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ORIGINAL ARTICLE

Mitigation policies buffer multiple climate stressors in a socio-ecological salt marsh habitat

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Abstract
Climate change strains human and natural system sustainability worldwide. Plum Island Estuary, Massachusetts (PIE MA) salt marshes are socio-environmental ecosystems experiencing two such climate stressors: sea level rise (SLR) and the mud fiddler crab *Minuca pugnax* (= *Uca pugnax* Smith) range expansion. Salt marshes are important sources of ecosystem functioning and ecosystem services. Uncertainties remain, however, whether SLR and the fiddler crab range expansion will affect PIE ecosystem functioning and services over time by changing marsh area. We, therefore, determined in this study: (1) to what degree PIE marshes provide residents with cultural ecosystem services (e.g., recreation); (2) whether SLR and the fiddler crab range expansion influence marsh area; and (3) whether policy measures influence the direction of marsh services in the face of SLR and multiple potential impacts of range expanding fiddler crabs. We developed a system dynamics model, parameterized with data from stakeholder surveys, the IPCC Report, and a literature review. We modeled low, moderate, and high SLR both with fiddler crabs enhancing marsh erosion and growth, and with and without mitigation strategies on marsh area and recreation. The multi-stressor effects of fiddler crab *erosion* enhancement and high SLR rates decreased marsh area by 2250. Future losses to marsh area caused declines in recreational days. Policy interventions (e.g., erosion reduction and tidal flood mitigation) largely mitigated these losses. Fiddler crab marsh *growth* by itself also strongly mitigated the effects of SLR. These results provide critical transdisciplinary insight for residents, scientists, and practitioners working to enhance PIE sustainability, and for researchers studying how to support environmental sustainability at scale.

Keywords System dynamics · Sustainability · Sea level rise · Species range expansion · Mitigation policy

Introduction

Human activities affect nearly all ecosystems at various spatial and temporal scales (Burrows et al. 2011). Global environmental alterations including climate change, natural resource overexploitation, and land use changes are placing significant burdens on ecosystem resiliency and sustainability by altering ecosystem functioning (i.e., primary production), plant–microbe interactions, biodiversity, and species range limits (Beamish et al. 2004; Mcdonald et al. 2008; Paradis et al. 2008; Poloczanska et al. 2016; Rudgers et al. 2020; Trisos et al. 2020). Large-scale natural system disruptions limit provisioning of ecosystem services to humans, such as food production (e.g., crop pollination), disease mitigation (e.g., bat and bird consumption of mosquitos), protection from storms (e.g., wave energy mitigation by salt marshes), and support for intrinsic well-being (e.g., recreation) (Balvanera et al. 2006; Balvanera et al. 2014; Gedan et al. 2009; Milcu et al. 2013; Twilley et al. 2016). Local-scale policy decisions, however, can mitigate such global or regional environmental changes in coupled human and natural systems via policies grounded in systems thinking (i.e., determining system structure, feedbacks, non-linearities, and leverage points). Sustainability scientists, therefore, need to understand the factors driving human and natural systems over time that matter to stakeholders (e.g., government officials and conservationists) to identify

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leverage points for implementation of effective mitigation policies (Sterman 2012). Therefore, we modeled how climate policies mitigate two climate stressors in the Plum Island Estuary (PIE), Massachusetts, the largest continuous salt marsh in northeastern United States.

The PIE great marsh is an ideal ecosystem to understand system dynamics for sustainability. Salt marshes represent the interface between the land and the sea (Barbier et al. 2011). They are ubiquitous around the world but have been in decline in the northeastern United States (where PIE marshes are located) over the past 400 years due to farming, industrialization, land reclamation, invasive species, and sea level rise (Bromberg and Bertness 2005). Salt marshes provide key ecosystem functions and services for both human and non-human communities (Gedan et al. 2010). PIE marshes in particular serve as critically important nursery habitat for species such as salt marsh sparrows (Wigand et al. 2017). Migratory birds such as the piping plover use PIE as a stopping area to feed, rest, and breed (MaCivor et al. 1990). Salt marshes are sources of various services for human society such as protection from coastal storms, and support for intrinsic well-being (i.e., cultural ecosystem services) (Feagin et al. 2010; Gedan et al. 2010). Cultural ecosystem services provide humans with esthetic, recreational, and intrinsic connections to nature, representing a growing research area (Vejre et al. 2010; Milcu et al. 2013; Kobryn et al. 2018). PIE marshes span multiple towns including Ipswich, Rowley, Newbury, and Newburyport MA (Langston et al. 2020, for a map of PIE, see Supplemental Figure SF1), and likely plays a significant role in providing cultural (i.e., recreational) services to the residents living in these four towns.

Understanding the extent that PIE marshes provide cultural ecosystem services is particularly important given that these services could be threatened by sea level rise and the range expansion of the mud fiddler crab *Minuca pugnax* (= *Uca pugnax* Smith). Due to increased saltwater inundation, sea level rise is projected to shrink total marsh area and cause the dominant species of marsh grass to shift from higher elevation *Spartina patens* (marsh hay) to lower elevation *S. alterniflora* (cordgrass) (Kirwan and Temmerman 2009; Gedan et al. 2010). *M. pugnax* is a burrowing crab that shifted its historic range north to the Plum Island Estuary in 2014 likely due to a warming Gulf of Maine (Sanford et al. 2006; Johnson 2014). The mud fiddler crab influences ecosystem functioning in its historic range south of Cape Cod MA. It increases rates of biogeochemical and nutrient cycling, alters patterns of decomposition, and both increases and decreases sediment stability (i.e., erosion) in the *S. alterniflora*-dominated low marsh (Katz 1980; Bertness 1985; Gribsholt et al. 2003; Holdredge et al. 2010; Johnson 2014; Smith and Green 2015). The mud fiddler crab affects ecosystem functioning in similar ways at PIE versus its historic (i.e., south of Cape Cod) range (Roy et al. in review). Whether these changes in PIE will lead to marsh growth or collapse, however, remains unknown.

Further, it is unclear how changes to the total area of PIE marshes will impact the ability of this ecosystem to provide cultural ecosystem services to local human communities. Reductions in marsh area could change the provisioning of recreational ecosystem services such as fishing, clamming, boating, and birding (Gedan et al. 2010; Carus et al. 2016). Previous research shows feedbacks between environmental changes (such as sea level rise) and a reduced ability for ecosystems to provide ecosystem services (Schröter et al. 2005; Vanbergen et al. 2013). Less well known, however, is the role that mitigation policies play in influencing the direction, degree, and rate of such feedbacks. These uncertainties confound the ability for stakeholders to make decisions that support ecosystem sustainability both globally and in PIE marshes locally as climate change threats emerge.

To better understand the dynamics of this system to support PIE as a vital ecosystem and a resource over time for the public, scientists, and practitioners alike, here we show (1) to what degree Plum Island Estuary (PIE) marshes provide residents with cultural ecosystem services (e.g., recreation); (2) whether sea level rise and the fiddler crab range expansion influence marsh area; and (3) whether policy measures influence the direction of marsh services in the face of sea level rise and multiple potential impacts of range expanding fiddler crabs. Our study explicitly applies causal linkages and feedbacks among marsh area, ecosystem services, and policy measures in this coupled human and natural salt marsh system.

Methods

We utilized a mixed-methods approach that incorporated surveys of key community stakeholders (e.g., scientists, conservationists, and educators), system dynamics modeling (SDM), and data cleaning and calculations of average yearly recreational days in R version 4.0.2 (R Core Team 2021).

Stakeholder surveys

From December 2020 until April 2021, we distributed surveys to stakeholders who lived or worked in Ipswich, Rowley, Newbury, or Newburyport MA. Our survey was developed using the survey platform software Qualtrics XM 2020–2021 and administered online by sending a Qualtrics link to each respondent via email. We sent our survey to an initial list of stakeholders generated by connecting with the PIE Long Term Ecological Research site (PIE LTER) network. To expand our initial pool of stakeholder survey respondents, we utilized a snowball sampling method (Goodman 1961; Reed 2008). This involved interviewing each survey respondent who agreed to be interviewed and asked whether they recommended we contact additional stakeholders to survey and interview (the results of the interviews are not reported here, and data from the interviews are out of the scope of the present research study). We then sent our survey to this new list of stakeholders and continued the process until April 2021. Of the 20 stakeholders we sent surveys to, 16 responded and completed their surveys. The *SARS-CoV-2* pandemic limited the number of individuals we could survey due to an inability to meet respondents in person. In addition, emailing survey stakeholders limited the size of the respondent pool to individuals who were willing to provide email addresses and respond.

Data from the survey responses were used to parameterize the social component of our model (outlined below) and served to answer our first research question: whether and to what extent PIE marshes provide recreational services. For details on our survey questions, see Supplemental Methods 1 (SM1). For a description of how we utilized the survey data to parameterize our model, see "System dynamics model construction".

System dynamics model construction

Concurrent to the survey, we developed a system dynamics model (SDM) using STELLA Architect (ISEE Systems version 1.8, see Fig. 1). SDM, often utilized in sustainability science, connects human and natural systems by creating Stock and Flow Diagrams (SFDs), which define interrelationships among variables within complex systems (Sterman 2012; Mavrommati et al. 2013; Nabavi et al. 2017). SDM has many advantages such as using both numerical and mental data (i.e., qualitative understanding of system structure), as well as demonstrating system behavior *over time*. However, the goal of SDM is *not* for predictive analytics; instead, SDM is utilized to better understand *model behavior and system structure under a variety of scenarios* to inform environmental decision making.

Our system dynamics model determined interrelationships and feedbacks among: (1) PIE marsh area (km^2); (2)

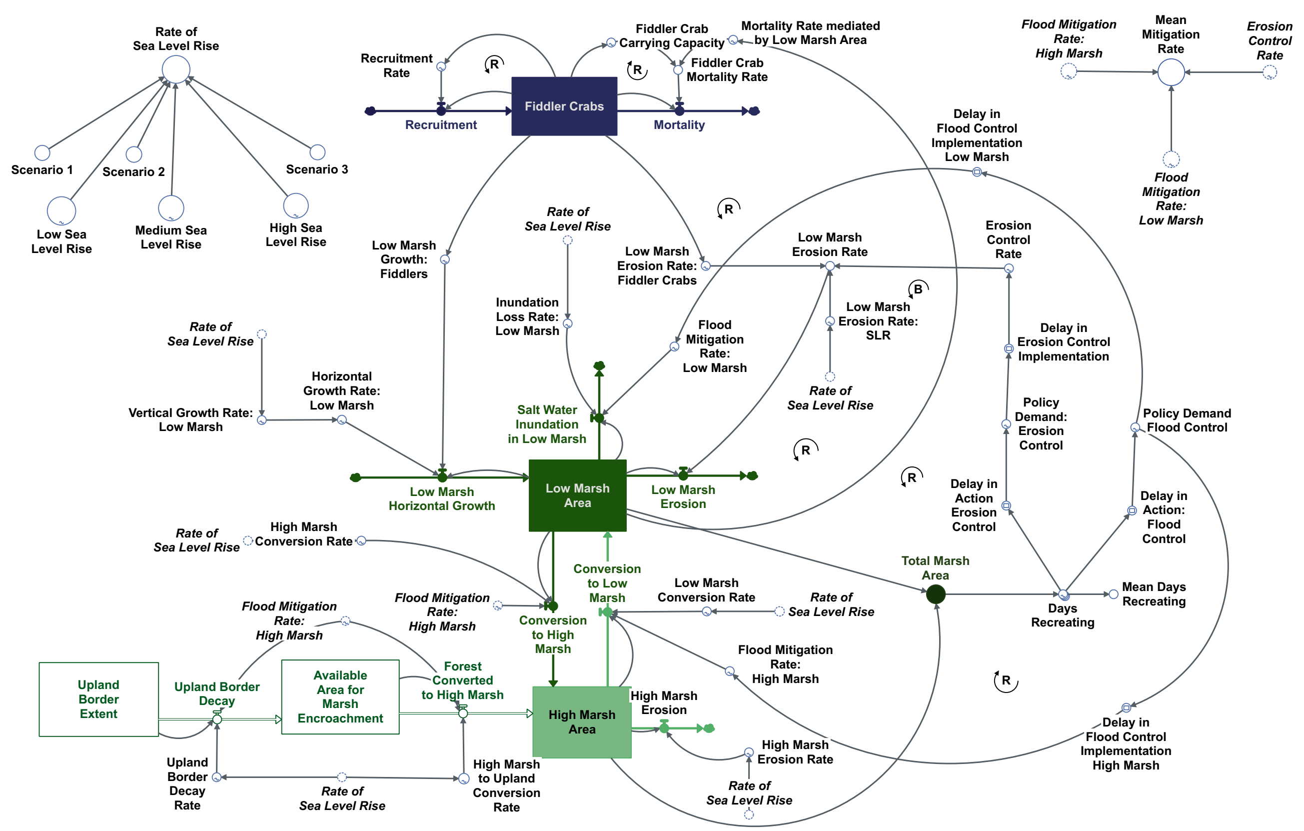

Fig. 1 System dynamics model. STELLA Architect generated stock and flow diagram of Plum Island Estuary (PIE) socio-environmental dynamics. Stocks include (listed in boxes) low marsh area (m^2), high marsh area (m^2), fiddler crab density (fiddler crabs m^{-2}), upland border extent (m^2), and a filler stock representing available area for marsh encroachment (m^2). The key convertor is rate of sea level rise. The model was parameterized and validated with data from the IPCC Fifth Assessment Report, published literature in the PIE LTER network, a literature review, and Roy et al. in review. Note bolded parameters define balancing and reinforcing feedback loops related to the marsh area, recreation, policy demand, and mitigation rate connections

sediment accretion rates (mm year^{-1}); (3) tidal inundation loss rate (% year^{-1}); (4) PIE upland border extent (km^2); (5) sea level rise (mm year^{-1}); (6) fiddler crab population densities (crabs m^{-2}); (7) mean yearly recreation days (days year^{-1}); (8) public demand for policy interventions (% year^{-1}); and (9) marsh loss mitigation rates (% year^{-1}). Our model connects the following stocks: high and low marsh area (i.e., high and low marsh elevation zones) and their feedbacks, upland border extent, and fiddler crab densities. Convertors (rates of change) link these stocks together and define interrelationships among model feedback loops. The following describes in detail model structure, parameterization, and feedback structure. For a visual representation, see Fig. 1.

To parameterize marsh area, upland border extent, and sea level rise, we utilized: (1) the IPCC Fifth Assessment Report (IPCC 2014); (2) a literature review; (3) published research from the PIE LTER network; and (4) a fiddler crab caging field experiment in PIE (Roy et al. in review). Data and model output from Langston et al. (2020) were particularly valuable to compare our model behavior to established literature on PIE salt marsh change over time (especially high, low, and total marsh area and upland border extent). For a description of the three sea level rise rates used in this study, see "Simulation scenarios". We defined starting fiddler crab densities in our model as ~ 1 crabs m^{-2}, which represented fiddler crab densities in 2014 (i.e., their first recorded incidence in PIE, Johnson 2014; Martinez-Soto and Johnson 2020). Using Langston et al. (2020), we defined vertical growth rate (i.e., sediment accretion rate) as 3.04 mm year^{-1} during historical sea level rise (i.e., 2.83 mm year^{-1}) to 9.33 mm year^{-1} for high sea level rise (18.5 mm year^{-1}; for a description of sea level rise scenarios, see "Simulation scenarios"). We then defined tidal inundation loss rate based on an assessment by Watson et al. (2018), who indicate that marsh area in southern New England salt marshes (the closest region to PIE marshes with reported long-term loss rates) was reduced on average by 17.3% over the last 40 years. Therefore, our starting yearly marsh loss rate due to low levels of sea level rise was based on their minimum reported loss rate (1.6%/40 years) and marsh loss rate due to high levels of sea level rise was based on their maximum reported loss rate (40.8%/40 years).

To parameterize mean yearly recreation days, we used data from our stakeholder surveys. All survey data were cleaned and analyzed using the *tidyverse* family of packages version 1.3.1 (Wickham 2021) in the programming language R (R Core Team 2021). Specifically, we determined the mean days the residents engage in each recreational activity across recreational activities per year. To do this, we calculated the total number of days respondents recreated in each activity across respondents. We then determined the aggregate recreation time per year by calculating mean yearly recreational days across all activities. We defined recreation in this study as any activity the community engages in at PIE salt marshes (including salt marsh research). A key assumption in our model is the direct, linear connection between marsh area and recreation. To our knowledge, there are no studies that directly connect ecosystem area to mean days recreating. Vedogbeton and Johnston (2020) in an economic valuation study, however, indicate that increased marsh area provides both available habitat structure and services such as recreation. Therefore, as marsh area increases or decreases in our model, so does potential mean days recreating. Due to the inherent uncertainty here, we kept the relationship between area and recreation days linear.

An important feedback loop in our system dynamics model connected marsh area, recreation, mitigation policies, policy demand, and mitigation rates; subsequently, the relationships among these variables defined the causal connections driving changes in our system. Using these causal connections, we tracked demand by PIE public to implement two mitigation policies: tidal flood mitigation (e.g., runnel digging) and erosion control (e.g., thin layer deposition). For a description of rationale for choosing erosion control and tidal flood mitigation, see Supplemental Methods SM2. We measured policy demand as the percent of PIE public per year who demand policy makers implement both tidal flood mitigation and erosion control, averaged between the policies. We also tracked the rate of salt marsh loss mitigation, measured by the percent of the marsh that is prevented from being lost vertically due to erosion and horizontally due to tidal flooding per year, averaged among low and high marsh tidal flood mitigation and erosion control. We were unable to directly measure either demand or mitigation rates. In addition, a thorough literature review on Google Scholar using the keywords "climate" AND "policy" AND "demand" AND "mitigation" AND "salt marsh" resulted in no relevant publications that explicitly quantified demand for policy interventions in the salt marsh context and determined the rate at which policies will mitigate loss of marsh area. Therefore, both mitigation rate and policy demand were categorized as 'uncertain' parameters. It is also unclear whether fiddler crabs will enhance salt marsh growth, salt marsh erosion, or both; therefore, we also include this uncertainty in our model.

We constructed our model such that, for our uncertain parameters, we utilized conservative estimates of rates of change. These conservative estimates were quantitatively derived based on model behavior over many iterations and scenarios of what represented realistic change over time (i.e., no inflated growth or excessive declines in marsh area). We used this methodology to define starting values for each uncertain parameter. For policy demand by PIE public, we used a starting value ~ 14% year^{-1}. For mitigation rates, we calculated the mean rate among low marsh tidal flood mitigation rate (0.20% year^{-1}), high marsh tidal flood mitigation rate (0.16% year^{-1}), and erosion control (0.41% year^{-1}) as ~ 0.26% year^{-1} total marsh loss mitigation rate (when implemented in our scenarios, see "Simulation scenarios"). For a crab effect on marsh erosion and growth rates, we capped total marsh lost or gained by fiddler crab marsh erosion or growth enhancement, respectively, at 0.35% year^{-1}.

Simulation scenarios

To determine the relationships among marsh area, recreation, policy, and mitigation *over time* (i.e., achieve our second research objective), we used our SDM described above and developed a series of scenarios of environmental change and mitigation. We defined three sea level rise scenarios: low (less than 3 mm per year by 2100), moderate (~ 9 mm per year by 2100), and high (~ 18.5 mm per year by 2100) based on IPCC FAR projections (IPCC 2014) and Langston et al. (2020). For each sea level rise scenario, we simulated two levels of a fiddler crab effect on PIE marsh area and mean yearly recreation days: enhancement of marsh erosion and promotion of marsh growth. We then simulated, for each sea level rise scenario and level of crab effect, the combined effects that both erosion control and tidal flood mitigation have in influencing both the direction of marsh stability (i.e., either loss, growth, or stasis) and recreational services (i.e., mean yearly recreation days).

We ran each simulation in STELLA Architect for 236 years (i.e., 2014–2250), with a delta time (DT) of 0.25 (modeling each year quarterly/seasonally). We chose to begin our simulations in 2014 because this was the first year that fiddler crabs were observed in PIE (Johnson 2014). DeConto et al. (2021) demonstrate the need to develop climate models and models of environmental change that extend beyond the year 2100. Their research shows that without modeling well past the end of this century to the year 2300, we miss many of the tipping points that drive long-term environmental change. Therefore, we chose to run our model to 2250 because it provided us with a robust time series to observe the effects of environmental change and intervention options on marsh area in PIE over time. Nevertheless, many climate models, including the IPCC FAR, run their simulations to 2100 (to represent conditions at the end of the current century); therefore, we also regularly report data that compares model output for 2014, 2100, and 2250 to paint a more complete picture of how our model captures dynamic behaviors of environmental change in this system over time. We also ran sensitivity tests on our baseline scenarios (i.e., low to high sea level rise) for 86 years (i.e., 2100, for the reasons outlined above). We ran all simulations using the built-in interface in STELLA Architect using graphical parameters and switches.

Validation

We validated fiddler crab burrow density against a time-series of fiddler crab densities from 2014 (~ 1 crab m^{-2}) to 2020 (~ 6 crabs m^{-2} by 2020, Martinez-Soto and Johnson 2020). To validate the total spatial area of salt marsh, area of each zone, and upland border extent over time, we utilized data from Kirwan et al. (2011), Langston et al. (2020), and published literature as part of the PIE LTER network (see Table 1). To formally validate burrow density, marsh area, and upland border extent, we used Thiel's U, calculated using the R function TheilU in the *DescTools* package version 0.99.39 (Signorell 2021). To access our data and R scripts, please see https://github.com/MRoy120/PIE_SDM_Validation for validation, and https://github.com/MRoy120/PIE_SDM_Visualizations for data visualization. Stakeholder survey data is not provided to maintain the confidentiality of the survey respondents.

Results

Recreational activities—survey results summary

We surveyed a broad cross-section of stakeholders and experts to represent the community including undergraduate students, naturalists, conservationists, scientists, educators, artists, and employees of the federal government. All surveyed stakeholders lived or worked in Ipswich, Rowley, Newbury, and Newburyport MA at the time of this research study. They all indicated that they have strong cultural connections to PIE marshes, particularly through recreation. Individuals engaged in the following recreational activities: birding, hiking, walking through the marsh, driving by the marsh, fishing by boat and on land, clamming, pleasure boating, and scientific/field research (Table 2). The activities the community engaged with the most, on average, were birding (~ 24 days year^{-1}), scientific/field research (~ 36 days year^{-1}), walking in the marsh (~ 40 days year^{-1}), and driving by the marsh (~ 56 days year^{-1}). The activities the community engaged with the least were hiking (on average ~ 1 day year^{-1}), fishing by land (on average ~ 2 days year^{-1}), and duck hunting (on average ~ 2 days year^{-1}). Despite low averages, many respondents indicated that both the community and the respondents themselves regularly engaged in boating for pleasure (on average ~ 6 days year^{-1}) and fishing (on average ~ 3 days year^{-1}); the highest reported values for boating for pleasure and fishing were 50 and 20 days year^{-1}, respectively. For a complete list of activities, the minimum and maximum times spent engaging in those activities as reported by respondents, and their averages, see Table 2.

Table 1 Model parameter values, uncertain parameters were model derived

Parameter	Initial value: 2014	Unit	Source
Available area for marsh encroachment	0	m^2	Model derived
Fiddler crab erosion and growth	0.35	% $year^{-1}$	Model derived
Fiddler crab mortality	Depends on the starting crab pop	Crabs $year^{-1}$	Cammen et al. (1980)
Fiddler crab mortality rate via low marsh area	Depends on the low marsh extent	Crabs $year^{-1}$	Bertness (1985), Luk and Zajac (2013)
Fiddler crab recruitment	Depends on the starting crab pop	Crabs $year^{-1}$	Cammen et al. (1980)
Fiddler crabs	1	Crabs m^{-2}	Martinez-Soto and Johnson (2020)
High marsh area	~ 33,000	m^2	Langston et al. (2020)
High marsh conversion to low marsh	0.005	m^2 $year^{-1}$	Model derived
High marsh erosion rate	0 – i.e., exceedingly low	m^2 $year^{-1}$	Redfield (1972)
Horizontal growth rate low marsh	0.00017 – calculated from sources	m^2 $year^{-1}$	Kirwan et al. (2011), Langston et al. (2020)
Inundation rate low marsh	0.0005—calculated from source	% $year^{-1}$	Watson et al. (2018)
Low marsh area	~ 7000	m^2	Langston et al. (2020)
Low marsh conversion to high marsh	0	m^2 $year^{-1}$	Model derived
Low marsh erosion rate	0.000022—calculated from sources	m^2 $year^{-1}$	Redfield (1972), Langston et al. (2020)
Mean days recreating	19	% days $year^{-1}$	Survey derived
Mean mitigation rate	0.26	% $year^{-1}$	Model derived
Mean policy demand	14	% $year^{-1}$	Model derived
Sea level rise	2.83	mm $year^{-1}$	IPCC Fifth Assessment Report, Langston et al. (2020)
Total marsh area	~ 40,000	m^2	Deegan et al. (2012), Langston et al. (2020)
Upland border decay rate	0	m^2 $year^{-1}$	Model derived
Upland border extent	~ 2400	m^2	Langston et al. (2020)
Vertical growth rate low marsh	3.04	mm $year^{-1}$	Langston et al. (2020)

Starting values were determined for 2014, when available

Table 2 Recreational activities (days $year^{-1}$) averaged across survey respondents and reported minimum and maximum time spent recreating across respondents

Recreational activity	Mean time (days $year^{-1}$)	Minimum time (days)	Maximum time (days)
Fishing by land	~ 1	0	5
Hiking	~ 2	0	10
Fishing by boat	~ 3	0	20
Duck hunting	~ 3	0	40
Pleasure boating	~ 6	0	50
Scientific/field research	~ 14	0	120
Birding	~ 24	0	340
Walking through the marsh	~ 40	5	340
Driving by the marsh	~ 56	0	300

Simulation series 1—low sea level rise

Mitigation policies implemented in low sea level rise scenarios had small impacts on marsh area or recreation (Fig. 2A–D). In addition, early implementation of tidal flood mitigation in the high marsh *slightly reduced* marsh area relative to low sea level rise by itself (i.e., independent of a fiddler crab effect) by 2250. When we included both tidal flood and erosion control measures, total marsh area slightly increased from the 40.0 km^2 2014 baseline to ~ 40.8 km^2 by 2100 and ~ 41.6 km^2 by 2250 (Fig. 2A). This, however, was lower than when we simulated low sea level rise alone by 2250; marsh area increased to ~ 40.7 km^2 by 2100 and 41.9 km^2 by 2250 from ~ 40 km^2 in 2014 without mitigation strategies (Fig. 2A). In other words, mitigation measures had an insubstantial effect by 2100 and reduced marsh area by 0.3 km^2 by 2250 at low sea level rise. These results represent mostly stable total marsh area over time.

Stability in marsh area was partly caused by low demand for policy change via limited alterations to baseline mean yearly recreational days. Mean yearly recreational days were ~ 20 days $year^{-1}$ by 2250 with and without mitigation policies at low sea level rise compared to ~ 19 days $year^{-1}$ in the present (Fig. 2B). Stable mean yearly recreational days through this period caused nearly no change in demand for policy intervention from ~ 14% $year^{-1}$ of PIE public in the present to ~ 13% $year^{-1}$ of PIE public by 2250 (Fig. 2C). Mean mitigation rates were largely unchanged in this scenario from 0.26% $year^{-1}$ marsh loss prevention in the present to 0.23% $year^{-1}$ marsh loss prevention in 2250 (after

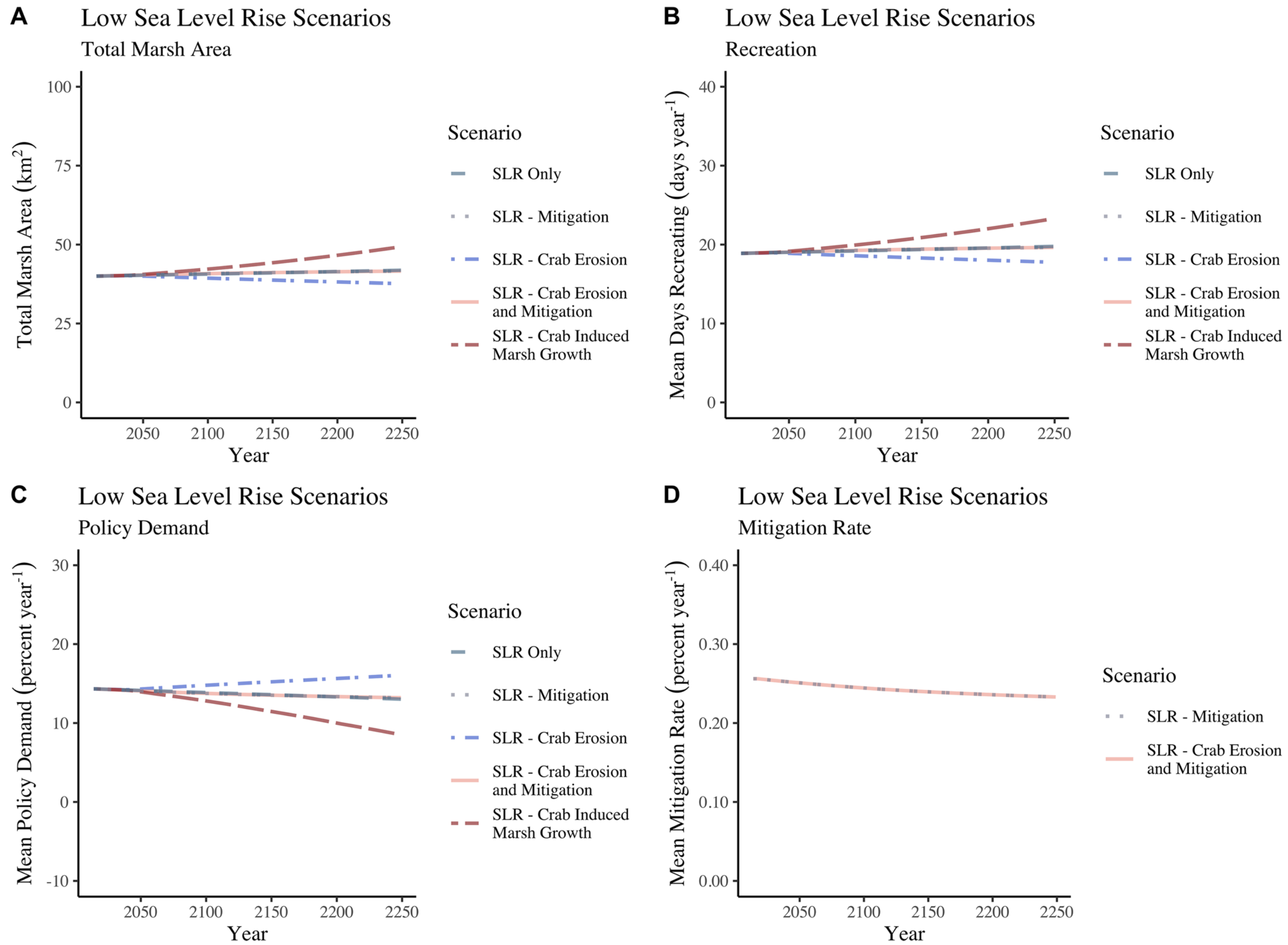

Fig. 2 Low sea level rise scenarios. Plots demonstrating change over time for five scenarios of low sea level rise (~3 mm year^{-1} by 2100): (1) sea level rise (SLR) only (dark gray dashed line); (2) SLR with mitigation policies (light gray dotted line); (3) SLR with fiddler crab erosion enhancement (lavender dashed line); (4) SLR with fiddler crab erosion enhancement and mitigation policies (peach solid line); and (5) SLR with crab induced marsh growth (dashed red line). Figure legends are consistent throughout Figs. 2, 3 and 4. **A** Change in total marsh area (m^2); **B** change in mean yearly number of days residents recreate (i.e., mean days recreating, days year^{-1}); **C** change in mean yearly percent of Plum Island Estuary (PIE) public who demand implementation of marsh loss mitigation policies (i.e., mean policy demand, % year^{-1}); **D** change in mean yearly percent of the marsh that was prevented from being lost by mitigation policies (i.e., mean mitigation rate, % year^{-1}). All models were run from 2014 to 2250 for a total of 236 years

including policy implementation, Fig. 2D). Stability was also facilitated by nearly unchanged rates of vertical marsh growth or tidal flood inundation. Vertical growth changed from 3.04 mm year^{-1} in the present to 3.06 mm year^{-1} by 2250. Tidal flood inundation changed from 0.049% year^{-1} to 0.051% year^{-1} by 2250. As a result, *Spartina patens* dominance in spatial extent over *S. alterniflora* was maintained throughout all low sea level rise scenarios.

We also observed marsh stability over time when we modeled crabs enhancing marsh erosion at low sea level rise. In addition, marsh area was nearly unaffected whether we did or did not include mitigation policies in our simulations (Fig. 2A). Marsh extent slightly decreased with crab erosion from 40 km^2 in the present to ~ 39.4 km^2 by 2100 and ~ 37.6 km^2 by 2250, without mitigation (Fig. 2A). With mitigation policies, marsh extent increased to 40.8 km^2 by 2100 and 41.6 km^2 by 2250; this represents a gain of 1.4 km^2 by 2100 and 4.0 km^2 by 2250 from the baseline with crabs. With crab erosion enhancement at low sea level rise, mean yearly recreation days ranged from ~ 18–20 days year^{-1} by 2250 without and with mitigation policies implemented, respectively (Fig. 2B). Mostly stable mean yearly recreational days again drove low demand for mitigation policies whether we did or did not include mitigation policies; ~ 16% year^{-1} of PIE public without including mitigation and ~ 13% year^{-1} of PIE public including mitigation policies by 2250 from ~ 14% year^{-1} of PIE public in the present (Fig. 2C). We observed the same mean mitigation rates as sea level rise without crab erosion; 0.26% year^{-1} marsh loss prevention in the present to 0.23% year^{-1} marsh loss prevention in 2250 (Fig. 2D).

Our model also demonstrates that crabs could be a net benefit to marsh area as an additional boost to marsh stability. When fiddler crabs were modeled as *enhancing marsh growth* at low sea level rise, total marsh area increased to 42.2 km^2 by 2100 and ~ 49.3 km^2 by 2250 (Fig. 2A); this represents an increase from the baseline in 2014 of ~ 1.5 km^2 by 2100 and ~ 7.4 km^2 by 2250. The moderate increases in spatial extent due to crabs enhancing marsh growth in this scenario increased average recreational days from ~ 19 days year^{-1} in the present to ~ 24 days year^{-1} by 2250 (Fig. 2B). Increased yearly recreational days caused a drop in policy demand to 9% year^{-1} of PIE public by 2250 from 14% year^{-1} of PIE public in the present (Fig. 2C). Finally, in all low marsh scenarios, marsh plant heterogeneity was maintained, such that, the high marsh dominated land cover versus the low marsh (see Supplemental Figure SF2).

Simulation series 2—moderate sea level rise

Moderate sea level rise (i.e., ~ 9 mm year^{-1}) increased total marsh area in the short and long terms, and mitigation policies enhanced these gains. Overall gains came at a cost to marsh plant heterogeneity, however. Without mitigation, total marsh area increased to ~ 41.1 km^2 by 2100 and ~ 44.6 km^2 by 2250 from ~ 40.0 km^2 in 2014 (Fig. 3A). Mitigation policies then caused further increases in marsh growth. By 2100, marsh area was maintained at ~ 41.6 km^2; however, by 2250, marsh area increased to ~ 53.6 km^2, a 9.0 km^2 increase from moderate sea level rise without mitigation policies (Fig. 3A). Mean days recreating increased from ~ 19 days year^{-1} in the present to ~ 26 days year^{-1} and ~ 21 days year^{-1} by 2250 with and without mitigation, respectively (Fig. 3B). This increase in mean yearly recreational days at moderate sea level rise with mitigation policies cut policy demand

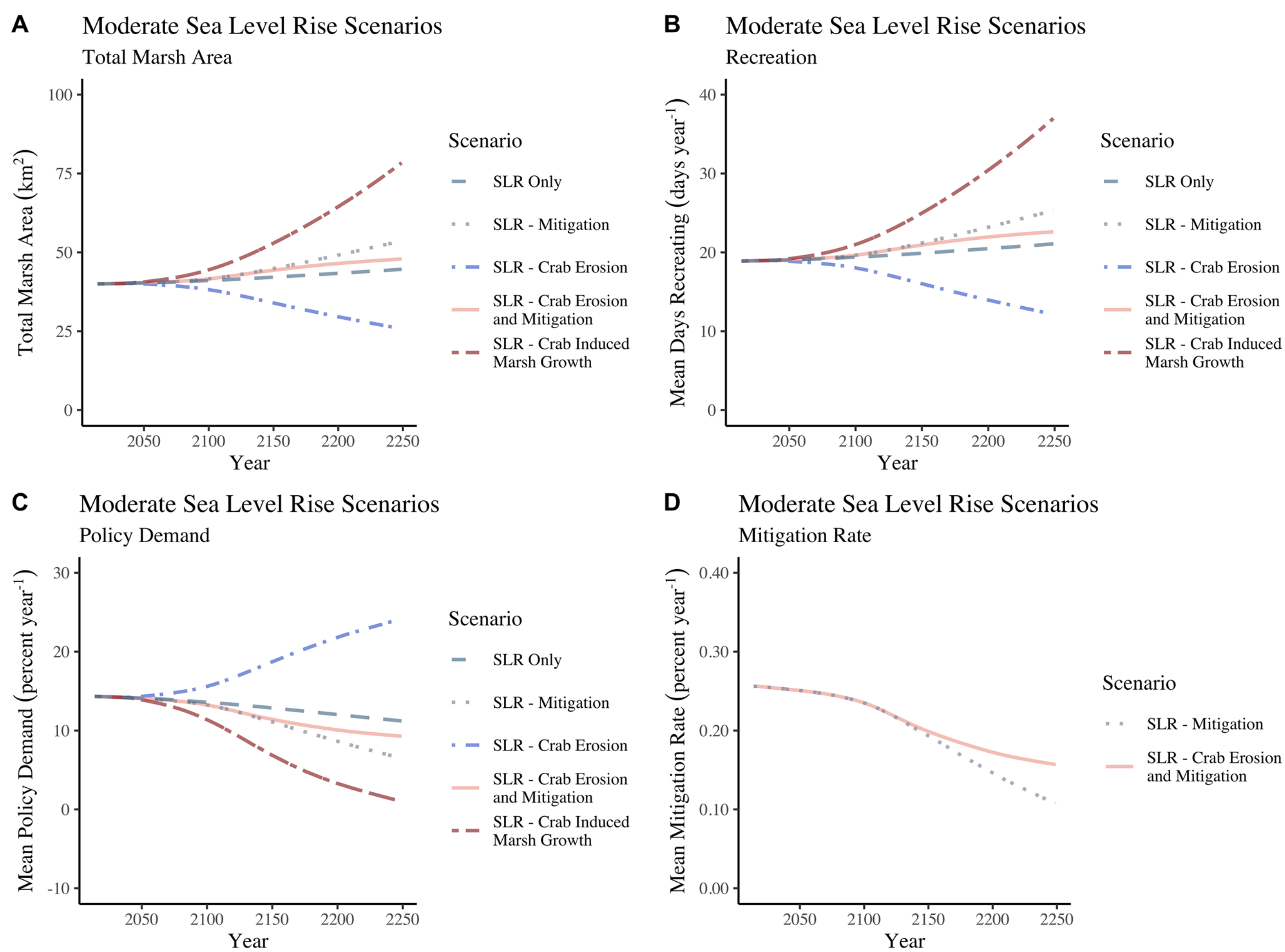

Fig. 3 Moderate sea level rise scenarios. Plots demonstrating change over time for five scenarios of moderate sea level rise (~ 9 mm year−1 by 2100). **A** Change in total marsh area (m^2); **B** change in mean yearly number of days residents recreate (i.e., mean days recreating, days year^{-1}); **C** change in mean yearly percent of Plum Island Estuary (PIE) public who demand implementation of marsh loss mitigation policies (i.e., mean policy demand, % year^{-1}); **D** change in mean yearly percent of the marsh that was prevented from being lost by mitigation policies (i.e., mean mitigation rate, % year^{-1}). All models were run from 2014 to 2250 for a total of 236 years

roughly in half from ~ 14% year^{-1} of PIE public in the present to between ~ 6.6 and 11.1% year^{-1} PIE public by 2250 (lower end represents demand with mitigation implemented, higher end without mitigation, Fig. 3C). Mean mitigation rates also dropped from 0.26% year^{-1} marsh loss prevention per year in the present to 0.11% year^{-1} marsh loss prevention by 2250 (Fig. 3D).

By ~ 2070, however, PIE marshes transition from a high marsh to a low marsh dominated ecosystem. Therefore, PIE is a bigger marsh by 2250, due to increased vertical growth rates resulting from sediment accretion (3.04 mm year^{-1} in the present to 3.65 mm year^{-1} by 2250). However, it is almost entirely composed of *Spartina alterniflora* cordgrass because baseline marsh loss rates more than doubled due to increased tidal flood inundation (0.049% year^{-1} in the present to 0.12% year^{-1} by 2250, Supplemental Figure SF3). The inundation loss rate we observed by 2250 was not high enough to cause marsh loss, but it was high enough to cause marsh zonal transition. We observed cordgrass dominance throughout all moderate sea level rise scenarios in this study.

Fiddler crab erosion enhancement reversed the gains of moderate sea level rise and caused marsh loss in the short and long terms. Erosion and tidal flood mitigation policies offset these losses, however. With crab erosion and without mitigation at moderate sea level rise, salt marsh extent *declined* to ~ 38.2 km^2 by 2100 and ~ 25.8 km^2 by 2250 (Fig. 3A). In other words, crabs reduced marsh area by ~ 2.9 km^2 in 2100 and ~ 18.9 km^2 in 2250 relative to moderate sea level rise simulated alone. This drop in marsh area resulted in a decline of ~ 6 mean recreational days year^{-1} by 2250: ~ 19 days year^{-1} in the present to ~ 13 days year^{-1} by 2250 (Fig. 3B). Mean policy demand increased from 14% year^{-1} in the present to 24% year^{-1} by 2250 (Fig. 3C). After adding both tidal flood mitigation and erosion control to this scenario, however, marsh area increased to ~ 41.6 km^2 by 2100 and 47.9 km^2 by 2250 (Fig. 3A). This represented an increase in marsh area of ~ 3.4 km^2 in 2100 and ~ 22.2 km^2 in 2250 versus crab erosion without mitigation policies. Mean yearly recreational days increased to 23 days year^{-1} (Fig. 3B), which caused policy demand to decrease to ~ 9% year^{-1} by 2250 (Fig. 3C). Mitigation rates declined slightly to ~ 0.24% year^{-1} by 2100, but then continued to decline to ~ 0.16% year^{-1} by 2250 due to an increase in marsh area after 2100 (Fig. 3D). Therefore, implementing mitigation policies early had positive impacts both in the short and long terms.

Fiddler crabs *enhancing marsh growth* had a dramatic impact on marsh area. In this scenario, total marsh area increased to 44.4 km^2 by 2100 and ~ 78.4 km^2 by 2250 (Fig. 3A). The extreme increases in spatial extent due to crabs enhancing marsh growth in this scenario increased average recreational days to ~ 37 days year^{-1} (Fig. 3B). Increased yearly recreational days caused a drop in policy demand to 0.01% year^{-1} of PIE public by 2250 from 14% year^{-1} of PIE public in the present (Fig. 3C).

Simulation series 3—high sea level rise

High sea level rise rates (i.e., ~ 18.5 mm year^{-1}) caused dramatic declines in marsh area over time; mitigation policies prevented some but not all losses. In addition, as with moderate sea level rise, high sea level rise caused the marsh to switch from a high marsh to a low marsh dominated ecosystem throughout all high sea level rise scenarios (Supplemental Figure SF4). Despite gains in marsh extent by 2100 (~ 41.0 km^2), total marsh area *decreased* past this century to ~ 18.5 km^2 by 2250 (Fig. 4A): a reduction of ~ 21.5 km^2 by 2250. Mean days recreating dropped to ~ 9 days year^{-1} by 2250 from ~ 19 days year^{-1} in the present (Fig. 4B), which caused a spike in demand for both tidal flood mitigation and erosion control from ~ 14% year^{-1} of PIE public in the present to ~ 26% year^{-1} of PIE public by 2250 (Fig. 4C).

Implementing mitigation policies prevented some losses in marsh area by high sea level rise, such that marsh area was 41.9 km^2 by 2100 and 29.0 km^2 by 2250 (Fig. 4A): an increase of ~ 10.5 km^2 in 2250 relative to high sea level rise without mitigation. Mean recreational days decreased from ~ 19 days year^{-1} in the present to ~ 14 days year^{-1} by 2250 with mitigation implementation. As Fig. 4C demonstrates, policy demand slightly declined to 13% year^{-1} in 2100 from 14% year^{-1} in the present as both high sea level rise and early mitigation implementation caused the marsh to increase in the short term. Marsh losses after 2100 caused policy demand to spike to 22% year^{-1} (Fig. 4C), however, which caused an increase in the mitigation rate to 0.43% year^{-1} by 2250 (Fig. 4D). Overall marsh losses at high sea level rise were triggered by competing forces. Despite high vertical growth rates by 2250 (9.33 mm year^{-1} by 2250), inundation loss rates jumped 20-fold from 0.05% year^{-1} in the present to 1.0% year^{-1} by 2250.

Marsh erosion caused by fiddler crabs at high sea level rise rates amplified the negative effects of high sea level rise. In this case, tidal flood and erosion control measures mitigated most of the effects of crab erosion on marsh area and recreation. Without mitigation, total salt marsh extent decreased to ~ 37.8 km^2 by 2100 and ~ 10.2 km^2 by 2250 (Fig. 4A). This caused a drop in mean yearly recreational days to ~ 5 days year^{-1} (Fig. 4B). This in turn caused policy demand to peak at ~ 26% year^{-1} by 2189, which then fell slightly to ~ 23% year^{-1} by 2250 (Fig. 4C). After implementing tidal flood and erosion control measures, we saw a similar pattern as with high sea level rise alone (i.e., absent fiddler crab erosion). Marsh area increased to ~ 41.9 km^2 by 2100 and decreased to ~ 28.8 km^2 by 2250 (Fig. 4A). In other words, mitigation policies buffered nearly all negative impacts of simulated fiddler crab enhanced marsh erosion.

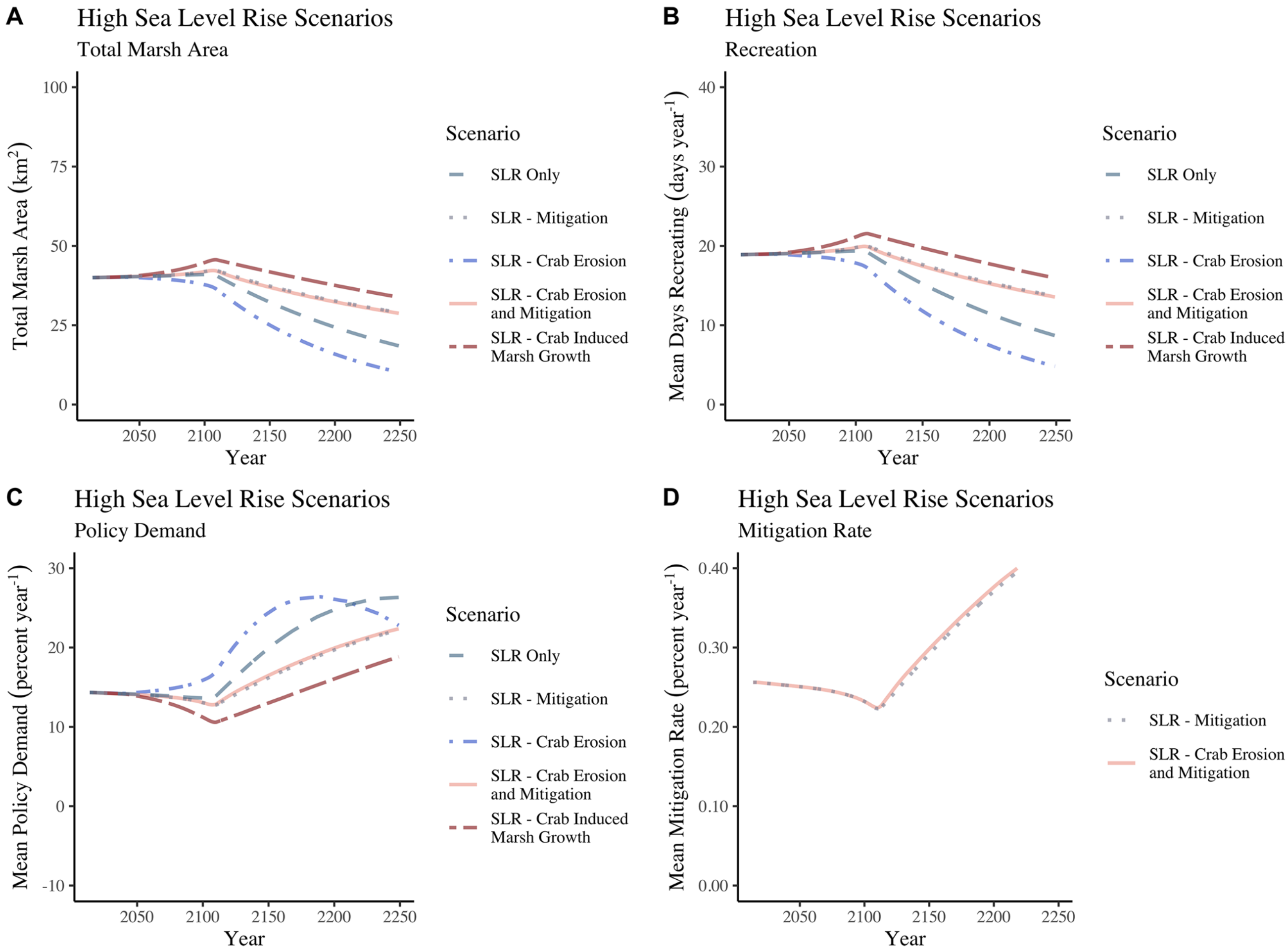

Fig. 4 High sea level rise scenarios. Plots demonstrating change over time for five scenarios of high sea level rise (~18.5 mm year^{-1} by 2100). **A** Change in total marsh area (m^2); **B** change in mean yearly number of days residents recreate (i.e., mean days recreating, days year^{-1}); **C** change in mean yearly percent of Plum Island Estuary (PIE) public who demand implementation of marsh loss mitigation policies (i.e., mean policy demand, % year^{-1}); **D** change in mean yearly percent of the marsh that was prevented from being lost by mitigation policies (i.e., mean mitigation rate, % year^{-1}). All models were run from 2014 to 2250 for a total of 236 years

Mean yearly recreational days declined slightly to ~ 14 days year^{-1} by 2250 compared to ~ 19 days year^{-1} in the present (Fig. 4B). As with high sea level rise alone, policy demand slightly declined to ~ 13% year^{-1} by 2100 as gains in marsh area in the short term and benefits of the mitigation policies cause marsh area to increase by the end of this century, even when including crabs (Fig. 4C). By 2250, policy demand then increased steadily to ~ 22% year^{-1} (Fig. 4C), which then caused mitigation rates to rise to ~ 0.43% year^{-1} (Fig. 4D).

Simulated marsh growth enhancement by fiddler crabs buffered much of the effects of high sea level rise. Marsh area increased to ~ 44.7 km^2 by 2100 and then declined to ~ 33.8 km^2 by 2250 (Fig. 4A). This represented a decrease in marsh area of ~ 6.2 km^2 by 2250, versus 15.3 km^2 total marsh area lost by 2250 with just high sea level rise alone (i.e., no positive crab effect). Mean yearly recreation days decreased from 19 days year^{-1} baseline in the present to 16 days year^{-1} (Fig. 4B). The loss in area caused policy demand to increase to 19% year^{-1} by 2250 from 14% year^{-1} in the present (Fig. 4C). Note that modest increases in marsh area by 2100 caused a deeper drop in policy demand to 11% year^{-1} by 2110. Demand began to increase again as marsh area began to steadily decline (Fig. 4C). For a description of fiddler crab density, upland border extent, and model validation, see Supplemental Results and Supplemental Figure SF5.

Discussion

Overall, our results show a feedback loop among recreational services, marsh area, policy demand, and mitigation policies. Plum Island Estuary (PIE) salt marshes are important sources of cultural ecosystem services. According to stakeholders, residents in Ipswich, Rowley, Newbury, and Newburyport, MA engage in a diverse set of activities in this ecosystem, including birding, fishing, and hiking. Environmental change, however, altered PIE marsh extent and recreational ecosystem service provisioning. When marsh area declined, so did mean number of yearly recreational days, which increased demand by the public for stakeholders to implement mitigation policies. Implementing mitigation policies (i.e., thin layer deposition and digging runnels) buffered the effects of both sea level rise (SLR) and crab erosion, stabilized marsh area, and maintained baseline recreational services. This positive feedback loop of area, recreation, demand, and mitigation was the key to preventing lost marsh area and services at high SLR rates and crab erosion enhancement in the long term. Feedbacks were less important in the short term in all sea level rise only scenarios due to initial increases in sediment accretion rates from SLR before loss. This sediment effect rendered policy feedbacks less important under moderate SLR only scenarios at all time scales. Synergisms between SLR and fiddler crabs with a negative impact, however, led to enhanced loss in both the short term under all scenarios and under moderate SLR. This enhanced loss led to, again, the renewed importance of ecosystem-service-policy feedbacks.

Based on stakeholder responses in our survey, PIE residents broadly have strong positive connections to PIE marshes. Recreation in particular plays a critical role in contributing to the overall human well-being for residents. Deriving intrinsic value via recreational services provisioning from our surveys proved a vital component to our understanding of the human–natural dynamics of PIE salt marshes. Our model demonstrates that when SLR and fiddler crab erosion decrease marsh area, demand increases for policies that mitigate environmental change and prevents further loss to marsh area. Without mean yearly recreational days in our model, we would miss a key component to the feedback loops that define this system. In addition, the 16 stakeholders in our survey were from a range of career stages and connections with the marsh, including undergraduate students, conservationists, scientists, educators, artists, federal employees, and naturalists. Many of the respondents have been working or living among PIE salt marshes for more than 3 decades. Therefore, our 16 stakeholder responses provide a robust cross-section of the population to represent the recreational interests of this community. Taken together, our results demonstrate the importance of both understanding intrinsic well-being and looking to diverse perspectives for both cultural ecosystem service valuation and sustainable environmental conservation.

Marsh area increased by the year 2250 at moderate sea level rise rates. Marsh area also increased in all scenarios by 2100 (absent crab erosion). Increased area at moderate sea level rise in the long term and for all scenarios in the short term was due to increased accretion rates caused by higher oceanic imports of sediment included in our model. Several studies indicate that moderate sea level rise will cause marsh area to increase as a result of increased import of both sediment and nutrients (Kirwan and Temmerman 2009; Kirwan et al. 2011; Kirwan and Megonigal 2013; Wigand et al. 2017). Langston et al. (2020) observed similar short-term increases in marsh area, where PIE marsh area increased by 4, 7, and 9% from the present to the year 2100 at low, moderate, and high sea level rise rates, respectively. This came at a cost to marsh heterogeneity in our study, however. PIE marshes shift from a *Spartina patens* to a *S. alterniflora*-dominated system at moderate and high sea level rise scenarios, likely due to an inability for *S. patens* to tolerate increased salt stress (Morris et al. 2013; Johnson and Williams 2017a, b). When fiddler crabs enhance marsh erosion, they reverse gains in marsh area in all scenarios in the short term (i.e., by 2100) and at moderate sea level rise in the long term. Although a larger marsh, PIE marsh resiliency was subsequently weakened by the homogenizing effect of moderate sea level rise. Mitigation policies buffered crab induced losses, however. Taken together, when sea level rise and fiddler crab erosion enhancement reduced marsh area and services provisioning, mitigation policies prevented lost area and recreational services in PIE salt marshes.

Before utilizing either or both thin layer deposition or runnel digging, however, it is important to empirically study the cost, feasibility, and overall multiple benefits to their use moving forward in PIE. Our research offers insight into this endeavor by providing stakeholders with a tool to track the effectiveness of the mitigation policies utilized here, and other policies unexplored in this model (e.g., directed purchase of private land for conservation). We explicitly defined how alterations to marsh area and recreational services impacted demand for policy measures, and how this in turn affected the rate that mitigation policies prevented losses in marsh area directly and mean yearly recreational days indirectly (via connections to marsh area). These parameters were defined by relying on mental (i.e., qualitative) data and model output over many iterations; however, we challenge scientists, practitioners, and policymakers to measure and define uncertain parameters in this model for PIE marshes locally and for salt marsh ecosystems globally. By engaging in such model updating, stakeholders will be better equipped to determine whether and to what extent mitigation strategies buffer environmental change impacts in both PIE and global marsh ecosystems.

Low and high marsh ecosystems support different assemblages of species and are structured differently based on the plants living in each zone (high versus low, Levin et al. 2001; Feagin et al. 2010). High marsh systems support a greater mix of marine and terrestrial species relative to low marsh systems, which tend to be predominantly marine

(Johnson et al. 2016; Nelson et al. 2018). This is in part due to the proximity of the high marsh to the terrestrial boundary and the low marsh to brackish creeks that are connected to oceanic sources (Levin et al. 2001). In addition, *S. patens* (i.e., high marsh dominated plant) provides a thick "thatch" like covering, creating a microclimate that reduces ambient temperature and increases relative humidity (Johnson and Williams 2017a; b). Many species such as air breathing snails and terrestrial arthropods (e.g., spiders) rely on the *S. patens* microclimate for their survival to prevent overheating and desiccation (Johnson and Williams 2017a; b). The low marsh also has higher rates of sediment deposition relative to the high marsh (Kirwan et al. 2012; Langston et al. 2020). Proximity to tidal creeks and morphological characteristics of *S. alterniflora* enhance its ability to baffle wave energy and trap suspended sediment (Wang et al. 2008). Therefore, food web dynamics and ecosystem structure, functioning, and services provisioning could change as percent cover of high marsh *S. patens* is reduced and *S. alterniflora* is increased, despite increases in total area (Johnson 2011; Sotka and Byers 2019).

Our research simulates both positive and negative effects of fiddler crabs on marsh stability over time. Researchers should utilize randomized controlled laboratory and field experiments that determine the role that fiddler crabs play in driving either salt marsh stability or collapse in PIE. This is especially important considering what we observed when we simulated fiddler crabs enhancing marsh *growth*. Strikingly, fiddler crabs could negate the necessity for policymakers to implement mitigation strategies described above. When we modeled fiddler crabs as enhancers of low marsh growth rather than erosion, their effect is strong enough to diminish the effects of sea level rise. For high sea level rise rate simulations, *M. pugnax* prevented some marsh losses by the year 2250. The effect of moderate sea level rise and fiddler crab growth is especially prominent: marsh area nearly doubled by the end of 2250. Fiddler crabs as marsh growth enhancers were better mitigators of severe sea level rise than the combination of erosion mitigation and tidal flood mitigation. Therefore, it is critical to experimentally determine whether and to what extent fiddler crabs influence dynamics of marsh structure over time. Altering structure will likely significantly change historic ecosystem functioning in PIE marshes through shifts in species composition and diversity.

If crabs display marsh growth enhancement in PIE as demonstrated in our model, the fiddler crab range expansion could by itself provide an additional novel ecosystem function and service to PIE marshes. Roy et al. (in review) demonstrated that *M. pugnax* burrowing behavior indirectly facilitated an increase in aboveground biomass of *Spartina alterniflora* in the low marsh, an important factor in promoting both vertical and horizontal marsh growth (Kirwan et al. 2012; Wigand et al. 2017). Much contemporary research analyzes the ability for the natural world to prevent or mitigate the effects of environmental change on global ecosystems (Heal et al. 2005; Goldstein et al. 2012). The recent appearance of the fiddler crab could provide much needed natural capital, particularly at high rates of sea level rise and as crab density increases over time in PIE. In addition, *Minuca pugnax*, based on our simulations, could reach densities in its non-historic habitat (i.e., PIE) comparable to historic (i.e., south of Cape Cod MA) populations by the end of this century (Aspey 1978; Martínez-Soto and Johnson 2020). The combination of increased crab densities and an enhancement of marsh stability could help support PIE marsh ecosystem services that we observed in this study. Scientists empirically studying the exact impacts of fiddler crabs on the competing dynamics of marsh growth and loss in PIE will be more important than ever with continued persistent environmental change.

Conclusion

Our study is the first of its kind to explicitly define feedbacks among marsh area, cultural ecosystem services (i.e., recreation), policy demand, and mitigation policies (i.e., thin layer deposition and runnel digging). Moreover, we demonstrated how environmental changes, and their mitigation, influence marsh sustainability in the short and long terms. Progressively higher rates of sea level rise and the range expansion of *Minuca pugnax* (when modeled as a marsh *erosion* enhancer) created a positive feedback loop of decreased marsh area. A smaller marsh meant residents had fewer recreational days in Plum Island Estuary (PIE) salt marshes. As an important source of recreation for PIE residents, fewer recreational days caused residents to demand that policymakers (e.g., conservationists, elected officials, state and federal employees, etc.) implement conservation measures to mitigate both environmental stressors. Digging runnels and thin layer deposition prevented lost area, buffering the impacts of sea level rise and the fiddler crab range expansion. In addition, when we modeled *M. pugnax* as marsh *growth* enhancers, they prevented decreased marsh area better than the combined positive effects of digging runnels and applying thin layer deposition. We, therefore, challenge researchers to study specific crab effects, and the overall success of mitigation efforts modeled here, on marsh erosion and accretion over time.

Transdisciplinary methodologies that integrate natural science, social science, modeling, and policy used in climate science and sustainability bring together disparate fields to tease apart so-called 'wicked problems'. Our study provides additional support to use systems thinking and dynamics, as well as transdisciplinary methodologies, to support environmental sustainability. While our research focused locally on

PIE salt marshes, this research also provides sustainability scientists in general with best practices to support a range of ecosystems and determine otherwise unobserved socio-environmental feedbacks. Therefore, our study builds on existing literature and provides a novel tool for stakeholders and researchers alike to promote environmental sustainability at both local and global scales.

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References

Aspey WP (1978) Fiddler crab behavioral ecology: burrow density in *Uca pugnax* (Smith) and *Uca pugilator* (Bosc) (Decapoda Brachyura). Crustaceana 34(3):235–244

Balvanera P et al (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol Lett 9(10):1146–1156. https://doi.org/10.1111/j.1461-0248.2006.00963.x

Balvanera P et al (2014) Linking biodiversity and ecosystem services: current uncertainties and the necessary next steps. Bioscience 64(1):49–57. https://doi.org/10.1093/biosci/bit003

Barbier EB et al (2011) The value of estuarine and coastal ecosystem services. Ecol Monogr 81(2):169–193. https://doi.org/10.1890/10-1510.1

Beamish RJ et al (2004) Regimes and the history of the major fisheries off Canada's west coast. Prog Oceanogr 60(2–4):355–385. https://doi.org/10.1016/j.pocean.2004.02.009

Bertness MD (1985) Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. Ecology 66(3):1042–1055

Bromberg DK, Bertness MD (2005) New England salt marsh losses using reconstructing historical maps. Estuaries 28(6):823–832

Burrows MT et al (2011) The pace of shifting climate in marine and terrestrial ecosystems. Science 334(6056):652–655. https://doi.org/10.1126/science.1210288

Cammen LM, Seneca ED, Stroud LM (1980) Energy flow through the fiddler crabs Uca pugnax and U. minax and the Marsh Periwinkle Littorina irrorata in a North Carolina Salt Marsh. Am Midl Nat 103(2):238–250

Carus J, Paul M, Schröder B (2016) Vegetation as self-adaptive coastal protection: reduction of current velocity and morphologic plasticity of a brackish marsh pioneer. Ecol Evol 6(6):1579–1589. https://doi.org/10.1002/ece3.1904

DeConto RM et al (2021) The Paris Climate Agreement and future sea-level rise from Antarctica. Nature 593(7857):83–89. https://doi.org/10.1038/s41586-021-03427-0

Deegan LA et al (2012) Coastal eutrophication as a driver of salt marsh loss. Nature 490(7420):388–392. https://doi.org/10.1038/nature11533

Feagin RA et al (2010) Salt marsh zonal migration and ecosystem service change in response to global sea level rise: a case study from an urban region. Ecol Soc. https://doi.org/10.5751/ES-03724-150414

Gedan KB, Silliman BR, Bertness MD (2009) Centuries of human-driven change in salt marsh ecosystems. Annu Rev Mar Sci 1(1):117–141. https://doi.org/10.1146/annurev.marine.010908.163930

Gedan KB et al (2010) The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. Clim Change 106(1):7–29. https://doi.org/10.1007/s10584-010-0003-7

Goldstein JH et al (2012) Integrating ecosystem-service tradeoffs into land-use decisions. Proc Natl Acad Sci USA 109(19):7565–7570. https://doi.org/10.1073/pnas.1201040109

Goodman LA (1961) Snowball sampling. Ann Math Stat 32(1):148–170. https://doi.org/10.1214/aoms/1177705148

Gribsholt B, Kostka JE, Kristensen E (2003) Impact of fiddler crabs and plant roots on sediment biogeochemistry in a Georgia saltmarsh. Mar Ecol Prog Ser 259:237–251. https://doi.org/10.3354/meps259237

Heal GM et al (2005) Protecting natural capital through ecosystem service districts. SSRN Electron J. https://doi.org/10.2139/ssrn.279114

Holdredge C et al (2010) Fiddler crab control of cordgrass primary production in sandy sediments. Mar Ecol Prog Ser 399:253–259. https://doi.org/10.3354/meps08331

IPCC (2014) Climate change 2014: impacts, adaptation and vulnerability—contributions of the Working Group II to the Fifth Assessment Report—summary for policymakers. Climate change 2014: impacts, adaptation and vulnerability—Contributions of the Working Group II to the Fifth Assessment Report, pp 1–32. https://doi.org/10.1016/j.renene.2009.11.012

Johnson DS (2011) High-marsh invertebrates are susceptible to eutrophication. Mar Ecol Prog Ser 438:143–152. https://doi.org/10.3354/meps09306

Johnson DS (2014) Fiddler on the roof: a northern range extension for the marsh fiddler crab, *Uca pugnax*. J Crustac Biol 34(5):671–673. https://doi.org/10.1017/CBO9781107415324.004

Johnson DS, Williams BL (2017a) Sea level rise may increase extinction risk of a saltmarsh ontogenetic habitat specialist. Ecol Evol 7(19):7786–7795. https://doi.org/10.1002/ece3.3291

Johnson DS, Williams BL (2017b) Sea level rise may increase extinction risk of a saltmarsh ontogenetic habitat specialist. Ecol Evol. https://doi.org/10.1002/ece3.3291

Johnson DS et al (2016) Saltmarsh plant responses to eutrophication. Ecol Appl 26(8):2647–2659

Katz LC (1980) Effects of burrowing by the fiddler crab, *Uca pugnax* (Smith). Estuar Coast Mar Sci 11(2):233–237. https://doi.org/10.1016/S0302-3524(80)80043-0

Kirwan ML, Megonigal JP (2013) Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504(7478):53–60. https://doi.org/10.1038/nature12856

Kirwan M, Temmerman S (2009) Coastal marsh response to historical and future sea-level acceleration. Quat Sci Rev 28(17–18):1801–1808. https://doi.org/10.1016/j.quascirev.2009.02.022

Kirwan ML et al (2011) Rapid wetland expansion during European settlement and its implication for marsh survival under modern sediment delivery rates. Geology 39(5):507–510. https://doi.org/10.1130/G31789.1

Kirwan ML et al (2012) On the relationship between sea level and *Spartina alterniflora* production. Ecosystems 15(1):140–147. https://doi.org/10.1007/s10021-011-9498-7

Kobryn HT et al (2018) Cultural ecosystem values of the Kimberley coastline: an empirical analysis with implications for coastal and marine policy. Ocean Coast Manag 162:71–84. https://doi.org/10.1016/j.ocecoaman.2017.09.002

Langston AK et al (2020) Modeling long-term salt marsh response to sea level rise in the sediment-deficient Plum Island Estuary, MA.

Limnol Oceanogr 65(9):2142–2157. https://doi.org/10.1002/lno.11444

Levin LA et al (2001) The function of marine critical transition zones and the importance of sediment biodiversity. Ecosystems 4(5):430–451. https://doi.org/10.1007/s10021-001-0021-4

Luk YC, Zajac RN (2013) Spatial ecology of fiddler crabs, Uca pugnax, in Southern New England Salt Marsh landscapes: potential habitat expansion in relation to salt marsh change. Northeast Nat 20(2):255–274. https://doi.org/10.1656/045.020.0213

MaCivor LH, Melvin SM, Griffin CR (1990) Effects of research activity on piping plover nest predation. J Wildl Manag 54(3):443–447

Martínez-Soto KS, Johnson DS (2020) The density of the Atlantic marsh fiddler crab (*Minuca pugnax*, Smith, 1870) (Decapoda: Brachyura: Ocypodidae) in its expanded range in the Gulf of Maine, USA. J Crustac Biol 40(5):544–548. https://doi.org/10.1093/jcbiol/ruaa049

Mavrommati G, Bithas K, Panayiotidis P (2013) Operationalizing sustainability in urban coastal systems: a system dynamics analysis. Water Res 47(20):7235–7250. https://doi.org/10.1016/j.watres.2013.10.041

Mcdonald RI, Kareiva P, Forman RTT (2008) The implications of current and future urbanization for global protected areas and biodiversity conservation. Biol Conserv 141(6):1695–1703. https://doi.org/10.1016/j.biocon.2008.04.025

Milcu AI et al (2013) Cultural ecosystem services: a literature review and prospects for future research. Ecol Soc. https://doi.org/10.5751/ES-05790-180344

Morris JT, Sundberg K, Hopkinson CS (2013) Salt marsh primary production and its responses to relative sea level and nutrientsin estuaries at plum island, Massachusetts, and North Inlet, South Carolina, USA. Oceanography 26(3):78–84. https://doi.org/10.5670/oceanog.2013.48

Nabavi E, Daniell KA, Najafi H (2017) Boundary matters: the potential of system dynamics to support sustainability? J Clean Prod 140:312–323. https://doi.org/10.1016/j.jclepro.2016.03.032

Nelson JA et al (2018) Feedbacks between nutrient enrichment and geomorphology alter bottom-up control on food webs. Ecosystems 22(2):229–242. https://doi.org/10.1007/s10021-018-0265-x

Paradis A et al (2008) Role of winter temperature and climate change on the survival and future range expansion of the hemlock woolly adelgid (*Adelges tsugae*) in eastern North America. Mitig Adapt Strat Glob Change 13(5–6):541–554. https://doi.org/10.1007/s11027-007-9127-0

Poloczanska ES et al (2016) Responses of marine organisms to climate change across oceans. Front Mar Sci 3(May):1–21. https://doi.org/10.3389/fmars.2016.00062

R Core Team (2021) R: a language and environment for statistical computing. Foundation for Statistical Computing. Vienna, Austria. Available at: https://www.r-project.org/

Redfield AC (1972) Development of a New England salt marsh. Ecol Monogr 42(2):201–237

Reed MS (2008) Stakeholder participation for environmental management: a literature review. Biol Conserv 141(10):2417–2431. https://doi.org/10.1016/j.biocon.2008.07.014

Rudgers JA et al (2020) Climate disruption of plant–microbe interactions. Annu Rev Ecol Evol Syst 51:561–586. https://doi.org/10.1146/annurev-ecolsys-011720-090819

Sanford E et al (2006) Larval tolerance, gene flow, and the northern geographic range limits of fiddler crabs. Ecology 87(11):2882–2894. https://doi.org/10.1890/0012-9658(2006)87

Schröter D et al (2005) Ecology: ecosystem service supply and vulnerability to global change in Europe. Science 310(5752):1333–1337. https://doi.org/10.1126/science.1115233

Signorell A (2021) Package " DescTools"

Smith SM, Green CW (2015) Sediment suspension and elevation loss triggered by atlantic mud fiddler crab (*Uca pugnax*) bioturbation in salt marsh dieback areas of Southern New England. J Coast Res 31(1):88–94. https://doi.org/10.2112/JCOASTRES-D-12-00260.1

Sotka EE, Byers JE (2019) Not so fast: promoting invasive species to enhance multifunctionality in a native ecosystem requires strong(er) scrutiny. Biol Invasions 21(1):19–25. https://doi.org/10.1007/s10530-018-1822-0

Sterman JD (2012) Sustaining sustainability: creating a systems science in a fragemented academy and polarized world. In: Sustainability science: the emerging paradigm and the urban environment, pp 21–56

Trisos CH, Merow C, Pigot AL (2020) The projected timing of abrupt ecological disruption from climate change. Nature 580(7804):496–501. https://doi.org/10.1038/s41586-020-2189-9

Twilley RR et al (2016) Co-evolution of wetland landscapes, flooding, and human settlement in the Mississippi River Delta Plain. Sustain Sci 11(4):711–731. https://doi.org/10.1007/s11625-016-0374-4

Vanbergen AJ et al (2013) Threats to an ecosystem service: pressures on pollinators. Front Ecol Environ 11(5):251–259. https://doi.org/10.1890/120126

Vedogbeton H, Johnston RJ (2020) Commodity consistent meta-analysis of wetland values: an illustration for coastal marsh habitat, environmental and resource economics. Springer Netherlands, Amsterdam

Vejre H, Jensen FS, Thorsen BJ (2010) Demonstrating the importance of intangible ecosystem services from peri-urban landscapes. Ecol Complex 7(3):338–348. https://doi.org/10.1016/j.ecocom.2009.09.005

Wang A, Wang Y, Chen J (2008) Role of *Spartina alterniflora* on sediment dynamics of coastal salt marshes—case study from central Jiangsu and middle Fujian coasts. Front Earth Sci China 2(3):269–275. https://doi.org/10.1007/s11707-008-0021-1

Watson EB et al (2018) Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt for Southern New England. Estuar Coasts 40(3):662–681. https://doi.org/10.1007/s12237-016-0069-1.Wetland

Wickham H (2021) Package "tidyverse"

Wigand C et al (2017) A climate change adaptation strategy for management of coastal marsh systems. Estuar Coasts 40(3):682–693. https://doi.org/10.1007/s12237-015-0003-y