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RESEARCH ARTICLE

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How grazing management can maximize erosion resistance of salt marshes

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Abstract

1. Combining natural saltmarsh habitats with conventional barriers can provide a sustainable and cost-effective alternative for fully engineered flood protection, provided that a minimal salt marsh width can be guaranteed for a long period. Hence, it is essential to understand both the key factors and management options driving the lateral erodibility/stability of salt marshes.
2. We aimed to determine how salt marsh management (i.e. grazing by large vs. small grazers vs. artificial mowing), marsh elevation and marsh age affect soil stability (i.e. soil collapse) and intrinsic lateral erodibility of salt marshes (i.e. particle-by-particle detachment). Soil cores were collected in high and low marshes (above and below 0.5 m MHWL, respectively) of different ages. At these locations, we compared cores from grazed areas to cores inside grazer exclosures, with and without artificial mowing. All cores were exposed to waves in flumes to test their stability and lateral erodibility.
3. All soil cores were characterized by a stable fine-grained layer deposited on top of readily erodible sand. The thickness of the fine-grained layer was a key parameter in reducing salt marsh instability (cliff collapse). This layer thickness increased with marsh age and at lower elevations, but decreased with cattle grazing due to compaction.
4. The erosion resistance of the fine-grained layer increased with (a) large grazers that compacted the soil by trampling, (b) mowing that excluded soil-bioturbating species, and (c) grazing by small grazers that promoted vegetation types with higher root density.
5. *Synthesis and applications.* Overall, marshes with thinner cohesive and/or fine-grained top layers are more sensitive to lateral erosion than marshes with deep cohesive soils, independently of the management. Grazing and artificial mowing can reduce the erodibility of fine-grained soils, making salt marshes more resilient to lateral erosion. However, compaction by large grazers simultaneously leads to thinner fine-grained layers and lower elevation, potentially leading to more inundation under sea-level rise. Hence, to effectively manage salt marshes to enhance their contribution to coastal protection, we recommend (a) moderate/rotational

livestock grazing, avoiding high intensity grazing in sediment-poor systems sensitive to sea-level rise and (b) investigating measures to preserve small grazers.

KEYWORDS

cliff erosion, coastal protection, ecosystem-based coastal defence, grazing management, salt marsh, soil stability, wave mesocosm

1 | INTRODUCTION

Many countries around the world are currently facing the challenge of flood risk due to sea-level rise, land subsidence and frequent storm surges (IPCC, 2014; Syvitski et al., 2009; Vousdoukas et al., 2018). The adoption of hard engineering solutions is associated with the destruction of the coastal ecosystems, and related loss of the ecosystem services (e.g. Lai et al., 2015). Ecosystem-based coastal defence may offer a more sustainable approach towards coastal protection, by combining hard engineering with natural coastal ecosystems into hybrid designs (Schoonees et al., 2019; Temmerman et al., 2013). A recent analysis of historic flood disasters revealed that salt marshes in front of a dike reduce both the chance of a dike breaching and the impact of a dike breach (Zhu et al., 2020). Despite being increasingly recognized for their coastal protection services (Gedan et al., 2009; Shepard et al., 2011; Temmerman et al., 2013), implementing the use of salt marshes into hybrid ecosystem-based flood-designs is hampered by the lack of in-depth understanding of their long-term dynamics (Bouma et al., 2014).

Salt marshes are dynamic ecosystems that vary in width in time, either due to intrinsic biogeomorphic and ecological processes or by changes in physical factors such as storms and sea-level rise (Phillips, 1986; Van de Koppel et al., 2005). With climate change, the frequency and intensity of storms and storm surges are expected to increase (IPCC, 2014). Extreme storms may induce the formation of a marsh cliff (Bouma et al., 2016), where after especially frequently occurring moderate (winter) storms determine the rate of lateral retreat (Leonardi et al., 2016). Landward migration of marshes, for example, to keep pace with sea-level rise, is often prevented due to human land use. This may eventually result in the complete loss of a marsh by coastal squeeze (Doody, 2013). Hence, understanding the susceptibility of salt marshes to this type of edge erosion (i.e. known as cliff erosion, scarp erosion or lateral erosion, also found in marsh creeks: Eerdt, 1985; Pagés et al., 2018; Sharma et al., 2016) is of key importance for being able to integrate marshes as sustainable flood defence strategies.

The susceptibility of marsh edges to lateral erosion can be divided into soil stability and lateral erodibility. Soil stability refers to marsh-edge collapse due to undercutting by wave erosion (Francalanci et al., 2013; Priestas et al., 2015; Schwimmer et al., 2001). Lateral erodibility refers to the gradual detachment of soil particles from the sediment due to hydraulic pressure (Bouma et al., 2009; Feagin et al., 2009). Previous studies showed that sandy soils erode faster than fine-grained soils (De Battisti et al., 2019; Feagin et al., 2009; Lo et al., 2017). Below-ground plant structures (roots and rhizomes) also were found to decrease erodibility (De Battisti et al., 2019; Ford et al., 2016; Wang et al., 2017), with the strongest effects found in sandy soils (De Battisti et al., 2019; Lo et al., 2017). Coarse detritus including roots and decaying plant parts seem to promote erosion while fine-grained detritus may reduce erosion (Feagin et al., 2009).

Grazing is often used for managing biodiversity in salt marshes (Bakker, 1989; Davidson et al., 2017), but is known to affect marsh properties in many ways. For example, trampling by large grazers increases soil bulk density and root density by soil compaction (Elschot et al., 2013, 2015; Howison et al., 2015; Schrama et al., 2013). A study by Pagés et al. (2018) using an overshot-weir flume related the higher compaction of the trampled soil by cattle with less lateral erosion. On the other hand, ungrazed soils may be colonized by tall plant communities and *Orchestia* sp. (Friese et al., 2018; Olff et al., 1997), a bioturbator that has been related to higher soil porosity (Howison et al., 2015), which in return could increase erodibility. In addition, small grazers have also shown to structure salt marsh vegetation communities (Chen et al., 2019; Kuijper & Bakker, 2005), although previous studies show no significant effect on soil properties (Elschot et al., 2013). On sandy barrier islands, older marshes and low marshes (below mean high water level) may have deeper fine-grained soil layers due to increased accretion (Elschot et al., 2013; Olff et al., 1997), which may increase salt marsh erodibility resistance and stability. An integrated view on how grazing management in combination with abiotic factors, such as marsh age, marsh elevation and sediment layering, affect the susceptibility of marsh edges to lateral erosion remains however lacking.

To facilitate the use of salt marshes as part of a nature-based flood protection, we aimed to enhance the understanding on how soil stability and erodibility processes are affected by management in relation to the physical setting of a marsh. The aims of this study hence are (a) to assess how salt marsh management (including grazing by small and large grazers and artificial mowing), elevation and age affect relevant biotic and abiotic factors related to sediment erosion, (b) how these changes affect the lateral erosion in the studied system and (c) discuss the implications of the results on the management of the salt marshes for coastal protection.

2 | MATERIALS AND METHODS

2.1 | Study area

The study sites were located in the back-barrier marsh of the island of Schiermonnikoog (the Netherlands; Figure 1a, 53°29′N 6°14′E). This island is expanding to the east, creating a salt marsh chronosequence (Olff et al., 1997). The western and oldest marsh has been grazed by large grazers (cattle) until 1958, ungrazed between 1958 and 1972 and grazed again thereafter (Bakker & de Vries, 1992). The eastern younger marshes have been only grazed by small grazers (hare and geese) but never by cattle (Figure 1a; Kuijper & Bakker, 2005). The tidal range is approximately 2.5 m and mean high tide (MHT) is 1 m.

2.2 | Experimental set-up

To study the effect of management, salt marsh age and salt marsh elevation on soil properties and how this is related to marsh stability and lateral erodibility, soil samples were collected from six locations along the back-barrier marsh of Schiermonnikoog (Figure 1a). Because this salt marsh is a well-protected National Park, human impact is not expected to have an effect on the marsh, neither close nor far from build-up area. In 2018, the ages of each study site were approximately 23, 38, 53, 63, 128 and 200 years (Olff et al., 1997). In 1972, large-grazer exclosures (8 m × 42 m) were deployed in the high (above 0.5 m + MHWL) and low (below 0.5 m + MHWL) 200-year-old salt marsh, excluding large grazers but not small grazers (Bos et al., 2002). Artificial mowing treatments were established inside the exclosures to test the effect of grazing (above-ground removal) without trampling (Veeneklaas et al., 2011). The vegetation was mown twice annually (June–July and August–September; Chen et al., 2020). One exclosure in the high marsh and one in the low marsh were sampled inside in the non-mowed area, with dominant *Elytrigia atherica*; inside in the mowed area, with *E. atherica* mixed with other species such as *Festuca rubra* and *Plantago maritima*, hereafter called mixed *E. atherica*; and outside in the trampled soil with *Juncus gerardii*.

In 1994, small-grazer exclosures (7 m × 7 m) were deployed in the low and high marsh in each of the five youngest stages, not grazed by cattle (further details in Kuijper & Bakker, 2005). Samples were collected in the high marsh outside and inside of the exclosure, both dominated by *E. atherica*; in the low marsh outside the exclosures with mixed species with predominance of *Atriplex portulacoides*, hereafter called mixed *A. portulacoides*; and inside the exclosure with *E. atherica* (excluding stages 23 and 63 with mixed *A. portulacoides*). Therefore, the treatments consisted of soil samples from high and low marshes grazed and ungrazed by small grazers at the stages 23-, 38-, 53-, 63- and 128-year-old; and high and low marsh grazed, ungrazed and mowed at the stage 200-year-old from the cattle grazed area (Figure 1b).

2.3 | Salt marsh soil stability and lateral erodibility

Soil samples were collected during April 2018. A total of 78 sediment cores of 15 cm diameter and 20 cm depth were extracted from the 6 locations (Figure 1a,b), with 3 replicates per treatment, by inserting PVC pipes in the soil and carefully digging them out. The roots sticking out of the bottom of the core were cleanly cut and a plastic lid was placed below to keep the sediment intact. Following the protocol of Lo et al. (2017), sediment cores were transported to the wave

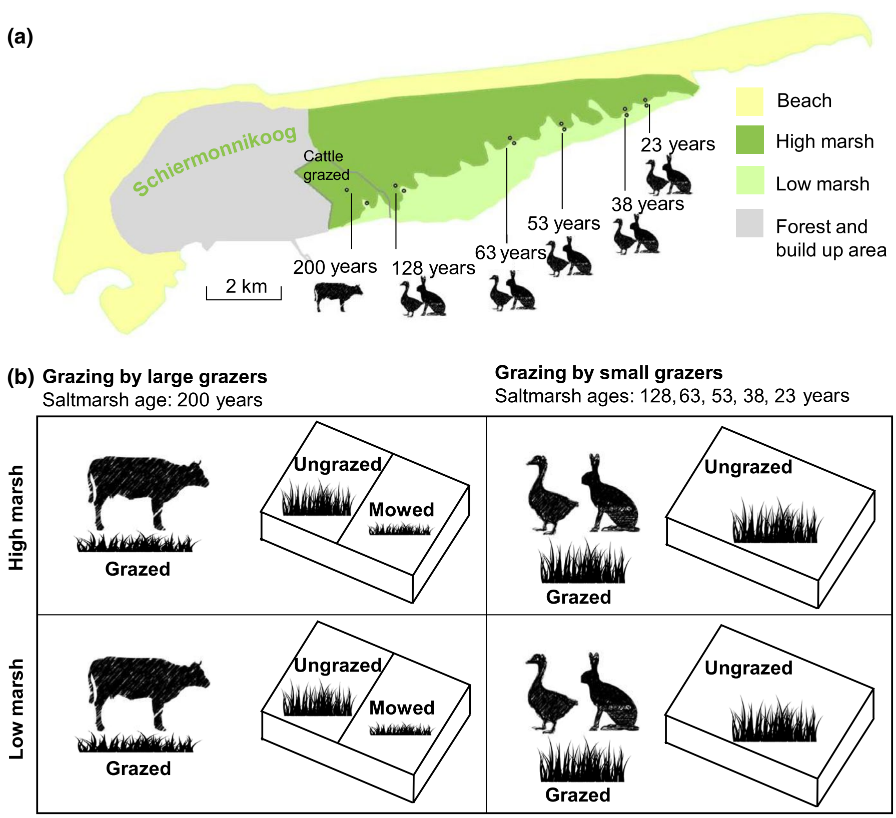

FIGURE 1 (a) Locations of the sampling sites in the island of Schiermonnikoog (b) Experimental treatments in the different salt marsh ages and elevations

mesocosms installations in the Royal Netherlands Institute for Sea Research (NIOZ) in Yerseke, the Netherlands, and stored in tanks with seawater from the Oosterschelde estuary until the erosion experiments. The elevation of each plot was determined with DGPS measuring 5–8 paired points inside and outside the exclosures.

Soil stability and lateral erodibility were determined in four wave flumes of 360 cm long × 90 cm high × 82 cm wide at the Royal Netherlands Institute of Sea Research with the same set-up as Lo et al. (2017) and Wang et al. (2017; Figure 2a). The same wave settings as in Lo et al. (2017) were utilized, with wave heights of ~8 cm in front of the cores, and the swash intersecting the whole sample. The sediment cores taken in the field were transferred into the metal cores which had an opening of 12 cm width in the longitudinal side. The exposed lateral side was cleanly sliced into a flat surface in the opening of the metal core and adjusted to 15 cm high (Figure 2b). Lateral erodibility from the fine-grained layer was measured after 1, 2, 4, 8, 16, 24, 32 and 38 hr of wave exposure. Soil stability was measured as the time when the fine-grained layer collapsed forward or slid to the bottom after the erosion of the sand layer (Figure 2c). Sediment collapsing was considered an artefact for the volume loss calculation and these data were excluded from the lateral erodibility analyses. The volume eroded was assessed by photogrammetry (structure from motion) technique as developed in Nieuwhof et al. (2015), and applied by Lo et al. (2017) and Wang et al. (2017). Some samples with sand layer on the bottom did not collapse even with the sand eroded, and the window to calculate the volume loss was adjusted to the fine-grained layer.

2.4 | Below-ground vegetation properties

Below-ground biomass was cleaned over a 1-mm sieve after the wave exposure and the roots and rhizomes were separated. After sieving the sediment, we soaked the remaining roots in a bucket with water and poured the floating roots into another bucket while the small sediment aggregates remained in the bottom and were discarded. A representative subsample of (a) rhizomes, (b) coarse roots (diameters between 0.5 and 2 mm for the plant species of this experiment) and (c) fine roots (diameters <0.5 mm), which include some fine dead root material that could not be distinguished and removed (cf. De Battisti et al., 2019), was separated from each sample. The subsamples of rhizomes and coarse roots were displayed in a calibrated red tray and photographed with a compact camera (model Nikon Coolpix W300). The subsamples and the rest of the sample were dried at 60°C until constant weight to obtain the biomass (g). The biomass of coarse roots and fine roots was extrapolated from

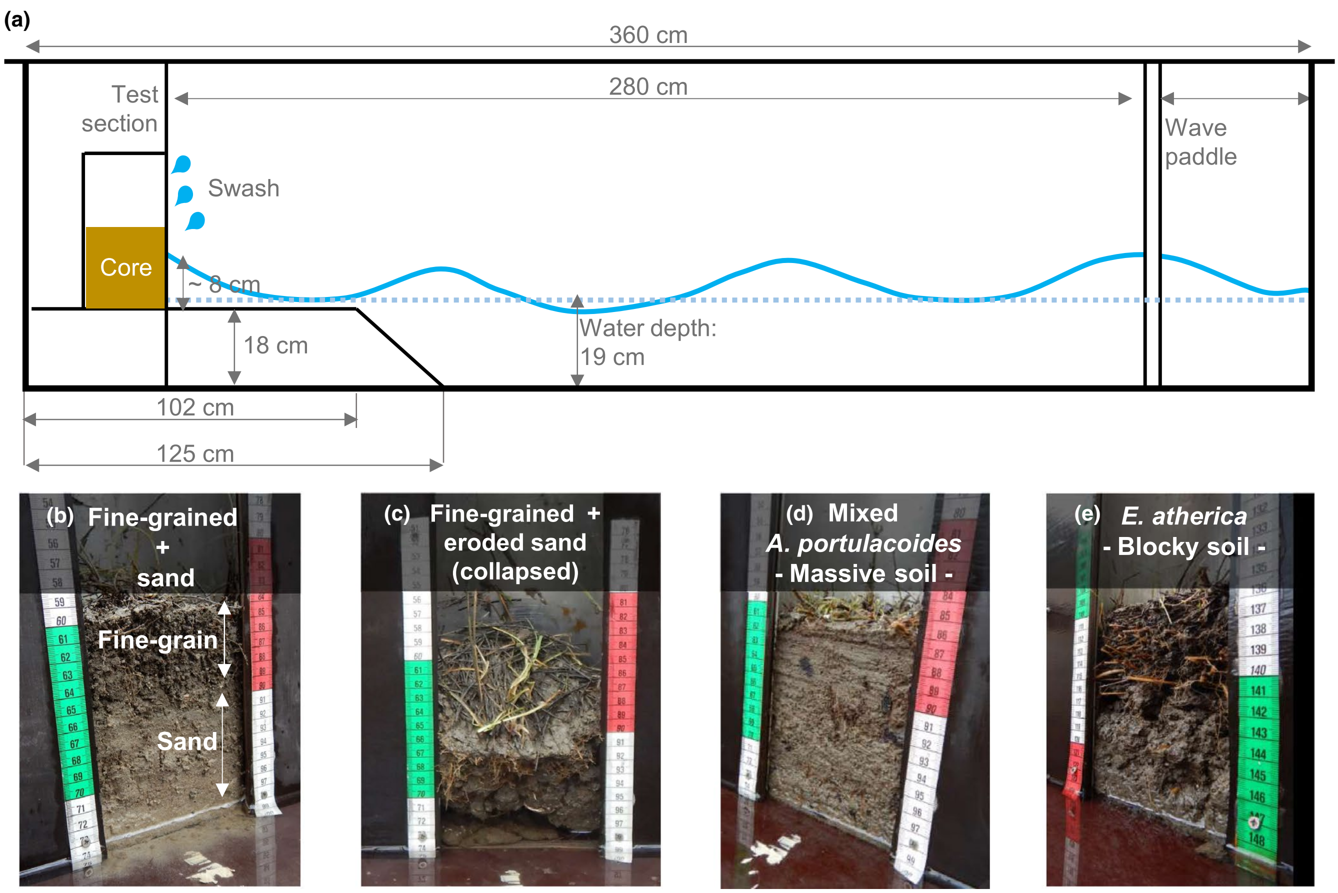

FIGURE 2 (a) Schematic side view of the wave mesocosms, (b) sample with fine-grained layer on top of sandy layer, (c) collapsed sample after the sand layer eroded, (d) soil structure with mixed *Atriplexportulacoides* and (e) soil structure with *Elytrigia atherica*

the dry weights of the subsamples and the dry weight of the whole sample. Diameters of the rhizomes and coarse roots were measured from the images of the subsamples with the software ImageJ (Schneider et al., 2012). Total root density, coarse roots density, fine roots density and rhizome density were calculated as in Table S1.

2.5 | Soil properties

Soil properties were determined from small cores of 2.2 cm Ø and 20 cm depth collected next to each 15 cm Ø sediment cores taken for the erosion experiment. After measuring the length of the fine-grained layer depth, the small cores were sliced separating the fine-grained layer from the sand layer to determine the properties of each layer separately. Sediment samples were weighed and freeze-dried during 4 days. Bulk density was calculated as the sediment dry weight in a known volume (g/cm^3). Water content was calculated as the difference of wet and dry weight. Organic content was determined by lost on ignition method after burning the sediment sample for 6 hr at 450°C. The sediment grain size was analysed using a Malvern® Mastersizer 2000. Soil strength at 5 cm depth (Mpa) was measured in the field with an Eijkelkamp Penetrologger with a 1 cm Ø cone. Deeper measurements of soil strength were not utilized because the intersection with the sand layer biased the results. Soil structure was determined visually from the soil cores and categorized into granular, blocky, massive or single grained, based on the Soil Survey Manual (Soil Science Division Staff, 2017).

2.6 | Bioturbators

The presence and absence of the bioturbator *Orchestia gammarellus* was noted in all the sampling locations by superficial inspection of the area where the cores were extracted and within the core.

2.7 | Data analysis

The lateral erodibility of the fine-grained layer was only studied for the samples that did not collapse (all the samples from the large-grazers area, the low marsh samples from the small-grazers area and the high marsh samples from stage 63). Estimates of the possible maximum volume loss over time (E_{max}) were calculated fitting a saturating-type function (Appendix S1). Full-factorial, well-replicated design was limited by the layout of successional stages and long-term grazing treatments in the field. For this reason, data were analysed separately for the large grazers and small grazers. In addition, because vegetation communities varied between successional stages in unreplicated way, below-ground biomass properties were primarily utilized as variables to relate to erosion. Principal component analysis (PCA) without rotation and Pearson correlation matrices were performed to find relationships between E_{max}, vegetation and soil properties. All the variables were log-transformed to account for nonlinear relationships. Because of high correlation among vegetation and soil variables, bulk density, soil structure and total root density were used in the further analysis as representative of the soil and vegetation properties. For the cattle-grazed area, we used a full-factorial two-way ANOVA to test the effect of management and elevation on a set of response variables: the fine-grained layer depth (cm), maximum volume loss (%), bulk density and root density. For the small-grazers area, the analysis was done for high and low marsh separately. We first used a two-way ANOVA to test the effect of grazing by small grazers and age on the set of response variables (the fine-grained layer depth [cm], maximum volume loss [%], bulk density and root density). Because the interaction of grazing and age was not significant, and to obtain an overall effect of grazing on the set of response variables, we followed the analysis with a linear mixed model (LMM) to test the effect of management (as fixed factor) on the set of response variables with age as random factor. One-way ANOVAs were used to test the effect of age on the small-grazers area on the set of response variables. The same analysis for large and small herbivores separately was done to test the effect of soil structure on the maximum volume loss (%) as response variable. In addition, for the large-grazers area, two-way ANOVA was used to test the effect of (a) soil texture and elevation on bulk density (response variable) and (b) vegetation type and elevation on root density (response variable). Similarly, for the small-grazers area, we used one-way ANOVA to test the effect of (a) soil texture on bulk density and (b) vegetation type on root density. Where necessary, the response variables were log-transformed to meet normality and homogeneity assumptions. To test differences in soil elevation between treatments, *T* test was performed among the large-grazer treatments, and Paired Wilcoxon signed-rank test was performed between the small-grazer treatments due to intrinsic variability of the topography. Statistical analyses were performed using R 3.5.0 (R Development Core Team, 2018). The DRC (Ritz et al., 2015) package was used for the saturating-type function.

3 | RESULTS

3.1 | Lateral stability and fine-grained layer erodibility across different managements, elevations and ages

All the samples from high marsh with a fine-grained sediment layer less than 12.5 cm (i.e. all but stages 63 years and the 200 years large-grazers area) collapsed within 2 hr of wave exposure, when the sand below was completely eroded (Figure 3a). We did not observe collapse in any of the other locations, further indicating that the collapse threshold indeed occurred around with a fine-grained layer <12.5 cm (Figure 3a,b). The properties of the fine-grained layer and sand layer are found in Tables S2 and S3. Sandy soils eroded with any type of vegetation and management. In contrast, the fine-grained layer only eroded up to 9.4% during the experimental period. In the large-grazers area, erodibility was significantly lower in the grazed treatment (~0%), followed by the mowed, compared to the ungrazed (up to 9.4%; $F_{2,12} = 61.7$, $p < 0.001$), without effect nor interaction of marsh elevation ($F_{2,12} = 0.2$, $p > 0.05$ and $F_{2,12} = 0.3$, $p > 0.05$, respectively; Figure 3c; Figure S2a). Fine-grained layer erodibility in the small-grazers area was significantly lower in the grazed (~1%) than ungrazed (up to 6%; $\chi^2(1) = 12.28$, $p < 0.001$), although we can observe the exceptions at stages 23 and 63 years (Figure 3c; Figure S2b). Erodibility differed significantly among locations but not in relation to age ($F_{5,29} = 10.7$, $p < 0.001$).

3.2 | Effect of soil properties and vegetation characteristics on the fine-grained layer erodibility

Both in the large- and small-grazers area, fine-grained layer erodibility was significantly reduced by soils with higher bulk density, higher soil strength and higher below-ground biomass (Figures 4 and 5; Figures S3 and S4). Additionally, lower silt % and soil water content in the large-grazers area and higher soil water content in the small herbivores area were also associated with less erosion (Figures 4 and 5; Figures S3 and S4). Organic content for both areas and rhizome density and silt % in the small-grazers area were the only variables not correlated to erosion (Figures 4 and 5). Erodibility also differed significantly between massive and both granular and blocky soil structure, but not between the latter two (large-grazers area: $F_{2,12} = 61.7$, $p < 0.001$, small-grazers area: $\chi^2(2) = 73$, $p < 0.001$; Figures 4 and 5). Massive structure appeared with *J. gerardii* and mixed *A. portulacoides*, platy structure with mixed *E. atherica* and blocky structure with *E. atherica*. Granular structure was the exception, present with both mixed *A. portulacoides* and *E. atherica* at stage 53 years. Below-ground biomass was mainly explained by the root density, specifically by the compartment of the fine roots, and less by the rhizomes or coarse roots (Tables S4 and S6). For the sake of simplicity, bulk density, soil structure and total root density were used in the further analysis as representative of highly correlated soil and vegetation properties. Therefore, similar relations are expected for all these variables. Total root density was higher in *J. gerardii* (trampled by cattle) and mixed *A. portulacoides* compared to

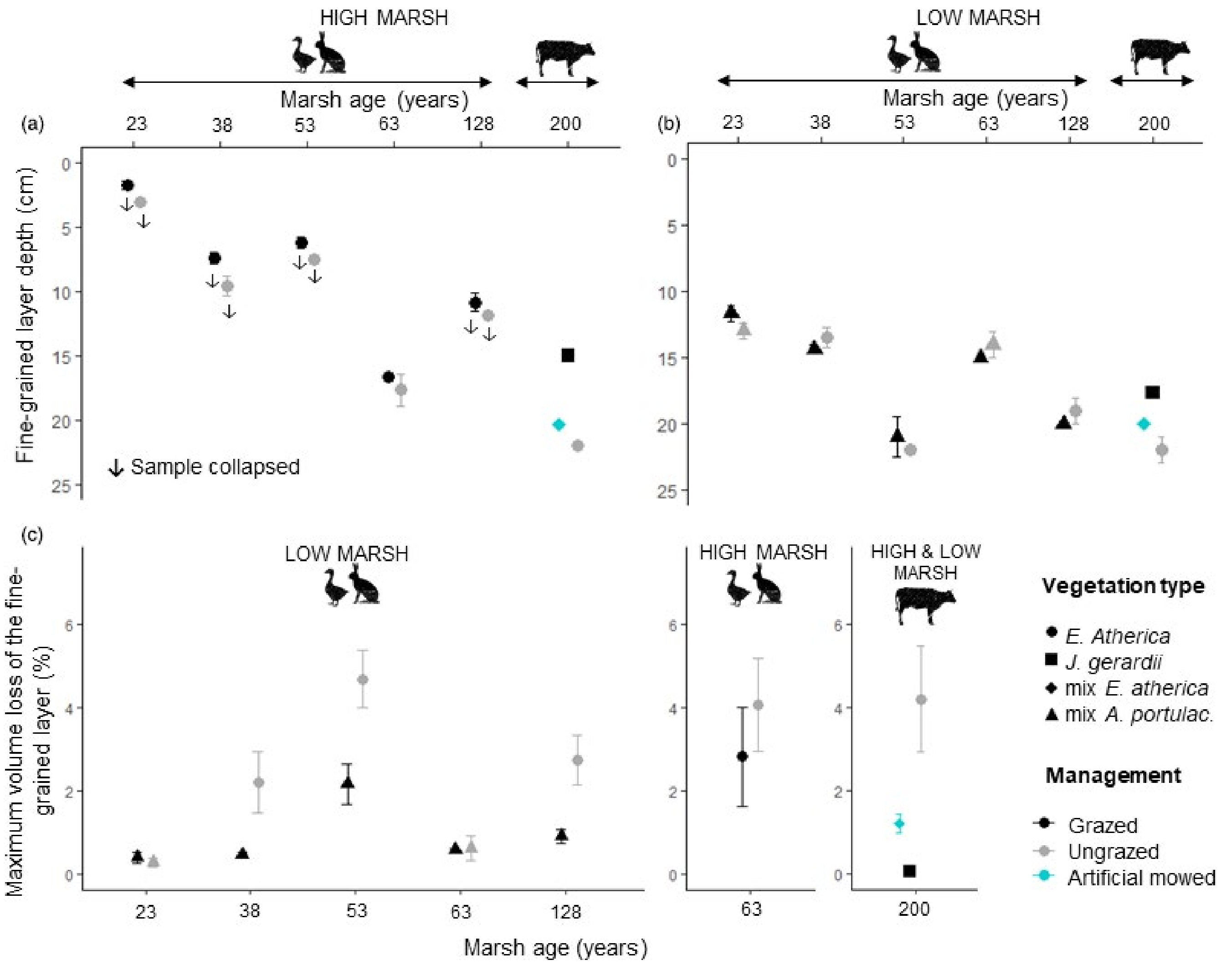

FIGURE 3 Fine-grained layer depth for each treatment in the (a) high marsh and (b) low marsh. (c) Maximum volume loss of the fine-grained layer for each treatment. Error bars represent standard errors

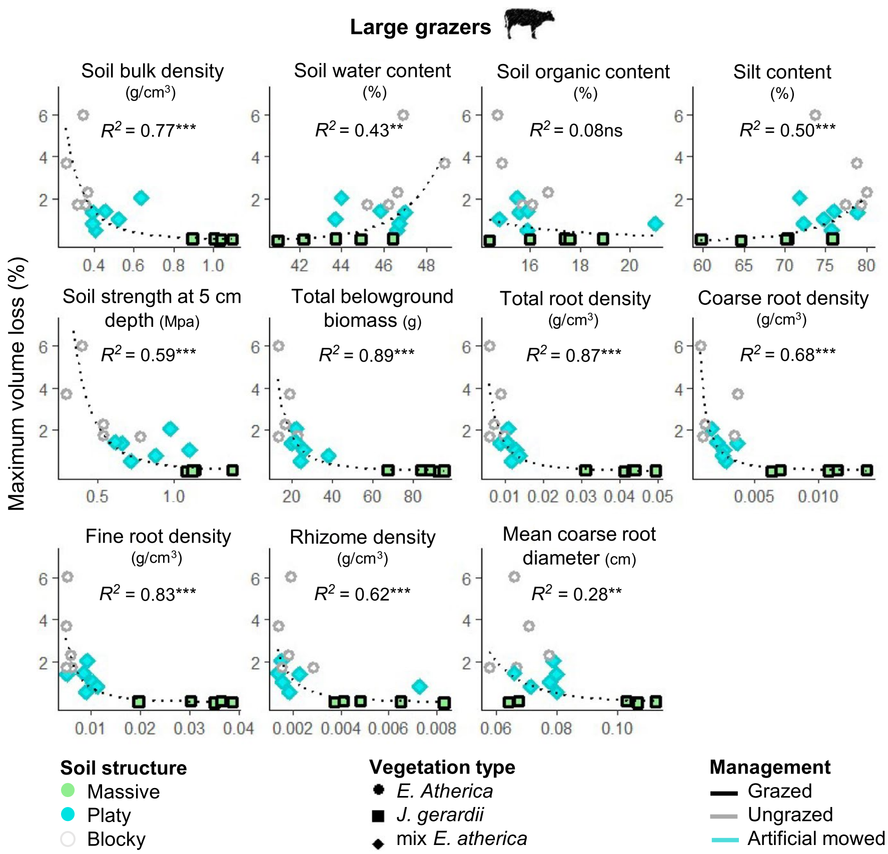

FIGURE 4 Relationships between maximum volume loss (%) and the soil and vegetation variables in the large-grazers area. R^2 and significance values were obtained from power-law regressions ($y = ax^b$). Significance codes refer to $p < 0.001$ (***), $p < 0.001$ (**), $p > 0.05$ (ns)

E. atherica (large-grazers area: $F_{2,12} = 132$, $p < 0.001$; small-grazers area: $F_{1,3} = 95.2$, $p < 0.001$, respectively; Figures 4 and 5). Bulk density was significantly higher in massive soils (Figure 2d) compared to platy, granular (Figure S1) and blocky soil structure (Figure 2e), but did not differ between the latter two (large-grazers area: $F_{2,12} = 44.9$, $p < 0.001$; small-grazers area: $F_{2,31} = 16.2$, $p < 0.001$; Figures 4 and 5). Granular structure from stage 63 years explained the high erodibility even with mixed vegetation, which cannot be explained by any other variable (Figure 5).

3.3 | Effects of management, elevation and age on soil properties, vegetation characteristics and bioturbators

The effect of management, elevation and age on fine-grained layer depth was investigated in relation to the lateral stability. Grazing by large grazers reduced the fine-grained layer depth followed by the mowed treatment, compared to the ungrazed ($F_{2,12} = 102.6$, $p < 0.01$), being ~2 cm deeper in the low marsh ($F_{2,12} = 10.7$, $p < 0.01$; Figure 3a,b). In the small-grazers area, the fine-grained layer was up to 15 cm deeper in the low marsh and tended to increase with age; however, these changes were not completely linear along the age gradient (high marsh: $F_{4,25} = 129$, $p < 0.001$; low marsh: $F_{4,25} = 54.6$, $p < 0.001$; Figure 3a,b). The fine-grained layer depth was overall significantly thinner in the small herbivores grazed treatment in the high marsh ($\chi^2(1) = 15.9$, $p < 0.001$), but not in the low marsh ($\chi^2(1) = 0.03$, $p > 0.05$; Figure 3a,b). The compaction of the fine-grained layer by large grazers was related to lower soil elevation, with grazed soils in average 5 cm lower than ungrazed in the high marsh and 2 cm in the low marsh ($t_{17} = 5.1$, $p < 0.01$ and $t_{21} = 2.7$, $p < 0.05$, respectively) and up to 7 cm lower than the artificial mowed in the high marsh and 2 cm in the low marsh ($t_2 = 12$, $p < 0.01$ and $t_6 = 1.3$, $p > 0.05$, respectively). Small grazers only reduced the elevation significantly in the low marsh at stage 23 years by on average 1.3 cm ($Z = 3$, $p < 0.05$).

The effect of management, elevation and age on soil structure, soil bulk density, total root density and bioturbators, as representative soil and vegetation variables, was investigated in relation to the fine-grained layer erodibility. Large grazers led to the highest bulk density and root density with massive soil structure, followed by the mowing treatment with platy structure, compared to the ungrazed with *E. atherica* and blocky structure ($F_{2,12} = 44.9$, $p < 0.001$, $F_{2,12} = 132$, $p < 0.001$, respectively; Figure 4). Grazing by small herbivores indirectly affected the root density, bulk density and soil structure by suppressing *E. atherica* with exception of stages 23 and 63 years with mixed *A. portulacoides* inside the exclosure, and stage 53 years with granular structure (Figure 5; Table S2). In the large-grazers area, bulk density and root density did not differ between the high and low marsh ($F_{1,12} = 2.5$, $p > 0.05$ and $F_{1,12} = 0.8$, $p > 0.05$, respectively). In the small-grazers area, bulk density differed significantly among locations, but not in relation to age ($F_{4,30} = 10.3$, $p < 0.001$) and root density was not affected by age ($F_{4,30} = 0.6$, $p > 0.05$). Lastly, large grazers and mowing suppressed the presence of bioturbators, except for one mowed sample, and it may be related to the differences in bulk density and soil

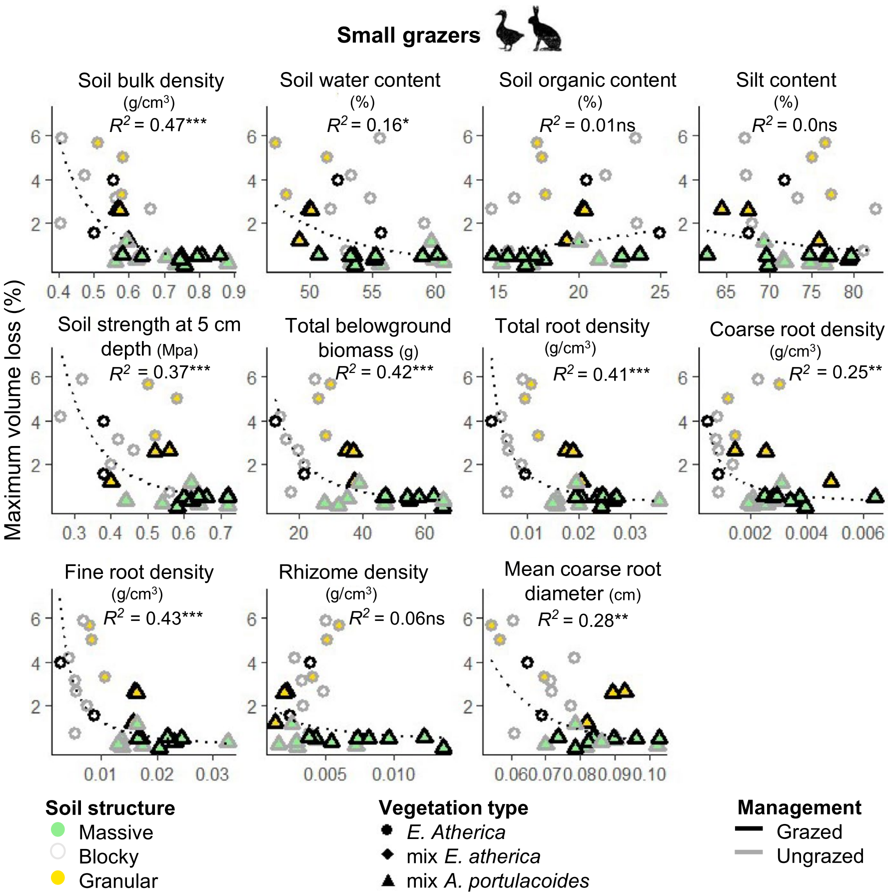

FIGURE 5 Relationships between maximum volume loss (%) and the soil and vegetation variables in the small-grazers area. R^2 and significance values were obtained from power-law regressions ($y = ax^b$). Significance codes refer to $p < 0.001$ (***), $p < 0.001$ (**), $p < 0.01$ (*), $p > 0.05$ (ns)

structure in the large-grazers area (Table S5). In the small-grazers area, bioturbators did not have an effect on erosion as they were found both in grazed and ungrazed sites (Table S5).

4 | DISCUSSION

Ecosystem-based coastal defence, combining traditional hard engineering with natural coastal ecosystems such as salt marshes, may offer a more sustainable approach towards future flood risk prevention (Temmerman et al., 2013). Hence, it is essential to understand key factors driving the lateral erodibility and stability of salt marshes to optimize their coastal protection management. We here investigated how salt marsh management can be used to optimize the erosion resistance of salt marshes and how it interacts with marsh age and elevation. In this study, we found that erosion-resistance of fine-grained soils can be increased by management that enhances (a) large grazers, which compact the soil by trampling, (b) mowing, which excludes soil-bioturbating arthropods and (c) small grazers (hares, geese) which promotes vegetation types with higher below-ground biomass, mostly explained by higher root density. More specifically, we observed faster erodibility for fine-grained soils with low bulk density (<0.6 g/cm^3), in combination with blocky or granular soil structure and low root density (<0.015 g/cm^3), as typical for *E. atherica* vegetation zones. However, bulk density should be used with caution as a proxy for erodibility, because soils with the same bulk density can have different grain size and soil structure, and therefore different erodibility. Furthermore, the stability of the fine-grained soils depended on the thickness of the fine-grained layer, which generally increased in sandy barrier islands with marsh successional age and at lower elevations, but decreased with cattle grazing. Sandy marshes with thin cohesive layers will still provide coastal protection (e.g. wave attenuation; Shepard et al., 2011), but their lateral erosion rate may be higher than in marshes with deeper cohesive soils. These results can be used to formulate recommendations to optimize the management of salt marshes for ecosystem-based-coastal-protection (Figure 6).

4.1 | Ecosystem services provided by grazing management: Increased soil resistance by livestock grazing as a new management application

Livestock grazing has been traditionally used as a management tool for preserving plant biodiversity in grasslands and salt marshes world-wide (Davidson et al., 2017; Olff & Ritchie, 1998). In this study area, grazing has increased the plant diversity in the long term by suppressing *E. atherica* dominance (Chen et al., 2020; Kuijper & Bakker, 2005). Furthermore, our experiment shows that the effect of suppressing *E. atherica*, together with the compaction of the soil and exclusion of bioturbators, can reduce the fine-grained soil erodibility. In the case of mowing for promoting plant diversity, the

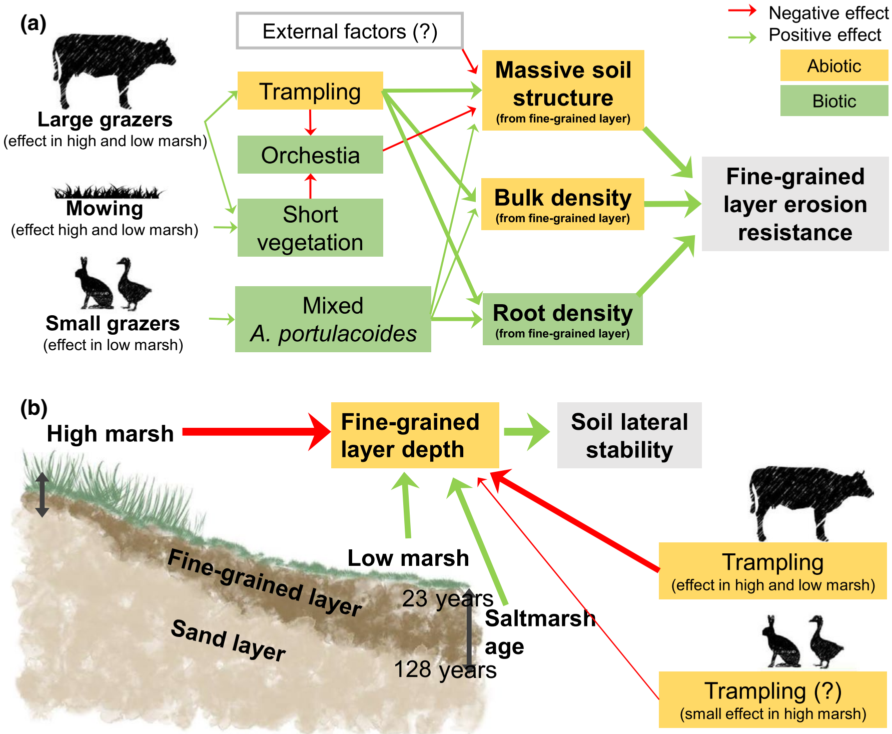

FIGURE 6 Conceptual model of the variables analysed in this study that can affect (a) the fine-grained layer erodibility (i.e. gradual erosion) and (b) lateral soil stability (i.e. cliff collapse) of salt marshes on sandy barrier islands and spits

reduction on erodibility may be explained by the platy soil structure from past grazing, which has not yet been converted to blocky as in the ungrazed treatment, which is likely due to the low abundance of bioturbators in the mowed areas (Howison et al., 2015, 2017; Schrama et al., 2015).

4.2 | Surprising positive effect by small grazers on soil resistance

Hares and geese grazing can also suppress the dominance of *E. atherica*, but only in the low marsh (Chen et al., 2019; Kuijper & Bakker, 2005). This study shows that the shift from *E. atherica* to vegetation with higher below-ground biomass, higher bulk density and massive soil structure can reduce the erodibility of the fine-grained layer on the low marsh. The importance of high below-ground biomass is in accordance with previous salt marsh studies (Brooks et al., 2020 and references therein). More specifically, the importance of fine root density (<1 or 0.5 mm Ø, depending on the author) is in accordance with previous terrestrial system studies (e.g. Baets et al., 2006, 2007; Li et al., 1991). For these small grazers, we cannot attribute the higher erodibility to bioturbators effects, because they were found in both grazed and ungrazed treatments. The 53-year stage was an exception compared to stages 23, 38, 63 and 128 years, in that it had a more granular soil structure and a higher erodibility. We hypothesize that this was due to the presence of more large detritus (dead below-ground biomass), which we observed but did not quantify. However, the presence of more large detritus has been previously related to increase the soil erodibility (Feagin et al., 2009). In contrast to the cattle grazing, small grazers overall did not cause soil compaction and thus did not reduce the fine-grained layer depth nor soil elevation.

4.3 | Grazing management under global change

Livestock grazing as a management tool for coastal protection should be taken cautiously because it can reduce soil elevation by compaction, as shown in this study and others (Bakker et al., 2020; Elschot et al., 2013). This could promote marsh vegetation mortality in face of sea-level rise, although livestock grazing may only enhance this problem in sediment-poor systems with low accretion rates (Kirwan et al., 2016; Törnqvist et al., 2020). Low sediment input can occur due to changes in the sediment supply (Ladd et al., 2019) or in microtidal and/or organogenic marshes (e.g. found in the United States), where the accretion is mainly due to decay of organic material (Kearney & Turner, 2016). In contrast, accretion rates of minerogenic marshes (common in Europe and south-east USA) are higher and depend on suspended sediment levels in the tidal input (Bakker et al., 2015). Additionally, too high livestock grazing can have negative effects in other trophic levels or soil carbon content in organogenic marshes (Davidson et al., 2017, 2020). Moderate intensity grazing and rotational grazing may be the best solution for reducing soil erodibility without excessive soil compaction as well as maintaining biodiversity at different trophic levels (Bakker et al., 2020; Bouchard et al., 2003; van Klink et al., 2016). Low marshes with low sediment input will benefit from being grazed by natural occurring small grazers (hares or geese) preventing the expansion of *Elytrigia* spp. without decreasing the elevation. However, small grazers like geese increasingly prefer higher fertilized agricultural grasslands over nearby natural salt marshes (Dokter et al., 2018); therefore, we recommend management at the landscape scale to recover small grazers abundance in the natural marshes.

4.4 | Relevance to other coastal regions

Livestock grazing in coastal marshes occurs world-wide, especially in European marshes, Asia and South America while less commonly in North America (Davidson et al., 2017). Using cattle grazing for soil stabilization in organogenic marshes as found in North America needs to be further studied because in addition to be sediment-poor systems, the effect of trampling could reduce even more the elevation and speed up soil decomposition (Davidson et al., 2017; Nolte et al., 2013). The effect of small herbivores through limiting the expansion of *E. atherica* can be applied to other low marshes with dominant *Elytrigia* sp. as found in Europe but may be different in other regions, like America, where the marshes are commonly dominated by taller, less palatable species as *Spartina* spp. (Davidson et al., 2017).

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AUTHORS' CONTRIBUTIONS

B.M.-D., L.L.G., D.v.d.W., T.J.B. and H.O. conceived the ideas and designed the methodology; B.M.-D. collected and analysed the data; H.O. contributed to the data analysis; B.M.-D. led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

DATA AVAILABILITY STATEMENT

Data available via the 4TU. Research data Repository https://doi.org/10.4121/14199176 (Marin-Diaz et al., 2021).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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