RESEARCH ARTICLE

Title: Severe drought events trigger long-term salt marsh dieback in the Southeastern and Gulf of Mexico USA

Running Title: Severe drought triggers long-term marsh dieback

Authors: Rolando, JL[a]; Hodges, M[b]; Garcia, KD[a].; Krueger, G[c].; Williams, N.[d]; Carr Jr., J.[e]; Robinson[d,f]; J.; George, A.[g]; Morris; J[h]; Kostka, JE[a,i,j].

Affiliations:

[a]Georgia Institute of Technology, School of Biological Sciences, Atlanta, GA 30332, USA

[b]South Carolina Department of Natural Resources, Charleston, SC 29412, USA

[c]Ashland University, Department of Biology and Toxicology, Ashland, OH 44805, USA

[d]Robinson Design Engineers, Charleston, SC 29407, USA

[e]Ashleyville Neighborhood Association, Charleston, SC 29407, USA

[f]College of Charleston, Marine Biology Graduate Program, Charleston, SC 29424, USA

[g]South Carolina Aquarium, Conservation Department, Charleston, SC 29401, USA

[h]University of South Carolina, Belle Baruch Institute for Marine & Coastal Sciences, Columbia, SC 29208, USA

[i]Georgia Institute of Technology, School of Earth and Atmospheric Sciences, Atlanta, GA 30332, USA

[j]Center for Microbial Dynamics and Infection, Georgia Institute of Technology, Atlanta, GA 30332, USA

Corresponding author:

Kostka, JE. e-mail: joel.kostka@biology.gatech.edu

Abstract

The frequency of salt marsh dieback events has been increasing in the last 25 years with unknown consequences to the resilience of the ecosystem to accelerated sea level rise (SLR). Sudden dieback events triggered by extreme climatic events were previously thought to recover naturally within a few months to years. We provide evidence that about 15% of total marsh area has not revegetated 10- and 20-years after dieback events in two case studies in the Southeast and Gulf of Mexico USA, respectively. In both case studies, dieback onset coincided with severe drought events. In West Ashley, SC, dieback areas were associated with lower elevation zones in the interior of the marsh. Restoration by grass planting was shown to be effective in the South Carolina urban marsh, indicating that the stressors that caused dieback may be overcome despite that most of dieback areas have not recovered naturally. A mechanistic modelling approach was used to predict the resilience of the South Carolina salt marsh to accelerated SLR under different scenarios of primary productivity, ecological management, and rates of SLR. Predictions indicate that a highly productive restored marsh (2000 g m^{-2} y^{-1}) would persist at a moderate SLR rate of 60 cm 100 y^{-1}, whereas a non-restored mudflat would rapidly lose all of its elevation capital. Further, at an elevated SLR rate of 122 cm 100 y^{-1}, most likely an extreme climate change scenario, even highly productive ecosystems augmented by sediment placement would not keep pace with SLR. Thus, both climate change adaptation and mitigation actions are urgently needed to preserve highly valuable present-day marsh ecosystems close to urban environments.

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

1. 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 and Bertness, 2001; Kirwan et al., 2010; Kirwan et al., 2016a; Kirwan and Megonigal, 2013; Morris et al., 2002; Langston et al., 2021; 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 sequestration, and land subsidence (Kirwan and Megonigal, 2013; Morris et al., 2016; Mudd et al., 2009). Plant primary productivity is a master variable modulating marsh vertical accretion not only by accruing organic matter from decomposing belowground biomass, but also by facilitating the deposition of suspended matter during tidal inundation (Baustian et al., 2012; Mudd et al., 2009, 2010). Conversely, relative marsh elevation or elevation capital (Cahoon and 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 Southeastern and Gulf of Mexico US (Alber et al., 2008; McKee et al., 2004). 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 and 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 and Alber, 2006; Schepers et al., 2020). In contrast, little is known about the causes and fate of long-term dieback in areas that transitioned from salt marsh to mudflat, as well as their vulnerability to accelerated SLR. Furthermore, to the best of our knowledge there are limited studies characterizing the trajectory of salt marsh dieback events at decadal timescales (e.g., Marsh et al., 2016).

At a global scale, the area of marshland that is projected to drown as a consequence of accelerated SLR is thought to be offset by salt marsh migration onto higher land (Kirwan et al., 2016a). Nevertheless, urban salt marshes are disproportionately vulnerable to accelerated SLR because built infrastructure associated with coastal cities acts as a physical barrier preventing marsh migration (Kirwan and Gedan, 2019; Schuerch et al., 2018). If no actions are taken to conserve 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 (Kulp and Strauss, 2019; Martínez et al., 2006). Moreover, lower income populations from underrepresented minority communities are more likely to occupy areas that are vulnerable to flooding and extreme climatic events (Chakraborty et al., 2014; Eisenman et al., 2007). Keeping urban marshes in place represents an effective climate change adaptation strategy with increasing relevance over time, insofar as the frequency and intensity of tropical storms and hurricanes are already increasing (Brown et al., 2019; Paerl et al., 2019). In comparison to their natural ecosystem counterparts, much less information is available on the vulnerability or resilience of urban salt marshes to accelerated SLR. Thus, there is 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 motivated by members of the Ashleyville Neighborhood Association in West Ashley, a densely populated low-lying area that borders on urban marshes in Charleston, South Carolina. Salt marshes in the area serve as essential habitat for many fish and shellfish species (e.g. red drum, sharks, 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 sea level rise (Crist et al. 2019). This area in the lower peninsula of Charleston, also known as Maryville, is of extraordinary cultural significance, since it served as the site of the first English settlement in SC in 1670 as well as the first African-American town in the state. Over the past 10 to 20 years, residents of the Gullah Geechee community noticed a large-scale dieback of vegetation and brought this to the attention of the South Carolina Aquarium.

Thus, this study was initiated to assess the vulnerability and restoration potential of salt marshes in South Carolina and then extended to areas where similar dieback events were observed in marshes of Louisiana. Here we provide evidence that about 15% of total salt marsh area have not recovered 20- and 10-years after dieback events triggered by extreme severe droughts in the Gulf of Mexico and Southeast USA, respectively. The Southeast case study was studied in greater detail due to existing knowledge gaps in urban marsh ecosystems, and because it has been targeted as a potential restoration site by stakeholders. The Charleston dieback area was preferentially located in lower elevation midmarsh zones. Transplanting seedlings grown from local botanical seeds was demonstrated as an effective restoration strategy. A mechanistic modelling approach (Morris et al., 2002) showed that while mudflat areas would rapidly lose elevation capital at a moderate SLR

rate of 60 cm in 100 y^{-1}; a healthy vegetated salt marsh ecosystem with an optimal primary productivity of 2000 g m^{-2} y^{-1} would persist at this SLR rate for 100 years, although it would lose about half of its elevation capital (20.5 cm). Under an extreme climate change scenario, with a SLR rate of 122 cm in 100 y^{-1}, even highly productive salt marsh ecosystems augmented by the direct application of external sediment (i.e., thin layer placement of 5 cm every 25 years) would lose all elevation capital after 100 years.

2. Materials and methods

2.1 Study Area

The study was conducted in *Spartina alterniflora*-dominated salt marshes located in West Ashley, Charleston, SC, and Lafourche Parish, LA. 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 Civil War and first permanent English settlement in the Carolinas in 1670, respectively. The dieback area was discovered by members of the local Gullah Geechee community in a salt marsh adjacent to Maryville (Fig. S1). The marshes in Lafourche Parish, LA are located 85 km south of New Orleans, and are previously studied marshes impacted by the Louisiana 1999-2001 sudden dieback event (Baustian et al., 2012).

In order to relate the expansion of dieback areas to drought, we retrieved monthly Palmer drought stress index (PDSI) values from the South Carolina Southern Division (January 2009 to August 2021), and the Louisiana Southeast Division (January 1998, January 2019) using NOAA´s National Centers for Environmental Information database (https://www.ncdc.noaa.gov/temp-and-precip/drought/historical-palmers/).

2.2 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, 2017), and aerial imagery from US Department of Agriculture's (USDA) National Agriculture Imagery Program (NAIP) and National Aerial Photography Program (NAPP). The NAIP dataset consists of 4-band [blue, green, red, and near infrared (NIR)], 1-meter resolution imagery (http://www.fsa.usda.gov/programs-and-services/aerial-photography/imagery-programs/naip-imagery). The NAPP dataset consists of 3-band (green, red, and NIR), 1-meter resolution imagery. The PlanetScope constellation acquires daily, 3-meter resolution imagery, using a 4-band (blue, green, red, 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. Imagery for South Carolina consisted of 8 images retrieved from 2009 up to 2021, while imagery for Louisiana consisted of 7 images from 1998 to 2019 (further details on 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):

$$NDVI = \frac{(NIR-Red)}{(NIR+Red)} \tag{1}$$

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

Total unvegetated area was estimated for all aerial images and the 2021 PlanetScope image. We visually inspected the coordinates of vegetated and unvegetated points (median observations per year: 330) 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 three/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 is unvegetated using the three/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 and van Etten, 2012; Liaw and Wiener, 2002; R Core Team, 2021).

In order to investigate if unvegetated areas were preferentially associated with creek bank or midmarsh zones of the West Ashley marsh, we drew all creek bank edges from the study area (Fig. 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. All pixels closer to 7.5 m to the edge of any creek bank were considered as creek bank, and pixels further than 7.5 m as midmarsh.

2.3 Field sampling in West Ashley, SC

Field sampling was performed during the summer of 2021 in West Ashley, SC. Two field campaigns were performed on 27-28 May, and 20-26 July of 2021. We sampled 60, and 119 locations, in May and July, respectively (total = 179 points). Field sampling was performed non-randomly by establishing transects across gradients of *S. alterniflora* biomass from the creek bank to the interior of the marsh (Fig. S1). The dieback area adjacent to Maryville was oversampled due to our interest 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 meters in the North American Vertical Datum of 1988 (NAVD88) GEOID 18. 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 at 82 locations using a 0.25 m^2 quadrat. Seventy-one *S. alterniflora* shoots of known height were harvested from 37 different sampling points. Harvested shoots were oven-dried at 60 ºC for one week and shoot height was used to construct a biomass allometric equation.

2.4 Acquisition of LIDAR derived DEM

A light detection and ranging (LIDAR) derived bare-earth digital elevation model (DEM) was constructed from LIDAR aerial acquisition conducted from February 25, 2017 to March 09, 2017 in Charleston County (South Carolina DNR, 2018). Axis GeoSpatial (Easton, MD, USA) operated 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 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 Table S2 (further details in South Carolina DNR, 2018).

2.5 Correction of the LIDAR derived DEM

High-accuracy RTK elevations were used to correct the LIDAR derived DEM. Error of the LIDAR derived DEM was defined as: Z_{LIDAR} - Z_{RTK} for each RTK observation, where Z is NAVD88 elevation. We used a multispectral PlanetScope satellite image retrieved 5 days before LIDAR acquisition (02/20/2017) to correct the DEM based on the known relation between LIDAR error and *S. alterniflora* height (Hladik and Alber, 2012). The 179 RTK observations were randomly split for model training and testing in a 7 to 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 and 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 root mean square error (RMSE) calculated as:

$$RMSE = \sqrt{\sum(Observed_i - Predicted_i)^2 / n},$$

where $Observed_i$ is the ith observation of the RTK elevation, and $Predicted_i$ 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.

2.6 Estimation of 2021 aboveground biomass

Aboveground biomass was estimated for the West Ashley, SC study area based on *in situ* observations made during the late July 2021 field campaign. Aboveground biomass (g m^{-2}) estimated from 82 locations (section 3.2) were randomly split into train and test 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 08/01/2021) as explanatory variables. The model grew 5000 trees; and in each split, 2 variables were randomly sampled as candidates. The performance of the model was assessed by calculating the RMSE of the train and test datasets (as in section 4.5). A raster containing the predicted aboveground biomass was constructed using the random forest model. Aboveground biomass was set to 0 in all unvegetated areas (as inferred in section 2.2). The model was built using the raster and random Forest R packages in the R v. 4.1.0 environment (Hijmans and van Etten, 2012; Liaw and Wiener, 2002; R Core Team, 2021).

2.7 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. The restoration effort consists of *S. alterniflora* planting using seedlings grown from local sources. Botanical seeds were annually collected in October thru December each year from 11 unique sites, all located in Charleston County. After collection, seeds were kept at 4ºC for eight weeks, after which they were germinated in planting trays filled with fresh water substrate. After 6-7 months, seedlings ~55.0 cm tall were transplanted in the dieback areas from May to August. When transplanting, a spacing of 30.5 cm was used. Restoration was conducted in 6 individual patches with dimensions ranging from 48.6 m^2 to 183.3 m^2. In order to assess the effectiveness of the pilot restoration project, aboveground biomass was estimated during the July field campaign in 5, and 12 0.25-m^2 quadrats from restored plots planted in 2020 and 2021, respectively. A second field assessment was performed in September 22nd 2021, in which 10 quadrats per plot were assessed in the 2019, 2020, and 2021 transplanted plots, and two patches of adjacent relict marsh. A one-way ANOVA with the restored/natural plots as a fixed factor was used to explain differences in *S. alterniflora* aboveground biomass. In case of finding statistically significance ($p < 0.05$), a Tukey's post-hoc test using the *emmeans* R package was performed (Lenth, 2016).

2.8 Modeling of marsh elevation under primary production, management, and sea level rise scenarios

The Marsh Equilibrium Model (MEM) v. 9.01 (Morris et al., 2002, 2021) was employed to predict the resilience of the West Ashley salt marsh to accelerated SLR under different scenarios of (i) *S. alterniflora* primary productivity, (ii) human intervention (none or thin layer placement), and (iii) rates of SLR (regional NOAA's intermediate low of 60 cm in 100 y^{-1} and intermediate of 122 cm in 100 y^{-1}). The Marsh Equilibrium Model 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 thin-layer placement (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 VBA 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 in 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 sea-level rise rate was set as 3.4 mm y^{-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 (Jan 2009 to August 2021) was 76 cm, 4 cm NAVD88, and 80 cm NAVD88, respectively. Optimal elevation was determined as 5 cm MSL based on a biplot between elevation and aboveground biomass from the study area (Fig. 4f). Four different optimal values of aboveground primary productivity were modelled: 0, 500, 1000, and 2000 g m^{-2} y^{-1} in order to assess the effects of plant primary productivity on marsh resilience to SLR. The median elevation of all dieback pixels in 2021 was set as the initial elevation: 0.478 m 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, a treatment with 5 cm of thin layer placement (TLP) every 25 years was included into the study. Thin layer placement consists in spreading sediment into the marsh platform to increase surface elevation. After TLP, marsh vegetation was simulated to be completely lost, and fully recovered 10 years after disturbance.

3. Results

3.1 Salt marsh dieback trajectory

We observed that expansions of large unvegetated areas coincided with severe droughts in both studied salt marshes in West Ashley, South Carolina and Lafourche Parish, Louisiana (Fig. 1, 2). West Ashley, SC, experienced its most severe drought over the last 100 years in 2011-2012, which coincided with an over 300 % increase of unvegetated marsh area (64 ha). A second dieback expansion was observed in 2017, concurrent with two major catastrophic tropical cyclones impacting West Ashley in October 2015 (Joaquin) and October 2016 (Matthew). In 2019, the vegetated area of West Ashley recovered up to 2013 levels which have persisted since (Fig. 1). In Louisiana, the drought that occurred between 1999 and 2001 has been described as the most extreme drought event in over 100 years, triggering more than 100,000 ha of marsh dieback across the state (McKee et al., 2004). We quantified a 60-ha increase of unvegetated area near Lafourche Parish from 1998 to 2004, which has not recovered after 20 years since the dieback event (Fig. 2). In the West Ashley marsh, the dieback event transitioned from vegetated marsh to mudflat, while in Louisiana the regime shift was from salt marsh to pond.

Motivated by stakeholders and due to the overall lack of data on urban marshes, the West Ashley salt marsh was studied in greater detail (study area in Fig. S1). Based on the well-known natural zonation of *S. alterniflora* (Mendelssohn and 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 majority of total unvegetated area was attributed to midmarsh across the time series investigated. Using the corrected 2017 LIDAR derived DEM,

we observed that both creek bank and midmarsh dieback areas were associated with lower elevation terrain (Fig. 3). The difference in median elevations between vegetated and unvegetated areas ranged from 13.3 cm to 27.4 cm, and 6.3 cm to 14.3 cm in creekbank and midmarsh zones across years, 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 (South Carolina DNR, 2009), finding similar results (Fig. S3). In the 2009 DEM, difference between median elevation from vegetated and unvegetated areas ranged from 4.4 to 23.0 cm, and 2.4 to 16.3 cm in creekbank and midmarsh zones, respectively. The results suggest that lower elevation in the unvegetated areas persisted before the dieback event.

3.2 *Spartina alterniflora* primary production during the summer of 2021

During July of 2021, field work was performed to calibrate a site-specific allometric equation (Fig. 4b), and to quantify the primary production of the West Ashley salt marsh (Fig. 4a). The random forest model based on the 2017 calibrated LIDAR derived DEM, vegetation indices, and satellite bands from an August 2021 color near infrared PlanetScope image had a $RMSE_{test}$ of 268.1 g m^{-2}. Median predicted aboveground biomass was 623.1 g m^{-2}, ranging from 0 to 1944 g m^{-2} (Fig. 4b). Biomass was within the expected range for West Ashley's latitude (predicted end of season productivity based on Kirwan et al., 2009: 574.8 g m^{-2} y^{-1}). Aboveground biomass was closely associated with elevation and distance to tidal creeks (Fig. 4e,f). Greater aboveground biomass was found in lower elevation areas within closer proximity to tidal creeks (Fig. 4e,f).

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

Similar to salt marshes throughout the eastern and Gulf coasts of the U.S., the West Ashley marsh was predominated by *Spartina alterniflora*. Thus, our team collected botanical seeds of *S. alterniflora* from the Charleston, SC area, and propagated the plants under greenhouse conditions for 6-7 months. Guided by the South Carolina Department of Natural Resources, seedlings were transplanted by community volunteers in July of 2019, 2020, and 2021 in a West Ashley dieback area that has been unvegetated since 2013 (Fig. 5). Aboveground biomass determined in July of 2021 was equivalent to that determined in the 2020 transplanted plot and compared to reference plots of natural midmarsh grass in the study area (Fig. 5). Repeated measurements conducted in September of 2021 showed that the restored plots planted in 2019 had greater aboveground biomass in comparison to reference sites adjacent to the restoration site (Fig. 5).

3.4 Project of marsh resilience under a range of SLR scenarios.

A mechanistic modelling approach (Morris et al., 2002, 2021) was used to assess the resilience of the West Ashley salt marsh to accelerated SLR under different scenarios of (i) *S. alterniflora* primary productivity, (ii) human intervention (none or thin layer placement), and (iii) rates of SLR (NOAA's regional intermediate low of 60 cm in 100 y^{-1} or intermediate of 122 cm in 100 y^{-1}). The model output indicated that mudflat areas (primary productivity = 0 g m^{-2} y^{-1}) rapidly lose elevation capital at a SLR rate of 60 cm in 100 y^{-1}; decreasing in elevation by a total of 38 cm after 100 years (Fig. 6). A salt marsh ecosystem with an optimal primary productivity of 2000 g m^{-2} y^{-1} would persist under the 60 cm in 100 y^{-1} SLR scenario, while losing 20.5 cm of elevation capital after 100 years (Fig. 6). At a SLR rate of 122 cm in 100 y^{-1}, such a healthy salt marsh with an optimal primary productivity of 2000 g m^{-2} y^{-1} would collapse. Simulating an external supply of sediment (i.e., thin layer placement) at a rate of 5 cm of inorganic sediment every 25 years would be just enough to persist after 100 years in the NOAA's intermediate SLR scenario (Fig. 6).

4. Discussion

4.1 The resilience of salt marshes to accelerated sea level rise and extreme climatic events.

Extreme drought events have been linked to sudden vegetation diebacks in a number of 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). 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 exceed the physiological tolerance of marsh plants (Marsh et al., 2016; Ramsey et al., 2012; Stagg et al., 2021). Our results reinforce previous studies which attribute marsh vegetation dieback to extreme climatic events in the USA Southeast and Gulf of Mexico.

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 15% loss of total vegetated area after 10 and 20 years in West Ashley, SC, and Lafourche Parish, LA, respectively. 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. From both case studies, most intriguing is the case of the West Ashley marsh, since 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 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 reasons explaining slow or no recovery 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 long-term marsh diebacks, thereby increasing the vulnerability of these ecosystems to SLR (Brown et al., 2019; Dai, 2013; Paerl et al., 2019). Thus, the effects of extreme climatic events, such as drought and extreme precipitation, must be incorporated into assessments of the resilience of salt marsh ecosystems 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 mean sea level (Morris et al, 2002; Kirwan et al., 2010, 2016b). In other words, the marsh drowns with the loss of elevation capital. Failing to incorporate the increasing frequency and intensity of extreme climatic events which trigger irreversible marsh diebacks will lead to an underestimation of the vulnerability of salt marshes to climate change. For example, our modelling approach forecasted that after 100 years under a moderate climate change scenario, the elevation of a highly productive marsh would exceed that of an unvegetated dieback marsh by 17.5 cm (see below for further discussion).

4.2 Call for restoration of urban marshes

Urban salt marsh ecosystems are inherently more vulnerable than natural marshes due to their limited migration space (Schuerch et al., 2018). At the same time, the value of the ecosystem services they provide are very 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 (Temmerman et al., 2013; Geedicke et al., 2018). 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 natural counterparts (Alldred et al., 2020).

Our results corroborate evidence from natural marshes to show that marsh grass planting is an effective restoration practice in an urban marsh that experienced dieback (Baustian et al., 2012; Linthurst and Seneca, 1980; Ogburn and 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 marsh stable 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). Because plant primary production is a key variable modulating marsh accretion, rapid restoration of marsh dieback is crucial in order to avoid further degradation, especially in stressed urban salt marshes (Morris et al., 2002; Mudd et al., 2009, 2010; Silliman et al., 2019). The urgency increases when taking in 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 in the long term must take elevation capital and hydrology in special consideration. 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).

4.3 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 modelling approach suggests that even highly productive marshes will likely drown under more extreme climate change scenarios. Our results are supported by long-term modelling 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 enough to keep up marsh elevation to 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). However, even periodically placement of dredge material may not be enough under extreme climate change scenarios (Fig 6). Thus, both climate change adaptation and mitigation actions are urgently needed to preserve highly valuable present-day marsh ecosystems close to urban environments.

5. **Acknowledgements**

This research was funded by grant 66115 (to J.E.K.) from the National Fish and Wildlife Foundation under the National Coastal Resilience Fund program. The views and conclusions contained in this document are those of the authors and should not be interpreted as representing the opinions or policies of the U.S. Government or the National Fish and Wildlife Foundation and its funding sources. Mention of trade names or commercial products does not constitute their endorsement by the U.S. Government, or the National Fish and Wildlife Foundation or its funding sources. Grass planting

in Maryville, South Carolina was implemented through funding received from NOAA National Marine Fisheries Service (to M.H.).

6. Conflict of Interest

The authors declare that they have no conflict of interest.

7. Contributions

J.LR., J.E.K., M.H., N.W., J.R., A.G., J.C., and J.T.M. designed the research. J.L.R., M.H., K.D.G., G.K., N.W., J.R., and J.E.K performed field work, J.L.R., and M.H., conducted laboratory work, J.L.R. analyzed the data with input from J.T.M., and J.E.K. J.L.R. wrote the first draft of the manuscript with inputs from J.E.K. and all authors equally contributed to revisions and gave approval for publication

8. Data availability statement

Remote sensed imagery was retrieved from publicly available repositories (https://earthexplorer.usgs.gov/ and https://planet.com/). All code and metadata can be found in the following GitHub Repository: https://github.com/kostka-lab/Longterm_Dieback_Spartina/

9. References

1. Alber, M., Swenson, E.M., Adamowicz, S.C., Mendelssohn, I.A., 2008. Salt marsh dieback: an overview of recent events in the US. Estuarine, Coastal and Shelf Science, 80:1-11.
2. Alldred, M., Borrelli, J.J., Hoellein, T., Bruesewitz, D., Zarnoch, C., 2020. Marsh plants enhance coastal marsh resilience by changing sediment oxygen and sulfide concentrations in an urban, eutrophic estuary. Estuaries and Coasts, 43:801-813.
3. Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecological Monographs, 81:169-193.
4. Baustian, J.J., Mendelssohn, I.A., & Hester, M.W., 2012. Vegetation's importance in regulating surface elevation in a coastal salt marsh facing elevated rates of sea level rise. Global Change Biology, 18:3377-3382.
5. Brown, V.M., Keim, B.D., Black, A.W., 2019. Climatology and trends in hourly precipitation for the southeast United States. Journal of Hydrometeorology, 20:1737-1755.
6. Cahoon, D.R., Guntenspergen, G.R., 2010. Climate change, sea-level rise, and coastal wetlands. National Wetlands Newsletter, 32:8-12.
7. Calcagno, V., de Mazancourt, C., 2010. glmulti: An R package for easy automated model selection with (generalized) linear models. Journal of Statistical Software, 34:1–29.
8. Chakraborty, J., Collins, T.W., Montgomery, M.C., Grineski, S.E., 2014. Social and spatial inequities in exposure to flood risk in Miami, Florida. Natural Hazards Review, 15:04014006.
9. Crist, P.J., White, R., Chesnutt, M., Scott, C., Sutter, R., Cutter, P., Dobson, G., 2019. Coastal Resilience Assessment of the Charleston Harbor Watershed. National Fish and Wildlife Foundation
10. Dai, A., 2013. Increasing drought under global warming in observations and models. Nature Climate Change, 3:52-58.
11. Donnelly, J.P., Bertness, M.D., 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. Proceedings of the National Academy of Sciences, 98:14218-14223.
12. Eisenman, D.P., Cordasco, K.M., Asch, S., Golden, J.F., Glik, D., 2007. Disaster planning and risk communication with vulnerable communities: lessons from Hurricane Katrina. American Journal of Public Health, 97:S109-S115.
13. Evans, B.R., Möller, I., Spencer, T., 2021. Topological and morphological controls on morphodynamics of salt marsh interiors. Journal of Marine Science and Engineering, 9:311.
14. Ford, M.A., Cahoon, D.R., Lynch, J.C., 1999. Restoring marsh elevation in a rapidly subsiding salt marsh by thin-layer deposition of dredged material. Ecological Engineering, 12:189-205.
15. Ganju, N.K., Defne, Z., Kirwan, M.L., Fagherazzi, S., D'Alpaos, A., Carniello, L., 2017. Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes. Nature communications, 8:1-7.
16. Geedicke, I., Oldeland, J., Leishman, M.R., 2018. Urban stormwater run-off promotes compression of saltmarshes by freshwater plants and mangrove forests. Science of The Total Environment, 637:137-144.

17. Hijmans, R.J., van Etten, J., 2012. raster: Geographic analysis and modeling with raster data. R package version 3.4-13. http://CRAN.R-project.org/package=raster
18. Hladik, C., Alber, M., 2012. Accuracy assessment and correction of a LIDAR-derived salt marsh digital elevation model. Remote Sensing of Environment, 121:224-235.
19. Hopkinson, C.S., Morris, J.T., Fagherazzi, S., Wollheim, W.M., Raymond, P.A., 2018. Lateral marsh edge erosion as a source of sediments for vertical marsh accretion. Journal of Geophysical Research: Biogeosciences, 123:2444-2465.
20. Kirwan, M.L., Guntenspergen, G.R., Morris, J.T., 2009. Latitudinal trends in *Spartina alterniflora* productivity and the response of coastal marshes to global change. Global Change Biology, 15:1982-1989.
21. Kirwan, M.L., Guntenspergen, G.R., D'Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophysical Research Letters, 37:L23401.
22. Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature, 504:53-60.
23. Kirwan, M.L., Temmerman, S., Skeehan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016a. Overestimation of marsh vulnerability to sea level rise. Nature Climate Change, 6:253-260.
24. Kirwan, M.L., Walters, D.C., Reay, W.G. and Carr, J.A., 2016b. Sea level driven marsh expansion in a coupled model of marsh erosion and migration. Geophysical Research Letters, 43:4366-4373.
25. Kirwan, M.L., Gedan, K.B., 2019. Sea-level driven land conversion and the formation of ghost forests. Nature Climate Change, 9:450-457.
26. Kulp, S.A., Strauss, B.H., 2019. New elevation data triple estimates of global vulnerability to sea-level rise and coastal flooding. Nature communications, 10:1-12.
27. Langston, A.K., Alexander, C.R., Alber, M., & Kirwan, M.L., 2021. Beyond 2100: Elevation capital disguises salt marsh vulnerability to sea-level rise in Georgia, USA. Estuarine, Coastal and Shelf Science, 249:107093.
28. Lenth, R.V., 2016. Least-squares means: The R package lsmeans. Journal of Statistical Software, 69:1–33.
29. Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. R News 2, 18-22.
30. Lindstedt, D.M., Swenson, E.M., Reed, D., Twilley, R., Mendelssohn, I.A., 2006. The case of the dying marsh grass. Report submitted to the Louisiana Department of Natural Resources, Baton Rouge, LA.
31. Linthurst, R.A., Seneca, E.D., 1980. Dieback of salt-water cordgrass (*Spartina alterniflora* Loisel.) in the lower Cape Fear estuary of North Carolina: an experimental approach to re-establishment. Environmental Conservation, 7:59-66.
32. Liu, Z., Fagherazzi, S., She, X., Ma, X., Xie, C., Cui, B., 2020. Efficient tidal channel networks alleviate the drought-induced die-off of salt marshes: Implications for coastal restoration and management. Science of The Total Environment, 749:141493.
33. Liu, Z., Fagherazzi, S., Cui, B., 2021. Success of coastal wetlands restoration is driven by sediment availability. Communications Earth & Environment, 2:1-9.
34. Marsh, A., Blum, L.K., Christian, R.R., Ramsey, E., Rangoonwala, A., 2016. Response and resilience of *Spartina alterniflora* to sudden dieback. Journal of Coastal Conservation, 20:335-350.
35. Martínez, M.L., Intralawan, A., Vázquez, G., Pérez-Maqueo, O., Sutton, P., & Landgrave, R., 2007. The coasts of our world: Ecological, economic and social importance. Ecological Economics, 63:254-272.
36. Maune, D.F., Maitra, J.B., McKay, E.J., 2007. Accuracy standards and guidelines. In D. Maune (Ed.), Digital elevation model technologies and applications: The DEM user's manual (pp. 65–97). (2nd Edition). Bethesda, Maryland: American Society for Photogrammetry and Remote Sensing.
37. McKee, K.L., Mendelssohn, I.A., Materne, M.D., 2004. Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? Global Ecology and Biogeography, 13:65-73.
38. Mendelssohn I.A., Morris J.T., 2002. Eco-Physiological Controls on the Productivity of Spartina Alterniflora Loisel. In: Weinstein M.P., Kreeger D.A. (eds) Concepts and Controversies in Tidal Marsh Ecology. Springer, Dordrecht.
39. Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. Ecology, 83:2869-2877.
40. Morris, J.T., Sundberg, K., Hopkinson, C.S., 2013, Salt marsh primary production and its responses to relative sea level and nutrients in estuaries at Plum Island, Massachusetts, and North Inlet, South Carolina, USA. Oceanography 26:78-84.

41. Morris, J.T., Barber, D.C., Callaway, J.C., Chambers, R., Hagen, S.C., Hopkinson, C.S., et al., 2016. Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. Earth's Future, 4:110-121.
42. Morris, J.T., Renken, K.A., 2020. Past, present, and future nuisance flooding on the Charleston peninsula. Plos one, 15:e0238770.
43. Morris, J. T., Cahoon, D. R., Callaway, J. C., Craft, C., Neubauer, S. C., Weston, N. B., 2021. Marsh Equilibrium Theory: Implications for Responses to Rising Sea Level. In: FitzGerald, D. M. and Hughes, Z. J. (eds) Salt Marshes: Function, Dynamics, and Stresses. Cambridge: Cambridge University Press, pp. 157–177.
44. Mudd, S.M., Howell, S.M., Morris, J.T., 2009. Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. Estuarine, Coastal and Shelf Science, 82:377-389.
45. Mudd, S.M., D'Alpaos, A., Morris, J.T., 2010. How does vegetation affect sedimentation on tidal marshes? Investigating particle capture and hydrodynamic controls on biologically mediated sedimentation. Journal of Geophysical Research: Earth Surface, 115:F03029.
46. Ogburn, M.B., Alber, M., 2006. An investigation of salt marsh dieback in Georgia using field transplants. Estuaries and Coasts, 29:54-62.
47. Paerl, H.W., Hall, N.S., Hounshell, A.G., Luettich, R.A., Rossignol, K.L., Osburn, C.L., Bales, J., 2019. Recent increase in catastrophic tropical cyclone flooding in coastal North Carolina, USA: Long-term observations suggest a regime shift. Scientific reports, 9:1-9.
48. Planet Team, 2019. Planet Imagery Product Specifications. San Francisco, CA. https://assets.planet.com/docs/Planet_Combined_Imagery_Product_Specs_letter_screen.pdf
49. QGIS.org, 2021. QGIS Geographic Information System. QGIS Association. http://www.qgis.org
50. R Core Team, 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.
51. Ramsey III, E., Werle, D., Suzuoki, Y., Rangoonwala, A., Lu, Z., 2012. Limitations and potential of satellite imagery to monitor environmental response to coastal flooding. Journal of Coastal Research, 28:457-476.
52. Raposa, K.B., Ekberg, M.L.C., Burdick, D.M., Ernst, N.T., Adamowicz, S.C., 2017. Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise. Regional Environmental Change, 17:389-397.
53. Reed, D.J., 1995. The response of coastal marshes to sea-level rise: Survival or submergence? Earth Surface Processes and Landforms, 20:39-48.
54. Rogers, K., Kelleway, J.J., Saintilan, N., Megonigal, J.P., Adams, J.B., Holmquist, J.R., et al., 2019. Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. Nature, 567:91-95.
55. Schepers, L., Brennand, P., Kirwan, M.L., Guntenspergen, G.R., & Temmerman, S., 2020. Coastal marsh degradation into ponds induces irreversible elevation loss relative to sea level in a microtidal system. Geophysical Research Letters, 47:e2020GL089121.
56. Schuerch, M., Spencer, T., Temmerman, S., Kirwan, M.L., Wolff, C., Lincke, D., et al., 2018. Future response of global coastal wetlands to sea-level rise. Nature, 561:231-234.
57. Silliman, B.R., He, Q., Angelini, C., Smith, C.S., Kirwan, M.L., Daleo, P., Renzi, J.J., Butler, J., Osborne, T.Z., Nifong, J.C., van de Koppel, J., 2019. Field experiments and meta-analysis reveal wetland vegetation as a crucial element in the coastal protection paradigm. Current Biology, 29:1800-1806.
58. South Carolina DNR, 2018. Report of the South Carolina Charleston County Lidar. https://prd-tnm.s3.amazonaws.com/StagedProducts/Elevation/metadata/SC_CoastalCounties_2017_A17/SC_Charleston_2016/reports/South_Carolina_Lidar_Charleston_Report_180720.pdf
59. South Carolina DNR, 2009. Report of the South Carolina Charleston County Lidar. https://www.fisheries.noaa.gov/inport/item/49978
60. Stagg, C.L., Osland, M.J., Moon, J.A., Feher, L.C., Laurenzano, C., Lane, T.C., Jones, W.R., Hartley, S.B., 2021. Extreme Precipitation and Flooding Contribute to Sudden Vegetation Dieback in a Coastal Salt Marsh. Plants, 10:1841.
61. Sweet, W.V., Kopp, R.E., Weaver, C.P., Obeysekera, J., Horton, R.M., Thieler, E.R., et al., 2017. Global and Regional Sea Level Rise Scenarios for the United States. NOAA Technical Report NOSCO-OPS 086; 44pp.
62. Temmerman, S., Moonen, P., Schoelynck, J., Govers, G., Bouma, T.J., 2012. Impact of vegetation die-off on spatial flow patterns over a tidal marsh. Geophysical Research Letters, 39:L03406.

63. Temmerman, S., Meire, P., Bouma, T.J., Herman, P.M., Ysebaert, T., De Vriend, H.J., 2013. Ecosystem-based coastal defence in the face of global change. Nature, 504:79-83.
64. Temmink, R.J., Christianen, M.J., Fivash, G.S., Angelini, C., Boström, C., Didderen, K., Engel, S.M., Esteban, N., Gaeckle, J.L., Gagnon, K., Govers, L.L., 2020. Mimicry of emergent traits amplifies coastal restoration success. Nature communications, 11:1-9.
65. Torres, R., Goni, M.A., Voulgaris, G., Lovell, C.R., Morris, J.T., 2004. Effects of low tide rainfall on intertidal zone material cycling. Coastal and Estuarine Studies, 59:93-114.
66. Valiela, I., Teal, J.M., Deuser, W.G., 1978. The nature of growth forms in the salt marsh grass *Spartina alterniflora*. The American Naturalist, 112:461–470
67. Webb, E.C., Mendelssohn, I.A., Wilsey, B.J., 1995. Causes for vegetation dieback in a Louisiana salt marsh: a bioassay approach. Aquatic Botany, 51:281-289.
68. Weston, N.B., 2014. Declining sediments and rising seas: an unfortunate convergence for tidal wetlands. Estuaries and Coasts, 37:1-23.
69. Wilsey, B. J., McKee, K.L., & Mendelssohn, I.A., 1992. Effects of increased elevation and macro-and micronutrient additions on *Spartina alterniflora* transplant success in salt-marsh dieback areas in Louisiana. Environmental Management, 16:505-511.
70. Zhu, Z., Vuik, V., Visser, P.J., Soens, T., van Wesenbeeck, B., van de Koppel, J., Jonkman, S.N., Temmerman, S., Bouma, T.J., 2020. Historic storms and the hidden value of coastal wetlands for nature-based flood defence. Nature Sustainability, 3:853-862.

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) over time. Values lower than -2 represent moderate drought (dotted line), while values lower than -4 extreme drought conditions (dashed line). **c** Maps showing the probability of an area being unvegetated. Based on individual random forest models per image.

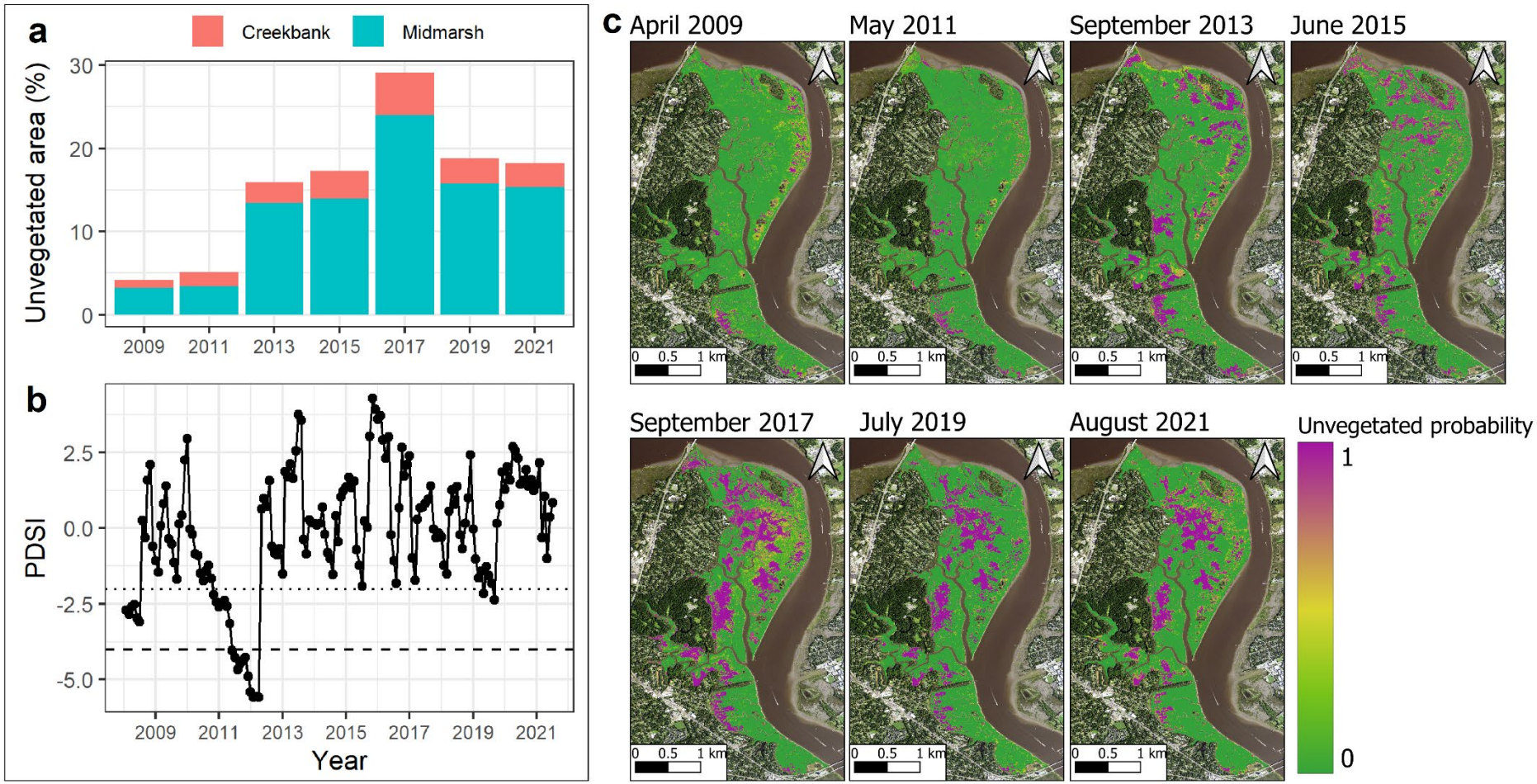

Figure 2: Dieback trajectory of the saltmarsh located in Lafourche Parish, LA. **a** Percentage of unvegetated marsh area over time. **b** Palmer drought severity index (PDSI) over time. Values lower than -2 represent moderate drought (dotted line), and values lower than -4 extreme drought conditions (dashed line). **c** Study area and maps showing the probability of an area being unvegetated. Based on individual random forest models per image.

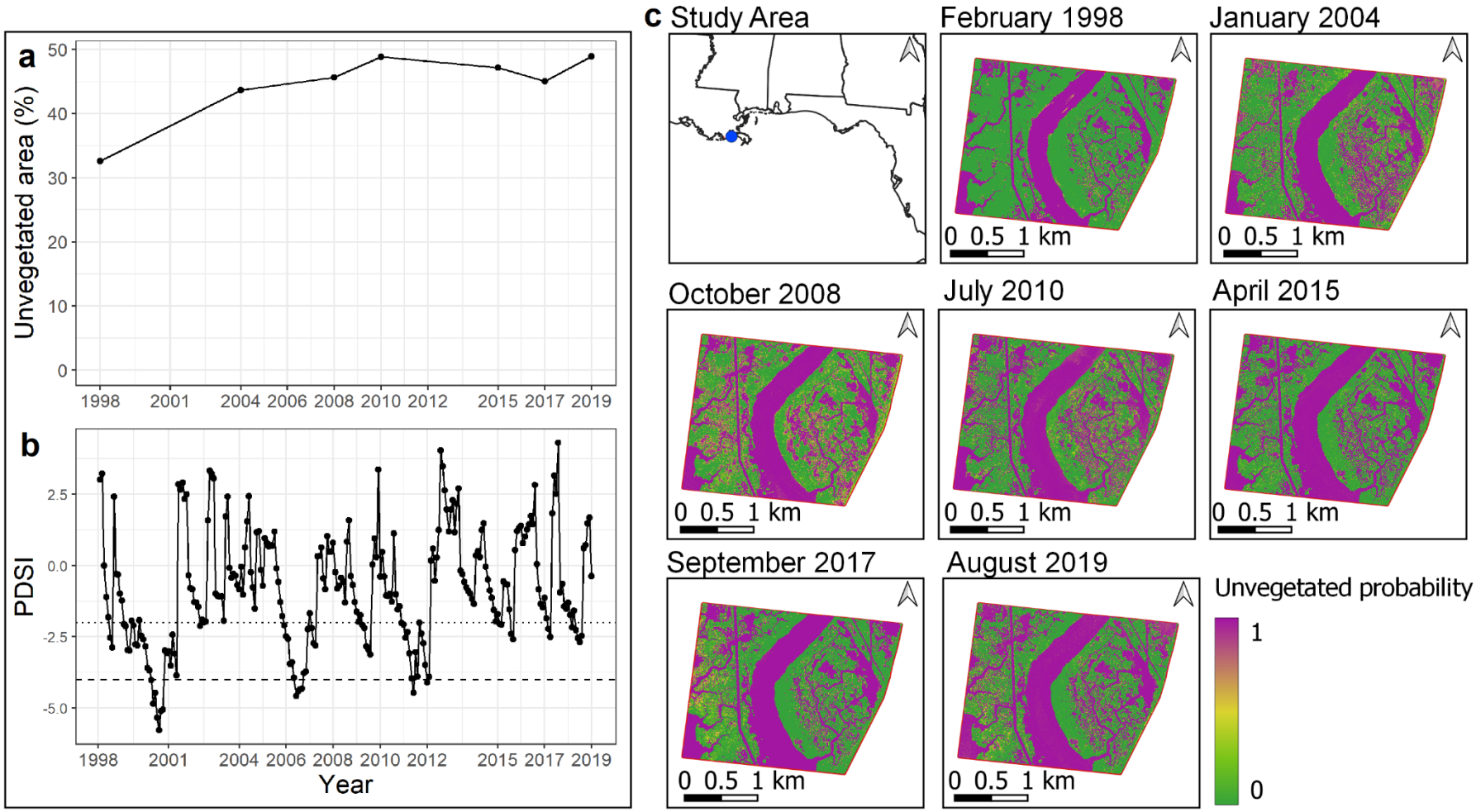

Figure 3: Characterization of unvegetated and vegetated areas within the West Ashley, SC salt marsh. **a** Boxplots representing the interquartile range of corrected elevation values generated from a 2017 light detection and ranging (LIDAR) derived digital elevation model (DEM). Boxplots grouped in vegetated and unvegetated areas over time. **b** Boxplots representing the interquartile range of pixel values from a distance from closer tidal creek raster from vegetated and unvegetated areas over time.

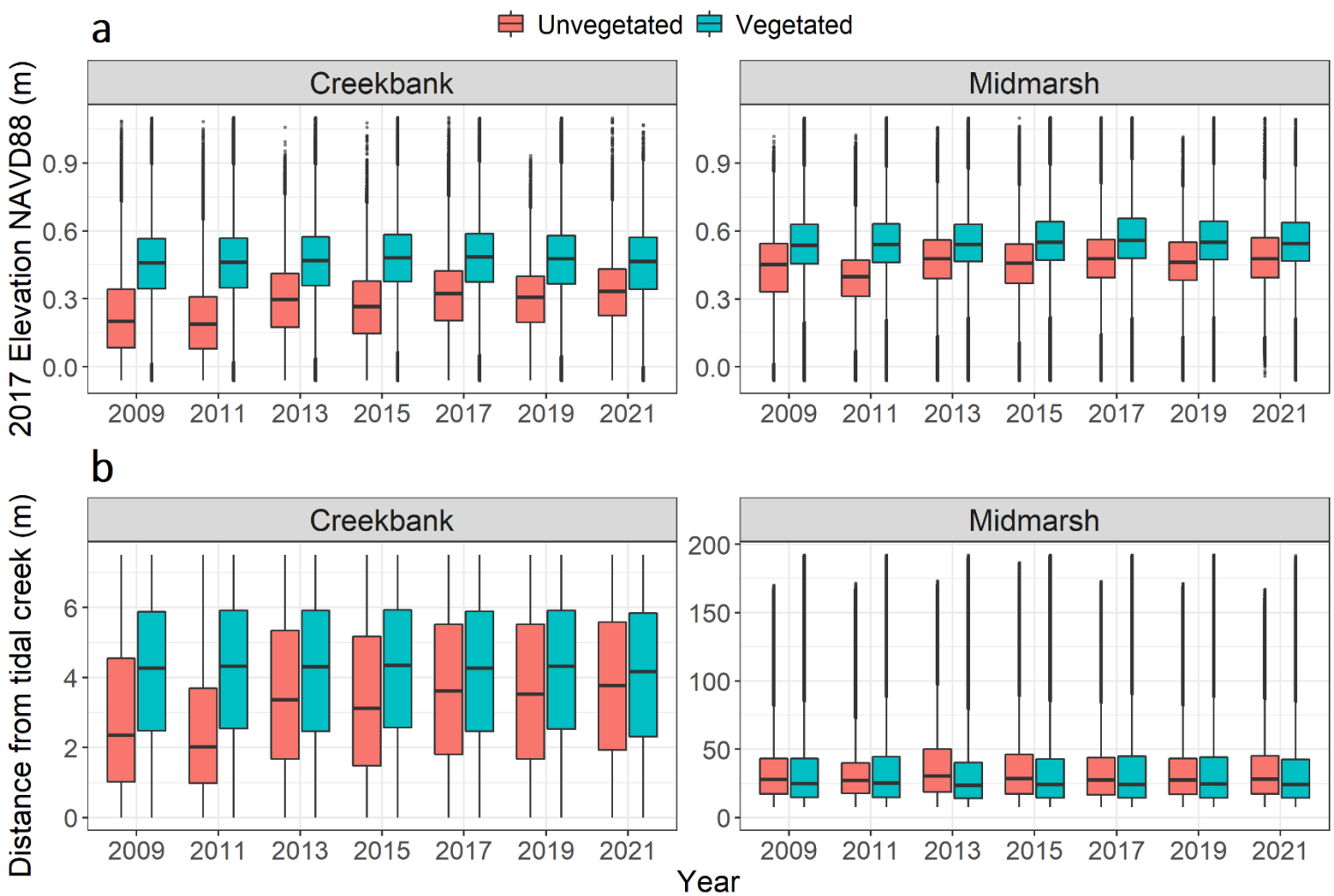

Figure 4: Aboveground biomass (AGB) estimation of the West Ashley, SC salt marsh during July 2021. **a** Map of AGB based on a random forest model using the 2017 elevation digital elevation model, vegetation indices and bands from a PlanetScope 4-band image. **b** Density plot of the predicted AGB values. **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 2017 elevation model **(f)**. Points and error bars represent AGB median and interquartile range on equidistant increments, respectively.

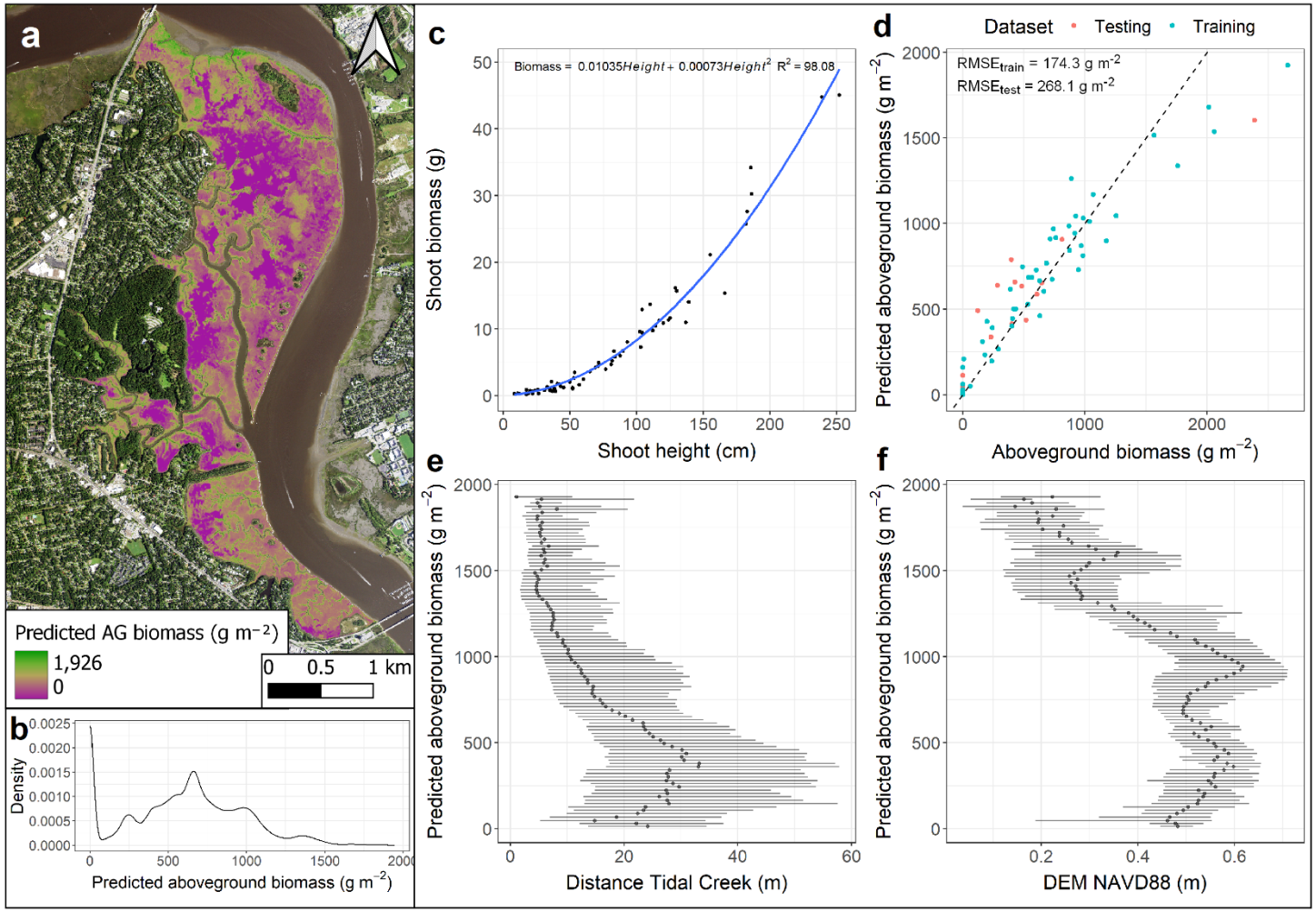

Figure 5 Grass planting is an effective restoration practice in Maryville, SC. **a** Drone image from the restoration site. Plots sampled are labelled on top in white. Photo acquired by the South Carolina Department of Natural Resources Shellfish Research Section on 21 July of 2021. **b** Boxplot representing the interquartile range of aboveground biomass from all sampled creek bank, midmarsh, and 2020 and 2021 restored plot 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).

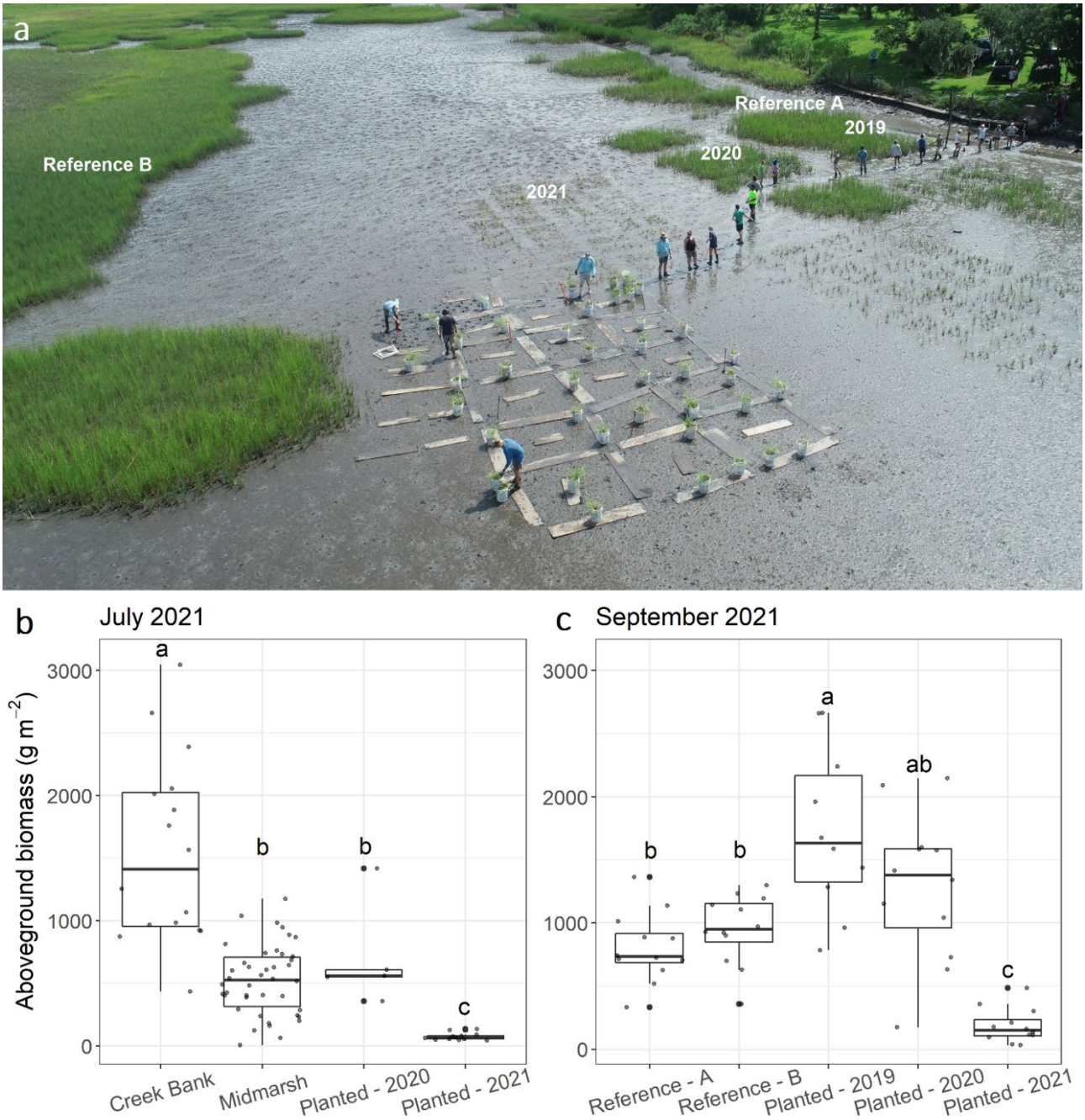
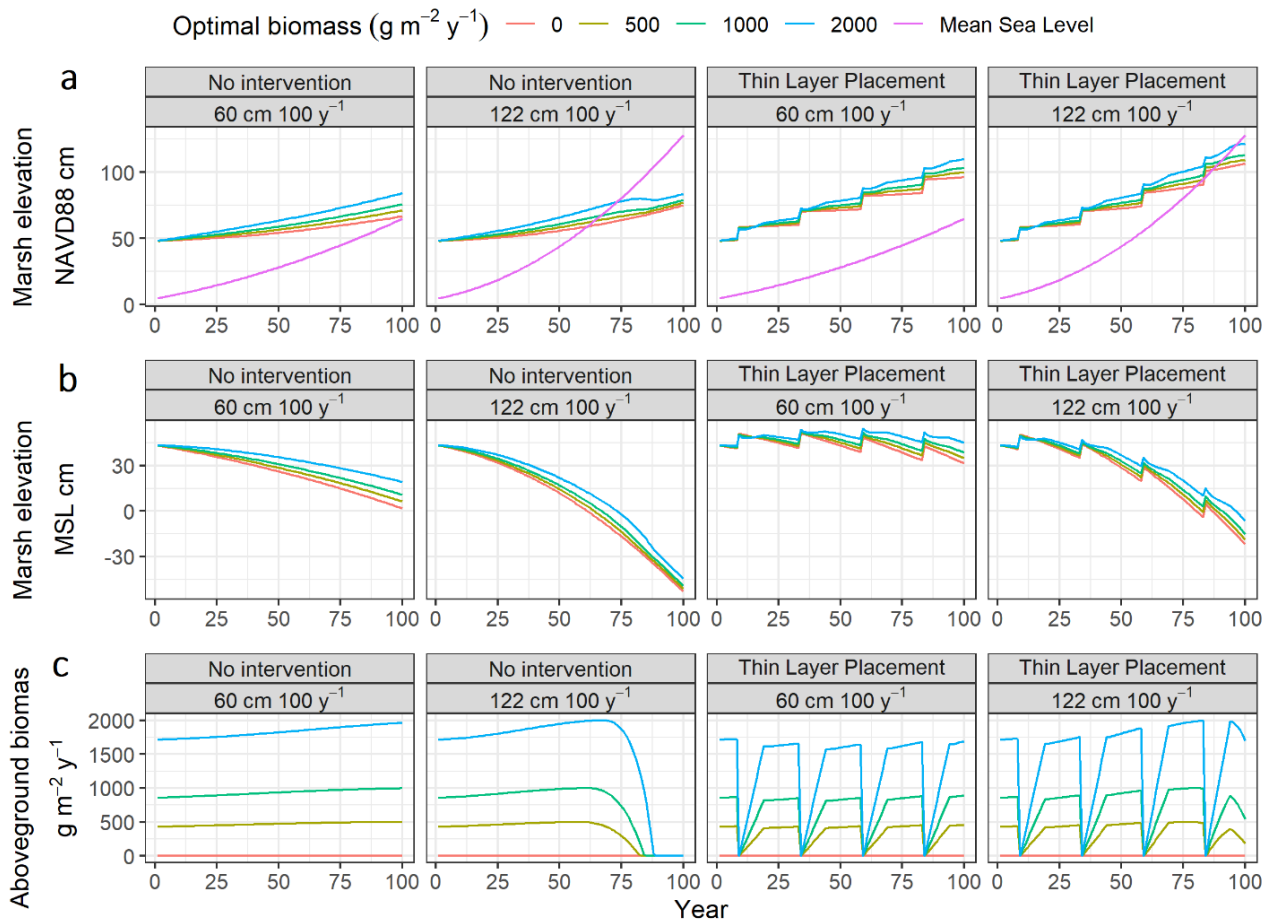
Figure 6 Modelled salt marsh resilience under different scenarios of sea level rise, human intervention, and plant primary production using the Marsh Equilibrium Model (MEM) (Morris et al., 2002). Modelled optimal biomass values ranged from 0 g m^{-2} y^{-1} (mudflat) to highly productive salt marshes of 2000 g m^{-2} y^{-1}. Thin Layer Placement was assumed to be performed every 25 years by applying 5 cm of inorganic sediment. Vegetation was assumed to fully recover after 10 years of disturbance. The two sea level rise scenarios modelled were based on regional NOAA´s intermediate low, and intermediate of 60 cm and 122 cm in 100 y, respectively (Sweet et al., 2017). Marsh elevation (cm) over time projected to NAVD88 (**a**), and mean sea level (MSL) (**b**). **c** Projected marsh aboveground biomass over time.