Evaluating indicators of marsh vulnerability to sea level rise along a historical marsh loss gradient

Lennert Schepers[1*], Matthew Kirwan[2], Glenn Guntenspergen[3], Stijn Temmerman[1]

[1]Ecosystem Management Research Group, University of Antwerp, Antwerp, Belgium.

[2]Department of Physical Sciences, Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, Virginia, USA.

[3]Patuxent Wildlife Research Center, U.S. Geological Survey, Laurel, Maryland, USA.

Corresponding author: Lennert Schepers (lennert.schepers@uantwerpen.be)

(running head: marsh vulnerability indicators along historical marsh loss gradient)

1 Abstract

Sea level rise (SLR) is threatening coastal marshes, leading to large-scale marsh loss in several micro-tidal systems. Early recognition of marsh vulnerability to SLR is critical in these systems to aid managers to take appropriate restoration or mitigation measures. However, it is not clear if current marsh vulnerability indicators correctly assess long-term stability of the marsh system. In this study, two indicators of marsh stress were studied, (i) the skewness of the marsh elevation distribution, and (ii) the abundance of codominant species in mixtures. We combined high-precision elevation measurements (GPS), LiDAR imagery, vegetation surveys and water level measurements to study these indicators in an organogenic micro-tidal system (Blackwater River, Maryland, U.S.A.), where large-scale historical conversion from marshes to shallow ponds resulted in a gradient of increasing marsh loss. The two indicators reveal increasingly stressed marshes along the marsh loss gradient, but suggest that the field site with the most marsh loss seems to experience less stress. For the latter site, previous research indicates that wind waves generated on interior marsh ponds contribute to lateral erosion of surrounding marsh edges and hence marsh loss. The eroded marsh sediment might temporarily provide the remaining marshes with the necessary sediment to keep up with relative SLR. However, this is only a short-term alleviation as lateral marsh edge erosion and sediment export lead to severe marsh loss in the long term. Our findings indicate that marsh elevation skewness and the abundance of codominant species in mixtures can be used to supplement existing marsh stress indicators, but that additional indices such as fetch length and the sediment budget should be included to account for lateral marsh erosion and sediment export and to correctly assess long-term stability of micro-tidal marshes.

2 Introduction

Coastal marshes provide critical ecosystem services, such as carbon sequestration (Chmura et al., 2003; McLeod et al., 2011; Ouyang and Lee, 2014), shoreline protection against storm impacts (Temmerman et al., 2013; Möller et al., 2014) and providing habitat for commercial fisheries (Boesch and Turner, 1984; Barbier et al., 2011), but the persistence of marshes over the next decades to centuries is threatened by accelerating sea level rise (SLR). When marshes cannot adapt to SLR by vertical accretion, submerging marsh vegetation dies off and is replaced by bare mudflats or open water areas, which is evident on large scales in the Mississippi River Delta (Day et al., 2000), the Venice Lagoon (Carniello et al., 2009) and the Chesapeake Bay (Kearney et al., 1988).

Elevated, vegetated marsh platforms and low, unvegetated mudflats or interior open water areas have been identified as a two stable states, each with positive feedback mechanisms that provide long-term stability (Fagherazzi et al., 2006; Marani et al., 2010; Wang and Temmerman, 2013; van Belzen et al., 2017). These feedbacks imply that once marshes convert to open water, it is very hard to reverse the process and to restore marshes. Early recognition of marsh vulnerability to SLR rise is thus critical to foresee these pending shifts and to take early management measures to preserve marshes and their highly valued ecosystem services in face of accelerating SLR.

Two indices, (i) elevation skewness and (ii) the abundance of codominant species in mixtures, have been proposed as indicators of marsh stress, but have never been tested empirically in marshes experiencing large-scale marsh loss. First, the skewness of the elevation distribution has been introduced by Morris et al. (2005). In their study, Morris et al. (2005) argue that resilient marshes have a negative elevation skewness, i.e. with more high marsh than low marsh area, because resilient marshes have the tendency to build-up vertically until their elevation approaches mean high water level (Pethick, 1981; Allen, 1990; Temmerman et al., 2003). The highest high water level is the upper limit for marsh vegetation as sediment deposition must approach zero at elevations near that of the highest high tide (Morris et al., 2002). Further, Morris et al. (2005) argue that an increase in SLR rate would result in increased flooding and would provide accommodation space for marshes to grow, therefore temporarily forming normal elevation distributions within the tidal frame. With higher rates of SLR, the marsh elevation is expected to lag behind sea level, hence to lower within the tidal frame, and consequently to approach their inundation tolerance limit (Kirwan and Murray, 2008; Kirwan and Temmerman, 2009). Marshes below this limit will die off, which results in a positively skewed elevation distribution of the remaining marsh portions above this elevation limit (Morris et al., 2005). Hence the change of elevation skewness from negative to positive values is interpreted as the effect of SLR on the ‘elevation capital’ of a marsh system, i.e. the position of the wetland elevation within the tidal zone (Cahoon and Guntenspergen, 2010).

A second indicator of marsh vulnerability to SLR is (ii) the co-occurrence of species in mixtures. A theoretical background for this hypothesis is the stress gradient hypothesis (Bertness and Callaway, 1994), which postulates that in harsh environments, positive interactions and facilitation between multiple species prevail. In low stress environments, however, competition leads to dominance by few or even one single species. Bertness and Hacker (1994) indeed show that species interactions shift from competitive interactions at the highest marsh elevations to positive interactions when stress levels are high in the low elevated marsh areas. Examples of positive interactions are shade provision, which limits surface evaporation and salt accumulation

(Callaway, 1994) and rhizosphere oxidation which alleviates anoxic substrate conditions (Bertness, 1991a; Boaga et al., 2014). The hypothesis that marshes dominated by codominant species in mixtures are indicative for higher stress levels and higher marsh vulnerability to submergence, has however never been tested.

Wetland assessment methods usually rely on multiple indicators, which are evaluated and combined into a score (Carullo et al., 2007; Rogerson et al., 2010). Recently, Raposa et al. (2016) and Cole Ekberg et al. (2017) both have developed an indicator scheme specifically targeted to assess marsh vulnerability (or resilience) to SLR. Raposa et al. (2016) were the first to develop their MArsh Resilience to SLR (MARS) indices aiming to determine the ability of wetlands to resist SLR. Raposa et al. (2016) combined variables (see Table 1) measuring 'elevation capital' with variables measuring accretion rates and SLR rates to compute a final score. Interestingly, one of their elevation indices is the elevation skewness. Cole Ekberg et al. (2017) included in their assessment also different vegetation types, but relies heavily on model outputs (SLAMM, see Table 1).

[Table 1 here]

Although the development of such indices is highly important to assess marsh vulnerability to SLR, there is a need for validating such indices against observations of long-term (i.e. over decades) historical marsh loss, in order to increase trust in these indices to correctly assess long-term stability of marshes with SLR. In this study, we combined field data and LiDAR imagery to test the skewness of the elevation distribution and the abundance of codominant species in mixtures as indicators of marsh vulnerability with SLR. We collected our data in the Blackwater River marshes (Blackwater Estuary, Maryland, U.S.A.), an organogenic micro-tidal marsh system where widespread historical marsh loss over the last 80 years has resulted in a spatial gradient of increasing marsh loss (Schepers et al., 2017). We test if the skewness of the elevation distribution and the abundance of codominant species differ the indicator scores by Raposa et al. (2016) and Cole Ekberg et al. (2017). Based on our evaluation of the indicator values against observed marsh loss rates, we argue that lateral erosion of marsh edges and consequent sediment export important mechanisms of marsh loss in our study area that are not assessed by the marsh vulnerability indicators, hence we propose indicators that should be included in future further developments of vulnerability indicators. This will aid managers to assess the marsh condition and to take early adaptation measures.

3 Study area

The Blackwater marshes (Maryland, USA; 38°24' N, 76°40' W) are a brackish coastal marsh system along the Blackwater river that connects Lake Blackwater, a large (> 5 km diameter) shallow open water area, with the Fishing Bay, an interior bay of the Chesapeake Bay (Fig. 1.). Since 1938 more than 2,000 ha or 51 % of the marshes have been converted to shallow open water, with most marsh habitat lost at or near Lake Blackwater and leaving the most downstream marshes closest to the Fishing bay relatively intact (Stevenson et al., 1985; Scott et al., 2009; Schepers et al., 2017). As a result, there is a spatial gradient of decreasing marsh area and increasing shallow open water area in upstream direction along the Blackwater River (Fig. 1). The marsh loss has been attributed to insufficient surface accretion relative to SLR (Stevenson et al., 1985), lateral marsh erosion along the pond edges (Stevenson et al., 1985; Ganju et al., 2013) and vegetation disturbance by rodents (Kendrot, 2011).

[Figure 1 here]

Short-term measurements showed that salinity varies little (<2 ppt) between the most downstream site near Fishing Bay and upstream site at Lake Blackwater, but the salinity might change significantly on seasonal timescales (Fleming et al., 2011). The tidal range (from 1 to <0.05 m) and allochthonous sediment input decreases from the Fishing Bay to Lake Blackwater (Ganju et al., 2013). At the most upstream areas, frequent northwestern storms export sediment out of the system (Stevenson et al., 1985; Ganju et al., 2013, 2015).

4 Methods

4.1 Field data

We selected four sites with an increasing proportion of open water areas as a measure of marsh loss (Fig. 1) along the Blackwater River, in such a way that transects of 1,000 m length and perpendicular to the river would not cross other river bends or upland areas. At each site three transects of 1,000 m (250 m apart) straddling the Blackwater River were created in a GIS system (ESRI ArcGIS 10.1). During a field campaign in August 2014 the elevation relative to the North American Vertical Datum of 1988 (NAVD88) was recorded with a high-precision GPS (Trimble R8 RTK-GPS, vertical error <2 cm) every 10 m along each transect. At the same time, we determined the species composition by recording dominant plant species (all species with >30% cover) within a circular, 0.5 m diameter plot centered on each GPS measuring point. Points located in tidal channels or ponds were excluded in this study.

During the same period (August 14 to October 29, 2014), we recorded water level measurements at each field site with pressure transducers (Hobo U20L-02, Onset, MA, USA). The measurements were recalculated to the NAVD88 vertical datum after local atmospheric pressure compensation. Tidal characteristics, including mean tidal range and mean high water level (MHW), were calculated from this dataset with the Tides package in R (Cox and Schepers, 2017).

4.2 LiDAR imagery

Additional to the GPS elevation data, we used LiDAR (Light Detection and Ranging) elevation data that were recorded in spring 2004 and downloaded from Maryland's GIS & Data portal (see Data Availability section) as a 0.91 m resolution digital terrain model. Comparison with our GPS-data (n = 737) revealed a high overall accuracy of the LiDAR data, with an average difference of 0.08 m (RMSE 0.11 m) and normally distributed residuals without trends or spatial patterns (see Supporting Information). Elevation changes in the area are lower than 1 cm/year (Cahoon et al., 2010), which implies that in the 10 year between the LiDAR data (2004) and the field campaign (2014), maximum elevation changes of around 10 cm are in the same order of magnitude as the vertical error on the LiDAR data. For each of the four sites, we extracted the LiDAR elevation data within an area of 125 m around the three transects. Based on an aerial image classification of 2010 (Schepers et al., 2017), only the marsh elevation was retained, resulting in >300,000 elevation points at the most intact site and >150,000 for the most degraded site.

4.2 Marsh vulnerability indices

Several indices of wetland vulnerability have been calculated. The skewness of the elevation distribution was calculated for each site with both the GPS measurements and the LiDAR measurements. We defined the Pearson's moment coefficient of skewness as:

$$skew = \frac{\frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})^3}{\left[\frac{1}{n-1}\sum_{i=1}^{n}(x_i - \bar{x})^2\right]^{3/2}} \qquad \textit{(equation 1)}$$

where n = the number of measurements, x_i = each measurement (here each elevation data point), $\bar{x}$ = the average of all measurements (all elevation data points).

In order to assess whether the skewness of the marsh elevation distribution is an indicator for marsh drowning driven by relative SLR, we simulated marsh drowning at field site 1 by removing the lowest quantiles of the LiDAR marsh elevation distribution in four different steps. These four steps were chosen to simulate an increasing percentage of marsh loss (1.86%; 12.20%; 34.33%; 57.89%) that matches the increasing percentages of marsh loss observed from field site 1 to 4. The skewness was then calculated on the remaining LiDAR marsh elevations after simulated marsh drowning, and it was evaluated whether the simulated marsh drowning resulted in increased skewness of the remaining marsh elevations.

Species composition was determined by recording the dominant plant species (all species with >30% cover) within a circular, 0.5 m diameter plot centered on each GPS measuring point. Species abundances along the marsh loss gradient were calculated by summing the number of occurrences for each species in each field site. The total number of points with at least two dominant species present, i.e. codominant species in a mixture, were also summed for each field site. All numbers were scaled to proportions for each field site because a wider river or more marsh loss results in a different number of marsh point measurements per field site. For example, an abundance of 0.4 for a species in field site 2 means that the species is (co)dominant at 40% of all marsh points in field site 2. Similarly, a mixture of two species with an abundance of 0.12 means that these species are present together (each >30% cover) at 12% of all marsh points of that field site. Site 2 has most points (n=243), slightly more points than field site 1 (n=222), where the river is a little wider, and more points than field sites 3 and 4 (n= 164 and 114, respectively) with more marsh loss. At three points *Spartina patens* (spp), *Spartina alterniflora* (spa) and *Distichlis spicata* (dsp) were present in equal coverages (all >30%). For these points we counted each specific mixture (spp – spa, spa – dsp, dsp – spp).

We verified the results of these two indicators (elevation skewness and vegetation composition) with the indicators of Raposa et al. (2016) and Cole Ekberg et al. (2017)(Table 1). We did not include all indicators because of limited data availability, and we modified two species indicators of Cole Ekberg et al. (2017). The percentage of low marsh is in Cole Ekberg et al. (2017) defined as regularly flooded areas dominated by tall-form (>50 cm) *Spartina alterniflora*. We defined this indicator as areas with *Spartina alterniflora* monocultures (the only species with a cover >30 %). Cole Ekberg et al. (2017) defined Perennial turfgrass type I as 'irregularly flooded, dominated by *Spartina patens, Distichlis spicata, Juncus gerardii,* with no *S. alterniflora*'. We did not record species with coverages <30 %, so we can not garantuee that *S. alterniflora* is not present at our GPS points. Therefore we modified this index as areas with *Distichlis spicata* or *Spartina patens* monocultures (or a mixture of both) but where *Spartina alterniflora* was not dominant (cover is <30 %).

To calculate the MARS average scores, we used the metric threshold table from Raposa et al. (2016, see Table 2). The average of the category scores (marsh elevation score and tidal range) produced a MARS average score for each field site. To convert the Cole Ekberg et al. (2017) indices to field site scores, we ranked the index scores from 1-4 with increasing vulnerability and calculated an average score. Note that contrary to Cole Ekberg et al. (2017), a higher score indicates a higher vulnerability.

4.3 Analyses and statistics

All data analyses, figures and maps were made in R (R Core Team, 2017). The non-parametric bootstrap technique was applied to determine if the skewness of the GPS elevation distribution was significantly different from zero: the elevation dataset was randomly resampled 10,000 times with replacement, each resample having the same size as the original dataset, and on each resample the skewness was calculated. If zero is within the 2.5^{th} and 97.5^{th} percentile of these 10,000 skewness values, the skewness is not significantly different from zero, similarly to a statistical test with 0.05 confidence level (α). Since the LiDAR-dataset consists of a huge dataset ($n > 150{,}000$) and includes the whole study area rather than a selection of sample points, the bootstrap technique was deemed to be not necessary for the LiDAR data.

5 Results

5.1 Elevation skewness

At the selected field sites the marsh loss ranges from 2 % at field site 1 to 58% at field site 4 (Fig. 2 top row). The skewness of the marsh elevation distribution at these field sites shows similar results for the LiDAR analyses (Fig. 2 row 2), the GPS measurements (Fig 2. row 3) as well as the GPS bootstrap method (Fig. 2 bottom row). The skewness shifts from significantly negative at the most intact field site 1, to non-skewed at site 2, to a positive skewness at site 3 (Fig. 2). Field site 4 has a non-skewed elevation.

[Figure 2 here]

[Figure 3 here]

5.2 Vegetation community structure

[Figure 4 here]

The most abundant species in the Blackwater Marshes are *Schoenoplectus americanus*, *Spartina alterniflora* and *Spartina patens* (Fig. 4). Although *Spartina patens* is considered as a high marsh species (Bertness, 1991b; Smith, 2009; Wigand et al., 2011; Raposa et al., 2017) and *Schoenoplectus americanus* and *Spartina alterniflora* are considered low marsh species (Bertness, 1991b; Nyman et al., 1994; Donnelly and Bertness, 2001), these species do not show a clear abundance shift with increasing marsh loss (Fig. 4) or with decreasing marsh surface elevation (data not shown). A less abundant high marsh species is *Distichlis spicata* (dsp), which does show a clear decline along the marsh loss gradient (Fig. 4). *Spartina cynosuroides* (spc) occurs in more or less similar abundances throughout the study area (Fig. 4)*. Bolboschoenus robustus*, *Juncus gerardii* and *Iva fructusea* were sparsely present (<10 points) and not included in our analysis.

[Figure 5 here]

Specific mixtures between two species show various patterns with increasing marsh loss (i.e. from field site 1 to field site 4). However, when summing all the mixture points, our results show that field site 3 has the highest proportion of mixture points (Fig. 5), whereas field site 4 has a lower proportion of mixtures.

5.3 Comparison with other indices indicating marsh vulnerability to SLR.

In Table 2 the results are shown of the MArsh Resilience to SLR (MARS) indices, calculated for our study area. Darker colors are indicative of more vulnerable marshes. In Table 3 we calculated indices that were mentioned in the recent work of Cole Ekberg et al. (2017). Although there are some differences between individual indices, the average scores indicate similar results, with increasing marsh loss from field site 1 to 4, the indices indicate an increasing vulnerability to SLR from field site 1 to 3. At field site 4, the site with most marsh loss, the vulnerability decreases again compared to field site 3.

[Table 2 here]

[Table 3 here]

6 Discussion

6.1 Elevation skewness

The skewness of the marsh elevation distribution shifts from significantly negative at the most intact field site 1, to non-skewed at site 2, to a positive skewness at site 3 (Fig. 2). This indicates that the marshes at the intact site 1 are mostly high elevation marshes close to their upper limit with a large elevation capital, and therefore highly resilient to SLR. In contrast, the positive skewness at field site 3 with considerable marsh loss indicates that there are more low elevation marshes close to their lower elevation limit, and hence vulnerable to SLR. Field site 4 deviates from this trend of increasing positive skewness with increasing marsh loss percentage, as field site 4 has the highest marsh loss percentage but a non-skewed elevation distribution. Potential explanations for this observation will be discussed at the end of this section.

An increase in the rate of SLR will lower the elevation of the marshes compared to the mean high water level and create accommodation space and increased flooding (Morris et al., 2005). Since marshes need time to build-up to adjust to a new equilibrium rate, the skewness of the elevation distribution becomes more positive (Kirwan and Murray, 2008; Kirwan et al., 2010). The shift from negative to positive skewness with increasing marsh loss from site 1 to 3 can be explained by the gradient in marsh loss. First, marshes with lower tidal ranges will be more impacted by a similar sea level rise because of a proportionally bigger shift of their growth range (Kirwan and Guntenspergen, 2010; Kearney and Turner, 2016). More marsh loss thus results in a more positive skewness (Fig. 3). This is the case in our study area, where tidal range decreases in upstream direction from field site 1 (0.63 m tidal range) to field site 3 (0.20 m tidal range). Simulated marsh drowning of the lowest elevated marsh portions (Fig. 3) indeed shows that increasing marsh loss by drowning results in an increasingly positive skewness. This is not surprising since we cut off lower values, but the fact that the patterns of simulated marsh loss resemble the actual observed patterns (Fig. 3, compare top row with bottom row), supports the hypothesis that marsh die-off at the lowest areas is at least partly responsible for the observed shift in skewness from field site 1 to 3.

At field site 4 the elevation has a slightly positive (LiDAR data) or no significant skewness (GPS bootstrap method, see Fig. 2 right column). This is remarkable; from our marsh loss simulation (Fig. 3) we would expect the most vulnerable, positive skewness at site field site 4 as this site has experienced most marsh loss. We hypothesize that is due to increased sediment availability generated by eroding marsh edges or pond bottoms at the most degraded field site 4. Studies of Ganju et al. (2013, 2015) in the same area support this hypothesis, showing appreciable suspended sediment concentrations (55 mg/L) and high accretion rates (>5 mm/yr) within marsh vegetation close to field site 4, likely originating from sediments eroding from pond bottoms and marsh edges. Furthermore, the remaining marsh portions in field site 4 are mostly located along the edges of ponds and channels (see Fig. 2) and therefore close to the local source areas of suspended sediments. These sediments build up the few remaining marsh areas, leading to non-skewed elevation distributions (Fig. 2). This hypothesis implies then that the lowest, most vulnerable areas have already converted to open water ponds, and that the remaining marsh portions fringing along channel and pond edges are only moderately vulnerable to SLR.

6.2 Vegetation community structure

The most abundant species in the Blackwater Marshes are *Schoenoplectus americanus*, *Spartina alterniflora* and *Spartina patens* (Fig. 4). *Spartina patens* is considered as a high marsh species (Bertness, 1991b; Smith, 2009; Wigand et al., 2011; Raposa et al., 2017) and *Schoenoplectus americanus* and *Spartina alterniflora* are considered low marsh species (Bertness, 1991b; Nyman et al., 1994; Donnelly and Bertness, 2001), but these species do not show a clear abundance shift with increasing marsh loss (Fig. 4) or with decreasing marsh surface elevation (data not shown). We hypothesize that clear abundance shifts with marsh loss are absent because they occupy elevations far below their optimum range, as flooding experiments of *Schoenoplectus americanus* and *Spartina patens* in the Blackwater Marshes have demonstrated (Kirwan and Guntenspergen, 2012; 2015). In these cases, other factors such as interspecific interactions might become more important.

6.3 Indices neglect erosional feedback mechanisms

Most indicators (% elevation below third elevation, elevation skewness, tidal range, marsh orthometric height, percentage unvegetated) show the consistent pattern of increasing vulnerability with increasing marsh loss from the intact marshes at site 1 to the degraded marshes at field site 3. However, field site 4 shows a consistent decrease in vulnerability for nearly all the indices. This is especially noteworthy because field site 4 is the area with the highest proportion of marsh loss, where more than 58% of the area consists of ponds that once were vegetated marshes (e.g. Schepers et al. 2017).

Site 4 has not only the highest proportion of marsh loss, but also the largest (Schepers et al., 2017) and deepest ponds (Schepers et al., 2019), providing favorable conditions for wind-generated wave erosion of the marsh edges surrounding the ponds. Also previous studies in other marsh areas with interior marsh ponds have highlighted that wind waves generated on the ponds play a key role in erosion of the marsh edges surrounding ponds. For example, Mariotti et al. (2013) and Ortiz et al. (2017) have demonstrated that interior marsh ponds larger than a certain threshold width (300 m in Mississippi marshes, 200-1000 m in US Atlantic marshes) are susceptible to runaway expansion due to the positive feedback between increasing wind fetch length and wave erosion of the surrounding marsh edges. Lateral expansion of ponds enhances the chance for pond merging, leading to larger ponds, larger wind fetch length and hence higher rates of wave-driven marsh edge erosion. This is also evident along the die-off gradient in our study area, where the average fetch length of the ponds increases suddenly from ca. 200 m at field site 4 to >1000 m just upstream at Lake Blackwater (Schepers et al., 2017) (Fig. 1). Ganju et al. (2013) demonstrate that marsh edge erosion along Lake Blackwater occurs predominantly into the direction of the most energetic wind (and ostensibly wave) direction, highlighting the role of wind-induced erosion as a mechanism for marsh loss. Schepers et al. (2017) show that over the last 80 years, open water areas have expanded considerably also at field site 4.

Furthermore, the sediments that are brought into suspension in the ponds due to surrounding marsh edge erosion, are subsequently exported from the marsh system. Ganju et al. (2013) measured tidal sediment fluxes through the main tidal channel draining the Blackwater marshes (our study area), and demonstrated a clear net sediment export from the marsh system during periods of dominant northwesterly winds. In a follow-up study, Ganju et al. (2015) demonstrated that upstream Blackwater marshes (close to field site 4) are unstable and exporting sediment, whereas marshes close to field site 1 are stable and importing sediment (Ganju et al. 2015, Guntenspergen 2017, unpublished data). The observation that ponds in site 4 are mostly connected to the tidal channel network (Schepers et al., 2017) might also facilitate export of eroded material with tidal ebb currents (Ganju et al. 2013; Schepers et al., 2019). In a further study by Ganju et al. (2017), including 8 different micro-tidal marsh systems, the Blackwater marshes were the only marsh system having a clear negative sediment flux. These observations all suggest that the marshes at field site 4 are vulnerable to an irreversible trajectory of continuing marsh edge erosion, rather than being only moderately vulnerable to SLR as suggested by the vulnerability indices.

6.4 Accounting for lateral erosional processes to determine marsh loss risk

We highlighted that existing indicators for marsh vulnerability to SLR do not account for lateral marsh edge erosion, a process that can erode the whole marsh by positive feedback mechanisms between increasing wind fetch length over increasing open water area (ponds), and increasing erosion rate of the adjacent marsh edge (Fagherazzi et al., 2013; Mariotti and Carr, 2014). To overcome this problem, we recommend including (i) the fetch length of interior marsh ponds and (ii) tidal channel sediment flux to assess the risk of marsh loss. Fetch length is a straightforward indicator for wind-generated wave erosion of marsh edges surrounding ponds, and is defined as the distance over continuous water surface in one direction, from one side of the pond to the other over which wind can generate waves (Rohweder et al., 2012; Ortiz et al., 2017; Schepers et al., 2017). Schepers et al. (2017) quantified the average fetch length oriented along 4 directions (N-S, E-W, NW-SE, NE-SW). If we calculate the average fetch length at each point in our transect that is located in a pond, we see an increase from 4.3 m to 281.0 m (Table 4), hence larger fetch lengths at site 4 means higher vulnerability to lateral marsh erosion.

[Table 4 here]

Large fetches can activate wave-induced marsh erosion, but the question is where the eroded sediment is transported to and eventually deposited, as this determines marsh stability on the long term (e.g. Ganju et al. , 2013, 2015, 2017). Does the eroded sediment nourish nearby marshes (Hopkinson et al 2018)? Or is it exported from the marsh system? The sediment budget of a marsh system will ultimately determine the final course of the marsh (French, 2006; Mariotti and Carr, 2014; Ganju et al., 2015, 2017; Mariotti, 2016). The connection of large ponds with the tidal channel system might be an important factor influencing tidal import or export (Millette et al., 2010; Schepers et al., 2019; Wilson et al., 2010, 2014). Sediment budgets have been measured through these connections and tidal channels, by intensive and long-term sediment flux measurements (Ganju et al., 2015, 2017). However, Ganju et al. (2017) suggest, based on a sediment flux study in eight micro-tidal marshes along the Atlantic and Pacific coasts, that the ratio between unvegetated-vegetated marsh area can be used as a single snapshot to infer the sediment budget. When this metric is calculated for our field sites, the ratio increases from 0.02 at field site 1 to 1.37 at field site 4 (Table 4). In fact, this metric is similar to the 'percentage unvegetated' as proposed by Cole Ekberg et al. (2017), the only metric that did not show a decrease in vulnerability at field site 4 (Table 3).

In conclusion, we propose to include fetch length of ponds and ratio of unvegetated versus vegetated marsh area as two additional indices to assess the long-term vulnerability of marshes, in particular to wind wave driven marsh edge erosion and tidal export of eroded sediments. The values for these two metrics show an increase from field site 1 towards field site 4, and indeed identify field site 4 as the most vulnerable to marsh loss in our study area.

7 Conclusion

Along a spatial gradient of increasing marsh loss, the skewness of the marsh elevation distribution shifts from negative to positive, indicating a shift from marshes that are resilient to SLR to marshes that are highly vulnerable. The abundance of individual species was not found to be a good indicator for marsh vulnerability to SLR, but the total proportion of codominant species in mixtures increased along the marsh die-off gradient and resulted in the same vulnerability trend as the elevation skewness. These vulnerability trends corroborate previously proposed indices of marsh vulnerability to SLR.

A remarkable result was that all above-mentioned indices suggested a lower vulnerability at the site with the largest area of marsh loss. This highlights a shortcoming of the current indices because they do not account for marsh loss by lateral erosional feedback mechanisms by wind waves and tidal currents. We recommend including two parameters, fetch length and sediment budget, to correctly assess long-term vulnerability which should aid managers to take the appropriate restoration or mitigation measures.

Our results demonstrate that there might be two management strategies, based on the extent of marsh loss. The elevation skewness and the proportion of mixtures of marshes, in combination with newly proposed indicators by Raposa et al. (2016) and Cole Ekberg et al. (2017), can be used initially as indictors of vertical marsh vulnerability to SLR. This will aid managers to monitor existing marshes and to recognize early signs of marsh stress to SLR. If needed, the managers can take actions to alleviate the stress e.g. by ensuring that enough sediment reaches marshes to keep up with SLR or sediment is applied to marshes to recover optimum elevations. However, if marshes are already experiencing marsh loss, our results indicate that managers should be especially cautious about lateral processes that are not included in typical vulnerability assessments, and can result in runaway erosion of large marshes. Preventing wave erosion by nature based strategies such as building oyster reefs might be a suitable option (Walles et al., 2015; Salvador de Paiva et al., 2018). Importantly, maintaining existing marshes is essential for either strategy since recovery of lost marshes may be far more difficult to achieve due to feedback mechanisms (Slocum and Mendelssohn, 2008; Day et al., 2011; Baustian et al., 2012; Mariotti, 2016).

Data availability

The LiDAR data was originally extracted from Maryland's GIS & Data portal (http://imap.maryland.gov), but is currently no longer available. We have stored our extract with metadata in the Marine Data Archive at http://mda.vliz.be/directlink.php?fid=VLIZ_00000640_5d7e98c6af96d

Acknowledgements

This project was financed by an UA-BOF DOCPRO grant (to L.S. and S.T.), the Research Foundation Flanders (FWO, PhD grants L.S., 11S9614N & 11S9616N, travel grants L.S. V428214N and S.T. K217414N), by the U.S. Geological Survey, Climate and Land-Use Research and Development Program (G.G.), by NSF GLD 1529245, NSF SEES 1426981, NSF LTER 1237733 (M.K.). We would like to thank the managers and biologists of the Blackwater National Wildlife Refuge for their assistance and valuable comments; and P. Brennand (USGS) for indispensable field assistance. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Conflict of interest

The authors declare no conflict of interest.

References

Allen, J.R.L., 1990. Salt-marsh growth and stratification: A numerical model with special reference to the Severn Estuary, southwest Britain. Mar. Geol. 95, 77–96. doi:10.1016/0025-3227(90)90042-I

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. doi:10.1890/10-1510.1

Baustian, J.J., Mendelssohn, I. a., Hester, M.W., 2012. Vegetation's importance in regulating surface elevation in a coastal salt marsh facing elevated rates of sea level rise. Glob. Chang. Biol. 18, 3377–3382. doi:10.1111/j.1365-2486.2012.02792.x

Bertness, M.D., 1991a. Interspecific Interactions among High Marsh Perennials in a New England Salt Marsh. Ecology 72, 125–137. doi:10.2307/1938908

Bertness, M.D., 1991b. Zonation of Spartina Patens and Spartina Alterniflora in New England Salt Marsh. Ecology 72, 138–148. doi:10.2307/1938909

Bertness, M.D., Callaway, R., 1994. Positive interactions in communities. Trends Ecol. Evol. 9, 187–191. doi:10.1016/0169-5347(94)90087-6

Bertness, M.D., Hacker, S.D., 1994. Physical Stress and Positive Associations Among Marsh Plants. Am. Nat. 144, 363. doi:10.1086/285681

Boaga, J., D'Alpaos, A., Cassiani, G., Marani, M., Putti, M., 2014. Plant-soil interactions in salt marsh environments: Experimental evidence from electrical resistivity tomography in the Venice Lagoon. Geophys. Res. Lett. 41, 6160–6166. doi:10.1002/2014GL060983

Boesch, D.F., Turner, R.E., 1984. Dependence of Fishery Species on Salt Marshes: The Role of Food and Refuge. Estuaries 7, 460–468. doi:10.2307/1351627

Cahoon, D.R., Guntenspergen, G.R., 2010. Climate Change, Sea-Level Rise, and Coastal Wetlands. Natl. Wetl. Newsl. 32, 8–12.

Cahoon, D.R., Guntenspergen, G.R., Baird, S., 2010. Do Annual Prescribed Fires Enhance or Slow the Loss of Coastal Marsh Habitat at Blackwater National Wildlife Refuge? Final Report to Joint Fire Science Program Project Number 06-2-1-35.

Callaway, R.M., 1994. Facilitative and Interfering Effects of Arthrocnemum Subterminale on Winter Annuals. Ecology 75, 681–686.

Carniello, L., Defina, A., D'Alpaos, L., 2009. Morphological evolution of the Venice lagoon: Evidence from the past and trend for the future. J. Geophys. Res. Earth Surf. 114, 1–10. doi:10.1029/2008JF001157

Carullo, M., Carlisle, B.K., Smith, J.P., 2007. A New England rapid assessment method for assessing condition of estuarine marshes: A Boston Harbor, Cape Cod and Islands pilot study. Massachusetts Off. Coast. Zo. Manag. Boston, USA.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochem. Cycles 17. doi:10.1029/2002GB001917

Cole Ekberg, M.L., Raposa, K.B., Ferguson, W.S., Ruddock, K., Watson, E.B., 2017. Development and Application of a Method to Identify Salt Marsh Vulnerability to Sea Level Rise. Estuaries and Coasts 40, 694–710. doi:10.1007/s12237-017-0219-0

Cox, T., Schepers, L., 2017. Tides: Quasi-Periodic Time Series Characteristics. R package version 2.0. https://cran.r-project.org/web/packages/Tides.

Day, J.W., Britsch, L.D., Hawes, S.R., Shaffer, G.P., Reed, D.J., Cahoon, D.R., 2000. Pattern and Process of Land Loss in the Mississippi Delta: A Spatial and Temporal Analysis of Wetland Habitat Change. Estuaries 23, 425–438. doi:10.2307/1353136

Day, J.W., Kemp, G.P., Reed, D.J., Cahoon, D.R., Boumans, R.M., Suhayda, J.M., Gambrell, R., 2011. Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: The role of sedimentation, autocompaction and sea-level rise. Ecol. Eng. 37, 229–240. doi:10.1016/j.ecoleng.2010.11.021

Donnelly, J.P., Bertness, M.D., 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. Proc. Natl. Acad. Sci. U. S. A. 98, 14218–14223. doi:10.1073/pnas.251209298

Fagherazzi, S., Carniello, L., D'Alpaos, L., Defina, A., 2006. Critical bifurcation of shallow microtidal landforms in tidal flats and salt marshes. Proc. Natl. Acad. Sci. U. S. A. 103, 8337–8341. doi:10.1073/pnas.0508379103

Fagherazzi, S., Mariotti, G., Wiberg, P., McGlathery, K., 2013. Marsh Collapse Does Not Require Sea Level Rise. Oceanography 26, 70–77. doi:10.5670/oceanog.2013.47

Fleming, B.J., DeJong, B.D., Phelan, D.J., 2011. Geology, hydrology, and water quality of the Little Blackwater River watershed, Dorchester County, Maryland, 2006–09: U.S. Geological Survey Scientific Investigations Report 2011–5054. http://pubs.usgs.gov/sir/2011/5054/.

French, J., 2006. Tidal marsh sedimentation and resilience to environmental change: Exploratory modelling of tidal, sea-level and sediment supply forcing in predominantly allochthonous systems. Mar. Geol. 235, 119–136. doi:10.1016/j.margeo.2006.10.009

Ganju, N.K., Defne, Z., Kirwan, M.L., Fagherazzi, S., D'Alpaos, A., Carniello, L., 2017. Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes. Nat. Commun. 8, 14156. doi:10.1038/ncomms14156

Ganju, N.K., Kirwan, M.L., Dickhudt, P.J., Guntenspergen, G.R., Cahoon, D.R., Kroeger, K.D., 2015. Sediment transport-based metrics of wetland stability. Geophys. Res. Lett. 42, 7992–8000. doi:10.1002/2015GL065980

Ganju, N.K., Nidzieko, N.J., Kirwan, M.L., 2013. Inferring tidal wetland stability from channel sediment fluxes: Observations and a conceptual model. J. Geophys. Res. Earth Surf. 118, 2045–2058. doi:10.1002/jgrf.20143

Hopkinson, C. S., Morris, J. T., Fagherazzi, S., Wollheim, W. M., Raymond, P. A. 2018. Lateral marsh edge erosion as a source of sediments for vertical marsh accretion. J. Geophys. Res. Biogeosci. 123(8), 2444-2465. doi: 10.1029/2017JG004358

Kearney, M.S., Grace, R.E., Stevenson, J.C., 1988. Marsh Loss in Nanticoke Estuary, Chesapeake Bay. Geogr. Rev. 78, 205–220. doi:10.2307/214178

Kearney, M.S., Turner, R.E., 2016. Microtidal Marshes: Can These Widespread and Fragile Marshes Survive Increasing Climate–Sea Level Variability and Human Action? J. Coast. Res. 319, 686–699. doi:10.2112/JCOASTRES-D-15-00069.1

Kendrot, S.R., 2011. Restoration through eradication : protecting Chesapeake Bay marshlands from invasive nutria (Myocastor coypus), in: Veitch, C.R., Clout, M.N., Towns, D.R. (Eds.), Island Invasives: Eradication and Management. Proceedings of the International Conference on Island Invasives. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB, pp. 313–319.

Kirwan, M.L., Guntenspergen, G.R., 2010. Influence of tidal range on the stability of coastal marshland. J. Geophys. Res. 115, F02009. doi:10.1029/2009JF001400

Kirwan, ML and Guntenspergen, GR. 2012. Feedbacks between inundation, root production, and shoot growth in a rapidly submerging brackish marsh. ***Journal of Ecology***, 100, 764-770.

Kirwan, M.L. and Guntenspergen, G.R., 2015. Response of Plant Productivity to Experimental Flooding in a Stable and a Submerging Marsh. ***Ecosystems*** 18, 903-913.

Kirwan, M.L., Guntenspergen, G.R., D'Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophys. Res. Lett. 37, L23401. doi:10.1029/2010GL045489

Kirwan, ML and Murray, AB, 2008. Tidal marshes as disequilibrium landscapes? Lags between morphology and Holocene sea level change. ***Geophysical Research Letters*** *35*, L24401 doi:10.1029/2008GL036050.

Kirwan, M.L., Murray, a. B., 2008. Ecological and morphological response of brackish tidal marshland to the next century of sea level rise: Westham Island, British Columbia. Glob. Planet. Change 60, 471–486. doi:10.1016/j.gloplacha.2007.05.005

Kirwan, M and Temmerman, S, 2009. Coastal marsh response to historical and future sea level acceleration. ***Quaternary Science Reviews,*** v. 28, p. 1801-1808, doi:10.1016/j.quascirev.2009.02.022.

Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., Rinaldo, A., 2010. The importance of being coupled: Stable states and catastrophic shifts in tidal biomorphodynamics. J. Geophys. Res. Earth Surf. 115, 1–15. doi:10.1029/2009JF001600

Mariotti, G., 2016. Revisiting salt marsh resilience to sea level rise: Are ponds responsible for permanent land loss? J. Geophys. Res. Earth Surf. 121, 1391–1407. doi:10.1002/2016JF003900

Mariotti, G., Carr, J., 2014. Dual role of salt marsh retreat: Long-term loss and short-term resilience. Water Resour. Res. 50, 2963–2974. doi:10.1002/2013WR014676

McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H., Silliman, B.R., 2011. A blueprint for blue carbon: Toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front. Ecol. Environ. 9, 552–560. doi:10.1890/110004

Millette, T.L., Argow, B.A., Marcano, E., Hayward, C., Hopkinson, C.S., Valentine, V., 2010. Integration of Multitemporal Multispectral Remote Sensing with LIDAR and GIS. J. Coast. Res. 265, 809–816. doi:10.2112/JCOASTRES-D-09-00101.1

Möller, I., Kudella, M., Rupprecht, F., Spencer, T., Paul, M., van Wesenbeeck, B.K., Wolters, G., Jensen, K., Bouma, T.J., Miranda-Lange, M., Schimmels, S., 2014. Wave attenuation over coastal salt marshes under storm surge conditions. Nat. Geosci. 7, 727–731. doi:10.1038/ngeo2251

Morris, J.T., Porter, D., Neet, M., Noble, P.A., Schmidt, L., Lapine, L.A., Jensen, J.R., 2005. Integrating LIDAR elevation data, multi-spectral imagery and neural network modelling for marsh characterization. Int. J. Remote Sens. 26, 5221–5234. doi:10.1080/01431160500219018

Morris, J.T., Sundareshwar, P. V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of Coastal Wetlands to Rising Sea Level. Ecology 83, 2869–2877. doi:10.1890/0012-9658(2002)083[2869:ROCWTR]2.0.CO;2

Nyman, J.A., Carloss, M., DeLaune, R.D., Patrick, W.H., 1994. Erosion rather than plant dieback as the mechanism of marsh loss in an estuarine marsh. Earth Surf. Process. Landforms 19, 69–84. doi:10.1002/esp.3290190106

Ortiz, A.C., Roy, S., Edmonds, D.A., 2017. Land loss by pond expansion on the Mississippi River Delta Plain. Geophys. Res. Lett. 44, 3635–3642. doi:10.1002/2017GL073079

Ouyang, X., Lee, S.Y., 2014. Updated estimates of carbon accumulation rates in coastal marsh sediments. Biogeosciences 11, 5057–5071. doi:10.5194/bg-11-5057-2014

Pethick, J.S., 1981. Long-term Accretion Rates on Tidal Salt Marshes. J. Sediment. Res. 51, 571–577. doi:10.1306/212F7CDE-2B24-11D7-8648000102C1865D

R Core Team, 2017. R: A Language and Environment for Statistical Computing, Vienna, Austria.

Raposa, K.B., Wasson, K., Smith, E., Crooks, J.A., Delgado, P., Fernald, S.H., Ferner, M.C., Helms, A., Hice, L.A., Mora, J.W., Puckett, B., Sanger, D., Shull, S., Spurrier, L., Stevens, R., Lerberg, S., 2016. Assessing tidal marsh resilience to sea-level rise at broad geographic scales with multi-metric indices. Biol. Conserv. 204, 263–275. doi:10.1016/j.biocon.2016.10.015

Raposa, K.B., Weber, R.L.J., Ekberg, M.C., Ferguson, W., 2017. Vegetation Dynamics in Rhode Island Salt Marshes During a Period of Accelerating Sea Level Rise and Extreme Sea Level Events. Estuaries and Coasts 40, 640–650. doi:10.1007/s12237-015-0018-4

Rogerson, A., Mclaughlin, E., Havens, K., 2010. Mid-Atlantic Tidal Wetland Rapid Assessment Method Version 3.0.

Rohweder, J., Rogala, J.T., Johnson, B.L., Anderson, D., Clark, S., Chamberlin, F., Potter, D., Runyon, K., 2012. Application of wind fetch and wave models for habitat rehabilitation and enhancement projects—2012 update. U.S. Army Corp. Eng. Contract Rep. doi:10.1017/CBO9781107415324.004

Salvador de Paiva, J.N., Walles, B., Ysebaert, T., Bouma, T.J., 2018. Understanding the conditionality of ecosystem services: The effect of tidal flat morphology and oyster reef characteristics on sediment stabilization by oyster reefs. Ecol. Eng. 112, 89–95. doi:10.1016/j.ecoleng.2017.12.020

Schepers, L., Brennand, P., Kirwan, M., Guntenspergen, G., Temmeraman, S., 2019. Coastal Marsh Degradation into Ponds Induces Irreversible Elevation Loss Relative to Sea Level. Manuscript submitted for publication.

Schepers, L., Kirwan, M., Guntenspergen, G., Temmerman, S., 2017. Spatio-temporal development of vegetation die-off in a submerging coastal marsh. Limnol. Oceanogr. 62, 137–150. doi:10.1002/lno.10381

Scott, M., McDermott, L., Silva, E., Watson, E., 2009. Project report: Digital Spatial Data Capture of Marsh Extent in Blackwater National Wildlife Refuge, 1938 and 2006. Eastern Shore Regional

GIS Cooperative, Salisbury University.

Slocum, M.G., Mendelssohn, I.A., 2008. Use of experimental disturbances to assess resilience along a known stress gradient. Ecol. Indic. 8, 181–190. doi:10.1016/j.ecolind.2007.01.011

Smith, S.M., 2009. Multi-Decadal Changes in Salt Marshes of Cape Cod, MA: Photographic Analyses of Vegetation Loss, Species Shifts, and Geomorphic Change. Northeast. Nat. 16, 183–208. doi:10.1656/045.016.0203

Stevenson, J.C., Kearney, M.S., Pendleton, E.C., 1985. Sedimentation and erosion in a Chesapeake Bay brackish marsh system. Mar. Geol. 67, 213–235. doi:10.1016/0025-3227(85)90093-3

Temmerman, S., Govers, G., Meire, P., Wartel, S., 2003. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium. Mar. Geol. 193, 151–169. doi:10.1016/S0025-3227(02)00642-4

Temmerman, S., Meire, P., Bouma, T.J., Herman, P.M.J., Ysebaert, T., De Vriend, H.J., 2013. Ecosystem-based coastal defence in the face of global change. Nature 504, 79–83. doi:10.1038/nature12859

van Belzen, J., van de Koppel, J., Kirwan, M.L., van der Wal, D., Herman, P.M.J., Dakos, V., Kéfi, S., Scheffer, M., Guntenspergen, G.R., Bouma, T.J., 2017. Vegetation recovery in tidal marshes reveals critical slowing down under increased inundation. Nat. Commun. 8, 15811. doi:10.1038/ncomms15811

Walles, B., Salvador de Paiva, J., van Prooijen, B.C., Ysebaert, T., Smaal, A.C., 2015. The Ecosystem Engineer Crassostrea gigas Affects Tidal Flat Morphology Beyond the Boundary of Their Reef Structures. Estuaries and Coasts 38, 941–950. doi:10.1007/s12237-014-9860-z

Wang, C., Temmerman, S., 2013. Does biogeomorphic feedback lead to abrupt shifts between alternative landscape states?: An empirical study on intertidal flats and marshes. J. Geophys. Res. Earth Surf. 118, 229–240. doi:10.1029/2012JF002474

Wigand, C., Carlisle, B., Smith, J., Carullo, M., Fillis, D., Charpentier, M., McKinney, R., Johnson, R., Heltshe, J., 2011. Development and validation of rapid assessment indices of condition for coastal tidal wetlands in southern New England, USA. Environ. Monit. Assess. 182, 31–46. doi:10.1007/s10661-010-1856-y

Wilson, C.A., Hughes, Z.J., FitzGerald, D.M., Hopkinson, C.S., Valentine, V., Kolker, A.S., 2014. Saltmarsh pool and tidal creek morphodynamics: Dynamic equilibrium of northern latitude saltmarshes? Geomorphology 213, 99–115. doi:10.1016/j.geomorph.2014.01.002

Wilson, K.R., Kelley, J.T., Tanner, B.R., Belknap, D.F., 2010. Probing the Origins and Stratigraphic Signature of Salt Pools from North-Temperate Marshes in Maine, U.S.A. J. Coast. Res. 26, 1007–1026. doi:10.2112/JCOASTRES-D-10-00007.1

Figure Captions

Figure 1: Aerial image of the Blackwater Marshes. From lower right corner to upper left corner of the image (i.e. in upstream direction along the Blackwater River) marshes are changing from high marsh vegetation cover (reddish color) close to the Fishing Bay (SE-corner) to increasing open water areas (dark color) in upstream direction, and ultimately to Lake Blackwater (NW-corner). White lines indicate GPS measurement points. White shaded areas with dashed outlines are no marshes but upland areas. Inset: Location of the Blackwater marshes along the Chesapeake Bay (white rectangle)

Figure 2: Changes in skewness of elevation distributions along the gradient of increasing marsh loss from field site 1 (left) to 4 (right). Top to bottom: aerial images of field sites with increasing marsh loss (marshes represented by reddish color); LiDAR elevation distributions and calculated skewness; GPS elevation distributions and skewness; and bootstrap analysis of the skewness of the GPS elevation distributions, indicating if the skewness is significantly negative (resilient marsh), neutral or positive (vulnerable marsh) when the 0 skewness (red vertical line) is lower, in or higher than the skewness range (indicated by the grey vertical lines i.e. the 2.5th and 97.5th percentile) respectively.

Figure 3: Changes in skewness of elevation distributions along a gradient of simulated increasing marsh loss (from left to right). Top row: aerial images of field site 1 with simulated progressive marsh loss of the lowest areas. Marsh areas are represented by reddish colors. Middle row: LiDAR elevation distribution and skewness corresponding with the marsh loss simulations. Bottom row: The observed marsh loss at field sites 1 to 4. Note that the simulated marsh loss resemble the observed marsh loss.

Figure 4: Abundances of the most prevalent species (sa: *Schoenoplectus americanus*, spa: *Spartina alterniflora*, spp: *Spartina patens*, dsp: *Distichlis spicata* and spc: *Spartina cynosuroides*) in the four sites with increasing marsh loss (sites 1 to 4 indicated on the X-axes). The numbers do not show consistent trends with increasing marsh loss except for *Distichlis spicata*.

Figure 5: Proportion of mixtures in each field site with increasing marsh loss (sites 1 to 4 indicated on the X-axis). The total number of mixtures results in a similar trend as the skewness indicator (see Fig. 2). sa: Schoenoplectus americanus, spa: Spartina alterniflora, spp: Spartina patens, dsp: Distichlis spicata and spc: Spartina cynosuroides

Tables

Table 1: vulnerability indices with short explanation from Raposa et al. (2016) and Cole Ekberg et al. (2017).

Indices from (Raposa et al., 2016)
Percentage of marsh below Mean High Water level (MHW)
Percentage of marsh below lowest third of plant distribution
Elevation skewness (see equation 1)
Elevation change rate (mm yr^{-1})
Short-term accretion rate (mm yr^{-1})
Long-term accretion rate (mm yr^{-1})
Turbidity of the water (NTU)
Tidal range (m)
Long-term rate of Sea Level Rise (SLR) (mm yr^{-1})
Short-term inter-annual variability in water levels (mm)
Indices from Cole Ekberg *et al.* (2017)
Median marsh orthometric height (m NAVD88)
Median marsh elevation relative to MHW
Modelled marsh loss with 0.3, 0.9 and 1.5 m of SLR with the Sea Level Affecting Marshes Model (SLAMM)
Loading Response: the depth that the PVC tube and penetrometer (4.6 kg) sink into the marsh when placed on its surface,
Penetration depth: the depth of penetration of the PVC tube after five blows from the hammer attached to the penetrometer.
Percentage unvegetated marsh area
Percentage *S. alterniflora* monoculture (low marsh)
Percentage turf grass type I (irregularly flooded, dominated by *Spartina patens*, *Distichlis spicata, Juncus gerardii*, with no *S. alterniflora*)
Percentage of perennial turfgrass type II (identical to type I, but *S. alterniflora* may be present but not dominant.

Table 2: vulnerability indices according to (Raposa et al., 2016)* show increasing vulnerability from field site 1 to site 3 and again a decrease in vulnerability in site 4. A lower MARS score means higher vulnerability to SLR.

Colorscale	Resilient				Vulnerabl
Field site		1	2	3	4
% below MHW		79.7	96.3	93.9	86.8
% below third elevation		4.5	18.5	34.1	5.3
Skewness		-1.11	-0.12	0.49	-0.02
Average Marsh Elevation index		3.6	3	2.6	3.3
Tidal range (m)		0.63	0.31	0.20	0.06
MARS average score		2.5	2	1.5	2

*Not included: Elevation change rate, short-term and long-term accretion rate, turbidity, long-term rate of SLR, short-term inter-annual variability in water levels.

Table 3: vulnerability indices according to (Cole Ekberg et al., 2017)* show increasing vulnerability from field site 1 to site 3 and again a decrease in vulnerability in site 4.

Field site	1	2	3	4
Median marsh orthometric height (m NAVD88)	0.41	0.19	0.12	0.12
Median marsh elevation relative to MHW	-0.06	-0.12	-0.11	-0.06
Percentage unvegetated	1.55	11.42	33.25	58.24
Percentage *S. alterniflora* monoculture (low marsh)	23.42	9.88	34.15	15.79
Percentage turf grass type I (high marsh)	47.75	2.47	9.15	11.40
Average score (1 vulnerable – 4 resilient)	3.8	2.6	2	2.6

*Not included: average S. alterniflora height, Modelled loss with 0.3, 0.9 and 1.5 m of SLR, soil penetration depth and loading response, percent of perennial turfgrass type II.

Table 4: proposed indices to be include to assess long-term marsh vulnerability.

Field site	1	2	3	4
Average fetch length for all pond points (m)	4.3	68.3	60.5	281.0
Ratio Unvegetated/vegetated marsh area	0.02	0.14	0.52	1.37
Long-term score (1 vulnerable – 4 resilient)	4	2.5	2.5	1