiation of conservation easements, restrictive covenants, and outright purchases (Field et al., 2017).

5. Conclusion

Kirwan et al. (2016) observed that marsh vulnerability can be overstated if dynamic feedback between marsh elevation and inundation is not taken into account, and potential inland migration of marshes is ignored. Our analyses support the existence of dynamic feedback because marsh accretion keeps pace with RSLR up to a point. Our analysis supports estimates by Morris et al. (2016) that estimates 5 mm yr\(^{-1}\) as a typical upper limit for accretion in most Atlantic and Gulf Coast marshes are already near there. Using a median GT results our VR equation outputs a RSLR threshold of 6.7 mm yr\(^{-1}\). RSLR threshold may be spatially variable, lower in places with smaller tidal ranges and higher in places with larger tidal ranges. We see much of the Gulf Coast already passed their threshold, and much of the Atlantic Coast passing it by 2100 under even modest RSLR scenarios. However our analyses also take into account the magnitude and geography of projected RSLR, comparative trends in low elevation land cover, as well as the fact that microtidal marshes are additionally vulnerable to SLR. Given
our results, we believe that marsh sustainability into the future should not be generally assumed. In fact, if the amassed data and indices we present are representative, we expect future marsh vulnerability to be spatially variable with many currently stable marshes switching from aggrading to submerging, and some marshes failing to migrate inland because of topography and a lack of unoccupied low-elevation land cover.

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J.R.H., L.N.B., and G.M.M. jointly designed the research and created the dataset. J.R.H. wrote code, performed GIS analysis, and drafted the manuscript. J.R.H. lead manuscript writing. L.N.B. and G.M.M. worked on data interpretation and on manuscript drafts.
Figure 1: A. Map shows the distribution of accretion rate data used in the analyses.
Figure 2: Scatterplot showing the variables used in the vertical resilience index, relative sea-level rise, inundation category, and tide range. The frequently inundated datapoints were used to create the index. The dashed line shows a one to one ratio between relative sea-level rise and accretion rate, and which data used to calibrate the index are aggrading, above the line, and submerging, below the line.
Figure 3: Regional summaries of the vertical and lateral resilience properties. Top panel shows wetland area-weighted averages grouped by state-level boundaries making up the 6 regions analyzed, for each representative concentration pathway (RCP). Bottom panel shows the fractional area of various simplified land cover classes occupying the SLR zone under RCP 4.5.
Figure 4: A and B. show vertical and lateral resilience indices for representative concentration pathway (RCP) 4.5 extrapolated out at the level of intermediate watershed unit for the Pacific Coast of the United States. For A. and B. each point represents the centroid of a watershed unit. For C. watershed units’ full extent are shown with the dominant landcover type in their RCP 4.5 sea-level rise zone displayed. C. Shows the dominant landcover class of the sea-level rise zone under RCP 4.5. Blank watersheds had incomplete digital elevation model coverage.
Figure 5: A and B. show vertical and lateral resilience indices for representative concentration pathway (RCP) 4.5 extrapolated out at the level of the intermediate watershed unit for the Gulf Coast of the United States. For A. and B. each point represents the centroid of a watershed unit. For C. watershed units’ full extent are shown with the dominant landcover type in their RCP 4.5 sea-level rise zone displayed. C. Shows the dominant landcover class of the sea-level rise zone under RCP 4.5. Blank watersheds either were either dominated by mangroves or had incomplete digital elevation model coverage.
Figure 6: A and B. show vertical and lateral resilience indices for representative concentration pathway (RCP) 4.5 extrapolated out at the level of the intermediate watershed unit for the Atlantic Coast of the United States. For A. and B. each point represents the centroid of a watershed unit. For C. watershed units’ full extent are shown with the dominant landcover type in their RCP 4.5 sea-level rise zone displayed. C. Shows the dominant landcover class of the sea-level rise zone under RCP 4.5. Blank watersheds either were either dominated by mangroves or had incomplete digital elevation model coverage.
Figure 7: Ordinal plot showing the two resilience indices presented in this paper extrapolated at the intermediate watershed scale for Representative Concentration Pathway (RCP 4.5). Vertical Resilience Index is the modeled net accretion (accretion – relative sea-level rise). Lateral Resilience Index is amount of land available to facilitate lateral migration of wetlands and is expressed as a fraction of current tidal wetland area. Quadrats indicate values indicating vulnerable resilience (V+) and vertical vulnerability (V-) on the y-axis, and lateral resilience (L+) and later vulnerability (L-) on the x-axis. Note that the x-axis is log-transformed for the sake of visualization. Point shapes and colors indicate broad political-geographic regions and point size indicates the absolute area of tidal wetland coverage within the watershed.
Table 1: How Coastal Change Analysis Program (C-CAP) land cover classes were simplified and treated as candidates for lateral marsh migration in our lateral resilience analysis.

| Simplified Landcover Class | Original C-CAP Classes                                                                 | Considered Candidate for Lateral Saltmarsh Migration? |
|----------------------------|----------------------------------------------------------------------------------------|-------------------------------------------------------|
| Tidal Wetlands             | Estuarine Emergent, Forested and Scrub/Shrub Wetlands; Palustrine Emergent, Forested and Scrub/Shrub Wetlands below modern day mean higher high water for spring tides (MHHWS). | No                                                    |
| Non-Tidal Wetlands         | Palustrine Emergent, Forested, and Scrub/Shrub Wetlands above modern day MHHWS.         | Yes                                                   |
| Water                      | Open Water; Unconsolidated Shore; Estuarine and Palustrine Aquatic Bed                  | No                                                    |
| Developed                  | High Intensity, Medium Intensity, and Low Intensity Developed; Developed Open Space      | No                                                    |
| Upland                     | Grassland, Scrub/Shrub, Evergreen, Deciduous, and Mixed Forest                          | Yes                                                   |
| Cultivated                 | Cultivated; Pasture/Hay                                                                 | No                                                    |
| Bare                       | Bare                                                                                    | Yes                                                   |
Table 2: Linear Modeling Output

| n  | Model Form                        | R²  | p             | Intercept | Parameter Summary          | AICc |
|----|-----------------------------------|-----|---------------|-----------|-----------------------------|------|
| 1  | accretion ~ (inundation * (rsl + gt + fwa-ssc + time)) | 0.56 | 1.40E-12      | -1.8±2.9 | inundation: 3.4±3.3         | 437  |
|    |                                   |     |               |           | rsl: 0.8±0.1                |      |
|    |                                   |     |               |           | gt: 3.3±0.8                 |      |
|    |                                   |     |               |           | fwa-ssc: -0.02±0.0          |      |
|    |                                   |     |               |           | time: -0.0±0.0              |      |
|    |                                   |     |               |           | inundation:rsl: -0.6±0.2    |      |
|    |                                   |     |               |           | inundation:gt: -3.8±1.0     |      |
|    |                                   |     |               |           | inundation:fwa-ssc: 0.0±0.0 |      |
|    |                                   |     |               |           | inundation:time: 0.1±0.0    |      |
| 2  | accretion ~ (inundation * (rsl + gt + time)) | 0.55 | 2.55E-13      | -1.9±2.9 | inundation: 3.2±3.3         | 435  |
|    |                                   |     |               |           | rsl: 0.8±0.1                |      |
|    |                                   |     |               |           | gt: 3.3±0.8                 |      |
|    |                                   |     |               |           | time: -0.0±0.0              |      |
|    |                                   |     |               |           | inundation:rsl: -0.5±0.2    |      |
|    |                                   |     |               |           | inundation:gt: -3.4±1.0     |      |
|    |                                   |     |               |           | inundation:time: 0.1±0.0    |      |
| 3  | accretion ~ (inundation * (rsl + gt)) | 0.54 | 2.40E-14      | -2.5±1.4 | inundation: 5.4±1.8         | 431  |
|    |                                   |     |               |           | rsl: 0.8±0.1                |      |
|    |                                   |     |               |           | gt: 3.4±0.7                 |      |
| 4  | accretion ~ (inundation * (rsl + gt + time)) | 0.34 | <2.2E-16      | -0.4±1.1 | inundation: 2.9±1.6         | 1207 |
|    |                                   |     |               |           | rsl: 0.6±0.1                |      |
|    |                                   |     |               |           | gt: 1.5±0.3                 |      |
|    |                                   |     |               |           | time: 0.0±0.0               |      |
|    |                                   |     |               |           | inundation:rsl: -0.4±0.1    |      |
|    |                                   |     |               |           | inundation:gt: -1.6±0.4     |      |
|    |                                   |     |               |           | inundation:time: -0.0±0.0   |      |
| 5  | accretion ~ (inundation * (rsl + gt)) | 0.33 | <2.2E-16      | 0.4±0.7  | inundation: 2.9±0.9         | 1204 |
|    |                                   |     |               |           | rsl: 0.6±0.1                |      |
|    |                                   |     |               |           | gt: 1.4±0.3                 |      |
| 6  | accretion ~ (rsl + gt)            | 0.23 | <2.2e-16      | 3.1±0.4  | rsl: 0.30±0.04              | 1222 |
|    |                                   |     |               |           | gt: 0.01±0.1                |      |
| 7  | accretion ~ (rsl + gt)            | 0.30 | <2.2e-16      | 3.1±0.4  | rsl: 0.32±0.03              |      |
|    |                                   |     |               |           | gt: 0.05±0.01               |      |
Table 3: Summary of Wetland Area Resilient to Relative Sea-Level Rise

| Scenario | North West | South West | South Central | South East | Mid Atlantic | North East | Total |
|----------|------------|------------|---------------|------------|--------------|------------|-------|
| **Vertically Resilient** | | | | | | | |
| RCP2.6   | 40 (82)    | 44 (82)    | 20 (1)        | 156 (31)   | 5 (2)        | 95 (55)    | 360 (15) |
| RCP4.5   | 39 (79)    | 42 (79)    | 0             | 126 (25)   | 0            | 34 (20)    | 242 (10) |
| RCP8.5   | 36 (73)    | 40 (74)    | 0             | 0          | 0            | 26 (15)    | 102 (4)  |
| **Laterally Resilient** | | | | | | | |
| RCP2.6   | 0          | 0          | 703 (48)      | 148 (30)   | 79 (39)      | 7 (4)      | 937 (38) |
| RCP4.5   | 0          | 0          | 733 (50)      | 167 (34)   | 87 (43)      | 30 (18)    | 1018 (41)|
| RCP8.5   | 0          | <1 (1)     | 765 (52)      | 247 (50)   | 98 (48)      | 39 (23)    | 1151 (47)|
| **Doubly Resilient** | | | | | | | |
| RCP2.6   | 9 (18)     | 10 (18)    | 11 (1)        | 52 (10)    | 1 (<1)       | 26 (15)    | 108 (4)  |
| RCP4.5   | 10 (21)    | 11 (21)    | 0             | 52 (10)    | 0            | 8 (5)      | 81 (3)   |
| RCP8.5   | 13 (27)    | 13 (25)    | 0             | 0          | 0            | 8 (5)      | 35 (1)   |
| **Doubly Vulnerable** | | | | | | | |
| RCP2.6   | 0          | 0          | 745 (50)      | 141 (28)   | 120 (59)     | 44 (26)    | 1050 (43)|
| RCP4.5   | 0          | 0          | 746 (50)      | 152 (31)   | 117 (57)     | 100 (58)   | 1115 (45)|
| RCP8.5   | 0          | <1 (<1)    | 714 (48)      | 250 (50)   | 106 (52)     | 98 (57)    | 1168 (48)|

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