



Drivers of Elevation Change in a Retreating Coastal Forest

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Abstract

The future of coastal marshes depends in part on their ability to migrate inland with sea level rise into coastal forests, but projections of land conversion commonly assume that migration occurs over a static terrestrial topography. Here, we assess the assumption of a static topography in a rapidly retreating coastal upland forest by measuring surface accretion, subsurface expansion or subsidence, and net elevation change across a gradient of salinity and inundation stress. We found that salt-affected forest soils were largely stable over the 5-year study period, exhibiting limited elevation change (-0.2 to 1.5 mm yr⁻¹) in most stages of forest health. This stability was achieved by relatively equal and offsetting rates of accretion and subsidence that were each greater than expected. Surface accretion rates were positively correlated with the total cover of *Morella cerifera* shrubs rather than *Phragmites australis*, an invasive reed that dominates retreating coastal forests in the mid-Atlantic. Subsidence was greatest in higher elevation forest areas and attributed to root zone collapse that occurred prior to ongoing tree mortality. Net elevation-change rates were highest in the marsh-forest transition zone (2.9 mm yr⁻¹), which is characterized by dead trees and fully developed marsh vegetation. Though this study measured relatively low elevation change rates in most portions of a retreating coastal forest, similarity between transition zone and adjacent salt marsh rates suggests that elevation change rates respond quickly to marsh migration and are elevated by the time of extensive tree mortality.

Keywords Accretion · Coastal landscape change · Marsh migration · *Morella cerifera* · *Phragmites australis* · Surface elevation table

Introduction

Sea level rise is increasing tidal flooding, storm surges, and salinization of groundwater in ways that threaten both human infrastructure and natural ecosystems. Salt marshes are highly valued ecosystems along global coastlines that provide protection from storms (Temmerman et al., 2013), improve water quality (Barbier et al., 2011), support estuarine and marine food webs (Colombano et al., 2021), and sequester carbon (McLeod et al., 2011). Salt marshes grow vertically through soil organic matter accumulation and sediment trapping but must migrate inland to survive rates of sea

level rise that are faster than their ability to build soil (Kirwan & Megonigal, 2013). The replacement of coastal forests with marshes has been widely documented in regions of the USA where sea level rise intersects a gently sloping topography (Kirwan & Gedan, 2019; White et al., 2022). Rates of coastal forest retreat are accelerating in parallel with rates of relative sea level rise (Schieder & Kirwan, 2019), resulting in marsh migration rates that have compensated for erosion and drowning in portions of the mid-Atlantic and Gulf of Mexico (Schieder et al., 2018; Chen & Kirwan, 2022; Raabe & Stump, 2015).

Future projections of marsh area change under sea level rise indicate that the fate of marshes is largely tied to their ability to migrate inland (Kirwan et al., 2016; Osland et al., 2022; Schuerch et al., 2018). However, projections of upland conversion to marsh rely on a passive inundation, or “bathtub model,” approach in which upland topography is constant through time (Farron et al., 2020; Geselbracht et al., 2011; Linhoss et al., 2015; Molino et al., 2022; Osland et al., 2022; Schuerch et al., 2018). This approach predicts widespread conversion of uplands to wetlands with gradual

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sea-level rise, but observed rates of coastal forest retreat are only about half the rate expected by sea level rise propagating across a static topography (Chen & Kirwan, 2024). Furthermore, assumptions of a static topography contrast with the general understanding that feedbacks between flooding, plant productivity, and sediment transport control surface elevations of other coastal ecosystems such as mangroves, marshes, and tidal forests (FitzGerald et al., 2008; Kirwan & Megonigal, 2013; Krauss et al., 2024; Saintilan et al., 2023a, 2023b). Because coastal hydrology and ecology are intertwined (Fagherazzi et al., 2025; Kirwan et al., 2025), and both forests and marshes are sensitive to small changes in elevation (Krauss et al., 2010), gains or losses of mere centimeters may significantly impact coastal forest retreat rates.

Upland forest soils are usually considered geomorphically stable over decadal timescales, with detrital soil inputs largely balanced by respiratory losses (Schimel, 1995), but several geomorphic processes may contribute to elevation gain in coastal forests. Deposition of autochthonous material from litterfall is likely the dominant source of surface accretion. Litterfall can make up 60% of total aboveground productivity in tidal freshwater forests (Stahl et al., 2018) and > 70% in boreal forests (Bhatti & Jassal, 2014). Increased inundation with saltwater may inhibit both aerobic and anaerobic decomposition of soil organic matter (Setia et al., 2011; Qu et al., 2021), potentially allowing for greater surface accretion in salt-affected forests, but observed outcomes are subtle and highly variable (Stagg et al., 2018; Helton et al., 2019; Wen et al., 2019; Smith et al., 2024). Hurricanes and extreme flooding events may also deposit sediment and organic matter (i.e., wrack) at the marsh-upland transition (Tate & Battaglia, 2013). Additionally, colonization of the understory by salt- and flood-tolerant species can rapidly build elevation through increased litter accumulation, belowground biomass production, and sediment trapping. *Morella cerifera* thickets can substantially increase litterfall (Brantley & Young 2008), and accretion rates within mature *Phragmites australis* stands can reach 10 mm yr⁻¹ (Rooth et al., 2003), potentially increasing elevation gain where these species establish or increase in cover.

Alternatively, the introduction of labile carbon (dying roots) and pulses of sulfate-rich seawater may enhance decomposition, leading to elevation loss (Chambers et al., 2019; Weston et al., 2011). Whether saltwater intrusion facilitates or hinders decomposition is directly related to the duration of flooding, soil organic C, and previous hydrologic conditions (Stagg et al., 2018; Wen et al., 2018; Smith et al., 2024). Excessive flooding may also substantially alter organic soil structure, leading to destabilization (Chambers et al., 2019). Additionally, flooding-induced tree mortality can lead to subsidence due to loss of root turgor pressure and collapse of the rooting zone. Elevation losses of 0.5–10 cm yr⁻¹ following hurricanes have been documented

in mangroves, tidal freshwater forests, and pocosins (Cahoon et al., 2003; Middleton & David, 2022; Craft, 2012).

Breakthroughs in understanding of the ecogeomorphic processes driving elevation gain in coastal ecosystems have come largely through repeat elevation measurements associated with the surface elevation table-marker horizon (SET-MH) approach (Cahoon et al., 1995; Saintilan et al., 2022; Webb et al., 2013). The combination of shallow and deeply anchored SETs reveals elevation changes that occur in shallow (root zone) and deep (sub root zone) portions of the soil profile. Horizon marker layers in the soil can further differentiate surface accretion. Integrating the results of these tools can help to infer which processes are responsible for net gain or loss in elevation at the soil surface, which has clear feedbacks to inundation in the coastal zone. Surface elevation tables have been widely adopted in coastal ecosystems such as marshes (Covington, 2020; Saintilan et al., 2022), mangroves (Krauss et al., 2010; Lovelock et al., 2015), and, more recently, tidal freshwater forests (Stagg et al., 2018; Krauss et al., 2024), pocosins (Moorman et al., 2024), and coastal scrub/shrubland (Cook et al., 2016; Ardón et al., 2025). However, elevation change rates have not been reported for upland coastal forests.

Here, we investigate surface elevation change using the SET-MH approach to expand the coastal geomorphic framework into a salt-affected upland forest. Root zone subsidence in forest soils prior to forest retreat would indicate that forest retreat and marsh migration might advance more quickly than static models predict. Alternatively, substantial accretion of forest soils would potentially explain observed lags between sea level rise and forest retreat. More generally, we aim to understand how the elevation dynamics of salt-stressed forests may change over time and whether these processes enhance or mitigate future marsh migration.

Study Site

Our study focuses on a well-studied coastal maritime forest on Virginia's Atlantic Coast. This area is located in the U.S. mid-Atlantic sea level rise hotspot, where relative sea level rise rates are faster than eustatic rates (Sallenger et al., 2012) and influenced by deep subsidence rates of 1–3 mm yr⁻¹ (Eggleston & Pope, 2013; Ohenhen et al., 2023). The rural and low-lying mid-Atlantic coast experiences rapid marsh migration into uplands, making it an ideal location to study the elevation dynamics of marsh transgression. Approximately 220 km² of coastal forests have converted to marsh in the mid-Atlantic since 1984 (Chen & Kirwan, 2024), and an additional 1050–3748 km² of conversion is projected for the Chesapeake Bay by 2100 (Molino et al., 2022).

The study site is located within The Nature Conservancy's Brownsville Nature Preserve, Nassawadox, VA

(37.464814° N, −75.835185° W) and adjacent to Upper Phillips Creek marsh (UPC), a focal study area of the Virginia Coast Reserve (VCR) Long-Term Ecological Research (LTER) program (Fig. 1). Due to its exceedingly low slope (0.0006 m/m) (Fig. 1), this area has undergone significant coastal transgression, highlighted by the conversion of upland forest to organic-rich high marsh (Brinson et al., 1995). Marsh transgression is occurring at rates between 0.3 and 1 m yr^{−1} (Fig. 1) and total marsh area increased by 3% (0.9) ha between 2000 and 2017 (Flester & Blum, 2020).

The studied forest is an upland, maritime forest comprised mainly of loblolly pine (*Pinus taeda*) with black tupelo (*Nyssa sylvatica*), red maple (*Acer rubrum*), and American holly (*Ilex opaca*) interspersed at higher elevations (Langston et al., 2025). The forest understory is composed mainly of grasses *Phragmites australis*, *Panicum virgatum*, and *Chasmanthium laxum*, and the thicket-forming shrub *Morella cerifera*. Eastern red cedar (*Juniperus virginiana*), *P. australis*, and shrubs such as *Baccharis halimifolia* and *Iva frutescens*

dominate the forest-marsh interface (Messerschmidt et al., 2022). The average forest elevation is 1.15 m, which is only 20 cm higher than the adjacent marshes (Table 1). The entire study area is within 15 cm of the Highest Astronomical Tide (1.07 m NAVD88) converted from observed water levels at Wachapreague, VA (Station 8631044, NOAA Tides and Currents, Blum et al., 2021). Over the study period, groundwater levels exceed surface elevations upwards of 100 days yr^{−1} at lower elevations (Pratt et al., 2025).

The Brownsville forest was delineated into four forest health zones in 2019, representing a gradient of flooding and salinity stress: high, mid, low, and transitional forest (Fig. 1) (Smith & Kirwan, 2021; Langston et al., 2025). High forest was initially defined as largely unaffected by saltwater intrusion and containing a mixed species composition of both coniferous and deciduous tree species. The mid forest displays early signs of salt stress, including deciduous tree mortality, but contains a healthy coniferous population. Low forest is defined by higher salt stress, indicated

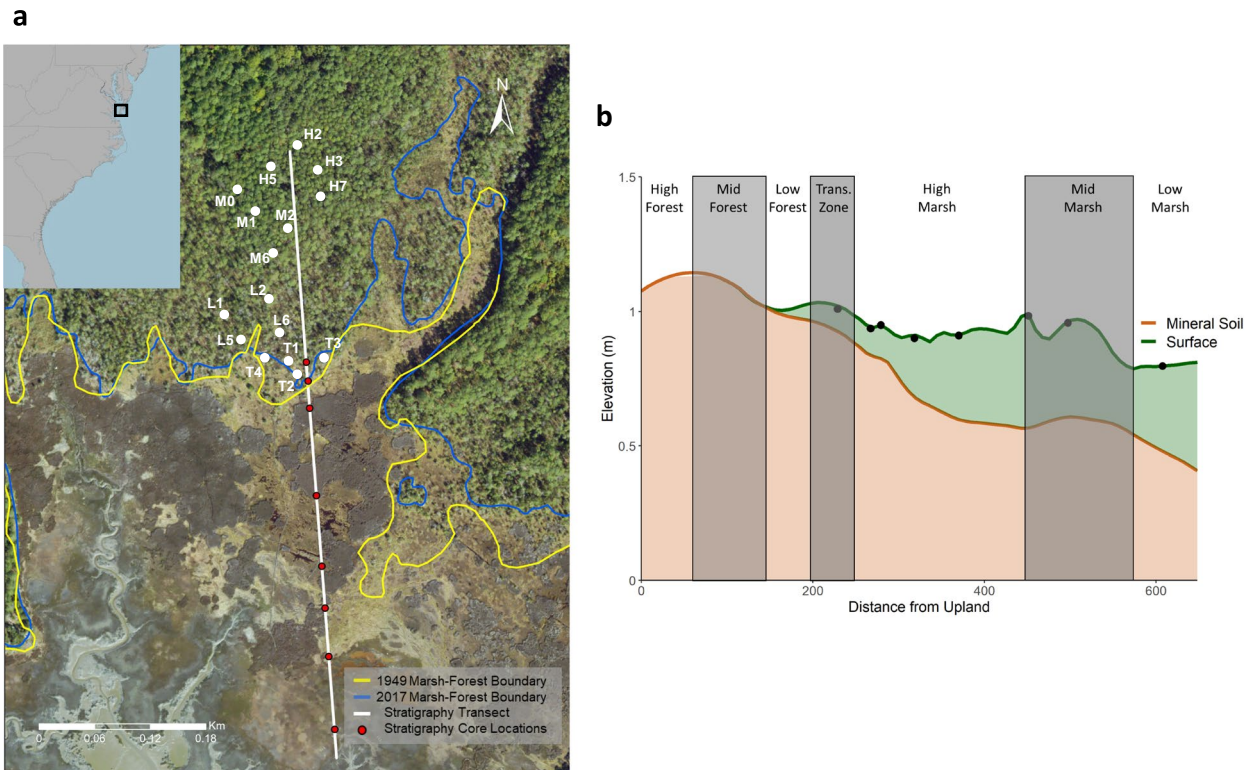


Fig. 1 Retreating coastal forest study site. **a** The Brownsville Preserve Forest is located near Phillips Creek (SW portion of image) on Virginia's Atlantic Coast (inset). Locations of the surface elevation tables (SETs) are shown in white. SETs are labeled by first letter denoting forest zone (high, mid, low, transition) and plot number. Yellow (1949) and blue (2017) lines represent historic marsh-forest boundaries delineated from aerial imagery from Virginia Department of Transportation and Virginia Base Mapping Program, respectively. Forest retreat rates (0.5 m yr^{−1} from 1949 to 2017) were determined using linear regression in Digital Shoreline Analysis System (DSAS).

The solid white line represents an elevation transect from healthy forest to low marsh with core locations (red dots) for the stratigraphic profile shown in Fig. 1b. **b** Surface elevation and stratigraphic profile that was determined via RTK GPS survey, where all elevations are relative to NAVD88, which approximates mean sea level in the region. The green-shaded area represents organic-rich marsh soils and the orange-shaded area represents the underlying mineral-rich, terrestrial sediment, both interpolated from eight sediment cores (Supp Fig. 2)

Table 1 Elevation, vegetation characteristics, and elevation change rates for each individual SET. Vegetation cover is expressed as a percent at each individual canopy height. Rates of surface accretion, netelevation change, and subsurface change expressed as mm yr⁻¹. A negative subsurface change indicates subsidence and a positive value indicates subsurface expansion

Forest zone	SET #	Elevation (NAVD88)	Shrub cover	Phragmites cover	Marsh cover	Surface accretion (mm yr ⁻¹)	Subsurface change (mm yr ⁻¹)	Net elevation change (mm yr ⁻¹)
High	2	1.12	80	0	0	4.2	-4.3	-0.1
High	3	1.23	20	5	0	1.1	-0.7	0.4
High	5	1.11	40	0	0	4.2	-5.8	-1.6
High	7	1.20	15	0	0	2.1	-1.8	0.3
Mid	0	1.21	60	0	0	5.2	-3.0	2.3
Mid	1	1.25	70	0	0	4.2	-2.8	1.4
Mid	2	1.22	30	0	0	4.2	-2.8	1.5
Mid	6	1.14	55	30	0	4.2	-3.3	0.9
Low	1	1.17	40	15	0	2.1	-2.0	0.1
Low	2	1.24	40	20	2	3.2	-1.9	1.2
Low	5	1.20	20	50	0	3.2	-2.6	0.5
Low	6	1.20	30	10	0	2.1	-1.2	0.9
Transition	1	0.94	0	10	80	3.9	0.5	4.3
Transition	2	.099	0	15	45	1.6	2.6	4.2
Transition	3	0.98	0	0	85	3.2	0	3.2
Transition	4	1.07	5	40	10	2.4	-2.4	0.0

by limited tree regeneration, increasing conifer stress, and standing dead trees. The transition zone contains ~50% adult tree mortality, no regeneration, and an understory of typical high marsh species. Since 2019, the shrub *M. cerifera* has expanded by 45–65% in high and mid forest plots, while in the low forest, there has been a steady spread of invasive *Phragmites australis* (Sward et al., 2023).

Methods

Changes in forest soil elevations were measured using the surface elevation table-marker horizon (SET-MH) approach that quantifies both surface accretion and net elevation change with an accuracy of 1–2 mm (Cahoon, 2002; Callaway, 2013; Lynch, 2015). SETs consist of a deep benchmark driven to refusal, on which a mechanical leveling arm is attached at fixed positions around the mark, allowing for repeated measurements of net elevation change with respect to the base of the benchmark (Cahoon et al., 2002). It is paired with a marker horizon, which consists of a feldspar layer established on the soil surface, on top of which surface accretion can be measured. The calculated difference between surface accretion (MH) and net elevation change (SET) allows for an estimate of total subsurface processes between the marker horizon and benchmark base (Cahoon et al., 1995). To examine the contribution of root zone processes to total subsurface elevation change, 3-inch diameter, capped,

aluminum pipes were installed to a depth of ~30 cm (Langley et al., 2009) (Fig. 2a). These pipes have 1-inch holes drilled into them to allow for root ingrowth (Walters et al. in prep).

Four SETs were installed in each zone to depths between 16 and 21 m. The high, mid, and low forest zones were installed in March 2019 and the transition zone in August 2020 (Table 1). Each SET was paired with a marker horizon and 1–2 shallow pipes driven to ~30 cm. Elevation measurements were taken approximately 3 times per year, beginning in October 2019 (high, mid, and low forest) and January 2021 (transition zone). To measure elevation at each SET, the arm was leveled at four equidistant positions approximately 0.5 m from the benchmark. At each position, leaf litter was gently removed by hand (Fig. 2b), taking care not to disturb the underlying surface. Partially decomposed litter less than ~2 cm in size was left in place. Pinecones and branches were removed from the measurement area only if their removal did not disturb the forest soil. Any foreign object (stick, rock, cone, etc.) partially buried beneath the surface was left in place but excluded from the analysis. Nine fiberglass pins were then lowered to the exposed surface of the underlying forest soil. Elevation was measured as the distance between the arm and the top of each pin. Leaf litter was removed (as above) from a small section of the marker horizon before measuring the depth of the feldspar layer from the revealed soil surface. Beginning in February 2022, the SET arm was leveled over the aluminum pipes and their heights were measured.

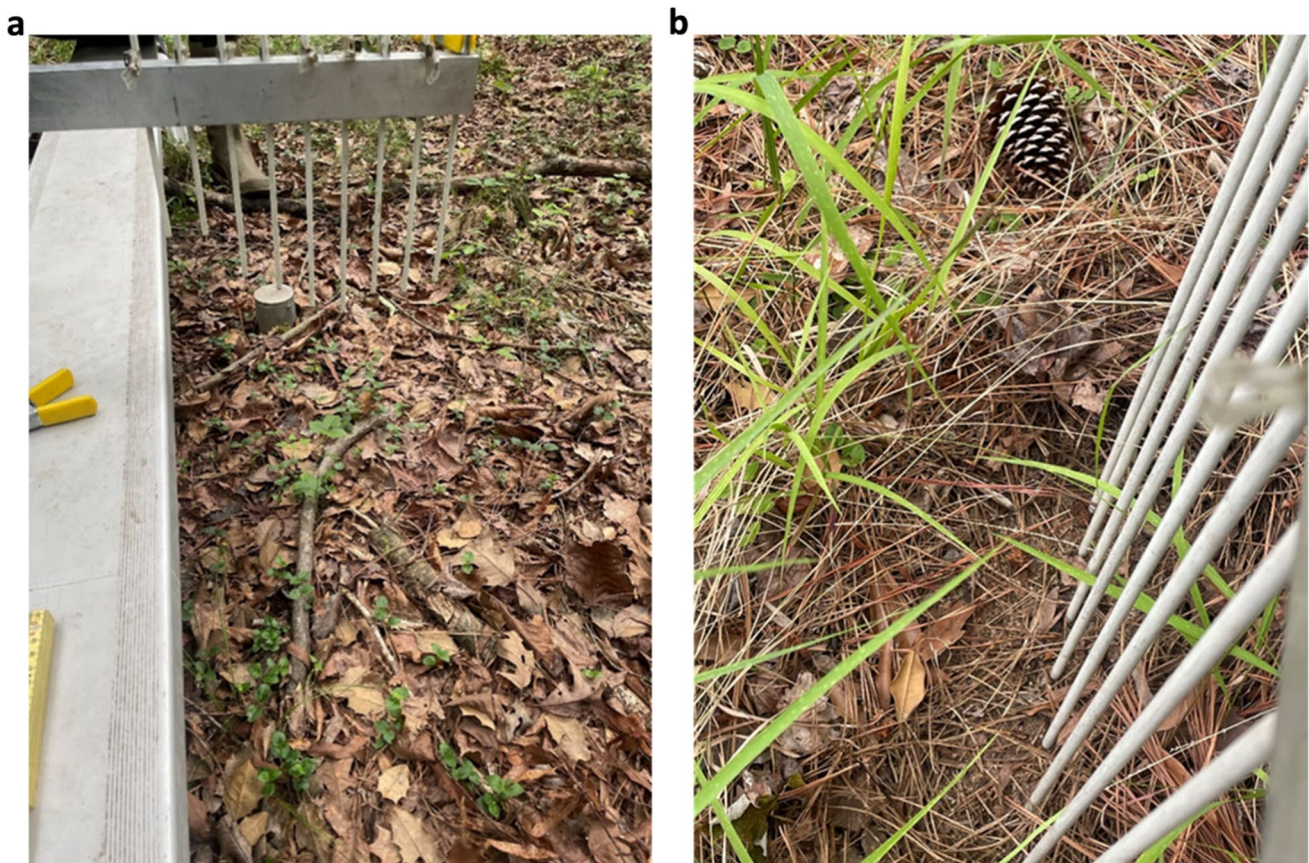


Fig. 2 Photographs of SET measurement approach. **a** Photograph of the high forest floor, with a shallow SET pipe in the background. **b** Photograph of forest floor, with leaf litter gently removed revealing

soil surface prior to measurement. Leaf litter is replaced upon completion of each measurement

After the measurements at each SET position and marker horizon were complete, the leaf litter was gently returned to its original position.

Statistical Approach

Net elevation change rates were calculated using the “regress then average” method (Lynch et al., 2015; Russel et al., 2022). Elevation trends were first regressed through time at the individual pin level, then averaged by arm position and finally aggregated to the entire plot. Each plot estimate, therefore, reflects the average of 36 individual pin measurements. Marker horizon derived accretion was then subtracted from plot level net elevation change to calculate total subsurface change, where a negative subsurface change indicates subsidence, while a positive value reflects subsurface expansion. Elevation change within the top 30 cm, here referred to as root zone subsurface change, is calculated as the difference between average SET cumulative change and shallow pipe derived cumulative change at each time step. Subsurface change below 30 cm, referred to as shallow hydro-geologic change, is the difference

between total subsurface and root zone elevation change. One-sided *t*-tests were used to determine if average rates of change were greater than 0 (no change) and 4.1 mm yr^{-1} (local relative sea level rise). Local relative sea level rise was calculated between 1997 and 2017 based on NOAA’s Wachapreague Station (ID 8631044) and converted to water levels at nearby Upper Phillips Creek (Blum et al., 2021). A paired *t*-test was used to compare net elevation change and surface accretion. The statistical significance of differences in net elevation change, accretion, and subsidence between forest zones was assessed using a one-way ANOVA in R Version (4.4.1) (R Core Team, 2023). Tukey’s HSD post hoc test was used to evaluate the significance of pairwise comparisons. All statistical tests were conducted with an α -level of 0.05 and $N=4$ for each forest zone.

Environmental Variables

Vegetation and Soils

Vegetation cover was measured in August 2024 within a $3 \text{ m} \times 3 \text{ m}$ quadrat, centered around each SET benchmark.

The percentage cover of each understory species was estimated via grid point-intercept at multiple height layers to account for vertical stratification; therefore, total cover could exceed 100%. This approach captured the complexity of vegetation structure in the plots. To simplify statistical analyses, species were subsequently grouped by functional plant types (i.e., upland grasses, shrubs, and marsh grasses).

Measurements of vegetation cover were supplemented with existing litterfall, porewater, and soil datasets collected in 2022 (Gedan et al., 2022, 2023). Litterfall was collected monthly using three replicate litter baskets, each measuring 25 cm × 25 cm, positioned randomly within each plot. Litter was oven-dried at 60 °C to a constant mass and then weighed to determine biomass contributions. In this study, only annual totals of litterfall biomass are presented. Soil porewater was sampled opportunistically from saturated soils following significant rainfall events to capture natural fluctuations in soil salinity. Samples were taken from the upper 20 cm of soil using a push-point sampler (MHE Products) and averaged to represent the salinity of the entire plot. Salinity was measured in situ using a handheld meter, following the methodology outlined by Sward et al. (2023). Surface soil samples were collected from each plot to assess soil conductivity and water content. These samples were air-dried and homogenized to ensure consistency. Conductivity was measured using a laboratory 5:1 dilution (He et al., 2012). Carbon and nitrogen contents of the surface soils were determined by LECO CN828 elemental analyzer.

Statistical Analyses

Stepwise regression analysis was conducted to identify the most significant predictors of net elevation change, accretion, and subsurface change. The stepwise regressions were run with and without transition zone elevation change metrics since the soil environmental data was not collected in the transition zone. In the model including all forest zones, the explanatory variables included elevation (NAVD88), shrub cover, *Phragmites* cover, and marsh grass cover. The model excluding the transition zone also included litterfall, soil moisture content, soil conductivity, and porewater salinity as variables. Starting with an initial intercept model, variables were added or removed based on Akaike Information Criterion (AIC) to balance model complexity with predictive accuracy. All analyses were performed in R version [4.4.1] (R Core Team, 2023).

Results

Elevation trends from the retreating upland forest indicate minimal net elevation change over the approximately 5-year record across the high, mid, and low forest, with rates of individual SETs ranging from -1.6 to 4.3 mm yr⁻¹

(Table 1). The transition zone gained elevation at the highest rate (2.9 mm yr⁻¹), followed by the mid (1.5 mm yr⁻¹), low (0.7 mm yr⁻¹), and high (-0.2 mm yr⁻¹) forest zones (Fig. 3). Net elevation change was significantly greater than zero in the mid ($t(3)=5.25$, $p=.007$), low ($t(3)=2.91$, $p=.031$), and transition zones ($t(3)=2.90$, $p=.0031$) (Suppl. Table 1) but not the high forest ($p=0.68$). Of the three zones with net elevation change rates greater than 0, only the transition zone was able to keep pace with local rates of sea level rise (4.1 mm yr⁻¹ for 2005–2020) (Blum et al., 2021).

Surface accretion rates were highly variable and ranged from 1.1 to 5.3 mm yr⁻¹ (Table 1) with the highest rates observed in the mid forest (4.5 mm yr⁻¹), though these were not significantly different from those in the high (2.9 mm yr⁻¹, $p=0.2$), low (2.6 mm yr⁻¹, $p=0.76$), and transition zones (2.8 mm yr⁻¹, $p=0.36$) (Table 2). Surface accretion rates did not differ from net elevation change in the transition zone ($p=0.88$) (Fig. 4). Conversely, surface accretion rates were significantly greater than net elevation change in the mid and low forest zones ($t(3)=23.54$, $p=0.0002$ and $t(3)=6.62$, $p=0.007$, respectively, Suppl. Table 2), while only marginally greater in the high forest ($t(3)=2.72$, $p=0.07$). This difference in surface accretion and net elevation gain highlights the importance of subsurface processes in driving elevation change rates in these zones.

Total subsurface change ranged from -5.8 to 2.6 mm yr⁻¹ (Table 1) with the greatest subsidence rates observed in the minimally salt-stressed high (-3.1 mm yr⁻¹) and mid forest (-3.0 mm yr⁻¹), although these were not significantly different from subsidence in the low forest (-1.9 mm yr⁻¹) (Table 2). Subsurface change in the transition zone was near zero (0.2 mm yr⁻¹) but highly variable (-2.4 to 2.6 mm yr⁻¹) (Fig. 3). Below the rooting zone (> 30 cm depth), the mid (1.6 mm yr⁻¹)

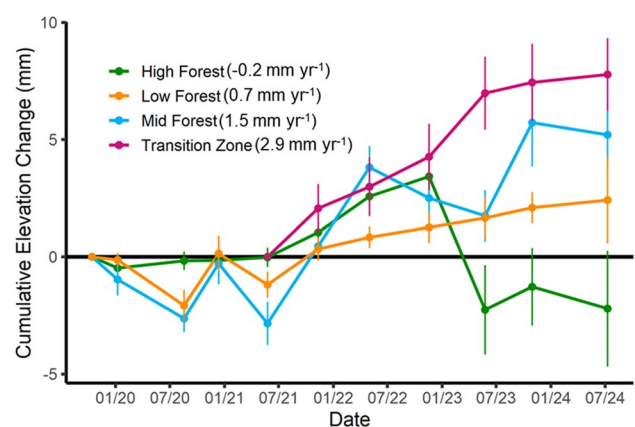
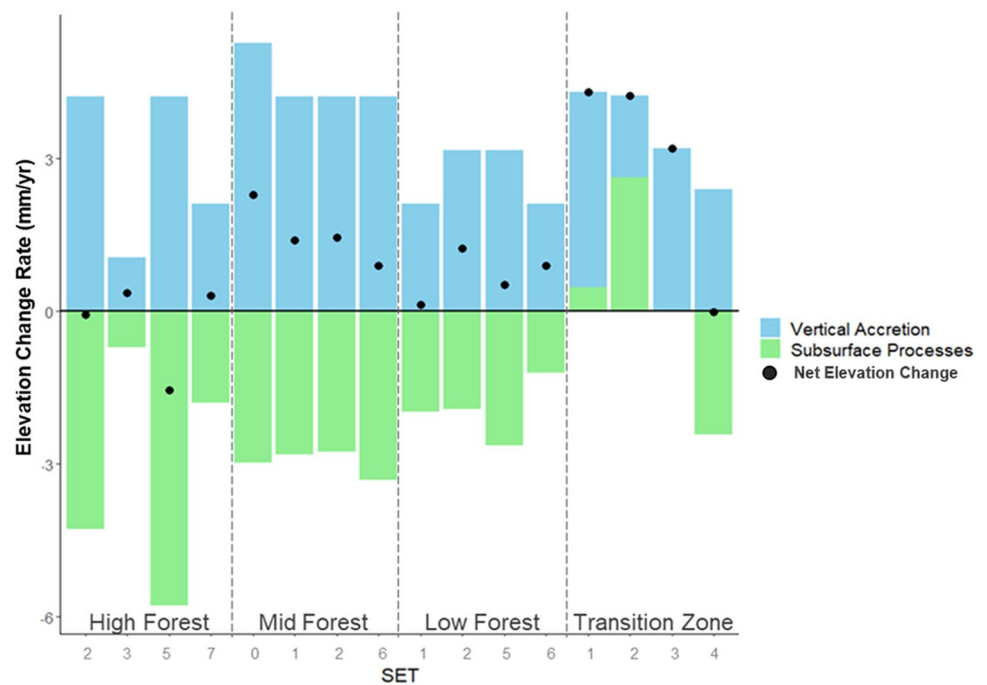


Fig. 3 Average cumulative elevation change (mm) of each forest zone ($n=4$) at Brownsville Forest shown in the high (green), mid (blue), and low (orange) forest. Error bars reflect standard error at each timestep. Average elevation change rates (mm yr⁻¹) calculated via linear regression are shown in the legend

Fig. 4 Rates of accretion (blue), subsidence (green), and net elevation change (black) for each individual SET. SET numbering scheme follows previously installed vegetation plots (Smith & Kirwan, 2021). Accretion rates are calculated as the thickness of material above the marker horizon divided by time since deployment. Net elevation change is calculated as the average change rate of each pin ($n = 36$) at a given SET. Subsidence is calculated as the difference between accretion and net elevation change. Negative values indicate subsidence (elevation loss) and positive values indicate subsurface expansion



and low forest (2.30 mm yr^{-1}) exhibited expansion, while the high forest (-0.6 mm yr^{-1}) and transition zone (-1.1 mm yr^{-1}) experienced subsidence. Within the rooting zone ($< 30 \text{ cm}$ depth), the high (-2.6 mm yr^{-1}), mid (-4.5 mm yr^{-1}), and low (-5.6 mm yr^{-1}) forest subsided, while only the transition zone (1.3 mm yr^{-1}) exhibited root zone expansion (Table 2).

The multiple linear regression analysis identified several significant interactions influencing net elevation change, accretion, and subsurface processes (Table 3). Net elevation change was strongly correlated with both elevation and percent shrub cover ($F(1,14) = 6.05$ and 3.0 , $p = .03$ and 0.11 , respectively). Net elevation change was positively correlated with the elevation of SET benchmarks in the high, mid, and low forest (Fig. 5a) but negatively correlated when the transition zone was included (Fig. 5b). In the high, mid, and low forest, shrub cover was positively correlated with net elevation change, though it was not a significant predictor across the entire forest health gradient (Fig. 5c, d). However, when analyzing surface accretion in isolation, shrub cover was the only significant predictor across all zones ($F(1, 10) = 8.92$, $p = .013$) (Table 3). While elevation was negatively correlated with accretion across all zones (Fig. 5), it was not statistically significant on its own ($F(1,10) = 0.67$, $p = 0.43$), but remained an important variable in our multiple linear model. Lastly, subsurface changes were positively correlated with elevation ($F(1,10) = 10.9$, $p = .008$) and soil conductivity, and negatively correlated with soil moisture and shrub cover (Figs. 5e and 6). However, only shrub cover significantly influenced subsurface change at all forest zones ($F(1,14) = 13.1$, $p = .003$) (Figs. 5f and 6).

Discussion

Future projections of marsh migration into retreating coastal forests rely on assumptions of a static topography, where the elevation of upland and wetland forests does not change through time (Linhoss et al., 2015; Molino et al., 2022; Osland et al., 2022; Schuerch et al., 2018). In contrast, detailed elevation-change measurements in coastal marshes (Blum et al., 2021; Saintilan et al., 2022), mangroves (Feher et al., 2024; Krauss et al., 2010; Rogers et al., 2005), and forested wetlands (Ardón et al., 2025; Krauss et al., 2024) indicate a dynamic topography where a variety of ecogeomorphic feedbacks influence rates of accretion, subsidence, and net elevation change (Cadot et al., 2014; Rogers et al., 2022; Lovelock et al., 2025). Although little work has been conducted in upland forests, observations from other coastal ecosystems indicate that vegetation type and inundation duration are primary drivers of elevation change through their impacts on surface accretion and belowground soil processes. In this study, we attempt to extend this conceptual framework to salt-affected upland forests.

Influence of Vegetation

In marshes, species composition directly influences rates of elevation change, largely due to differences in productivity (Calvo-Cubero et al., 2013) and sediment trapping capacity (Boorman et al., 1998; Rooth et al., 2003). Similarly, we found that vegetation type strongly influenced the rate of net elevation change across the retreating upland forest. Total shrub cover, dominated by the shrub *Morella cerifera*,

Table 2 Differences among forest and transition zones for surface accretion, net elevation change, and subsurface change expressed as mm yr^{-1} . Contribution of shallow root zone processes are calculated as the difference between total subsurface change and deep subsurface change (derived from shallow SET benchmarks). A negative subsurface value indicates subsidence, while a positive value indicates belowground expansion. Statistical differences between zones (determined via Tukey HSD) are shown in superscript

Forest zone	Surface accretion (mm yr^{-1})	Net elevation change (mm yr^{-1})	Subsurface change (mm yr^{-1})	Root zone change (mm yr^{-1})	Hydro-geologic zone change (mm yr^{-1})
High	2.9 ^a	-0.2 ^a	-3.1 ^a	-2.6 ^a	-0.6 ^a
Mid	4.5 ^a	1.5 ^a	-3.0 ^a	-4.5 ^a	1.6 ^b
Low	2.6 ^a	0.7 ^a	-5.6 ^a	-5.6 ^a	2.3 ^b
Transition	0.8 ^a	2.9 ^b	1.3 ^b	1.3 ^b	-1.1 ^a

was the most important variable controlling net elevation change, surface accretion, and subsurface expansion or contraction (Fig. 5). Over the study period, the mid-forest zone experienced significant increases in *M. cerifera* (Sward et al., 2023) and the greatest rates of surface accretion (Fig. 4). Litter inputs from densely packed shrubs may have contributed to the higher accretion rates observed in the mid-forest zone. In the subsurface, increased shrub cover resulted in greater elevation loss (Fig. 5e, f). *M. cerifera* encroachment at the study site is likely responding to increased light availability from tree stress and mortality (Sward et al., 2023). Therefore, subsurface subsidence in shrub-dominated plots may reflect root-zone collapse (e.g., Krauss et al., 2024), where root mortality and loss of turgor pressure occur even prior to tree mortality (Ding et al., 2023). The co-occurrence of shrub encroachment and tree root collapse in the mid-forest might explain why there were opposite responses in surface and subsurface contributions to elevation change. Alternatively, a nitrogen-fixing shrub, *M. cerifera*, increases soil nutrients, moisture, and winter temperatures (Thompson et al., 2017), potentially stimulating decomposition of soil organic matter in its rhizosphere (Smith et al., 2024).

Surprisingly, the invasive reed *Phragmites australis* did not significantly influence any of the elevation change metrics across forest zones (Figs. 4 and 5). Previous studies indicate that *Phragmites* contributes to rapid marsh elevation change through increased sediment trapping and high plant productivity (Rooth & Stevenson, 2000; Rooth et al., 2003). Accretion rates in *P. australis* stands may be 3–4 mm yr^{-1} greater than adjacent native marsh (Rooth et al., 2003), driven by above and belowground organic matter accumulation (Rooth & Stevenson, 2000). In contrast, we found that subsurface losses were highest in plots with the greatest *P. australis* cover (MF 6, TZ 4, LF 5; Table 2) (Fig. 4). Given its high abundance in stressed forest plots, the positive influence of *P. australis* on net elevation may be offset by subsidence related to tree mortality. Conversely, *P. australis* may contribute to subsurface losses due to decomposition, stimulated by increased standing stocks of nitrogen (Windham & Meyerson, 2003) and higher rhizosphere oxidation in *P. australis* stands (Windham & Lathrop, 1999). Additionally, its decomposition-resistant stems (Meyerson et al., 2000) may accumulate as standing dead material, taking up to 7 years for enhanced accretion to be reflected in the SET-MH record (Rooth et al., 2003). The full contribution of *Phragmites* to surface accretion is likely unrealized due to the thick thatch layer (pers. observation), short marker horizon record, and distance from tidal inputs.

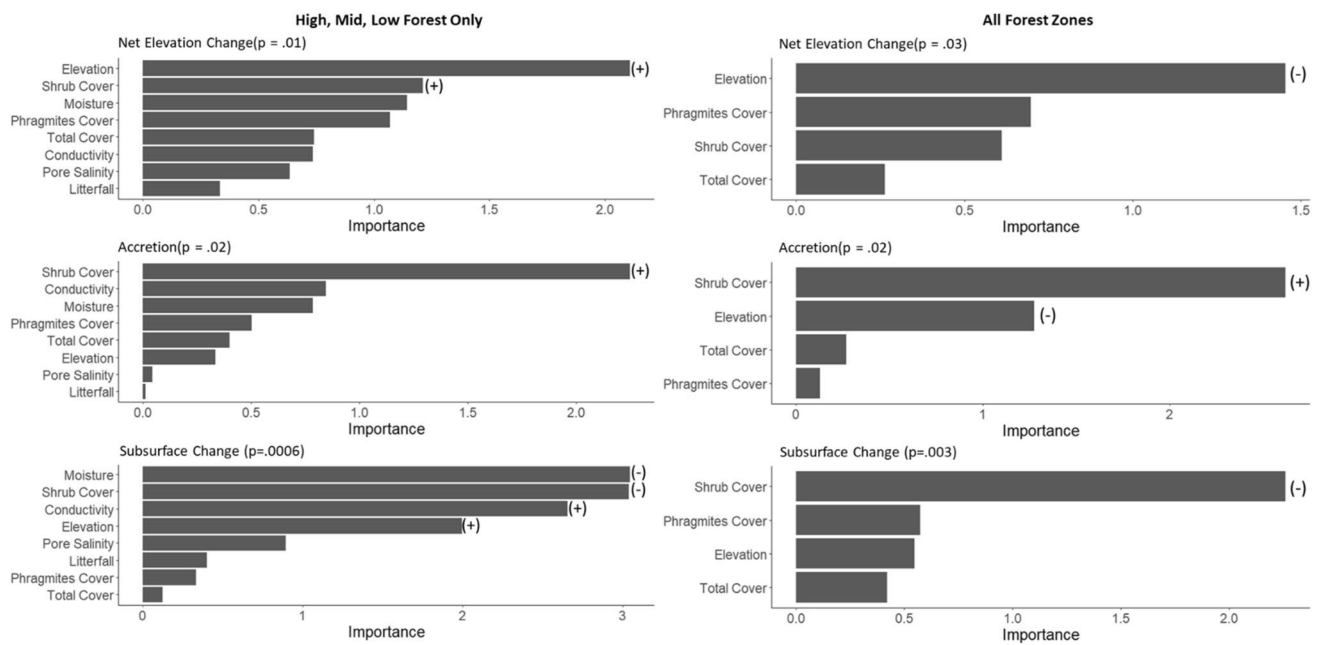


Fig. 5 Importance of each environmental variable shown. Importance calculated as the absolute value of the t -statistic for each model parameter in the full linear model. On the left shows only high, mid, and low forest while the plots on the right include the transition zone. The models including the transition zone are limited in their param-

eters due to data not being collected at these sites. The significant predictive variables in each multiple linear regression are denoted by a +/–, indicating the direction of the relationship. p -values of the multiple linear models containing only significant variables are shown in parentheses

Belowground Dynamics

Subsidence and elevation loss is common in coastal landscapes and has been documented in marshes (Keogh et al., 2021; Cahoon et al., 1995), mangroves (Cahoon & Lynch, 1997; Rogers et al., 2005), tidal freshwater forests (Krauss et al., 2024), and pocosins (Miller et al., 2021). Subsidence often occurs in the form of autocompaction (Cahoon et al., 1995; Kaye & Barghoorn, 1964; Saintilan et al., 2022), the physical compression of organic-rich peat beneath its own weight. Environmental stressors like salinization and inundation also drive subsidence across these ecosystems. For example, losses of 11 mm yr⁻¹ were documented in Honduran mangroves following a large hurricane (Cahoon et al., 2002) and a Mississippi River deltaic marsh lost 15 cm over several years in response to increased tidal inundation (Delaune et al., 1994). A study of Louisiana marshes indicates that over 60% of subsidence occurs in the top 5 m (Jankowski et al., 2017; Keogh & Tornqvist, 2019), while in Australia, it is largely confined to the organic-rich top 80 cm (Rogers & Saintilan, 2021).

Consistent with widespread observations of subsidence in other coastal ecosystems, shallow subsidence was observed in 13 of 16 SET locations across the studied

retreating coastal forest (Fig. 7). Although previous work links high subsidence rates exceeding 10 mm yr⁻¹ (Cahoon et al., 2002; Miller et al., 2021) to tree mortality, we found that total subsidence was greatest in the high and mid forest, where tree mortality is lower than in the low forest zone (Table 3). This is consistent with models showing that decreases in root biomass precede canopy stress and tree mortality in coastal forests (Ding et al., 2023). Root zone subsidence (< 30 cm) was greatest in the mid (–4.5 mm yr⁻¹) and low (–5.6 mm yr⁻¹) forest zones. In contrast with observations of tree mortality leading to elevation loss in other forested wetland ecosystems (Middleton & David, 2022; Cahoon et al., 2003), surface accretion (4.5 and 2.6 mm yr⁻¹) and expansion below the rooting zone (1.6 and 2.3 mm yr⁻¹, respectively) resulted in generally positive (0.7–1.5 mm yr⁻¹) net elevation change rates in the mid and low forest (Fig. 4). Similar rates of shallow subsidence have been documented in stressed tidal freshwater forests (–3.8 to –7.6 mm yr⁻¹), largely driven by losses in the rooting zone (Krauss et al., 2024; Stagg et al., 2016). Autocompaction is not a probable cause of forest subsidence, as the organic layer is thin (5–10 cm, Smith & Kirwan, 2021) and underlain by thick consolidated clay (Supp. Fig. 1). Instead, root zone subsidence was likely due to detrimental effects

Fig. 6 Relationship between shrub cover (%) and net elevation change (a), surface accretion (b), and total subsurface change (c) for each SET. Each panel shows a linear regression with 95% confidence intervals (gray shading)

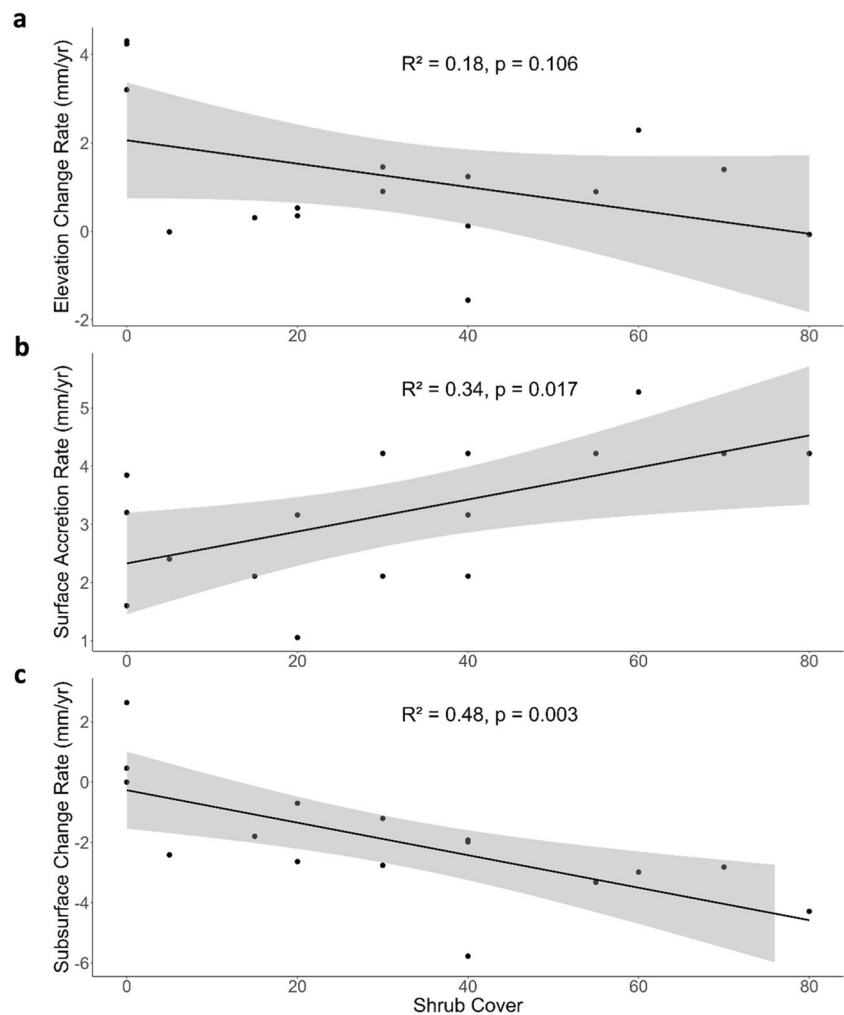


Table 3 p -Values from linear models comparing net elevation change, accretion, and subsurface change with environmental drivers (e.g., litter depth, soil moisture). p -values represent the significance of each environmental variable in predicting elevation change responses

	Litter depth	Annual Litterfall	Soil moisture	Soil conductivity	Soil carbon	Soil nitrogen	Porewater salinity	Elevation (NAVD88)	Shrub cover	Phragmites cover	Total cover
Elevation change	0.72	0.67	0.57	0.3	0.34	0.25	0.66	0.011	0.61	0.92	0.32
Accretion	0.11	0.71	0.92	0.49	0.96	0.58	0.30	0.43	0.013	0.69	0.66
Subsurface change	0.075	0.52	0.63	0.16	0.53	0.78	0.20	0.008	0.09	0.66	0.27

Significant values ($p < 0.1$) are highlighted in bold. Here we use $\alpha = 0.1$ to reduce the risk of missing potential correlations, as this was an exploratory analysis to identify patterns for further investigation

of inundation and salinity on mature Loblolly pine, which distribute up to 80% of salt-sensitive, fine roots within the top 20 cm of soil (Mou et al., 1995).

Surprisingly, given the small distance between zones, expansion below the rooting zone (> 30 cm) occurred only in the mid and low forest, while the high forest and transition zone subsided at depth (Table 2). Similar localized patterns of expansion and subsidence were observed in a South Carolina freshwater forest and were attributed to

small-scale differences in hydrology (Stagg et al., 2018). In the studied upland forest, expansion below the rooting zone (> 30 cm) in the mid and low forest may be due to the establishment of *M. cerifera* and *P. australis*, the roots of which frequently reach depths greater than 30 cm (Moore et al., 2012). Alternatively, the expansion may be due to the swelling of clay-rich soils in response to inundation with saline water. Clay matrix expansion by up to 10% has been documented in laboratory studies (Pedarla et al., 2012). Like

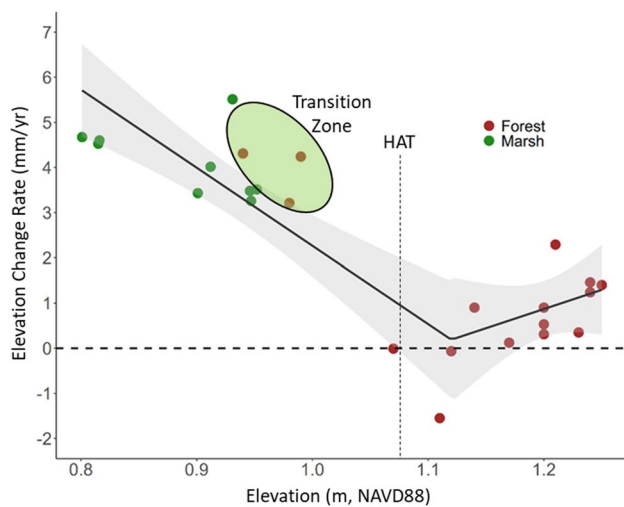


Fig. 7 Net elevation change for each forest SET (this study) compared to long-established SETs in the adjacent marsh (green dots, Blum et al., 2021). Transition zone SETs are highlighted in green circle. Best fit lines are shown in black and determined using the segmented package in R (Muggeo, 2008); 95% confidence intervals for each segmented regression are shown in gray

the high forest, the transition zone experienced shallow subsidence below the root zone. This subsidence may be at least partially due to autocompaction, as the depth of organic matter here is ~20 cm (Smith & Kirwan, 2021). Expansion in the root zone of 1.3 mm yr⁻¹ offset the subsidence below 30 cm, resulting in near-zero subsurface elevation change in the transition zone. This pattern of surface accretion and expansion in the shallow zone is similar to the elevation trends of the nearby high marsh, which from 2009 to 2024 gained elevation at a rate of 3.5 mm yr⁻¹, largely driven by surface accretion (3.1 mm yr⁻¹) and subsurface expansion (0.4 mm yr⁻¹) (Blum et al., 2021).

Influence of Elevation and Flooding Frequency

Across coastal ecosystems, rates of net elevation change are largely a function of elevation and flooding frequency (Kirwan & Megonigal, 2013). In coastal systems, lower elevations are more frequently flooded, resulting in increased potential for sediment deposition (Stumpf 1983, Friedrichs and Perry 2001) and enhanced exchange of nutrients, leading to greater above and belowground productivity (Morris et al., 2002; Kirwan & Guntenspergen, 2012). A recent global review of nearly 400 surface elevation tables identified rates of relative sea level rise and hydroperiod as the most important factors driving marsh net elevation change (Saintilan et al., 2022). We find that this relationship between elevation and elevation change generally holds true for the forest-to-marsh transition, where the lowest elevation transition zone SETs exhibit the greatest net elevation

change rates (Fig. 4), and elevation is a significant control of surface accretion rates across all forest zones (Fig. 5).

Unlike other coastal systems, such as marshes and mangroves, where mineral sediment deposition can reach several mm yr⁻¹ (Cahoon & Lynch, 1997; Kirwan & Megonigal, 2013; Krauss et al., 2024; Morris et al., 2002), the deposition of mineral sediment in coastal forests is unlikely. Due to the distance from the nearest tidal channel (> 500 m), any flood waters are likely to be depleted of sediment well before reaching the transition zone. This is supported by the high surface organic matter content of sediment cores in the transition zone and low forest (Supp. Fig. 1). Though the impacts of tidal inundation on sediment deposition are minimal in coastal upland forests, lower elevations are nonetheless associated with more frequent soil saturation (Nordio et al., 2024; Callahan et al., 2017) due to their proximity to shallow groundwater and propensity to collect surface rainwater. Like the adjacent high marsh, the waterlogged, anaerobic soils at the forest-marsh ecotone may limit decomposition (Piaszczyk et al., 2022; Janousek et al., 2017) and facilitate the establishment of productive marsh species (Walters et al., 2021; Shaw et al., 2022; Smith et al., 2013) that in turn increase belowground biomass relative to that of the nearby low and mid forests.

Elevation change rates that increase with increasing elevation between the low forest and high forest are in contrast with elevation change rates that decrease with increasing elevation in long-measured SETs in the adjacent salt marsh (Blum et al., 2021) (Fig. 7). By synthesizing upland forest and salt marsh elevation changes, we find that the relationship between elevation and net elevation change shifts from negative to positive at an elevation very close to the highest astronomical tide (HAT, 1.07 m; NOAA Station ID 8631044), suggesting a switch in the processes driving net elevation change associated with large changes in flooding regimes. For example, the low forest zone is flooded by approximately 33 tides per year, while the transition zone is flooded by approximately 140 tides per year (Supp. Table 3). This difference in flooding regime may correspond to a threshold in which additional flooding leads to enhanced soil organic matter accumulation in wetlands but enhanced subsidence in forest and shrub-dominated zones.

The observed trend of increasing elevation change with high elevations may lead to the development of hummock-hollow microtopography, a feature common in global wetlands (Brinson & Christian, 1999; Diamond et al., 2020) such as tidal freshwater forested wetlands (Krauss et al., 2024) and pocosins (Brinson, 1991) where small elevation variations drive dramatic differences in inundation, soil moisture, and salinity (Wallis & Raulings, 2011). Once established, this microtopography is often self-reinforcing, as higher, drier hummocks experience reduced hydraulic stress and are therefore more productive (Conner et al.,

2022), while shallow subsidence offsets accretion more in hollows than in hummocks (Hsueh et al., 2016; Krauss et al., 2024). A similar pattern of hummock-hollow formation and maintenance may be occurring in the studied coastal upland forest, where elevation gains are directly related to initial surface elevation (Fig. 7). This is consistent with elevation variance that is greater within individual plots and forest zones than between forest zones (pers. observation). Additionally, both rates of subsidence and accretion were more strongly controlled by elevation and vegetation type than by forest zone, suggesting the importance of localized, small-scale controls on elevation change rates. In tidal freshwater forested wetlands, maintenance of the “hummock-hollow” microtopography is an important factor for the resistance of transition to oligohaline marsh (Krauss et al., 2024; Noe et al., 2013). Similarly, poorly explained variance in the elevation of the marsh-forest boundary across the Chesapeake Bay region has been attributed to microtopography (Molino et al., 2022 JGR), and inundation relief provided by hummocks may slow the transition from forest to marsh.

Conclusions and Implications for Coastal Transgression

We measured relatively small changes in the elevation of a retreating upland forest soil (Fig. 3), which at least tentatively supports assumptions of a static upland topography in models of coastal transgression (Molino et al., 2022; Osland et al., 2022; Schuerch et al., 2018). This finding is in contrast to observations of extensive subsidence and net decreases in the elevation of other coastal forests following mortality events (e.g., Miller et al., 2021). Large storms, which were largely absent during the study period, may explain this deviation. For example, elevation losses of $\sim 1.1 \text{ cm yr}^{-1}$ in mangroves (Cahoon et al., 2003) and 10 cm yr^{-1} in forested wetlands (Middleton & David, 2022) have been documented using SET-MH following hurricanes. In each instance, tree-mortality-driven subsidence resulted in rapid conversion to marsh, whereas forest to marsh conversion is gradual at our study site (Fig. 1). Conversely, storms may increase accretion through a pulse of leaf litter and woody debris, wrack deposits, or inorganic sediment. For example, Hurricane Hugo deposited up to 3 cm of mud into a South Carolina pine forest (Gardner et al., 1991). In this study, we found no evidence of either a significant subsidence or depositional event, though we were not able to directly measure storm effects during the study period. Therefore, over short time scales and in the absence of large storms, assumptions of a static upland topography may be valid.

Observations of slow net changes in soil elevation belie more dynamic responses to vegetation and inundation

expressed in the individual components of elevation change (i.e., surface accretion and subsurface expansion or subsidence). Even in the absence of a major storm event, transgression of vegetation types is occurring rapidly in the studied forest, including the encroachment of woody shrubs (Sward et al., 2023) and invasive *Phragmites* (Shaw et al., 2022), and reductions in the biomass of even the high forest zones (Chen & Kirwan, 2022). While we measured subsidence rates of up to 5 mm yr^{-1} in most forest zones that may correspond with declines in forest health, subsidence was largely compensated for by surface accretion (Fig. 4). Finally, we observed net surface accretion paired with belowground expansion in the transition zone, such that net elevation change rates in developing marshes are nearly equivalent to adjacent, fully developed salt marshes (Fig. 7). Together, these observations suggest that the elevation of forest soils is relatively stable, but that elevation change rates and their components respond rapidly to marsh migration into retreating coastal forests.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s12237-025-01624-y>.

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Author Contribution Tyler C. Messerschmidt: conceptualization, data curation, formal analysis, methodology, visualization, writing—original draft, writing—review and editing; Keryn B. Gedan: conceptualization, resources, writing—review and editing; Matthew L. Kirwan: conceptualization, funding acquisition, methodology, project administration, resources, writing—original draft, writing—review and editing.

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Data Availability Data are published and archived by the Virginia Coast Long Term Ecological Research in the following data publication: Messerschmidt, T.C. and M.L. Kirwan. 2025. Surface Elevation (SET) Data for Brownsville Forest at the Virginia Coastal Reserve, 2019–2025 ver 4. Environmental Data Initiative. <https://doi.org/10.6073/pasta/f1356e575381d53b6b1b936cbb7eaf17>.

Declarations

Competing Interests The authors declare no competing interests.

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