Research Square

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Restoration of Sesarma-driven salt marsh dieback through direct crab management

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Abstract

Salt marsh ecosystems provide extensive resilience to coastal areas yet face increasing impacts from sea level rise, drought, human impacts, and, in some areas, dieback of salt marsh vegetation, exposing sensitive sediments to erosion. In New England, increased populations of the native purple marsh crab (*Sesarma reticulatum*) have primarily driven salt marsh dieback perpendicular to salt marsh creeks, as crabs actively graze on roots and shoots of smooth cordgrass (*Spartina alterniflora*) in. When dieback was first observed on Nantucket Island, MA (~ 10–15 years after observance on the mainland), research pointed to a lack of predators as the driving force and could not offer a practical restoration method. This study attempted to examine anthropogenic predation of *Sesarma* and document the resultant impact on salt marsh dieback areas in one marsh on Nantucket. From 2019–2025, we performed extensive crab trapping and documented end-of-season dieback areas in the corresponding marsh. Initially, we observed dramatic increases in smooth cordgrass (*Spartina alterniflora*) colonization within previous dieback areas. Crab trapping provided a non-invasive and time efficient way to reduce grazing impact and increase vegetation cover, protecting the salt marsh from erosion. In later years of the study, we began to observe increased dieback on the high marsh platform and adjacent to an undersized culvert through a dike road, all areas that crabs had not initially burrowed in and away from the salt marsh creek edge typical for crab grazing habitat. While this data complicates the results of the study, issues of climate change stressors including water impounding, increased salinity, and drought are likely additional stressors to be addressed at the marsh while continuing crab management.

Introduction

Coastal salt marsh wetlands are ecosystems that experience twice-daily flushing in New England with salt water brought in by tides. Steady inundation and drainage drive numerous ecological functions that these ecosystems provide, including acting as habitat for birds and fish, providing flood protection and minimizing storm surge, absorbing and transforming excess nutrients from runoff, and storing carbon in organic salt marsh peat (Elmer 2013). To provide adaptation to coastal change and gradual sea level rise, salt marshes depend on tidal inundation for vertical growth; as daily tides deposit marine sediments, halophytic plants trap sediment on the marsh platform (Morris et al. 2002). Furthermore, as salt marsh vegetation dies each year, detritus accumulates on the marsh and decomposes slowly, forming peat and storing carbon (Drexler et al. 2009). The steady accumulation of peat and growth of vegetation in salt marshes has historically allowed the marsh ecosystem to keep pace with natural fluctuations in sea level (Crosby et al. 2016). However, the effects of climate change threaten marshes globally, particularly the increase in frequency and intensity of storms coupled with anthropogenic-driven sea level rise outpacing vertical marsh growth (Kirwan 2010, Luo et al. 2026).

There are many indicators of salt marsh stress, but one of the most visually obvious is salt marsh dieback. Broadly, salt marsh dieback is the complete loss of stabilizing low marsh vegetation, most notably smooth cordgrass *Spartina alterniflora* (smooth cordgrass), which can result in large expanses of exposed marsh soil sediment within an established salt marsh (Marsh et al. 2016). The loss of low

marsh vegetation can severely impact salt marsh stability, leading to increases in peat erosion from wave and storm events, sediment subsidence, and exposure of the salt marsh to the amplifying threats associated with climate change and sea level rise (Elmer 2014).

Multiple drivers of salt marsh dieback have been documented, and studies have made clear that dieback is more likely to occur in stressed salt marshes (Elmer 2013). Dieback often occurs in marshes that may be waterlogged from storms, precipitation, or sea level rise, which flood vegetation beyond their tolerance (Stagg et al. 2021). Waterlogging decreases carbon dioxide available for plants and exposes them to excess salinity, increasing soil salinity and subsequent vegetation die off (Hou et al. 2020). Marshes may also undergo stress due to drought, which exposes peat to oxygen and changes its chemistry. Specifically, allowing more oxygen into salt marsh peat shifts decomposition from slow, anaerobic mechanisms occurring in soils flooded twice daily to fast, respiration mechanisms where aeration increases subsurface reduction-oxidation (redox) potential, ultimately releasing carbon dioxide (Guimond et al. 2020). Marshes also become stressed when exposed to plant pathogens, or when wrack or storm over-wash smothers peat, causing dieback (Elmer 2014).

Increased grazing pressure from purple marsh crabs *Sesarma reticulatum* (purple marsh crab) is another prominent driver of salt marsh dieback of *S. alterniflora* in New England, with crabs eating *S. alterniflora* shoots and roots and oxidizing the peat through burrowing (Coverdale et al. 2012; Holdredge et al. 2008). Crab-driven salt marsh dieback was first observed in 2003 in Cape Cod by Stephen Smith ad John Portnoy at the Cape Cod National Seashore (Smith 2009), in 2009 in Rhode Island (Bertness et al. 2014), and in 2012 on Nantucket Island, MA (Karberg, personal observation). A rapid rise in *S. reticulatum* populations leads to overconsumption of the entire *Spartina* plant which destabilizes salt marsh soils and facilitates peat erosion (Altieri et al. 2012; Bertness & Silliman 2008). The direct cause of increased *S. reticulatum* populations is debated, with causes linked to localized overfishing of *S. reticulatum* predators, existing marsh degradation, and softening of peat due to extended tidal inundation periods that make crab burrowing easier (Elmer 2013; Raposa 2018; Crotty et al. 2020).

According to Crotty et al. (2017), the growth of *S. reticulatum* populations has driven the loss of salt marsh area in New England more than sea level rise between 2005–2013. It is expected that projected sea level rise will exacerbate crab-driven dieback, with higher tides and storm surge exposing areas for crab burrows and herbivory (Szura et al. 2017). A synergistic interaction between the two stressors has the potential to leave 86% of current salt marsh areas vulnerable to herbivory by *S. reticulatum* in the next century (Crotty et al. 2017).

The pace at which crab herbivory has eroded salt marshes in New England and its projected impact on marsh areas calls for management strategies that protect vulnerable marsh ecosystems from these stressors, given that natural recovery from dieback is variable. According to Rolando et al. (2023), areas of dieback have had variable recovery periods, with some marshes recovering naturally within months to years, and others continuing to remain unvegetated. The reasons behind slow or no recovery must be further researched, but potential explanations include limited seed dispersal or poor hydrology and

drainage (Rolando 2023). Other studies note the need to replant salt marsh grasses within dieback to assist recovery, emphasizing that recovery from dieback may require intervention (Ogburn & Alber 2006).

In 2019, the Nantucket Conservation Foundation (NCF) observed heightened grazing pressure from *S. reticulatum* in marshes experiencing dieback on Nantucket Island, MA. Facing this pressure and potential loss of valuable salt marshes, we explored the restoration technique of acting as a predator to facilitate marsh recovery. Due to uncertainty about the driving force behind increased crab populations, we believed that the crab population itself was a controllable variable to restore salt marsh grass before sediment became too eroded.

Research Objectives

The purpose of this project was the removal of the native *Sesarma reticulatum* during the growing period to facilitate restoration of *Spartina alterniflora* in areas experiencing salt marsh dieback. We examined the effectiveness of trapping and removing *S. reticulatum* on slowing salt marsh dieback extent and assisting recolonization of *S. alterniflora* to increase salt marsh resilience. This was the first management project to explore active *S. reticulatum* population management to improve salt marsh health and has already informed adaptive restoration options for other salt marshes throughout New England.

Methods

Study Site

The Medouie Creek wetland complex (Medouie) (~ 7 hectares) is a conservation property owned by the Nantucket Conservation Foundation (NCF) in the eastern lobe of Polpis Harbor, a sub-embayment of Nantucket Harbor on Nantucket Island, MA (Fig. 1). This historically contiguous salt marsh experienced diking in the 1930s, creating a tidal restriction separating the Back marsh from the Front marsh which remained hydrologically connected to tidal Polpis Harbor (Fig. 1). In 2008, NCF initiated a large-scale restoration project to reconnect tidal hydrology and successfully restored salt marsh function, hydrology, and vegetation communities throughout Medouie (Karberg et al. 2018). In winter 2022, NCF installed an intertidal oyster reef perpendicular to the Front marsh creek outflow into Polpis Harbor. Consultations during permitting for this project indicated that the culvert placed to re-connect tidal flow in 2008 was now undersized due to sea level rise changes.

Marsh Dieback Areas

In 2012, 5 years post-restoration, NCF first observed growing expanses of bare, unvegetated salt marsh soil riddled with burrows in the Front marsh at Medouie (Karberg *pers obs*). Consultation with New England salt marsh dieback researchers and a pilot purple marsh crab trapping effort in 2015 confirmed the spread of *Sesarma*-driven salt marsh dieback in Medouie on Nantucket, approximately 13 years later

than *Sesarma*-driven dieback observed on Cape Cod (Holdredge et al. 2009; Smith 2009; Karberg *pers obs*).

Starting in 2018, NCF staff documented the extent of salt marsh dieback at the end of each field season (October 2018–2025) by walking the edge of dieback soils and using a handheld Trimble GPS unit to collect georeferenced polygons. As crab-driven salt marsh dieback expands in very regular grazing waves, dieback edge areas were easily and consistently identified on the landscape year to year and with field definitions being visually less than 10% of vegetation present as well as the presence of extensive crab holes in the bare surface soil. In 2021, salt marsh dieback was first observed in the Back marsh at Medouie; GPS documentation expanded to include these areas in 2022 (Fig. 3,4).

Crab Population Reduction

Crab trapping pit-fall arrays provide the most commonly used method to effectively trap and estimate salt marsh platform burrowing crab populations (Altieri et al. 2012; Angelini et al. 2018; Raposa et al. 2018). To act as crab predators in the salt marsh landscape, we adapted these trap arrays to effectively remove crabs from the dieback areas. Arrays were placed at the dieback fringe, the transition zone between dieback soils and vegetation farthest from the salt marsh creek edge, running parallel to the creek. This strategic placement covered the densest area of crab populations as *S. reticulatum* graze progressively away from the salt marsh creek (Angelini et al. 2018, Bertness et al. 2009, Morrison et al. 2024). Trap arrays consisted of empty tennis ball cans with three 1-inch slits in the bottom half of the can for drainage. A soil auger matching the diameter of the tennis ball can was used to dig a shallow hole allowing the can to be sunk until its rim was flush with the soil surface (Altieri et al. 2012; Angelini at al. 2018; Raposa et al. 2018). This mimicked a crab burrow opening which crabs would enter during nocturnal feeding, and the smooth can sides prevented crabs from escaping. Traps were placed approximately every five feet. Trap locations were re-used each field season unless the dieback fringe had moved significantly, then individual traps were shifted to a more optimal trap location.

Traps were deployed and open in the research area over the entire field season (typically from May to November). In 2019–2022, traps were placed solely in the Front marsh (n = 30) (Fig. 1). In 2023–2025, ~ 20 traps were added to the Back marsh, increasing total trap number (n = 50). Traps were visited 2–3 times each week, at which point all crabs in each trap were removed and counted. In addition to *S. reticulatum*, we also captured the invasive European green crab (*Carcinus maenas*) and the native fiddler crab (*Uca pugnax*). Fiddler crabs were recorded and released. *S. reticulatum* were euthanized by placing them in a small cooler and immediately freezing.

Analysis

Study Design

Study design consisted of annual observations at two sites: the Front marsh during 2019–2025 (n = 7) and the Back marsh during 2023–2025 (n = 3) with site-year as the observational unit. Annually, within each site we collected total seasonal catch of crabs and documented end-of-season dieback acreage. Seasonal trapping effort was calculated as the number of traps placed in each site multiplied by total number trap days, and catch per unit effort (CPUE) was calculated as total seasonal catch divided by effort, expressed as crabs per trap-day (Duncombe & Theirrault 2017). Effort was calculated as trap number multiplied by trapping days. CPUE was interpreted as an index of relative crab availability during trapping, conditional on reasonably comparable catchability across observations.

CPUE = (total number of crabs captured)/(Trap days * number of traps)

CPUE is often used to standardize crab trapping data and provides an index of relative crab abundance standardized to account for variable sampling effort and success (Raposa et al. 2018, 2019; Schweitzer and Withers 2009).

Annual dieback trajectories were calculated as current-year end-of-season dieback minus previous-year dieback; negative values indicated reduced dieback area.

Observed saltmarsh dieback change and crabs trapped over time

As an exploratory assessment of saltmarsh dieback acreage change over time, we calculated consecutive-year differences in dieback acreage separately for each marsh, including the 2018 Front and 2022 Back baseline measurements. Additionally, we calculated and graphed annual number of crabs removed through trapping in the Front and Back marsh.

Saltmarsh dieback and crab removal

We used two Gamma generalized models with a log link to evaluate our two primary questions: the association between crab availability and end-of-season dieback acreage and the associate between number of crabs removed and end-of-season dieback acreage. We selected the Gamma distribution due to the strictly positive, continuous end-of-season dieback response; the log link ensured positive fitted means.

The primary model evaluated whether greater same-year CPUE (crab availability) was associated with greater end-of-season dieback:

$$\log(\mu_{s,t}) = \beta_0 + \beta_1((\mathrm{CPUE}_{s,t})/0.1) + \beta_2 D_{s,t-1} + \beta_3 I_s.$$

The secondary model evaluated whether greater total crab removal in a year was associated with lower end-of-season dieback acreage:

$$\log(\mu_{s,t}) = \alpha_0 + \alpha_1((R_{s,t})/100) + \alpha_2 D_{s,t-1} + \alpha_3 I_s.$$

In both model equations, $\mu_{s,t}$ is end-of-season dieback acreage at site s in year t, $D_{s,t-1}$ is previous-year dieback acreage, $R_{s,t}$ is total annual crab removal, and I_s identifies site (Front = 0; Back = 1). Both models assume a conditional Gamma distribution with variance $\varphi\mu^2_{s,t}$, where φ is the dispersion parameter.

Previous-year dieback was included to account for differences in the previous marsh condition and the fixed site effect accounted for persistent differences between the two marsh sites (Front and Back). Models estimated a common CPUE or crab removal (predictor) slope across sites with a site-specific intercept. Site-by-predictor interactions were not fitted because the dataset contained only ten site-year observations with only three from the Back marsh. CPUE was expressed per 0.1 crab per trap-day to facilitate interpretation while crab removal was expressed per 100 crabs removed; this rescaling did not alter hypothesis tests.

Exponentiated Gamma coefficients were reported as ratios of expected dieback. Tests were two-sided, with α = 0.05, and p-values were unadjusted. Gamma confidence intervals were calculated using t-based Wald intervals with six residual degrees of freedom. Model assessment included residual plots, simulation-based Gamma diagnostic checks, influence plots and leave-one-out estimates to assess sensitivity. Gaussian linear models with identical predictors were fitted to assess whether conclusions depended on the response-family specification. Given the small sample size and limited spatial replication, all inference was considered exploratory. Analyses were performed in R; the exact version and session information are saved with the reproducible output.

Results

Observed saltmarsh dieback change over time

Front marsh dieback acreage declined from 0.424 acres in 2018 to 0.042 acres in 2023, a reduction of 82.6% (Fig. 2). Dieback acreage subsequently increased to 0.068 acres in 2024 and 0.103 acres in 2025 but remained 57.4% below its 2018 value.

In the Back marsh, dieback acreage decreased from the 2022 baseline of 0.250 acres to 0.135 acres in 2023, increased to 0.247 acres in 2024, and returned to 0.135 acres in 2025. Thus, Back marsh dieback fluctuated, with no net change between the study time frame (2023–2025) (Fig. 2, 4).

Mean annual dieback change was − 0.0199 acres in the Front marsh and − 0.0383 acres in the Back marsh.

Marsh Site	Monitoring Interval	Start Dieback (Acres)	End Dieback (Acres)	Net Change (Acres)	Ecological Trajectory
Front	2018 → 2019	0.2420	0.1670	-0.0750	Surface Recovery
Front	2019 → 2020	0.1670	0.1280	-0.0390	Surface Recovery
Front	2020 → 2021	0.1280	0.1020	-0.0260	Surface Recovery
Front	2021 → 2022	0.1020	0.0670	-0.0350	Surface Recovery
Front	2022 → 2023	0.0670	0.0420	-0.0250	Surface Recovery
Front	2023 → 2024	0.0420	0.0680	+ 0.0260	Dieback Expansion
Front	2024 → 2025	0.0680	0.1030	+ 0.0350	Dieback Expansion
Back	2022 → 2023	0.2500	0.1350	-0.1150	Surface Recovery
Back	2023 → 2024	0.1350	0.2470	+ 0.1120	Dieback Expansion
Back	2024 → 2025	0.2470	0.1350	-0.1120	Surface Recovery

Net change = (End Dieback − Start Dieback); negative values indicate decreased dieback and positive indicate increased dieback. Ecological trajectories are descriptive: “Surface Recovery” denotes reduced dieback.

Table 1

Front and Back marsh dieback acreage change over time illustrating salt marsh surface recovery and expanded dieback.

The ecological trajectory of the Front Marsh showed a consistent decrease in dieback acreage (Surface Recovery) from 2018–2023 after crab trapping began. Following 2023, the Front marsh showed an increase in Dieback acreage (Dieback Expansion) (Table 1) although the location of the dieback on the marsh surface occurred in new areas outside the typical *S. reticulatum* grazing patterns (Bertness et al. 2009, Morrison et al. 2024). While dieback acreage continued to decrease within the primary crab grazing area along the salt marsh creek, increases in dieback in 2023–2025 occurred adjacent to the restricted culvert and ditch road as well as along the shrub line at the extent of the Front marsh (Fig. 3).

Total crab removal fluctuated year to year but small sample size prohibited statistical analyses of change over time. In the Front marsh, removal increased by an average of 11.5 crabs per year over 2019–2025. And in the Back marsh, removal increased by an average of 37.5 crabs per year over 2023–2025 (Fig. 5). Trapping itself likely did not influence year to year crab numbers, this marsh is not an isolated system.

Saltmarsh dieback and crab removal

For the primary model which evaluated whether greater same-year CPUE (crab availability) was associated with greater end-of-season dieback: after adjustment for previous-year dieback and site, same-year CPUE was not significantly associated with end-of-season dieback (log-scale coefficient per 0.1 crab per trap-day = − 0.0257, SE = 0.1475, p = 0.867). The estimated dieback ratio was 0.975 (approximate 95% CI: 0.679–1.398) and the point estimate indicated 2.5% lower expected dieback per 0.1-unit increase in CPUE, but the interval encompassed both lower and higher dieback (Table 2).

Analysis	Predictor increment	Model	Estimate (95% CI)	p
Primary	0.1 crab/trap-day	Gamma	0.975 (0.679–1.398)	0.867
Secondary	100 crabs removed	Gamma	0.891 (0.658–1.206)	0.388

Table note. All models include previous-year dieback acreage and site and use ten observations. Gamma estimates are ratios of expected dieback; linear estimates are acreage differences. Gamma intervals are approximate t-based Wald intervals. P values are two-sided and unadjusted. The primary and secondary predictors were analyzed separately; rescaling is for interpretation only.

Table 2

Model results for the primary and secondary research questions.

For the secondary model which evaluated whether greater total crab removal in a year was associated with lower end-of-season dieback: the estimated association between total crab removal and dieback was negative but imprecise (log-scale coefficient per 100 crabs = − 0.1152, SE = 0.1237, p = 0.388). Each additional 100 crabs removed corresponded to an estimated dieback ratio of 0.891 (approximate 95% CI: 0.658–1.206) (Table 2).

Discussion

While ecosystem degradation can be driven by complicated and interactive factors, often the simplest interventions can provide successful restoration (Jones et al. 2018). In salt marshes experiencing top-down control through *Sesarma*-driven decreases in *S. alterniflora* coverage, increase factors such as higher salinity and drought can further drive salt marsh dieback (Angelini et al. 2019) but complicated restoration may not always be the simplest solution for reducing dieback area. This study shows strong

preliminary support for active *S. reticulatum* management to restore salt marshes experiencing dieback due to crab grazing.

In acting as anthropogenic predators of *S. reticulatum*, we saw a strong initial response in reduced dieback in the Front marsh at Medouie. Field observations showed dramatic regrowth of *S. alterniflora* in the initial years of crab control (Fig. 7) and field documentation of dieback areas showed decreases in the dieback area in the primary crab grazing area adjacent to the salt marsh ditch. Top-down control of salt marsh dieback through herbivory management showed overall success at Medouie as seen in smaller scale, fenced crab exclusion studies which used localized fenced areas, sunk into the salt marsh soil to exclude crabs from dieback areas, allowing regrowth of *S. alterniflora* (Bertness & Silliman 2008; Holdredge et al. 2009). Fencing itself can be destructive to salt marsh soils that are already compromised, and fencing an entire salt marsh area to control crab access is both impractical and destructive (Holdredge et al. 2009). Additional restoration suggestions include increasing predator populations (Bertness et al. 2014),which can be a more time-consuming restoration than using anthropogenic predation to control crab populations. Our study showed human-control of crab populations to be similarly effective to both crab exclusion (Holdredge et al. 2009) and increasing predator populations (Bertness et al. 2014) in reducing grazing pressure; *S. alterniflora* had time to recruit in dieback areas. This methodology involved only 2 hours of staff time per week with little impact to the salt marsh soils.

2019 (August)

Figure 7

Photomonitoring of the Front marsh facing into Polpis Harbor taken in August, 2019, 2020 and 2022 respectively showing progressive colonization of bare dieback soils by healthy *Spartina alterniflora* in crab trapping locations.

This study initially focused on crab removal in the Front marsh, only adding removal of crabs and dieback area monitoring in the Back marsh in 2023, providing limited sample size to analyze the Back marsh. The early and rapid decrease in salt marsh dieback area in the Front marsh (Table 1, Fig. 2, Fig. 7) provides field-based evidence that sustained trapping during the growing season likely reduced grazing pressure enough for *S. alterniflora* to recolonize. The Back Marsh, although limited in research time frame, also showed an overall decline in dieback from 2022–2025 (Table 1, Fig. 2), offering additional evidence that this management may support salt marsh recovering from crab-driven dieback.

With changing environmental conditions, crab grazing may not be the only environmental pressure driving dieback in salt marshes (Elmer et al. 2014; Marsh et al 2016). In the initial years of this study, crab trapping and removal showed a directionally positive effect in the reduction of dieback mapped in the same growing season (Table 2). However, between 2023–2025, dieback began to increase in the Front marsh (Table 2 and Fig. 3). A visual comparison of the locations of dieback showed dieback increasing in new areas beyond the primary crab grazing and trapping area, particularly along the culvert

and dike road and in high marsh areas along the shrubline at the extent of highest tides (Fig. 3). The Foundation is developing adaptation plans to address this undersized culvert (designed in 2008), with recent modeling indicating its inability to handle current high tides and storms related to sea level rise. This had led to increased impoundment of saltwater at the shrub line and near the culvert following high tides and storm events. While data was not collected to document soil salinity and water impoundment during this study, dieback in these areas may be driven by compounding environmental factors related to climate change and sea level rise (Morris et al. 2002; Alldred et al. 2017; Crotty et al. 2017). The recent decline in salt marsh dieback recovery may indicate that climate-related stressors are beginning to overwhelm the gains achieved through crab-removal.

Crab removal appears to create an opportunity for salt marsh dieback recovery, but whether that opportunity translates into sustained marsh regrowth and expansion may increasingly depend on climate-driven site conditions. In this study, anthropogenic predation of crabs proved an initially successful tool to preserve salt marsh area, but additional management will be needed to adapt to changing environmental pressures with climate change. Monitoring and understanding additional environmental variables including tidal flooding frequency, marsh elevation, and sediment dynamics will be useful in creating a restoration plan for salt marshes impacted by both increased crab grazing pressure and climate change. Additional restoration methods can also be considered including small scale crab exclusion with consideration for salt marsh peat impacts, planting of viable *S. alterniflora* plugs, and thin layer deposition (which was not permitted at the time of this study) (Smith 2025). Preservation of salt marsh areas is of high priority for conservation and coastal resilience, indicating a need to explore all options for restoration.

Declarations

Author Contribution

Karberg, J. contributed to the study conception and design. All authors contributed to field work, data collection, analyses and all drafts of the manuscript. All authors read and approved the final manuscript.

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Data Availability

The data that support the findings of this study are available from the corresponding author and the Nantucket Conservation Foundation, but restrictions apply to the availability of these data so are not publicly available. The data are, however, available upon request and with the permission of the Nantucket Conservation Foundation (NCF).

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Figures



Figure 1

Project Locus at Medouie Creek Wetland Complex in the Polpis Harbor embayment of Nantucket Harbor, Nantucket Island MA. Project area consisted of the Front marsh, south of a historic dike road and connected to the Back marsh by a now-undersized culvert installed as restoration in 2008. X's denote general crab trap locations with the study site.

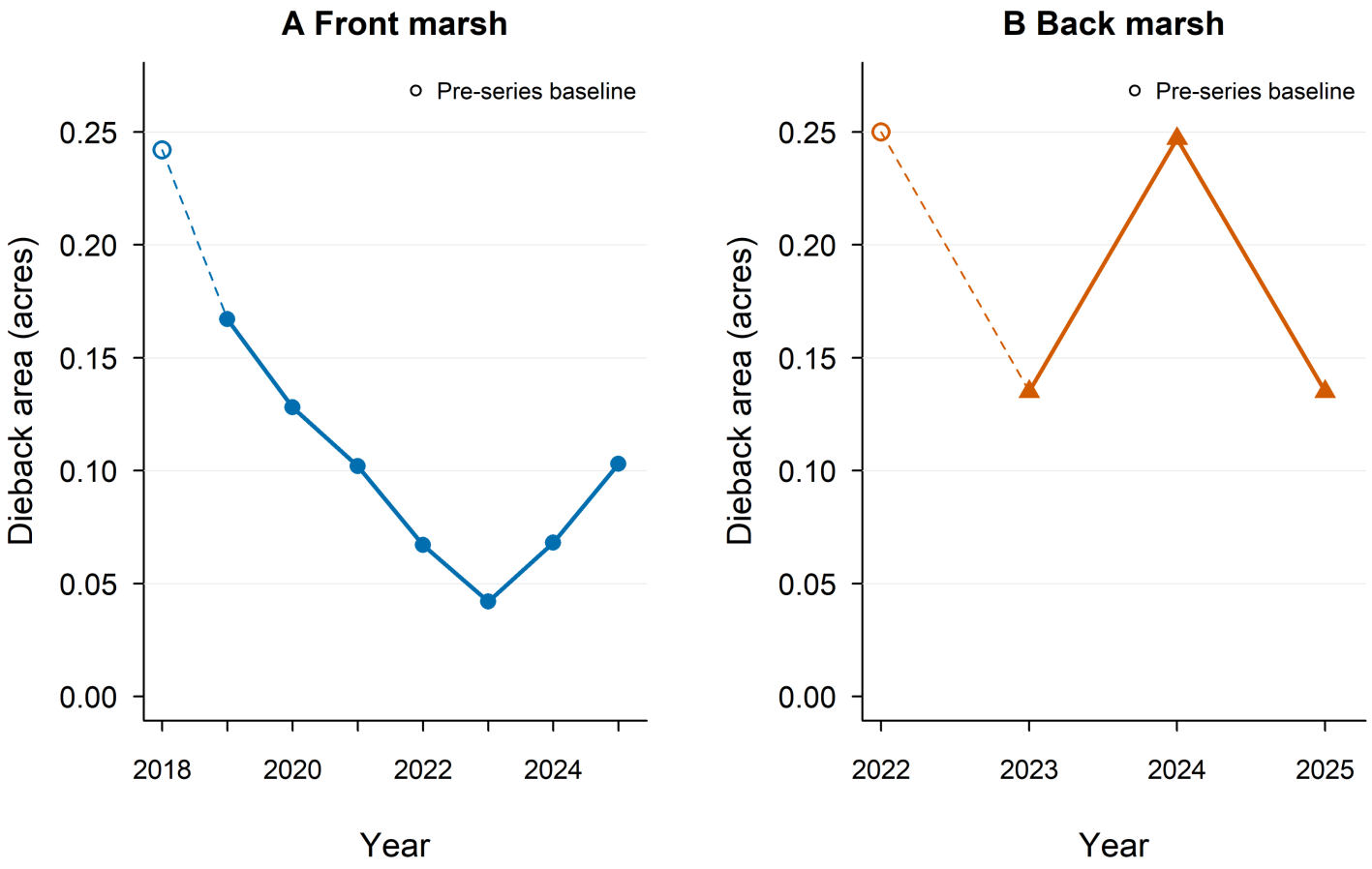


Figure 2

Patterns in annual dieback change for the Front and Back marsh including acreage from the year prior to crab removal in the Front marsh (2018) and the Back marsh (2022).

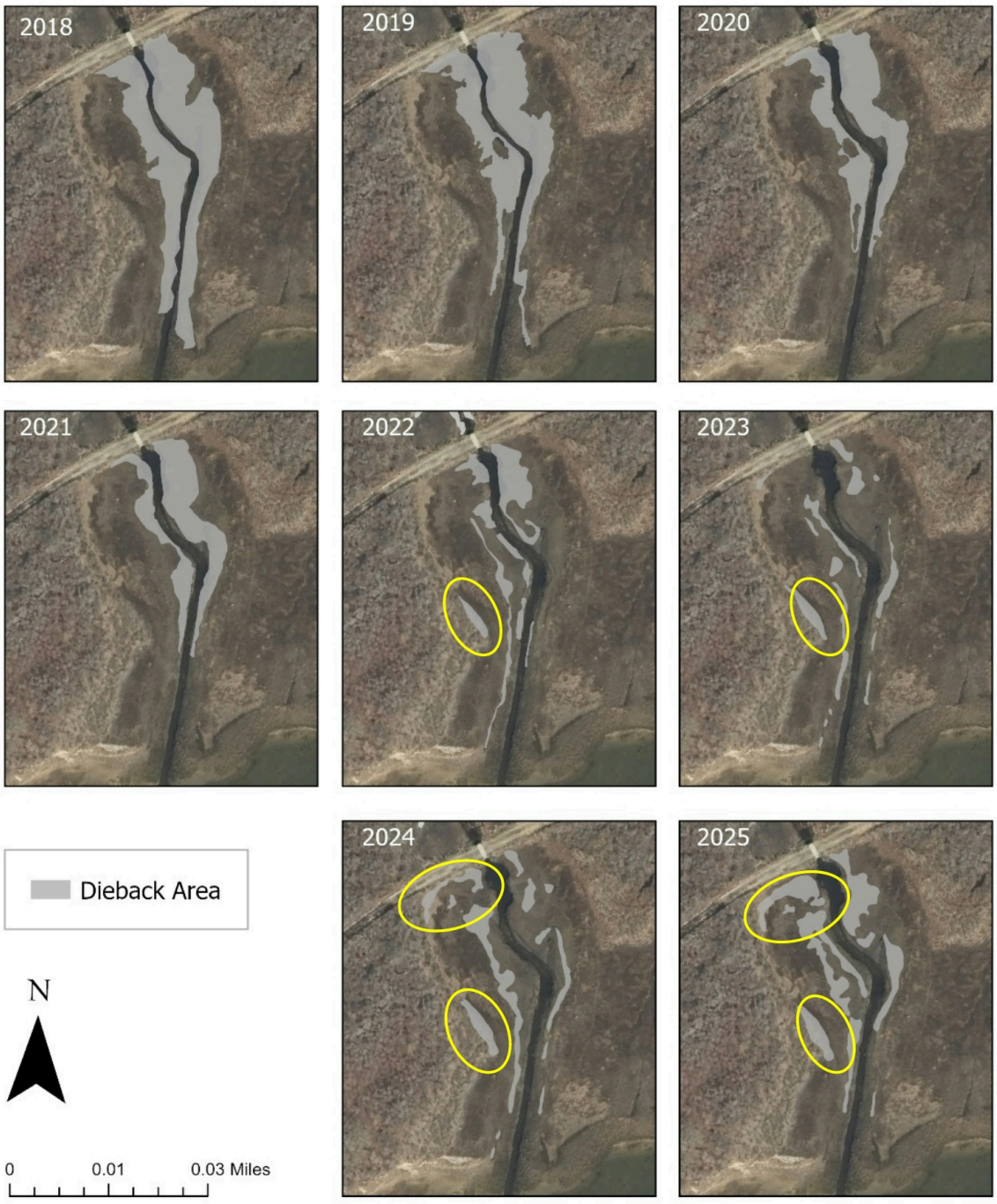


Figure 3

Extent of dieback in the Front marsh from 2018-2025 showing initial dieback along the salt marsh creek with decreases each year up to 2023. In 2022 dieback can be observed along the shrub line in the expansion circle as well as along the dike road adjacent to the undersized culvert.



Figure 4

Extent of Dieback documented in the Back marsh from 2022 prior to crab trapping through 2025. Patterns of dieback were variable annually.

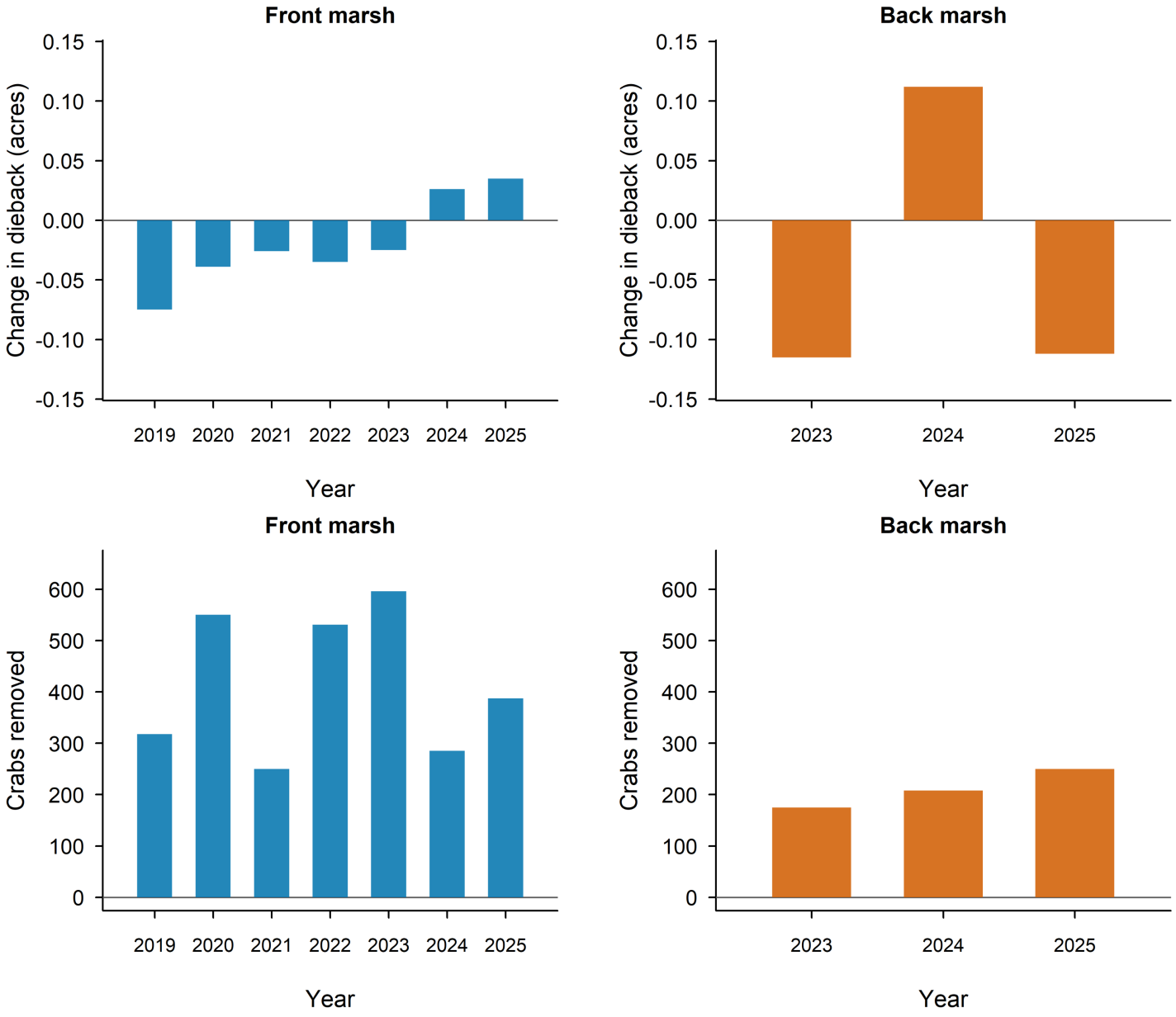


Figure 5

Year to year change in dieback acreage (upper panels) and total crab removal (lower panels) in the Front and Back marsh. Bars denote descriptive observations, not fitted trends.

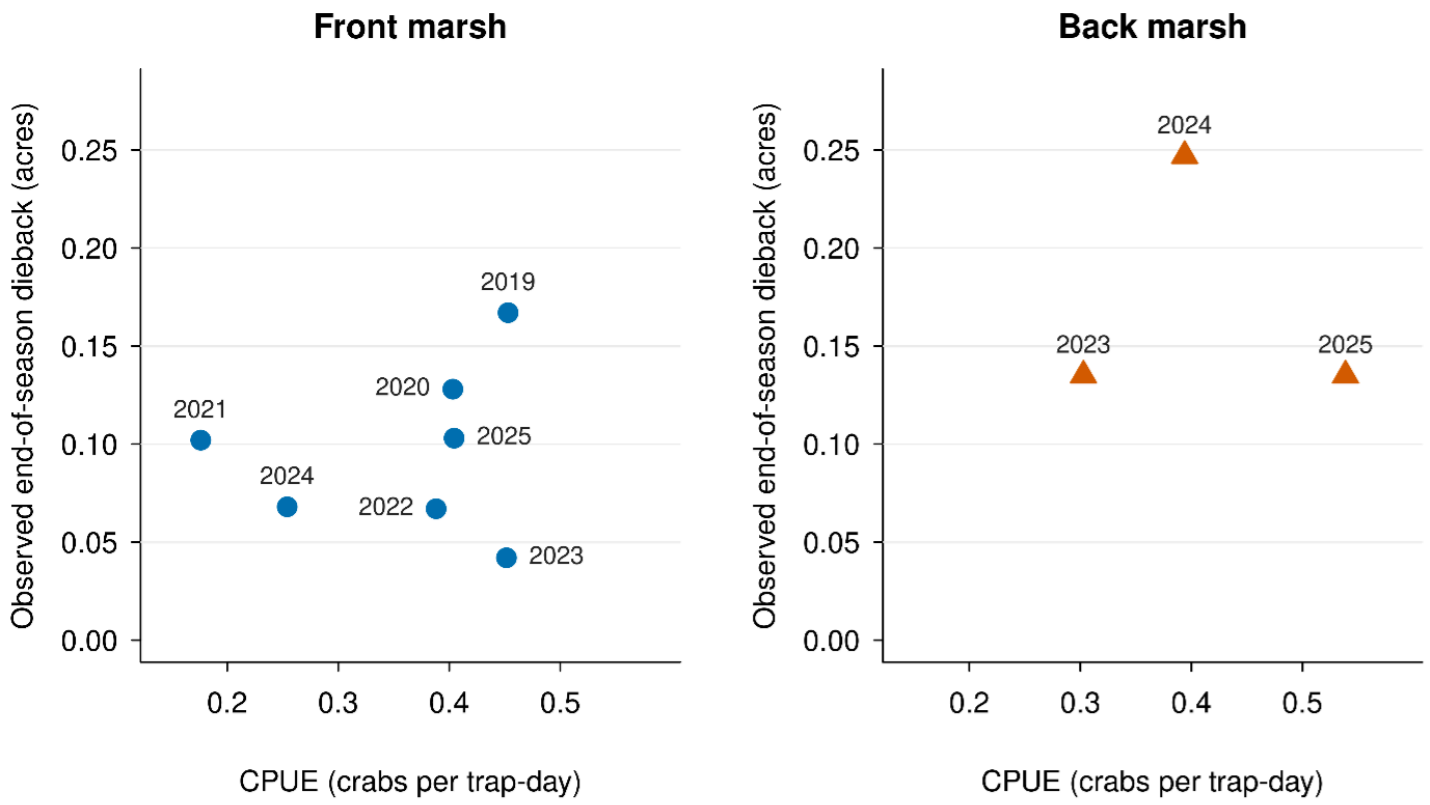


Figure 6

Observed relationship between same-year crab catch per unit effort (CPUE) and end-of-season dieback acreage in the Front and Back marshes.

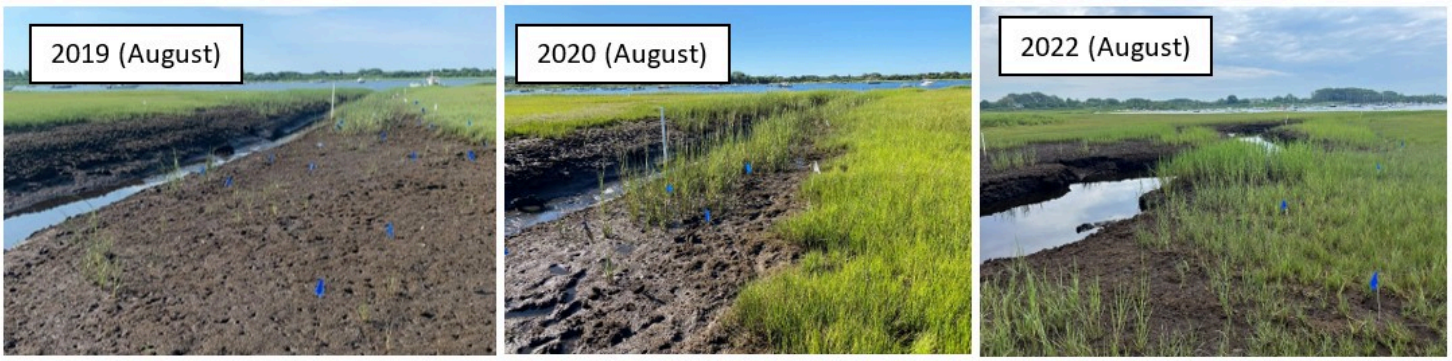


Figure 7

Photomonitoring of the Front marsh facing into Polpis Harbor taken in August, 2019, 2020 and 2022 respectively showing progressive colonization of bare dieback soils by healthy *Spartina alterniflora* in crab trapping locations.