Modeling the spatial dynamics of marsh ponds in New England salt marshes

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Abstract

Ponds are common features on salt marshes, yet it is unclear how they affect large-scale marsh evolution. We developed a spatially explicit model that combines cellular automata for pond formation, expansion, and drainage, and partial differential equations for elevation dynamics. We use the mesotidal Barnstable marsh (MA, USA) as a case study, for which we measured pond expansion rate by remote sensing analysis over a 41-year time span. We estimated pond formation rate by comparing observed and modeled pond size distribution, and predicted pond deepening by comparing modeled and measured pond depth. The Barnstable marsh is currently in the pond recovery regime, i.e., every pond revegetates and recovers the necessary elevation to support plant growth after re-connecting to the channel network. This pond

dynamic creates an equivalent (i.e., spatially and temporally averaged over the whole marsh) 0.5-2 mm/yr elevation loss that needs to be supplemented by excess vertical accretion. We explore how the pond regime would change with decreased sediment supply and increased relative sea-level rise (RSLR) rate, focusing on the case in which the vegetated marsh keeps pace with RSLR. When the RSLR rate remains below the minimum unvegetated deposition rate, the pond dynamics is nearly unaltered and ponds always occupy ~10% of the marsh area. However, when RSLR rate exceeds this threshold, the ponds in the marsh interior – which receive the least amount of suspended sediment – do not recover after drainage. These ponds transition to mudflats and permanently occupy up to 30% of the marsh area depending on RSLR rate. For marshes with a small tidal range, such as the microtidal Sage Lot Pond marsh on the opposite side of the peninsula from Barnstable marsh, high RSLR rates could bring every portion of the marsh into the pond runaway regime, with the whole marsh eventually converting into mudflats. In this regime, the existing marsh would disappear within centuries to millennia depending on the RSLR rate. Because of the spatial and temporal components of marsh evolution, a single RSLR threshold value applied across the entire marsh landscape provides a limited description of the marsh vulnerability to RSLR.

1. Introduction

Marsh evolution models commonly assume a homogenous marsh platform, in which both elevation and plant biomass change smoothly except close to a few ecotones (Marani et al., 2013). On the contrary, many marsh platforms are highly heterogeneous because of the presence of ponds – semicircular depressions permanently inundated even at low tide. This heterogeneity is particularly relevant for the Mid-Atlantic and New England coast of the USA, where ponds are ubiquitous features (Adamowicz and Roman, 2005; Harshberger, 1909; Koop-Jakobsen and Gutbrod, 2019; Redfield, 1972; Schepers et al., 2017). Thus, considering the presence of ponds is necessary to accurately predict the landscape-level evolution and persistence of these salt marshes under climate change.

Ponds haven been simulated in a simplified 2D model (Kirwan et al., 2008) by lowering the elevation and preventing vegetation growth in a few random locations on the marsh platform. After this temporary disturbance, the vegetation was allowed to regrow and the bed quickly regained its original elevation by accreting faster than RSLR rate. Because of these localized and temporary bursts of vertical accretion, the model provided a simple explanation for the presence of spatial heterogeneities in marsh vertical accretion and for the widely observed mismatch between marsh vertical accretion and RSLR. Nonetheless, the model overlooked several physically-based processes that play a key role in pond dynamics: pond deepening by organic matter oxidation, pond expansion by erosion of the pond edge, and pond drainage by channel interception.

A simple lumped model that considered the evolution of a single pond (Mariotti, 2016) suggested that marshes are either in a regime of pond recovery where ponds form, expand, reconnect with the nearby tidal creek and recover to the surrounding marsh elevation, or in a regime of pond runaway (also called pond collapse regime) in which ponds do not revegetate and marsh area is lost even if the vegetated platform keeps pace with sea-level rise. Vertical accretion by *in situ* plant production does not contribute to pond recovery if, at the time of pond drainage, the elevation of the pond bed is below the limit for vegetation growth, which is roughly equal to mean sea level along the Atlantic and Gulf Coast of the

USA (McKee and Patrick, 1988). Indeed, if the bed elevation lies below the limit for vegetation growth, vertical accretion is only provided by deposition of suspended sediment. Vertical accretion by *in situ* plant production would be present only after the elevation deficit for vegetation growth is overcome. Accordingly, the model found that the marsh enters the pond runaway regime when the RSLR rate is larger than the minimum unvegetated accretion rate (Mariotti, 2016), which is equal to $C \cdot r/(2T \cdot \rho_m)$, where C is the suspended sediment concentration, r is the tidal range, T is the tidal period, and ρ_m is the mud dry bulk density. Marshes are thus more likely to be in the pond runaway regime if the tidal range is small, but a very large sediment supply can allow for pond recovery in microtidal systems, as shown by an example in coastal Louisiana (Mariotti, 2016). A major limitation of the model, however, was that it neglected the interactions between multiple ponds, which likely represent important spatial dynamics over long time scales. As a consequence, the model was not able to predict spatially averaged metrics, such as the amount of pond area on the marsh platform in the pond recovery regime or the rate of pond area increase in the pond runaway regime.

When considering the dynamics of ponds, it is also important to simulate key aspects of sediment transport. For example, it is crucial to simulate how the suspended sediment concentration (SSC) that drives inorganic deposition (i.e., mud) varies in space, such as the SSC decreases with distance from the channels (Christiansen et al., 2000; Fagherazzi et al., 2012), thus making the marsh interior more susceptible to entering the pond runaway regime. In addition, lateral sediment redistribution by soil diffusion (or soil creep), which is present along marsh boundaries with sharp elevation gradients such as the banks tidal creeks (Kirwan and Murray, 2007; Mariotti, 2018; Mariotti et al., 2019, 2016), could transfer sediment from high to low elevation areas and thus affect the rate of vertical elevation change.

Another important aspect to consider is the biogeochemical dynamics of ponds. The emergent vegetation that dominates the marsh platform (e.g., *Spartina alterniflora, S. patens*, *Distichlis spicata*, etc.) cannot survive in ponds because of inundation stress or because of the high salinity that results from the absence of regular flushing (Pethick, 1974). Instead, ponds in New England marshes are colonized by

communities of micro- and macro-algae and submerged grasses such as *Ruppia maritima* (Harshberger, 1909; Spivak et al., 2017). The organic matter composition of pond surface sediments reflects inputs from algal and plant communities that colonize ponds during formation (Spivak et al., 2018). Yet, the contribution of submerged algae and grasses to vertical accretion in ponds is likely negligible, given that ponds tend to be net heterotrophic (Spivak et al., 2017) and thus do not accumulate carbon but rather remove the existing one.

In addition to the absence of vertical accretion, ponds might actually experience a net elevation loss. Indeed, the high rates of respiration, driven primarily by sulfate reduction (van Huissteden and van de Plassche 1998), convert buried organic matter into dissolved inorganic carbon and thereby contribute to soil column collapse and pond deepening (Johnston et al., 2003; Spivak et al., 2018, 2017). The elevation trajectory of ponds is thus diametrically opposite to that of the marsh platform, which gains elevation through time via suspended sediment deposition and *in situ* plant production. As a result, ponds in New England marshes are likely experiencing a gradual deepening with respect to the surrounding marsh platform (Wilson et al., 2009, 2010).

Here, we develop a spatially explicit model for pond dynamics. We apply the model to the Barnstable marsh (MA, USA), which is a typical mesotidal marsh in New England, and constrain the model with field data and remote sensing analysis. We also consider the case with a smaller tidal range, using as reference a nearby microtidal marsh with limited sediment supply (the Sage Lot Pond marsh). Then, we explore the model for different RSLR rates and sediment supply to predict future marsh trajectories. As suggested by a previous idealized model (Mariotti, 2016), we expect to find a threshold above which ponds never revegetate and thus lead to permanent marsh loss. By including pond dynamics within the model, we predict marsh loss when the marsh complex enters the runaway pond regime, which occurs even when deposition on the vegetated marsh platform is equal or greater than the RSLR rate, a commonly used threshold for marsh drowning (Morris et al., 2002).

2. Site description

125 The study area is in the New England region of the United States, which experiences a temperate climate,
126 about 1000 mm of precipitation per year, a growing season from May to September, and overnight freezes
127 during the winter (Mariotti et al., 2019). Ice rafting has been documented to occur, especially in
128 concomitance with winter storm surges (Argow et al., 2011).

129 Barnstable is a mesotidal backbarrier marsh, located on the north side of the Cape Cod peninsula (Fig. 1).
130 The spring tidal range is about 3.6 m (NOAA Sesuit Harbor station 8447241). The marsh is dominated by
131 *S. alterniflora* in the low marsh and *S. patens* in the high marsh (Redfield, 1972). The majority of the
132 Barnstable marsh has been ditched since the early 1900s. For simplicity, in this study we focus on an area
133 that does not have man-made ditches (Fig. 1). This choice allows us to isolate and understand the spatially
134 explicit pond dynamics under natural conditions, without the confounding effects of mechanical
135 anthropogenic disturbances.

136 Sage Lot Pond is a polyhaline, microtidal (0.7 m spring tidal range, NOAA Woods Hole Station 8447930)
137 backbarrier marsh located on the south side of the Cape Cod peninsula (Fig. 1). The marsh is dominated
138 by short form *S. alterniflora* on the low marsh and *Distichilis spicata, Juncus gerardii*, and some *S.*
139 *patens* at higher elevations (Gonneea et al. 2019). This marsh is also ditched.

140 According to the nearby NOAA station 8443970, mean sea level was -9.2 cm in the NAVD88 datum
141 during the epoch 1983-2001. Considering the recent increase in sea level, we assume that MSL is now at
142 0 m NAVD88, which agrees with the datum analysis in the nearby Plum Island Estuary marsh
143 (Hopkinson et al., 2018).

144 **3. Methods**

145 **3.1 Field measurements and remote sensing analysis**

146 For the Barnstable marsh, suspended sediments in tidal creek water were collected in triplicate 1-liter pre-
147 combusted (450 ºC) glass bottles from four different locations (Fig. 1) between June 21st and August 1st
148 2018. All samples were collected during the incoming tide. Samples were filtered through pre-combusted
149 and pre-weighed glass fiber filters (nominal pore size 0.45 μm). The filters were then dried to constant

mass (60 ºC) and concentrations were calculated by normalizing suspended sediment mass to the volume filtered. While this sampling is not a comprehensive characterization of the sediment dynamics, it provides at least a magnitude estimate, which is often the only data available (Cavatorta et al., 2003).

Sediment cores were collected from three transects at approximately 2 m (channel adjacent) and 100 m (marsh interior) from the creek bank (Table 1). All cores were taken in the vegetated platform, i.e., not in a pond. Cores were collected with a modified piston core where a plastic core liner (11 cm diameter) was fitted with a gasketed piston and placed on the sediment surface. The core liner was pushed ~70 cm into the marsh subsurface while the piston was maintained at the surface to minimize compaction, which was observed through the clear core liner. The cores were split vertically, sampled at 1 cm intervals to 45 cm below soil surface and 2 cm intervals thereafter, frozen, and then freeze-dried. Sediment samples were prepared for gamma analysis by sealing approximately 5 g of dried, homogenized peat for 3 weeks, and counting on a planar-type gamma counter for 24 to 48 hours to measure ^{7}Be, ^{137}Cs, ^{210}Pb, and ^{226}Ra at 477, 661.6, 46.5 and 352 KeV energies respectively (Canberra Inc., USA). Detector efficiency was determined from EPA standard pitchblende ore in the same geometry as the samples. Activities of ^{7}Be, ^{137}Cs, and ^{210}Pb were decay corrected to time of collection. Suppression of low energy peaks by self-adsorption was corrected for according to (Cutshall et al., 1983). Excess ^{210}Pb was calculated by subtracting supported ^{226}Ra from total ^{210}Pb with a detection limit of 0.1 dpm g^{-1}. Sediment ages were calculated using the continuous rate of supply ^{210}Pb age model, a variant on the advection-decay equation (Appleby and Oldfield, 1978). This age model assumes that ^{210}Pb down-core activity is a function of decay and variable sediment accretion, however, the full ^{210}Pb profile must be captured to prevent bias towards ages that are too old and accretion rates that are too low at depth (Binford, 1990). Vertical accretion rates were calculated as the age difference between each interval, with temporal resolution typically 2 to 10 years depending on marsh, treatment, and sediment age.

Ponds within a few hundred meters from the sites where cores were collected were surveyed in the summer of 2019 in one of two ways (Fig. 1G). First, 200 marsh-pond boundaries were surveyed in 2019

by measuring the elevation of the marsh adjacent to the pond edge and the elevation of the pond bed adjacent to the pond edge using an RTK-GPS with vertical accuracy of 2 cm. The height of the pond bank was calculated as the difference between these measurements. Second, elevation transects running through the diameter of the pond were surveyed for thirteen of the 200 ponds (five small, five medium, and three large ponds). The absolute bed elevation was referenced to the elevation of the adjacent marsh surface, which was surveyed with the RTK-GPS.

We performed a remote sensing analysis using two datasets: USGS Single Frames Aerial Photos from March 1973 with resolution of ~0.5 m and USGS high resolution orthoimages from April 2014 with resolution of 0.075 m, both of which were collected at low tide (earthexplorer.gov). We also used USGS Lidar collected in 2014 with resolution of 1 m (OCM Partners, 2019). For consistency we resampled all datasets to a 1 m resolution. In both images we defined isolated ponds as those cells with standing water at low tide, which were easily identifiable using a threshold on the intensity of the visible band. Pond cells were clustered together using a five-point stencil connection, such that pond cells that touched each other either on the top, bottom, right, or left boundary were grouped together. Then, we identified the same ponds in each image and calculated the rate of area change. We excluded from the analysis those ponds that either merged together or reconnected to the channel network between 1973 and 2014. From the rate of area change, we calculated the equivalent rate of pond radius expansion. Change was measured as a difference in area rather than the retreat of the pond edge, eliminating any error due to image misalignment.

For the Sage Lot Pond marsh, SSC was measured from 2011 to 2018 across all seasons and tides. A total of 258 grab samples (500 mL each) were collected and stored at 4 ºC for a maximum of 1 week and then filtered through pre-combusted borosilicate glass fiber filters (0.7 um pore size), rinsed thoroughly with deionized water to remove salts, and dried at 105 ºC for >4h. SSC was determined as the ratio of dried filtrate mass total per sample volume. Existing vertical accretion measurements for Sage Lot Pond marsh (Gonneea et al., 2019, 2018) are also reported in Table 1.

3.2 Model for pond dynamics

Within the model, the pond dynamics is implemented as a cellular automata, in which each cell is described by its state (which can only take a finite number of values) without explicitly simulating the elevation. Channels are special cells that are fixed in time, which do not widen, narrow, or migrate laterally. These cells don't take part in the evolution of the cellular automata and are collectively defined as "channel network". Cells that are not part of the channel network are marsh cells, which collectively constitute the marsh domain. Marsh cells are either connected or isolated. Connected cells experience tidal fluctuations, and are defined as vegetated platform if their elevation is above the lower limit for vegetation growth (z_{min}) or mudflat if their elevation is below (z_{min}). Isolated cells, also called isolated ponds or ponds, do not experience tidal fluctuations or exchange with the channel network. Pond cells with an elevation below z_{min} are defined as drowned ponds (Fig. 2).

Two processes allow for connected marsh cells to become isolated cells: pond formation and pond expansion. In the real marsh, formation of new ponds can take place by a variety of mechanisms. A slight depression in the marsh might lead to water logging and rapid decay of the marsh peat. Deposition of rafts composed of either eroded plant material or macroalgae might kill the standing vegetation and temporarily prevent new plant growth (Pethick, 1974). Ice rafting might compress the marsh (Argow and FitzGerald, 2006) or remove large pieces of standing vegetation. Sub-surface piping might collapse the marsh from below (Kesel and Smith, 1978). Large patches of vegetation can be removed by grazers such a geese herbivory (Kirwan et al., 2008) and major storms (Howes et al., 2010). Rather than explicitly modeling these different mechanisms, we simply assume that individual ponds form at a constant rate k_{seed} [#ponds/m^2/yr], that is, during each time step Δt, each vegetated marsh cell is transformed into a pond cell with a probability $k_{seed}\Delta t\Delta x^2$. Pond expansion allows existing ponds (i.e., clusters of connected pond cells) to enlarge. This process is implemented at every boundary between a pond cell and a vegetated cell. Expansion is simulated using a probabilistic approach (Mariotti and Canestrelli, 2017), in which a

vegetated cell adjacent to a pond cell is converted to a pond cell with a probability $p=k_{exp}\Delta t/\Delta x$, where k_{exp} is the expansion rate [m/yr].

Pond cells transform into vegetated cells though pond drainage. In the real marsh, pond drainage occurs when a pond become close enough to the channel network that a new connecting channel forms, as exemplified in Fig. 1E. Then, the pond fills in quickly (assuming that there is enough sediment supply) and the connecting channel, now lacking the tidal prism associated with the pond, eventually silts in. The model is unable to explicitly simulate the formation or closure of these transient connecting channels, since the model does not simulate tidal hydrodynamics and the main channel network is assumed to be fixed in time. We thus parameterize this effect by assuming that pond drainage instantaneously occurs when a connected cell becomes closer than a critical distance L from the channel network. When this happens, the whole pond, which consists of all pond cells that are contiguous to at least one pond cell with distance L from the channel network, are drained and all its cells become connected cells. The distance L is the distance at which the pressure gradient generated by the water in the pond is able to start scouring the marsh soil – possibly by seepage and piping first – and eventually form a connecting channel. In theory this distance might be calculated using a hydrodynamic and sediment transport model, but for simplicity this distance is calibrated to reproduce the observation that ponds tend to drain when they get about 20 m away from the main channel network (Fig. 1E).

3.3 Model for elevation dynamics

The elevation of each marsh cell relative to mean sea level, $z(x,y,t)$, is described as

$$\frac{\mathrm{d}z}{\mathrm{d}t} = D_p + D_m - S - P - R \qquad \text{(Eq. 1)}$$

where D_p is the *in situ* organic accretion by marsh plants, D_m is the accretion from suspended sediment, which we assume is composed of mud and thus transported in suspension, S is the divergence of the creep flux F, P is the elevation loss by pond dynamics (which is further explained below), and R is the relative sea-level rise rate.

The organic accretion by *in situ* plant production is $D_p = D_{pMAX}B$, where D_{pMAX} is the maximum accretion rate [mm/yr] and B is a function that describes the dependence on the bed elevation and serves as a proxy for the hydroperiod, $B = \max[0, 4(z - z_{min})(z_{max} - z)/(z_{max} - z_{min})^2]$, where z_{min} and z_{max} are the minimum and maximum elevation for vegetation growth. B is set equal to zero in isolated ponds, simulating the absence of marsh plants and assuming that any macrophytes and microphytobenthos present in ponds do not contribute to substantial vertical accretion.

Mud accretion depends on hydroperiod and the local SSC. The spatial pattern of SSC is described by

$$C = C_o\left(\alpha + (1 - \alpha)e^{-\beta l}\right) \quad \text{(Eq. 2)}$$

where C_o is the time-average suspended sediment concentration in the channels, $l(x,y)$ is the distance to the closest cells that belong to the channel network, β is the decay factor for SSC with distance from the channel network, and α is the factor describing the SSC that is spatially uniform. This last parameter allows transport of sediment even to large distances from the channels, and thus can parameterize processes such as storm-induced deposition on the marsh platform. Mud accretion thus equals

$$D_m = \frac{C\min(h, r)\min(1, h/\Delta r)}{T\rho_m} \quad \text{(Eq. 3)}$$

where h is the depth with respect to MHHW, T is the tidal period, ρ_m is the dry bulk density of the mud, and Δr is the difference between the spring and neap high tide. The last factor in the numerator is introduced to account for modulation introduced by the spring-neap variability. If the marsh elevation is at least Δr lower than MHHW, the marsh is inundated every tidal cycle and this factor is equal to one. If the marsh is higher than this elevation, the marsh is only inundated during spring tides, and the factor will be less than one. This allows mud to deposit at high marsh elevations but at a lower rate.

In theory, ponds could accrete vertically by suspended sediment deposition akin to the marsh platform. Yet, several reasons suggest that this deposition is small. First, despite ponds having a large depth, the water in the pond is not exchanged during tides. Only the thin layer of water that flows over the pond

should count toward sediment deposition as formulated in Eq. 3 (which assumes that the water depth is equal to the depth of the flow that floods that cell). Second, ponds lack the emergent marsh vegetation that enhances sediment trapping. As a consequence, for the same water depth and sediment concentration, a vegetated pond should experience a lower sediment deposition that a vegetated marsh. Third, ponds are generally far from channels, specifically they are always farther than the distance L. As such, the suspended sediment concentration above ponds is always lower than on the marsh close to the channel edge. Furthermore, field data suggested that ponds in a New England salt marsh are isolated from the tidal network and do not import sediment (Spivak et al., 2017). Overall, sediment deposition in the ponds is likely very small, and for simplicity is set equal to zero.

The creep flux F parameterizes all processes that cause soil diffusion and is set equal to $\mu\nabla z$, where μ is the soil diffusivity coefficient, whose value was empirically found equal to ~0.1 m^2/yr (Mariotti et al., 2019). The creep flux is set equal to zero within ponds and, more importantly, at the pond edges. This choice reflects the absence of tidal oscillations, a process that creates "tidal fatigue" and is responsible for the fast creep in channel banks (Mariotti et al., 2019). This is due to the standing water in the isolated ponds maintaining hydrostatic pressure on the pond bank and preventing slumping during low tide. Thus, the model allows isolated ponds to have relatively steep banks, as observed in the field (Wilson et al., 2010). When pond cells reconnect, they drain instantaneously, and creep can take place. Excluding the creep process at the edge of isolated ponds results in isolated pond expansion only occurring according to the pond expansion term k_{exp}, which can be calibrated by direct comparison with the observed historical expansion rate of isolated ponds. To summarize, creep does not affect isolated pond dynamics, but does affect the dynamics of mudflats (i.e., connected ponds) and prevents the formation of an unrealistically large elevation gradient in the marsh interior, which in reality might have been smoothed by hydrodynamic processes other than creep but that the model is unable to explicitly simulate.

The channel dynamics is not explicitly simulated, and the channel elevation is kept equal to the lower limit for vegetation growth. This elevation acts as a boundary condition for the bed elevation gradients

and thus affects the sediment creep flux near the banks. Noticeably, creep from the marsh platform into the channels moves material out of the marsh domain and thus acts as a sink. Creep can also redistribute sediment within the marsh domain; this does not count as either a sink or source because it conserves sediment.

Finally, the term P includes the platform elevation loss through pond processes. When a connected cell becomes a pond cell by either pond formation or pond expansion, it loses the elevation $Y=\min(Y_{max}, \max(0, z-z_{min}))$, where Y_{max} is the maximum scour thickness, calibrated with field data, and z_{min} is the lower elevation limit below which pond scour does not occur. Y is the thickness of the material that is instantaneously removed from the system, such that the elevation z is lowered by the amount Y.

The term P also includes pond deepening, which simulates organic matter decomposition via sulfate reduction in the bed of the pond. Pond deepening is set equal to a constant rate ($P_{deepening}$) if $z>z_{min}$ and to zero if $z<z_{min}$. Even though we introduced active pond deepening to simulate the loss of elevation caused by organic matter decomposition, this term could be more generally considered as the net effect of all the processes that change pond elevation. For example, if we assume that some sediment settles in the pond during high tide, the term $P_{deepening}$ should be interpreted as the net deepening, including the actual pond deepening by organic matter decomposition minus the elevation gain caused by sediment deposition. If sediment deposition was larger than organic matter decomposition, $P_{deepening}$ could be negative, and the pond might gain elevation through time. Given that this parameter is calibrated by comparing measured and modeled pond depths, not by directly measuring the deepening or accretion, we are unable to isolate these two components.

To summarize, the pond depth, relative to the surrounding marsh platform, is controlled by three mechanisms: 1) the initial scour, which takes place by either pond formation or expansion, 2) the active pond deepening, and 3) the relative deepening caused by the surrounding marsh gaining elevation through organic and inorganic accretion.

3.4 Coupling between pond dynamics and elevation dynamics

As outlined above, the bed elevation is affected by pond processes of formation, expansion, and deepening through the term P (Eq. 1). While the cellular automata model formulated in Section 2.2 is independent of bed elevation, the elevation dynamics is introduced by allowing ponds to form and expand only in vegetated marsh cells. This rule keeps ponds from forming and expanding into a mudflat, which is defined as a connected cell with an elevation lower than z_{min}. This model implementation is a result of only allowing the processes associated with isolated pond formation and expansion to occur in highly organic soil that can be oxidized and compressed.

If the pond elevation at the time of drainage is higher than z_{min}, the pond revegetates just after it is drained. In other words, the cell instantaneously switches from a pond to a vegetated marsh status. In this scenario, every cell that is not a pond is a vegetated marsh that can be transformed back to a pond through either pond formation or pond expansion. In this case, the pond dynamics without elevation is equivalent to the pond dynamics with elevation, so the cellular automata can be run independently of the elevation. On the other hand, if the connected ponds do not revegetate immediately, they become mudflat cells in which ponds are not allowed to form (because $z<z_{min}$). These areas affect the dynamics of pond formation and expansion, and thus the cellular automata model needs to be run together with the elevation model.

A special consideration should be given to cells whose elevation drops below the vegetation limit z_{min}. These cells can be either drowned ponds or mudflats (Fig. 2). Neither has organic accretion by vegetation (Eq. 3), but only the mudflat cells receive inorganic sediment and are allowed to creep under the model specifications. If a drowned pond is surrounded by mudflats, the pond cannot expand. As a result, drowned ponds will never intercept a channel and will never drain in the model domain. In reality, drowned ponds should connect to the channel network through channels that quickly form in the mudflats, but these are not explicitly simulated in the model. Connecting these drowned ponds to creeks will not immediately allow for pond revegetation since the pond is still below the vegetation limit, but it will allow for inorganic accretion and creep, which will both increase the bed elevation and potentially allow for revegetation. In the model we account for this dynamic by introducing the rule that any drowned

pond that touches a mudflat becomes a mudflat itself, and receives inorganic sediment. Then, the evolution of the mudflat would follow the elevation dynamics as previously described.

To summarize, the pond dynamics can be affected by the presence of mudflat cells, since ponds cannot form or expand into these cells. The formation of mudflats is controlled by active pond deepening ($P_{deepening}$) and by RSLR; the recovery of mudflats is controlled by mud accumulation and by RSLR (Fig. 2). If the amount of mudflats at any time is negligible (a few percent of the total marsh area), then the elevation dynamics would in practice not affect the pond dynamics.

4. Results

4.1 Field measurements

The pond expansion rate, calculated using remote sensing, was equal to ~1.5 cm/yr, with a weak dependence on the pond size (Fig. 3). This is calculated for ponds that did not merge between 1973 and 2014 and thus only includes ponds smaller than about 20 m. We were not able to measure expansion for larger ponds. Detailed transects across thirteen ponds indicate that ponds are 0.3-1.2 m deep and that their depth increases with diameter. The survey of 200 marsh-pond boundaries revealed that the depth of the pond immediately adjacent to the bank was 0.46±0.21 m. As such, the depth of the pond close to the edge is relatively uniform, whereas the depth in the middle of the pond has a greater variability. Field measurements at Barnstable marsh show an accretion of 4.6±2.5 mm/yr near the channel and 5.1±3.3 mm/yr in the marsh interior over the past 100 years (Table 1). Total suspended sediment concentrations were 29.3±3.6 mg/l for Barnstable and 5.0±2.8 mg/l for Sage Lot pond.

4.2 Pond dynamics not coupled to elevation

The model simulations were run in a portion of the Barnstable marsh about 1.2 x 1.2 km (Fig. 1C). Using the 2014 Lidar topography, the channel network was defined as the area with an elevation lower than the limit for vegetation growth, which was assumed to be equal to MSL. Areas that were identified as ponds (Section 2.1) were excluded from the channel network.

The pond model is first run without including the requirement that ponds can only form and expand where the marsh elevation is greater than z_{min}. In this case, the pond dynamics does not depend on bed elevation. This simplification is equivalent to the fully coupled model if the pond elevation never drops below z_{min}, and it is nearly equivalent to the fully coupled model if the amount of mudflats is negligible, which is the case if mudflats recover quickly. The advantage of running the model without the elevation dynamics is to emphasize a key feature of pond dynamics, that is, the spatial distribution of ponds does not depend on RSLR and sediment supply. As such, the parameters for pond evolution (k_{seed} and k_{exp}) can be calibrated against observations without being affected by the uncertainties associated with RSLR (and its variability during the last century) and sediment supply (which is estimated using a limited number of samples).

Starting with a marsh without ponds, we run the model for 1000 years to reach a steady state, defined as the period in which the time-averaged statistical distribution of ponds does not change. We found that pond dynamics does not depend on the size of the ponds formed via the seeding mechanism, rather the same steady state pond distribution is obtained if the ponds formed by the seeding mechanism are composed of a single or several cells (and thus the initial ponds is larger). This occurs because the amount of pond area created by seeding, where ponds are introduced into the model, is small compared to the area gained by pond expansion. As such, the main effect of pond formation is not to directly increase the pond area, but rather to create "seeds" that allow for pond expansion. This also confirms that pond formation should have units of #ponds$\cdot m^{-2}yr^{-1}$ and not of $m^2m^{-2}yr^{-1}$.

We found that the steady state pond size distribution only depends on the ratio between pond formation and expansion rates. This is confirmed by noticing that the ratio between the two rates has units of #ponds/m^3 and is independent of time. The actual values of the rates only affect the time needed to reach the steady state distribution. The ratio between pond formation and expansion rate that best fits the pond size distribution (Fig. 4) is 0.027 #ponds/m^3. If the ratio decreases, the number of small ponds decreases while allowing for only a few large ponds (Fig. 4). Intuitively, in this scenario each pond has more time to

395 expand before eventually being drained. Using the measured pond expansion rate (1.5 cm/yr), the pond
396 formation rate is estimated to be $4 \cdot 10^{-4}$ ponds$\cdot m^{-2} yr^{-1}$. We could not directly measure this rate because of
397 the difficulty of detecting small ponds.

398 **4.3 Elevation dynamics**

399 Simulations that include the elevation dynamics (Section 2.3) are run for 1000 years, in which the first
400 900 years have R of 1 mm/yr and the last 100 years have R of 2.9 mm/yr, thus representing the RSLR
401 acceleration in the 19th century (NOAA Woods Hole Station 8447930). Thus, year 900 of the simulation
402 roughly corresponds to year 1914, whereas year 1000 roughly corresponds to year 2014 (i.e., present
403 time). The model at year 0 is initialized with an elevation equal to the 90th percentile of the vegetation
404 range, i.e., $0.9(r/2)=1.62$ m. Despite this initialization of marsh elevation being arbitrary, it does not affect
405 the results after about 500 years. Put differently, any initial marsh elevation would lead to the
406 (statistically) same results during the last few hundred years of the simulation. As such, this approach
407 recreates a synthetic marsh representative for the beginning of the 20th century that is 1) at steady state
408 according to the processes included in the model, 2) independent of any arbitrary initial marsh elevation,
409 3) independent of any topography measured at present time. Specifically, the measured Lidar is not used
410 to initialize the marsh elevation nor the pond distribution. Only the geospatial distribution of channels
411 (which could be reconstructed from aerial images without elevation) is needed to initialize the model.

412 First, we consider the elevation dynamics without the presence of ponds (Fig. 5). We analyze this
413 scenario by considering the spatially averaged vertical fluxes over the marsh domain (Fig. 6). As
414 expected, the net vertical accretion is equal to the RSLR rate. When RSLR rate increases to 2.9 mm/yr,
415 the net vertical accretion lags behind, but it nearly matches the RSLR rate after about 100 years (i.e., at
416 present times). The gross vertical accretion at present times is 5.1 mm/yr. This value represents the
417 spatially averaged accretion rate; in reality the accretion is much higher in the marsh adjacent (<10 m) to
418 channels, where it can be up to 20 mm/yr (Figs. 5, 7C). This additional accretion is balanced by bank
419 creep, which transports sediment out of the marsh domain. When spatially averaged over the entire marsh,

this creep flux creates an equivalent deficit of 2.2 mm/yr, which allows the net vertical accretion to balance the RSLR rate (Fig. 6).

Next, we run the model including the mechanisms of pond formation, expansion, and deepening. The maximum initial pond depth Y_{max} (which is both the maximum initial scour of new ponds, as well as the maximum initial scour when the pond edge expands) is set equal to the measured height of the pond edge, which was measured for Barnstable marsh to be 0.46 m. The consequences of this choice in the model are that 1) small ponds would have a nearly uniform depth across the pond, and that depth would be about 0.46 m, 2) larger ponds would have a depth greater than 0.46 m in their center, which is older and thus may have deepened over time due to organic matter decomposition as well as accretion of the adjacent marsh platform, whereas their edges, which formed recently, would have a depth of ~0.46 m. The active pond deepening $P_{deepening}$ was instead calibrated to match the observed depth in the middle of the ponds (Fig. 8). As expected, the pond depth is much greater in the middle than at the edge, and the depth in the middle of the pond increases with the pond diameter.

Ponds do not affect the marsh dynamics in the buffer zone close to the channel network, where ponds are drained instantaneously. This is equivalent to assuming that they do not form at all and cannot expand. As such, the model that includes pond dynamics has a similar pattern of bank creep and excess accretion on the marsh adjacent to the channel as the model without pond dynamics (Fig. 6). On the other hand, ponds drastically change the dynamics of the marsh interior. Ponding creates a heterogeneous landscape, with low elevation areas surrounded by high marsh areas. This elevation pattern is mirrored by a spatially heterogeneous vertical accretion, where the net vertical accretion in ponds is zero (if not negative, because of active pond deepening), whereas recently connected ponds experience enhanced vertical accretion. Compared to the case without ponds, the average marsh elevation is also lower.

Because pond reconnection is a stochastic process, the vertical accretion rate is not constant in time. In particular, there are periods when large ponds are drained and the vertical accretion suddenly increases (Fig. 6). Over a sufficient time period (e.g. 50 years), however, the net vertical accretion rate is equal to R.

4.4 Exploring the effects of RSLR rate, sediment supply, and tidal range

To generalize our results, we perform a series of explorative simulations with different sediment supplies (C_o), RSLR rates (R), and tidal ranges (r). In order to consistently initialize the simulations, we start with a uniform elevation equal to $0.9(r/2)$ with no ponds, and then run the model with R equal to 1 mm/yr for 500 years, which is enough to establish a steady state pond distribution (Fig. 6). As already pointed out, the choice of the initial elevation does not affect the results after about 500 years. After 500 years, we changed a single parameter (either R or C_o) while keeping all the other parameters fixed, and run the simulations for an additional 1000 years (Fig. 9). We focused on scenarios in which the vegetated marsh kept pace with RSLR and thus all marsh loss is due to pond expansion.

First, we consider the effects on increasing R or decreasing C_o in a mesotidal marsh such as Barnstable marsh. For the relatively high sediment supply of Barnstable marsh (C_o=30 mg/l), the RSLR rate does not affect the dynamics of ponds, which always occupies ~10% of the marsh. For the case with a lower C_o=10 mg/l, lower R gives nearly identical results as for the case with a higher C_o, with pond area remaining at ~10% of the total marsh area, whereas high R starts to affect pond area (Fig. 9A). For example, for C_o=10 mg/l and R=8 mm/yr, the total unvegetated area (ponds plus mudflats) increases to about 25% within 500 years (Fig. 9A). Noticeably, under these conditions, the unvegetated area stabilizes and does not increase indefinitely even after 1000 years.

Next, we consider the case of a marsh with a smaller tidal range. We use as a reference Sage Lot Pond marsh, which has a spring tidal range of 0.7 m (Gonneea et al, 2019). Field measurements indicate that this marsh has a small sediment supply equal to 5 mg/l. For simplicity, we consider the same channel geometry as of Barnstable marsh but with a smaller tidal range. Accordingly, we modify the vegetation limit, the initial marsh elevation, the maximum initial pond elevation, and the spring-neap variability (Table 2). For the microtidal case, we found that for low R the unvegetated area is still ~10%, but for higher rates the unvegetated area increases quickly, and can be up to 100%, indicating loss of the entire

469 the marsh platform (Fig. 9B). The model predicts that the marsh would enter the pond runaway regime for
470 R~3 mm/yr (Fig. 9B).

471 We also compared the model predictions with the measured accretion rates in Barnstable and Sage Lot
472 Pond marsh. We separated the spatially averaged accretion predicted by the model between the marsh
473 adjacent to channels (less than 10 m from the nearest channel, which roughly coincides with the low
474 marsh, and is about 29% of the total marsh area) and the marsh interior (more than 10 m from the nearest
475 channel, where bank creep is virtually zero, and is about 71% of the total marsh area). For a marsh with
476 low tidal amplitude and sediment supply such as Sage Lot Pond, the model predicts that when RSLR rate
477 was 1 mm/yr , the gross vertical accretion rates was ~2.6 mm/yr in the marsh adjacent to channels and
478 ~1.4 mm/yr for the marsh interior (Fig. 7B). After the acceleration in RSLR to 2.9 mm/yr the model
479 predicts that the gross vertical accretion increases to ~4.1 mm/yr in the marsh adjacent to the channels and
480 to ~3 mm/yr in the marsh interior (Fig. 7B). For comparison, the field measurements of accretion over a
481 100 year period, which includes the period during modern RSLR acceleration, indicate an accretion rate
482 of 3.7±2.0 mm/yr close to the channels and 1.4±0.3 mm/yr in the interior (Table 1). For the Barnstable
483 marsh, with a higher tidal range and sediment supply, the model predicts gross vertical accretion rates of
484 ~9.5 mm/yr near channels and ~2.8 mm/year in the marsh interior when RSLR rate is 1 mm/yr, and ~11.5
485 mm/yr near channels and ~4.0 mm/year in the marsh interior when RSLR rate is 2.9 mm/yr (Fig. 7A). For
486 comparison, field measurements at Barnstable marsh show an accretion of 4.6±2.5 mm/yr near the
487 channel and 5.1±3.3 mm/yr in the marsh interior over the past 100 years.

488 As an indicator of the overall marsh status, we calculated the spatially-averaged primary productivity
489 normalized by the maximum productivity (Fig. 9). The productivity of the vegetated marsh depends on
490 the bed elevation following the function B (section 3.3), whereas the productivity in the ponds is equal to
491 zero. As the RSLR rate increases, the marsh elevation decreases and thus the productivity increases. As
492 the unvegetated pond area starts to increases, however, the overall marsh productivity starts to decrease.
493 This is particularly evident for the Sage Lot Pond marsh for a RSLR rate greater than 4 mm/yr (Fig. 9B).

Finally, we expanded these simulations by systematically considering different tidal ranges (0.7, 1.6, 3.6 m), RSLR rates (1 to 8 mm/yr), and sediment supply concentrations (5 to 30 mg/l) (Fig. 10). We found that, for small tidal ranges, the rate of marsh loss by pond expansion is highly sensitive to the RSLR rate. When the RSLR rate increases to 3 mm/yr, marsh loss occurs but is relatively slow, and the amount of unvegetated area (which includes both ponds and mudflats) increases from 10% to 15% of the total marsh area during the first 100 years (Fig. 9). A further increase in the RSLR rate (5 mm/yr) drastically increases the rate of marsh loss, with the amount of unvegetated area increasing from 10% to 40% of the total marsh area during the first 100 years.

5. Discussion

5.1 The pervasive effect of ponds in salt marshes

Ponds in Barnstable marsh expand at a very slow rate (1.5 cm/yr) compared to exposed marsh creekbank edges that retreat by wave-induced erosion (0.5-10 m/yr) (Elsey-Quirk et al., 2019; Hopkinson et al., 2018; Leonardi et al., 2016; Marani et al., 2011). Yet, ponds are numerous and collectively have a long perimeter, and thus are able to affect nearly the entire marsh surface on millennial time scales. Considering an average unchanneled length of 50 m (Marani et al., 2003), a pond formed in the middle of the marsh would take ~2000 years to reach an area influenced by channel network and drain. Since some ponds form closer to the channel network initially, 2000 years is likely an upper estimate of the time needed to reconnect ponds to the channel network and drain. In addition, ponds that merge together will drastically reduce the time needed for a pond to reconnect to the channel network.

Given that the Barnstable marsh is ~4000 years old (Redfield, 1965), we expect that any point on the marsh was at least once a pond. Indeed, stratigraphic evidence of ponds were found in nearly all cores taken in a New England salt marsh (ME, USA) (Wilson et al., 2009). However, many historical records within salt marshes only extend ~100 years, due to age model constraints of the commonly used lead-210 methods, and thus may miss some of these millennial scale dynamics. In any case, the assumption of steady state marsh accretion often used while interpreting the sedimentary record (Morris et al., 2016)

519 should be evaluated, since even if RLSR rate and sediment supply are constant, a specific site within the
520 marsh may not be at steady state at any given time due to pond dynamics.

521 Another consequence of pond dynamics is that recently connected ponds (as identified in Fig. 1E) are a
522 common feature and should be interpreted as part of the basic marsh dynamics as opposed to an indicator
523 of changes in boundary conditions. Thus, ponds do not necessarily indicate an acceleration in RSLR or
524 other anthropogenic modifications, but are rather a natural occurrence within many salt marshes. Ponds
525 also create diverse habitats and ecological niches for wildlife (Brush et al., 1986), and thus should not
526 necessarily be considered a negative feature within salt marshes.

527 **5.2 Vertical accretion in the pond recovery regime**

528 When the marsh is in the pond recovery regime, pond dynamics in the marsh interior increased the
529 spatially-averaged gross vertical accretion by 0.5-2 mm/yr (Fig. 7), a result that was previously predicted
530 (Kirwan et al., 2008). This excess accretion is larger for larger tidal ranges, because marshes with a larger
531 tidal range have a thicker soil profile that can be removed by the ponding processes. Ponding also results
532 in highly heterogeneous accretion across the marsh platform (Fig. 5), and thus the excess accretion rate
533 during pond recovery could locally be much higher than 2 mm/yr. For example, the model predicts that
534 recovering ponds might have accretion rates up to 20 mm/yr (Fig. 5). Indeed, Wilson et al., (2014)
535 reported vertical accretion rates of 8 mm/yr in recently recovered ponds in a New England salt marsh,
536 despite the fact that the sediment supply in that marsh was extremely low (~5 mg/l). The model predicts
537 that the excess accretion rate persists from when a pond revegetates until it reaches the equilibrium
538 elevation with the marsh surface. In other words, ponds that were drained centuries ago might not have
539 fully equilibrated and thus might still accrete slightly faster than the RSLR rate. These predictions are
540 consistent with the 5.1 mm/yr vertical accretion measured in the marsh interior of Barnstable marsh
541 during a period when relative sea-level rise rate was 2.9 mm/y (NOAA Woods Hole Station 8447930)
542 (Table 1).

Our model also predicts channel bank creep causes an equivalent elevation drop of ~2 mm/yr when averaged over the whole marsh platform (Fig. 6,7). This loss is balanced by an excess vertical accretion (Mariotti et al., 2016), which can be observed by the gross vertical accretion rate being much larger than the RSLR rate. Contrary to ponding, this bank creep is localized to the low marsh immediately adjacent to channels, where the excess vertical accretion can be an order of magnitude higher than RSLR rate (i.e., up to 20 mm/yr, Fig. 5). This prediction is consistent with the larger vertical accretion rates measured on the marsh adjacent to channels than on the marsh interior at Sage Lot Pond marsh (Table 1). For Barnstable marsh, the model predicts that the vertical accretion on the marsh adjacent to channels is ~11 mm/yr (Fig. 7A), which is much larger than the measured 4.6±2.5 mm/yr (Table 1). One possible explanation is that the channel-adjacent area in the model integrates portions of the marsh that are extremely low (e.g., the vegetated bank located a few meters from the channel). These low areas have a disproportionally high excess accretion (Fig. 5 and Fig. 7C,D) and strongly influence the average value over the area here defined as channel-adjacent (<10 m from the channel network). Field measurements in the channel-adjacent area, in contrast, are generally taken a few meters inland from the edge of the marsh, with a higher elevation than the slumping blocks and where the excess accretion is much lower. Indeed, the cores in Barnstable collected ~2 m from the marsh edge are actually ~5 m from the lowest vegetated point (Fig. 1H). Noticeably, the model predicts a minimum in vertical accretion, equal to about 3 mm/yr, located between 5 and 20 m from the channel (Fig. 7C). This is the region where neither bank creep nor ponding are present, and thus the gross accretion rate matches RSLR rate.

The model also recreates the lagged vertical accretion that follows the acceleration in RSLR rate, a disequilibrium effect has been previously identified with simplified marsh models (Kirwan and Temmerman, 2009; Kirwan and Murray, 2008). This lag should be present in every salt marsh that experienced accelerated RSLR, regardless of pond occurrence or the influence of bank creep.

To summarize, the model is able to combine three previously identified mechanisms by which vertical accretion rates deviate from the RSLR rate, emphasizing that a direct comparison between vertical

accretion and RSLR rate can be misleading at best, and overly optimistic for projecting marsh elevation trajectories.

5.3 Vertical accretion in the pond runaway regime

The simple lumped model (Mariotti, 2016) identified the minimum unvegetated accretion, $D_{cr}=C\cdot r/(2T\cdot\rho_m)$, as the threshold between pond recovery and pond runaway regimes. Because C is not spatially uniform, D_{cr} varies among the domain. D_{cr} is lowest in the marsh interior, where $D_{cr}=\alpha\cdot C_o\cdot r/(2T\cdot\rho_m)$.

For $R<D_{cr}$, the lumped pond model predicts that every pond recovers. Indeed, the spatially explicit model predicts that, for any RSLR rate lower than D_{cr}, ponds always occupy ~10% of the marsh area (Figs. 9,10). This prediction agrees with the finding that stable salt marshes are associated with an unvegetated-vegetated ratio of ~0.1 (Wasson et al., 2019) (a pond area of 10% corresponds to an unvegetated-vegetated ratio of 0.11). As for the lumped model, the spatially explicit model predicts an increase in the total unvegetated area as $R>D_{cr}$. The spatially explicit model provides more information than the lumped model, and specifically predicts that the rate at which marsh loss by pond expansion takes place strongly increases as R further exceeds D_{cr}. Put differently, when R is just slightly above D_{cr}, the rate of marsh loss by pond expansion is extremely low, and the marsh could last thousands of years. On the other hand, once the threshold for pond runaway is exceeded, even small increases in R could have catastrophic consequences for marsh loss.

Using D_{cr} as the threshold for the pond runaway regime is further complicated by the presence of creep, which can take place at the edge of mudflats (i.e., connected ponds). Creep transfers sediment from high to low elevations, and thus tends to increase vertical accretion in the mudflats and to lower vertical accretion on the vegetated platform. For example, a very small mudflat might recover even if R is slightly larger than D_{cr} (which explains why the Sage Lot Pond marsh does not enter the pond runaway regime for R=1 mm/yr even if D_{cr}=0.6 mm/yr). Lateral transport of sediment not associated with creep, including sediment transport by waves or sheet flow over the marsh, might slightly alter the predictions purely

based on vertical accretion rates. Nonetheless, the parameter D_{cr} captures the transition to the pond runaway regime within an uncertainty of about 1 mm/yr (Fig. 10).

5.4 Pond formation

A crucial parameter in the model is the rate of pond formation. If ponds do not form, neither the pond recovery nor the pond runaway regime occurs. In this case, the marsh is either fully vegetated or disappears by drowning, and thus the marsh landscape is dramatically different than what simulated in our model (Fig. 5). As such, it is noteworthy to emphasize that ponds are not always present in salt marshes. Ponds have been identified in the Mid-Atlantic and New England Coast of the USA (Adamowicz and Roman, 2005; Mariotti, 2016; Schepers et al., 2017) as well as in the northern Gulf of Mexico (Mariotti, 2016; Nyman et al., 1994). Yet, marshes in the South-Atlantic Coast of the USA, including sites with relatively little human modifications such as in Virginia and Georgia, seem to have few ponds if any at all. For marshes in which ponds are not common, it is possible that either ponds do not form at all or that the pond formation rate (the parameter k_{seed}) is extremely low. In this latter case the model would predict that the number of ponds would be highly reduced (Fig. 4) and that their size distribution would be highly skewed towards a few large ponds.

In this study, we assumed that ponds form without explicitly simulating the mechanisms that lead to pond formation in the first place. In particular, we did not investigate whether the rate of pond formation changes as a function of environmental drivers. We can only speculate that pond formation might be related to disturbances related to climate via ice rafting and scour, excessive wrack accumulation, grazing/bioturbation such as by *Sesarma reticulatum* crab or burrowing species such as fiddler crabs (*Uca* sp.), and to biogeochemical and hydrological processes associated with microtopography.

We further emphasize that pond formation *per se* is not directly driving permanent marsh loss, but it could lead to marsh loss in the pond runaway regime. In the pond runaway regime, an increase in the pond formation rate would directly increase the rate at which the marsh is lost, mainly by increasing the "seeds" from which ponds can expand. Pond formation is thus an example of a relatively secondary

mechanism that, through spatial interactions (Larsen, 2019), could lead to rapid marsh loss. Thus, we argue that future research should develop a mechanistic understanding of pond formation and determine whether environmental changes such climate or burrowing and grazing pressure, might increase the rate of pond formation.

5.5 Marsh management

All simulations were performed for scenarios in which the vegetated marsh keeps pace with RSLR. As such, if pond dynamics were not included, the marsh would have been preserved indefinitely and there would have not been any unvegetated area occupied by either ponds or mudflats. Thus, including pond dynamics in marshes where ponds are known to form is an essential step to simulate marsh evolution.

Including pond dynamics in marsh models is also essential for predicting ecosystem productivity. Marsh grasses remove CO_2 from the atmosphere and bury it in soils, support coastal food webs, and filter inorganic nutrients washed from the landscape, among other ecosystem services. As such, changes in productivity have consequences for ecological and biogeochemical processes within marshes and adjacent ecosystems. In both the mesotidal (Barnstable) and microtidal (Sage Lot Pond) simulations, normalized primary productivity increases monotonically with the RSLR rate (Fig. 9). The expansion of the unvegetated area for very high RSLR rates, however, starts to decrease productivity. For the microtidal case, a decrease in productivity associated with pond expansion could be observed even at the decadal time scale (Fig. 9B), and thus could be relevant for coastal management.

As previously identified (Mariotti, 2016), neither the presence of ponds nor the expansion of ponds is a sign of permanent marsh loss. On the other hand, the absence of pond recovery is a potential indicator of a regime shift. Monitoring pond recovery, such as by measuring their accretion rates once reconnected to the channel network, should thus become a routine marsh assessment, alongside measurements of vertical accretion in the vegetated marsh.

One management question is whether to promote or prevent pond recovery. One example includes selectively digging ditches to reconnect ponds, a strategy referred to as "quality ditching" or Open Water Marsh Management (Wolfe, 1996). Our model suggests that the appropriate management (in terms of morphological evolution of the salt marsh) depends on whether the marsh is in the pond recovery or pond runaway regime. In the former case, favoring pond drainage should accelerate vertical accretion, in the latter it would accelerate marsh loss.

As previously shown in a model that only considers one pond (Mariotti, 2016), determining whether a marsh is in the pond runaway or pond recovery regimes requires an estimate of the minimum inorganic deposition in the marsh interior (D_{cr}), and specifically to estimate the suspended sediment concentration. Because this parameter is difficult to determine without long-term monitoring (Ganju et al., 2017), we suggest that analyzing the trajectory of recently connected ponds is important to determine whether ponds are recovering or expanding indefinitely.

5.6 Model limitations and future directions

The model assumes that any connected area above z_{min} revegetates instantaneously. Therefore, a mudflat is only present if $z<z_{min}$. In reality it could take several years for a connected pond to revegetate even if $z>z_{min}$. For example, the recently connected ponds in Barnstable have not fully revegetated even if their elevation is ~1 m above MSL (i.e., $z>z_{min}$). Even though this delay could be included in the dynamics, we argue that it is not crucial, because revegetation likely will occur within shorter times scales (decades) than the time scale considered in this study. More importantly, large vertical accretion rates by mud deposition could take place even though recently connected ponds are not fully revegetated.

Channels are assumed to be fixed. When analyzing historical imagery (~50 years), channels in Barnstable marsh are relatively stable, even though they could have migrated over longer time scales. Migrating channels would likely increase the rate at which ponds are intercepted and thus drained. Channel migration could be parameterized as a larger value for the reference drainage distance L, allowing ponds within the model to drain even if located at a large distance from channels. Indeed, this could be a

dominant mechanism for pond drainage at sites where channel migration is relatively fast (Finotello et al., 2018). Channel migration would likely result in similar marsh evolution dynamics and pond recovery, with marsh loss at the eroding bank but, at the same time, induce accretion that is faster than the RSRL rate at the accreting bank.

The model does not conserve sediment in the channel domain: the sediment transported from the bank to the channel through the creep mechanism does not accumulate in the channel, and the sediment transported from the channel to the marsh, which is responsible for the term D_m, does not cause a sediment deficit in the channel network. That is, we assume that there is an infinite supply of new sediment from the channels. In reality, a portion of the bank material that creeps into the channel would be resuspended and re-transported to the marsh platform, thus allowing for sediment recycling. This component cannot be reproduced in the model, since an explicit representation of the channel dynamics would be needed.

Finally, the model does not include the dynamics of hydrological alterations such as ditching, which is a common feature in New England salt marshes. Ditching could impact the marsh elevation dynamics in at least two ways. On one hand, ditches could cause a nearly instantaneous marsh-wide pond drainage, which could temporarily increase the spatially averaged vertical accretion rate. On the other hand, ditches could lower the water table and cause a combination of carbon oxidation and compaction, which in turn could lower the marsh elevation. Such effects could be included in the model by 1) allowing ponds close to the ditches to be drained similarly to the ponds drained by the natural channel network, and 2) adding an additional term in Eq. 1 to simulate a localized bed lowering close to the ditches.

6. Conclusions

The proposed model simulates the spatially explicit dynamics of marsh ponds, and thus allows simulation of the aggregated effect that ponds have on the evolution of the marsh platform. The Barnstable marsh is predicted to be in a pond recovery regime: pond expansion does not cause net marsh loss because they recover the marsh elevation once drained. Even in the pond recovery regime, however, ponds drastically

affect the marsh elevation and vertical accretion. In particular, pond dynamics explain why marsh vertical accretion, even away from channels, could be several times larger than the rate of RSLR, and also why the accretion rate is highly variable in space.

The model predicts that future increases in RSLR rate at Barnstable marsh would not cause the marsh to transition to the full pond runaway regime. Yet, with a large RSLR rate and a reduced sediment supply, ponds in the marsh interior might become permanent mudflats and thus increase the unvegetated area to ~20% of the total marsh. Monitoring of pond recovery, such as by measuring their accretion rates once reconnected to the channel network, could be used as a landscape-level indicator of regime shifts.

Marshes with a smaller tidal range and sediment supply, such as Sage Lot Pond marsh, are more prone to enter the pond runaway regime. For RSLR rates just above the critical threshold (e.g., 3 mm/yr), the rate of marsh loss by pond expansion is still relatively low, and the marsh could take several centuries (if not millennia) before completely disappearing. A further acceleration in RSLR rate (e.g., 5 mm/yr) would drastically increase the rate at which the marsh is lost by pond expansion – which will take place even though the vegetated marsh keeps pace with RSLR.

Future research needs include understanding the mechanism of pond formation, evaluating how hydrological alterations such as ditching affects pond dynamics, and quantifying how pond dynamics affect blue carbon accumulation and preservation.

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D
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F
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856

857 Figure 1. A) Map of the study area including Barnstable marsh and Sage Lot Pond marsh (Massachusetts,
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859 the model. The white dots indicate the locations where SSC was measured in triplicates. The red dots
860 indicated the location were cores for vertical accretion were collected. D,E,F) Example of pond
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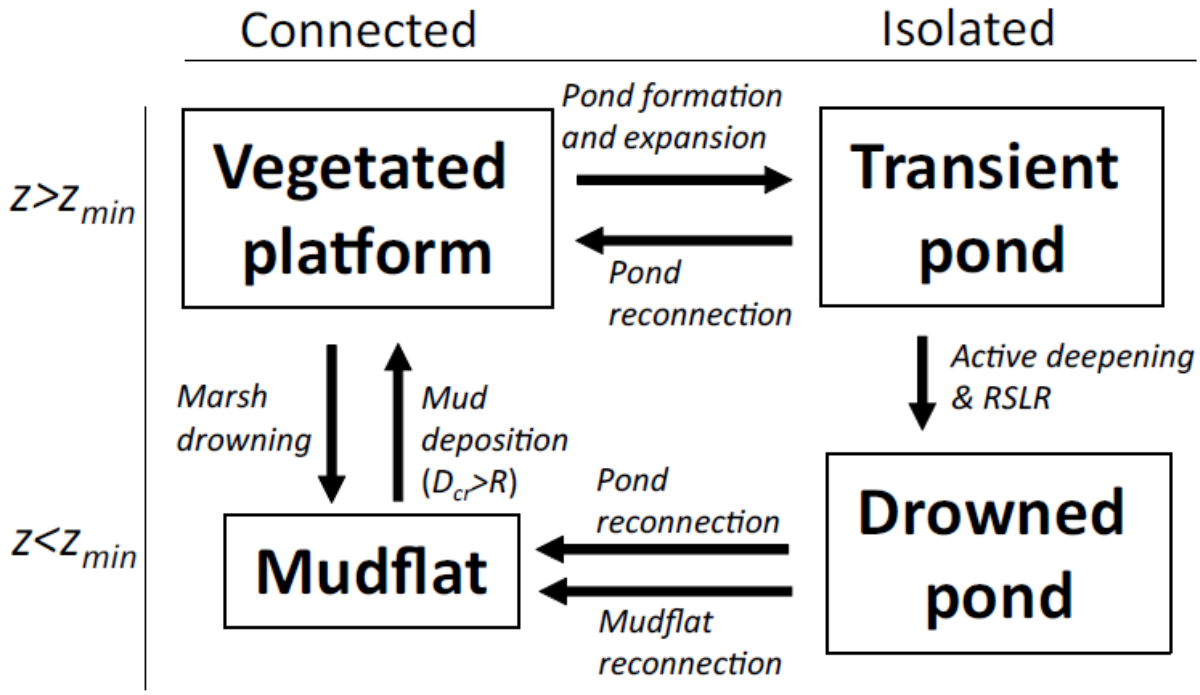
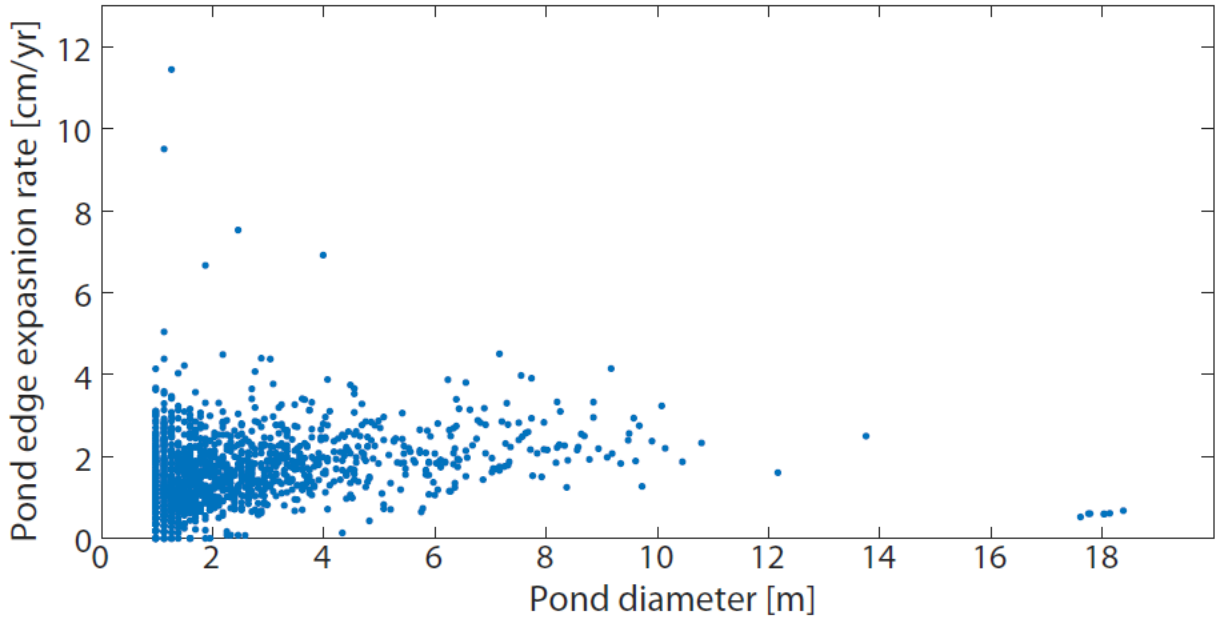
863 white dots indicate ponds only surveyed at one edge. H) Detail of a marsh core collected adjacent to the
864 channel. Images from Google Earth (NASA, USDA) accessed on October 2019.



865

866 Figure 2. Summary schematic of the marsh and pond dynamics and scheme of the various sediment fluxes
867 in the model. Net vertical accretion is equal to the gross vertical deposition minus bank creep and
868 ponding.



869

870 Figure 3. Measured pond edge expansion rate as the increase of the pond radius through time in

871 Barnstable marsh from 1973 to 2014 as a function of the pond diameter.

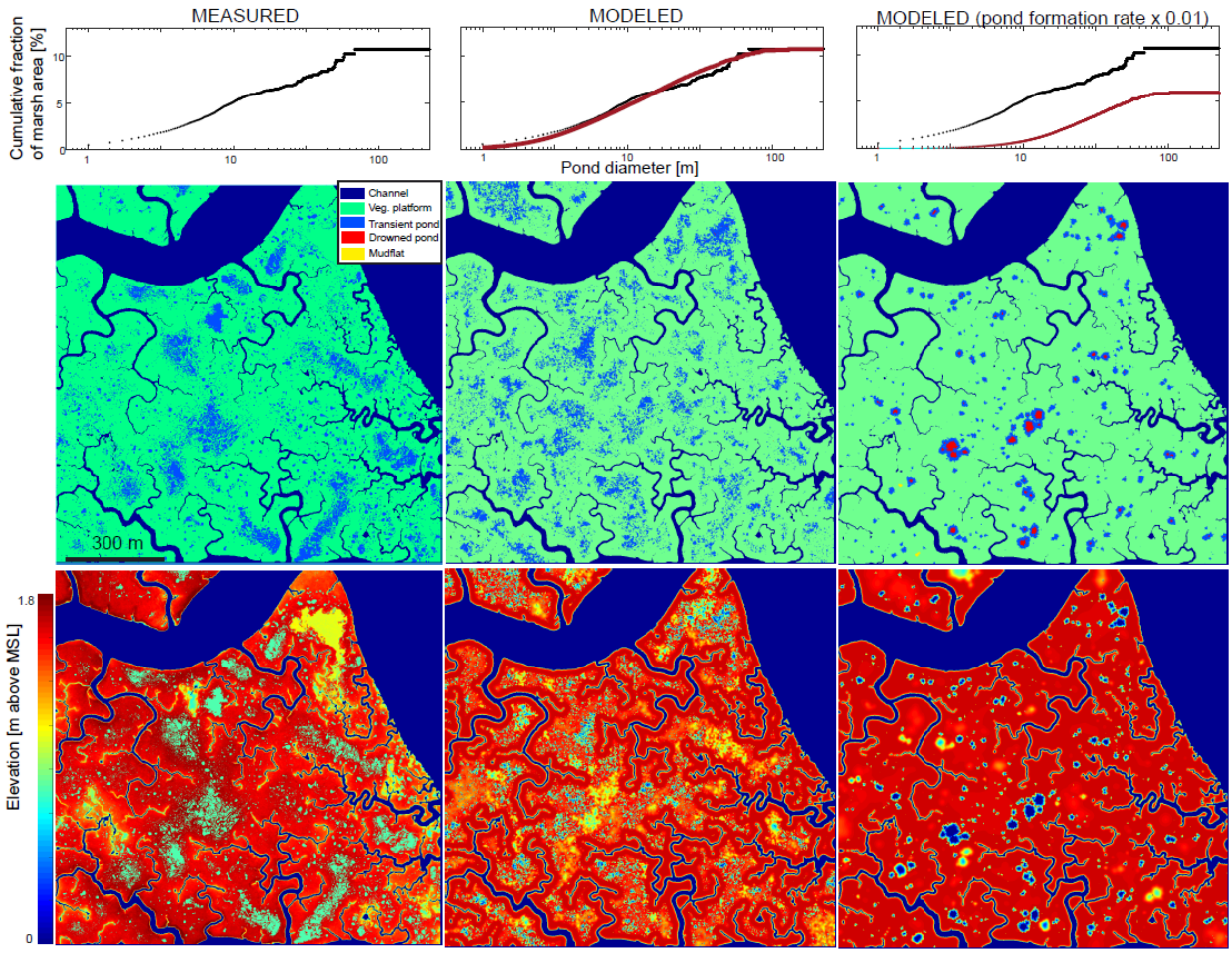
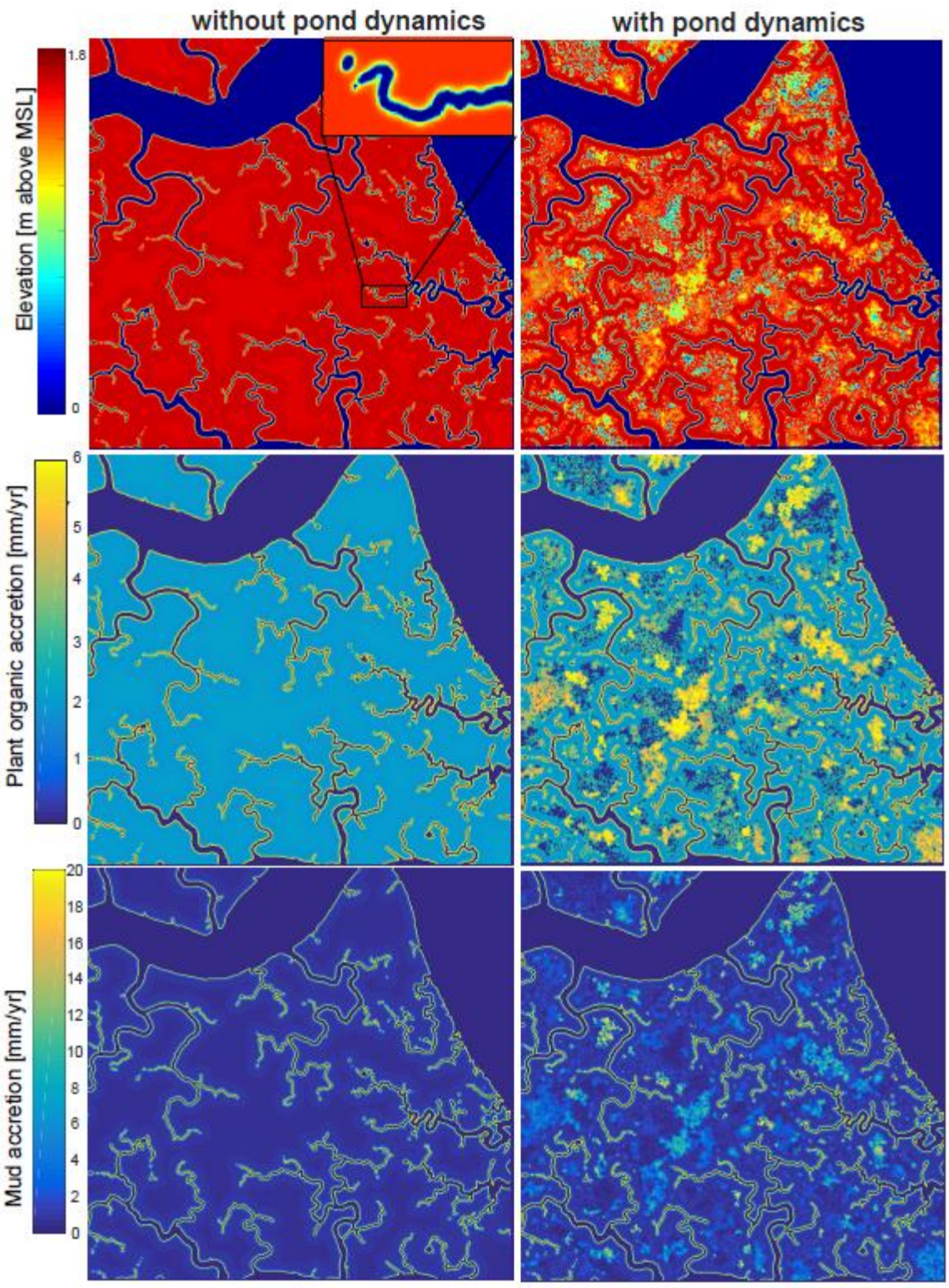


Figure 4. A,B,C) Comparison of measured and simulated (at steady state after 1000 years) pond size distribution. A) Measured pond size distribution, showing that the total fraction of pond area is about 10% of the marsh surface. B) Best fit for the parameters k_{seed} and $k_{exp.}$. C) Model results for the case in which k_{seed} is decreased by a factor 100. D,G) Measured pond spatial distribution and elevation E,F,H,I) Model snapshot after 1000 years showing the pond spatial distribution and the elevation (as the last datapoint in Fig. 6).


without pond dynamics
with pond dynamics
Elevation [m above MSL]
1.8
0
Plant organic accretion [mm/yr]
6
5
4
3
2
1
0
Mud accretion [mm/yr]
20
18
16
14
12
10
8
6
4
2
0

880

Figure 5. Vertical accretion rate by *in situ* plant production and by mud accumulation for the case with and without pond dynamics calculated at year 1000 (as the last datapoint in Fig. 6). The inset in the top left panels shows the detail of the low-marsh adjacent to a channel.

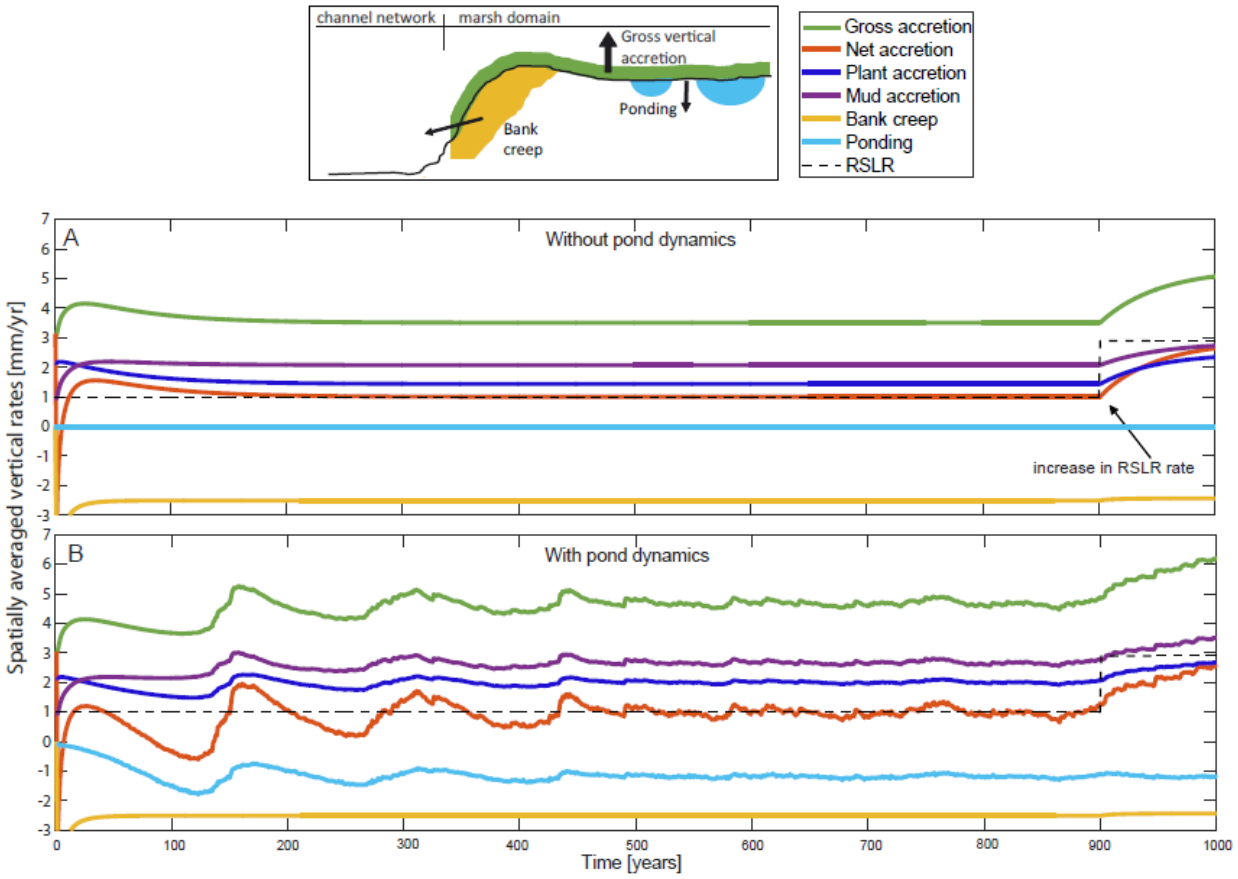


Figure 6. Reconstructed time series of spatially-averaged elevation change for Barnstable marsh, comparing the case without ponds dynamics (A) and with pond dynamics (B). Gross accretion is the sum of plant and mud accretion. Net accretion is the gross accretion minus ponding and bank creep. Note that during the first ~500 years the marsh is equilibrating from the initial conditions without ponds.

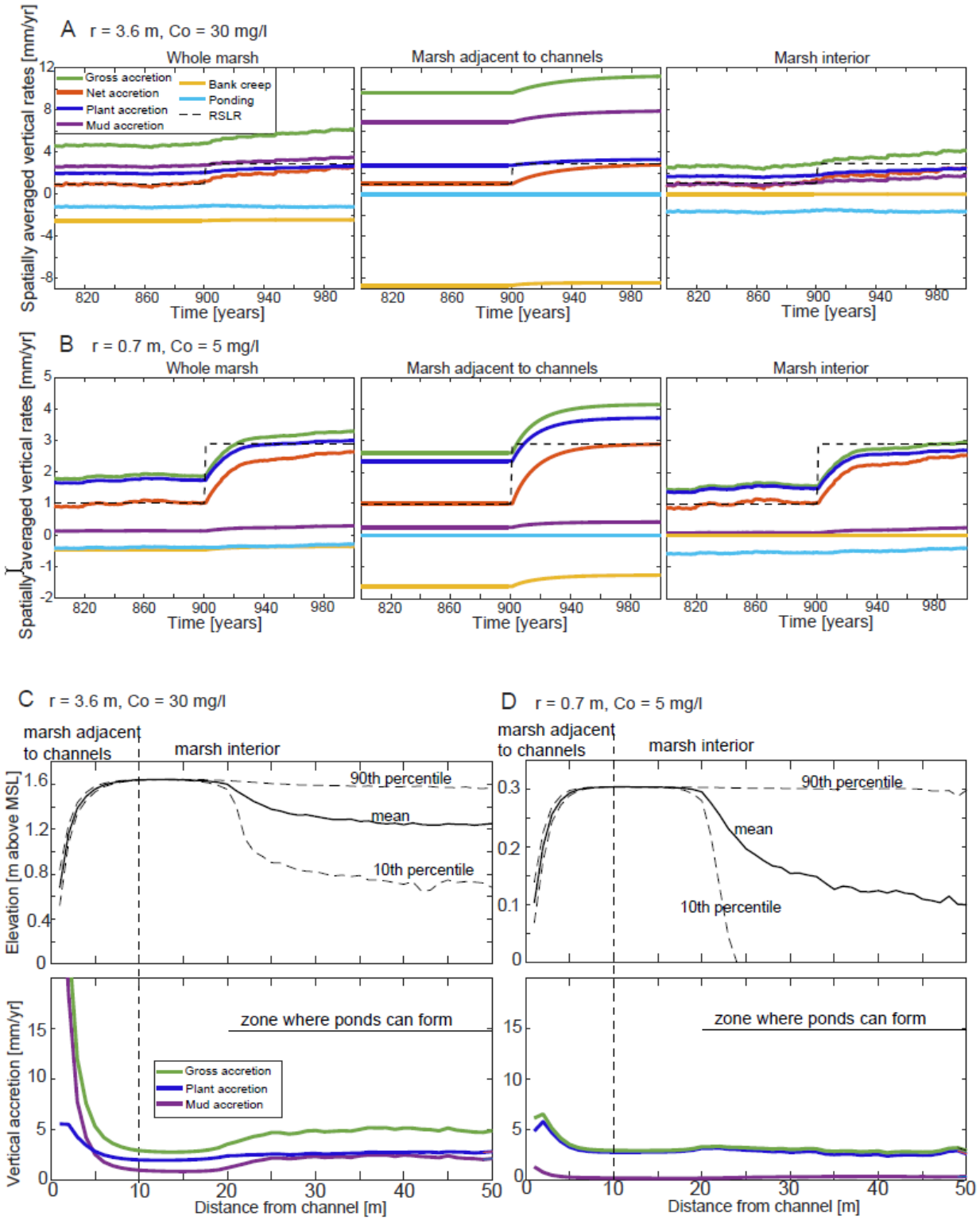
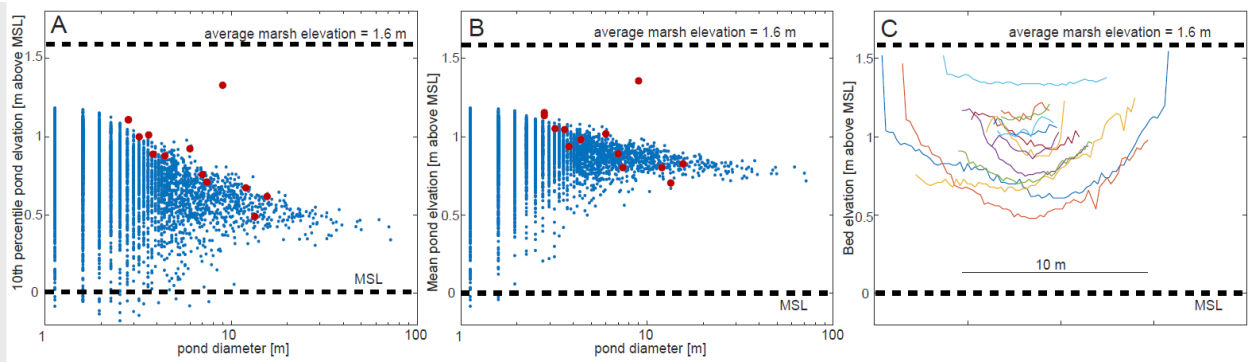


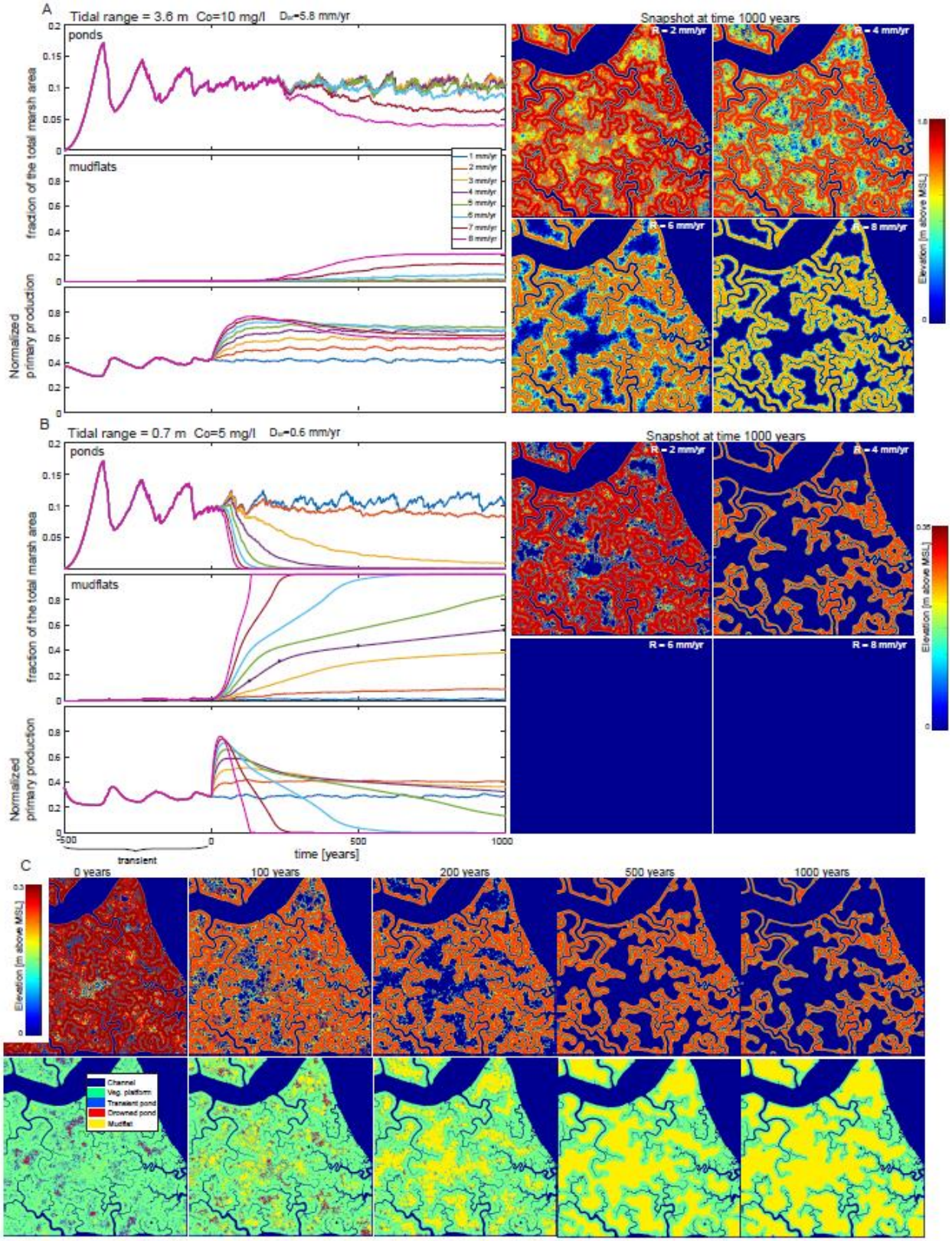
Figure 7. A,B) Spatially-averaged elevation change during the transition from *R*=1 mm/yr to R=2.9 mm/yr (as in Fig. 6B), considering the whole marsh area, the area adjacent to channels, and the marsh

894 interior. Note that in the marsh interior the bank creep is zero. C,D) Snapshot at year 1000 showing the
895 how the vertical accretion varies rapidly with the distance from the channel network. A,C) Conditions
896 representing Barnstable marsh (r=3.6 m, C_o=30 mg/l), B,D) Conditions representing Sage Lot Pond
897 marsh (r=0.7 m, C_o=5 mg/l).



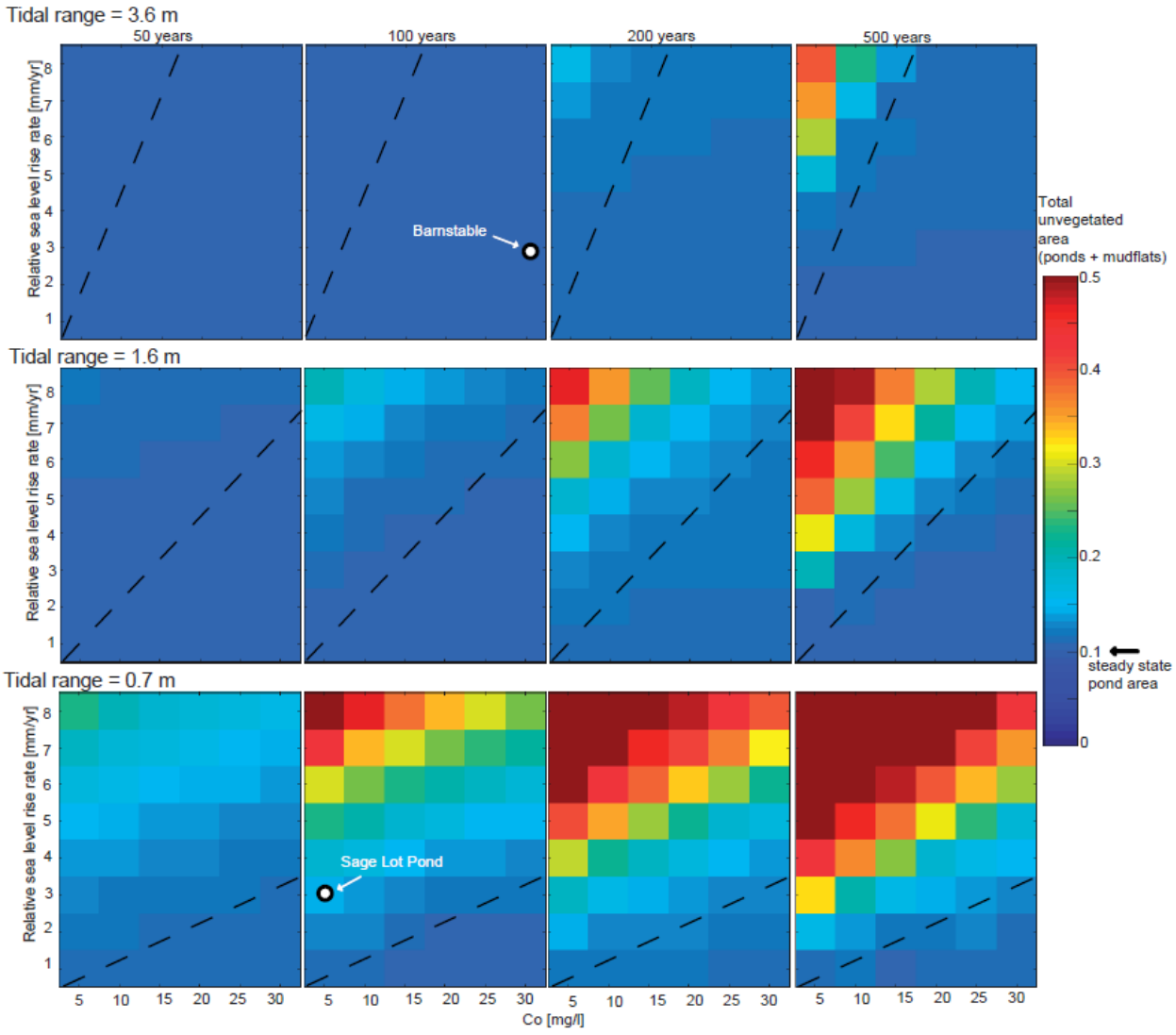
898

899 Figure 8. A,B) Modeled 10th percentile (A) and mean (B) pond elevation as a function of pond size. The
900 blue dots are the individual points from the model, the red dots are the survey of the 13 ponds in
901 Barnstable marsh. C) Cross sections of the 13 ponds surveyed in Barnstable marsh (see Fig 1G).


A
Tidal range = 3.6 m Co=10 mg/l Dw=5.8 mm/yr
ponds
mudflats
fraction of the total marsh area
Normalized primary production
1 mm/yr
2 mm/yr
3 mm/yr
4 mm/yr
5 mm/yr
6 mm/yr
7 mm/yr
8 mm/yr
Snapshot at time 1000 years
R = 2 mm/yr
R = 4 mm/yr
R = 6 mm/yr
R = 8 mm/yr
Elevation [m above MSL]
B
Tidal range = 0.7 m Co=5 mg/l Dw=0.6 mm/yr
ponds
mudflats
fraction of the total marsh area
Normalized primary production
Snapshot at time 1000 years
R = 2 mm/yr
R = 4 mm/yr
R = 6 mm/yr
R = 8 mm/yr
Elevation [m above MSL]
-500
0
500
1000
transient
time [years]
C
0 years
100 years
200 years
500 years
1000 years
Elevation [m above MSL]
Channel
Veg. platform
Transient pond
Drowned pond
Mudflat

902

Figure 9. A,B) Amount of pond and mudflat area through time for different RSLR rates for the case of Barnstable marsh (r=3.6 m) with a reduced sediment supply (C_o=10 mg/l) (A), and for the case of Sage Lot Pond marsh (r=0.7 m and C_o=5 mg/l) (B). The normalized primary production is calculated averaging the plant primary production divided by the maximum primary production (which is attained when the marsh is at the elevation optimum). The normalized primary production initially increases because the marsh attains a lower elevation but subsequently decreases when the pond area (whose primary production is assumed to be zero) increases. C) Snapshots of marsh configurations at different times (as indicated by the black dots in panel B) for the case of Sage Lot Pond marsh with R=4 mm/yr.



912 Figure 10. Predictions of total unvegetated area (ponds + mudflats) at different time intervals after the
913 increase in RSLR rate. The dashed lines indicate the threshold between pond recovery and pond runaway
914 regime using the lumped pond model of Mariotti (2018), with $R=D_{cr}=\alpha C_o r/(2T\rho_m)$. The comparison with
915 Barnstable and Sage Lot Pond marsh is made assuming that the increase in relative sea level rise rate
916 started about 100 years ago at the beginning of the 20th century.

917

918 Table 1. Sediment accretion rates measured on 1 cm sediment core intervals in Barnstable (measured
919 for this work) and Sage Lot Pond marshes (Gonneea et al., 2019, 2018).

920

Site	**Sediment accretion rate (1900 to 2018) mm/y**
Sage Lot Pond	
Interior (1 core, 13 intervals)	1.4 ± 0.3
Channel-adjacent (1 core, 27 intervals)	3.6 ± 2.0
Barnstable	
Interior (3 cores, 134 intervals)	5.1 ± 3.3
Channel-adjacent (3 cores, 130 intervals)	4.6 ± 2.5

921

922 Table 2. List of parameters used in the model.

	Symbol	**Description**	**Value**	**Reference**
	Δx	Spatial discretization	1 m	
	Δt	Temporal discretization	1 year	
Physical parameters	C_o	SSC in channels	30 mg/l or 5 mg/l	Measured
	ρ_m	Mud dry bulk density	650 kg/m^3	Assumed
	β	Horizontal decay rate of SSC with distance from channel network	0.05 m^{-1}	(Christiansen et al., 2000)
	α	Fraction of spatially uniform SSC	0.3	Assumed

	r	Spring tidal range	3.6 m or 0.7 m	NOAA Station 8447930 and Station 8447241
	Δr	Spring-neap variability	0.05r	NOAA
	T	Tidal period	12.5 hr	NOAA
	R	Relative Sea Level Rise rate	2.9 mm/yr	NOAA
	μ	Soil creep diffusivity	0.1 m^2/yr	(Mariotti et al., 2019)
Vegetation parameters	z_{min}	Min elevation for vegetation	0	Assumed
	z_{max}	Max elevation for vegetation	r/2	Assumed
	D_{pMAX}	Max in-situ organic deposition	6 mm/yr	Assumed
Pond Parameters	k_{seed}	Pond formation rate	$4 \cdot 10^{-4}$ #ponds/m^2/yr	Calibrated
	k_{exp}	Pond expansion rate	0.015 m/yr	Measured
	L	Drainage influence length	20 m	Measured
	Y_{pond}	Max initial depth of new ponds	$0.25(z_{max}-z_{max})$	Calibrated
	$P_{deepening}$	Active deepening of ponds	3 mm/yr	Calibrated

923

924