

1 Transient coastal landscapes: Rising sea level threatens salt marshes

2

3 Ivan Valiela^a, Javier Lloret^{a,1}, Tynan Bowyer^{a,b}, Simon Miner^a, David Remsen^a, Elizabeth
4 Elmstrom^{a,2}, Charlotte Cogswell^c, and E. Robert Thieler^d

5

6 ^aThe Ecosystems Center, Marine Biological Laboratory, Woods Hole MA US 02543; ^bThe
7 University of Chicago, Chicago IL US 60637; ^cCR Environmental, Inc. 639 Boxberry Hill Road,
8 East Falmouth, MA US 02536; ^dUnited States Geological Survey, 384 Woods Hole Road,
9 Woods Hole, MA US 02543

10

11 ¹Corresponding author. Email: jlloret@mbl.edu. Phone: +1 508 289 7699.

12 ²Current address: School of Aquatic and Fishery Sciences, University of Washington,
13 1122 NE Boat St, Box 355020, Seattle, WA 98195

14

15 **Abstract**

16 Salt marshes are important coastal environments that provide key ecological services. As sea
17 level rise has accelerated globally, concerns about the ability of salt marshes to survive
18 submergence are increasing. Previous estimates of likely survival of salt marshes were based on
19 ratios of sea level rise to marsh platform accretion. Here we took advantage of an unusual, long-
20 term (1979-2015), spatially detailed comparison of changes in a representative New England salt
21 marsh to provide an empirical estimate of habitat losses based on actual measurements. We show
22 prominent changes in habitat mosaic within the marsh, consistent and coincident with increased
23 submergence and coastal erosion. Model results suggest that at current rates of sea level rise,

24 marsh platform accretion, habitat loss, and with the limitation of the widespread “coastal
25 squeeze”, the entire ecosystem might disappear by the beginning of the next century, a fate that
26 might be likely for many salt marshes elsewhere.

27

28 **Keywords:** salt marsh loss, sea level rise, ecological services, vegetation mosaics, coastal
29 squeeze.

30

31 **Highlights:**

32 Accelerated sea level rise increased submergence in a New England salt marsh.

33 Increased sea level collapsed creek banks and widened channels.

34 Low marsh vegetation cover increased while high marsh cover decreased.

35 Habitat, sediment stability, C burial, foraging and nursery services were affected.

36 Vegetation cover seems likely to disappear by the beginning of the next century.

37 **1. Introduction**

38 Global sea levels have been rising since the last glacial maximum, at rates of around 1 mm yr⁻¹
39 during late Holocene (Kemp et al., 2011). These moderate sea level rise rates allowed
40 development of coastal salt marshes, through vertical build-up of salt marsh platforms, by
41 accumulation of underground organic matter and sediment deposition, at rates greater than or
42 equal to sea level rise (Redfield, 1972).

43 Sea levels have variably increased across recent centuries (Church and White, 2011), but
44 a significant recent acceleration of sea level rise is apparent across many coasts of the world
45 (Kopp et al., 2016; Merrifield et al., 2009), with hotspots including the Atlantic coast of North
46 America (Sallenger et al., 2012). Accelerated sea level rates are evident in Cape Cod (Fig. 1a),
47 where recent rate of sea level rise is almost six times higher than the rates observed during the
48 late Holocene. Such increases in sea level force a plethora of environmental effects on coastal
49 ecosystems such as salt marshes (Morris et al., 2002; Valiela, 2006).

50 Accelerated sea level rise rate, if not matched by accretion of the marsh platform, should
51 significantly increase submergence and shift the vegetation mosaic within salt marshes (Miller et
52 al., 2001; Smith, 2014). Plant zonation within salt marshes is vertically stratified, with species of
53 plants and algae constrained within certain elevation ranges (Valiela and Rietsma, 1995) (Fig.
54 1b, c). As sea level rises, the vegetation strata can only survive if they migrate into shallower
55 sites in a landward direction. Rising sea levels also force accompanying changes in coastline
56 position and re-distribute sediment within coastal wetlands (Kolker et al., 2009).

57 Concerns about sea level-driven alteration of vegetation and sediments in salt marsh
58 ecosystems are based on potential loss of important ecological services that these coastal
59 environments furnish—stabilization of sediment, shoreline protection, interception of nutrients,

60 provision of habitat use for shellfish, birds, and other fauna, sequestration of atmospheric carbon,
61 plus a number of other functions (Craft et al., 2009; Valiela, 2006). The plethora of ecological
62 services furnished by salt marshes result from ecological, biogeochemical, and geological
63 mechanisms that differ among the various vegetation and sediment habitats within salt marshes.
64 Thus, as sea level alters marsh habitat mosaics, it seems reasonable to expect potential changes
65 in services provided by marsh ecosystems, and loss of these services if the marsh disappears.

66 There have been differing conclusions about salt marshes facing rising sea levels, ranging
67 from exceptionally vulnerable (Watson et al., 2017), to threatened (Crosby et al., 2016), to
68 susceptible (Valiela, 2006), to over-estimated (Kirwan et al., 2016). These studies rely on meta-
69 analyses that compare estimated sea level rise to estimated salt marsh platform accretion.

70 Adaptability of salt marshes to rising sea levels depends on their ability to build vertically at
71 rates greater than or equal to relative sea level rise, or else to migrate inland at rates faster than
72 erosion at their seaward boundary. Moderate increases of flooding duration can increase mineral
73 sediment deposition rates (Vandenbruwaene et al., 2011), and plant productivity (Kirwan and
74 Guntenspergen, 2012), contributing to increased vertical accretion, and reduced erosion rates. In
75 addition, spatial and temporal variations in accretion, and plant-mediated changes of
76 hydrodynamics and sediment transport mechanics along the marsh platform, may also alter
77 wetland stability during periods of elevated sea level rise rates (Rodriguez et al., 2017; Sandi et
78 al., 2018; Alizad et al., 2016; Belliard et al., 2016). Such biogeomorphic feedbacks allowed
79 marshes to keep pace with rising sea levels during recent centuries (Kirwan et al., 2016; Morris
80 et al., 2002).

81 Current and likely future conditions forecast a more problematic future for salt marshes
82 through this century. First, comparisons with historical fluctuations in global sea level

83 reconstructions show that recent accelerated rates of global sea level rise are unprecedentedly
84 high, and expected to continue accelerating in the future (Merrifield et al., 2009). Second,
85 susceptibility of coastal wetlands to sea level rise might be greatest in coasts with lower fluvial
86 sediment sources to support platform accretion (Kirwan et al., 2010). Third, in populated areas,
87 salt marsh inland migration can be limited by the effects of the “coastal squeeze”, a phenomenon
88 that occurs when structures built landward of coastal wetlands, or steep upland topography,
89 prevent landward incursion of marsh vegetation as sea level rises (Doody, 2004).

90 Here we examine the effects of recent rising sea level on the habitat mosaic within a
91 coastal salt marsh by taking advantage of an unusual, spatially detailed comparison of decadal
92 changes in Great Sippewissett Marsh, on Cape Cod. Vegetation in the marsh is characteristic of
93 New England and other regions along the North American Atlantic Coast, with a variety of plant
94 species located at specific elevations within the tidal range according to their relative tolerance to
95 submergence (Ewanchuck and Bertness, 2004; Nixon, 1982, and Fig. 1b, and c). This marsh is,
96 as are many wetlands around the world, surrounded by human development. The marsh is
97 exposed to mean tidal range of about 1.65 m, and sediment accretion takes place largely through
98 accumulation of below ground plant biomass, rather than via riverine sediment inputs (Valiela,
99 2015).

100 In this paper we test whether 1) there have been recent decadal changes in the vegetation
101 mosaic of a representative salt marsh in New England; 2) the changes in salt marsh vegetation
102 were associated with parallel rise in sea level; and 3) knowledge of the links of recent vegetation
103 and sea level changes could provide the means, combined with forecasted future sea level rise, to
104 predict future trajectories of the marsh mosaic. Expanding scale from this empirically based

105 synthesis, we then carry out a literature meta-analysis to elucidate the fate of salt marshes across
106 the world's coasts to the end of this century in the context of accelerated sea level rise.

107

108 **2. Materials and Methods**

109 **2.1 Sea level rise data and calculations.**

110 Sea level data were obtained from a tide gauge station located eight km south of Great
111 Sippewissett Marsh, in Woods Hole, MA (NOAA Station ID#: 8447930). 1972 to 2013 monthly
112 sea level data were used to estimate sea level rise rates. To determine whether there had been a
113 significant shift in sea level rise, and when it took place, we carried out a piecewise linear
114 regression analysis of the sea level rise time series. The intent of this statistical analysis was to
115 objectively determine the best fits of regression models to segments of the time series of sea
116 level rise data, to estimate significant differences of sea level rise rates during different segments
117 of the data series, as well as identify the hinge years when significant shifts may have taken
118 place.

119 **2.2 Field mapping and GIS processing.**

120 The 1979 field-recorded mapping involved examination of the entire salt marsh. We subdivided
121 an outline map of the marsh, obtained from a detailed aerial photograph, into sub-units of the
122 marsh approximately of 100 m². These sub-units were then closely examined, and habitat data
123 recorded while walking through each area and recording position and taxonomic make up of
124 parcels with different vegetative cover, aiming at a resolution of at least 0.5 m.

125 To draw the habitat map for 2015, we developed an ortho-rectified map of Great
126 Sippewissett Marsh using an unpiloted aerial vehicle flown at an altitude of 50m. A DJI Phantom
127 III Professional quadcopter provided multiple overlapping aerial images using a built-in 12-

128 megapixel camera borne by the unpiloted aerial vehicle, and fitted with an f/2.8 lens with a 94-
129 degree field of view. Photographs were captured in an overlapping grid pattern optimized for the
130 generation of a composite orthomap. The application, Map Pilot 1.41, by DronesMadeEasy©
131 automatically generated the photographic grid and photo sequence through a simple user
132 interface where we defined the target area, altitude, and percentage overlap. We specified an
133 80% overlap to ensure a high-quality image. The unpiloted aerial vehicle automatically followed
134 the specified flight path and captured over 1,000 individual, overlapping images. A web service
135 provided by MapsMadeEasy.com allowed the resultant images to be uploaded for rendering.
136 The individual images were then aligned into a composite orthorectified image with a resolution
137 of 0.3 m, suitable for geospatial analysis, and comparable to the resolution of the 1979 map (Fig.
138 SI 1).

139 We used QGIS Semi-Automatic Classification Plugin 4.9.5 (SACP) to classify the color-
140 coded vegetation types present in the 1979 vegetation map to convert them to polygons in a
141 shapefile. To create a corresponding 2015 vegetation map, the unpiloted aerial vehicle-obtained
142 composite image was divided into ten sections of approximately equal size, each of which was
143 then classified individually. Open water, creek bank, sand dune and beach, and upland vegetation
144 categories were all manually classified and converted to polygons. High and low marsh
145 vegetation were then isolated and classified using SACP, according to the differences in texture
146 and coloration of the two vegetation types. This classification was then visually checked for
147 accuracy against the stitched image, and then further verified by confirming category boundaries
148 observed in situ. We then extracted vegetation type areas and point-by-point conversion data
149 from the resulting spatially referenced shapefiles. We measured shoreline movement by
150 averaging 10 measurements taken at equidistant points along the shore. Dune area was measured

151 above 41°34'52.4"N and below 41°35'16.2"N. Marsh vegetation data were not measured above
152 41°59'28.6"N, and the lower boundary was defined by a road to the south (Fig. SI 1).

153 All GIS analyses were completed using the QGIS 2.14 software. Both the 1979
154 vegetation map and the 2015 unpiloted aerial vehicle-obtained composite image were geo-
155 referenced against fully referenced aerial and satellite imagery corresponding to the years each
156 was completed, using buildings, roads and other fixed structures and features as reference points.
157 We used 23 control points for geo-referencing. The root mean square error during the process
158 was 3.3 m.

159 **2.3 Numerical modeling of future marsh habitat trajectories.**

160 The modeling effort aimed to predict trajectories of the different vegetation habitats through the
161 21st century as sea level rose. The strategy was to use habitat and vegetation changes we
162 observed in the real-time comparisons from the 1979 and 2015 maps, plus sea level and elevation
163 constraints for each marsh habitat (Fig. 1c), as the basis for anticipating future changes
164 throughout Great Sippewissett Marsh.

165 The modeling conservatively assumed that the current effective sea level rise as 2.93
166 mm·yr⁻¹ [the difference between current sea level rise rate of 5.83 mm yr⁻¹ (Fig. 1a) and
167 accretion rate (~2.9 mm yr⁻¹, Kinney and Valiela, 2013)], would be maintained through the
168 century. Some studies of salt marsh accretion assume increase in water depth favors accretion by
169 increasing external sediment deposition (Fagherazzi et al., 2012), but in Great Sippewissett
170 Marsh and other New England salt marshes, accretion is mainly from burial of roots and
171 rhizomes, rather than from fluvial sediment supply (Drake et al., 2015; Valiela, 2015). We need
172 to emphasize here that 1) fluvial transport of terrestrial sediment is minimal, and freshwater
173 enters the marsh via groundwater flow through sand-gravel aquifers (Valiela and Teal, 1979),

174 and 2) change in vegetation-stimulated accretion is captured in the model by use of the data on
175 vegetation cover itself. We conservatively assumed that the constrained vertical distribution of
176 the several vegetation zones (Fig. 1c), defined by different tolerance of plant species to
177 inundation and salinity (Kirwan and Guntenspergen, 2010), will not change. The model applied
178 current annual rates of change in area of vegetation type for each type of habitat, as a point of
179 departure for the simulation.

180 We then defined, at annual time-steps, the area of each elevation-constrained habitat (Fig.
181 1c) that would be present as effective sea level rose through the years. This defined conservative
182 future trajectories for the several habitats found within Great Sippewissett Marsh. Examination of
183 the surrounding terrain, and the presence of a constructed embankment on the east side of Great
184 Sippewissett Marsh (see Fig. SI 1) suggested that landward incursion by wetland vegetation
185 would be prevented (the “squeeze” effect), a ceiling to landward incursion that was also included
186 in the model.

187 From recent trajectories of sea level rise in the area (Fig. 1 a), we assumed that the areas
188 of the various habitats were relatively constant in the period between 1979 and 1992, when sea
189 level rise rate was relatively low, at 0.99 mm yr^{-1} . Long-term (1932-1992) rate of sea level rise in
190 this location was 2.54 mm yr^{-1} . Both, long- and short-term pre-1992 rates fall below the average
191 accretion rates in Great Sippewissett Marsh of around 2.9 mm yr^{-1} (Kinney and Valiela, 2013).
192 We therefore concluded that observed habitat changes took place after the acceleration of rates of
193 sea level rise to 5.83 mm yr^{-1} after 1992.

194 Observed rates of habitat change were used to numerically model possible post-2015
195 trajectories of these habitats in Great Sippewissett Marsh. For each year, changes in the different

196 habitats were calculated as the balance between gains and losses of areas of the specific habitats,
 197 and propagated into the future as

$$198 \quad A_{X,t} = A_{X,t-1} + \text{Again}_X - \text{Aloss}_X, \quad (1)$$

199 where $A_{X,t}$ is the area covered by habitat X in year t , $A_{X,t-1}$ the area in the previous year, and
 200 Again_X and Aloss_X the amount of area gained or lost by habitat X each year.

201 For un-vegetated sandy habitats and dunes, we assumed no gains in area as sea level
 202 rises. Our data revealed partial colonization by low marsh of sandy habitats and of the landward
 203 side of the dunes as these habitats are progressively inundated (see Fig. SI 2). We assumed this
 204 colonization to continue at current rates until the total colonization of available sandy habitat
 205 area. Changes in area of sandy habitats were calculated as

$$206 \quad A_{SH,t} = A_{SH,t-1} - \text{Col}_{LMtoSH}, \quad (2)$$

207 where Col_{LMtoSH} the amount of area of sandy habitats colonized by low marsh each year.

208 For dunes, measured losses were due to low marsh colonization in the landward side and
 209 submergence in the seaward side. We assumed that initially both sides of the marsh were
 210 approximately equal so low marsh colonization will only occur until the complete eastern half
 211 was covered by this type of vegetation. Changes in dune habitats were calculated as

$$212 \quad A_{DH,t} = A_{DH,t-1} - \text{Col}_{LMtoDH} - \text{Subm}_{DH}, \quad (3)$$

213 where Col_{LMtoDH} is the area of dunes colonized by low marsh, and Subm_{DH} the area dunes lost
 214 by submergence each year.

215 We assumed, from knowledge of the local landscape (and Fig. SI 1) that the “squeeze”
 216 effect exerted on high marsh vegetation by the berm on the east side of the marsh, and *P.*
 217 *australis*, will continue into the future, preventing inland expansion of the marsh. Therefore, for

218 high marsh vegetation, the coastal “squeeze” prevented up-gradient incursion, and hence
 219 constrained the gains in area in the upper range of topography,

$$220 \quad A_{HM,t} = A_{HM,t-1} - Subm_{HM} - Col_{PhragtoHM}, \quad (4)$$

221 where $Subm_{HM}$ is the area of high marsh lost by submergence of its lower elevation limit and
 222 $Col_{PhragtoHM}$ is the area of high marsh colonized by *P. australis* each year.

223 Most of the 1978 area of high marsh lost in 2015 was invaded by low marsh. We
 224 assumed that the area of high marsh lost by submergence each year was initially colonized by
 225 low marsh,

$$226 \quad Subm_{HM} = Col_{LMtoHM}, \quad (5)$$

227 where Col_{LMtoHM} is the area of high marsh colonized by low marsh each year.

228 To model changes in area of *P. australis* we assumed a constant rate of colonization while high
 229 marsh habitats are still present. After high marsh areas disappear, and in spite of tolerance to
 230 salinity of the new genetic variant, *P. australis* area was assumed to recede and be invaded by
 231 low marsh at the same rate as high marsh,

$$232 \quad A_{Phrag,t} = A_{Phrag,t-1} + Col_{PhragtoHM} \text{ if } A_{HM,t} > 0, \quad (6)$$

$$233 \quad A_{Phrag,t} = A_{Phrag,t-1} - Subm_{HM} \text{ if } A_{HM,t} = 0. \quad (7)$$

234 Future trajectories of low marsh vegetation were calculated as the balance between the
 235 areas gained by colonization of high marsh, sand and dune habitats, and the by the effects of sea
 236 level rise as

$$237 \quad A_{LM,t} = A_{LM,t-1} + Col_{LMtoHM} + Col_{LMtoSH} + Col_{LMtoDH} - Subm_{LM}, \quad (8)$$

238 where $Subm_{LM}$ is the area of low marsh lost by submergence of its lower elevation limit each
 239 year.

240 To solve equation 8, two different periods were defined. In the first period, where high
 241 marsh and sandy habitats were still present, low marsh upland colonization was still possible.
 242 The processes of colonization of new areas and accretion would redistribute much of the area
 243 covered by low marsh in the optimum central part of its elevation range (see Fig. 1b, c). We
 244 assumed that low marsh area cover follows a normal distribution (Tabak et al., 2016), with
 245 99.9% of the area of low marsh distributed in the 30 cm of its elevation range, and much of its
 246 area distributed around the central part of this range. The losses caused by the effects of sea level
 247 rise would then occur in the lower elevation limit of low marsh distribution. We assumed that the
 248 amount of low marsh area at this lower elevation limit would be minimum and, therefore, we
 249 expect minimum losses in terms of area per year as sea level rises during this initial period,

$$250 \quad Subm_{LM} = minSubm_{LM} \text{ if } t < T_{Sq}, \quad (9)$$

251 where T_{Sq} is the year where the sum of high marsh, *P. australis* and sandy habitats is null, and
 252 upland colonization of low marsh is not possible.

253 After the total loss of higher elevation areas, the “squeeze” effect will be exerted on low
 254 marsh. The cumulative losses of the normally distributed low marsh area would accelerate each
 255 year until half of the elevation range has been inundated and then decrease until the entire range
 256 is inundated. To simulate the changes in the rate of low marsh loss in this period, as the normal
 257 distribution of areas are inundated, we assumed that

$$258 \quad Subm_{LM} = minSubm_{LM} + \frac{maxA_{LM}}{\sqrt{2\sigma^2\pi}} e^{-\frac{(t-\mu)^2}{2\sigma^2}} \text{ if } t > T_{Sq}, \quad (10)$$

259 where $maxA_{LM}$ is the area of low marsh in the year T_{Sq} . The parameters for the calculation of
 260 losses of a normal distribution of low marsh area were calculated as

$$261 \quad \mu = \frac{1}{2} \frac{Range_{LM}}{SLR - Accr_{LM}}, \text{ and} \quad (11)$$

$$262 \quad \sigma = \frac{1}{6} \frac{Range_{LM}}{SLR - Accr_{LM}}, \quad (12)$$

263 where $Range_{LM}$ is the elevation range covered by low marsh, SLR the rate of sea level rise, and
 264 $Accr_{LM}$ is the rate low marsh accretion each year.

265 For the case of the creek banks, the gains in area by colonization of low marsh in the
 266 higher elevation limit of its distribution, and the losses by inundation of its lower limit, both
 267 caused by sea level rise, were considered equal during the first period of low marsh expansion.
 268 During this period, observed losses were assumed to be caused only by edge slumping,

$$269 \quad A_{CB,t} = A_{CB,t-1} - Slump_{CB} \text{ if } t < T_{Sq}, \quad (13)$$

270 where $Slump_{CB}$ is the rate of creek bank edge slumping each year.

271 After low marsh starts to recede, further losses of creek bank took place at a rate
 272 proportional to low marsh losses, and to the relative areas of both habitats. Creek bank
 273 vegetation area was then calculated as

$$274 \quad A_{CB,t} = A_{CB,t-1} - Slump_{CB} + (A_{LM,t} - A_{LM,t-1}) \frac{A_{CB,t-1}}{A_{LM,t-1}} \text{ if } t > T_{Sq}, \quad (14)$$

275 The calculation of the different terms used in the previous equations were based on
 276 observed changes in areas resulting from the comparison of 1979 and 2015 datasets, the
 277 elevation ranges of the various Great Sippewissett Marsh habitats (Valiela and Rietsma, 1995),
 278 measured accretion rates in Great Sippewissett Marsh (Kinney and Valiela, 2013), and the post-
 279 1992 local rate of sea level rise of 5.83 mm yr⁻¹. Values for these variables are in Table SI 1.

280 To assess uncertainty in sea level rise effects on the different habitat covers, the standard
 281 error of the post-1992 sea level rise estimates was used to recalculate model parameters,
 282 assuming the various habitat parameter responses are proportional to the difference between sea
 283 level rise and habitat accretion rates,

$$284 \quad P_{X,SLR \pm S.E.} = P_{X,SLR} \frac{SLR \pm S.E. - Accr_X}{SLR - Accr_X}. \quad (15)$$

285

286 **3. Results and discussion**

287 **3.1 Sea level rise changes.**

288 Sea level rise was variable throughout the study period (Fig. 1a), but remained at about 1 mm yr⁻¹
 289 up to the hinge year of 1992, and then rose to 5.83 mm yr⁻¹ after that year. This difference
 290 appears large enough to expect significant effects on the vegetation and habitats of Great
 291 Sippewissett and similar salt marshes.

292 **3.2 Overall changes in Great Sippewissett Marsh, 1979-2015.**

293 The total area of Great Sippewissett Marsh system changed slightly between 1979 and 2015 (Fig.
 294 2a, b, and Table SI 2). The marsh includes areas of vegetated cover, sand deposits (beach and
 295 dunes), and open water (Fig. 2a). The vegetated area was about twice that of open water, a
 296 proportion that suggests that Great Sippewissett is a reasonably mature marsh, where platform
 297 accretion historically exceeded, or kept up with sea level rise, vegetated areas predominated, and
 298 where exports of particulate organic materials to the adjacent sea were likely (Valiela, 1983).

299 Between 1979 and 2015 there was an almost 6% increase in open water area, and 3% and
 300 19% decreases in vegetated cover and sand deposits, respectively (Fig. 2b). These recent changes
 301 suggest modest but detectable changes for the marsh as a whole, effects consistent with increased
 302 vegetation submergence and erosion of beach and dunes.

303 **3.3 Alteration of the vegetation mosaic.**

304 There were substantial changes in the specific units of the vegetation and habitat mosaic within
305 Great Sippewissett Marsh between 1979 and 2015 (Fig. 2c, d and Fig. 3).

306 *3.3.1 Loss of creek banks.*

307 There was a 14% loss of creek banks and low-lying habitats that supported tall *Spartina*
308 *alterniflora* (Fig. 2d). Creek bank loss was by slumping following increased submergence (Fig.
309 SI 3a, b). Slumping of creek banks has been attributed to joint effects of nitrogen-related
310 eutrophication and sea level rise (Deegan et al., 2012); both these drivers favor conversion of
311 stands of short into taller *S. alterniflora*, changes that reduces marsh sediment cohesion (Deegan
312 et al., 2012; Valiela, 2015). We found, however, that concentrations of dissolved inorganic
313 nitrogen in waters of Great Sippewissett Marsh, while variable, have not increased across recent
314 decades (Table SI 3), during which sea level has risen substantially in excess of local accretion
315 of salt marsh platform (Valiela, 2015). This suggests that sea level effects might be the prime
316 driver of creek bank loss in Great Sippewissett Marsh.

317

318 *3.3.2 Increase in low marsh and decrease in high marsh.*

319 The shift in vegetation mosaic included a 43% increase in area of low marsh vegetation,
320 dominated by short *S. alterniflora* (Fig. 2d, Fig. 3a, and Table SI 2). During the same period,
321 area covered by high marsh vegetation (mainly *Spartina patens*) decreased by nearly 60% (Fig.
322 2d, Fig. 3b, and Table SI 2). Increased submergence owing to recent sea level rise therefore
323 shifted much of what was high to low marsh species cover, accounting for 66.5% of the gain in
324 low marsh area cover; the remaining gain in low marsh cover was at the expense of newly
325 submerged sand habitats. These major changes in the vegetated mosaic took place within the

326 marsh even though there were only minor changes in total area cover of vegetation and open
327 water (Fig. 2b, and Table SI 2). The increase in lower-lying low marsh at the expense of high
328 marsh and sand habitats is clearly consistent with increased submergence as sea level rose, as
329 reported for century-scale and decadal- and shorter-scale effects of sea level rise (Kolker et al.,
330 2009; Raposa et al., 2017; Rietsma et al., 2011; Watson et al., 2016).

331 An obvious question is why high marsh plants did not simply migrate towards upland
332 areas as sea level rose? We already mentioned two possible mechanisms in the discussion of the
333 model approach. First, such a movement was made difficult by the presence of an embankment
334 or berm, built during the 1870s to support a railroad (now a bicycle path) along the eastern
335 boundary of the marsh (Fig. SI 1). Second, during the nearly four decades of our study, the
336 landward edges of Great Sippewissett Marsh have been subject to invasion (Fig. 2d, and Table SI
337 2) by a genetic variant of common reed, *Phragmites australis*, as reported for other NE US
338 coasts (Hazelton et al., 2014).

339 Instead, it is thought that the new variant is favored by its increased salinity tolerance and
340 affinity for uptake of increased concentrations of dissolved inorganic nitrogen in freshwater
341 flowing into salt marshes (Valiela, 2015). This invasive form of *P. australis* prevents landward
342 movement of high marsh (Smith, 2013) owing to its characteristic growth as a narrow but
343 aggressive phalanx along landward margins of salt marshes (Fig. SI 3c). *P. australis* is taller than
344 high marsh plants, has robust belowground roots and rhizomes, and easily shades and excludes
345 high marsh grasses.

346 **3.4 Consequences for salt marsh ecological services.**

347 Shifts in the vegetation mosaic following sea level rise may also alter the many other ecological
348 services provided by salt marshes, including stability of underlying sediments, sequestration of
349 carbon, and provision of food and nursery roles for animals.

350 *3.4.1 Marsh sediment stability.*

351 The changes recorded in the distribution of creek banks, low, and high marsh species are likely
352 important for maintenance of stability of underlying sediments, an important ecological service
353 provided by marsh vegetation. *S. alterniflora*, the dominant species in most Western Atlantic salt
354 marshes, grows in taller, sparser stands low in the intertidal, and as dwarf, dense stands higher in
355 the tidal range (Fig. 1b). The interwoven mass of roots and rhizomes of salt marsh plants is
356 largely responsible for the consolidation of the otherwise loose sediments in marshes (Deegan et
357 al., 2012; Valiela, 2015), but this stabilizing service is not uniform across the tidal range. The
358 short *S. alterniflora* confers considerably greater binding to the underlying sediment because its
359 root and rhizome mass is much denser than the tall form. Sediments in low-lying creek banks,
360 usually supporting tall *S. alterniflora*, are less stable, and more likely to suffer erosion and
361 slumping (Fig. SI 3a, b). On the whole, therefore, increased submergence weakens stability of
362 salt marsh platforms.

363 *3.4.2 Carbon sequestration.*

364 Sea level-influenced vegetation shifts alter sequestration of carbon (Kirwan and Mudd, 2012;
365 Morris et al., 2002). C burial rates in creek banks is about 4.7 times the rates measured in high
366 marsh; carbon burial in low marsh is about 1.9 times those recorded in high marsh sediments
367 (Ouyang and Lee, 2014). The stimulation of carbon sequestration associated with stands of *S.*
368 *alterniflora* is consistent with observations in China and the British Isles (Gao et al., 2012;
369 Ouyang and Lee, 2014).

370 3.4.3 Foraging and nursery services.

371 Sea level-related shifts of area of low and high marsh have consequences for consumers that feed
372 and reproduce in creek bank, low marsh, and high marsh habitats.

373 Loss of creek bank habitat will impair certain species with major ecological and
374 biogeochemical roles within salt marsh ecosystems. Creek bank and low-lying habitats are
375 essential for a number of key marsh species. Fiddler crabs (*Uca pugnax*) play fundamental roles
376 in sediment biogeochemistry and turnover (Wang et al., 2010), and their numerous burrows are
377 most abundant within low-lying marsh sediments, mainly where tall *S. alterniflora* grows
378 (Bertness and Miller, 1984; Krebs and Valiela, 1978). A number of bivalves, including oysters
379 (*Crassostrea virginica*) and ribbed mussels (*Geukensia demissa*) live largely on salt marsh creek
380 banks (Evgenidou and Valiela, 2002). Ribbed mussels by themselves were numerous enough in
381 Great Sippewissett marsh in the 1980s to filter more than the whole volume of tidal exchange of
382 water daily (Jordan and Valiela, 1982), and hence maintained water clarity. The narrow and
383 patchy strips of tall stands of *S. alterniflora* on salt marsh creek banks are the sole nesting habitat
384 of seaside sparrows (*Ammodramus caudacutus*), whose nests are extremely vulnerable to even
385 slight increases in sea level (Gjerdrum et al., 2005) and whose populations are declining (Berry
386 et al., 2015).

387 Expansion of low marsh at the expense of high marsh favors feeding by *Fundulus*
388 *heteroclitus* and other fish species, predators that forage within the low marsh canopy during
389 high tide (Werme, 1981), but are unable to penetrate into and feed within the much denser stands
390 of high marsh vegetation (Vince et al., 1976). Sharp-tailed sparrows (*Ammodramus maritimus*)
391 nest in high marsh sites, and nesting success of this vulnerable species is highly susceptible to

392 flooding regimes (Gjerdrum et al., 2005), so that reproduction is reduced by rising sea level
393 (Rush et al., 2009).

394 **3.5 Erosion, transport, and biological effects of changes in sediments.**

395 Beach and dunes along the seaward margin of Great Sippewissett Marsh (Fig. SI 1) receded to
396 the East by a mean of 22 m between 1979 and 2015 (Fig. 3c, and Fig. SI 4). The removed sand,
397 transported by storm over-wash and net tidal exchange into the marsh, seems likely to be
398 responsible for the 82% increase in area of sand-covered bare bottom across the period (Fig. 2c,
399 Fig. 3d, and Table SI 2). This extension of sand cover was synchronous with a 47% decrease in
400 mud-covered bottom (Fig. 2c, Fig. 3d, and Table SI 2).

401 Continued reduction of beach and dune areas will have effects on a number of species,
402 including nest success and population recovery by piping plover (*Charadrius melodus*) and least
403 tern (*Sterna albifrons*), federally listed species that use these limited habitats as nesting areas.
404 Reduction of nesting habitat changes are therefore of avian conservation concern in the region.
405 The transition of mud to sand bottoms under open water (Fig. 2c,d) should have had other
406 biological effects, such as increased supply of food for consumers entering marshes to feed, since
407 production of benthic invertebrate biomass in sandy bottoms of Great Sippewissett Marsh is 2.5
408 to 16 times that reported for muddy sediment (Sarda et al., 1995). The shift in benthic sediment
409 cover towards sandy sediments therefore has likely increased food supply for commercially
410 important fish [flounder species, menhaden (*Brevoortia tyrannus*), bluefish (*Pomatomus*
411 *saltatrix*), and other species], as well as for shrimp and crabs, that use open water habitats within
412 salt marshes as early-life history foraging areas in New England and elsewhere in the world
413 (Hampel et al., 2005; Sá et al., 2006).

414 **3.6 Future of Great Sippewissett Marsh.**

415 The modeled potential future trajectories of the total area and the vegetation mosaic of Great
416 Sippewissett, yielded a forecast of the future of the marsh.

417 We assumed no changes in vegetated habitats of Great Sippewissett Marsh (creek bank,
418 low marsh, and high marsh) between 1978 and 1992, a period during which platform accretion
419 rates approximately matched long-term sea level rise rates (left side of Fig. 4, whole lines). As
420 sea level rise rates accelerated by about six-fold after 1992, trajectories were set to match
421 measured habitat changes that took place in the marsh until 2015 (right side of Fig. 4, whole
422 lines). After 2015, and as sea level continues to rise, the model anticipates major changes in
423 vegetated habitats (Fig. 4a, dashed lines), and predicts a progressive transformation of the salt
424 marsh into open water areas (Fig. 4b, dashed lines).

425 With regard to vegetated habitats, modeled trajectories predict that creek banks will
426 continue to slump, and the tall stands of *S. alterniflora* growing on the banks will largely
427 disappear from Great Sippewissett Marsh by about the end of the century. Increased
428 submergence will initially foster expansion of low marsh vegetation to almost twice its original
429 area by 2030 (Fig. 4a). Most of the low marsh expansion will be at the expense of high marsh
430 areas (Fig. 4a). After disappearance of high marsh around 2030, low marsh extent will decrease
431 rapidly as it is further submerged, and has no available elevated areas to invade. If the land
432 margin adjacent to the marsh were not steep or populated and protected from flooding by people,
433 the high marsh could have grown landward, and could have survived considerably longer.

434 Our simulation did not allow high nor low marsh species to expand onto upland areas
435 because of the coastal “squeeze” against the built environment (such as the embankment
436 supporting the bicycle path to the East of the marsh, Fig. SI 1), and because of the dominant
437 stand of *P. australis* along the landward margin of Great Sippewisset Marsh (Fig. SI 3c). Both

438 low and high marsh will likely be gone by 2100—with an uncertainty of 30-40 years (Fig. 4).
439 Great Sippewissett Marsh will then have largely disappeared, and the coast will show a shallow
440 embayment in its place (Fig. 4b).

441 A recent evaluation of salt marsh vulnerability to current sea level rise rates showed that,
442 based on the quantification of the marsh's sediment budget, many other North American marshes
443 are likely to disappear in a similar time period (Ganju et al., 2017). According to the calculations
444 of that same study, and with a ratio of open water to vegetated marsh of 0.59 in 2015, the likely
445 lifespan of the marsh complex in Great Sippewissett could be just slightly more than 100 years, a
446 result that matches almost exactly the calculations from our modeled trajectories.

447 An earlier study of sediment cores taken from Great Sippewissett Marsh suggested that
448 about 2500 years ago, the area that is now Great Sippewissett Marsh included a shallow coastal
449 embayment with a tidal mudflat, a rather narrow salt marsh rim, and some freshwater swamp
450 towards land (Treggor, 1983). That description matches what the model anticipates will be the
451 case after 2100. These conclusions point out that salt marshes such as Great Sippewissett are
452 indeed transient coastal environments at geological scales, with lifetimes of a few thousand
453 years.

454 **3.7 Future trajectories of salt marshes in general.**

455 The future of salt marshes and other coastal wetlands across the world is a fraught with
456 uncertainty; the only guarantee is that major changes are inevitable. Although sea level rise is a
457 major driver of changes (Tabak et al., 2016), the future of salt marshes also depend on the
458 relative magnitude of sea level rise vs. sediment accretion, two variables that differ greatly
459 among coastal wetlands (Fig. 5). The local variability in such variables is reflected in the diverse
460 results reported. Kirwan et al. (2016) reviewed ratios of sea level rise and accretion in salt

461 marshes, and suggested that marsh loss might be overestimated. Other meta-analyses
462 (Blankespoor et al., 2014; US Fish and Wildlife Service, 2012; Watson et al., 2017) concluded
463 that marsh loss may indeed be large.

464 The comparison of sea level rise vs accretion pointed out many points above and below
465 the 1:1 line that indicates a balance between the two processes. Enumeration of the points above
466 and below the 1:1 line suggested that 40.3% of salt marshes around the world could be fated to
467 be submerged (Fig. 5). The meta-analyses included sites where riverine sediment inputs support
468 accretion, a condition that may not be true of all salt marsh sites, and did not at all consider the
469 widespread occurrence of coastal squeeze effects owing to human settlement of coastal areas
470 (Valiela, 2006). The meta-analyses of Fig. 5 also assumed no further increases in sea level rise.
471 Sea level rise, however, has accelerated during recent decades. Since the early 1990s, current
472 rates of global sea level rise have almost doubled (Chen et al., 2017; Dangendorf et al., 2017;
473 Merrifield et al., 2009). The acceleration will, conservatively, result in a doubling of rise of
474 global sea level by 2100, forcing a rise of 65 cm by 2100, relative to sea level in 2005 (Nerem et
475 al., 2018). Salt marshes, much like other coastal wetlands, are therefore facing a threatening
476 future rise in sea level. To project a probable future, we re-drew the 1:1 line in Fig. 5, to a
477 position likely to exist at the end of this century, assuming that accretion will remain unchanged,
478 but sea level will rise as presently forecasted (Nerem et al., 2018). Under these conditions, by
479 2100, 94.9% of the 315 salt marsh sites—particularly those where platform accretion is largely
480 through below-ground biomass accumulation—would likely become submerged. Although this
481 oversimplifies conditions and does not account for future changes in accretion, these numbers
482 still suggest that salt marshes are, and will continue to be, a significantly threatened major
483 coastal environment.

484 The magnitude of changes forced by sea level rise on coastal wetlands is and will be
485 significant, with greater submergence followed by slumping creek banks, less stable sediments,
486 more low marsh, changing un-vegetated sediments, and likely decreased ability of high marsh to
487 migrate landward. Further sea level rise will thoroughly alter the way salt marsh ecosystems
488 convey their important ecological services—stabilization of coastlines, carbon sequestration,
489 provision of wildlife habitats, and other services such as interception of nitrogen (Valiela, 2006).

490 Sea level rise will have measurable effects on the functioning of these ecosystems, with
491 consequences for ecological services, and will make developing management measures a serious
492 challenge. Moreover, the habitat mosaic and the topographic and hydrodynamic setting of
493 different salt marshes will differ. Predictions of future trajectories in function and services, and
494 planning strategies toward marsh preservation, will therefore have to be locality-specific. There
495 is little doubt, however, that the best and most broadly meaningful preservation plan would
496 include preventive global-scale measures aimed at diminishing sea level rise by decreasing
497 atmospheric warming.

498

499 **Acknowledgements**

500 Ivan Valiela and Elizabeth Elmstrom were supported by Woods Hole Sea Grant, NOAA grant
501 no. NA14OAR4170074. Javier Lloret was supported by a Rosenthal Postdoctoral Fellowship
502 Award from the Marine Biological Laboratory, and by a Northeast Climate Science Center
503 Fellowship. Tynan Bowyer was supported by a Metcalf Research Fellowship of the University of
504 Chicago. David Remsen was supported by MBL Cox and Bernstein funds. USGS supports the
505 conclusions of research conducted by their employees, and peer reviews and approves all of their

506 products consistent with USGS Fundamental Science Practices. Mention of trade names or
507 commercial products does not constitute endorsement or recommendations for use.

508

509 **References**

510 Alizad, K., Hagen, S.C., Morris, J.T., Medeiros, S.C., Bilskie, M. V., Weishampel, J.F., 2016.

511 Coastal wetland response to sea-level rise in a fluvial estuarine system. *Earth's Futur.* 4, 483–

512 497. <https://doi.org/10.1002/2016EF000385>

513 Belliard, J.P., Di Marco, N., Carniello, L., Toffolon, M., 2016. Sediment and vegetation spatial

514 dynamics facing sea-level rise in microtidal salt marshes: Insights from an ecogeomorphic

515 model. *Adv. Water Resour.* 93, 249–264. <https://doi.org/10.1016/j.advwatres.2015.11.020>

516 Berry, W.J., Reinert, S.E., Gallagher, M.E., Lussier, S.M., Walsh, E., 2015. Population status of

517 the seaside sparrow in Rhode Island: A 25-year assessment. *Northeast. Nat.* 22, 658–671.

518 <https://doi.org/10.1656/045.022.0403>

519 Bertness, M.D., Miller, T., 1984. The distribution and dynamics of *Uca pugnax* (Smith) burrows

520 in a New England salt marsh. *J. Exp. Mar. Bio. Ecol.* 83, 211–237.

521 [https://doi.org/10.1016/S0022-0981\(84\)80002-7](https://doi.org/10.1016/S0022-0981(84)80002-7)

522 Blankespoor, B., Dasgupta, S., Laplante, B., 2014. Sea-level rise and coastal wetlands. *Ambio*

523 43, 996–1005. <https://doi.org/10.1007/s13280-014-0500-4>

524 Chen, X., Zhang, X., Church, J.A., Watson, C.S., King, M.A., Monselesan, D., Legresy, B.,

525 Harig, C., 2017. The increasing rate of global mean sea-level rise during 1993-2014. *Nat. Clim.*

526 *Chang.* 7, 492–495. <https://doi.org/10.1038/nclimate3325>

527 Church, J.A., White, N.J., 2011. Sea-level rise from the late 19th to the early 21st century. *Surv.*

528 *Geophys.* 32, 585–602. <https://doi.org/10.1007/s10712-011-9119-1>

- 529 Craft, C., Clough, J., Ehman, J., Joye, S., Park, R., Pennings, S., Guo, H., Machmuller, M., 2009.
530 Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. *Front.*
531 *Ecol. Environ.* 7, 73–78. <https://doi.org/10.1890/070219>
- 532 Crosby, S.C., Sax, D.F., Palmer, M.E., Booth, H.S., Deegan, L.A., Bertness, M.D., Leslie, H.M.,
533 2016. Salt marsh persistence is threatened by predicted sea-level rise. *Estuar. Coast. Shelf Sci.*
534 181, 93–99. <https://doi.org/10.1016/j.ecss.2016.08.018>
- 535 Dangendorf, S., Marcos, M., Wöppelmann, G., Conrad, C.P., Frederikse, T., Riva, R., 2017.
536 Reassessment of 20th century global mean sea level rise. *Proc. Natl. Acad. Sci.* 114, 5946–5951.
537 <https://doi.org/10.1073/pnas.1616007114>
- 538 Deegan, L.A., Johnson, D.S., Warren, R.S., Peterson, B.J., Fleeger, J.W., Fagherazzi, S.,
539 Wollheim, W.M., 2012. Coastal eutrophication as a driver of salt marsh loss. *Nature* 490, 388–
540 392. <https://doi.org/10.1038/nature11533>
- 541 Doody, J.P., 2004. “Coastal squeeze” – an historical perspective. *J. Coast. Conserv.* 10, 129–138.
542 [https://doi.org/10.1652/1400-0350\(2004\)010\[0129:CSAHP\]2.0.CO;2](https://doi.org/10.1652/1400-0350(2004)010[0129:CSAHP]2.0.CO;2)
- 543 Drake, K., Halifax, H., Adamowicz, S.C., Craft, C., 2015. Carbon sequestration in tidal salt
544 marshes of the Northeast United States. *Environ. Manage.* 56, 998–1008.
545 <https://doi.org/10.1007/s00267-015-0568-z>
- 546 Evgenidou, A., Valiela, I., 2002. Response of growth and density of a population of *Geukensia*
547 *demissa* to land-derived nitrogen loading in Waquoit Bay, Massachusetts. *Estuar. Coast. Shelf*
548 *Sci.* 55, 125–138. <https://doi.org/10.1006/ecss.2001.0891>
- 549 Ewanchuck, P.J., Bertness, M.D., 2004. Structure and organization of a northern New England
550 salt marsh plant community. *J. Ecol.* 92, 72–85.

- 551 Fagherazzi, S., Kirwan, M.L., Mudd, S.M., Guntenspergen, G.R., Temmerman, S., D'Alpaos, A.,
552 van de Koppel, J., Rybczyk, J.M., Reyes, E., Craft, C., Clough, J., 2012. Numerical models of
553 salt marsh evolution: Ecological, geomorphic, and climatic factors. *Rev. Geophys.* 50, RG1002.
554 <https://doi.org/10.1029/2011RG000359>
- 555 Ganju, N.K., Defne, Z., Kirwan, M.L., Fagherazzi, S., D'Alpaos, A., Carniello, L., 2017.
556 Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes. *Nat.*
557 *Commun.* 8, 14156. <https://doi.org/10.1038/ncomms14156>
- 558 Gao, J., Bai, F., Yang, Y., Gao, S., Liu, Z., Li, J., 2012. Influence of *Spartina* colonization on the
559 supply and accumulation of organic carbon in tidal salt marshes of northern Jiangsu Province,
560 China. *J. Coast. Res.* 280, 486–498. <https://doi.org/10.2112/JCOASTRES-D-11-00062.1>
- 561 Gjerdrum, C., Elphick, C.S., Rubega, M., 2005. Nest site selection and nesting success in
562 saltmarsh breeding sparrows: The importance of nest habitat, timing, and study site differences.
563 *Condor* 107, 849–862. <https://doi.org/10.1650/7723.1>
- 564 Hampel, H., Cattrijsse, A., Elliott, M., 2005. Feeding habits of young predatory fishes in marsh
565 creeks situated along the salinity gradient of the Schelde estuary, Belgium and The Netherlands.
566 *Helgol. Mar. Res.* 59, 151–162. <https://doi.org/10.1007/s10152-004-0214-2>
- 567 Hazelton, E.L.G., Mozdzer, T.J., Burdick, D.M., Kettenring, K.M., Whigham, D.F., 2014.
568 *Phragmites australis* management in the United States: 40 years of methods and outcomes. *AoB*
569 *Plants* 6, plu001. <https://doi.org/10.1093/aobpla/plu001>
- 570 Jordan, T.E., Valiela, I., 1982. A nitrogen budget of the ribbed mussel, *Geukensia demissa*, and
571 its significance in nitrogen flow in a New England salt marsh. *Limnol. Oceanogr.* 27, 75–90.
572 <https://doi.org/10.4319/lo.1982.27.1.0075>

- 573 Kemp, A.C., Horton, B.P., Donnelly, J.P., Mann, M.E., Vermeer, M., Rahmstorf, S., 2011.
574 Climate related sea-level variations over the past two millennia. *Proc. Natl. Acad. Sci. U. S. A.*
575 108, 11017–22. <https://doi.org/10.1073/pnas.1015619108>
- 576 Kinney, E., Valiela, I., 2013. Changes in $\delta^{15}\text{N}$ in salt marsh sediments in a long-term fertilization
577 study. *Mar. Ecol. Prog. Ser.* 477, 41–52. <https://doi.org/10.3354/meps10147>
- 578 Kirwan, M.L., Guntenspergen, G.R., 2012. Feedbacks between inundation, root production, and
579 shoot growth in a rapidly submerging brackish marsh. *J. Ecol.* 100, 764–770.
580 <https://doi.org/10.1111/j.1365-2745.2012.01957.x>
- 581 Kirwan, M.L., Guntenspergen, G.R., 2010. Influence of tidal range on the stability of coastal
582 marshland. *J. Geophys. Res. Earth Surf.* 115, F02009. <https://doi.org/10.1029/2009jf001400>
- 583 Kirwan, M.L., Guntenspergen, G.R., D’Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S.,
584 2010. Limits on the adaptability of coastal marshes to rising sea level. *Geophys. Res. Lett.* 37,
585 L23401. <https://doi.org/10.1029/2010GL045489>
- 586 Kirwan, M.L., Mudd, S.M., 2012. Response of salt-marsh carbon accumulation to climate
587 change. *Nature* 489, 550–553. <https://doi.org/10.1038/nature11440>
- 588 Kirwan, M.L., Temmerman, S., Skeeahan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016.
589 Overestimation of marsh vulnerability to sea level rise. *Nat. Clim. Chang.* 6, 253–260.
590 <https://doi.org/10.1038/nclimate2909>
- 591 Kolker, A.S., Goodbred, S.L., Hameed, S., Cochran, J.K., 2009. High-resolution records of the
592 response of coastal wetland systems to long-term and short-term sea-level variability. *Estuar.*
593 *Coast. Shelf Sci.* 84, 493–508. <https://doi.org/10.1016/j.ecss.2009.06.030>
- 594 Kopp, R.E., Kemp, A.C., Bittermann, K., Horton, B.P., Donnelly, J.P., Gehrels, W.R., Hay, C.C.,
595 Mitrovica, J.X., Morrow, E.D., Rahmstorf, S., 2016. Temperature-driven global sea-level

- 596 variability in the Common Era. *Proc. Natl. Acad. Sci.* 113, 1–8.
- 597 <https://doi.org/10.1073/pnas.1517056113>
- 598 Krebs, C.T., Valiela, I., 1978. Effect of experimentally applied chlorinated hydrocarbons on the
599 biomass of the fiddler crab, *Uca pugnax* (Smith). *Estuar. Coast. Mar. Sci.* 6, 375–386.
- 600 [https://doi.org/10.1016/0302-3524\(78\)90129-9](https://doi.org/10.1016/0302-3524(78)90129-9)
- 601 Merrifield, M.A., Merrifield, S.T., Mitchum, G.T., 2009. An anomalous recent acceleration of
602 global sea level rise. *J. Clim.* 22, 5772–5781. <https://doi.org/10.1175/2009JCLI2985.1>
- 603 Miller, W.D., Neubauer, S.C., Anderson, I.C., 2001. Effects of sea level induced disturbances on
604 high salt marsh metabolism. *Estuaries* 24, 357. <https://doi.org/10.2307/1353238>
- 605 Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of
606 coastal wetlands to rising sea level. *Ecology* 83, 2869. <https://doi.org/10.2307/3072022>
- 607 Nerem, R.S., Beckley, B.D., Fasullo, J.T., Hamlington, B.D., Masters, D., Mitchum, G.T., 2018.
608 Climate-change–driven accelerated sea-level rise detected in the altimeter era. *Proc. Natl. Acad.*
609 *Sci.* 115, 201717312. <https://doi.org/10.1073/pnas.1717312115>
- 610 Nixon, S.W., 1982. *The Ecology of New England High Salt marshes: a Community Profile*. U.S.
611 Fish and Wildlife Service, Office of Biological Services, Washington DC.
- 612 <https://doi.org/FWS/OBS-81/55>
- 613 Ouyang, X., Lee, S.Y., 2014. Updated estimates of carbon accumulation rates in coastal marsh
614 sediments. *Biogeosciences* 11, 5057–5071. <https://doi.org/10.5194/bg-11-5057-2014>
- 615 Raposa, K.B., Cole Ekberg, M.L., Burdick, D.M., Ernst, N.T., Adamowicz, S.C., 2016.
616 Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise.
617 *Reg. Environ. Chang.* 1–9. <https://doi.org/10.1007/s10113-016-1020-5>

- 618 Raposa, K.B., Weber, R.L.J., Ekberg, M.C., Ferguson, W., 2017. Vegetation dynamics in Rhode
619 Island salt marshes during a period of accelerating sea level rise and extreme sea level events.
620 *Estuaries and Coasts* 40, 640–650. <https://doi.org/10.1007/s12237-015-0018-4>
- 621 Redfield, A.C., 1972. Development of a New England salt marsh. *Ecol. Monogr.* 42, 201–237.
622 <https://doi.org/10.2307/1942263>
- 623 Rietsma, C.S., Monteiro, R.O., Valiela, I., 2011. Plant cover, herbivory, and resiliency in a Cape
624 Cod salt marsh: Multi-year responses and recovery following manipulation of nutrients and
625 competition. *Estuaries and Coasts* 34, 198–210. <https://doi.org/10.1007/s12237-010-9337-7>
- 626 Rodríguez, J.F., Saco, P.M., Sandi, S., Saintilan, N., Riccardi, G., 2017. Potential increase in
627 coastal wetland vulnerability to sea-level rise suggested by considering hydrodynamic
628 attenuation effects. *Nat. Commun.* 8. <https://doi.org/10.1038/ncomms16094>
- 629 Rush, S.A., Soehren, E.C., Woodrey, M.S., Graydon, C.L., Cooper, R.J., 2009. Occupancy of
630 select marsh birds within northern Gulf of Mexico tidal marsh: Current estimates and projected
631 change. *Wetlands* 29, 798–808. <https://doi.org/10.1672/08-174.1>
- 632 Sá, R., Bexiga, C., Veiga, P., Vieira, L., Erzini, K., 2006. Feeding ecology and trophic
633 relationships of fish species in the lower Guadiana River Estuary and Castro Marim e Vila Real
634 de Santo António Salt Marsh. *Estuar. Coast. Shelf Sci.* 70, 19–26.
635 <https://doi.org/10.1016/j.ecss.2006.05.038>
- 636 Sallenger, A.H., Doran, K.S., Howd, P.A., 2012. Hotspot of accelerated sea-level rise on the
637 Atlantic coast of North America. *Nat. Clim. Chang.* 2, 884–888.
- 638 Sandi, S.G., Rodríguez, J.F., Saintilan, N., Riccardi, G., Saco, P.M., 2018. Rising tides, rising
639 gates: The complex ecogeomorphic response of coastal wetlands to sea-level rise and human

- 640 interventions. *Adv. Water Resour.* 114, 135–148.
641 <https://doi.org/10.1016/j.advwatres.2018.02.006>
- 642 Sarda, R., Foreman, K., Valiela, I., 1995. Macroinfauna of a Southern New England salt marsh:
643 Seasonal dynamics and production. *Mar. Biol.* 121, 431–445.
644 <https://doi.org/10.1007/BF00349452>
- 645 Smith, J.A.M., 2013. The role of *Phragmites australis* in mediating inland salt marsh migration
646 in a Mid-Atlantic estuary. *PLoS One* 8, e65091. <https://doi.org/10.1371/journal.pone.0065091>
- 647 Smith, S.M., 2014. Vegetation change in salt marshes of Cape Cod National Seashore
648 (Massachusetts, USA) between 1984 and 2013. *Wetlands* 35, 127–136.
649 <https://doi.org/10.1007/s13157-014-0601-7>
- 650 Tabak, N.M., Laba, M., Spector, S., 2016. Simulating the effects of sea level rise on the
651 resilience and migration of tidal wetlands along the Hudson River. *PLoS One* 11, e0152437.
652 <https://doi.org/10.1371/journal.pone.0152437>
- 653 Treggor, J., 1983. The Development and Geomorphology of the Great Sippewissett Marsh.
654 Masters Thesis. Central Connecticut State University.
- 655 US Fish and Wildlife Service, 2012. Application of the Sea-Level Affecting Marshes Model
656 (SLAMM 6) to Swanquarter National Wildlife Refuge.
- 657 Valiela, I., 2015. The Great Sippewissett salt marsh plots—Some history, highlights, and
658 contrails from a long-term study. *Estuaries and Coasts* 38, 1099–1120.
659 <https://doi.org/10.1007/s12237-015-9976-9>
- 660 Valiela, I., 2006. *Global Coastal Change*. Blackwell Publishing, 368 pp.
- 661 Valiela, I., 1983. Nitrogen in salt marsh ecosystems, in: Carpenter, E.J., Capone, D.G. (Eds.),
662 *Nitrogen in the Marine Environment*. Academic Press, pp. 649–678.

- 663 Valiela, I., Rietsma, C.S., 1995. Disturbance of salt marsh vegetation by wrack mats in Great
664 Sippewissett Marsh. *Oecologia* 102, 106–112. <https://doi.org/10.1007/BF00333317>
- 665 Valiela, I., Teal, J.M., 1979. The nitrogen budget of a salt marsh ecosystem. *Nature* 280, 652–
666 656. <https://doi.org/10.1038/280652a0>
- 667 Vandenbruwaene, W., Maris, T., Cox, T.J.S., Cahoon, D.R., Meire, P., Temmerman, S., 2011.
668 Sedimentation and response to sea-level rise of a restored marsh with reduced tidal exchange:
669 Comparison with a natural tidal marsh. *Geomorphology* 130, 115–126.
670 <https://doi.org/10.1016/j.geomorph.2011.03.004>
- 671 Vince, S., Valiela, I., Backus, N., Teal, J.M., 1976. Predation by the salt marsh killifish *Fundulus*
672 *heteroclitus* (L.) in relation to prey size and habitat structure: Consequences for prey distribution
673 and abundance. *J. Exp. Mar. Bio. Ecol.* 23, 255–266. [https://doi.org/10.1016/0022-](https://doi.org/10.1016/0022-0981(76)90024-1)
674 [0981\(76\)90024-1](https://doi.org/10.1016/0022-0981(76)90024-1)
- 675 Wang, J.Q., Zhang, X.D., Jiang, L.F., Bertness, M.D., Fang, C.M., Chen, J.K., Hara, T., Li, B.,
676 2010. Bioturbation of burrowing crabs promotes sediment turnover and carbon and nitrogen
677 movements in an estuarine salt marsh. *Ecosystems* 13, 586–599. [https://doi.org/10.1007/s10021-](https://doi.org/10.1007/s10021-010-9342-5)
678 [010-9342-5](https://doi.org/10.1007/s10021-010-9342-5)
- 679 Watson, E.B., Raposa, K.B., Carey, J.C., Wigand, C., Warren, R.S., 2017. Anthropocene
680 survival of southern New England’s salt marshes. *Estuaries and Coasts* 40, 617–625.
681 <https://doi.org/10.1007/s12237-016-0166-1>
- 682 Watson, E.B., Szura, K., Wigand, C., Raposa, K.B., Blount, K., Cencer, M., 2016. Sea level rise,
683 drought and the decline of *Spartina patens* in New England marshes. *Biol. Conserv.* 196, 173–
684 181. <https://doi.org/10.1016/j.biocon.2016.02.011>

685 Werne, C.E., 1981. Resource Partitioning in a Salt Marsh Fish Community. Ph.D. Thesis.

686 Boston University.

687

688

689

690 **Figure legends**

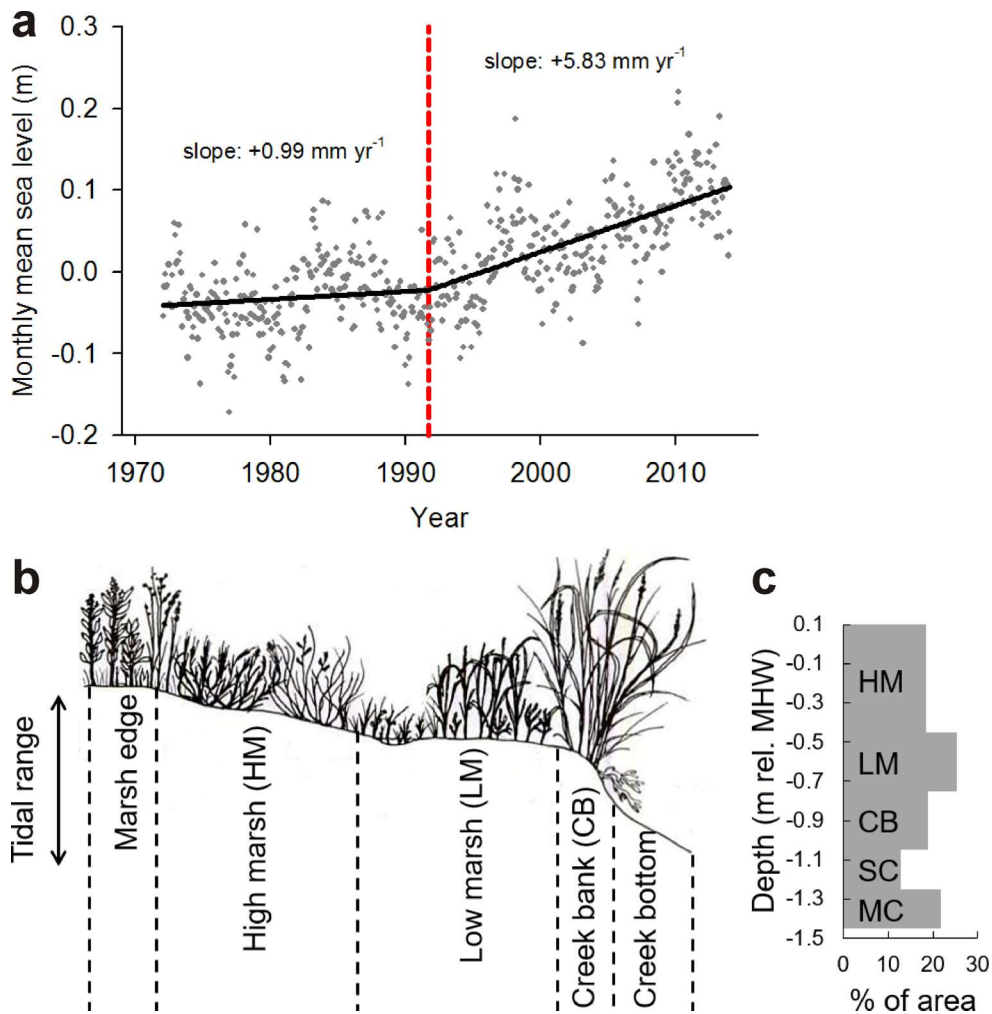
691 **Fig. 1** Recent rates of sea level rise and habitat distribution across the elevation gradient. **a)**
 692 Record of sea level rise, 1972-2013, for the Woods Hole, Cape Cod, MA US, station. Plotted
 693 values are sea levels relative to NAVD88. Dashed red line represents the hinge year of 1992,
 694 when slopes changed significantly according to the piecewise regression analysis of data. **b)**
 695 Diagrammatic section through a New England salt marsh, from the upper marsh edge (on the
 696 left) to the tidal creek bottom (on the right); figure adapted from Valiela (2015). In sequence,
 697 from left to right: the upper tidal margin of the marsh is defined by a narrow band of marsh elder,
 698 *Iva frutescens*, and black rush, *Juncus maritima*; the high marsh platform is covered by marsh
 699 hay, *S. patens*, with some spikegrass, *Distichlis spicata*; intermediate and short forms of *S.*
 700 *alterniflora* grow on the low marsh platform, with the glasswort, *Salicornia europaea* growing in
 701 temporary open gaps; tall form of cordgrass, *S. alterniflora* occurs on creek banks—with a
 702 limited understory of brown algae; at the lower limit of the tidal range, there are un-vegetated
 703 bare sediments. **c)** Elevation range for high marsh, low marsh, and creek bank vegetation, and for
 704 un-vegetated areas covered by sand (SC) or mud (MC) sediments; from Valiela and Rietsma
 705 (1995).

706 **Fig. 2** Changes in areas of major habitats of Great Sippewissett Marsh between 1979 and 2015.
 707 **a)** Areas \pm s.e. of total marsh and major habitats. **b)**
 708 Percent changes between 2015 and 1979, for the same categories. **c)** Area \pm s.e. of major
 709 vegetation types. **d)** Percent changes between 2015 and 1979, for the same categories, where
 710 blue shows gains, and red shows losses.

711 **Fig. 3** Spatial changes in major habitats of the Great Sippewissett Marsh between 1979 and
712 2015, for **a)** low marsh, **b)** high marsh, **c)** beaches and dunes, and **d)** un-vegetated mud and sand
713 sediments.

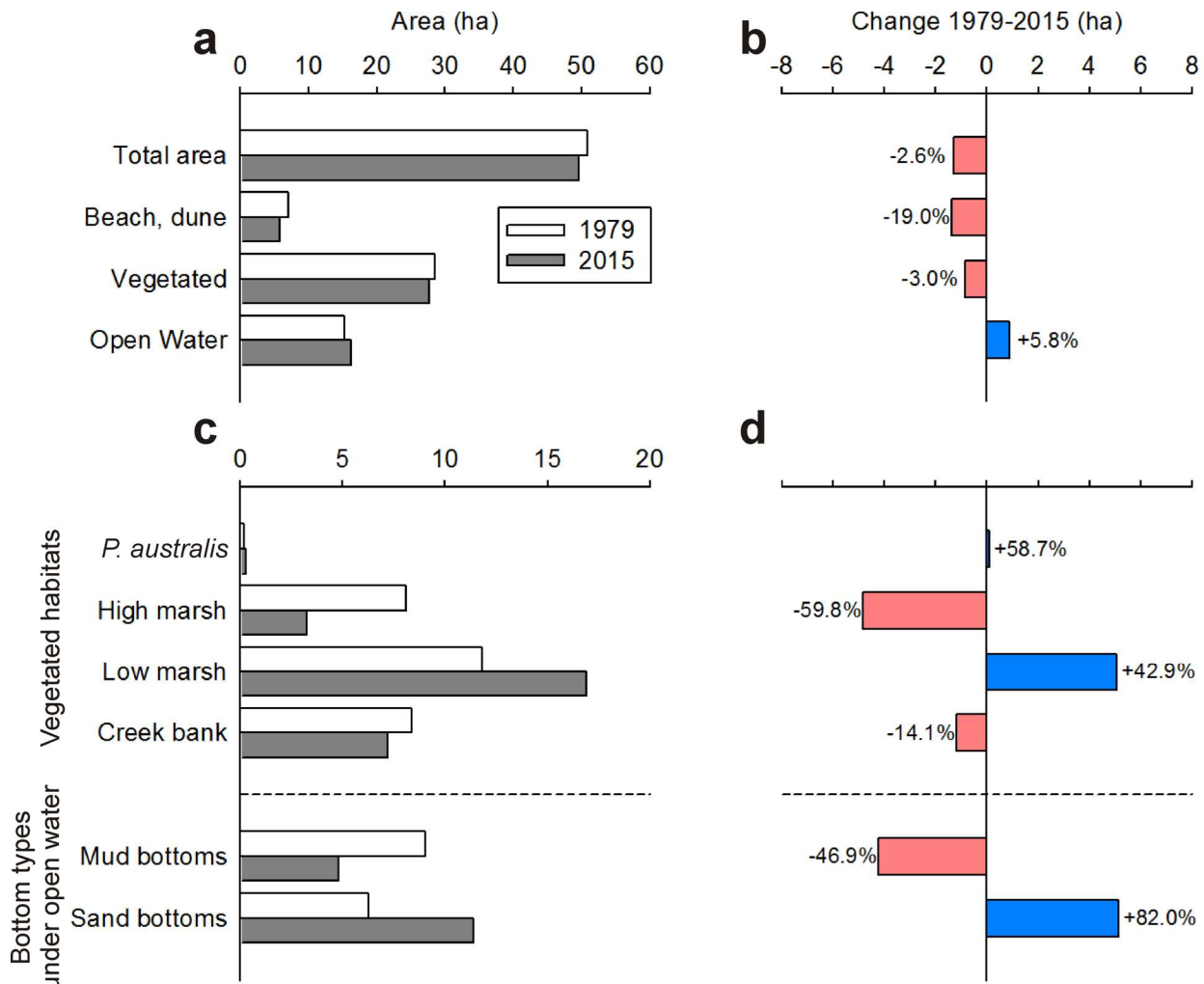
714 **Fig. 4** Modeled trajectories of Great Sippewissett Marsh for **a)** vegetated habitats, and **b)** open
715 water area. Values expressed as percentages of total area of the marsh in 1979. Solid lines
716 represent changes in areas between 1979 and 2015. Dashed lines represent modeled changes in
717 areas into the future. Shaded areas represent the propagated uncertainty associated with the
718 estimated areas of each habitat.

719 **Fig 5** Sediment accretion rates versus mean sea level rise in salt marshes around the world. The
720 graph includes data from 315 locations. Data from compilations in Crosby et al. (2016), Kirwan
721 et al. (2016), Raposa et al. (2016), and Valiela (2006). The black dashes show the 1:1 line
722 indicating agreement between accretion and sea level rise during recent historical times. The
723 grey dashes show the same agreement line, modified to show position in 2100, assuming no rise
724 in accretion, and increased sea level to levels predicted for the end of this century.



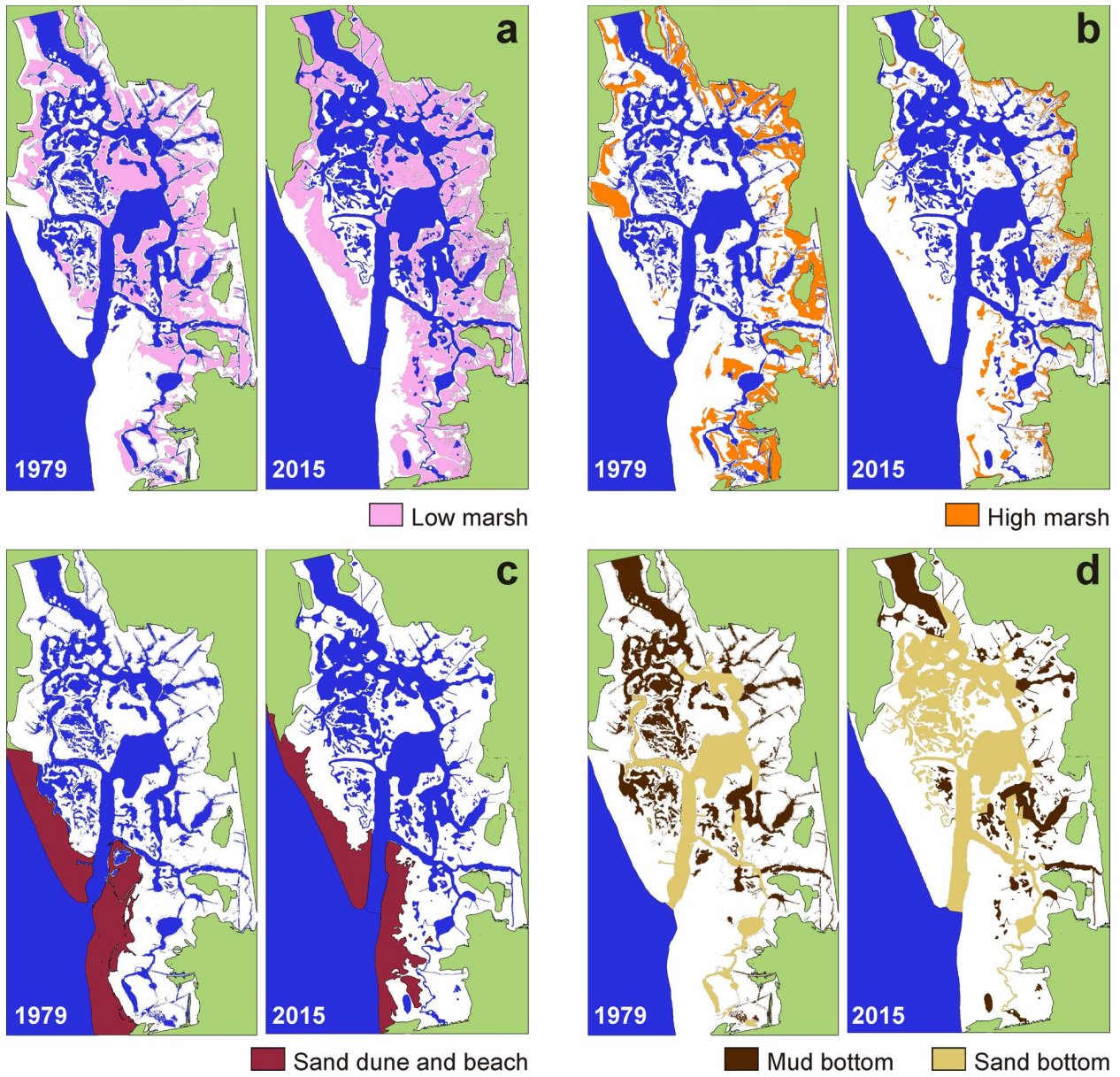
725

726 Fig.1



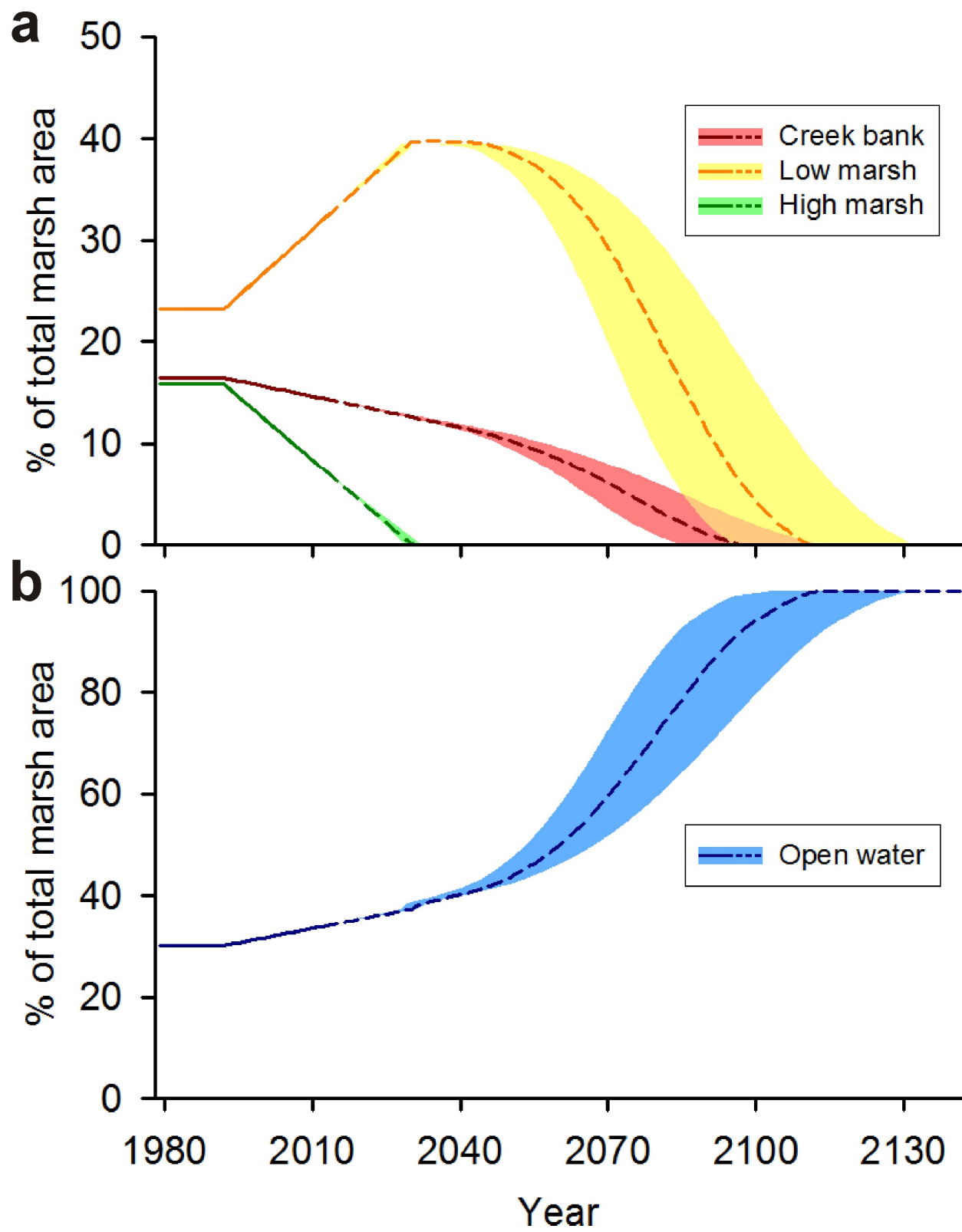
727

728 Fig.2



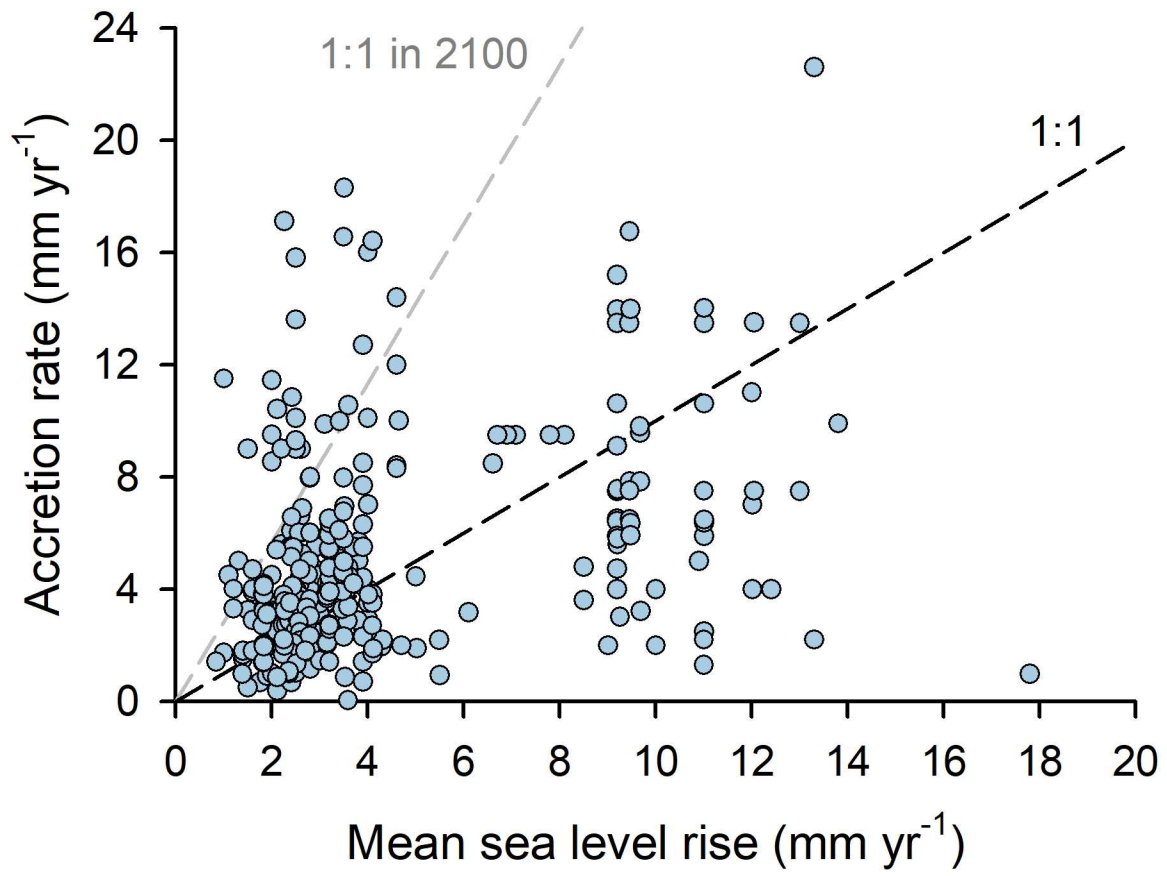
729

730 Fig.3



731

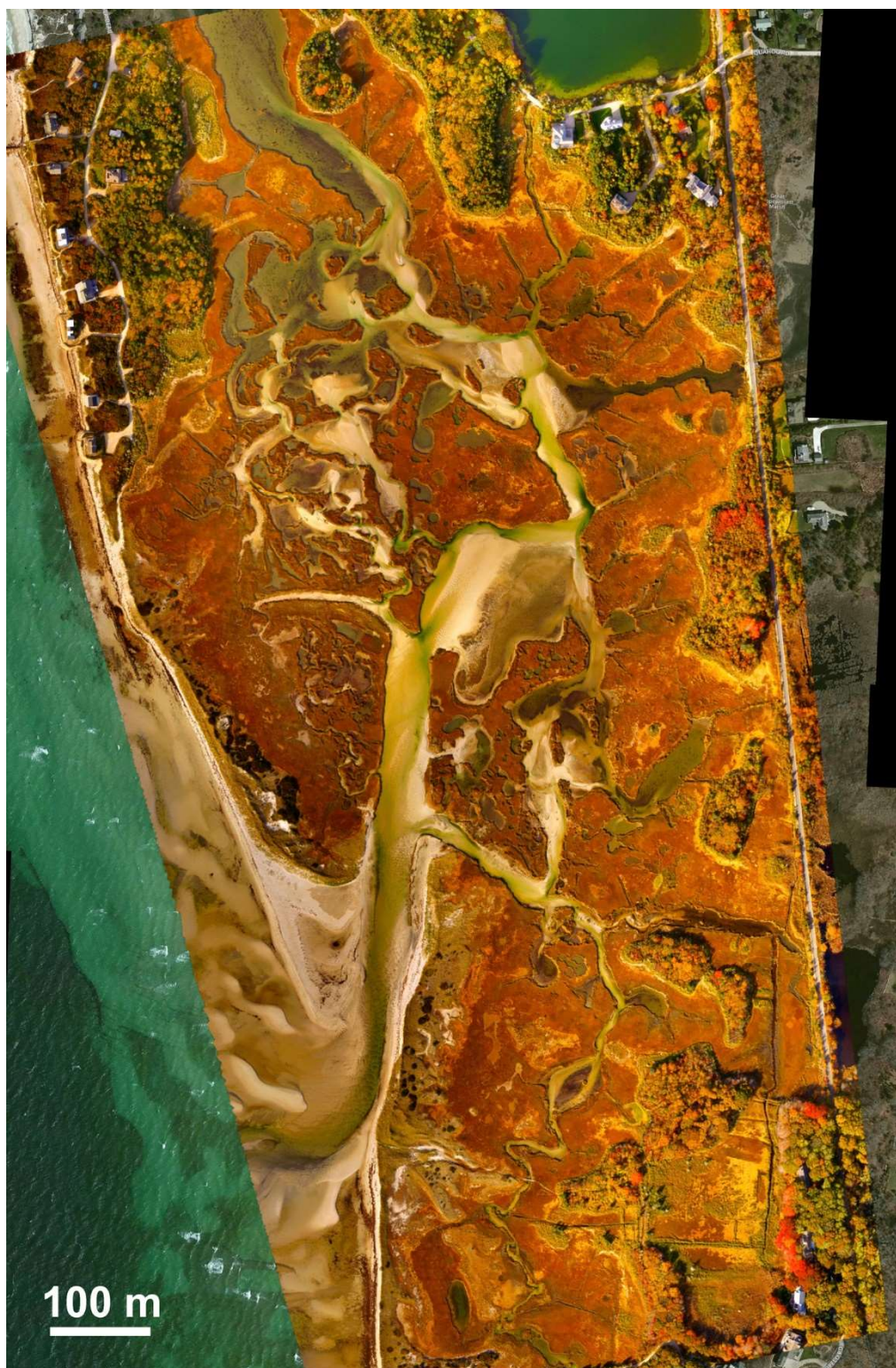
732 Fig.4



733

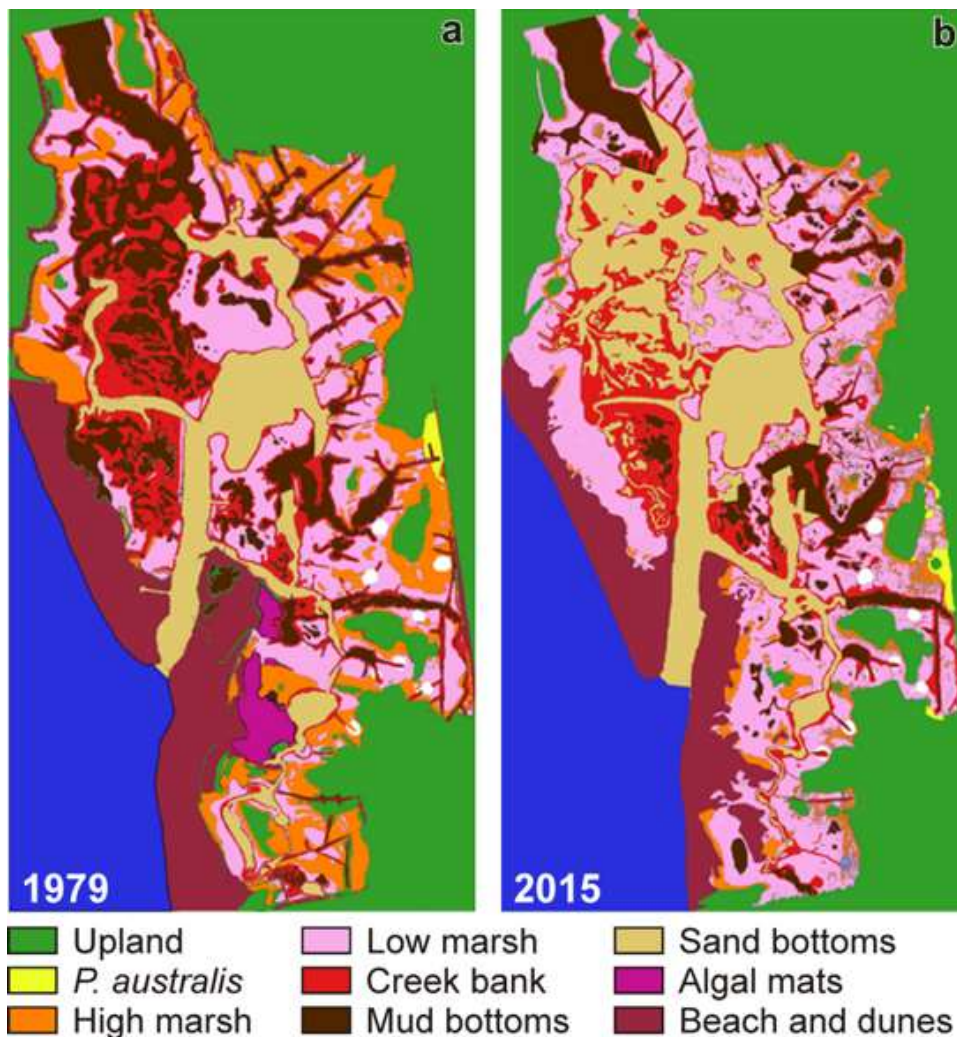
734 Fig.5

735

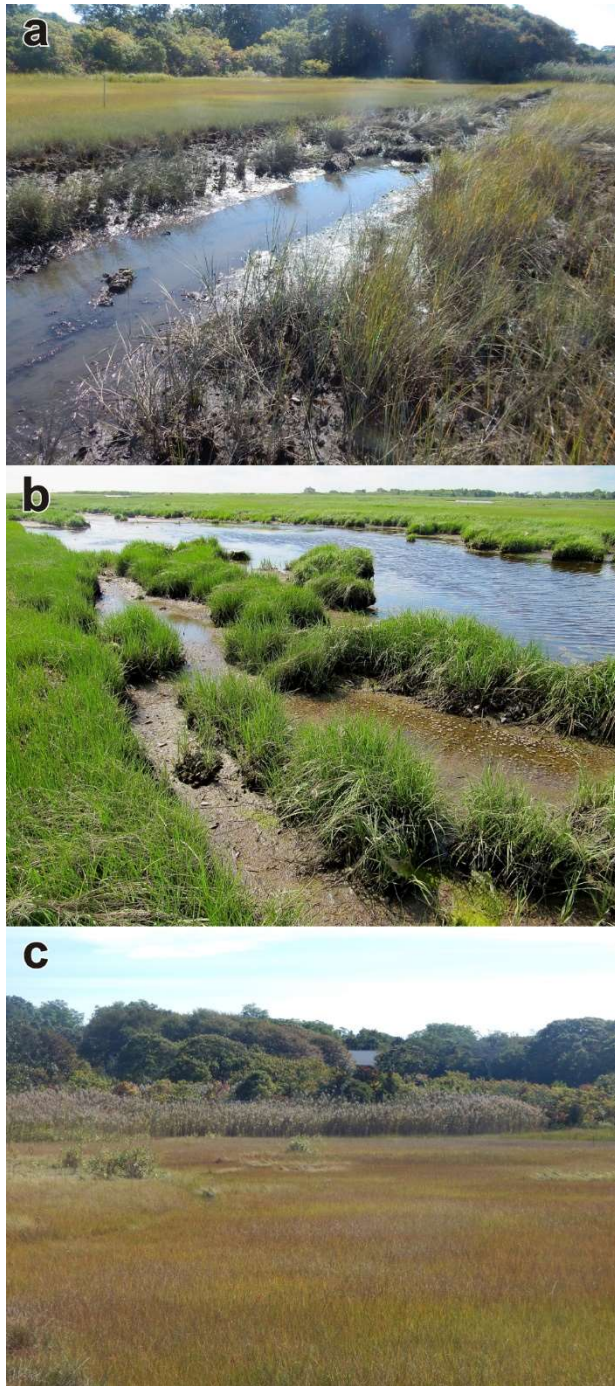
736 **Supplementary Information**

737

738 **Fig. SI 1. Aerial image of Great Sippewissett salt marsh, Cape Cod, MA. The image was compiled by digitally**739 **stitching a series of images obtained by unpiloted aerial vehicle flights done in September 2015.**



740
 741 **Fig. SI 2. Great Sippewissett Marsh habitat maps. a,** Maps of habitat and vegetation mosaic for 1979 generated
 742 from data collected from field surveys. **b,** Maps of habitat and vegetation mosaic for 2015 generated from images
 743 obtained by Unpiloted Aerial Vehicle flights
 744



745
 746 **Fig. SI 3. Some views of Great Sippewissett Marsh.** a) Tidal creek within Great Sippewissett Marsh, 2015,
 747 showing slumping creek banks on either side of creek; the vegetation ranges from tall *S. alterniflora* on the
 748 collapsing creek banks, to shorter *S. alterniflora* on the platform farther away from the creek, and high marsh
 749 species nearer to land. b) Slumping creek banks along a tidal creek in Great Sippewissett Marsh, 2014, after
 750 increased submergence stimulated growth of sparser tall grasses, sediment weakened, bands of marsh platform and
 751 creek banks separated from the platform, and collapsed into the pond, and slowly disappeared. c) View of part of
 752 Great Sippewissett Marsh, looking toward land, 2015, showing short *S. alterniflora* in foreground, giving way to
 753 high marsh vegetation farther away. The upper edge of high marsh vegetation is prevented from landward migration
 754 by the solid phalanx of the much taller *P. australis* new variant.



755
756
757
758
759
760
761
762

Fig. SI 4. Shoreline recession. Detail of Figure 2, showing the area near the inlet of Great Sippewissett marsh, an indicating the retreat in position of the vegetated wetland edge between 1979 (shown as the black line), compared to 2015

763 **Table SI 1.** Values of variables in modeling of Great Sippewissett Marsh future trajectories

Parameter	Description	Value	Equations
<i>Range_{LM}</i>	Amplitude of low marsh elevation range	0.3 m	11, 12
<i>Accr_{LM}</i>	Low marsh accretion rate	2.95 mm yr ⁻¹	11, 12, 15
<i>Accr_{HM}</i>	High marsh accretion rate	2.25 mm yr ⁻¹	15
<i>Col_{LMtoSH}</i>	Low marsh colonization rate of sandy habitats	311 m ² yr ⁻¹	2, 8
<i>Col_{LMtoDH}</i>	Low marsh colonization rate of dune habitats	132 m ² yr ⁻¹	3, 8
<i>Subm_{DH}</i>	Inundation effect on dunes	457 m ² yr ⁻¹	3
<i>Subm_{HM}</i>	Inundation effect on high marsh	2077 m ² yr ⁻¹	4, 7
<i>Col_{LMtoHM}</i>	Low marsh colonization rate of high marsh	2077 m ² yr ⁻¹	5, 8
<i>Col_{PhragtoHM}</i>	<i>P. australis</i> colonization rate of high marsh	27 m ² yr ⁻¹	4, 6
<i>minSubm_{LM}</i>	Minimum rate of Inundation effect on low marsh	313 m ² yr ⁻¹	9, 10
μ	Half-life for the total area of low marsh	52 yr	10, 11
σ	Standard deviation of the distribution of low marsh loss	17 yr	10, 12
<i>Slump_{CB}</i>	Creek bank edge slumping rate	515 m ² yr ⁻¹	13, 14

764

765

766

767

768

769

770 **Table SI 2.** Area of different habitats (m^{-2}) \pm s.e. within Great Sippewissett marsh during 1979
 771 and 2015, and % changes between these years, per habitat and for entire marsh system

Habitat	1979	2015	Difference	% change
Wetland vegetation	284,674 \pm 5,991	276,154 \pm 5,901	-8,520	-3.0
Creek bank ^a	83,747 \pm 3,249	71,900 \pm 3,011	-11,847	-14.1
Low marsh	118,320 \pm 3,862	169,080 \pm 4,617	47,902	+42.9
High marsh	80,910 \pm 3,194	32,509 \pm 2,025	-50,760	-59.8
<i>P. australis</i>	1,697 \pm 463	2,665 \pm 580	968	+57.0
Open water	152,819 \pm 4,389	161,749 \pm 4,516	8,930	+5.8
Mud bottoms	90,292 \pm 3,374	47,930 \pm 2,458	-42,362	-46.9
Sand bottoms	62,527 \pm 2,808	113,819 \pm 3,788	51,292	+82.0
Beach and dunes	71,258 \pm 2,997	57,706 \pm 2,6978	-13,553	-19.0
Total area	508,571 \pm 8,008	495,609 \pm 7,905	-12,962	-2.5

772 ^aThis habitat includes creek banks as well as isolated low-lying islands supporting growth of tall form *S.*
 773 *alterniflora*.

774

775

776 **Table SI 3.** Concentrations (μM) of nutrients in water samples, and number of stations (n)
 777 sampled within Great Sippewissett marsh, 1970-2014

Year	Nitrate		Ammonium		Phosphate	
	Mean \pm se	n	Mean \pm se	n	Mean \pm se	n
1970	0.74 \pm 0.20	20	3.26 \pm 0.64	19	0.94 \pm 0.25	20
1971	2.23 \pm 0.74	6	3.56 \pm 0.16	3	3.33 \pm 2.17	6
1972	1.28 \pm 0.04	2	13.23 \pm 5.07	2	3.56 \pm 3.53	2
1975	2.30 \pm 0.45	16	0.50 \pm 0.10	12	0.72 \pm 0.11	12
1977	0.39 \pm 0.09	13	2.78 \pm 0.29	13	0.60 \pm 0.23	13
1984			10.90 \pm 0.00	1		
2007	1.06 \pm 0.28	28	1.72 \pm 0.21	28	0.96 \pm 0.12	28
2009	0.71 \pm 0.30	13	1.63 \pm 0.46	13	0.45 \pm 0.11	13
2012	2.40 \pm 1.62	6	0.14 \pm 0.03	6	1.28 \pm 0.24	6
2014	3.22 \pm 0.71	15	4.75 \pm 0.59	14	7.89 \pm 0.24	15

778

779

780

781