


Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise

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Abstract Salt marshes persist within the intertidal zone when marsh elevation gains are commensurate with rates of sea-level rise (SLR). Monitoring changes in marsh elevation in concert with tidal water levels is therefore an effective way to determine if salt marshes are keeping pace with SLR over time. Surface elevation tables (SETs) are a common method for collecting precise data on marsh elevation change. Southern New England is a hot spot for SLR, but few SET elevation change datasets are available for the region. Our study synthesizes elevation change data collected from 1999 to 2015 from a network of SET stations throughout Rhode Island (RI). These data are compared to accretion and water level data from the same time period to estimate shallow subsidence and determine whether marshes are tracking SLR. Salt marsh elevation increased at a mean overall rate of 1.40 mm year⁻¹ and

ranged from -0.33 to 3.36 mm year⁻¹ at individual stations. Shallow subsidence dampened elevation gain in mid-Narragansett Bay marshes, but in other areas of coastal RI, subsurface processes may augment surface accretion. In all cases, marsh elevation gain was exceeded by the 5.26 mm year⁻¹ rate of increase in sea levels during the study period. Our study provides the first SET elevation change data from RI and shows that most RI marshes are not keeping pace with short- or long-term rates of SLR. It also lends support to previous research that implicates SLR as a primary driver of recent changes to southern New England salt marshes.

Keywords New England · Resilience · Surface elevation table · Accretion

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Introduction

The ability of salt marshes to sequester carbon, provide habitat for fish and wildlife, and protect shorelines from flooding and storms, among other ecosystem services, (Valiela and Cole 2002; McKinney and Wigand 2006; Shepard et al. 2011; Kirwan and Mudd 2012) is increasingly threatened by accelerating global sea-level rise (SLR) (Craft et al. 2009; Nicholls and Cazenave 2010). Marshes in southern New England (USA) are particularly at risk due to the confluence of multiple factors including low sediment supplies and tidal ranges (Weston 2014) and regional rates of SLR that are among the highest in the world (Sallenger et al. 2012). The problem appears particularly acute in Rhode Island (RI), where SLR is now assumed to be the primary driver of accelerating rates of high marsh vegetation loss and the proliferation of marsh surface vegetation dieback and ponding (Donnelly and Bertness 2001; Raposa et al. 2015; Cole Ekberg et al. 2015). Additional research demonstrates that the vast majority of RI and other southern New England salt marshes are already too low in elevation to avoid further dieback and reductions in plant growth as sea levels continue to rise (Watson et al. 2014a). These studies consistently identify SLR as a driver of ongoing changes to RI marshes and potential marsh loss, but data that actually quantify overall marsh elevation change have not been reported for RI. This is a critical gap; quantifying rates of marsh elevation change relative to changes in local tidal water levels is an essential step in quantifying marsh vulnerability to SLR (Cahoon and Guntenspergen 2010). Determining whether RI marshes are keeping pace with SLR will in turn provide improved clarity as to whether or not SLR is a key driver of the changes to RI marshes that are currently being reported.

Previous studies in RI and nearby Connecticut using radionuclide dating of peat cores show that in some cases marsh accretion rates can be commensurate with long-term rates of SLR (Orson et al. 1998; Bricker-Urso et al. 1989). However, accretion rates are only one component of multiple processes that interactively affect marsh elevation change; accretion can actually be considerably higher than elevation gain due to subsidence and compaction of subsurface peat and may not reflect true changes in marsh elevation (Cahoon et al. 1995). One approach for quantifying precise rates of marsh elevation change over time is with surface elevation tables (SETs; Cahoon et al. 2002a, b). Elevation change data from SETs can also be coupled with short-term accretion rate data derived from marker horizons (MH) that are generally placed in conjunction with SET stations to inform processes affecting overall elevation change (SET–MH stations are hereafter referred to as SET stations). Although a growing number of

SET stations now exist throughout New England, it takes years to build elevation change datasets that can account for sometimes high intra- and inter-annual variability and reveal accurate trends in marsh elevation change over time. Consequently, only a limited number of elevation change estimates from SETs are currently available from New England marshes (e.g., Erwin et al. 2006; Anisfeld and Hill 2012; Carey et al. 2015a; Burdick and Peter 2015; Anisfeld et al. 2016); very few datasets are available for the southern New England subregion, and, until now, no SET elevation change data were available from any RI marshes.

There is clearly a need to compile and synthesize elevation change (and short-term accretion) data from existing SET stations in southern New England to document whether marshes are keeping pace with local rates of SLR and to provide further insight into the drivers of ongoing marsh change and loss. The goals of this study, therefore, are to (1) compile data from all existing and active SET stations in RI, (2) calculate rates of marsh elevation change and accretion, and (3) compare these rates to concomitant changes in sea levels derived from a tide station in Narragansett Bay, RI. Our study provides the first elevation change data for RI, expands the spatial extent of SET-derived marsh elevation change and accretion datasets in southern New England, and provides further insight into the dynamics of the region's salt marshes during the current period of accelerating SLR.

Materials and methods

Elevation change and accretion data were compiled from 49 existing and active SET stations in RI (Table 1). These stations are located in Narragansett Bay proper (three marshes on Prudence Island; 16 SETs), the adjoining Sakonnet River (two marsh sections; six SETs); Rhode Island Sound (one marsh; three SETs); the southern coastal ponds (three marsh sections; nine SETs) and in the Narrow River Estuary (five marsh sections; 15 SETs) (Fig. 1). Installation dates ranged from 1999 to 2015, although only a small minority of stations (10, or 29 %) were installed prior to 2004 and have datasets that span a period of 10 years or more. Most SETs have been installed in marsh platform habitats that include classic *Spartina patens*/*Distichlis spicata*-dominated salt meadow, short-form *Spartina alterniflora*, and areas where these two habitats currently mix. A small number of SETs have also been installed in bare, unvegetated areas on the marsh platform, but these are specific to marsh restoration/adaptation projects.

SETs have been installed in RI by four primary organizations to either monitor temporal trends in reference

Table 1 Descriptions of all active surface elevation tables (SETs) in Rhode Island salt marshes

	Salt marsh	Region	No. of SETs	Installation year	Organization	Habitat type (s)	Sampling years
Reference	Coggeshall, PI	Narragansett Bay	1 (1)	1999	UNH	Sm	1999–2000, 2014–2015
			6 (5)	2012	NBNERR	Sm; Sa-s	2012–2015
	Nag West, PI	Narragansett Bay	2 (2)	1999	UNH	Sm	1999–2000, 2014–2015
			6 (5)	2012	NBNERR	Sm; Sa-s	2012–2015
	Nag Pond, PI	Narragansett Bay	1 (1)	1999	UNH	Sm	1999–2000, 2014–2015
	Sachuest Point	Sakonnet River	3 (3)	2003	STB/USFWS	Mix	2003–2007; 2009–2014
Managed	John H. Chafee	Narrow River	15 (7)	2011; 2013; 2014	USFWS	Sm; Sa-s; Mix	Variable
	Ninigret	Coastal Ponds	6 (0)	2013; 2014	USFWS	Sm	Variable
	Sachuest Point	Sakonnet River	3 (0)	2011	USFWS	Mix	2011–2015
	Gooseneck Cove	Rhode Island Sound	3 (0)	2003	STB	Ba; sparse Sa-s	2003–2007; 2009–2014
	Ninigret	Coastal Ponds	3 (0)	2015	STB/NBNERR	Ba; Sa-s	2015

The Coggeshall, Nag West, and Nag Pond marshes are all located on Prudence Island (PI). For each marsh, the total number of active SETs is shown along with the number of SETs used in our analyses in parentheses. See manuscript text for agency and organization names. For habitat type, Sm = salt meadow, Sa-s = short-form *Spartina alterniflora*, Ba = bare; Mix = mixture of *S. alterniflora* and salt meadow. In the John H. Chafee Marsh in the Narrow River, seven SETs were installed in 2011 and read most years from 2011 to 2015. Five more were installed in 2013 and read in 2014 and 2015; three more were installed in 2014 and read in 2015. In Ninigret reference marshes, three SETs were installed in 2013 and read from 2014 to 2105; three more were installed in 2014 and read in 2015

marshes or assess changes due to different types of restoration and adaptation projects (Table 1). The University of New Hampshire (UNH) installed RI's first six SETs in salt meadow habitats in three Narragansett Bay National Estuarine Research Reserve (NBNERR) marshes on Prudence Island in 1999 (four were recovered and used in this study). These were augmented when the NBNERR installed an additional 12 SETs in two of these same marshes in 2012, equally divided between short-form *S. alterniflora* and salt meadow habitats. Save The Bay (STB) installed three SETs in the Gooseneck Cove hydrologic restoration marsh and three in the nearby Sachuest Point reference marsh in 2003. The US Fish and Wildlife Service (USFWS) later added three additional SETs to a restoration section of the Sachuest Point Marsh in 2011. The USFWS also installed 15 SETs in multiple sections of the John H. Chafee (JHC) marsh complex in the Narrow River from 2011 through 2014 and six more in reference salt marshes within Ninigret Pond along RI's southern shore as part of its long-term monitoring program. Most recently, the NBNERR and STB installed three additional SETs in a nearby Ninigret Pond marsh that is slated to receive sediment slurry to build elevation capital and marsh resilience against SLR.

In this study, we used a subset of all available SET stations to calculate rates of elevation change and accretion. For inclusion in our analyses, SET stations had to be located in marshes not impacted by any major management activities (i.e., reference marshes), on the vegetated marsh platform (e.g., not in low marsh or other habitats such as ditches), in polyhaline areas (defined by

a lack of brackish marsh vegetation species such as *Scirpus* spp. and *Typha* spp.), and had to have datasets spanning at least three years, not including baseline installation readings. Using these criteria, we included 24 SET stations in our analyses (Online Resource 1). Excluded were all nine SET stations that were located in managed marshes, two NBNERR SETs that were located in a vegetated drainage ditch and in a brackish area, and eight USFWS SETs in JHC and six in Ninigret Pond with datasets spanning less than 3 years.

It is important to note that all study marshes containing the SETs used in our analyses are generally similar based on vegetation and habitat composition and on their proximity to major sediment sources. Each of the marshes is dominated by *S. alterniflora*, *S. patens*, *D. spicata*, and *Juncus gerardii*, although the relative proportion of each species varies among sites. All sites also contain a small number of minor tidal creeks and/or marsh surface pools and have been subject to a similar degree of historic grid-ditching. None of the SETs included in our study, however, were located within or along the direct edges of any of these hydrologic features that potentially receive elevated sediment inputs. At a broader scale, none of our study marshes were located in close proximity to any large rivers or other major sediment sources such as the Blackstone River in upper Narragansett Bay (Fig. 1), including the marshes in the Narrow River and Sakonnet River (contrary to their colloquial names, the former is an example of a fjord-type estuary and the latter is a subembayment of Narragansett Bay proper).



Fig. 1 Map of Rhode Island showing locations of marsh systems with surface elevation table and marker horizon (SET–MH) stations installed. Refer to Table 1 for descriptions of SET–MH stations in

each area. General regions are *italicized*; study marshes are in **bold**. Major rivers, representing potential sediment sources to marshes, are also shown for reference

Elevation change and accretion data were collected at all SET stations following the methods and protocols described by Lynch et al. (2015). The older UNH SETs were read using the original version of the SET (Cahoon et al. 2002a); all other stations were read using the newer deep-river SET (Cahoon et al. 2002b). Rates of marsh elevation change at each SET station were calculated by regressing

mean pin heights (averaged across all nine pins within each of the four cardinal directions, and then averaged across the four directions on each sampling date) over time. Similarly, accretion rates were calculated by regressing mean accumulated soil depth over the MH (averaged across all replicate soil plugs within each of the four cardinal directions, and then averaged across the four directions on each

sampling date) over time. Finally, SLR rates were calculated with linear regression using monthly mean sea-level data from the National Water Level Observation Network (NWLON) Newport tide station in Narragansett Bay, RI (station ID# 8452660). Rates of SLR were calculated over the long-term (1930–2015) and within the time frame of all RI SET measurements used in this study (1999–2015). All monthly tide station data were downloaded from www.tidesandcurrents.com.

Results

The rate of SLR from 1930 to 2015 was $2.78 \text{ mm year}^{-1}$, but this accelerated to $5.26 \text{ mm year}^{-1}$ during the period when SET measurements were taken in RI (1999–2015) (Fig. 2). Marsh elevation change rates varied somewhat among individual SET stations and among coastal regions in RI, but our SET data show that RI salt marshes are generally not keeping pace with either the short- or long-term rate of SLR. Rates from all 24 SET stations were well below the short-term SLR rate, and rates from 21 stations (88 %) were also below the long-term SLR rate. Using data from all 24 SETs, RI marshes gained elevation at a mean rate of $1.40 \text{ mm year}^{-1}$. Rates were lowest at the Prudence Island marshes (mean $1.12 \text{ mm year}^{-1}$; range -0.33 to $2.49 \text{ mm year}^{-1}$), followed closely by the JHC marshes (mean $1.75 \text{ mm year}^{-1}$; range -0.22 to $3.36 \text{ mm year}^{-1}$) and Sachuest Point (mean $1.89 \text{ mm year}^{-1}$; range 1.31 – $2.85 \text{ mm year}^{-1}$) (Fig. 3). Changes in elevation in RI marshes were very similar based on short (4–5 years) and longer-term (11–16 years) SET datasets (mean 1.41 and $1.38 \text{ mm year}^{-1}$, respectively).

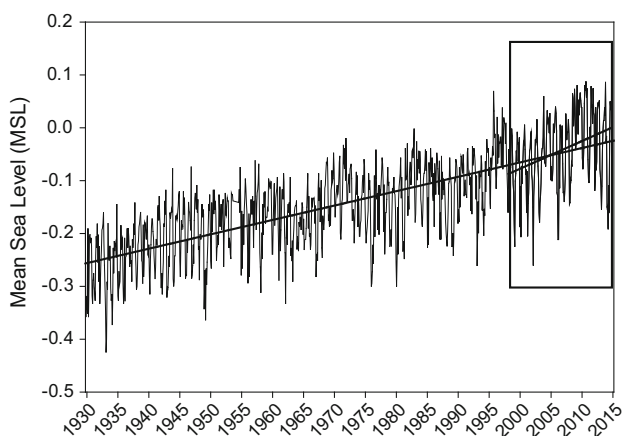


Fig. 2 Mean sea level (MSL) from the Newport RI NWLON tide station (ID# 8452660), with regression lines for long-term and short-term rates of sea-level rise (SLR). The long-term rate of SLR based on monthly MSL is $2.78 \text{ mm year}^{-1}$ (1930–2015); the short-term rate of SLR increased to $5.26 \text{ mm year}^{-1}$ (1999–2015; the years covering SET readings in our study)

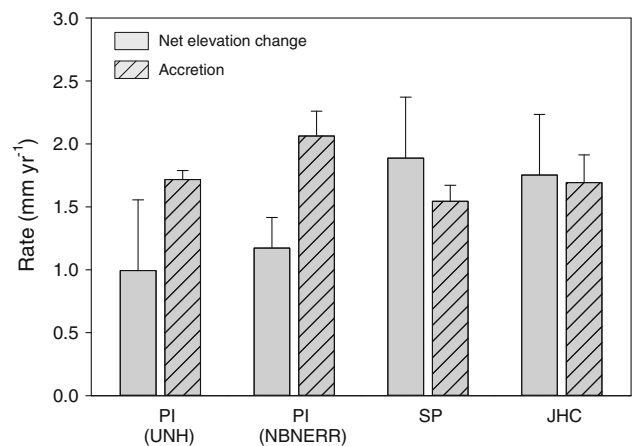


Fig. 3 Rates of elevation change and accretion over time in Rhode Island salt marshes. All data are shown as means ($+1$ SE) across all individual SET stations within each marsh or marsh complex. For individual stations, rates of change were calculated by regressing mean pin height (SETs) and soil depth (MH) on each reading date over time. Prudence Island (PI) UNH data span 1999–2015 ($n = 4$ SET–MH stations); Prudence Island NBNERR data span 2012–2015 ($n = 10$); Sachuest Point (SP) data span 2004–2014 ($n = 3$); John H. Chafee (JHC) data span 2011–2015 ($n = 7$)

Short-term accretion rates averaged $1.83 \text{ mm year}^{-1}$ across all 24 stations. Accretion rates were lowest at Sachuest Point (mean $1.54 \text{ mm year}^{-1}$; range 1.31 – $1.65 \text{ mm year}^{-1}$), followed closely by the JHC marshes (mean $1.69 \text{ mm year}^{-1}$; range 0.83 – $2.73 \text{ mm year}^{-1}$) and the Prudence Island marshes (mean $1.96 \text{ mm year}^{-1}$; range 0.69 – $2.64 \text{ mm year}^{-1}$). At the Prudence Island marshes, accretion was higher on average than elevation gain, showing a mean shallow subsidence rate of $0.84 \text{ mm year}^{-1}$; the opposite was true for Sachuest Point and the JHC marshes where shallow subsurface processes appear to be augmenting surface accretion (Fig. 3). There were no apparent patterns in the relationship between accretion and elevation change rates at the scale of individual stations; accretion rates were higher than elevation gain rates at 9 stations (38 %) and lower at 15 stations (62 %).

Discussion and conclusions

Our study provides the first data on marsh elevation change from SET stations in RI. Rates from all stations were consistently low (generally less than 2 mm year^{-1}) and were far below the rate of SLR during the same time period, thereby demonstrating a growing elevation deficit (Cahoon et al. 1995) between marsh elevations and tidal water levels that jeopardizes long-term marsh survival. Our elevation change data also support the growing notion that SLR is a primary driver of the recent and dramatic changes that are being observed in southern New

England salt marshes (Carey et al. 2015a; Raposa et al. 2015; Watson et al. 2016). In other regions, incidents of marsh surface ponding, vegetation shifts and dieback, peat subsidence, channel expansion, and changes in marsh fauna have all been attributed to marsh submergence (Rozas and Reed 1993; Hartig et al. 2002; Alber et al. 2008; Kirwan and Megonigal 2013). Some of these same changes are now occurring in marshes throughout southern New England and are coincident with greater marsh flooding associated with rising water levels. Coupled with new data showing that the majority of RI marshes sit well below the maximum productivity elevations for marsh plants (Watson et al. 2014a), our data demonstrate that RI marshes are at increased risk from accelerating SLR and are likely to continue exhibiting changes reflective of inundation stress.

Factors that are likely contributing to the low rates of marsh elevation gain presented here include low inorganic sediment supplies, increasing belowground decomposition rates associated with warmer temperatures, and reductions in plant growth due to inundation stress. Although comprehensive total suspended sediment (TSS) concentrations are not yet available for RI marshes, preliminary data from three Prudence Island marshes average 9.5 mg L^{-1} (Jun–Sep 2015; Raposa unpublished data), which is consistent with the finding of low suspended sediment loads in other areas of New England (Weston 2014) and does not bode well for marsh survival (Kirwan et al. 2010). At the same time, RI marshes are also experiencing a reduction in the contribution of organic matter to accretion, which is likely a result of increased belowground decomposition due to rising temperatures (Carey et al. 2015b). Recent work also shows that the vast majority of marshes in southern New England already sit at low elevations where inundation stress limits plant growth, resulting in a feedback loop that will likely reduce overall marsh resilience as SLR continues (Watson et al. 2014a; Alizad et al. 2016). Further, we believe it is unlikely that southern New England marshes will be able to keep up with the recent $>5 \text{ mm year}^{-1}$ rate of SLR even if these limiting biophysical conditions were mitigated. In fact, without a sizable increase in the rate of marsh elevation gain, our data suggest that most RI marshes will fail to keep up with future SLR even if it eventually reverts back to the long-term rate of $\sim 2.8 \text{ mm year}^{-1}$.

It is likely that historic ditching also contributes to the current inability of RI salt marshes to keep pace with SLR. During the 1930s, mosquito control ditches were excavated in order to drain surface waters that supported mosquito reproduction, but they ultimately had adverse effects on the marshes and marsh-dependent wildlife (Cottam 1938). Unlike creeks, ditches drain marsh peat at depth and result in elevation loss (Wright 2012; Vincent

et al. 2013). Side casting of excavated material during ditch construction also formed levees that restricted marsh interiors from receiving sediment input (Kennish 2001). The levees in turn curtail drainage so that interior vegetation is subject to prolonged inundation, which leads to stressed vegetation and pool formation (Miller and Egler 1950). Finally, natural clogs in unnatural ditch drainage networks act as ditch plugs (Adamowicz, unpublished data) that can augment stress (Vincent et al. 2013) and result in root collapse (DeLaune et al. 1994). This suite of impacts from ditching clearly has the potential to dampen elevation gain rates and increase the vulnerability of the region's marshes.

Rates of marsh elevation change were surprisingly similar across most SET stations in this study; combining data from all 24 SETs, the inter-quartile range of elevation gain was only $0.69\text{--}2.21 \text{ mm year}^{-1}$. Higher variability was expected because the marshes vary considerably with regard to geomorphology (e.g., the Sachuest Point, Nag, and Nag Pond marshes are all back-barrier systems; Coggeshall and the JHC marshes are more fringing and are open to the estuary) and tide range (e.g., Narrow River tide range is approximately half that of Narragansett Bay; Watson et al. 2014b). Instead, the similarity in rates of elevation gain may reflect the fact that all marshes with SETs in RI are located within the southern half of coastal RI where nutrient loadings and marsh elevations are relatively low, and where marshes are comprised of relatively more short-form *S. alterniflora* (Oczkowski et al. 2008; Cole Ekberg et al. 2015) than upper Bay marshes, which typically have more high marsh perennials. The rates of elevation gain presented here should therefore only be considered as representing a subset of marshes in RI. Because changes in marsh elevation can be affected by nutrients and vegetation composition (e.g., Anisfeld and Hill 2012), elevation change rates may differ in marshes at higher latitudes within coastal RI where these factors differ from marshes at lower latitudes.

It is difficult to compare short-term MH accretion rates in our study to other New England marshes because rates can vary dramatically over time and across sites (e.g., Roman et al. 1997). Even so, the mean RI accretion rate of $1.83 \text{ mm year}^{-1}$ is lower than rates reported for a number of Connecticut, Massachusetts, and New Hampshire salt marshes (Roman et al. 1997; Anisfeld and Hill 2012; Burdick and Peter 2015), which is likely due to a combination of low sediment supplies and a declining contribution from organic matter (Carey et al. 2015b). By combining SET elevation change and short-term MH accretion data, researchers have shown that shallow subsidence can be negligible in some marshes (e.g., Burdick and Peter 2015), while in others it can markedly counteract the contribution of accretion to elevation gains

(Anisfeld and Hill 2012). Our data show that patterns can vary dramatically even within the same estuarine system. Shallow subsidence appears to be prevalent in the Prudence Island marshes, but not at the Sachuest Point and JHC marshes where other subsurface processes (e.g., belowground plant growth) may be slightly augmenting gains in elevation due to accretion alone. Care must be taken when interpreting these data, however, due to the limited number of marshes with SET–MH datasets and to the likely low accuracy associated with measuring very small amounts of deposited material over a small number of years at some MH stations. The accuracy of accretion and subsidence rates should increase over time as more data are collected, and stronger spatial patterns may emerge as more SET–MH stations become established in RI.

Our findings from RI marshes add to the limited amount of elevation change data that are currently available from salt marshes elsewhere in southern New England. For example, Carey et al. (2015a) reported elevation change rates of 2.7 and 2.4 mm year⁻¹ in low marsh and high marsh habitats, respectively, in two Connecticut salt marshes from 2003–2013, and Anisfeld and Hill (2012) reported rates of approximately 1.5 mm year⁻¹ from another Connecticut marsh. In Massachusetts, Erwin et al. (2006) documented a mean rate of 3.1 mm year⁻¹ at the unditched Nauset Marsh on outer Cape Cod from 1999 to 2003, and the Waquoit Bay National Estuarine Research Reserve recorded a mean increase of only 1.7 mm year⁻¹ from 2013 to 2015 from six SETs in one ditched marsh on upper Cape Cod (Jordan Mora unpublished data). These data collectively demonstrate that elevations of most southern New England salt marshes are rising at rates that are much lower than the recent rate of SLR in the region presented here. If the trend continues, we predict that deficits between marsh and tidal water elevations will expand and lead to additional marsh degradation and, eventually, submergence.

Our new elevation change data can be used to improve forecasting of marsh responses to SLR. For example, recent Sea Level Affecting Marshes Model (SLAMM) runs for RI used an accretion rate of 3.8 mm year⁻¹ and predicted overall net gains in future salt marsh area (RI Coastal Resources Management Council 2015). A model re-run with our mean elevation change value (1.40 mm year⁻¹) is likely to produce different and more realistic outcomes and assist in improved policy and planning. SLAMM uses accretion rate as a static input parameter, but other models can be used to make predictions of future accretion rates under varying conditions (see review in Fagherazzi et al. 2012). In the latter case, our data could theoretically be used to calibrate or ground truth models aimed at forecasting accretion rates in RI marshes. Our data therefore set

the stage for more accurate modelling of marsh responses to SLR and for improving predictions of accretion rates in marshes that currently lack in situ field data from SET–MH monitoring.

In conclusion, our study provides further evidence that most RI salt marshes are not keeping pace with short- or long-term rates of SLR. These marshes remain highly vulnerable to continued inundation and many of the indicators of marsh drowning documented elsewhere are now being reported in southern New England. It is therefore likely that many of the changes now occurring in these marshes (e.g., channel expansion and erosion, and high marsh ponding and vegetation dieback) are driven in part by increased inundation associated with the growing deficit between marsh elevation and high tide water levels, a situation that could well have been exacerbated by historic ditching. The RI SET elevation change and accretion datasets presented here will become more robust as annual monitoring continues at existing stations. The 49 existing RI SET stations will also likely be augmented with additional stations that are already planned for marshes in the upper and lower reaches of Narragansett Bay and in a second coastal pond. Upon full build-out, the planned network of SET stations in RI will eventually provide marsh elevation change and accretion datasets along established gradients in marsh elevation, condition, and vegetation composition and provide a more complete picture of marsh vulnerability to SLR in RI and southern New England (Cole Ekberg et al. 2015; Raposa et al. 2016). The RI network will also advance the vision of a global SET–MH monitoring network to better understand marsh vulnerability to SLR at broad spatial scales (Webb et al. 2013).

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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