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ARTICLE

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Restoration and resilience to sea level rise of a salt marsh affected by dieback events

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Abstract

The frequency of salt marsh dieback events has increased over the last 25 years with unknown consequences to the resilience of the ecosystem to accelerated sea level rise (SLR). Salt marsh ecosystems impacted by sudden vegetation dieback events were previously thought to recover naturally within a few months to years. In this study, we used a 13-year collection of remotely sensed imagery to provide evidence that approximately 14% of total marsh area has not revegetated 10 years after a dieback event in Charleston, SC. Dieback onset coincided with a severe drought in 2012, as indicated by the Palmer drought stress index. A second dieback event occurred in 2016 after a historic flood influenced by Hurricane Joaquin in 2015. Unvegetated zones reached nearly 30% of the total marsh area in 2017. We used a light detection and ranging-derived digital elevation model to determine that most affected areas were associated with lower elevation zones in the interior of the marsh. Further, restoration by grass planting was effective, with pilot-scale restored plots having greater aboveground biomass than reference sites after two years of transplanting. A positive outcome indicated that the stressors that caused the dieback are no longer present. Despite that, many affected areas have not recovered naturally, even though they are located within the typical elevation range of healthy marshes. A mechanistic modeling approach was used to assess the effects of vegetation dieback on salt marsh resilience to SLR. Predictions indicate that a highly productive restored marsh (2000 g m^{-2} $year^{-1}$) would persist at a moderate SLR rate of 60 cm in 100 years, whereas a nonrestored mudflat would lose all its elevation capital after 100 years. Thus, rapid restoration of marsh dieback is critical to avoid further degradation. Also, failure to incorporate the increasing frequency and intensity of extreme climatic events that trigger irreversible marsh diebacks underestimates salt marsh vulnerability to climate change. Finally, at an elevated SLR rate of 122 cm in 100 years, which is most likely an extreme climate change scenario, even highly productive ecosystems augmented by sediment placement would not keep pace with SLR. Thus,

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climate change mitigation actions are also urgently needed to preserve present-day marsh ecosystems.

KEYWORDS
dieback, drought, elevation, resilience, salt marsh, sea level rise, urban ecosystem

INTRODUCTION

The resilience of salt marsh ecosystems to climate change has been a source of intense scientific debate, often focused on the response of marshes to accelerated sea level rise (SLR) (Donnelly & Bertness, 2001; Kirwan et al., 2010; Kirwan & Megonigal, 2013; Kirwan, Temmerman, et al., 2016; Langston et al., 2021; Morris et al., 2002; Raposa et al., 2017; Rogers et al., 2019; Schuerch et al., 2018; Weston, 2014). Ultimately, the capacity of present-day marshes to persist under accelerated SLR will depend on the ability to increase in surface elevation at a rate that at least matches that of SLR (Morris et al., 2002; Reed, 1995). Salt marsh surface elevation is determined by the balance between sediment deposition and erosion, organic matter accumulation, and land subsidence (Kirwan & Megonigal, 2013; Morris et al., 2016; Mudd et al., 2009). Plant productivity is a key variable modulating marsh vertical accretion not only by accruing organic matter from belowground biomass, but also by facilitating the deposition of suspended matter during tidal inundation (Baustian et al., 2012; Mudd et al., 2009, 2010). Moreover, relative marsh elevation, or elevation capital (Cahoon & Guntenspergen, 2010), the distance of the wetland surface to the lowest elevation at which plants can survive, largely determines the vulnerability to SLR (Morris et al., 2021).

In the last 25 years, sudden salt marsh dieback events, in which extensive areas of vegetation die off in short periods of time, have been increasing in frequency with little understanding of the implications for the long-term resilience of marshes to SLR (Alber et al., 2008; McKee et al., 2004). Sudden salt marsh dieback events have been associated with drought and suggested to be caused by the interaction of multiple physical, chemical, and biological stressors in the USA Southeast and Gulf of Mexico (Alber et al., 2008). Specifically, elevated porewater salinity, acidification of air-exposed sediment, fungal infections, and increased snail herbivory have all been reported in previous events (Alber et al., 2008; Hughes et al., 2012; McKee et al., 2004; Silliman et al., 2005). In many cases, areas affected by sudden salt marsh dieback have naturally recovered within a few months to years (Alber et al., 2008; Lindstedt et al., 2006; Linthurst & Seneca, 1980; Marsh et al., 2016). However, there are also reports of marshes that have not recovered to date (Alber et al., 2008; Baustian et al., 2012; Marsh et al., 2016), with most of these cases representing low-elevation zones that experienced an irreversible regime shift to ponds, requiring active restoration for recovery (Baustian et al., 2012; McKee et al., 2004; Ogburn & Alber, 2006; Schepers et al., 2020). In contrast, little is known about the causes and fate of dieback events causing long-term vegetation loss in areas that transitioned from salt marsh to mudflat, as well as their vulnerability to accelerated SLR. Salt marsh trajectories (i.e., the pathway of ecosystem function over time) have been previously used to predict and assess the persistence of marshes under climate change and other types of ecosystem perturbation or to study ecosystem recovery after restoration or marsh creation (e.g., Fleeger et al., 2020; Morgan & Short, 2002; Wasson et al., 2019; Wu et al., 2022). However, to the best of our knowledge, there are limited studies characterizing the trajectory and recovery of salt marshes affected by dieback events over decadal timescales (e.g., Marsh et al., 2016). Thus, the dynamic or change in ecosystem function over time, along with the ecological forces driving the trajectory of dieback events, require further study over longer timescales.

At a global scale, the area of marshland that is projected to drown as a consequence of accelerated SLR has been suggested to be offset by salt marsh migration onto higher land (Kirwan, Temmerman, et al., 2016). Nevertheless, even if that holds true, urban salt marshes are highly vulnerable to accelerated SLR because the built infrastructure associated with coastal cities acts as a physical barrier preventing marsh migration (Kirwan & Gedan, 2019; Schuerch et al., 2018). If no actions are taken to conserve present-day urban salt marshes, the ecosystem services they provide will be lost. For example, salt marshes provide a natural defense to storm surge for coastal populations and cities (Barbier et al., 2011; Temmerman et al., 2013). Up to 41% of the world's population lives on the coastline, with 630 million people living below projected flood levels for 2100, and lower income populations from underrepresented minority communities are more likely to live in flood-prone zones (Chakraborty et al., 2014; Eisenman et al., 2007; Kulp & Strauss, 2019; Martínez et al., 2007). Conservation of urban marshes represents an effective climate change adaptation strategy with increasing relevance over time, as the frequency and intensity of tropical storms and hurricanes are already increasing (Brown et al., 2019; Paerl et al., 2019). To the best of our knowledge, little information is available on the vulnerability or resilience of urban salt marshes to accelerated SLR. Thus, there is a great need to investigate the potential response of urban marshes to various climate change, ecological, and management scenarios in order to support adaptive management of the coastline.

This study was initiated to assess the vulnerability and restoration potential of salt marshes in Charleston, SC, motivated by members of the Ashleyville Neighborhood Association in West Ashley, a densely populated low-lying area that borders on urban marshes. Salt marshes in the area serve as essential habitat for many fish and shellfish species (e.g., red drum, sharks, and oysters), and the National Fish and Wildlife Foundation (NFWF)-funded Coastal Resilience Assessment prioritized the West Ashley area for restoration, because its habitats are generally in relatively poor condition, wildlife is exposed to high stress, and human assets show a very high vulnerability to storms along with SLR (Crist et al., 2019). Thus, we used this case study to expand the current knowledge on the trajectory of salt marsh dieback events and their effects on salt marsh resilience to climate-change-induced accelerated rate of SLR. Objectives of this study were to: (1) characterize the onset, dynamic, and trajectory of salt marsh dieback events in Charleston, SC; (2) correlate the onset of dieback with climatological and salt marsh geophysical properties; (3) evaluate the effectiveness of salt marsh restoration by grass planting in an unvegetated marsh affected by a dieback event; and finally, (4) model the resilience of dieback-affected salt marshes to SLR under contrasting restoration scenarios.

MATERIALS AND METHODS

Study area

The study was conducted in *Spartina alterniflora*-dominated salt marshes located in West Ashley, Charleston, SC. The watershed west of the Ashley River has great cultural and historical relevance since it contains Maryville and Charles Towne Landing. Maryville and Charles Towne Landing are the sites of a Gullah Geechee community established after the American Civil War and the first permanent English settlement in the Carolinas in 1670, respectively. The dieback area was first discovered by members of the local Gullah Geechee community in a salt marsh adjacent to Maryville (Appendix S1: Figure S1).

In order to relate the expansion of dieback areas to drought and extreme precipitation, we retrieved monthly Palmer drought stress index (PDSI) values from the South Carolina Southern Division (from January 2009 to August 2021) using NOAA's National Centers for Environmental Information database (https://www.ncei.noaa.gov/access/monitoring/historical-palmers/) and monthly precipitation values in the Charleston area, SC, (ThreadEx) from NOAA's Online Weather Data (NOWData, https://www.weather.gov/wrh/Climate?wfo=chs). The South Carolina Southern Division includes the 10 southernmost counties in South Carolina, including the Charleston county (https://www.cpc.ncep.noaa.gov/products/analysis_monitoring/regional_monitoring/CLIM_DIVS/south_carolina.gif). The PDSI is a widely used drought index based on the supply and demand of a water balance equation of precipitation, evapotranspiration, soil moisture loss and recharge, and runoff that has been used to study the effects of water limitation on agricultural and ecological systems (Palmer, 1965).

Characterization of marsh dieback trajectories using multispectral, high-resolution imagery

Multispectral imagery was acquired from three sources: satellite imagery from the PlanetScope constellation (Planet Team, 2019) and MAXAR's WorldView-3 satellite and aerial imagery from the US Department of Agriculture's (USDA) National Agriculture Imagery Program (NAIP). The NAIP dataset consists of four-band (blue, green, red, and near infrared [NIR]), 1-m resolution imagery (http://www.fsa.usda.gov/programs-and-services/aerial-photography/imagery-programs/naip-imagery). The PlanetScope constellation acquires daily 3-m resolution imagery, using a four-band (blue, green, red, and NIR) sensor. We used the Planet's Surface Reflectance product, processed to top-of-atmosphere reflectance and atmospherically corrected to surface reflectance (Planet Team, 2019). PlanetScope images were acquired during low tide and on cloudless days. The MAXAR WorldView-3 acquires four-band (blue, green, red, and NIR), 30-cm resolution images. The used View-Ready WorldView-3 image was retrieved during low tide on a cloudless day, was radiometrically and sensor corrected, and atmospherically corrected to surface reflectance (MAXAR, 2021). The complete imagery used in this study consisted of nine images retrieved from 2009 to 2021 (further details in Appendix S1: Table S1).

Two vegetation indices were calculated from all acquired images: the normalized difference vegetation index (NDVI) and the modified soil-adjusted vegetation index 2 (MSAVI2):

$$\mathrm{NDVI} = \frac{(\mathrm{NIR} - \mathrm{Red})}{(\mathrm{NIR} + \mathrm{Red})}. \quad (1)$$

$$\mathrm{MSAVI2} = \frac{2\mathrm{NIR} + 1 - \sqrt{(2\mathrm{NIR} + 1)^2 - 8(\mathrm{NIR} - \mathrm{Red})}}{2}. \quad (2)$$

Total unvegetated area was estimated for all NAIP images and the 2016 WorldView-3, and 2021 PlanetScope satellite images. We visually inspected the coordinates of vegetated and unvegetated points (median observations per year: 218) using high-resolution images at a scale of 1:500 to 1:1000 in QGIS 3.16.5 (QGIS.org, 2021). Raster values of the four bands and NDVI were extracted from the pixels containing the vegetated and unvegetated points. Random forest models were used to estimate the probability that a pixel was unvegetated using the four bands and NDVI as explanatory variables. Out-of-bag estimate of error rate was lower than 3.5% for all models. Pixels with a greater than 75% probability of being unvegetated were defined as such, while pixels with a lower than 25% likelihood of being unvegetated were defined as vegetated areas. This analysis was performed using the raster and randomForest R packages in the R v. 4.1.0 environment (Hijmans & van Etten, 2012; Liaw & Wiener, 2002; R Core Team, 2021).

In order to investigate if unvegetated areas were preferentially associated with creek bank or midmarsh zones, we drew all creek bank edges from the study area (Appendix S1: Figure S1). Edges were drawn using the 2019 NAIP aerial image and the corrected 2017 LIDAR-derived DEM at a scale of 1:250 to 1:500 in QGIS 3.16.5 (QGIS.org, 2021). We generated a raster measuring the distance of each pixel to the closest creek bank edge using the "Join attributes by nearest" algorithm in QGIS with a resolution of 1 m. Based on the well-known natural zonation of *S. alterniflora* (Mendelssohn & Morris, 2002), with taller plants found closer to tidal creeks, areas within 7.5 m of a tidal creek were defined as "creek bank marsh," whereas areas >7.5 m distal to creeks were defined as "midmarsh" (Valiela et al., 1978). The 7.5-m threshold characterizing the ecosystem state change was defined by visually inspecting an aboveground biomass versus distance from tidal creek biplot (Appendix S1: Figure S2; method for aboveground biomass estimation below). The creek bank against midmarsh zones were only defined by distance to tidal creek, without consideration of marsh elevation.

Field sampling

Field sampling was performed during spring and summer of 2021. Two field campaigns were performed on May 27–28 and July 20–26 of 2021. We sampled 60 and 119 locations, respectively, in May and July (total = 179 points). Field sampling was performed nonrandomly by establishing transects across gradients of *S. alterniflora* biomass from the creek bank to the interior of the marsh (Appendix S1: Figure S1). The dieback area adjacent to Maryville was oversampled due to our interest in it as a potential restoration site. At all locations ($n = 179$), a high-accuracy real-time kinematic (RTK) survey was conducted using a Trimble GNSS GPS receiver (Trimble, Sunnyvale, CA, USA). Elevation was expressed in centimeter in the North American Vertical Datum of 1988 (NAVD88) using the GEOID 18 model. Instrument accuracy is of 2-cm root mean square error (RMSE), horizontal and vertical. During the July field sampling, *S. alterniflora* shoot density and height were measured using a 0.25-m^2 quadrat at 82 points sampled in parallel for RTK surface elevation. Shoot density was measured by counting all *S. alterniflora* stems inside the 0.25-m^2 quadrat. Shoot height was measured randomly from six stems per quadrat. In order to estimate aboveground biomass, 71 *S. alterniflora* shoots of known height were harvested from 37 sampling points across contrasting *S. alterniflora* growth zones. Harvested shoots were oven-dried at 60°C for one week, and their shoot height and dry mass were used to construct a biomass allometric equation. For all sampling points, biomass was calculated for individual shoots using the constructed allometric equation and mean shoot biomass ($n = 6$) multiplied by shoot density to calculate aboveground biomass at the plot level (in grams per square meter).

Acquisition of LIDAR-derived DEM

A light detection and ranging (LIDAR)-derived bare-earth digital elevation model (DEM) constructed from LIDAR aerial acquisition conducted from February 25, 2017 to March 9, 2017 in Charleston County was used in this study (South Carolina DNR, 2018). LIDAR acquisition involved Axis GeoSpatial (Easton, MD, USA) operating a Cessna 206H outfitted with a Riegl LMS-Q1560 dual-channel laser scanner system at 2043 m above ground level. Scanner pulse rate was 800 kHz with an unlimited number of returns per pulse. The DEM vertical datum used was NAVD88 Geoid 12B and was projected to the NAD83 (2011) South Carolina State Plane at a pixel resolution of 1 m. System parameters for LIDAR acquisition are described in Appendix S1: Table S2 (further details can be found in South Carolina DNR, 2018).

Correction of the LIDAR-derived DEM

High-accuracy RTK elevations were used to correct the bare-earth LIDAR-derived DEM. Error of the

LIDAR-derived DEM was defined as: $Z_{\mathrm{LIDAR}} - Z_{\mathrm{RTK}}$ for each RTK observation, where Z is NAVD88 elevation. We used a multispectral PlanetScope satellite image retrieved 5 days before LIDAR acquisition (February 20, 2017) to correct the DEM based on the known relation between LIDAR error and *S. alterniflora* height (Hladik & Alber, 2012). The 179 RTK observations were randomly split for model training and testing in a 7:3 ratio, respectively. Error of the LIDAR-derived DEM was fitted to a multiple lineal regression model using automated model selection based on Akaike information criterion (AIC) as implemented in the glmulti R function (method = "h") (Calcagno & de Mazancourt, 2010). We used the four bands of the PlanetScope image (red, green, blue, and NIR), two vegetation indices (NDVI and MSAVI2), and the original LIDAR-derived DEM elevations as potential explanatory variables. The DEM was corrected by solving the multiple linear regression for the RTK-observed elevation. Performance of the best model was tested based on RMSE calculated as: $\mathrm{RMSE} = \sqrt{\sum (\mathrm{Observed}_i - \mathrm{Predicted}_i)^2 / n}$, where $\mathrm{Observed}_i$ is the ith observation of the RTK elevation and $\mathrm{Predicted}_i$ is the ith observation of the LIDAR-derived DEM (Maune et al., 2007). We calculated the RMSE for the test and train datasets of both the corrected and uncorrected LIDAR-derived DEM.

Estimation of 2021 aboveground biomass

Aboveground biomass was estimated for the total study area based on in situ biomass observations made during the July 2021 field campaign. Aboveground biomass (in grams per square meter) estimated from 82 sampling points (see *Field sampling*) was randomly split into training and testing datasets (80% and 20%, respectively). A random forest model was constructed using the corrected 2017 LIDAR-derived DEM, MSAVI2, NDVI, and the green, red, and NIR bands from a PlanetScope image (acquired in January 08, 2021) as explanatory variables. The model grew 5000 trees, and in each split, two variables were randomly sampled as candidates. The performance of the model was assessed by calculating the RMSE of the training and testing datasets (as explained in *Correction of the LIDAR-derived DEM*). A raster containing the predicted aboveground biomass for the whole study area was constructed using the random forest model. Aboveground biomass was set to 0 in all unvegetated areas (as inferred in the *Characterization of marsh dieback trajectories using multispectral high-resolution imagery*). The model was built using the raster and randomForest R packages in the R v. 4.1.0 environment (Hijmans & van Etten, 2012; Liaw & Wiener, 2002; R Core Team, 2021).

Pilot grass planting restoration

A pilot restoration project led by the South Carolina Department of Natural Resources (SC-DNR) was initiated in the Maryville marsh in 2019 to test the effectiveness of marsh restoration in long-term affected ecosystems impacted by dieback events. The restoration effort consisted of planting *S. alterniflora* using seedlings grown from local sources. Botanical seeds were annually collected in October through December each year (2019, 2020, and 2021) from 11 unique sites, all located in Charleston County. After collection, seeds were kept at 4°C for eight weeks, after which they were germinated in planting trays filled with freshwater and substrate. After 6–7 months, seedlings ~55.0 cm tall were transplanted into the dieback area from May to August. When transplanting, a spacing of 30.5 cm was used. Restoration was conducted in six individual patches with dimensions ranging from 48.6 to 183.3 m^2. Plots from transplants performed in 2019, 2020, and 2021 were studied. In order to assess the effectiveness of the pilot restoration project, aboveground biomass was estimated during the July field campaign in 5 and 12 quadrats (0.25 m^2) from restored plots planted in 2020 and 2021, respectively. Aboveground biomass was calculated based on shoot height and density as previously described (see *Field sampling*). A one-way ANOVA with restored versus natural marsh plots across the entire study area (grouped in either creek bank or midmarsh) was used to explain differences in *S. alterniflora* aboveground biomass. Further, to have a complete and balanced statistical design at the end of the growing season, a second field assessment was performed on September 22, 2021. In this assessment, 10 quadrats per plot were sampled in plots planted in 2019, 2020, and 2021, as well as two patches of adjacent reference marsh (i.e., natural salt marsh patches that have not experienced dieback). For the September assessment, a one-way ANOVA was used to explain differences in *S. alterniflora* aboveground biomass between restored and reference plots. In both July and September evaluations, in the case of finding statistical significance in the ANOVA ($p < 0.05$), a Tukey's post hoc test using the *emmeans* R package was performed (Lenth, 2016).

Modeling of marsh elevation under primary production, management, and SLR scenarios

The Marsh Equilibrium Model (MEM) v. 9.01 (Morris et al., 2002, 2021) was employed to assess the resilience of the Charleston salt marsh to accelerated SLR under different scenarios of (1) *S. alterniflora* primary productivity, (2) human intervention (none, grass planting after dieback events, or thin layer placement [TLP]), and (3) rates of

SLR (regional NOAA's intermediate low of 60 cm in 100 years and intermediate of 122 cm in 100 years). The rationale for utilizing the MEM was to assess the long-term effect (100 years) of marsh vegetation loss with and without restoration in Charleston, SC. The MEM is a one-dimensional mechanistic model that projects change in marsh elevation with SLR as a function of primary production and sediment accretion. It was modified to simulate TLP of mineral sediment by allowing the user to specify periodic sediment additions to a specified thickness, net of compaction. The model was coded in Visual Basic for Applications behind an Excel user interface.

We simulated two rates of SLR, based on regional NOAA's intermediate low (60 cm in 100 years) and intermediate (122 cm in 100 years) projections by 2100 (Sweet et al., 2017). The intermediate low scenario has a 73% and 96% probability of being exceeded by 2100 under RCP4.5 and RCP8.5 scenarios, respectively, while the intermediate scenario has a 3% and 17% probability of being exceeded by 2100 under RCP4.5 and RCP8.5 scenarios, respectively (Sweet et al., 2017). Starting SLR rate was set as 3.4 mm year^{-1}, as calculated for the Charleston Harbor by Morris and Renken (2020). The elevation range of marsh vegetation was assumed to be within 25 cm below mean sea level (MSL) and 30 cm above mean high water (MHW), as characterized in South Carolina salt marshes (Morris et al., 2013). Tidal range, MSL, and MHW during the study period (January 2009–August 2021) were 76 cm, 4 cm, and 80 cm NAVD88, respectively. Optimal elevation was determined as 5 cm above MSL based on a biplot of elevation and aboveground biomass from the study area (see *S. alterniflora primary production*). Four different optimal values of aboveground primary productivity were modeled: 0, 500, 1000, and 2000 g m^{-2} year^{-1} to assess the effects of plant productivity and restoration by grass planting on marsh resilience to SLR. The 0 g m^{-2} year^{-1} simulation was used to simulate a marsh affected by dieback without restoration. The median elevation of all dieback pixels in 2021 was set as the initial elevation: 47.8 cm NAVD88. Total suspended matter (TSM) was estimated as 23 mg L^{-1} from remotely sensed GlobColour OLCIA imagery (http://hermes.acri.fr/) as the average of monthly values from 2016 to 2021. TSM was extracted from the closest pixels to the southern border of our study area in the Charleston Harbor. Finally, two restoration treatments were added: The first one modeled a marsh affected by dieback events every 20 years, followed by immediate restoration by grass planting. In this case, planted grasses were simulated to fully recover after 2 years as observed in our pilot study (see *Grass transplanting is a successful restoration method in West Ashley, SC*). The second treatment modeled 5 cm of TLP every 25 years to simulate a more aggressive restoration action. TLP consists of spreading sediment onto the marsh platform to increase surface elevation (Ford et al., 1999). After TLP, marsh vegetation was simulated to be completely lost and fully recovered 10 years after disturbance. A conservative 10 years until recovery was used, even though previous studies have shown no vegetation loss after thin-layer sediment applications of between 5 and 7 cm in the US Southeast (Davis et al., 2022).

RESULTS

Salt marsh dieback trajectory

We observed an expansion of large unvegetated areas in 2013 that coincided with a severe drought in 2011–2012 (Figure 1). Charleston, SC, experienced its most severe drought over the last 100 years in 2011–2012, concurrent with an over 200% increase of unvegetated marsh area at our study site (64 ha, Figure 2a). The South Carolina Southern NOAA division, which contains the city of Charleston, experienced an extreme drought (PDSI < -4.0) from June of 2011 to April of 2012 (Figure 1). A second large expansion of unvegetated area was observed in 2016–2017 that was not drought-induced. The onset of the 2016–2017 dieback event coincided with the historic flood of the city of Charleston caused by record precipitation associated with the tropical cyclone Joaquin in October 2015 (Figure 1). In 2019, the vegetated area of West Ashley had recovered up to 2013 levels, which have persisted since. Based on field inspection and aerial imagery, we found that the affected salt marsh ecosystem transitioned from vegetated salt marsh to mudflat after both dieback events.

The dieback of 2013 caused a 300% expansion of unvegetated area in the midmarsh zone (Figure 2b). In the 2016 event, the midmarsh and creek bank experienced a 50% and 33% increase of unvegetated area, respectively (Figure 2b,c). Moreover, from 2011 to 2017, there was a constant loss of about 50% of creek bank salt marsh every two years (Figure 2c). A LIDAR-derived DEM acquired in 2017 was corrected for the studied area, improving the error estimation by approximately 30% on both training and testing datasets (RMSE after correction: less than or equal to 8 cm in both train/test datasets, more detailed information in Appendix S1: Supplementary Text S2 and Figure S3). On average, creek bank marshes were located at lower elevation terrain, and thus at greater nondimensional depth (Appendix S1: Figure S4). Greater nondimensional depth is associated with longer hydroperiods (Alizad et al., 2016; Morris et al., 2002). Furthermore, using the corrected 2017 LIDAR-derived DEM, we observed that both creek bank and midmarsh unvegetated areas were associated with lower elevation terrain (Figure 3; Appendix S1: Figure S4).

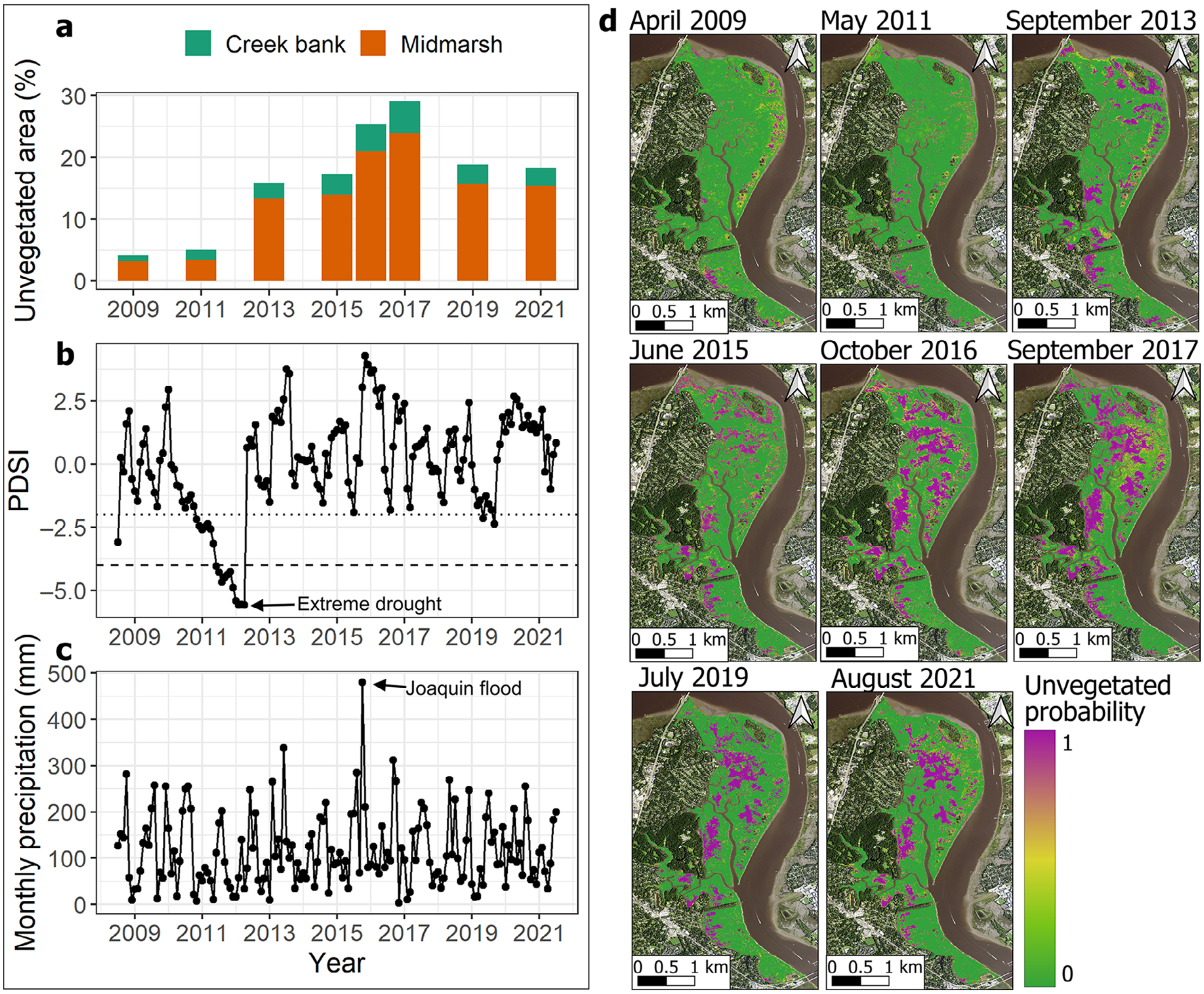

FIGURE 1 Dieback trajectory of the saltmarsh located in West Ashley, Charleston, SC. (a) Percentage of unvegetated marsh area over time. Total area is partitioned into creek bank or midmarsh categories based on areas closer or further than 7.5 m to its closest creek bank, respectively. (b) Palmer drought severity index (PDSI) from the South Carolina Southern Division over time. Values lower than −2 represent moderate drought (dotted line), while values lower than −4 represent extreme drought conditions (dashed line). (c) Monthly precipitation from the Charleston, SC, area retrieved from NOAA's Online Weather Data. (d) Maps showing the probability of an area being unvegetated. Based on individual random forest models per image.

The difference in median elevations between vegetated and unvegetated areas across years ranged from 13.3 to 27.4 cm and 6.3 to 14.3 cm in creek bank and midmarsh zones, respectively. Since the difference in elevation could have been caused by land subsidence after the dieback event, we repeated the analysis with an uncalibrated 2009 LIDAR-derived DEM from the same study area (South Carolina DNR, 2009), finding similar results (Appendix S1: Figure S5). Because the 2009 DEM was acquired before the dieback events, both current unvegetated and vegetated marshes have a similar elevation bias. In the 2009 DEM, difference in median elevation between vegetated and unvegetated areas ranged from 4.4 to 23.0 cm and 2.4 to 16.3 cm in creek bank and midmarsh zones, respectively. The results suggest that lower elevation persisted in the unvegetated areas prior to the dieback event. Intriguingly, the elevation of most unvegetated areas is located within MSL and MHW, where healthy vegetated marshes are predicted based on the MEM (Morris et al., 2002, 2021).

S. alterniflora primary production

A site-specific allometric equation was constructed to quantify the primary production of the Charleston salt marsh during July of 2021 (Figure 4a,c). The random forest model used to estimate plant aboveground biomass was based on the 2017 calibrated LIDAR-derived DEM, vegetation

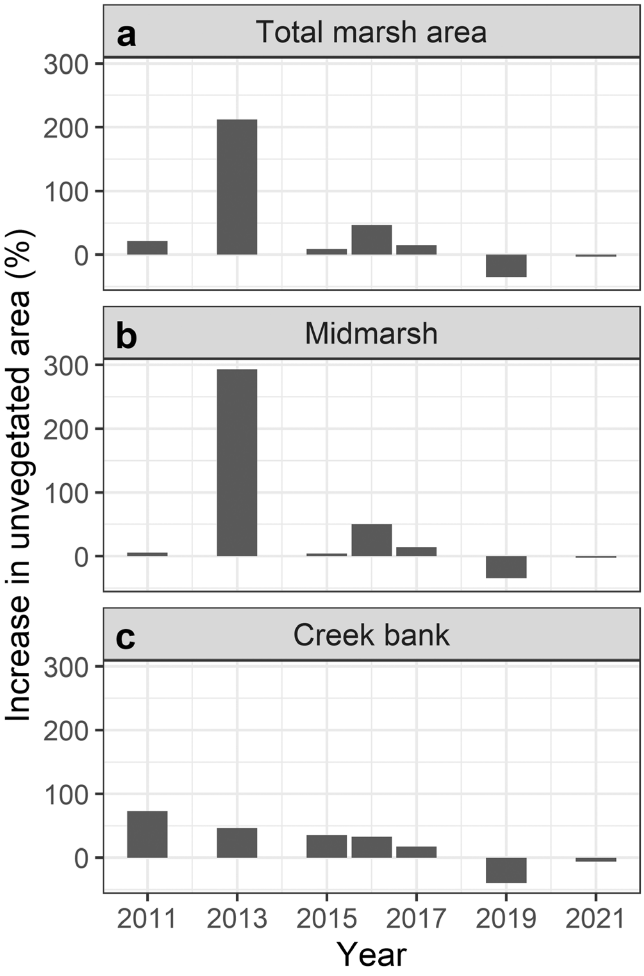

FIGURE 2 Dieback events were associated with an expansion of unvegetated area in the midmarsh zone. Barplots showing the percent increase in unvegetated areas since the last evaluation (1 or 2 years). Analysis was performed for the total marsh, midmarsh, and creek bank zones.

indices, and satellite bands from an August 2021 color NIR PlanetScope image. The $\mathrm{RMSE}_{\mathrm{test}}$ of the model was 268.1 g m^{-2}. Median predicted aboveground biomass was 623.1 g m^{-2}, ranging from 0 to 1944 g m^{-2} (Figure 4b). Biomass was within the expected range for Charleston's latitude (predicted end-of-season productivity based on Kirwan et al., 2009: 574.8 g m^{-2} year^{-1}). Aboveground biomass was closely associated with elevation and distance to tidal creeks (Figure 4e,f; Appendix S1: Figure S2), with greater aboveground biomass observed in lower elevation areas within closer proximity to tidal creeks (Figure 4e,f).

Grass transplanting is a successful restoration method in West Ashley, SC

Guided by the SC-DNR, seedlings from botanical seeds of *S. alterniflora* were collected in Charleston, SC, propagated under greenhouse conditions, and transplanted by community volunteers in July of 2019, 2020, and 2021 in a West Ashley dieback area that has been unvegetated since 2013 (Figure 5; Appendix S1: Figure S1). Aboveground biomass was determined during the July 2021 field trip for the 2020 and 2021 transplanted plots compared to reference plots of natural midmarsh and creek bank grass across the entire study area (Figure 5). Aboveground biomass of the naturally vegetated midmarsh and the 2020 transplanted plot were nonstatistically different and had an average ± SE of 527 ± 43 g m^{-2} and 700 ± 185 g m^{-2}, respectively. We conducted a complete sampling design in September of 2021 to study the 2019, 2020, and 2021 restored and two patches of reference marsh adjacent to the restoration site. We found that restored plots planted in 2019 had greater aboveground biomass than reference marshes (Figure 5), whereas biomass in the plot transplanted in 2020 was nonstatistically different from that of reference marshes (Figure 5). Aboveground biomass was 1726 ± 207 g m^{-2} and 1290 ± 168 g m^{-2} (mean ± SE) for the 2019 and 2020 transplanted plots, respectively. Average RTK elevations of the two reference patches were 41.7 and 60.4 cm NAVD88, while elevation of the three restored plots ranged from 36.1 to 45.9 NAVD88 cm. No statistical relationship ($p > 0.05$) was observed between aboveground biomass and elevation for the reference and restored plots. Thus, statistical differences in biomass between restored and reference plots are the result of restoration action and not marsh elevation.

Salt marsh resilience to SLR under contrasting restoration scenarios

The model output indicated that mudflat areas (*S. alterniflora* primary productivity = 0 g m^{-2} year^{-1}) rapidly lose elevation capital at a SLR rate of 60 cm in 100 years; decreasing in elevation by a total of 38 cm after the 100 years (Figure 6). Vegetated salt marsh ecosystems with an optimal primary productivity of 500, 1000, and 2000 g m^{-2} year^{-1} would persist under the 60 cm in 100-year SLR scenario, but losing 37.4, 33.0, and 24.5 cm of elevation capital after 100 years, respectively (Figure 6). In general, marshes restored by immediate grass planting after dieback events did not differ from nonimpacted marshes (Figure 6). However, at a SLR rate of 122 cm in 100 years, even a healthy vegetated salt marsh with an optimal primary productivity of 2000 g m^{-2} year^{-1} would collapse in less than 75 years (Figure 6). Simulating an external supply of sediment (i.e., TLP) at a rate of 5 cm of inorganic sediment every 25 years would maintain the elevation capital of all simulated salt marsh ecosystems at 60 cm in 100-year SLR scenario. However, TLP of a highly

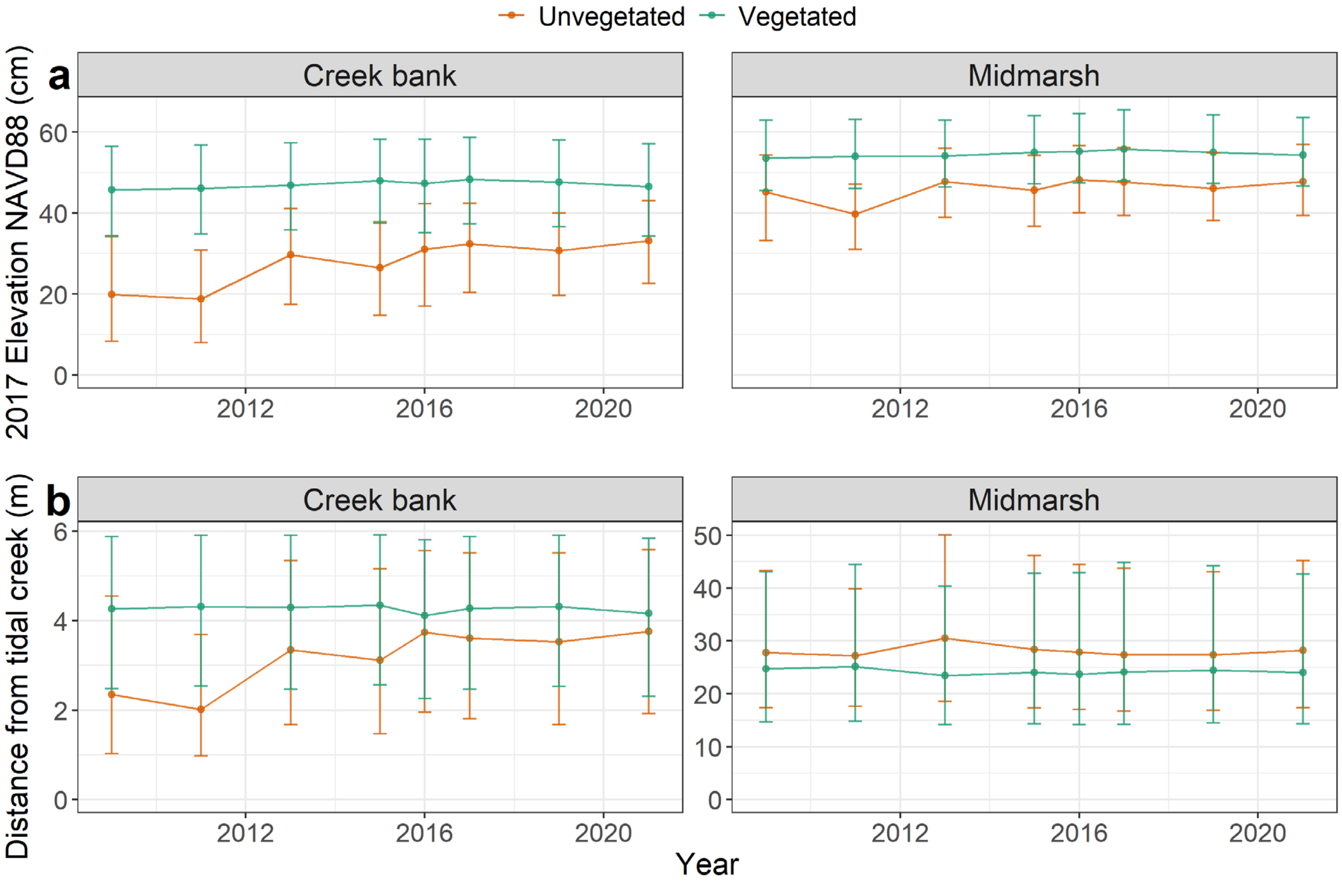

FIGURE 3 Characterization of unvegetated and vegetated areas within the West Ashley, SC, salt marsh. (a) Median ± interquartile range of the field-corrected 2017 light detection and ranging derived digital elevation model of vegetated and unvegetated areas over time. (b) Median ± interquartile range of the distance from the closest tidal creek of vegetated and unvegetated areas over time. NAVD88, North American Vertical Datum of 1988.

productive salt marsh (2000 g m^{-2} $year^{-1}$) under the extreme 122 cm in 100-year SLR scenario would result in the persistence of the ecosystem for only 100 years (Figure 6).

DISCUSSION

The resilience of salt marshes to accelerated SLR and extreme climatic events

Extreme drought events have been linked to sudden vegetation diebacks in several salt marshes in the USA Southeast and Gulf of Mexico over the last 25 years (Alber et al., 2008; Lindstedt et al., 2006; McKee et al., 2004). The drought-induced mechanism of sudden marsh dieback has been associated with the interaction and cascading effects of multiple stressors, such as extreme porewater salinity, sediment acidification due to the oxidation of air-exposed sulfur minerals into sulfuric acid, infection by fungal pathogens, and an exponential increase of periwinkle snail herbivory (Alber et al., 2008; Hughes et al., 2012; McKee et al., 2004; Silliman et al., 2005). More recently, extreme precipitation from catastrophic tropical cyclones was suggested to trigger marsh vegetation diebacks by either intrusion of salt water into brackish/freshwater marshes and/or by extending periods of tidal inundation, which in turn exceed the physiological tolerance of marsh plants (Marsh et al., 2016; Ramsey et al., 2012; Stagg et al., 2021). The co-occurrence of vegetation loss with extreme drought and flooding events in our study provides substantial evidence that marsh vegetation dieback is attributed to extreme climatic events in the USA Southeast.

Salt marsh dieback events are often described as transient phenomena (Alber et al., 2008). In contrast, we show that marsh dieback triggered by extreme climatic events can be long-lasting, with an approximate 14% loss of total vegetated area after 10 years. To the best of our knowledge, this is the first time a decadal timeframe is used to characterize a long-term salt marsh dieback event

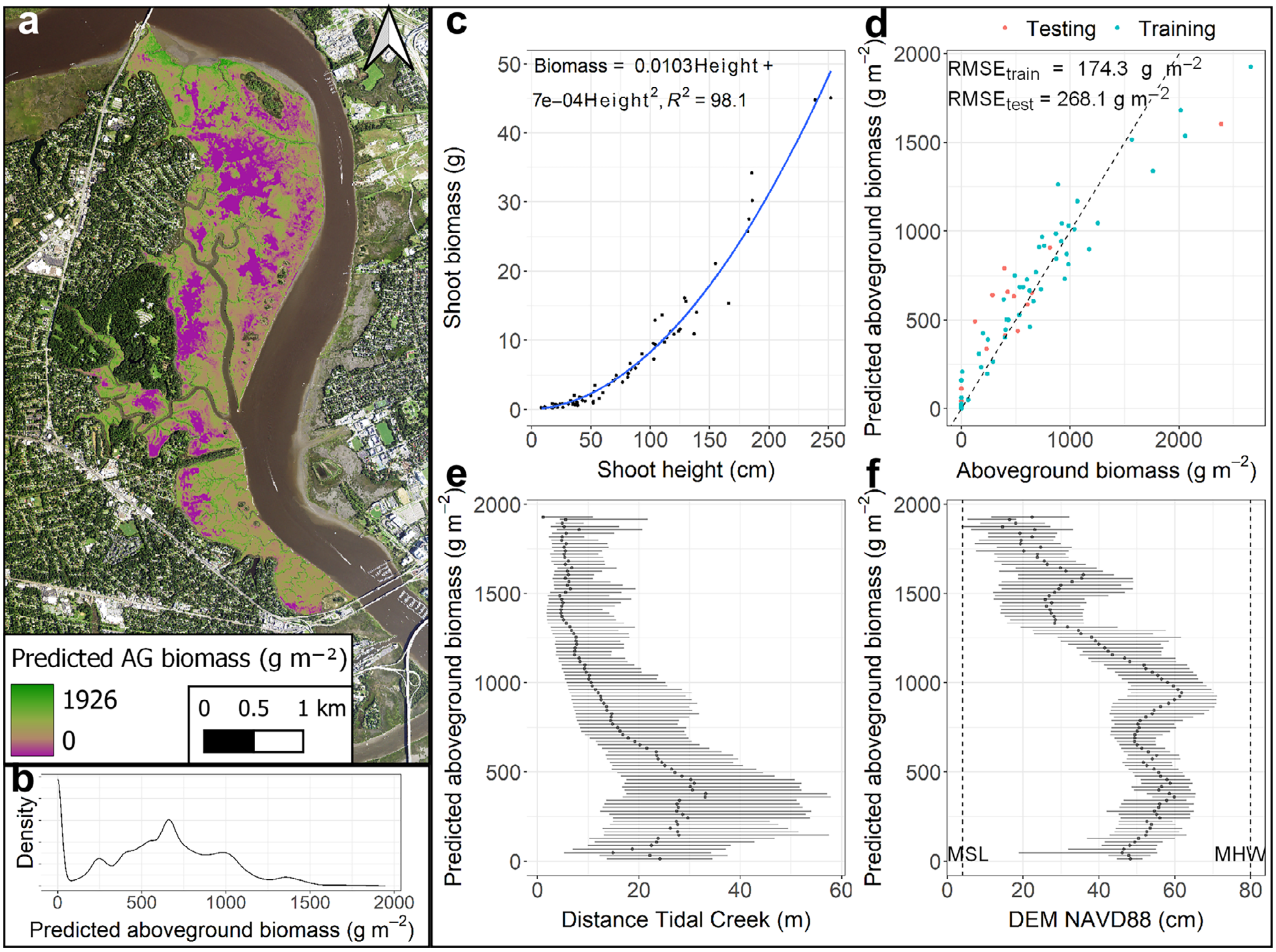

FIGURE 4 Aboveground biomass (AGB) estimation of the West Ashley, SC, salt marsh during July 2021. (a) Map of AGB based on a random forest model using the 2017 elevation digital elevation model, vegetation indices, and bands from a PlanetScope four-band image. (b) Density plot of the predicted AGB values. (c) Allometric biomass equation based on shoot biomass. (d) Goodness of fit of the random forest model in the testing and training datasets. Relation between predicted AGB and distance from tidal creek (e), and the corrected 2017 elevation model (f). Points and error bars represent AGB median and interquartile range on equidistant increments, respectively. Elevations (North American Vertical Datum of 1988 [NAVD88]) of the study area mean sea level (MSL) and mean high water (MHW) are shown as dashed lines in panel (f). DEM, digital elevation model; RMSE, root mean square error.

induced by an extreme climatic event. It is intriguing that the surface elevation of the dieback areas, or elevation capital, remains within the range of a hypothetically healthy marsh (Morris et al., 2002). Furthermore, we provide empirical evidence that the marsh may be restored by grass planting, suggesting that the stressors that caused dieback may be overcome, despite the fact that the majority of dieback areas have not revegetated naturally. Potential explanations for slow or no recovery require further research and may be associated with limited seed dispersion and viability, poor vegetative reproduction, as well as disturbed marsh hydrology and geomorphology (i.e., tidal creek erosion and sedimentation limiting previously normal tidal flow).

The increasing frequency and intensity of severe climatic events due to climate change is likely to further develop marsh diebacks similar to those we observed in Charleston (Brown et al., 2019; Dai, 2013; Paerl et al., 2019). Long-term vegetation loss, as shown in this study and previous work, increases the vulnerability of salt marsh ecosystems to SLR (Baustian et al., 2012; Mudd et al., 2009, 2010; Figure 6). Thus, the effects of extreme climatic events on marsh vegetation must be incorporated into assessments of the resilience of the ecosystem to climate change. Current models designed to project marsh resilience to accelerated SLR often assume that the ecosystem collapses due to the progressive loss of elevation relative to MSL (Kirwan et al., 2010; Kirwan, Walters, et al., 2016; Morris et al., 2002). In other words,

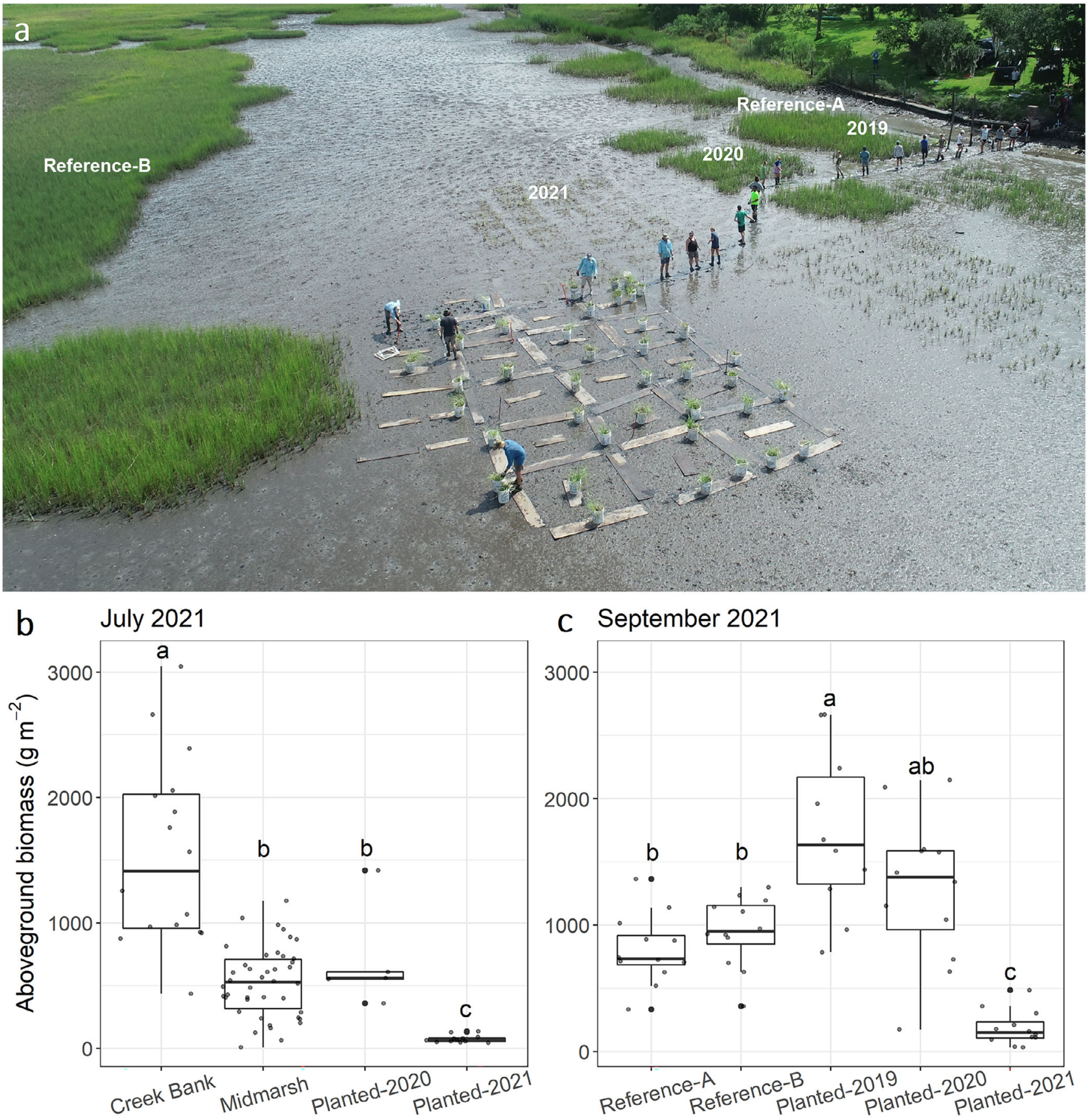

FIGURE 5 Grass planting is an effective restoration practice in Maryville, SC. (a) Drone image from the restoration site. Plots sampled are labeled on top in white. Photo acquired by Gary Sundin (South Carolina Department of Natural Resources Shellfish Research Section) on July 21 of 2021. (b) Boxplot representing the interquartile range of aboveground biomass from sampled creek bank and midmarsh plots across the entire study area, and 2020 and 2021 restored plots during the July 2021 field campaign. (c) Boxplot representing the interquartile range of aboveground biomass from 10 quadrats sampled from the two reference sites, and the 2019, 2020, and 2021 restored plot during September of 2021. Different letters indicate statistically significant differences ($p < 0.05$) between treatments according to a Tukey test using *emmeans* R package (Lenth, 2016).

the marsh drowns with the loss of elevation capital. Failing to incorporate the increasing frequency and intensity of extreme climatic events that trigger irreversible marsh diebacks will lead to an underestimation of the vulnerability of salt marshes to climate change. For example, our modeling approach forecasted that after 100 years under a moderate climate change scenario, the elevation of a highly productive marsh

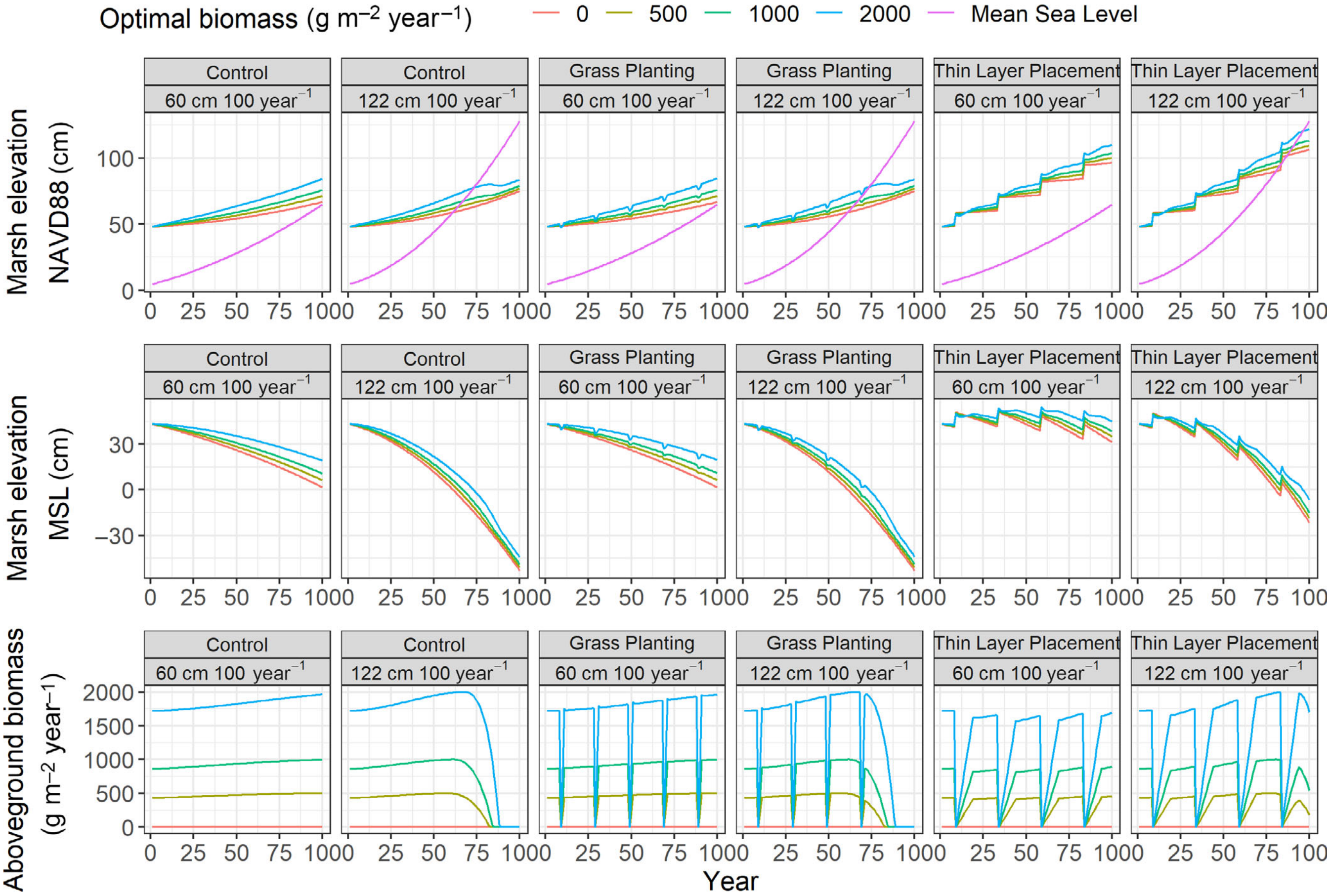

FIGURE 6 Modeled salt marsh resilience under different scenarios of sea level rise, human intervention, and plant primary production using the Marsh Equilibrium Model (Morris et al., 2002). Modeled optimal biomass values ranged from 0 g m^{-2} $year^{-1}$ (dieback and mudflat) to highly productive salt marshes of 2000 g m^{-2} $year^{-1}$. Three restoration scenarios were modeled: (1) Control without dieback events and restoration actions (control), (2) A marsh impacted by dieback events every 20 years with restoration by grass planting and planted vegetation fully mature after two years (grass planting), and (3) thin layer placement (TLP) performed every 25 years by applying 5 cm of inorganic sediment. In the TLP treatment, vegetation is considered to fully recover after 10 years of disturbance. The two sea level rise scenarios modeled were based on regional NOAA's intermediate low and intermediate of 60 and 122 cm in 100 years, respectively (Sweet et al., 2017). Shown are marsh elevation over time projected to NAVD88 and mean sea level (MSL) and modeled salt marsh aboveground biomass over time. NAVD88, North American Vertical Datum of 1988.

exceeds that of an unvegetated dieback marsh by 17.5 cm (see below for further discussion).

Call for restoration of urban marshes

Urban salt marsh ecosystems are more vulnerable than nonurban marshes due to their limited migration space (Schuerch et al., 2018). At the same time, the value of the ecosystem services they provide is remarkably high due to their close proximity to heavily populated areas. Thus, their conservation and restoration have been proposed as an effective adaptation strategy to climate change, with special emphasis on their ability to attenuate storm surge and coastal erosion (Geedicke et al., 2018; Temmerman et al., 2013). Hybrid solutions, in which engineered structures are combined with coastal wetlands to protect built infrastructure from storm surge, were shown to be more effective than engineered defenses alone (Temmerman et al., 2013; Zhu et al., 2020). However, despite the enhanced value and vulnerability of urban salt marshes, these ecosystems remain understudied in comparison to their nonurban counterparts (Alldred et al., 2020).

Our results corroborate evidence from nonurban marshes to show that marsh grass planting is an effective restoration practice in an urban marsh that experienced dieback (Baustian et al., 2012; Linthurst & Seneca, 1980; Ogburn & Alber, 2006). The multiple stressors that lead to sudden vegetation dieback have yet to be completely revealed (Evans et al., 2021). The ability of unvegetated marshes to respond positively to grass planting is likely dependent on the strength of the marsh platform and whether it is located within the elevation range of a stable marsh domain (sensu Morris et al., 2002), along with sufficient sediment supply from the surrounding estuary (Baustian et al., 2012; Liu et al., 2021; Webb et al., 1995; Wilsey et al., 1992). As shown in our modeling approach and by previous work, plant primary production is a key variable modulating marsh accretion (Morris et al., 2002; Mudd et al., 2009, 2010; Silliman et al., 2019). Thus, rapid restoration of marsh dieback is crucial to avoid further degradation, especially in stressed urban salt marshes. The urgency increases when taking into consideration that dieback areas in West Ashley and other studied locations are already associated with lower elevation terrain (Alber et al., 2008; Marsh et al., 2016). Moreover, unvegetated marshes are prone to fill adjacent tidal channels and to experience marsh platform erosion, which is exacerbated during low tide rainfall events (Ganju et al., 2017; Temmerman et al., 2012; Torres et al., 2004). Thus, rapid restoration by grass planting in marshes that have experienced long-term dieback would not only reinstate the marsh's ability to accrete sediment but also prevent further disturbance of marsh hydrology. The effectiveness of any restoration effort over the long term must include careful consideration of elevation capital and hydrology. Recent studies show that drainage density and distance from a channel network increase marsh resilience to future drought-induced dieback events (Liu et al., 2020).

Adaptive management of salt marsh ecosystems under climate change

Grass transplanting is an effective restoration practice that bolsters high rates of sediment accretion under conservative SLR scenarios (Silliman et al., 2019; Temmink et al., 2020). However, our model results indicate that even highly productive marshes will likely drown under more extreme climate change scenarios. Our results are supported by long-term modeling performed in Georgia, USA, which predicts an 88% loss of initial marsh area by 2160 (Langston et al., 2021). Field verification of remotely sensed data and site-specific monitoring are critical for the assessment of salt marsh resilience to accelerated SLR in urban environments. If grass planting alone is not sufficient to maintain marsh elevation under accelerated SLR, natural accretion could be augmented by the direct application of sediment. This can be performed by either beneficial use of dredging material or by sediment redistribution within the same marsh system from edge marsh erosion, or other processes (Ford et al., 1999; Hopkinson et al., 2018). Beneficial use of dredged material by TLP has been successfully applied to salt marshes in the US Southeast without vegetation loss if performed in small increments of less than 10 cm and by low-impact sediment application methods (Davis et al., 2022; Ford et al., 1999). However, to restore large areas of impaired marsh ecosystem, a source of dredged material must be present in the area, which represents a common limitation of the TLP approach. Moreover, further research is warranted on the ecosystem response to TLP in salt marshes that are prone to dieback events, which may lead to unintended consequences. However, even under successful TLP conditions in our modeling exercise, periodic placement of dredged material was not sufficient for marsh elevation to keep pace with SLR under extreme climate change scenarios (Figure 6). Thus, constant monitoring, rapid restoration, and climate change mitigation actions are all urgently needed to preserve highly valuable present-day marsh ecosystems close to urban environments.

AUTHOR CONTRIBUTIONS

J. L. Rolando, J. E. Kostka, M. Hodges, N. Williams, J. Robinson, A. George, J. Carr Jr., and J. Morris designed the research. J. L. Rolando, M. Hodges, K. D. Garcia, G. Krueger, N. Williams, J. Robinson, and J. E. Kostka performed field work. J. L. Rolando and M. Hodges conducted laboratory work. J. L. Rolando analyzed the data with input from J. Morris and J. E. Kostka. J. L. Rolando wrote the first draft of the manuscript with inputs from J. E. Kostka, and all authors equally contributed to revisions and gave approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors declare that they have no conflict of interest.

DATA AVAILABILITY STATEMENT

Remotely sensed imagery was retrieved from https://earthexplorer.usgs.gov/, https://planet.com/explorer/, and https://discover.maxar.com/. Details for retrieval, including query IDs, of the remotely sensed images used can be found in Appendix S1: Supplementary Text S1 and Table S1. R code, field data, and resulting maps (Kostka Lab, 2022) are available from Zenodo: https://doi.org/10.5281/zenodo.6980907.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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