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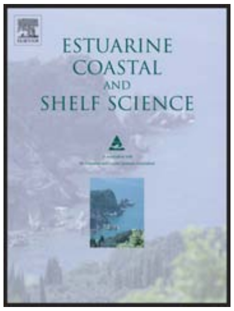

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Title: Does livestock grazing affect sediment deposition and accretion rates in salt marshes?

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Abstract

Accretion rates, defined as the vertical growth of salt marshes measured in mm per year, may be influenced by grazing livestock in two ways: directly, by increasing soil compaction through trampling, and indirectly, by reducing aboveground biomass and thus decreasing sediment deposition rates measured in g/m² per year . Although accretion rates and the resulting surface elevation change largely determine the resilience of salt marshes to sea-level rise (SLR), the effect of livestock grazing on accretion rates has been little studied. Therefore, this study aimed to investigate the effect of livestock grazing on salt-marsh accretion rates. We hypothesise that accretion will be lower in grazed compared to ungrazed salt marshes. In four study sites along the mainland coast of the Wadden Sea (in the south-eastern North Sea), accretion rates, sediment deposition rates, and soil compaction of grazed and ungrazed marshes were analysed using the ^{137}Cs radionuclide dating method. Accretion rates were on average 11.6 mm yr^{-1} during recent decades and thus higher than current and projected rates of SLR. Neither accretion nor sediment deposition rates were significantly different between grazing treatments. Meanwhile, soil compaction was clearly affected by grazing with significantly higher dry bulk density on grazed compared to ungrazed parts. Based on these results, we conclude that other factors influence whether grazing has an effect on accretion and sediment deposition rates and that the effect of grazing on marsh growth does not follow a direct causal chain. It may have a great importance when interacting with other biotic and abiotic processes on the marsh.

1. Introduction

Many coasts of the world show an enhanced rate of sea-level rise (SLR) over the past century, and studies predict it to accelerate in the future (IPCC, 2007; Vermeer and Rahmstorf, 2009). Global SLR was 3.1 mm yr^{-1} between 1993 and 2003 (IPCC, 2007). For the Wadden Sea, a long-term SLR of 1-2 mm yr^{-1} was reported for the last 50 to 100 years while mean high tide (MHT) even increased by 2-2.5 mm yr^{-1} (Oost et al., 2009, citing several authors). However, these rates might be lower if datasets were corrected for the lunar nodal cycle as calculated for the short-term local SLR of the years 1995-2010 (0.7 mm yr^{-1} and 2.3 mm yr^{-1}, with and without correction for the lunar nodal cycle, respectively; Baart et al., 2012). As a consequence of SLR, 5-20% of all global coastal wetlands could be lost until 2080 due to both lateral erosion at the wetlands seaward edge as well drowning, if vertical accretion cannot keep pace with sea level rise (Nicholls, 2004). Among these coastal ecosystems are mangroves (e.g. Krauss et al., 2010), tidal freshwater forests (e.g. Craft, 2012) and salt marshes (e.g. Morris et al., 2002), for example. Salt marshes provide many ecosystem services (Short et al., 2000), such as improving coastal protection by attenuating wave energy (Möller, 2006), sequestering carbon (Callaway et al., 2012), and harbouring a unique flora and fauna (Schmidt et al., 2012).

Given that lateral erosion is not occurring, the resilience of salt marshes to SLR is largely determined by their ability to compensate higher water levels by increased vertical accretion and/or reduced soil subsidence rates leading to increased surface elevation. Only if accretion rates and the resulting increase in surface elevation are higher than rates of SLR, a salt marsh will be able to keep pace with relative SLR. The surface elevation change in salt marshes is the sum of sediment accretion, erosion, compaction processes, and possible regional crustal movements (French, 1993). Marsh accretion, in this context, is defined as the increase in surface elevation relative to a marker horizon or a local measuring device (Cahoon et al., 1995; van Wijnen and Bakker, 2001), but not relative to a fixed benchmark as the surface elevation change (Cahoon et al., 1995; Nolte et al., 2013). It is driven by sediment deposition (g m^{-2} yr^{-1}) that is usually measured over shorter timescales compared to accretion (e.g. French and Spencer 1993; Nolte et al., 2013), but can also be calculated for longer timescales as *e.g.* the total sediment deposition since marsh formation (Elschot et al., in press). The accreted material above and below the marker horizon may be subject to subsidence caused, for instance, by autocompaction (Cahoon et al., 1995; Bartholdy et al., 2010). If accretion and surface elevation change is measured simultaneously, accretion may exceed elevation change, as the latter takes subsidence into account. However, in minerogenic marshes rates of subsidence are usually low (Allen, 2000; French, 2006) and, consequently, differences between surface elevation change and accretion are negligible (French et al., 2003). Many studies have investigated accretion rates in salt marshes (e. g. Cahoon and Turner, 1989; Dijkema, et al. 1990; Dijkema, 1997; Bellucci et al., 2007; Baustian et al., 2012), and several models exists to predict the future development of salt marshes (e.g. Allen, 1990; Temmerman et al., 2003; Bartholdy et al., 2004; French, 2006, Schuerch et al., 2013). Yet, the question of whether accretion rates and the resulting surface elevation change in salt marshes will suffice to outpace SLR is still a point of discussion (e.g. Suchrow et al., 2012).

In general, important factors influencing sediment deposition and accretion rates in tidal marshes on different spatial and temporal scales (French and Spencer 1993) are distance to the sediment source, such as creeks or marsh edges (e.g. Esselink et al., 1998; Reed et al., 1999; Bartholdy et al., 2004), elevation affecting flooding frequency and duration (e.g. Richard et al., 1978; Stoddart et al., 1989; Temmerman et al., 2003), and suspended sediment concentration (SSC) of the inundating water (Kirwan et al., 2010). An important mechanism for the spatial variability of sediment deposition is the reduction of the flow velocity above the vegetated marsh surface (Temmerman et al., 2012), which can lead to increased sediment deposition at sites with higher biomass (Morris et al., 2002) and/or in the vicinity of tidal creeks or marsh edges (Christiansen et al., 2000; Temmerman et al., 2004; van Proosdij et al., 2006).

Whether and how grazing management affects sediment deposition and accretion rates on these marshes has been scarcely studied (e.g. Andresen et al., 1990; Neuhaus et al., 1999; Stock, 2011; Suchrow et al., 2012). Mainland salt marshes at the Wadden Sea coast, located along the south-eastern North Sea, represent about 10% of all European temperate salt marshes (Bakker et al., 1997). Here, livestock grazing for agricultural purposes used to be common (Esselink et al., 2000). Since the 1980s, however, grazing was reduced in many of these salt marshes primarily for nature conservation reasons (Esselink et al., 2009). We expect grazing livestock to influence accretion rates in salt marshes in different ways. Firstly in a direct way, by increasing soil compaction through trampling (Olsen et al., 2011) and thereby reducing accretion. This process could, however, be mitigated, because an increased compaction lowers the marsh elevation, which in turn increases flooding frequency and thus might lead to increased sediment deposition. Secondly, grazing livestock might influence accretion in an indirect way by reducing aboveground biomass (Kiehl et al., 1996), which was found to increase water currents and thus lower sediment deposition in grazed marshes. Furthermore, reduced aboveground biomass reduces the direct sediment capture of vegetation structures.

This study aimed to investigate effects of livestock grazing on the resilience of salt marshes to SLR by quantifying accretion and sediment deposition rates as well as soil compaction on a long time-scale in grazed and adjacent ungrazed parts of four salt marshes along the mainland coast of the Wadden Sea. We thereby neglect lateral marsh dynamics (such as edge erosion through wave impacts), since the effects of grazing are primarily expected to modified the vertical marsh growth, rather than edge erosion. The investigation of long-term accretion rates often leads to a small spatial resolution, hence measurement locations should be representative for the studied marshes (Nolte et al., 2013), since sediment dynamics in marshes are three dimensional processes and can hardly be represented in one point (e.g. van der Wal et al. 2004, van de Koppel et al. 2005, de Groot et al. 2011).

We tested the following hypotheses:

(1) Vertical accretion rates are lower in grazed compared to ungrazed salt marshes. To test this we calculated accretion rates by radionuclide dating of sediment horizons in soil cores.

(2) Sediment deposition rates are lower in grazed compared to ungrazed sites. This was investigated by calculating the annual amount of settled sediment per unit area.

(3) Soil compaction is higher on grazed compared to ungrazed sites. This hypothesis was tested by comparing the dry bulk density of the soil, which was assumed to be a measure for grazing-induced soil compaction.

2. Materials and Methods

2.1 Study sites

The study was carried out on four different salt marshes in The Netherlands and Germany along the mainland coast of the European Wadden Sea, a shallow depositional coastal system, stretching from the Netherlands to Denmark (Fig. 1). The Wadden Sea has the largest salt marsh area in Europe in one entity with barrier sand dunes and the tidal flat area (Dijkema 1987). The main types of Wadden Sea salt marshes are barrier connected or lay in front of the mainland. Three German study sites are part of the Schleswig-Holstein Wadden Sea National Park, which was established in 1985. One Dutch study site is protected as a national nature conservation area. Traditionally, all study sites were used for intensive livestock grazing and are characterised by a history of coastal engineering. The construction of ditched sedimentation fields, enhancing sediment deposition and establishment of salt-marsh vegetation, led to a relatively flat topography (Esselink et al., 1998). With increasing importance of nature conservation, drainage and grazing have been reduced or stopped in many Wadden Sea salt marshes since the 1980s (Esselink et al., 2009). Each of our study sites is subdivided into a grazed and an ungrazed part. Grazing treatments were underway for at least 20 years before sampling and have been maintained ever since. The change in grazing treatment led to a change in vegetation composition in most of the ungrazed and in some of the grazed parts of the study sites between 1988 and 2010 (Tab. 1; Esselink et al., 2009). Vegetation on ungrazed parts of the study sites generally developed from *Puccinellia maritima* or *Festuca rubra* types to the *Elymus athericus* type, which typically implied a development from rather short to high and biomass-rich canopies (Kiehl et al., 2001). Vegetation on grazed parts of the marshes often developed from *Puccinellia maritima* to *Festuca rubra* types or stayed the same (Esselink et al., 2009). Rates of accretion or surface elevation change between 7 and 43 mm yr^{-1} were reported for salt marshes at the Dutch coast (Dijkema, 1997; Esselink et al., 1998; Hazelden and Boorman, 1999; Dijkema et al., 2010), which is a higher range than values communicated for most salt marshes in Germany (6-26 mm yr^{-1}; Dittmann and Wilhelmsen, 2004; Stock, 2011; Suchrow et al., 2012).

Figure 1

The elevation of sampling locations was measured using a levelling instrument (Spectra precision® laser LL500 and laser receiver HR500 by Trimble) or extracted from a digital elevation model using the Software ArcGIS 10 (Tab. 1). The same software was used to assess the distance to the next creek and the distance to the marsh edge by means of an aerial photograph.

The salt marsh Noord-Friesland Buitendijks (NFB), The Netherlands (53°20'11", 5°43'40"), is exposed to a tidal range of about 2.1 m. The sedimentation fields, leading to marsh development, were installed in the years 1952 to 1960. Large parts of the area have been purchased by the NGO 'It Fryske Gea' for nature conservation. Drainage ditches have not been maintained since the year 2000 (Dijkema et al., 2011). The part of the site in which sampling took place is moderately grazed by horses (Dijkema et al., 2011). The ungrazed part was abandoned approximately 30 years ago and lies at a distance of 1.8 km to the grazed part.

Dieksanderkoog (DSK) at the mouth of the Elbe estuary, Germany, is a wide salt marsh, which stretches up to 2,000 m from the seawall to the intertidal flats (53°58'23", 8°53'8") and is exposed to a tidal range of 3 m. The marsh started to develop after 1935, when the present seawall and a system of sedimentation fields were constructed (Kohlus, 2000). One part of the salt marsh is intensively grazed by sheep and an adjacent part remained ungrazed since the early 1990s (Stock, 2005). On the latter part, maintenance of the ditches was stopped after the abandonment of grazing. On the grazed part, however, ditches are still renewed every five years.

The study site Hamburger Hallig (HH) is situated behind a small remnant of a former island 3 km off the coast (54°36'8", 8°49'27"). The tidal range at this site is 3.4 m. After the construction of a dam connecting the island with the mainland in 1874, salt marshes began to expand alongside the dam (Palm, 2000). The whole salt marsh was intensively grazed by sheep until 1991. Since then, 26% of the area is moderately grazed and 21% is still intensively grazed, while grazing was abandoned on 53% of the area (Esselink et al., 2009).

The study site Sönke-Nissen-Koog (SNK) is situated 3.5 km north of HH (54°38'4", 8°50'2") and experiences the same tidal range (3.4 m). After the construction of the present seawall and adjacent sedimentation fields in 1925 (Kunz and Panten, 1997), a salt marsh developed with a current extent of approximately 1,000 m. The study site is part of a grazing experiment (Kiehl et al., 1996), which started in 1988 and guarantees a continuous intensive grazing on the grazed part. The ungrazed part of the marsh is situated adjacent to the grazed one.

2.2 Core sampling

In 2010, we collected four soil cores from each of the study sites. NFB, HH and SNK were sampled in April; DSK was sampled in December. Two cores were taken from the ungrazed part and two from the grazed part of each site. One of the two cores per grazing treatment and site was collected at a sampling location close to the seawall (hereafter termed ‘landwards’) and one close to the intertidal flats (hereafter termed ‘seawards’). Soil cores were taken by removing the vegetation at the sampling location and driving a PVC tube (11.8 cm inner diameter) down to a depth of 80 cm into the soil. Sampling compaction was measured while taking the cores in the field. For doing this, the actual length of each soil core (distance upper to lower end of the core) was related to its original length (distance soil surface to lower end of the core) resulting in sampling compaction (%) for every core. After sampling, the tubes were sealed with plastic bags to avoid loss of soil moisture.

2.3 Core processing and soil properties

In the laboratory, each core was cut twice along its side in order to remove one half of the tube. For NFB and HH, processing of the cores was carried out at Groningen University. Here, the cores were cut into 2 cm sections. Each section of soil was weighed, dried in the oven at 105 °C to constant weight, and then weighed again to determine soil moisture. Dry weight and volume per section were used to calculate dry bulk density (g cm^{-3}). The dried material was ground using a Culatti rotor mill to disaggregate the soil particles. The grain size was then analysed by laser diffractometry (Mastersizer S – long bench MAM 5005) assessing volumetric concentrations of different grain-size classes. Organic matter content was determined as the weight loss after ignition of a 5 g subsample of each section at 550 °C for four hours. For DSK and SNK, processing of the cores was carried out at the University of Hamburg. Here, the cores were cut into 1 cm sections in the upper 24 cm and into 2 cm sections below. For SNK, sections below 40 cm were cut to 5 cm. Soil moisture, dry bulk density, and organic matter content were determined as described for NFB and HH, and samples were manually ground afterwards. Grain size distribution was analysed using a laser diffraction sensor (HELOS H2249).

In order to measure the activity of the radionuclide ^{137}Cs, the ground soil material of all samples was filled into 120 ml containers. Measurements were performed in the Laboratory for Radioisotopes at Göttingen University, Germany, for a minimum counting time of 250,000 seconds using a low-background coaxial Ge(Li)detector (Schuerch et al., 2012). As ^{137}Cs sorbs strongly onto small particles, the ^{137}Cs activity of each section was normalised to the mean organic matter content and mean percentage of grain sizes smaller than 20 µm of the whole core (Kirchner and Ehlers, 1998).

2.4 The ^{137}Cs dating method

The ^{137}Cs radionuclide ($t_{1/2}$= 30.2 years) is anthropogenic in origin and produced by nuclear fission. In Europe, sediment cores usually show two peaks of increased ^{137}Cs activity, which can be attributed to two historic nuclear events: The upper peak is usually related to the accident at the Chernobyl nuclear power plant in 1986, while the lower peak is caused by the nuclear bomb tests during the 1960s with its maximum in 1963 (Kirchner and Ehlers, 1998). If only one peak was present in cores of our study, the ^{137}Cs activity below the peak was considered to relate the peak to either 1963 or to 1986. If the activity was approaching zero below the peak, the peak was regarded as resulting from 1963 and not from 1986 since anthropogenic emission of ^{137}Cs only started in the 1950s (Pennington et al., 1973).

We attempted to validate the measurements of the ^{137}Cs-method using the simultaneously measured ^{210}Pb (Nolte et al. 2013). Reliable age calculations from ^{210}Pb, however, require undisturbed sediment layers. Many of the cores presented here were disturbed in larger depths, as the removal of sediment from ditches and its deposition on the marsh surface was a common practice before 1986. Therefore the ^{210}Pb-measurements could not be used in this study.

2.5 Accretion and sediment deposition rates

Before calculating salt-marsh accretion rates, we corrected each core for sampling compaction by adding the compaction term (in %) to each single section. This corrected section thickness was used for all further calculations. The mean sampling soil compaction was 7.5% and ranged from 0.3 to 12.4%. After the correction for sampling compaction, marsh accretion rates (mm yr^{-1}) were derived from the identified ^{137}Cs peak by dividing the respective depth by the time period since 1986 or 1963 (see Dyer et al., 2002).

To determine the input of sediment to a site, we calculated the sediment deposition rate (kg m^{-2} yr^{-1}; Callaway et al., 1996). For this, the dry bulk density (*ρ)* and thickness of each section (a) above the soil depth with the identified ^{137}Cs peak were multiplied, summed up and divided by the years passed (t; Eq. 1). The thickness of each section represents the original thickness of each slice (*e.g.* 10 mm) corrected for the sampling compaction (*e.g.* 10%) by adding the percentage compaction to the thickness of each slice (*e.g.* + 0.1 mm). The sediment deposition rate includes both mineral sediment deposition and organic deposition.

$$\textbf{sediment deposition rate} = \frac{\sum_{i-1}^{n}(a_i * \rho_i)}{t} \qquad \text{(Eq. 1)}$$

2.6 Soil compaction (dry bulk density)

Dry bulk density was used as a proxy for soil compaction. We compared mean dry bulk densities above the identified ^{137}Cs peaks between grazed and ungrazed cores of the same site. Grazing-induced compaction is assumed to be an important parameter in influencing dry bulk density as found amongst others by Schrama et al. (2013). Also, source material could possibly influence the dry bulk density. This influence should, however, be similar in all four cores within one site and should therefore not influence the pair-wise comparisons.

2.7 Statistical data Analysis

The data did not meet the assumptions of normality and homogeneity and therefore non-parametric tests were used. Wilcoxon signed-rank and Kruskal-Wallis tests were applied to compare site and core characteristics between sites, grazing treatments and among sampling locations. These site and core characteristics were elevation above MHT, distance to the marsh edge and to the next creek, as well as mean organic matter content and mean percentage of grain sizes smaller than 20 µm in the upper 50 cm of the soil.

Differences in accretion rates, sediment deposition rates and dry bulk density between grazed and ungrazed parts were analysed with a Wilcoxon signed-rank test. The same test was used for analysing differences between seaward and landward sampling locations. Differences among the four study sites were analysed with Kruskal-Wallis tests.

The relation of accretion rate, sediment deposition rate and dry bulk density to elevation above MHT, distance to the marsh edge, distance to the next creek, mean organic matter content and mean percentage of grain sizes smaller than 20 µm in the upper 50 cm of soil was tested with Spearman's rank correlations. Statistical significance in all tests was determined using a 95% confidence interval with the probability $p<0.05$. All analyses were conducted with SPSS 19.

3. Results

3.1 Peak identification

In 14 out of the 16 cores, peaks of ^{137}Cs activity could be identified. Seven cores showed the expected pattern with two peaks (Fig. 2A; S 1), which could be identified as 1986 and 1963, respectively. In five cores, we found a single peak only, which was identified as 1986. In two further cores, single peaks were found and identified as 1963 as the ^{137}Cs activity approached zero below the respective peak. No peak could be detected in the core of the seaward ungrazed sampling location at NFB. Here, it is likely that both the 1986- and 1963-peak were below the sampling depth of 80 cm. We therefore calculated a minimum accretion rate and sediment deposition rate for this core assuming the 1986-peak to be just below 80cm and used it for further analysis. In the core of the landward ungrazed sampling location at HH, a high activity of ^{137}Cs in a layer close to the marsh surface was found. This peak could not be clearly identified; its position was too shallow to be identified as the 1986-peak. We assume some disturbance in this core and excluded it from the calculation of accretion and sediment deposition rates and of mean dry bulk densities above the ^{137}Cs peak.

Figure 2

3.2 Site and core characteristics

Organic matter content, elevation above MHT, and distance to the marsh edge did not differ significantly between grazed and ungrazed parts of the marshes (Tab. 1 and 2). The only difference was observed for the average distance to the next creek, which was slightly smaller for ungrazed parts (median 49 m, range 14-74 m) compared to grazed parts (median 51 m, range 40-100 m $Z= -2.371$; $p<0.05$; $N=16$; Wilcoxon-Test). In addition to distance to the marsh edge, which is of course smaller for the seaward location, none of the abovementioned factors differed significantly between seaward and landward sampling locations.

3.3 Accretion rates

The mean accretion rate was 11.6 mm yr^{-1} and ranged from 5.4 to 34.6 mm yr^{-1} (Tab.2). In general, we found that the accretion rates calculated by ^{137}Cs dating agreed with literature data, except for a small number of values (Tab. 1). Rates did not significantly differ between the grazing treatments (Fig. 3). Highest values were achieved at the Dutch site NFB, where rates on the ungrazed parts were twice as high as on the grazed parts (medians 29.0 mm yr^{-1} and 13.4 mm yr^{-1}, respectively). The three German sites all had lower accretion rates of 8.2 mm yr^{-1} on average. In one further case we found a higher accretion rate in the ungrazed compared to the grazed part (DSK landwards; Tab. 2). The seaward locations at both HH and SNK showed the opposite pattern with higher accretion rates in the grazed compared to the ungrazed part. Accretion rates differed significantly between landward and seaward locations and were always higher at seaward locations (Fig. 3). We also found a negative correlation with distance to the marsh edge (r_s= 0.54; $p<0.05$; N=15). No significant correlations were found between accretion rate and distance to the next creek or to elevation above MHT.

Figure 3

3.4 Sediment deposition rates

The mean sediment deposition rate was 6.6 kg m^{-2} yr^{-1} and ranged from 2.8 to 11.9 kg m^{-2} yr^{-1}. It did not differ between the grazing treatments (Fig. 4). Also, differences between the study sites were not significant (H=6.57; p=0.09; N=15; Kruskal-Wallis-Test). However, we found a trend of higher values at NFB. Here, a median of 10.4 kg m^{-2} yr^{-1} was deposited on ungrazed and 5.8 kg m^{-2} yr^{-1} on grazed parts. At all four study sites, sediment deposition rates were found to be higher at seaward compared to landward sampling locations (medians 8.4 and 4.8 kg m^{-2} yr^{-1}, respectively; Fig. 4).

Figure 4

3.5 Compaction

Mean dry bulk density of sediment above the identified ^{137}Cs peak ranged from 0.34 to 1.10 g cm^{-3} and depended on the grazing treatment; it was significantly higher on grazed compared to ungrazed parts (medians 0.65 and 0.52 g cm^{-3}, respectively; Fig. 5). Mean dry bulk density increased with decreasing mean organic matter content in the upper 50 cm of the soil cores (r_s=-0.68; p<0.01; N=15). It further increased with decreasing mean percentage of soil particles smaller than 20 µm (r_s=-0.76; p<0.001; N=15).

Figure 5

4. Discussion

Accretion rates and grazing regimes

Our hypotheses that accretion and sediment deposition rates would be higher on ungrazed salt marshes, was not supported by our results. However, we see a different outcome for the German marshes compared to the Dutch marsh, where our hypothesis was clearly supported. Rather than depending on the grazing treatment or on elevation, as in other studies (e.g. Stoddart et al., 1989; Temmerman et al., 2003), accretion and sediment deposition rates depended on the distance to the marsh edge, explaining large scale patterns of sediment deposition (see also Esselink et al., 1998; Reed et al., 1999; Bartholdy et al., 2004; Dijkema et al., 2010; but see Craft, 2012 for a contrasting result). For study sites in the inner part of the marsh, this behaviour was already described by French and Spencer (1993) and van Proosdij et al. (2006) and infers that marsh accretion in the inner part of the marsh is primarily controlled by extreme flooding events rather than slowly changing hydroperiods. At NFB, the hypothesis of higher accretion and sediment deposition rates on the ungrazed part was supported. This might, however, possibly be caused by a shorter distance to the marsh edge on ungrazed locations (Table 1).

One explanation for the unexpected results that do not support the hypothesis of lower accretion and sediment deposition rates in grazed marshes, might be the relative importance of small scale patterns of sediment deposition in Germany. At the Dutch site NFB, ditches had silted up within the last ten years and inundating water enters the marsh mainly from the marsh edge or the major creek and only during storm events. This flow pattern leads to a large scale sedimentation pattern with high sediment deposition rates closer to the marsh edge and major creeks. In contrast to NFB, inundating water enters the marshes in Germany mainly from the still intact ditch system. Consequently, sediment deposition is highest along the small ditches, thereby leading to the formation of levees. This small-scale pattern of sediment deposition might be amplified by vegetation, which can slow down currents and enhance sedimentation (e.g. Christiansen et al., 2000; Baustian et al., 2012; Temmerman et al., 2012). Vegetation structure differed considerably between the grazed and ungrazed parts of the study sites (Tab. 1); ungrazed salt marshes were covered by tall and dense vegetation, which can be expected to trap large amounts of sediment. Flow velocities at the creek edge are 2-4 times lower on a vegetated marsh than on a marsh with no or only short vegetation (Temmerman et al., 2012). Therefore, in a marsh with tall and dense vegetation, more sediment settles close to the ditch and does not reach the central part of the marsh, where the cores for this study were taken.

Another explanation for the unexpected results might be the feedback of trampling causing soil compaction and thus lowering the surface elevation which could lead to an increased sediment deposition rate in grazed marshes against our expectation. However, sediment deposition rates did not differ between grazed and ungrazed marshes, but the interplay of these factors driving sedimentation on should be further investigated. The hypothesis that compaction is higher on grazed sites than on ungrazed sites was supported by our findings.

Mean dry bulk densities, which were used as an indicator for soil compaction, were significantly higher on grazed sites. Olsen et al. (2011) and Schrama et al. (2013) came to a similar conclusion. Soil compaction was especially pronounced at NFB. This very clear outcome might be caused by the livestock species used for grazing. While the German marshes are grazed by sheep, the study site at NFB is grazed by horses, which might cause more compaction because of their higher activity in comparison to cattle and sheep (Menard et al., 2002). In general, also organic matter content and grain size distribution can influence dry bulk densities (Kolker et al., 2009). However, since the mean organic matter content and the mean percentage of grain sizes smaller than 20 µm did not differ significantly between grazed and ungrazed parts of the study sites, these do not explain differences of mean dry bulk densities between the grazing treatments.

Measured accretion rates in the context of SLR

At all sites, accretion rates were well above rates of SLR mentioned in the introduction. It might be argued that because of further compaction of layers the surface elevation change, rather than accretion should be measured to assess the marshes resilience to SLR. Accretion rates were found to only slightly exceed elevation change measurements by French et al. (2003), because the autocompaction rates are small in minerogenic compared to organogenic marshes. Van Wijnen and Bakker (2001) also measured both accretion and surface elevation change on island marshes, but found an elevation deficit even though there was no accretion deficit. However, this study encompassed a relatively short time scale (3 years) and therefore freshly accreted and thus uncompacted layers played a large role in their study. In contrast, the accretion rates presented in our study represent 23 years (or more) of accretion and therefore include a high number of already largely compacted deeper layers. Therefore, accretion rates calculated with long-term methods give a better approximation of surface elevation change then short term measurements. Additionally, deep subsidence rates are low in the Wadden sea area (0.8 mm yr^{-1} for Dutch and 0.4 mm yr^{-1} for German sites (Veenstra, 1980). It therefore seems likely that most mainland Wadden Sea salt marshes outpace rates of current and projected SLR independent of the grazing treatment. Furthermore, if sea level rises, the frequency of inundations increases as well, initiating a positive feedback loop of enhanced sediment deposition on salt marshes (French, 2006). However, if the rate of SLR would strongly accelerate in the future, salt marshes with low sediment supply might be endangered in the long term (Kirwan and Temmerman, 2009).

Limitations

Potential limitations, caused by the methods applied in this study are related to the correction for sampling compaction and the possibility of ^{137}Cs to migrate deeper into the soil. We corrected each soil core for the sampling compaction assuming constant compaction throughout the whole core. However, sampling compaction may vary between the different layers. Generally, one would expect the lower part of the core to have a higher bulk density because of autocompaction and therefore less sampling compaction to occur here. Using a constant correction for sampling compaction may therefore lead to overcompensation in the lower parts and to undercompensation in higher parts of the core. As the ^{137}Cs-peak representing the year 1986 is mainly found in the upper part of the cores, the calculated accretion rates might slightly underestimate the actual rates. However, bulk density often showed no clear distribution within the core and therefore made a correction for sampling compaction related to depth or bulk density impossible.

The calculation of accretion rates might also be affected by the downwards migration of ^{137}Cs in the soil core, which would lead to an overestimation of accretion rates. As a validation of results the use of ^{210}Pb was not applicable due to disturbances in the soil core before 1986.

Conclusions and outlook

Our results indicate that salt marsh soils were becoming compacted by grazing, while accretion rates and sediment deposition rates were not affected by the grazing treatment. In areas with high minerogenic deposition rates like the Wadden Sea, the resilience of salt marshes to SLR thus seems not to be negatively influenced by livestock grazing. The influence of grazing on accretion rates is therefore likely to be also affected by interactions of grazing with other biotic and abiotic processes. For this reason, the effect of abiotic and biotic factors on accretion rates, the interaction of these factors and finally their alteration by livestock grazing should be further studied. This might be especially important in organogenic coastal systems (e.g. salt grasslands at the Baltic Sea coast; Callaway et al., 1996, Dijkema, 1990), where grazing for nature conservation (Sammul et al., 2012) might cause a larger degree of compaction compared to the minerogenic Wadden Sea salt marshes. In addition, small scale patterns of sediment deposition should be considered in future studies (e.g. Dijkema et al., 2010). For improved estimates on the importance of grazing treatment on marsh resilience, the marsh should be investigated in a three dimensional way rather than studying single points and include further biotic and abiotic controls (van der Wal et al. 2004, van de Koppel et al. 2005, de Groot et al., 2011). This could be done by combining methods with a high temporal resolution (e.g. ^{137}Cs dating) with methods with a high spatial resolution (e.g. sediment traps; Nolte et al., 2013). In the face of a rising sea level, the question whether or not grazing as a tool for salt marsh management might influence sedimentation processes, it is especially important to make sustainable management decisions. Models which aim to predict future marsh development therefore should seek to include the interplay between grazing and other factors influencing marsh accretion with respect to spatial patterns.

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Figure captions

Fig. 1: Location of the four study sites on the Wadden Sea mainland coast. Black markers=grazed salt marsh; white markers=ungrazed salt marsh; crosses=seaward sampling locations; dots=landward sampling locations; base maps: Amtliche Geobasisdaten Schleswig-Holstein, © VermKatV-SH and Ministry of Agriculture, Nature and Food Quality, Copyright Slagboom en Peeters.

Fig. 2: An example for the (A) normalized ^{137}Cs activity, (B) grain size distribution, (C) dry bulk density, and (D) organic matter content for all depths in the core from the landward grazed sampling location at NFB. An overview of all 16 soil cores can be found in supporting information 1.

Fig. 3: Accretion rates of grazed and ungrazed and of landward and seaward locations. Boxplots represent: median (middle line), interquartile range (box), 1.5 times interquartile range (bar) and outliers (dots). The grazing treatment had no significant effect ($Z=0.34$; $p=0.74$; $N=14$; Wilcoxon-Test), while accretion rates were significantly higher in seaward compared to landward locations ($Z=-2.37$; $p<0.05$; $N=14$; Wilcoxon-Test).

Fig. 4: Sediment deposition rates of grazed and ungrazed and of landward and seaward locations. Boxplots represent: median (middle line), interquartile range (box), 1.5 times interquartile range (bar) and outliers (dots). The grazing treatment had no significant effect ($Z=0.00$; $p=1.0$; $N=14$; Wilcoxon-Test), while sediment deposition rates were significantly higher in seaward compared to landward locations ($Z=-2.37$; $p<0.05$; $N=14$; Wilcoxon-Test).

Fig. 5: Mean dry bulk density above the identified ^{137}Cs peak of grazed and ungrazed and of landward and seaward locations. Peaks were identified to originate from 1986 (NFB, HH and SNK) and from 1963 (DSK). Boxplots represent: median (middle line), interquartile range (box), 1.5 times interquartile range (bar) and outliers (dots). Median dry bulk density was found to be significantly lower in ungrazed locations (Z=-2.01; $p<0.05$; N=14; Wilcoxon-Test), while no significant difference was found between landward and seaward locations (Z=-1,82; p=0.069; N=14; Wilcoxon-Test).

Table captions

Tab. 1: Key parameters of the study sites. Tidal amplitude data was retrieved from BSH (2011); NFB=Noord Friesland Buitendijks; DSK=Dieksanderkoog; HH=Hamburger Hallig; SNK=Sönke-Nissen-Koog; NAP=Normal Amsterdam Peil (Dutch Ordnance Datum); NHN=Normalhöhennull (German Ordnance Datum); MHT=mean high tide; TMAP=Trilateral Monitoring and Assessment Program (Esselink et al., 2009); SEB=sedimentation erosion bar. TMAP vegetation types were coded as follows: Agr (*Agrostis stolonifera* type), Atr/Puc (*Atriplex portulacoides* / *Puccinellia maritima* type), Ely (*Elymus athericus* type), Fes (*Festuca rubra* type), Puc (*Puccinellia maritima* type), Spa (*Spartina anglica* type).

Tab. 2: Mean proportion of grain sizes smaller than 20 µm and mean organic matter content in the upper 50 cm of each core, depth of the ^{137}Cs peaks, accretion rate, sediment deposition rate and mean dry bulk density above the identified peak.

Appendix captions

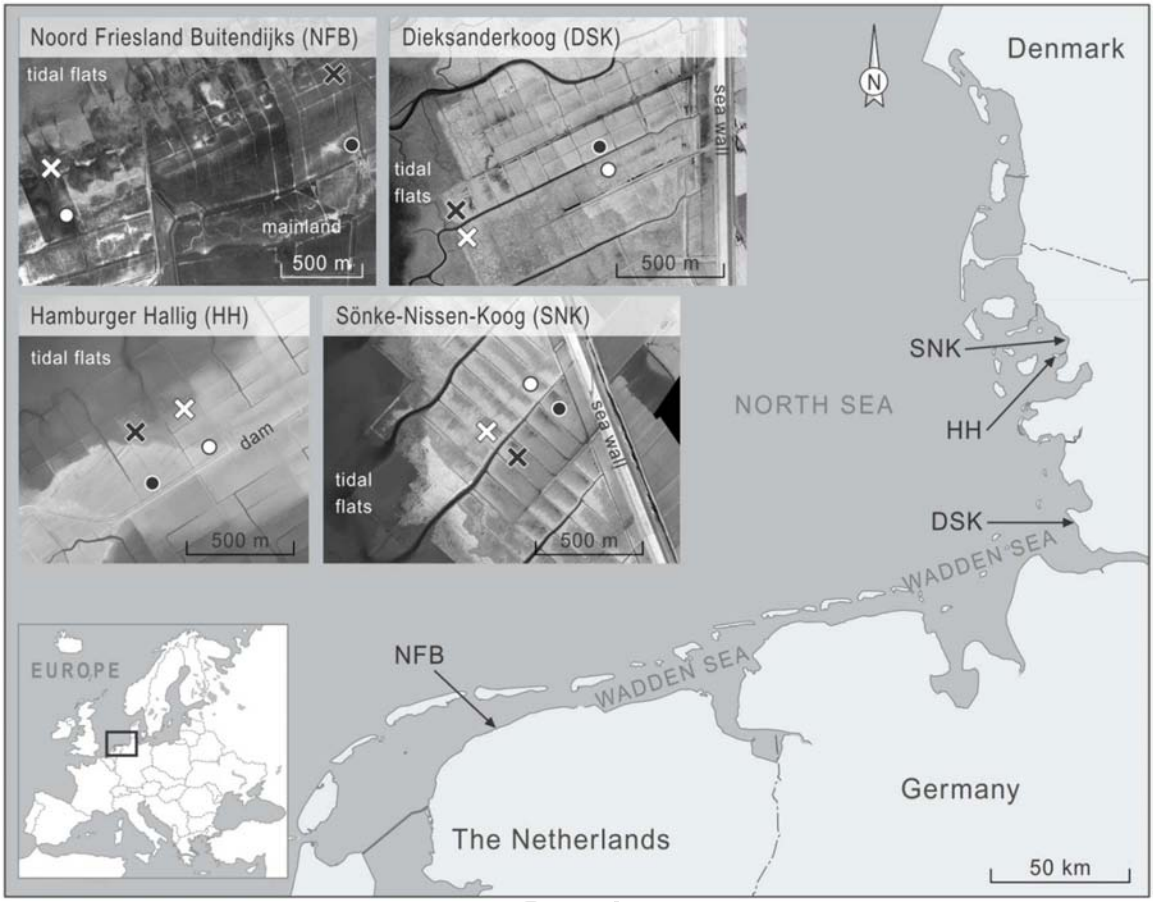
Appendix caption A: ^{137}Cs activity, grain size distribution, dry bulk density, and organic matter content for all depths in the 16 cores. The peaks from 1986 and 1963 are indicated with arrows.

Site	Location	Treatment	Elevation absolute [m]	Elevation above MHT [m]	TMAP vegetation in 1988	TMAP vegetation in 2010	Grazing animals	Grazing intensity	Distance to next creek [m]	Distance to marsh edge [m]	Reference accretion rates [mm yr^{-1}]	Method	Reference
NFB	Landwards	Ungrazed	1.55 m NAP	0.57	*Puc*	*Ely*	-	-	50	380	23.8	Plate	Nolte et al. unpubl. data
		Grazed	1.74 m NAP	0.76	*Puc*	*Agr*	Horses	moderate	50	630	7.2		
	Seawards	Ungrazed	1.29 m NAP	0.31	*Puc*	*Puc*	-	-	68	100	28.6		
		Grazed	1.49 m NAP	0.51	*Puc*	*Puc*	Horses	moderate	100	160	9.2		
DSK	Landwards	Ungrazed	2.11 m NHN	0.49	*Fes*	*Ely*	-	-	14	955	3.4	Levelling	WSV 2012
		Grazed	1.91 m NHN*	0.29	*Puc*	*Fes*	Sheep	intensive	48	920	5.2		
	Seawards	Ungrazed	2.40 m NHN	0.78	*Puc*	*Ely*	-	-	74	290	9.1		
		Grazed	2.25 m NHN*	0.63	*Spa*	*Ely*	Sheep	intensive	85	205	9.2		
HH	Landwards	Ungrazed	2.13 m NHN	0.54	*Puc*	*Ely*	-	-	55	250	3.3-4.8	SEB	Stock 2011
		Grazed	2.09 m NHN	0.50	*Puc*	*Fes*	Sheep	moderate	95	230			
	Seawards	Ungrazed	1.92 m NHN	0.33	*Atr/Puc*	*Atr/Puc*	-	-	35	35	3.2-6.3		
		Grazed	1.92 m NHN	0.33	*Puc*	*Fes*	Sheep	moderate	40	40			
SNK	Landwards	Ungrazed	2.00 m NHN	0.41	*Puc*	*Ely*	-	-	46	525	7.7	Levelling	Suchrow et al. 2012
		Grazed	2.07 m NHN	0.48	*Puc*	*Puc*	Sheep	intensive	51	685			
	Seawards	Ungrazed	2.04 m NHN	0.45	*Puc*	*Ely*	-	-	47	340	7.7		
		Grazed	2.04 m NHN	0.45	*Puc*	*Puc*	Sheep	intensive	48	450			

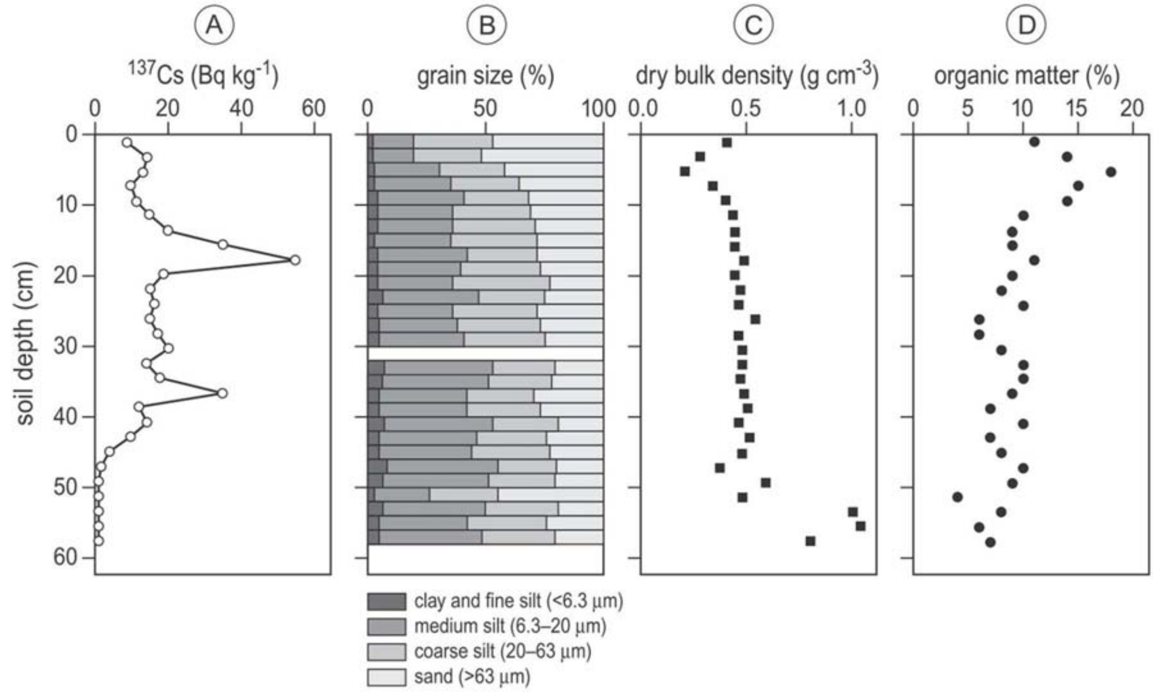
*elevation data of these sampling locations was extracted from a digital elevation model

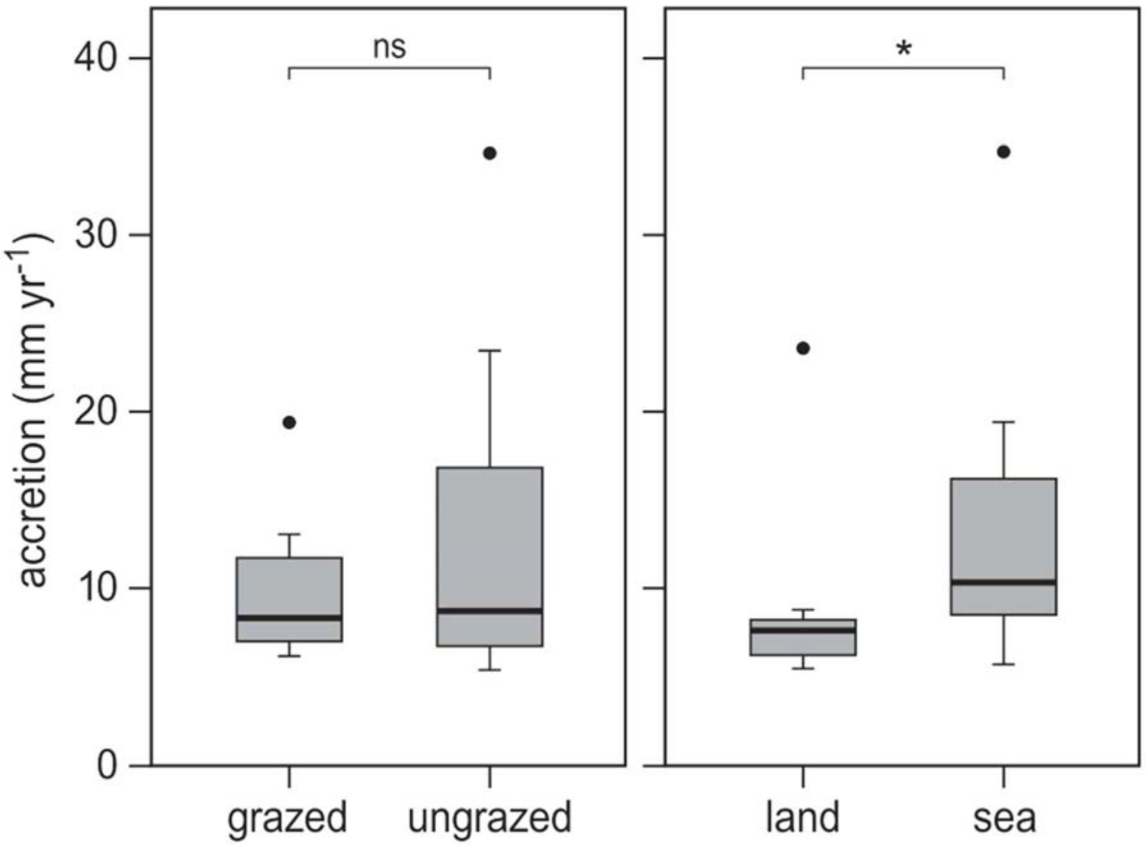
Site	Location	Treatment	mean percentage of		^{137}Cs peak depth		Accretion		Sediment deposition [kg m^{-2} yr^{-1}]	Mean dry bulk density above peak [g cm^{-3}]
			grain size <20 µm [%]	organic matter [%]	1986 [cm]	1960 [cm]	Reference year	Rate [mm yr^{-1}]		
NFB	Landwards	Ungrazed	45.79	9.72	56.1	none	1986	23.4	9.0	0.38
		Grazed	40.40	9.37	17.9	36.8	1986	7.5	3.0	0.38
	Seawards	Ungrazed	37.69	8.75	none	none	1986	34.6	11.9	0.34
		Grazed	41.39	9.26	46.4	none	1986	19.3	8.5	0.43
DSK	Landwards	Ungrazed	6.51	2.87	none	43.3	1963	8.7	8.9	0.98
		Grazed	22.64	6.42	14.4	26.7	1963	6.1	5.7	0.91
	Seawards	Ungrazed	22.37	5.39	19.4	47.6	1963	10.1	9.2	0.89
		Grazed	10.58	3.44	none	46.8	1963	10.4	11.6	1.09
HH	Landwards	Ungrazed	30.97	5.09	3.3	none	1986	1.4*	0.7*	0.38*
		Grazed	30.84	5.54	17.9	30.5	1986	7.5	3.5	0.44
	Seawards	Ungrazed	21.92	3.72	18.9	none	1986	7.5	4.0	0.48
		Grazed	23.76	4.18	31.1	none	1986	12.9	7.5	0.56
SNK	Landwards	Ungrazed	40.88	10.24	12.9	none	1986	5.4	2.8	0.52
		Grazed	32.53	7.29	15.2	53.4	1986	6.3	4.8	0.75
	Seawards	Ungrazed	30.25	6.91	14.5	33.7	1986	5.6	3.3	0.56
		Grazed	26.99	7.47	21.4	49.7	1986	8.9	8.3	0.93

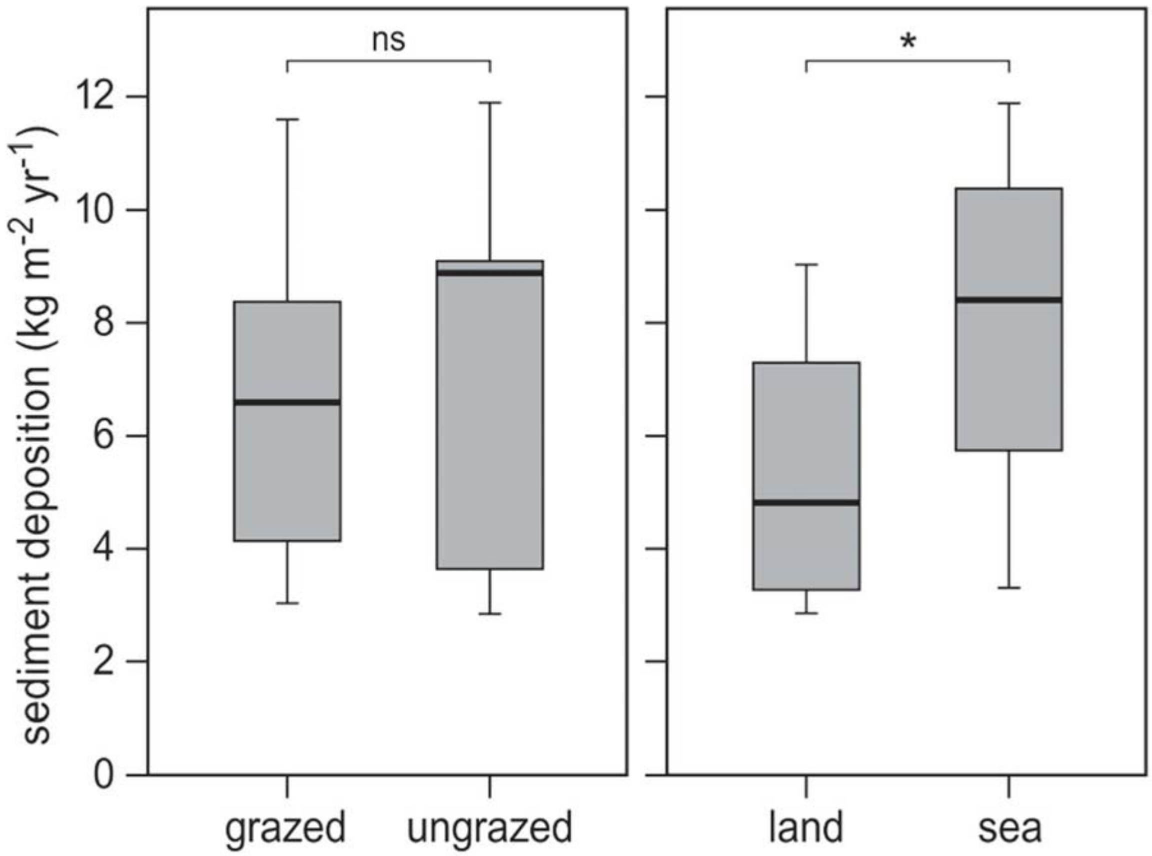
*Value excluded from the analysis (see text).

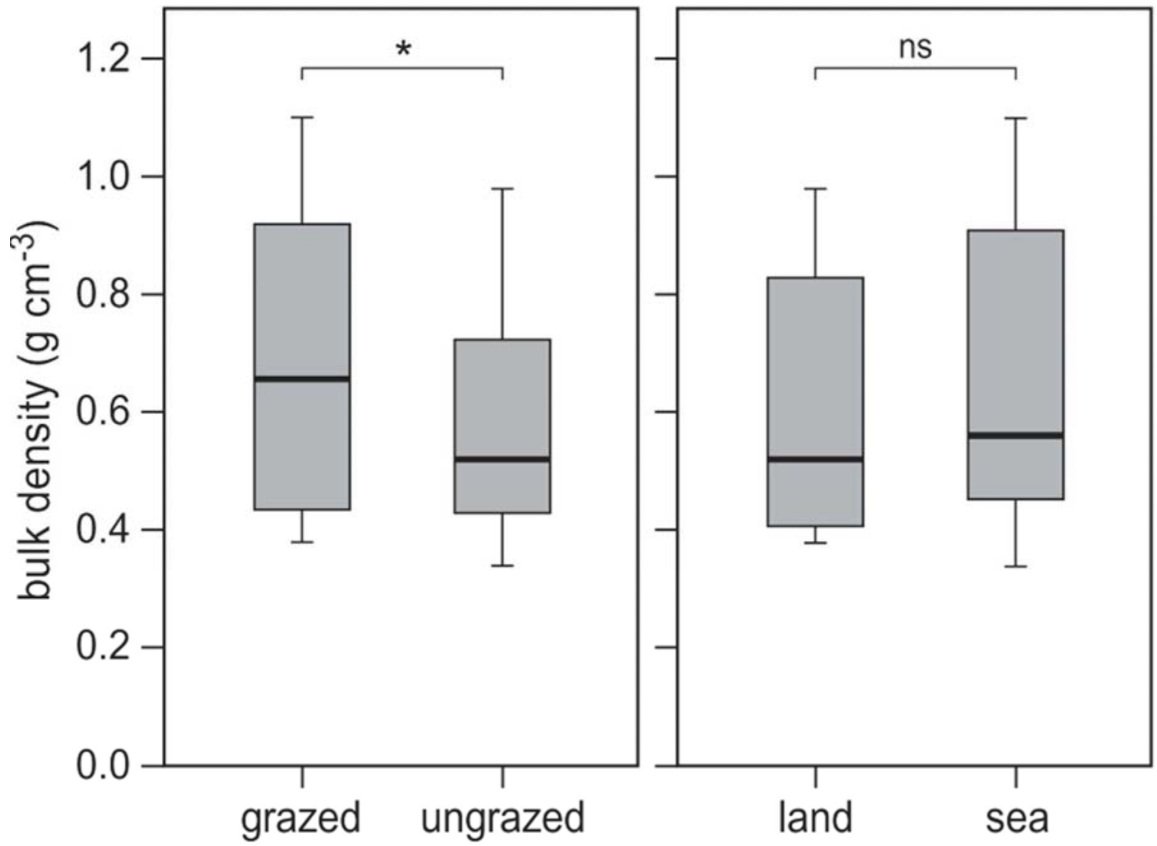


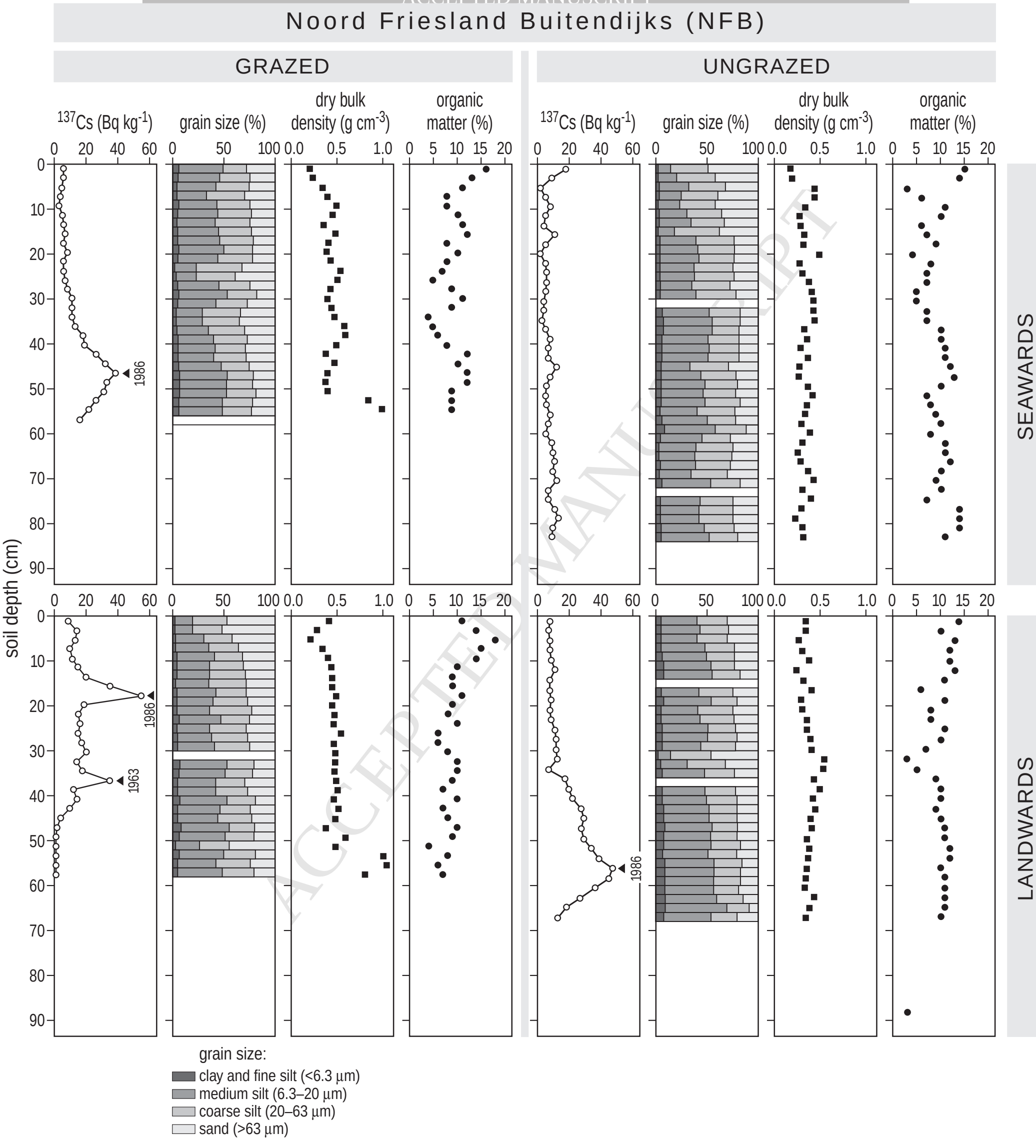
Noord Friesland Buitendijks (NFB)
tidal flats
mainland
500 m
Dieksanderkoog (DSK)
tidal
flats
sea wall
500 m
Hamburger Hallig (HH)
tidal flats
dam
500 m
Sönke-Nissen-Koog (SNK)
tidal
flats
sea wall
500 m
N
Denmark
SNK
HH
DSK
NORTH SEA
WADDEN SEA
WADDEN SEA
NFB
EUROPE
Germany
The Netherlands
50 km

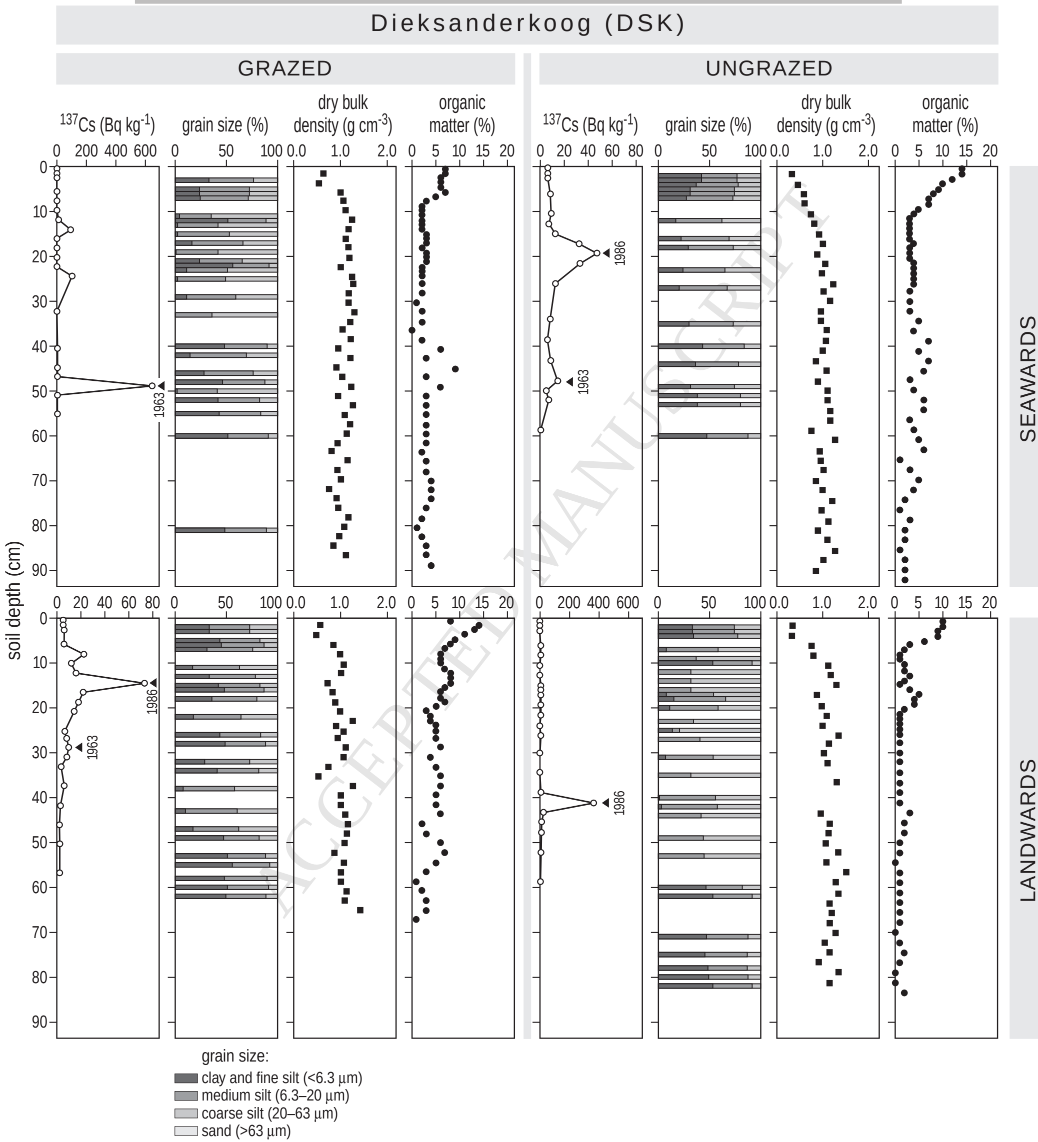

A
^{137}Cs (Bq kg^{-1})
B
grain size (%)
C
dry bulk density (g cm^{-3})
D
organic matter (%)
soil depth (cm)
clay and fine silt (<6.3 µm)
medium silt (6.3–20 µm)
coarse silt (20–63 µm)
sand (>63 µm)

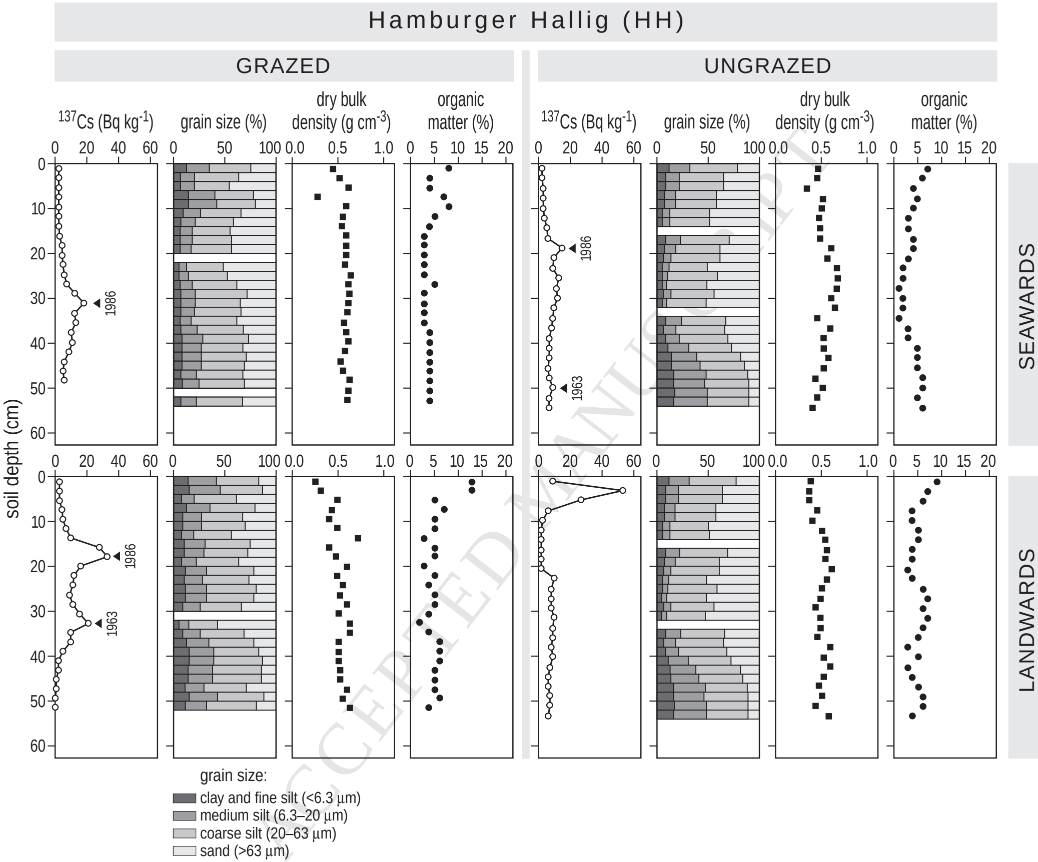

ns
*
40
30
20
10
0
accretion (mm yr-1)
grazed
ungrazed
land
sea


ns
*
sediment deposition (kg m-2 yr-1)
0
2
4
6
8
10
12
grazed
ungrazed
land
sea

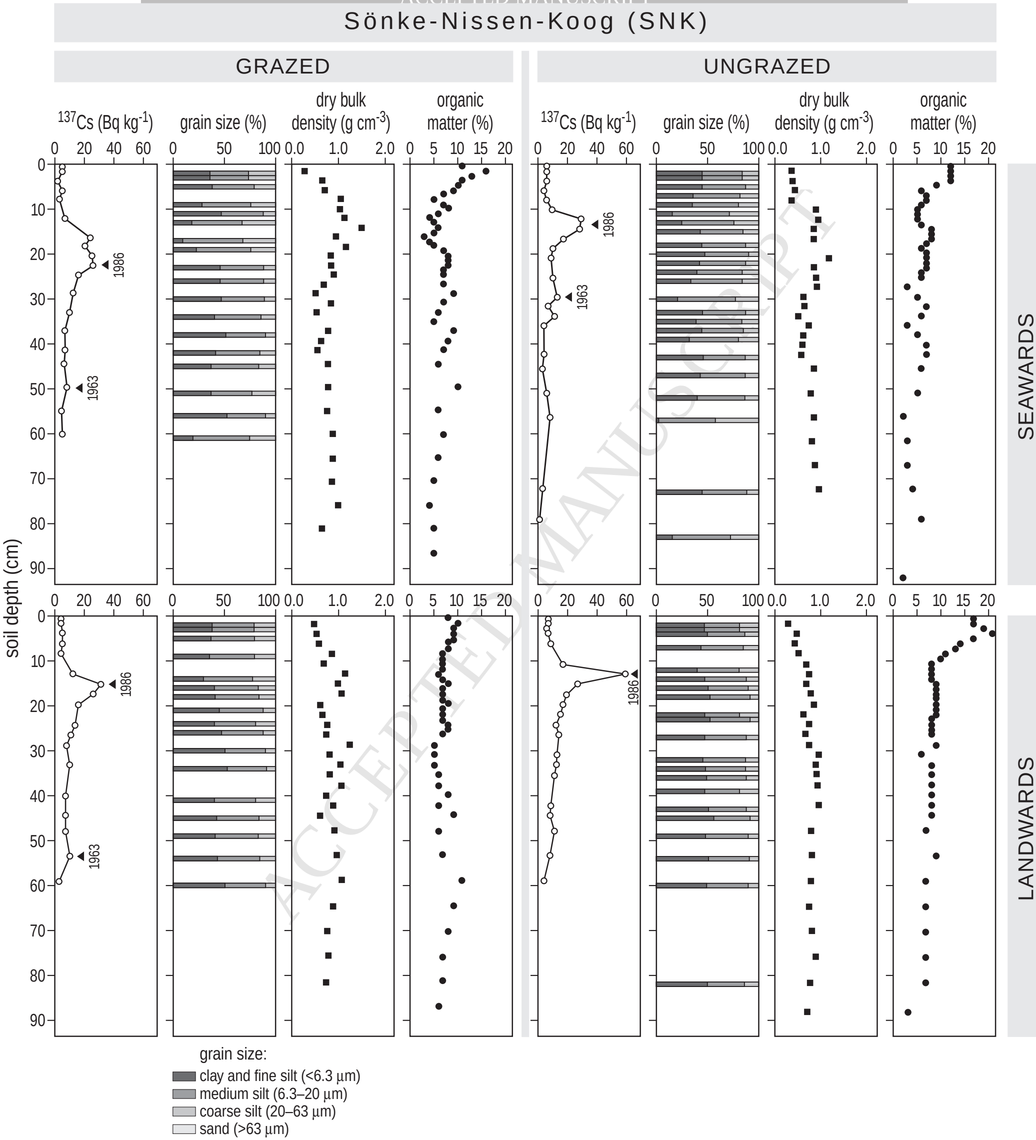

bulk density (g cm-3)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
*
ns
grazed
ungrazed
land
sea


Noord Friesland Buitendijks (NFB)
GRAZED
UNGRAZED
137Cs (Bq kg-1)
grain size (%)
dry bulk density (g cm-3)
organic matter (%)
soil depth (cm)
SEAWARDS
LANDWARDS
1986
1963
grain size:
clay and fine silt (<6.3 µm)
medium silt (6.3–20 µm)
coarse silt (20–63 µm)
sand (>63 µm)


Dieksanderkoog (DSK)
GRAZED
UNGRAZED
137Cs (Bq kg-1)
grain size (%)
dry bulk density (g cm-3)
organic matter (%)
soil depth (cm)
1963
1986
SEAWARDS
LANDWARDS
grain size:
clay and fine silt (<6.3 µm)
medium silt (6.3–20 µm)
coarse silt (20–63 µm)
sand (>63 µm)



Hamburger Hallig (HH)
GRAZED
UNGRAZED
137Cs (Bq kg-1)
grain size (%)
dry bulk density (g cm-3)
organic matter (%)
soil depth (cm)
1986
1963
SEAWARDS
LANDWARDS
grain size:
clay and fine silt (<6.3 µm)
medium silt (6.3–20 µm)
coarse silt (20–63 µm)
sand (>63 µm)



Sönke-Nissen-Koog (SNK)
GRAZED
UNGRAZED
137Cs (Bq kg-1)
grain size (%)
dry bulk density (g cm-3)
organic matter (%)
soil depth (cm)
SEAWARDS
LANDWARDS
1986
1963
grain size:
clay and fine silt (<6.3 µm)
medium silt (6.3–20 µm)
coarse silt (20–63 µm)
sand (>63 µm)

Supporting information 1: ^{137}Cs-activity, grain size distribution, dry bulk density, and organic matter content for all depths in the 16 cores. The peaks from 1986 and 1963 are indicated with arrows.