- Korrigierte Fassung -

Tidal marshes of the Elbe Estuary – spatial and temporal dynamics of sedimentation and vegetation

Dissertation zur Erlangung des Doktorgrades an der Fakultät für Mathematik, Informatik und Naturwissenschaften

Fachbereich Biologie

der Universität Hamburg

vorgelegt von

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Hamburg, 2014

Tag der Disputation: 28.11.2014

Folgende Gutachter empfehlen die Annahme der Dissertation:

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Genius is 1 % inspiration and 99 % perspiration (Thomas Alva Edison)

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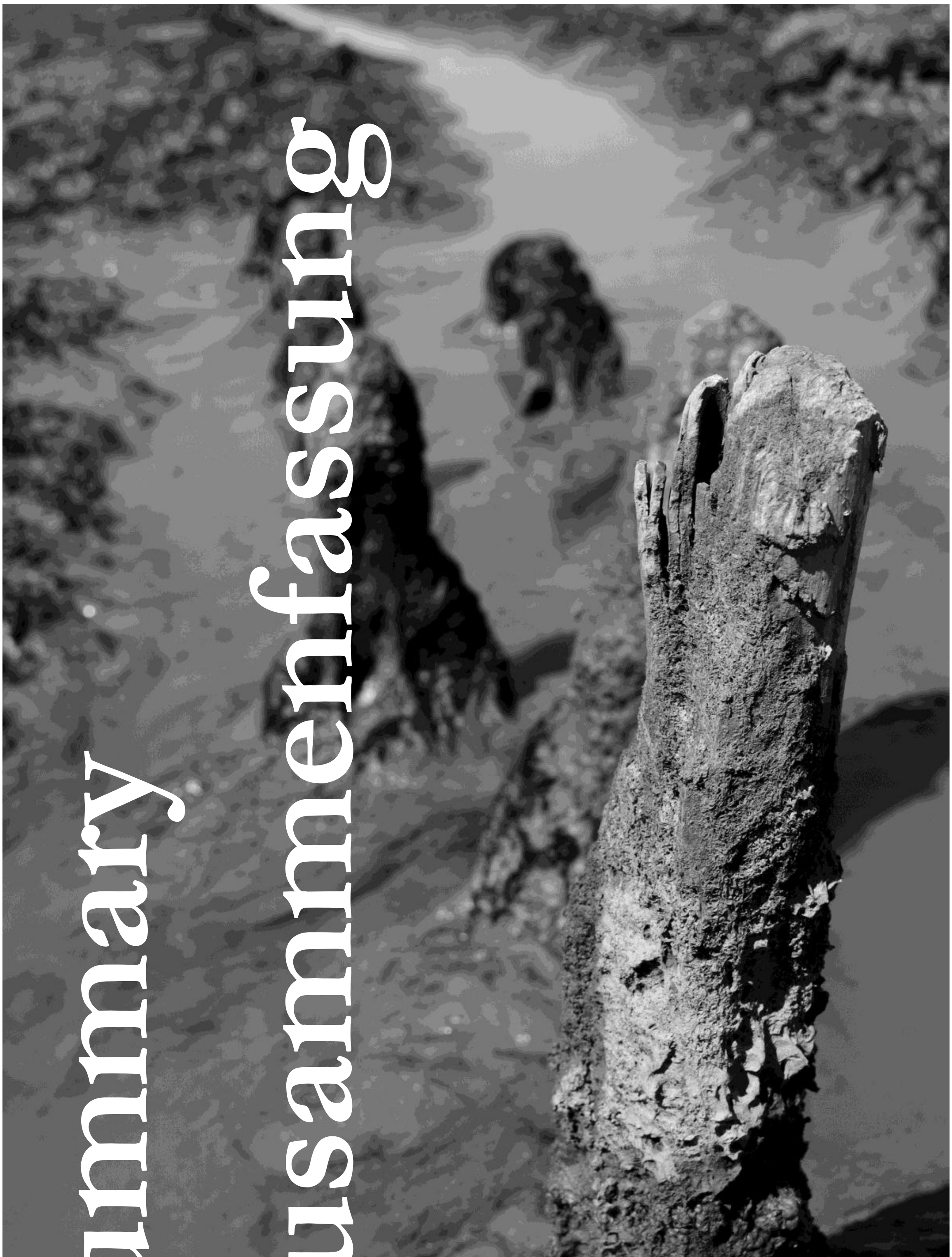
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Tidal marshes are vegetated ecosystems dominated by herbaceous plants, which are found along the shores of coasts and estuaries and are often bordered by tidal flats. Tidal marshes and tidal flats play an important role in coastal protection. They provide habitat for flora and fauna, they are a key component of element cycling between terrestrial and marine Systems, and they are used in different degrees for agricultural purposes. In these ecosystems, sediment deposition and tidal inundations cause distinct elevational zones: unvegetated tidal flats in the lowest and most frequently submerged areas, species-poor low marshes dominated by species with adaptations for regular inundations, and high marshes with high species richness and only sporadic inundations. This zonation along the elevational gradient can be seen as an outcome of progressive succession as a consequence of continual sediment deposition and an associated increase in surface elevation. A development in the opposite direction, e.g., from tidal marsh communities to tidal flats, can be interpreted as regressive succession. In addition, differences in salinity of the adjacent ocean and tidal river shape a salinity gradient along estuaries with highest levels at the mouth of estuaries and decreasing salinity upstream as freshwater mixes with seawater. This salinity gradient is usually reflected in the distinct vegetation of tidal freshwater, brackish and salt marshes.

Stability of tidal marshes in relation to rising sea-level depends on accretion rates. Vertical accretion in estuarine marshes is a complex function of different interacting factors with high variability in space and time. Variables such as inundation (frequency, duration, and height), distance to the sediment source (marsh edge and/or creek), suspended-sediment concentration of the inundation water, standing biomass, and seasonal variations of these factors have all been found to affect sediment deposition. Accretion rates are determined by rates of sediment deposition, erosion and compaction of minerogenic and organogenic particles, as well as subsurface accumulation of dead belowground biomass, and shallow subsidence processes. In times of climate change and associated sea-level rise, knowledge about the relationship between the above mentioned spatial and temporal factors affecting sedimentation and the resulting accretion rates is gaining high importance.

The Elbe Estuary is the largest estuary along the German coast. In 2011, tidal marshes covered an area of 75 km^2 and adjacent tidal flats extended over 187 km^2 between the city of Hamburg and the mouth of the estuary. However, almost all tidal marshes of the Elbe Estuary have been altered for decades by embankments, channel deepening, channel straightening, and other engineering activities. These activities affect sediment dynamics and may lead to temporal changes in sediment deposition and resulting vegetation patterns along estuarine gradients; processes which have been poorly documented for the Elbe Estuary.

The aim of this thesis was to determine the recent marsh development (1980–2010) and to identify factors that are influencing progressive and regressive succession of elevational zones. Furthermore, measurements of short-term sediment deposition with a high spatial resolution in salt, brackish and tidal freshwater marshes serve to identify the most important factors affecting sediment deposition and to investigate whether current marsh surface accretion rates are sufficient to compensate predicted sea-level rise. In addition, comparability of sediment-deposition rates of commonly used sediment-trap types and related studies is impeded by differences in trapping efficiency. For that reason, a flume-

study was conducted to compare the performance of different sediment-trap types for short-term measurements of sediment-deposition rates.

Chapter 1 provides a general introduction to tidal marshes, threats by sea-level rise, anthropogenic impacts, and sedimentation processes. Additionally, the objectives of the thesis and the author's contribution to each chapter are outlined.

Changes and pathways of marsh succession of the Elbe Estuary between 1980 and 2010, as well as influencing factors are described in **chapter 2**. First, the intertidal habitats were classified into three elevational, and three salinity zones. Then, vegetation maps of 1980 and 2010 were compared and the changes in total area of the distinguished habitats were calculated. To analyze the direction of temporal change, we differentiated between progressive and regressive succession. By using regression tree models (conditional inference trees), the most influential hydro-morphological factors to separate progressive and regressive succession of low marshes were identified. Total estuarine tidal marsh area at the Elbe increased by 2 % (150 ha) between 1980 and 2010. Changes were unequally distributed between salinity zones. While salt and brackish high marshes increased substantially, a decrease in tidal freshwater marshes was evident. Low marshes decreased in all salinity zones. Additionally, a high persistence of tidal flats (82–95 %) and high marshes (95–97 %) was determined, whereas only 19–28 % of 1980 low marshes remained. In salt and brackish marshes, more than two-thirds of the 1980 low marshes changed into high marshes (progressive succession). In contrast, less than one-third of low marshes underwent this development in tidal freshwater marshes. Here, 44 % of 1980 low marshes showed a regressive succession into tidal flats in 2010. Distance to the navigation channel was the major factor determining succession in salt and brackish marshes. Here, the closer the distance to the channel, the higher the risk of regressive succession was. In tidal freshwater marshes, river bank situation and distance to the navigation channel were identified as main factors for marsh succession. In this case, considerable channel engineering caused a strong decrease of mean low water and increase in mean high water between 1980 and 2010. It is quite likely that these interferences negatively modified marsh distribution, increasing regressive succession, and thus, limited the quantity of tidal freshwater marshes. The results of **chapter 2** provide valuable insight into the pathways of marsh succession and can further assist in recognizing problematic developments of estuarine tidal marshes.

Chapter 3 presents results of an annual sediment measuring campaign conducted in salt, brackish, and tidal freshwater marshes of the Elbe Estuary. The spatial and temporal variation in short-term sediment-deposition rates and its possible influencing factors in three marsh types along the estuarine salinity gradient were studied. Between March 2010 and March 2011, bi-weekly sediment deposition was quantified along three transects, reflecting the variability in elevation (low to high marsh) and distance to the sediment source, in a salt, brackish, and tidal freshwater marsh. Simultaneously, water-level fluctuations and suspended-sediment concentration (SSC) were recorded, and aboveground plant biomass was sampled once during the late summer (September 2010) and once at the end of winter (February 2011). Annual sediment deposition (17.5 ± 4.0 kg m^{-2}) and calculated accretion rates (20.3 ± 4.7 mm year^{-1}) were highest in the brackish low marsh and were between 51 and 71 % lower in the low tidal freshwater and the salt marsh, respectively. Highest SSC and longest inundations were found during the fall and winter. Flooding duration and frequency

were higher in the tidal freshwater than in the brackish and the salt marsh. Aboveground plant biomass of the regularly flooded vegetation stratum (0–50 cm above soil surface) did not differ between marsh types, but the spatial pattern changed between late summer and end of winter. In all three marsh types, decreasing sediment-deposition rates were recorded with increasing distances from the sedimentation source. The applied multiple regression models were able to explain 74, 79, and 71 % of variation in sediment- deposition patterns in tidal freshwater, brackish, and salt marshes, respectively. SSC was the most important factor of the model. The results emphasize the importance of considering spatial and temporal variations in sediment-deposition rates and its predictors. Findings of **chapter 3** showed that sediment-deposition rates in the tidal low marshes of the Elbe Estuary seem to be sufficient to compensate moderate rates of sea-level rise. Contrastingly, high salt marshes might be vulnerable due to insufficient input of sediment, and may regress into low marshes.

Chapter 4 shows the results of a flume study, which compared trapping efficiency of different types of sediment traps. Measurements of sediment-deposition rates are a common method to calculate tidal marsh accretion and estimate their stability with regard to rising sea-level. Missing standardization in measuring methods impede comparability of different studies. The trapping efficiency of two circular and two flat surface trap-types in a flume was determined. All trap types are frequently used to determine short-term sediment-depositions rates. Two scenarios of tidal inundations (short, 37 ± 2 min and long, 61 ± 2 min) and two scenarios of SSC (low, ~65 mg l^{-1} and high, ~100 mg l^{-1}) of the flooding water were simulated and the effects of these factors on sediment- deposition rates at different distances to the sediment source (inlet of the flume) were recorded. Highest sediment-deposition rates were found in circular traps, which were 20–45 % higher compared with floor mat and tile sediment-trap type, respectively. All types of sediment traps showed a strong decrease in sediment-deposition rates with increasing distance to the inlet of the flume, but no significant interaction between trap types and distance was found. These results show that different sediment-trap types differ in their trapping efficiency, but that these differences in trapping efficiency are independent of the tidal inundation scenario, SSC, and distance to the sediment source. To enhance the explanatory power of different sediment-trap types, additionally field studies under different environmental conditions are recommended.

Chapter 5 discusses and summarizes the findings of the thesis.

Overall, the results of this thesis indicate that sediment-deposition rates underlay strong spatio-temporal variability. Sediment deposition strongly differs between salinity zones. Highest sediment-deposition rates and resulting progressive succession were found in the estuarine maximum turbidity zone of the brackish marshes and in areas with large tidal flats in front of the vegetated marshes which are consistently far away from the navigation channel. Lowest sediment-deposition rates and overall shortest distance to the navigation channel induced a decrease of tidal freshwater marshes. These areas are affected by extensive anthropogenic engineering activities of the navigation channel, resulting in a remarkable increase of the tidal amplitude during the last decades. Tidal freshwater marshes are especially vulnerable to effects of climate change and accelerated sea-level rise. In addition to rising sea levels, increases of salinity may also compromise the habitat quality of the local endemic species *Oenanthe conioides* at the Elbe Estuary.

Zusammenfassung

Tidemarschen sind von krautigen Pflanzen bewachsene Ökosysteme, die entlang von Küsten und Ästuaren auftreten und seeseitig häufig von Watten begrenzt sind. Tidemarschen und Watten haben große Bedeutung für den Küstenschutz. Sie bieten Habitate für Fauna und Flora, erfüllen eine Schlüsselrolle im Stoffkreislauf zwischen terrestrischen und marinen Systemen und werden zum Teil landwirtschaftlich genutzt. In diesen Ökosystemen verursachen unterschiedliche Sedimentablagerungen und Überflutungsbedingungen eine ausgeprägte Zonierung. In den am tiefsten gelegenen und am häufigsten überfluteten Bereiche befinden sich Watten, die durch das Fehlen von Höheren Pflanzen charakterisiert sind. In Bereichen mit periodisch auftretender Überflutung befinden sich die relativ artenarmen Unteren Marschen. Nur von unregelmäßigen Überflutungen geprägt, entwickeln sich die Oberen Marschen, die sich im Gegensatz zur Unteren Marsch durch einen höheren Artenreichtum auszeichnen. Diese Zonierung resultiert aus einer „progressiven" (fortschreitenden) Sukzession durch fortlaufende Sedimentablagerungen und dem damit verbundenen Anstieg der Geländehöhe. Die entgegengesetzte Entwicklung wird als „regressive" (rückwärtsgerichtete) Sukzession bezeichnet. Dies ist beispielsweise der Fall, wenn sich die Untere Marsch zu Watten entwickelt. Als zweiter die Zonierung beeinflussende Gradient wird in der vorliegenden Arbeit die Salinität betrachtet. Sie wird maßgeblich durch angrenzende Ozeane bzw. einen tidegeprägten Fluss bestimmt. Unterschiede der Salinität sind ein typisches Merkmal in Ästuaren. Die Salinität nimmt mit zunehmendem Abstand zur Mündung und erhöhter Süßwasserbeimischung ab.

Die Stabilität der Tidemarschen ist eng vom Verhältnis zwischen Meeresspiegelanstieg und Auflandungsraten abhängig. Vertikale Auflandungsraten auf ästuarinen Marschflächen sind eine komplexe Funktion von verschiedenen interagierender Einflussfaktoren, die zusätzlich noch eine hohe räumliche und zeitliche Variabilität aufweisen. Als wichtige, die Sedimentablagerung beeinflussende Faktoren wurden die Überflutungshäufigkeit, -dauer und -höhe, die Entfernung zur Sedimentquelle (Marschkante und/oder Priel), die im Überflutungswasser suspendierten Schwebstoffe, der Pflanzenbewuchs sowie die jahreszeitlichen Unterschiede in diesen Faktoren identifiziert. Auflandungsraten setzen sich aus Sedimentablagerung, Verdichtungen sowie Erosion der mineralischen und organischen Partikel zusammen. Ebenso spielen eine unterirdische Anreicherung von abgestorbener Biomasse (z.B. Wurzeln) und Bodensenkungsprozesse eine Rolle. In Zeiten des Klimawandels und dem damit verbundenen Anstieg des Meeresspiegels sind Kenntnisse über räumliche und zeitliche Variabilität der aufgezählten Einflussfaktoren von entscheidender Bedeutung, da diese die Sedimentablagerungen und folglich die sich daraus ableitenden Auflandungsraten beeinflussen.

Das Mündungsgebiet der Elbe weist das flächenmäßig größte Ästuar an der deutschen Küste auf. Im Jahr 2011 erstreckten sich die Tidemarschen zwischen dem Hamburger Stadtgebiet und der Mündung bei Cuxhaven/Friedrichskoog Spitze über insgesamt 75 km². Die angrenzenden Watten bedeckten 187 km². Nahezu alle Tidemarschen im Elbe-Ästuar sind seit Jahrzehnten Maßnahmen wie Deichbau, Vertiefungs- und Begradigungsmaßnahmen des Flusses und diversen Strombaumaßnahmen ausgesetzt. Diese Maßnahmen beeinflussen die räumliche Sedimentationsdynamik, was zu einer Veränderung der Sedimentablagerung und den damit zusammenhängenden Vegetationsmustern führt. Diese Veränderungsprozesse sind für das Elbe-Ästuar bisher nur sehr unzureichend dokumentiert.

Die Zielsetzung der vorliegenden Doktorarbeit lag zum einen in der Darstellung der Marschentwicklung zwischen 1980 und 2010 und in der Identifizierung der zugrundeliegenden Einflussfaktoren, die für die progressive sowie regressive Sukzessionen der Marschen verantwortlich waren. Ebenso wurden räumlich hochaufgelöste Kurzzeit-Untersuchungen zur Sedimentablagerung in jeweils einer Salz-, Brack- und Süßwassermarsch durchgeführt, um die für Sedimentablagerung wichtigsten Einflussfaktoren zu ermitteln und aus den berechneten Auflandungsraten eine Abschätzung zur Widerstandsfähigkeit der Marschen im Hinblick auf die Meeresspiegelanstiegs-Prognosen geben zu können. Weiterhin wurde ein Vergleich der Effektivität („Fängigkeit“) von häufig verwendeten Sedimentfallentypen in einem Strömungskanal durchgeführt. Hierdurch sollte die Interpretation von Ergebnisse anderer Kurzzeit-Untersuchungen erleichtert werden.

Kapitel 1 enthält eine allgemeine Einführung zu Tidemarschen. Es werden Gefährdungen durch den Meeresspiegelanstieg, Auswirkungen durch menschliche Eingriffe und die verschiedenen Sedimentationsprozesse und zugrundeliegenden Faktoren aufgezeigt und beschrieben. Am Ende folgen die Zielsetzungen der Doktorarbeit und der Beitrag des Autors zu den jeweiligen Kapiteln wird dargestellt.

In **Kapitel 2** sind die Veränderungen und Bewegungsrichtungen der Sukzession in den Tidemarschen der Elbe im Zeitraum von 1980 bis 2010 sowie wichtige Einflussgrößen dargestellt und bewertet. Als ersten Arbeitsschritt wurden die im Gezeitenbereich liegenden Lebensräume in drei Höhen- und drei Salinitätszonen unterteilt. Danach folgten ein Vergleich der Vegetationskartierungen von 1980 und 2010, eine Berechnung der Flächengrößen und eine Darstellung der stattgefundenen Veränderungen zwischen den einzelnen Vegetationszonen. Um die Richtung der zeitlichen Veränderung zu analysieren, wurde eine Unterteilung in progressiver und regressiver Sukzession durchgeführt. Durch die Benutzung von Regressionsbaum-Modellen („Conditional Inference Trees“), konnten die wichtigsten hydro-morphologischen Einflussfaktoren bestimmt werden, die für die progressive und regressive Sukzession der Unteren Marsch verantwortlich waren. Insgesamt wurde zwischen 1980 und 2010 eine Vergrößerung der Elbe-Tidemarschen in Höhe von 2 % (150 ha) festgestellt. Allerdings verteilten sich die Veränderungen ungleichmäßig zwischen den einzelnen Salinitätszonen. Während in den Salz- und Brackwassermarschen eine umfangreiche Vergrößerung festgestellt wurde, nahm die Ausdehnung der Süßwassermarschen deutlich ab. In allen Salinitätszonen kam es zu einer Abnahme der Unteren Marschen. Zusätzlich wurde eine sehr hohe Persistenz der Watten (82–95 %) und der Oberen Marsch (95–97 %) festgestellt, während nur 19–28 % der 1980er Unteren Marschen im Jahr 2010 noch als die selbigen klassifiziert wurden. In den Salz- und Brackwassermarschen wurden mehr als zwei Drittel der 1980er Unteren Marschen im Jahr 2010 als Obere Marschen kategorisiert. Im Süßwasserbereich zeigten nur ein Drittel der 1980er Unteren Marschen diese progressive Entwicklung, 44 % hingegen unterlagen einer regressiven Sukzession in Watten. In den Salz- und Brackwassermarschen wurde der Abstand zur Fahrrinne als wichtigster sukzessionsbeeinflussender Faktor ermittelt. Näher an der Fahrrinne liegende Flächen unterlagen einem höheren Risiko einer regressiven

Sukzession als weiter entfernte Flächen. Im Süßwasserbereich wurden die Uferlage (Prallhang, Gleithang, gerade Fließstrecke) und die Entfernung zur Fahrrinne als wichtigste Faktoren für die Sukzessionsrichtung ermittelt. In dieser Salinitätszone erfolgten die umfangreichen Ausbaumaßnahmen der Elbe, die zu einem starken Abfall des Tideniedrigwassers und Anstieg des Tidehochwassers zwischen 1980 und 2010 geführt haben. Es ist davon auszugehen, dass diese Eingriffe die Verteilung der Marschen negativ beeinflusst und durch großflächige Zunahme der regressiven Sukzession deren Quantität reduziert haben. Die Ergebnisse im **Kapitel 2** liefern wertvolle Erkenntnisse über Wirkungspfade, welche die Sukzession in Marschen beeinflussen. Die gewonnenen Erkenntnisse können dazu verwendet werden, zukünftige, problematische Entwicklungen in ästuarinen Marschen zu erkennen.

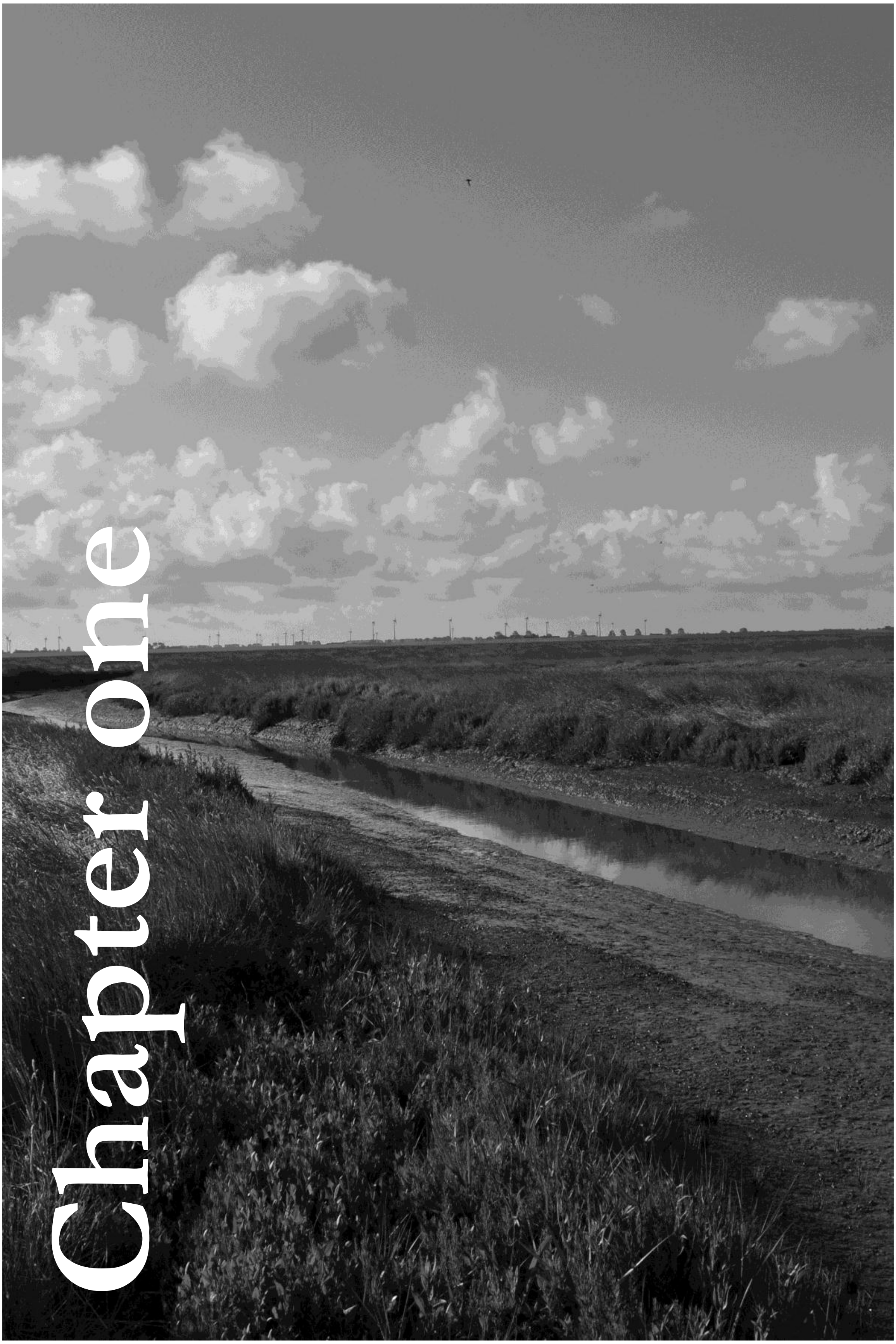
Kapitel 3 beschreibt die Ergebnisse einer einjährigen Sedimentmessungs-Kampagne im Elbe-Ästuar, die in jeweils einer Salz-, einer Brack- und Süßwassermarsch durchgeführt wurde. In dieser Untersuchung wurden räumliche und zeitliche Variationen von kurzzeitigen Sedimentablagerungsraten und die möglichen Einflussfaktoren entlang des Salinitätsgradienten bestimmt. Zwischen März 2010 und März 2011 wurden alle zwei Wochen die Sedimentablagerung entlang von drei Längsprofilen gemessen, welche die unterschiedlichen Geländehöhen (Untere zu Obere Marsch) sowie die Entfernung zur Sedimentquelle (Marschkante und/oder Priel) der drei Untersuchungsgebiete von der Salz-, Brack- und Süßwassermarsch widerspiegelten. Zeitgleich wurden Wasserstandsschwankungen und Schwebstoffwerte des Überflutungswassers während des gesamten Zeitraums aufgenommen. Eine Untersuchung der Vegetationsbedeckung erfolgte im Spätsommer (September 2010) und am Ende des Winters (Februar 2011). Die höchsten jährlichen Sedimentablagerungen (17.5 ± 4.0 kg m^{-2}) und die daraus berechneten Auflandungsraten (20.3 ± 4.7 mm $year^{-1}$) wurden in der Unteren Brackwassermarsch ermittelt. Die Unteren Süßwassermarsch und Salzmarsch wiesen eine um 51 % bzw. 71 % niedrigere Sedimentablagerung gegenüber der Unteren Brackwassermarsch auf. Die höchsten gemessenen Schwebstoffwerte und längsten Überflutungen traten im Herbst und Winter auf. Überflutungsdauer und -häufigkeit nahmen von den Süßwasser- zu den Salzmarschen ab. Die oberirdische Biomasse der regelmäßig überfluteten Pflanzenabschnitte (0–50 cm über der Geländeoberfläche) unterschied sich nicht zwischen den einzelnen Marschtypen, jedoch veränderte sich das räumliche Muster zwischen Spätsommer und dem Ende des Winters. In allen drei Marschtypen nahm die Sedimentablagerung mit zunehmender Entfernung von der Sedimentquelle ab. Die verwendeten multiplen Regressionsmodelle konnten 74 %, 79 % und 71 % der Variation der Sedimentablagerungen in den Süß-, Brackwassermarschen bzw. Salzmarschen erklären. Der Schwebstoffgehalt war immer der wichtigste Einflussfaktor. Die Ergebnisse unterstreichen die Bedeutung von räumlich-zeitlichen Variationen in den Sedimentablagerungsraten und die Betrachtung von einzelnen Einflussfaktoren. Die Ergebnisse aus **Kapitel 3** zeigen, dass die Sedimentablagerungen in den Unteren Marschen des Elbe-Ästuars ausreichend hoch zu sein scheinen, um einen mäßigen Meeresspiegelanstieg ausgleichen zu können. Die Oberen Salzmarschen könnten hingegen durch eine unzureichende Sedimentzufuhr gefährdet sein und sich in Untere Marschen entwickeln.

Kapitel 4 zeigt die Ergebnisse einer Strömungskanal-Studie, in der die Effektivität von verschiedenen Sedimentfallen-Typen getestet wurde. Untersuchungen der Sedimentablagerung sind eine häufig angewendete Methode, um ästuarine Auflandungsraten zu bestimmen und deren Stabilität im Hinblick auf den Meeresspiegelanstieg abschätzen zu können. Fehlende einheitliche Methoden erschweren aber die Vergleichbarkeit verschiedener Studien. In der durchgeführten Studie wurde die Effektivität der „Fängigkeit" von zwei kreisförmigen und zwei ebenerdigen Fallentypen, welche häufig in Kurzzeit-Untersuchungen zum Einsatz kommen, bestimmt und miteinander verglichen. Zwei unterschiedliche Überflutungs-Szenarien (kurz, 37 $\pm$ 2 min und lang, 61 $\pm$ 2 min) und zwei unterschiedliche Schwebstoff-Szenarien (gering, ~65 mg l^{-1} und hoch, ~100 mg l^{-1}) des Überflutungswassers wurden simuliert. Zusätzlich wurden die Auswirkungen dieser Einflussgrößen auf die Sedimentablagerungsraten in Abhängigkeit der Entfernung zur Sedimentquelle (Einlassöffnung des Strömungskanales) aufgezeichnet. Die höchsten Sedimentablagerungsraten wurden mit den kreisförmigen Fallentypen gemessen, die zwischen 20 % und 45 % höher als die Werte der Fußmatten- bzw. der Kachel-Sedimentfallen waren. Alle verwendeten Sedimentfallen-Typen zeigten einen starken Abfall der Sedimentablagerungen mit zunehmender Entfernung zur Einlassöffnung des Strömungskanals. Eine signifikante Wechselwirkung zwischen Fallentyp und Entfernung konnte nicht festgestellt werden. Die Ergebnisse zeigen, dass unterschiedliche Sedimentfallen-Typen sich in ihrer „Fängigkeit" unterscheiden, allerdings sind diese Unterschiede vom Überflutungs-Szenario, Schwebstoffgehalt und Entfernung zur Sedimentquelle unabhängig. Weitere in-situ Untersuchungen sollten durchgeführt werden, um die Vergleichbarkeit von verschiedenen Sedimentfallen-Typen zu erhöhen.

In **Kapitel 5** werden die Ergebnisse dieser Doktorarbeit zusammengefasst und diskutiert.

Die Ergebnisse dieser Doktorarbeit zeigen, dass Sedimentablagerungsraten einer umfangreichen räumlich-zeitlichen Variabilität unterliegen. Die Ablagerung von Sediment unterscheidet sich stark zwischen den Salinitätszonen. Die höchsten Sedimentablagerungsraten sowie die sich daraus ableitende Gebiete mit progressiver Sukzession, wurden im Bereich des ästuarinen Trübungsmaximums der Brackwassermarschen und in Marschen mit vorgelagerten, ausgedehnten Watten gefunden, die sich in großer Distanz zur Fahrrinne befanden. Niedrige Sedimentablagerungsraten und geringe Abstände zur Fahrrinne verursachten einen Rückgang der Süßwassermarschen. Diese Gebiete sind durch umfangreiche menschliche Eingriffe an der Fahrrinne der Elbe geprägt, was sich deutlich in einem Tidenhubanstieg in den letzten Jahrzehnten bemerkbar macht. Insbesondere die Süßwassermarschen sind zusätzlich durch die Folgen des Klimawandels und den beschleunigten Meeresspiegelanstieg gefährdet. Hier bedrohen nicht nur höhere Wasserstände, sondern auch eine Erhöhung der Salinität die Habitatqualität der endemischen Art *Oenanthe conioides* (Schierlingswasserfenchel).

Chapter one



1 General Introduction

1.1 Tidal Marshes – A Short Characterization and Overview

Tidal marshes are vegetated wetland ecosystems dominated by herbaceous plants and occur along shores of coasts and estuaries in the temperate zone. They are often bordered by unvegetated tidal flats in lower elevations and non-wetland ecosystems in higher elevations. An accretion rate sufficient to compensate rising sea-level is crucial for the survival of tidal marshes (Morris et al. 2002). Tidal inundations, salinity of the flooding water, water velocities, as well as river bank morphology, and exposition to navigation channel are major environmental factors influencing vegetation zonation in estuarine marshes (e.g., Odum 1988, Baldwin and Mendelssohn 1998, Butzeck et al. 2014). Adaptations to these physical factors in combination with biotic interactions between different species shape the vegetation zonation along estuarine gradients ('marsh zonation paradigm'; see, Grace and Wetzel 1981, Keddy 1989, Pennings and Callaway 1992, Pennings et al. 2005). Along the longitudinal estuarine salinity gradient, tidal freshwater, brackish, and salt marshes can be distinguished according to the occurrence of characteristic plant species (Engels and Jensen 2009). An elevational gradient perpendicular to the river axis further differentiates habitats into tidal flats in the lowest and twice daily submerged areas, low marshes with pioneer vegetation and periodic inundation, and high marshes with only sporadic inundations (Kötter 1961, Raabe 1986).

The global extent of tidal marshes has yet to be accurately estimated. Salt marshes in North America cover approximately 300,000 km² (Mitsch and Gosselink 2000), which is similar to the total worldwide area of freshwater coastal wetlands and tidal flats estimated by Wolanski et al. (2009). European salt marshes cover around 2,300 km² (Dijkema 1987), from which approximately 20 % are situated along the Wadden Sea coast of the Netherlands, Denmark, and Germany (Esselink et al. 2009). In Germany, the area of tidal marshes along the North Sea coast and the adjacent estuaries is estimated to be 300 km², including the tidal freshwater and brackish marshes of the Weser and Elbe Estuary (see Osterkamp et al. 2001, Esselink et al. 2009). Here, comparatively large tidal marshes (75 km²) are found along the shores of the Elbe Estuary, which is the largest estuary in Germany. The tidal marshes of the Elbe Estuary serve as main study subject in this thesis.

Overall, little attention has been paid to tidal freshwater marshes (defined as tidal marshes with salinities < 0.5 psu); possibly due to the less frequent occurrence of these habitats compared to salt marshes. The limitation of areas with freshwater influx of a river impedes the occurrence of tidal freshwater marshes in arid zones. Information regarding the areal extent of European tidal freshwater marshes is lacking (van den Bergh et al. 2009); although tidal freshwater marshes support hundreds of endangered species, and feature greater fish and bird populations than salt marshes (Odum 1988).

Nowadays, there is a growing appreciation for the importance of tidal marshes in delivering ecosystem services, e.g., storm and flood buffering, erosion control, nutrient cycling, breeding and resting habitats for birds, habitats of endangered and threaten plant and animal species, filter for pollutants, aquifer recharging, and economical uses such as fisheries, livestock farming and agriculture (e.g., Mitsch and Gosselink 2000b, Costanza et al. 1997, Costanza and Mageau 1999, Kirwan and Megonigal 2013). Estimations of the value of the world's ecosystem services by Costanza et al. (1997) illustrated the unique importance of estuarine habitats, which are estimated to be per unit area [US $ ha^{-1} $year^{-1}$] the most valuable ecosystems for nutrient cycles and food production. Therefore, a conservation of these areas should be of major relevance worldwide. Yet, the knowledge necessary for environmental management concerning estuarine marshes and the factors affecting their spatial and temporal change is still scarce.

1.2 Accelerated Sea-Level Rise – Uncertainty of Predictions and Influences on Tidal Marsh Stability

The ability of tidal marshes to maintain positive surface elevation relative to sea level is in part dependent upon sediment-deposition rates and the resulting accretion rates. During the last approximately 5,000 years, eustatic sea-level rise occurred slowly, allowing marshes to accrete at levels sufficient enough to persist (Warren and Niering 1993). Over the period of 1901–2010, global mean sea level rose by 0.19 m, and is expected to increase an additional 0.26 to 0.98 m by 2100 with projected global warming, depending on various interacting factors such as thermal expansion of sea water, melting of glaciers and pack ice (Church et al. 2013). Local and regional sea-level changes differ from global sea-level changes. Wahl et al. (2011) reported a current sea-level rise in the German Bight of about 3.6 ± 0.7 mm $year^{-1}$, based on gauge data from 1971–2008, whereas satellite based linear trends from 1993–2011 showed an annual sea level increase of 3.2 ± 0.5 mm for this area, which is 64 % faster than the highest estimation of IPCC (2007) for the same period (Rahmstorf et al. 2012). Future trends of sea-level rise for the German Bight is expected to be in the range of 0.40–0.80 m by 2100 (approximately 4.4–8.9 mm $year^{-1}$, Gönnert et al. 2009), which is comparable to the expected global trend. Overall, current accelerated sea-level rise is threatening the stability of coastal wetlands (Morris et al. 2002, Neubauer and Craft 2009, Stralberg et al. 2011). However, rates of sea-level rise and the response of tidal marshes differ worldwide. In general, increasing water tables induced by sea-level rise will change inundation parameters in tidal marshes, which are expected to adjust toward a new equilibrium (Morris et al. 2002) if accretion rates are able to compensate sea-level rise.

Tidal marsh vegetation strongly attenuates the hydrodynamics of flow velocity and wave energy (e.g., Temmerman et al. 2005b, Leonard and Croft 2006, Bouma et al. 2007). Tidal marsh vegetation also plays an important role in the evolution of intertidal landscapes (Temmerman et al. 2007) as typical marsh plants act as ecosystem engineering species (Jones et al. 1997, Bouma et al. 2009). Increases of sediment-deposition rates by a positive feedback loop between tidal marsh vegetation, hydrodynamic factors, and sediment-deposition rates are reported by several authors (e.g., Nyman et al. 1993, van de Koppel et al. 2005). Many marshes are characterized by a cyclic sequence of sedimentation and erosion. Sediment deposition usually occurs on the whole inundated marsh platform, while erosion mainly appears along the marsh and/or creek edges as so-called cliff erosion (van Proosdij et al. 2006a). Allen (2000) and van de Koppel et al. (2005) identified cliff erosion as an essential factor for the natural temporal (long-term) dynamics of tidal marshes. After a period of marsh succession and expansion, natural disturbances such as storm surges can initiate cliff erosion (Allen 2000), damaging the vegetation, causing a higher vulnerability of the marsh surface and supporting further erosion (van de Koppel et al. 2005). Increased hydrodynamic energy due to ship waves and increasing currents due to channel engineering may alter hydro-morphology, and thus result in erosion of estuarine marshes and the development of tidal flats. This development from tidal marsh communities to tidal flats can be interpreted as regressive succession. After some time, new pioneer vegetation may establish on these tidal flats in front of marsh cliffs during 'windows of opportunity' (van de Koppel et al. 2005, van der Wal et al. 2008, Balke et al. 2014). The establishment of vegetation then decreases the hydrodynamic energy, increases sediment-deposition rates (Brueske and Barrett 1994), slows down further erosion, and initiates new progressive tidal marsh succession, thus closing the cycle of marsh development.

Gradual decreases and/or subsidence of tidal marshes are mainly reported from organogenic marshes (e.g., Reed 2002), which can be explained by low and insufficient sediment-deposition rates, higher compaction rates of organic rich sediments, and/or by shallow subsidence processes (Cahoon et al. 1995, Turner et al. 2006). In contrast, minerogenic marshes showed lesser rates of mineralization and subsidence (French and Burningham 2003, Nolte et al. 2013b). In these minerogenic marshes, only insignificant amounts of autocompaction were found by Allen (1990) and French (1993). Bartholdy et al. (2010) verified these results, showing significant decreases of bulk density with increasing content of organic carbon.

Many studies have examined sedimentation deposition or accretion rates in salt marshes (e.g., Morris et al. 2002, Nielsen and Nielsen 2002, Reed 2002, Neumeier and Amos 2006, van Proosdij et al. 2006a) whereas studies in tidal freshwater marshes and comparisons between tidal marshes of different salinity zones (Odum 1988, Pasternack and Brush 1998, Neubauer et al. 2002, Temmerman et al. 2003a, 2005a) are scarce. However, tidal freshwater wetlands are extremely vulnerable to rising sea level through increasing inundation and salt water intrusion (Neubauer and Craft 2009). Increases in inundation may lead to a migration of tidal marshes, but flood protection infrastructures such as dikes often limit extensive areas and impede landward movements. In addition, increases of salinity play a decisive role in tidal freshwater marsh stability. Higher salinities and sulfate ions (SO_4^{2-}) from sea water may increase decomposition rates, due to a switch from methanogenis to sulfate reduction, which is the more energy efficient bacterial decomposition process (Capone and Kiene 1988). Additionally, higher salinities will reduce productivity of marsh plants, negatively affecting organic matter accumulation (Neubauer and Craft 2009), and increase the risk of submergence.

1.3 Anthropogenic Threats – Directly and Indirectly Impacts Altered Tidal Marsh Distribution

In addition to sea-level rise, projected higher temperatures, eutrophication, and contamination of sediments may lead to changes of species composition in tidal marshes. Most coastal ecosystems are N-limited (Bertness and Pennings 2000, Rozema et al. 2000), but human activities have multiplied N- and P-fluxes into the oceans, promoting eutrophication in tidal marshes (Mitsch et al. 2001, Howarth et al. 2002). Eutrophication leads to changes in species composition and dominance (Bertness and Pennings 2000, Rozema et al. 2000) which might affect the stability of tidal wetland systems by modifying physiological and morphological processes (Kirwan and Megonigal 2013). Furthermore, hydrocarbon gas extraction (Dijkema 1997), unsustainable groundwater abstraction (Harvey and Odum 1990), and ditching for drainage (Gedan et al. 2009) all increase subsidence rates, and the building of artificial structures such as dams and reservoirs prevent high amounts of sediments from reaching tidal marshes (Syvitski et al. 2005, Kirwan and Temmerman 2009).

Due to the high economic pressure and high population density, it is unlikely that any European estuary exsists in a natural state today (Zonneveld and Barendregt 2009). Morphological adjustments of the navigation channel by engineering activities have changed tidal conditions and altered marsh distribution and quality (Bundesanstalt für Gewässerkunde 2013). Disturbances due to increased hydrodynamic energy of ship waves and increased tidal currents alter hydro-morphology and might result in cliff erosion. All of these activities are strongly connected with changes in tidal levels (Cox et al. 2003, Kerner 2007). The deepening and broadening of estuarine navigation channels led to alterations of velocity and inundations. In the Elbe Estuary, water volume and flow velocity in anabranches have been diminished, causing here an increase of sediment deposition. In the longer term, these alterations might reduce shallow water habitats, which are ecologically important for juvenile fishes (Gerkens and Thiel 2001), and oxygen enrichment of the water column (Kerner 2007). All of these anthropogenic threats might influence the survival of tidal marshes. Hence, comprehensive knowledge of spatial and temporal sedimentation-deposition patterns and the interconnected influencing factors is an important requirement for being able to predict the future of tidal marshes under global climate change.

1.4 Sedimentation – Measuring Surface Elevation Changes and Identifying Driving Factors

Sediment deposition is the gravity based settlement of inorganic and organic particles during inundation. Sediment deposition itself and the controlling factors show a high spatial and temporal variability (e.g., Reed 1989, Allen 2000, Temmerman et al. 2005a, Bartholomä et al. 2009, Nolte et al. 2013a). To assess a marsh's resistance to sea-level rise, it is necessary to determine accretion, which is the vertical adjustment of the marsh surface in millimeters per year $^{-1}$ (Nolte et al. 2013a). Accretion rates are a balance of sediment deposition, erosion, and compaction processes (Neubauer et al. 2002) of organic and mineral particles. Additional processes such as subsurface accumulation of dead biomass (autochthonous sedimentation, Bricker-Urso et al. 1989, French 2006), local shallow subsidence (autocompaction, Cahoon et al. 1995), as well as large-scale glacial isostatic adjustments after the last Ice Age (Vink et al. 2007) also affect changes of marsh surface elevation.

Factors including inundation duration, frequency and height (e.g., Cahoon et al. 1995, Allen and Duffy 1998, Leonard 1997), distance to the sediment source (marsh edge and/or creek, Esselink et al. 1998, Temmerman et al. 2003a), suspended-sediment concentration of the inundation water (Fettweis et al. 1998, Butzeck et al. 2014), and seasonal variations in water levels and wind regime (Neumeier and Amos 2006, van Proosdij et al. 2006a) have all been found to affect sediment deposition. Furthermore, aboveground plant biomass plays an important role, but its importance for sediment deposition cannot yet be generalized. It is clear that standing biomass reduces the energy in the water column of flooding water and causes turbulence (Leonard and Luther 1995), thus, generally increasing sediment deposition, while decreasing erosion and remobilization of sediments (Christiansen et al. 2000, van Proosdij et al. 2006a). The effect of reducing water velocity can be described as the complex function of height, density, and relative stiffness of the vegetation (Boorman et al. 1998). However, the highest sedimentation-deposition rates are often found in low lying stands of pioneer vegetation, which are situated close to the sediment source, and show the highest inundations, although total biomass might be low (see Butzeck et al. 2014).

Depending on the temporal resolution and spatial scale of the study, a wide variety of methods exists for the measurement of sediment deposition and accretion rates (Nolte et al. 2013a). Short-term (tidal to bi-weekly) methods to study sediment deposition include ceramic tiles (e.g., Pasternack and Brush 1998), floor mats (e.g., Lamberg and Walling 1987) or circular sediment traps (e.g., Temmerman et al. 2003a). These inexpensive and easy to use short-term methods can be used to analyze the effects of environmental factors with high spatial and temporal variation on sediment-deposition rates to unravel the complex interrelationships that lead to characteristic deposition patterns in estuarine marshes. Long-term (seasonal to several years) methods to measure accretion rates include sedimentation-erosion bars (van Wijnen and Bakker 2001, Stock 2011), marker horizons (French and Spencer 1993, Bartholdy et al. 2004), sedimentation-elevation tables (Boumans and Day 1993), rod surface-elevation tables (Cahoon et al. 2002), or a joint methodology of marker horizons and sediment-elevation tables (Cahoon et al. 2000). In contrast to short-term methods, often only a low number of samples can be analyzed and, thus, the drivers of spatial and temporal variation in sediment deposition and/or accretion cannot be unraveled. A detailed review of sediment-deposition and accretion measuring methods for different temporal and spatial scales was recently completed by Nolte et al. (2013a).

1.5 Objectives of the Thesis

In times of climate change and sea-level rise, an understanding of patterns in sediment-deposition rates, their underlying mechanisms, and the consequences for marsh stability along estuarine gradients is urgently needed. The aim of the thesis was to analyze spatial and temporal variation in sedimentation and vegetation patterns along the whole salinity gradient of the Elbe Estuary and to determine the underlying environmental factors. Most sedimentation studies in tidal marshes have been carried out in salt marshes, whereas only few studies were conducted in tidal freshwater marshes or compared tidal freshwater and salt marshes. Studies along salinity gradients, comparing sedimentation and vegetation dynamics in tidal freshwater, brackish and salt marshes are almost completely lacking. Additionally, there is a lack of knowledge concerning the effects of multiple and interacting factors for sedimentation and vegetation dynamics in these tidal marshes. As the results of studies on sedimentation-deposition rates might largely depend on the methods which are used, a comparative study on trap efficiencies of different sediment traps was included in the thesis.

The objectives of the thesis can be summarized as follows:

1) Determining the historical and current distribution of tidal marshes within the Elbe Estuary (**chapter 2**).
2) Examining the temporal changes of tidal marshes between 1980 and 2010 within the Elbe Estuary (**chapter 2**).
3) Identifying factors that influence marsh succession (**chapter 2**).
4) Assessing selected anthropogenic impacts on marsh succession (**chapter 2**).
5) Quantifying the spatial and temporal variation in sediment-deposition rates and its predictors (**chapter 3**).
6) Evaluating the relative importance of different predictor factors for sediment-deposition rates (**chapter 3**).
7) Assessing the stability of tidal marshes within the Elbe Estuary under projected accelerated sea level rise (**chapter 3**).
8) Comparing the trapping performance of different sediment-trap types under controlled experimental conditions (**chapter 4**).

1.6 Outline of The Author's Contribution

The thesis consists of five chapters, including a general introduction, three manuscripts, and a synthesis. In the following I provide an overview of cooperation with other scientists and declare my own contribution to this thesis. My own contribution to the thesis includes:

- Writing the general introduction (chapter 1) and synthesis (chapter 5).
- Writing all manuscripts (chapter 2, 3, 4). Jens Oldeland contributed substantially to the sub-chapter "Model description (conditional inference trees)" of chapter 2.
- Sampling, sample processing, data preparation and data analysis for all manuscripts (chapter 2, 3, 4). Uwe Schröder contributed equally to the data preparation for chapter 2. Kerstin Hansen determined the bulk density for chapter 3.
- Preparing maps, figures, and tables for all chapters (chapter 2, 3, 4).

1.7 Reprint permission

Chapter 3 based on the paper:

Butzeck C, Eschenbach A, Gröngröft A, Hansen K, Nolte S, Jensen K (2014) Sediment Deposition and Accretion Rates in Tidal Marshes Are Highly Variable Along Estuarine Salinity and Flooding Gradients. Estuaries and Coasts. doi:10.1007/s12237-014-9848-8.*

Chapter two

2 Vegetation Succession of Estuarine Low Marshes is Affected by Interactions of Distance to Navigation Channel, and Water Level

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Submitted to Journal of Coastal Conservation

2.1 Introduction

Estuarine marshes typically occur along the banks of tidal rivers. Their vegetation and ecosystem properties are affected by several, partly interrelated factors such as inundation, salinity, flow velocity, river bank morphology, tidal amplitude and exposition to the navigation channel. The construction of dikes for flood protection and the creation of agricultural land have reduced the area of estuarine marshes in Northwest-Europe over the last century (Meire et al. 2005, van Koningsveld et al. 2008, Temmerman et al. 2012). During the last 150 years, channel deepening, straightening and other engineering activities have also altered the distribution and quality of estuarine marshes (e.g., the Elbe Estuary) (Bundesanstalt für Gewässerkunde 2013). However, tidal marshes are now considered important for delivering several ecosystem services including storm buffering (Costanza and Mageau 1999, Kirwan and Megonigal 2013). Therefore, their conservation has become a strong common interest. Yet, the necessary knowledge for environmental management concerning estuarine marshes and factors affecting temporal and spatial change is still scarce.

The major factors influencing vegetation zonation in estuarine marshes are tidal inundation and salinity (Baldwin and Mendelssohn 1998), which necessitate specific adaptions of the occurring plant species. Therefore, along the longitudinal estuarine salinity gradient, tidal freshwater, brackish and salt marshes can be distinguished according to the occurrence of characteristic plant species (Engels et al. 2011). Within these marshes, an elevational gradient perpendicular to the river axis further differentiates habitats into bare tidal flats in the lowest and twice daily submerged areas, low marshes with pioneer vegetation and periodic inundation, and high marshes with only sporadic inundations. This zonation along the elevational gradient can be seen as an outcome of progressive succession during which early pioneer species establish on bare tidal flats and later are replaced by high marsh species as a consequence of continual sediment deposition and an associated increase in surface elevation (for salt marshes see, e.g., Olff et al. 1997, Suchrow and Jensen 2010).

Surface-elevation change in estuarine marshes is influenced by various factors such as sediment deposition, above- and belowground plant-biomass production, compaction, and erosion (Bricker-Urso et al. 1989, Neubauer et al. 2002, Nolte et al. 2013b, Butzeck et al. 2014). Furthermore, rates of sea-level-rise (Wahl et al. 2011) and elastic aftereffects of the last Ice Age (Vink et al. 2007) resulting in deep subsidence, play an important role for the fate of estuarine marshes. The spatial pattern of sediment deposition in estuarine marshes is affected by inundation characteristics (e.g., Cahoon and Reed 1995), distance to the marsh edge and to creeks (e.g., Temmerman et al. 2003a), suspended-sediment concentration (SSC, e.g., Fettweis et al. 1998), and by aboveground plant biomass (e.g., Leonard and Luther 1995). Butzeck et al. 2014 found SSC to be the most important factor for differences in sediment-deposition rates along estuarine gradients. While sediment deposition usually occurs on the whole inundated marsh platform, erosion often appears along the marsh margin as so-called cliff erosion (van Proosdij et al. 2006a). Cliff erosion has been identified as an essential factor for natural temporal dynamics of tidal marshes (Allen 2000, van de Koppel et al. 2005). Cyclic sequences of increasing surface elevation and marsh succession on the one hand, and cliff erosion and marsh destruction on the other hand, are typical for many tidal marshes. After a period of marsh succession and expansion, natural disturbances such as storm surges can initiate erosion at the marsh edge (Allen 2000). If the vegetation is damaged, the marsh surface becomes more vulnerable to further disturbances (van de Koppel et al. 2005), enhancing erosion. Anthropogenic disturbances caused by increased hydrodynamic energy due to ship waves, and/or increasing currents due to channel adjustments may alter hydro-morphology, and thus result in cliff erosion of estuarine marshes and the development of tidal flats. This development from tidal marsh communities to tidal flats can be interpreted as regressive succession. After some time, new pioneer vegetation might establish on these tidal flats in front of marsh cliffs during 'windows of opportunity' (van de Koppel et al. 2005, van der Wal et al. 2008, Balke et al. 2014). The establishment of vegetation then decreases the hydrodynamic energy, increases sediment deposition (Brueske and Barrett 1994) slows down further erosion, initiating new progressive tidal marsh succession and closing the cycle.

Progressive and regressive successions are natural processes in floodplains of rivers. Changes in river morphology and tidal amplitude can induce succession; however, changes in flow, sediment and other interacting factors cause instabilities of river shorelines (Rosgen 1994). Hydro-morphologic factors such as distance to the sedimentation source (Esselink et al. 1998), inundation (duration, height, and frequency; Leonard 1997, Allen and Duffy 1998), and SSC (Fettweis et al. 1998, Butzeck et al. 2014) are highly variable in estuarine marshes. Anthropogenic impacts such as channel engineering activities alter marsh distribution and quality (Bundesanstalt für Gewässerkunde 2013) and morphology adjustments of the navigation channel are strongly connected with changes in tidal levels (Cox et al. 2003, Kerner 2007). Deepening and broadening of channels alters hydrodynamics (velocity and inundation), leading to a concentration of the available water volume in the navigation channel. Additionally, water volume and flow velocity in anabranches are diminished, causing an increase of sediment deposition (Kerner 2007). The consequences of these changes are increases in progressive succession and possibly an expansion of tidal marshes. In the longer term, these alterations might reduce shallow water habitats, which are ecologically important for juvenile fishes (Gerkens and Thiel 2001) and for oxygen enrichment of the water column (Kerner 2007).

Marsh succession has been analyzed in previous studies, but these studies only used single marshes for their analysis (e.g., Temmerman et al. 2003b) or considered only one salinity zone (e.g., Field and Philipp 2000). Higinbotham et al. (2004) investigated changes along a 16 km stretch from tidal freshwater to salt marshes, but did not differentiate between habitats along the elevational gradient (bare tidal flats, low marshes, high marshes; hereafter referred to as elevational zones). To our knowledge, no study has been carried out thus far, analyzing tidal marsh succession of elevational zones during several decades along the whole estuarine salinity gradient (including salt, brackish, and tidal freshwater marshes). Additionally, there is a lack of knowledge concerning the effects of multiple explanatory factors and their interactions (Meesters et al. 2007) on pathways of succession in estuarine habitats.

In this study, we aim to identify factors affecting marsh succession in salt, brackish, and tidal freshwater marshes. Therefore, we examined temporal changes of tidal flats into low marshes and of low marshes into high marshes (hereafter referred to as progressive succession), as well as changes of high marshes or low marshes into tidal flats (herafter referred to as regressive succession) during the last 30 years (1978/82 to 2010/11) using vegetation maps and aerial photographs. We applied conditional inference trees to identify the most important factors for low marsh succession. Additionally, we quantified the changes in total area of tidal marshes of the Elbe Estuary.

2.2 Methods

2.2.1 Study Area

The study was carried out at the largest German estuary, the Elbe Estuary (53° 40' N, 9° 31' E, Fig. 2.1), between stream-km 635 to 732. The Elbe Estuary is characterized by a semi-diurnal and meso- to macrotidal regime (Davies 1964). Tidal range increased from 1.8 to 3.6 m at the gauge of Hamburg-St. Pauli during the last century (Bergemann 2006, Kerner 2007), possibly as a result of several anthropogenic changes of the estuarine morphology, including channel deepening, broadening, and straightening (see Arbeitsgemeinschaft für die Reinhaltung der Elbe 2007). 139 km^2 of tidal marsh area were lost due to several embankments from 1896/1905 to 1981/1982 between the city of Hamburg and Cuxhaven (Arbeitsgemeinschaft für die Reinhaltung der Elbe 1984). Variability in freshwater discharge (average discharge into the North Sea is 860 $m^3 s^{-1}$, Bergemann 2006) can cause fluctuations of the upper limit of the brackish water zone in a range of 80 km (Bergemann 2004). The tide at Hamburg St. Pauli shows a pronounced tidal asymmetry with a flood period of approximately 5 hours and an ebb period of approximately 7 hours 20 min (Bundesamt für Seeschifffahrt und Hydrographie 2011) leading to much faster flood than ebb current (Dronkers 1986).

2.2.2 Study Sites

Based on Planungsgruppe Ökologie + Umwelt Nord (PÖUN) (1997), we distinguished three different marsh types along the salinity gradient: tidal freshwater (limnic, < 0.5 psu), brackish (oligohaline, 0.5–5 psu), and salt marshes (meso- and polyhaline, 5–30 psu). In addition, each marsh type was divided into three different elevational zones, tidal flats (TF), low marshes (LM), and high marshes (HM). Marsh types and elevational zones were classified based on vegetation communities, rather than topography, by using existing vegetation maps and aerial images of two time steps (1978/1982, 2010/2011). We defined TF as non-vegetated intertidal habitats, with a lower border of the mean low waterline (Dyer et al. 2000) and an upper border of adjacent tidal marshes. Tidal flats are submerged twice a day, have an upper border around the mean high water line, and are followed by low marshes with pioneer vegetation, mid and high marshes with only sporadic inundations were aggregated together as high marshes in this study. Mean high water, mean low water and tidal amplitude for 1980 and 2010 were calculated from tidal values from the period of 1971 to 1980 (hereafter referred to as 1980), and from the period of 2001 to 2010 (hereafter referred to as 2010). Tidal values are recorded and published by the Waterways and Shipping Office of Hamburg (http://www.portal-tideelbe.de).

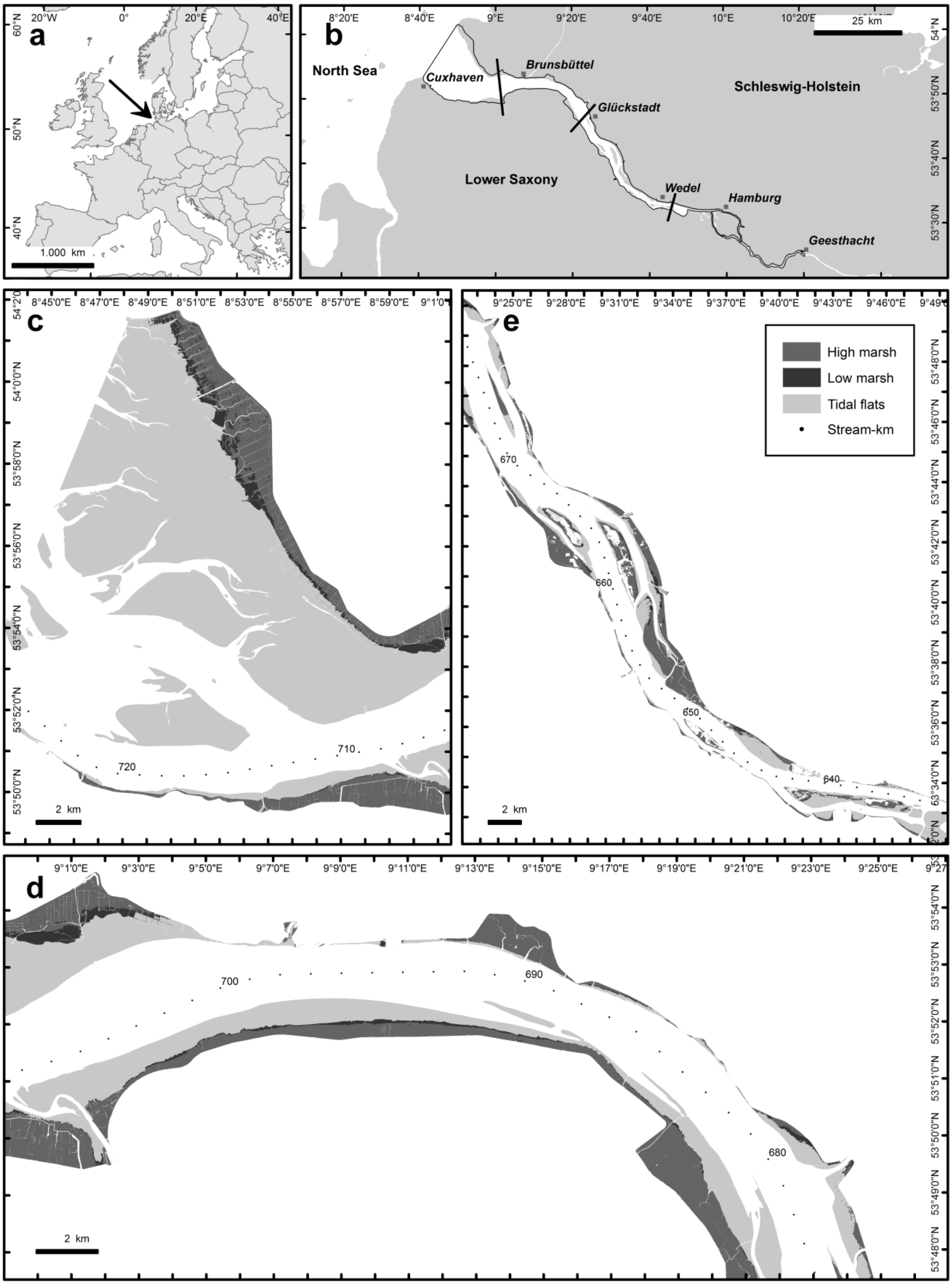

Fig. 2.1 (a) Location of the Elbe Estuary on the German North Sea coast, **(b)** Schematic extent of the Elbe Estuary, **(c)** Salt marshes, **(d)** Brackish marshes, **(e)** Tidal freshwater marshes, with the three elevational zones (tidal flats, low, and high marsh) and the navigation channel stream-km position for the period of 2010, respectively.

Salt marshes were located between stream-km 705 and km 732 (Fig. 2.1c). At the southern bank of the estuary, very few or no TF occurred in front of the salt marshes in 2010, whereas TF covered extensive areas in front of the salt marshes along the northern bank.

Mean high and low water values increased proportionally with sea-level rise during 1980 to 2010 (Fig. 2.2a). *Salicornia europaea* agg., *Spartina anglica* and *Puccinellia maritima* characterized the LM. The HM was mainly dominated by *Elymus athericus* and *Festuca rubra*.

Brackish marshes were found between stream-km 676 and 704 Fig. 2.1d). Here, mean high water level increases between 1980 and 2010 were slightly higher than those found in salt marshes. Mean low water level increased, but not as much as in salt marshes (Fig. 2.1a), and stayed almost the same at the upstream border of this zone in the investigation period. Vegetation of LM was mostly dominated by *Scirpus maritimus,* while in HM patches of *Phragmites australis,* and *Elymus athericus* were intermingled.

Tidal freshwater marshes occurred between stream-km 635 and 676 (Fig. 2.1e). Here, mean high water increased twice as much as in salt marshes and mean low water of 2010 fell below the mean values of 1980 Fig. 2.2a). *Scirpus maritmus, Scirpus tabernaemontani,* and *Typha angustifolia* dominated the LM. Vegetation of HM was dominated by *Phragmites australis*, *Lythrum salicaria*, and/or *Urtica dioica,* while in highest elevations shrubs and remnants of floodplain forests were found. Islands were a special feature in this salinity zone.

2.2.3 Data Processing – Preparing Vegetation Maps

Digital vegetation maps were processed in a geographical information system (ESRI ArcGIS 10.1). The extent and distribution of tidal marshes between Elbe River-km 635 and 732 in 1980 was compared to that in 2010. Two small areas of dike relocations (Hahnöfersand, 105 ha, finished in 2005) were not included. Different standards in mapping accuracy and mapping keys between the 1980 (reference scale 1:50,000) and the 2010 mappings (scale rate during processing 1:2,000) made data harmonizing and generalizing essential. For 2010, we used semi-automatic classified vegetation maps for the salt, brackish and tidal freshwater marshes (see Table 2.1: REF-1, REF-2) which were based on high resolution multispectral (color-infrared) aerial images by Federal Waterways and Shipping Agency Hamburg (WSA Hamburg) with a ground sample distance between 0.15 and 0.25 m. Maps for the southern shore between stream-km 710 and 725 were also created on the basis of aerial images from Federal Waterways and Shipping Agency Hamburg (see Table 2.1: REF-3). For 1980, the analog map of the vegetation of the tidal freshwater and brackish marshes (see Table 2.1: REF-4) was based on vegetation surveys from 1980/1981 by Schoen (1983). We digitized and improved the map by comparing and adjusting aerial images of the same period (see Table 2.1: REF-5). Salt marshes of 1980 were digitized from aerial images (see Table 2.1: REF-6).

Vegetation was classified according to the three differentiated elevational zones. We defined a minimum patch size of 0.01 ha and attached small isolated vegetation patches below this size to adjacent vegetation patches (hereafter referred to as polygons). Linear landscape structures such as drainage ditches or fascines visible in the 2010 mappings were merged with adjacent areas, because these structures were not mapped in 1980. Areas of water, or obviously heavily impacted by human interferences (e.g., dikes, enrockments, harbor areas, strongly influenced parts of islands by depositing dredged material) in at least one period were excluded from analyses. Tidal marshes grazed by livestock and/or mown for agricultural purpose were included in this study.

After classifying elevational zones, maps from 1980 and 2010 were intersected and clipped into a new map, to analyze successional pathways of low marsh between 1980 and 2010 for SM, BM and TFM, respectively.

(i) Progressive succession (TF to LM, LM to HM)

(ii) Regressive succession (HM to LM, LM to TF)

(iii) Persistence (no change of elevational zone)

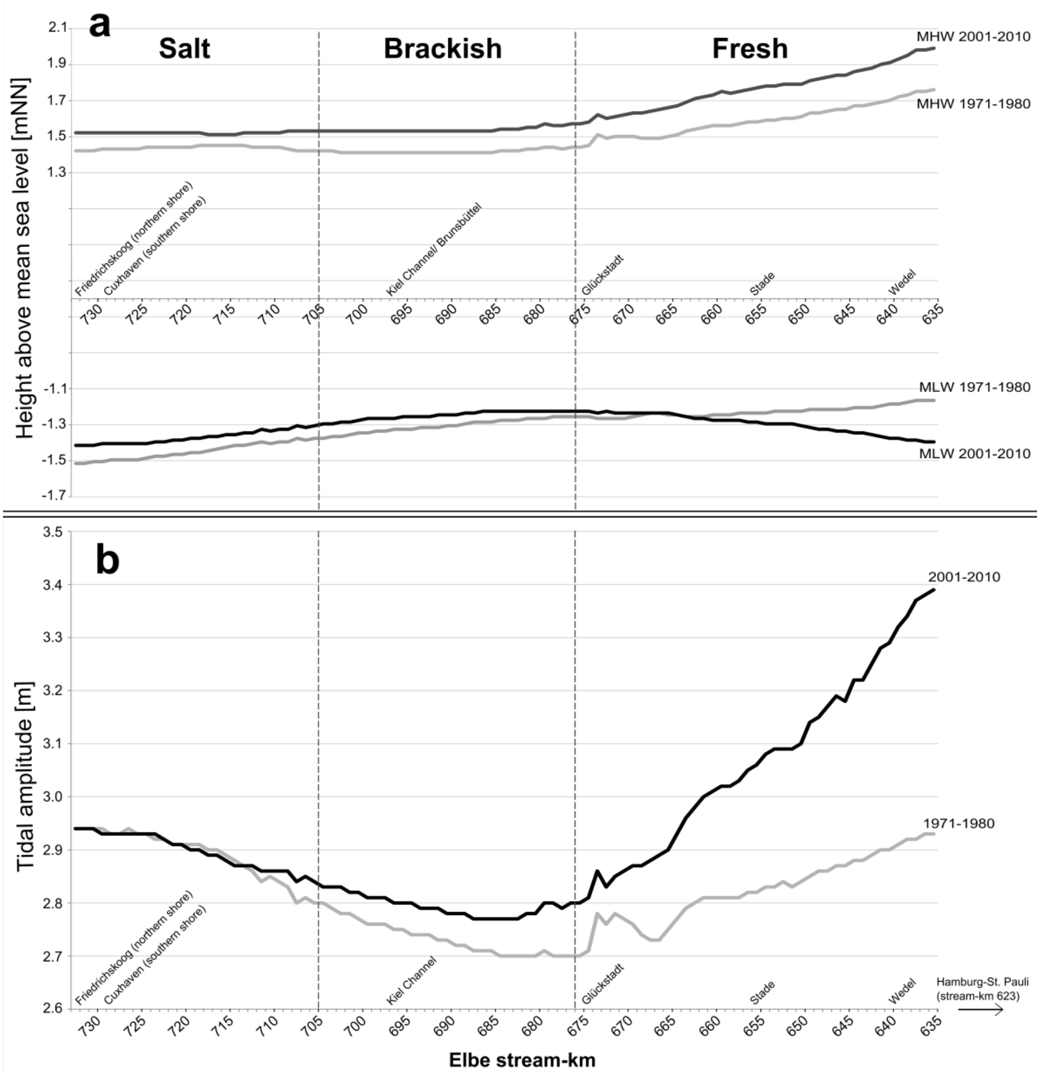

Fig. 2.2 (a) Mean high (MHW) and low water (MLW), and **(b)** mean tidal amplitude per stream-km for 1980 and 2010, respectively. Values for 1980 consist of mean tidal values from 1971 to 1980. Values for 2010 consist of mean tidal values from 2001 to 2010.

Table 2.1 Mappings and aerial images used in the study for the periods 1980 and 2010.

	Years	Type	Salinity Zone	Author/ Authority	Abbreviation
2010					
	2010-2011	mapping	brackish, fresh	Nature-Consult/ German Federal Institute of Hydrology (BfG)	REF-1
	2010-2013	mapping	salt	Leguan/ National Park Schleswig-Holstein Wadden Sea	REF-2
	2010	aerial images	salt (southern shore)	Federal Waterways and Shipping Office Hamburg (WSA Hamburg)	REF-3
1980					
	1980-1981	mapping	brackish, fresh	G. Schoen (1983)/ Waterways and Shipping Administration (WSV)	REF-4
	1978-1979	aerial images	brackish, fresh	Northern Region Office of the Federal Waterways and Shipping Agency (GDWS ASt Nord)	REF-5
	1976-1981	aerial images	salt	Schleswig-Holstein's Government-Owned Company for Coastal Protection, National Parks and Ocean Protection (LKN S-H)	REF-6

2.2.4 Randomized Point Distribution and Determining Morpho-Hydrologic Factors

Effects of environmental conditions on successional pathways of the different elevational zones were quantified with a randomized point distribution in GIS. First, we excluded all polygons with no changes between both periods from further analysis. Afterwards, we generated 6,000 points in residual polygons of which 2,000 were positioned in each salinity zone. Within each salinity zone, 1,000 points were positioned in progressive (*ProSucc*) and regressive (*RegSucc*) polygons, respectively. In addition, anabranches were found within the tidal freshwater zone due to the existence of islands. To detect possible divergent successional pathways for anabranches occurring within the tidal freshwater zone due to the existence of islands, we added 500 randomized points for *ProSucc* and *RegSucc*, respectively at anabranches. The minimum distance between all neighboring points was 10 m to minimize spatial autocorrelation.

Thereafter, five environmental factors were calculated for each of the 7,000 randomized points. (1) Distance to the axis of the navigation channel (*Channel*) was calculated automatically with the GIS. To define (2) the river shore situation (*Shore*), we created cross section areas perpendicular to the axis of the navigation channel every 100 m. The comparison of the opposite areas along the axis allowed us to classify the shore situation into outside, inside, or straight bank. Smaller areas (angles) indicate inside bank situation, bigger areas indicate outside bank situation, and equal areas indicated straight bank situation.

Tidal mean low water (MLW) and mean high water (MHW) levels per Elbe-kilometer were calculated with INFORM 3 (Giebel et al. 2011) for the period 1971–1980 and 2001–2010 by using data from 18 gauges from both bank sides and from the weir in Geesthacht down to the mouth of the estuary. MLW and MHW of both periods were used to calculate (3) the difference of mean low water level *(Dmlw)* and (4) the difference of mean high water level *(Dmhw)*, respectively. In addition, changes in bathymetry of the tidal flats were indicated by (5) shift of nearest distance to the MLW-line *(Dmlw_line)* between 1980 and 2010 (Fig. 2.3).

To achieve this, the MLW-line for 2010 was derived from the digital elevation model of 2010. For 1980, the minimum distance to the edge between tidal flats and permanent water body, which is synonymous with the MLW-line, were extracted from the vegetation map of Schoen (1983). Statistics of these environmental factors are summarized in Table 2.2. In a final step, all points were intersected with the polygons of the vegetation map to add the information of all factors to the point dataset.

We checked for correlation of the environmental factors and excluded *Dmhw* for tidal freshwater marshes and anabranches (Dmlw × Dmhw, $R^2 = 0.84$ and $R^2 = 0.67$, respectively). No correlations between Dmlw and Dmhw were found for brackish and salt marshes ($R^2 = 0.30$ and $R^2 = 0.06$, respectively). Therefore, analysis for tidal freshwater marshes (at the navigation channel) included four environmental factors, while all five environmental factors were used for brackish and salt marshes. We also excluded the factor *Channel* for the anabranch analysis of tidal freshwater marshes.

Table 2.2 (a) Description of the environmental factors used in the study and the differentiated directions of succession.

Name	Description	Type
Factors		
Channel	Distance to navigation channel [m]	Continuous
Dmlw_line	Shift of the shortest distance to the mean low tidal water (MLW)-line between 1980 and 2010, MLW-line_{2010} - MLW-line_{1980} [m]	Continuous
Dmlw	Difference of mean low water level between 1980 and 2010, Dmlw: MLW_{2010}-MLW_{1980} [m]	Continuous
Dmhw	Difference of mean high water level between 1980 and 2010, Dmhw: MHW_{2010}-MHW_{1980} [m]	Continuous
Shore	River shore situation: Inside bank, Outside bank, Straight bank	Categorical
Direction of succession		
ProSucc	Progressive Succession	Categorical
RegSucc	Regressive Succession	Categorical

Table 2.2 (b) Statistics of continuous environmental factors *(Channel, Dmlw_line, Dmlw, Dmhw,* and *Shore)* used in the conditional inference tree models (CTREE), divided into the salinity zones (salt marsh, brackish marshes, and tidal freshwater marshes) and anabranches.

Factors and marsh zones	Mean	Std.dev	Min	Max
Salt marshes				
Channel [m]	8,910	5,793	803	16,060
Dmlw_line [m]	-102	903	-1,362	2,460
Dmlw [m]	0.08	0.01	0.06	0.10
Dmhw [m]	0.09	0.01	0.06	0.12
Brackish marshes				
Channel [m]	2,052	1,112	546	4,938
Dmlw_line [m]	276	434	-230	2,425
Dmlw [m]	0.07	0.01	0.03	0.08
Dmhw [m]	0.12	0.003	0.12	0.13
Tidal freshwater marshes				
Channel [m]	1,179	433	439	2,716
Dmlw_line [m]	-2	163	-580	550
Dmlw [m]	-0.04	0.08	-0.25	0.04
Anabranch				
Dmlw_line [m]	22	66	-206	326
Dmlw [m]	-0.03	0.07	-0.23	0.04

2.2.5 Model Description (Conditional Inference Trees)

In order to identify the most influential factor for separating the two classes, progressive succession (*ProSucc)* and regressive succession (*RegSucc)*, we applied regression tree models. The most common implementation of the Classification and Regression Trees (CART) framework introduced by Breiman (1984) is a recursive binary partitioning algorithm that can be used as a tree-based classifier. Tree based models consist of nested if-else statements and partition the data into smaller subsets which are again partitioned by the most influential factor. The advantages of single tree-based classifiers are their simplicity and their ease of interpretation. However, the classical CART model is known to suffer from several statistical problems; overfitting and a selection bias due to exhaustive multiple comparisons throughout the tree structure (Kuhn and Johnson 2013). Hence, we applied conditional inference tree (CTREE) models as suggested by Hothorn et al. (2006). CTREE is an unbiased recursive partitioning tree-model that relies on penalizing the multiple comparisons based on statistical hypothesis tests, i.e. p-values (Kuhn and Johnson 2013). For technical details see Hothorn et al. (2006). We analyzed the four different subdatasets, i.e. salt, brackish, and tidal freshwater marshes, as well as the anabranches of tidal freshwater marshes with separate CTREE models. The algorithm is implemented in the *ctree* function of the party package (Hothorn et al. 2006) available for the R statistical software environment v.3.0.2 (R Core Team 2014). For each dataset, we split the dataset into training and test dataset with a 7:3 proportion. For the salt, brackish, and freshwater marshes we used the following settings: mincriterion = 0.95, minsplit = 70, minbucket = 35, maxdepth = 5, while for the TFM anabranch data we applied a slightly different setting, due to half as many points (i.e. mincriterion = 0.95, minsplit = 35, minbucket = 18, maxdepth = 5). Both models used the Bonferroni correction for penalizing the multiple comparisons at the splits. The resulting best single tree models were plotted and the tree split structure was interpreted visually.

For validation purposes, we predicted the best model on the test datasets. For the resulting classification tree models we prepared a confusion matrix which displays the number of correct/false classifications. Based on the confusion matrix, we calculated several measures for the classification accuracy, namely Overall Accuracy (OA), Sensitivity, Specificity, Cohens Kappa, and the Area under the Curve (AUC). In brief, OA reports the total number of correctly classified cases; Sensitivity and Specificity measures in percentage how many correct and false responses were classified (see Kuhn and Johnson 2013 for an in-depth description of these values). Cohens Kappa reports a weighted classification benchmark based on the number of observations and number correct/false classifications. Kappa ranges from -1 to 1 with values above 0 being rated as slight, above 0.2 as fair, above 0.4 as moderate, above 0.6 as substantial (“good”), while values above 0.8 are ranked as almost perfect (Landis and Koch 1977, Viera and Garrett 2005). The AUC weights the specificity and sensitivity of the classification and reports a value between 0 and 1; however models with values below 0.5 are worse than a random model.

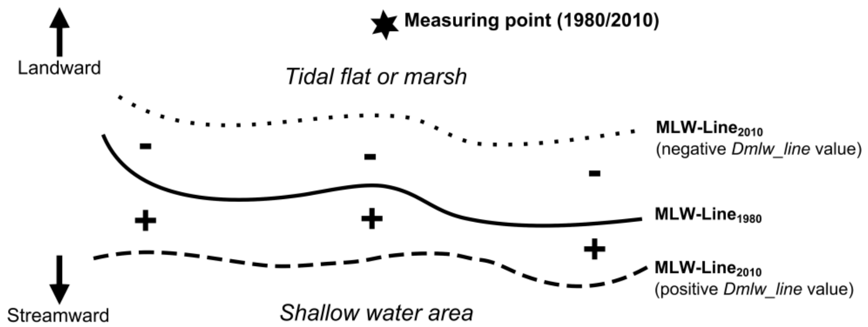

Fig. 2.3 Illustration of bathymetry changes of the tidal flats by the environmental factor change of the distance to the mean low water line (*Dmlw_line*). MLW-line represents the streamward border between tidal flats and the permanent water body. Negative values represent a decrease in distance to the MLW-line between 1980 and 2010. Positive values represent an increase in the distance to MLW-line between 1980 and 2010.

2.3 Results

2.3.1 Proportional Change and Persistence of Marsh Zones

In 2010, about 7,500 ha tidal marshes occurred between the downstream border of Hamburg (stream-km 635) and the mouth of the Elbe Estuary, which was an increase of 2 % compared to 1980. 13 % of these tidal marshes were classified as low marsh. Tidal flats extended to around 18,000 ha and were mostly bordered landward by low marshes. Analyzing the salinity zones showed a decrease of tidal freshwater marshes by 5 % within 30 years, whereas, salt and brackish marshes increased by 8 and 5 %, respectively. With exception of the tidal freshwater marshes, the high marshes increased by 4 to 22 %. However, low marshes decreased between 4 and 30 % in all three salinity zones (Fig. 2.1, Table 2.3).

Overall, low marshes showed the highest percentage of change with only 19 to 28 % of 1980 unchanged (persisted) in 2010. In salt and brackish marshes, 68 and 69 % expanded into high marshes (progressive succession), respectively. Tidal freshwater marshes showed a contrasting succession. 44 % of 1980 low marshes showed a regressive succession into tidal flats in 2010, whereas only 28 % developed into high marshes (Fig. 2.4b). Persistence of tidal flats (82 to 95 %, Fig 2.4a) and high marshes (95 to 97 %, Fig 2.4c) were considerably higher in comparison with low marshes.

Table 2.3 Distribution and marsh surface area [ha] of elevational zones (tidal flats, low marsh, high marsh) of salt, brackish and tidal freshwater marshes in 1980 and 2010 and changes in [%] within these 30 years at the Elbe Estuary (between stream-km 635 – 732), * Tidal flats extent of 1980, especially for the northern shore salt marshes, might slightly differ from reality through coarse nautical chart map resolution and missing area-wide aerial images.

	Tidal Flat			Low Marsh			High Marsh			Sum Tidal Marshes (Low & High Marsh)		
	1980*	2010	Change [%]*	1980	2010	Change [%]	1980	2010	Change [%]	1980	2010	Change [%]
Salt marshes	15,365	14,001	-9	745	522	-30	2,059	2,504	22	2,804	3,026	8
Brackish marshes	2,081	2,202	6	249	238	-4	1,583	1,649	4	1,832	1,887	3
Tidal freshwater marshes	2,018	2,068	2	262	234	-11	2,433	2,334	-4	2,695	2,568	-5

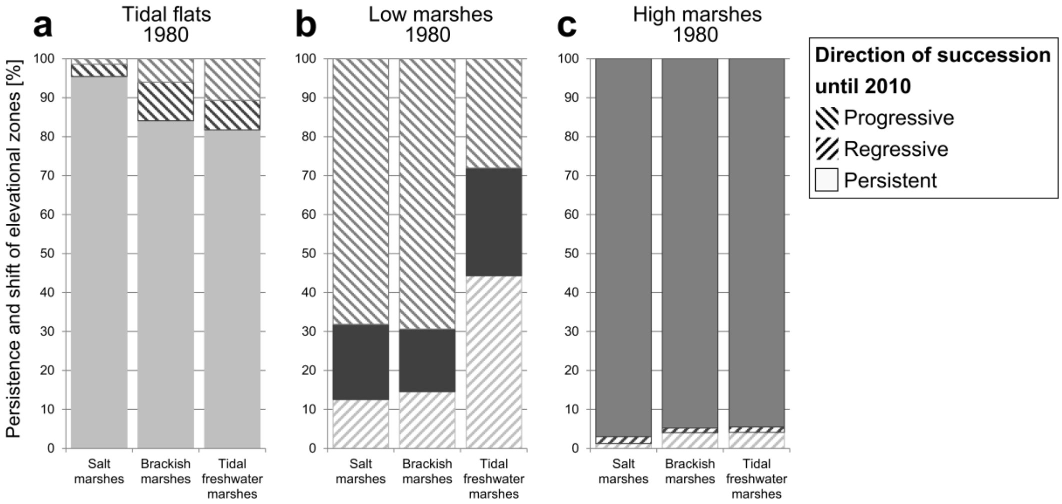

Fig. 2.4 Persistence and temporal changes of **(a)** tidal flats, **(b)** low, and **(c)** high marshes from salt, brackish and tidal freshwater marshes, in the Elbe Estuary between 1980 and 2010. Successional changes for different elevational zones are represented by different colors of patterning. Black-hatched: Succession towards low marsh, dark-grey-hatched: succession towards high marshes, light-grey-hatches: succession towards tidal flats (colors based on Fig. 2.1).

2.3.2 Hydro- Morphologic Factors Influencing the Direction of Succession

Using our model factors, we found a kappa of 0.74 showing a substantial agreement, and correctly classifying 87 % (accuracy value = 0.870, Table 2.4) of succession in low salt marshes of the test data. In brackish marshes, the value for kappa was 0.62 and indicated a substantial agreement (accuracy value = 0.808, Table 2.4). Tidal freshwater marshes and anabranches showed a moderate level of agreement of 0.54 (accuracy value = 0.772) and 0.47 (accuracy value = 0.733, Table 2.4), respectively.

The classification tree of the low marsh succession for salt marshes had six terminal nodes, with half of the terminal nodes classified as regressive succession and half as progressive succession (Fig 2.5a). The best tree for brackish marshes had two terminal nodes with one classified as regressive succession and the other as progressive succession (Fig 2.5b). The best tree for tidal freshwater marshes classified four terminal nodes as regressive succession and six as progressive succession (Fig 2.5c). Finally, the best tree for anabranches classified one terminal node for regressive and progressive succession, in each case.

The CTREE model for salt marshes (Fig 2.5a) showed that in areas up to 6,725 m from the navigation channel, regressive succession dominated, whereas in areas with longer distances, progressive succession occurred. Differences in MHW were determined as second split factor on the left (Fig 2.5a, node 2) and right (Fig 2.5a, node 7) branch. Differences within left and right branching were found in nodes 4 and 8 (Fig 2.5a), where the change of the distance to the MLW-line appeared to influence progressive and regressive succession.

The CTREE model developed for brackish marshes (Fig. 2.5b) identified the distance to navigation channel as the most important explanatory factor. Areas less than or equal to 1,575 m were mainly associated with regressive succession. Regressive succession also occurred in areas of distance greater than 1,575 m from the main channel with less pronounced changes (≤ 0.05 cm) of the MLW (Fig. 2.5b, node 4), whereas MLW changes greater than 0.05 cm promoted progressive succession (Fig. 2.5b, node 5).

In contrast to the salt and brackish marsh CTREE models, river shore situation was the main explanatory factors for tidal freshwater marsh succession. Areas situated along inside banks and with MLW changes less than or equal to -0.05 m showed regressive succession (Fig. 2.5c, node 16), whereas at MLW changes greater than -0.05 m, the distance to the main channel (Fig. 2.5c, node 17), affected regressive (≤ 747 m) and progressive succession (> 747 m). Regressive succession predominantly occurred in areas situated on outside and straight banks with changes of distance to MLW-line between 59 and 81 m (Fig. 2.5c, node 11). In contrast, MLW-line changes greater than 81 m are associated with progressive succession (Fig. 2.5c, node 13, and 14). In areas that show changes of the MLW-line of less than or equal to 59 m (Fig. 2.5c, node 4), the model identified a distance of 1,274 m to main channel as slip-up to divide into the direction of succession. Overall, more regressive succession occurred in areas closer to main channel (Fig. 2.5c, node 5), than in areas further away (Fig. 2.5c, node 8).

Table 2.4 Comparison of model characteristics and model accuracy of the conditional inference tree models (CTREE) for predicting the direction of succession as progressive *(ProSucc)*, or regressive (*RegSucc*) for low marshes at the Elbe Estuary between 1980 and 2010. In each case, the training dataset included 1,400 points and 600 points for the test dataset for each salinity zone. The anabranch dataset included 700 and 300 points for the learning (training) and test (validation) dataset, respectively.

CTREE model		Salt marshes	Brackish marshes	Tidal freshwater marshes	anabranches
	No. of included factors	5	5	4	3
Training dataset	Sensitivity	0.831	0.849	0.84	0.659
	Specificity	0.939	0.798	0.768	0.966
	Overall Accuracy	0.877	0.821	0.799	0.737
	Kappa Value	0.754	0.643	0.599	0.474
	Area under the curve (AUC)	0.887	0.821	0.799	0.737
Test dataset	Sensitivity	0.832	0.854	0.817	0.659
	Specificity	0.917	0.73	0.738	0.938
	Overall Accuracy	0.87	0.808	0.772	0.733
	Kappa Value	0.74	0.617	0.543	0.467
	Area under the curve (AUC)	0.87	0.808	0.772	0.733

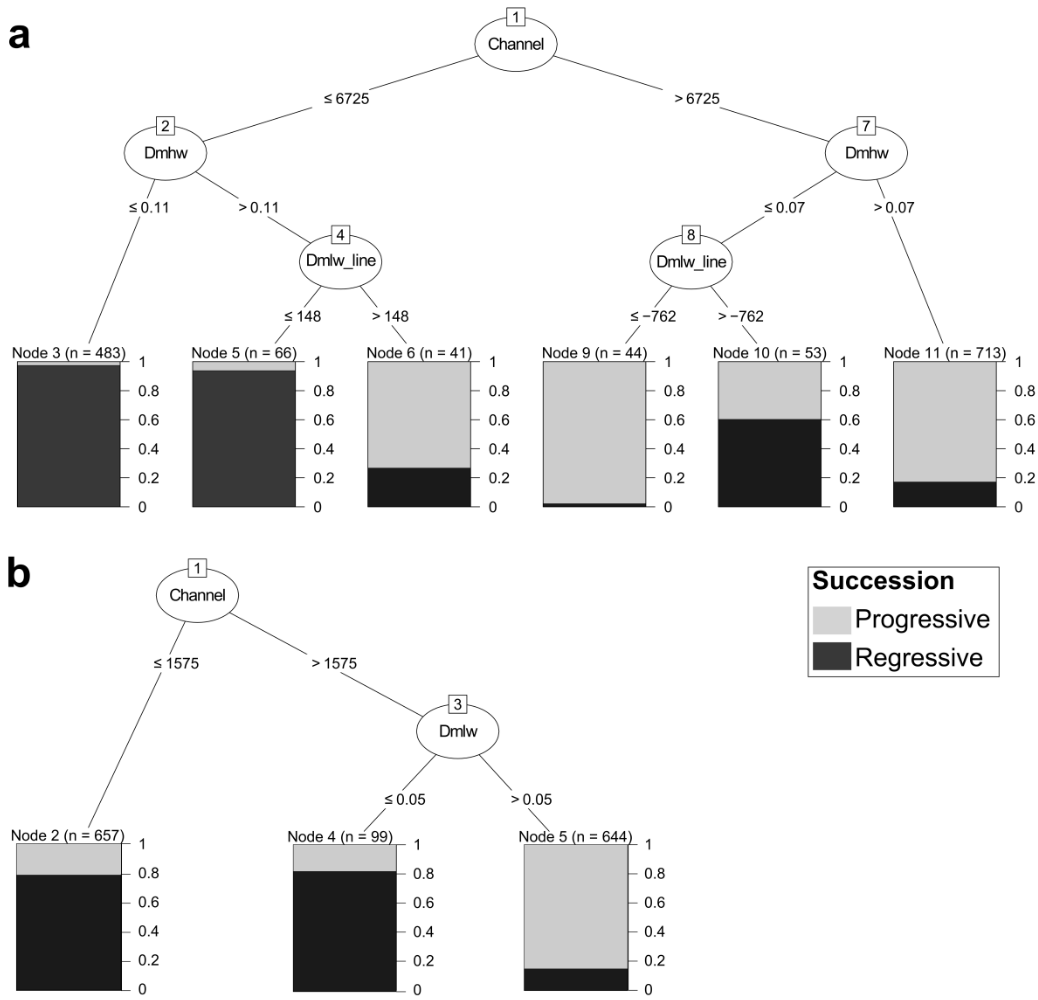

Fig. 2.5 Conditional inference tree models for progressive and regressive succession of the low marsh in **(a)** salt marshes, **(b)** brackish marshes on five environmental explanatory factors. Each of the two models was plotted from the training dataset containing 1,400 cases. The explanatory factors and the points, at which the split-up was made, are written as labels on the branches of the trees. Internal nodes are represented by circles. Rectangles on the bottom of the figure represent the terminal nodes of the tree. The number of cases classified in the terminal node and the percentage of the predicted direction of succession are represented in the terminal nodes. Node numbers (identifier) of internal or terminal nodes are shown above the circles and rectangles, respectively. Abbreviations used for factors are defined in Table 2.2.

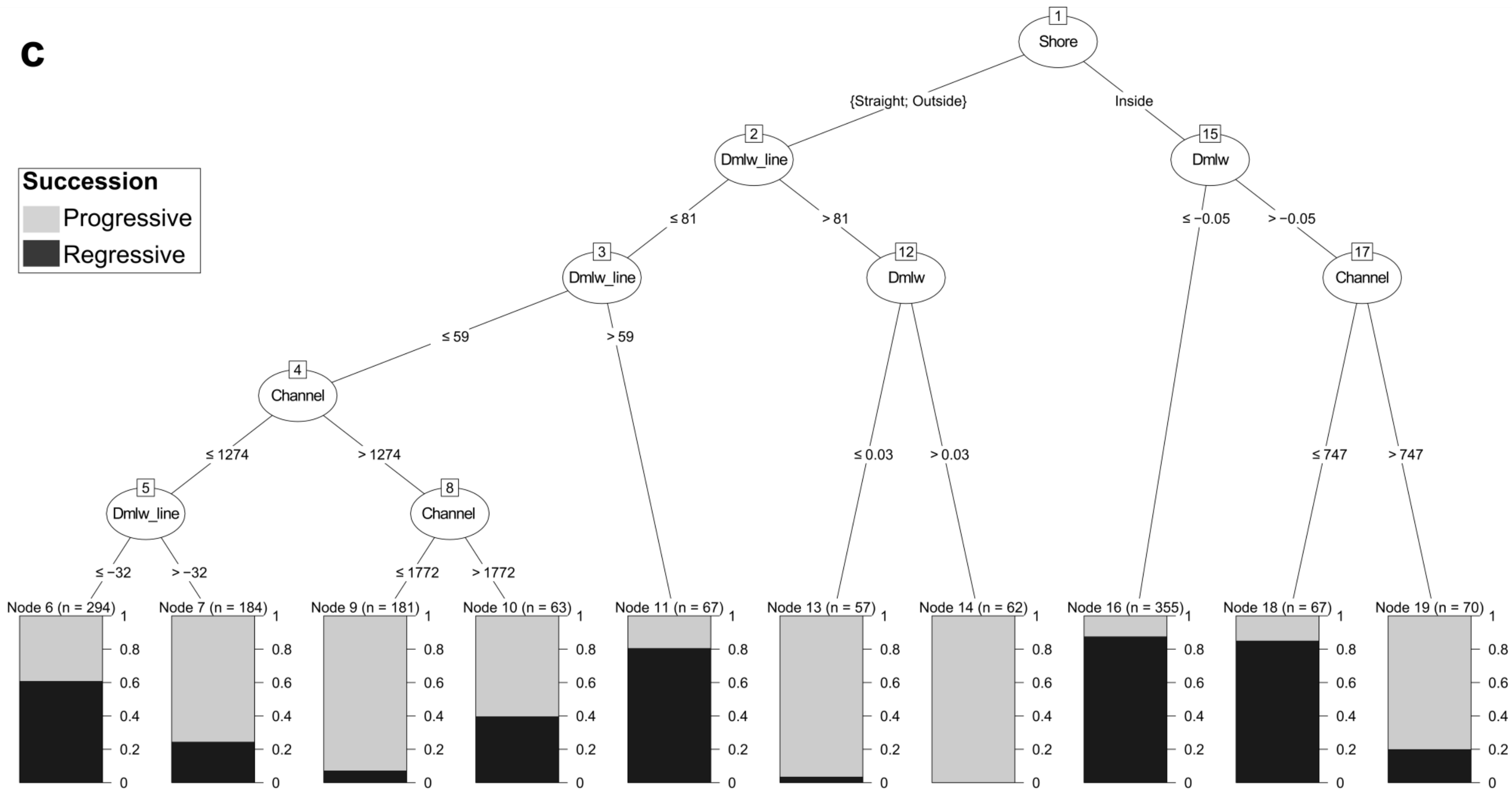

Fig. 2.5 (c) Conditional inference tree model for progressive and regressive succession of the low tidal freshwater marshes based on four environmental explanatory factors. The model was plotted from the training dataset containing 1,400 cases. The explanatory factors and the points, at which the split-up was made, are written as labels on the branches of the trees. Internal nodes are represented by circles. Rectangles on the bottom of the figure represent the terminal nodes of the tree. The number of cases classified in the terminal node and the percentage of the predicted direction of succession are represented in the terminal nodes. Node numbers (identifier) of internal or terminal nodes are shown above the circles and rectangles, respectively. Abbreviations used for factors are defined in Table 2.2.

2.4 Discussion

Total area of salt and brackish marshes increased, whereas tidal freshwater marshes of the Elbe Estuary decreased between 1980 and 2010. Low marshes decreased notably in all salinity zones. Changes in the total area of high marshes differed between salinity zones. For example, high marshes increased substantially in areas within the salt and brackish zone, while, the high marsh area within the tidal freshwater zone decreased. In salt and brackish marshes, substantial amounts of low marshes developed into high marshes, whereas tidal freshwater low marshes were subjected to a regressive succession into tidal flats. CTREE analyses showed the major importance of distance to navigation channel for marsh succession in salt and brackish marshes. Here, the closer the distance to the navigation channel, the higher the risk for regressive succession. In tidal freshwater marshes, river bank situation was identified as major factor for low marsh succession. However, tidal freshwater marshes of the Elbe Estuary were notably affected by channel engineering activities, causing a strong decrease of mean low water, which might have negatively modified the distribution and quality of marshes.

2.4.1 Changes in Tidal Marshes – Differences Between Elevational and Salinity Zones

Comparable changes in tidal marshes were observed for the Dutch and German part of the Wadden Sea, where tidal marshes increased in area by 5 % between two periods (1995/2001 and 2002/2007, Bakker et al. 1993, Esselink et al. 2009) Also, salt marshes in Wales increased around 3 % between 1989 and 2006/2009 (Environment Agency 2011). In contrast, at the North American Delaware estuary, marsh extent remained constant between 1977/1978 and 1997/1998, but a considerable replacement of high marsh by low marsh vegetation was found (Field and Philipp 2000). In the context of sea-level rise (e.g., Reed 2002), gradual decreases and/or subsidence of marshes were mainly reported from organogenic marshes. In these marshes, regressive succession can be explained by low and insufficient sediment-deposition rates and by shallow subsidence processes (Cahoon et al. 1995) which contrasts minerogenic marshes (French and Burningham 2003, Nolte et al., 2013b).

Our approach of analyzing large-scale changes of elevational zones allows us to derive conclusions about sediment-deposition rates in relation to sea-level rise. We can assume that regressive succession indicates insufficient deposition rates (Cahoon and Reed 1995), or local processes of edge erosion (Allen 2000, van Proosdij et al. 2006a). Deposition rates higher than sea-level rise promote progressive succession (Allen 1990). Therefore, the total increase of the marsh areas of the Elbe Estuary can probably be explained by deposition rates that overall exceed sea-level rise. Opposing directions of succession in tidal freshwater marshes, as compared to brackish and salt marshes, might however also be explained by the intensity of direct or indirect anthropogenic impacts such as morphological adjustments of the main channel and anabranches.

2.4.2 Environmental Factors Influencing Succession of Tidal Low Marshes

CTREE analysis showed the major importance of distance to the navigation channel in salt and brackish marshes, and river bank situation for low marsh succession in tidal freshwater marshes. The most important finding was that closer to the navigation channel, there was a higher chance of a regressive succession. It can be assumed that physical forces such as higher flow velocities (Leonard and Croft 2006) and wave activity (Temmerman et al. 2003a) promote regressive succession much stronger in areas situated closer to the navigation channel. These forces are more pronounced over tidal flats and decrease landwards from the vegetated marsh edge (Temmerman et al. 2005b). Different thresholds (splitting points) of distance to navigation channel resulted from differences in the average distances to the navigation channel in salt and brackish marshes. In contrast to salt and brackish marshes, regressive succession prevailed in tidal freshwater low marshes. Here, overall distance to the navigation channel was generally lower compared to salt and brackish marshes, which may help explain the regressive succession that has occurred. CTREE of tidal freshwater marshes showed a similar pattern for progressive and regressive succession for the factor *distance* as found in salt and brackish marshes. However, the importance of the distance to the navigation channel for determining progressive and regressive succession was lower, here.

River shore situation can also be highlighted as major predictor factor for the direction of succession in the tidal freshwater low marshes in our study. Areas located at inside bank situations primarily showed regressive succession, whereas areas at outside and straight banks showed progressive succession. This pattern varies from the situation found in natural rivers (Rosgen 1994) which might be explained by various anthropogenic influences such as the construction of enrockments, spur dikes, and the hydraulic fillings for bank protection at current-exposed areas (Garniel and Mierwald 1996). All these measures have most likely reduced regressive succession and thus affected our results.

Previous studies dealing with marsh succession almost entirely focused on effects of mean high water levels and neglected variations of low water (e.g., Field and Philipp 2000, Temmerman et al. 2003b, Higinbotham et al. 2004). We used the factor *Dmlw_line* to study effects of MLW changes (see below) on marsh succession as an expansion or constriction of tidal flats, which might be directly linked to the direction of succession of adjacent marshes. We found that a pronounced expansion of tidal flats in front of the marsh fostered progressive marsh succession, whereas a marginal expansion or a constriction promoted regressive succession (c.f., Fig 2.5a, internal node 4 and internal node 8, Fig 2.5c, internal node 2). These findings of progressive succession can be explained by the re-suspension of sediment from tidal flats which can then be deposited on the marsh (Uncles and Stephens 2010). In addition, wave energy dissipation by large tidal flats in front of the marshes can prevent regressive succession by attenuation of wave energy (Möller and Spencer 2002).

We identified distance to navigation channel, shift of the MLW-line, and river bank situation as most important factors for the explanation of progressive and regressive succession. However, our results proved hydrological factors as decisive as well. The hydrological system of the Elbe Estuary was altered by major engineering measures in 1974–1978 and 1998–2000 (Arbeitsgemeinschaft für die Reinhaltung der Elbe 2007), causing changes in MHW, MLW, and flow velocity (Kerner 2007, Niemeyer 2001).

Dmhw and *Dmlw* were found to be important for the direction of succession in all salinity zones and anabranches. Tidal values were also previously found to influence vegetation composition of tidal marshes. Tidal values can also serve as an indicator for variability in inundation frequency, duration, and depth (Field and Philipp 2000). Furthermore, increases of high water levels are also associated with higher deposition rates, which in turn promote succession (e.g., Morris et al. 2002, Bartholdy et al. 2004). In salt marshes, increases of MHW and MLW during the study period corresponded fairly well with the current mean sea-level rise of 3.6 ± 0.7 mm year^{-1} described for the area by Wahl et al. (2011). Under natural conditions, a similar relationship between sea-level rise and MLW and MHW should be expected further upstream in the brackish and freshwater zone. Yet, here an increase of MHW and, in contrast to expectations, a decrease of MLW was found, probably due to the engineering manipulations on the river. These changes of MHW, and especially MLW, were most pronounced in the tidal freshwater zone, where the width of the Elbe with adjacent tidal flats and marshes is much smaller than in the salt and brackish salinity zone. Anthropogenic interferences in these areas in turn might have had a strong impact on marsh persistence and the direction of succession as substantial changes of MHW and MLW between 1980 and 2010 in tidal freshwater marshes are associated with a high percentage of regressive succession here.

The trend that stronger increases of MHW promote progressive succession in salt marshes found in this study was previously described by other authors (e.g., Olff et al. 1997). Increases of MHW prolong inundation, which increases the opportunity of suspended sediment to deposit on the marsh surface, and results in an increase in elevation. Surprisingly, no influence of MHW on the direction of succession was found for brackish marshes in our study. Here, a more pronounced increase of MLW led to a higher probability of progressive succession. The increase of MLW seems to be associated with an increase in deposition rates and thus also promotes progressive succession, similar, to the positive effect of increases of MHW on sediment-deposition rates.

Strong increases of tidal amplitude (increase of MHW together with a decrease of MLW, cf. Fig 2.2b) and earlier described anthropogenic impacts were previously found to be related to regressive succession (Cox et al. 2003). In tidal freshwater marshes, regressive succession of low marshes seems to be slightly related to decreases of MLW. This relationship might be explained by an increase in flow velocity, especially in tidal freshwater marshes (Kappenberg and Grabemann 2001), which promote erosional processes of marshes and tidal flats. The opposite pattern was found in anabranches in tidal freshwater marshes. Here, progressive succession in areas with stronger decrease of MLW might be explained by a decrease of the frequency and duration of submergence promoting a streamward succession of elevational zones and an enlargement of tidal flats in these slow-flow areas.

2.5 Conclusions

The results of this work provides a comprehensive inventory of tidal marsh extent of the Elbe Estuary between 1980 and 2010, including detailed data of change and persistence of elevational and marsh zones in the three types of differentiated salinity zones (tidal freshwater, brackish and salt marshes). We demonstrated that the direction of succession can be explained by changes in estuarine morphology and in tidal values. In tidal freshwater marshes, where the most extensive channel engineering activities were conducted, considerably decreases in areal extent occurred. As planned future channel engineering activities are likely to further influence marsh succession and persistence, further studies are necessary to understand and predict these developments. Future assessments of tidal marshes of the Elbe Estuary will fortunately not be constrained by methodological problems, as the Trilateral Monitoring and Assessment Program (TMAP) of the Wadden Sea, as well as the river basin management plan of the EU Water Framework Directive (WFD) will provide maps on a common methodological basis. This will enable to identify pathways of marsh succession at a high spatial and temporal resolution and can assist to spot problematical developments of tidal marshes, also in relation to channel engineering work, which is regularly conducted in many estuaries worldwide.

2.6 Acknowledgements

I would like to thank the Federal Waterways and Shipping Office Hamburg (WSA Hamburg), Northern Region Office of the Federal Waterways and Shipping Agency (GDWS ASt Nord), and Schleswig-Holstein's Government-Owned Company for Coastal Protection, National Parks and Ocean Protection (LKN S-H), especially Martin Stock, for providing existing maps and aerial images. I thank all colleagues, especially Stefanie Nolte for her helpful comments on the manuscript, and Franziska Rupprecht for her R-statistics and CTREE introductions. The work was funded by the Federal Ministry for Education and Research within the research project KLIMZUG-NORD and supported by the Estuary and Wetland Research Graduate School Hamburg (ESTRADE) as member of the State Excellence Initiative (LExI) funded by the Hamburg Science and Research Foundation. Additionally, I thank Jenny Allen for proofreading.

Chapter three

3 Sediment Deposition and Accretion Rates in Tidal Marshes Are Highly Variable Along Estuarine Salinity and Flooding Gradients

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Published: Estuaries and Coast (2014). The final publication is available at Springer via http://dx.doi.org/10.1007/s12237-014-9848-8.

3.1 Introduction

Tidal marshes are vegetated intertidal ecosystems in which differences in salinity and tidal flooding regime are mirrored in vegetation composition (Engels and Jensen 2009). In times of global climate change, rising sea levels and saltwater intrusion might threaten these ecosystems (Neubauer and Craft 2009). The ability of tidal marshes to maintain positive surface elevation relative to sea level is in part dependent upon sediment deposition rates and the resulting accretion rates. So far, some studies revealed insufficient accretion rates (e.g., Bakker et al. 1993, van Wijnen and Bakker 2001), whereas others found accretion rates high enough to compensate moderate rates of sea-level rise (e.g., Morris et al. 2002, Neubauer et al. 2002, Temmerman et al. 2004). Most sedimentation studies in tidal marshes, however, have been carried out in salt marshes (e.g., Morris et al. 2002, Nielsen and Nielsen 2002, Reed 2002, Neumeier and Amos 2006, van Proosdij et.al 2006a), whereas only few studies were conducted in tidal freshwater marshes (e.g., Pasternack and Brush 1998, Neubauer et al. 2002), or compared tidal freshwater and salt marshes (Odum 1988, Temmerman et al. 2003a, 2005a).

Sediment deposition is the process of settlement of inorganic and organic particles during inundation by gravity (e.g., Allen 2000, Temmerman et al. 2005a, Bartholomä et al. 2009, Nolte et al. 2013a) and is the outcome of complex interactions of various factors with high temporal and spatial variability. However, to assess marsh resilience to sea-level rise it is necessary to investigate accretion. Accretion rates, in contrast to sediment deposition rates, describe vertical adjustments to a specific soil layer in millimeters per year $^{-1}$ (Nolte et al. 2013a), as a balance of deposition, erosion and compaction processes (Neubauer et al. 2002) of organic and mineral particles, and interstitial water. Additionally, accumulation of dead belowground (roots and rhizomes, Bricker-Urso et al. 1989) and aboveground biomass like stems, leafs, shells, and snails have to be considered. It is possible to calculate accretion rates based on sediment deposition measurements and soil bulk density (Nolte et al. 2013a).

Factors such as timing, frequency, and height of inundations (e.g., Cahoon and Reed 1995, Leonard 1997, Allen and Duffy 1998), distance to the sediment source (Esselink et al. 1998, Temmerman et al. 2003a), variability in suspended-sediment concentration (SSC) of the flooding water (Fettweis et al. 1998), and seasonal variations in water levels and wind regime (Neumeier and Amos 2006, van Proosdij et al. 2006a) have all been found to affect sediment deposition. Furthermore, aboveground plant biomass reduces the energy in the water column of flooding water (Leonard and Luther 1995), thus, generally increasing sediment deposition and decreasing erosion and remobilization of sediments (Christiansen et al. 2000). In contrast to summer, higher rain falls and less vegetation cover during winter could abate terrestrial erosion (Fettweis et al. 1998). To our knowledge, no study with a high temporal and spatial resolution of sediment-deposition rates in tidal marshes along the whole estuarine salinity gradient (including tidal freshwater, brackish, and salt marshes) has been carried out up to now. It could be expected, however, that the high spatial variation of SSC in estuarine waters might lead to strong differences in sedimentation deposition rates between different salinity zones.

In our research, we therefore want to quantify the variability of sediment-deposition and accretion rates in space and time in estuarine marshes. To quantify sediment-deposition rates, a wide range of measuring techniques for different temporal and spatial scales exists (Nolte et al. 2013a). Methods for measuring short-term (tidal to bi-weekly) sediment-deposition rates include ceramic tiles (Pasternack and Brush 1998, Christiansen et al. 2000, Neubauer et al. 2002) or circular sediment traps (Temmerman et al. 2003a). These sediment traps can be used in large numbers. Therefore, they allow analyzing effects of environmental variables with high spatial and temporal variations to understand the complex interrelations leading to sediment-deposition patterns in estuarine marshes. On the long-term (annual to several centuries), various methods are available to measure accretion rates such as, sedimentation-erosion bars (SEB, van Wijnen and Backer 2001), marker horizons (French and Spencer 1993, Bartholdy et al. 2004), sedimentation-elevation tables (SET, Boumans and Day 1993), rod surface-elevation table (RSET, Cahoon 2002), or a joint methodology of marker horizons and SET by Cahoon et al. (2000). However, using these long-term methods, often only a low number of samples or sites can be analyzed and, thus, the driving forces of spatiotemporal variation in sediment deposition cannot be unraveled. To investigate the small scale spatiotemporal factors driving sediment deposition on the one hand, and to assess the resulting accretion on the other hand, we calculated accretion rates using the sediment-deposition rate and the soil bulk density. Various error sources need to be considered when applying this approach. For example, the approach is not able to account for the deposition of dead organic material such as dead roots. Investigations by Nyman et al. (2006) showed no increase in accretion with increasing sediment-deposition rate. However, this study was conducted in an organogenic marsh, where organic deposition forms the main source of accretion. In contrast, mineral sediment deposition plays a key role in minerogenic marshes, such as those studied here.

In this study, we quantified the variability of bi-weekly spatial and temporal patterns of sediment-deposition rates and calculated accretion rates in three marsh types (tidal freshwater, brackish, and salt marsh) along the salinity gradient at the Elbe Estuary (Germany). At each marsh type, the within marsh spatial heterogeneity was considered by measuring sediment-deposition rates along three transects spanning the elevational gradient from low to high marsh zones. Simultaneously, we measured SSC of the flooding water, inundation parameters, distance to the sediment source and aboveground plant biomass as predictor variables and determined their relative importance for sediment-deposition rates with multiple regression models.

We aimed to (i) quantify spatial and temporal variations in sediment-deposition rates and its predictors, (ii) evaluate the relative importance of different predictor variables for sediment-deposition rates, and (iii) estimate whether sediment-deposition and calculated accretion rates in estuarine marshes of the Elbe Estuary are sufficient to compensate predicted sea-level rise.

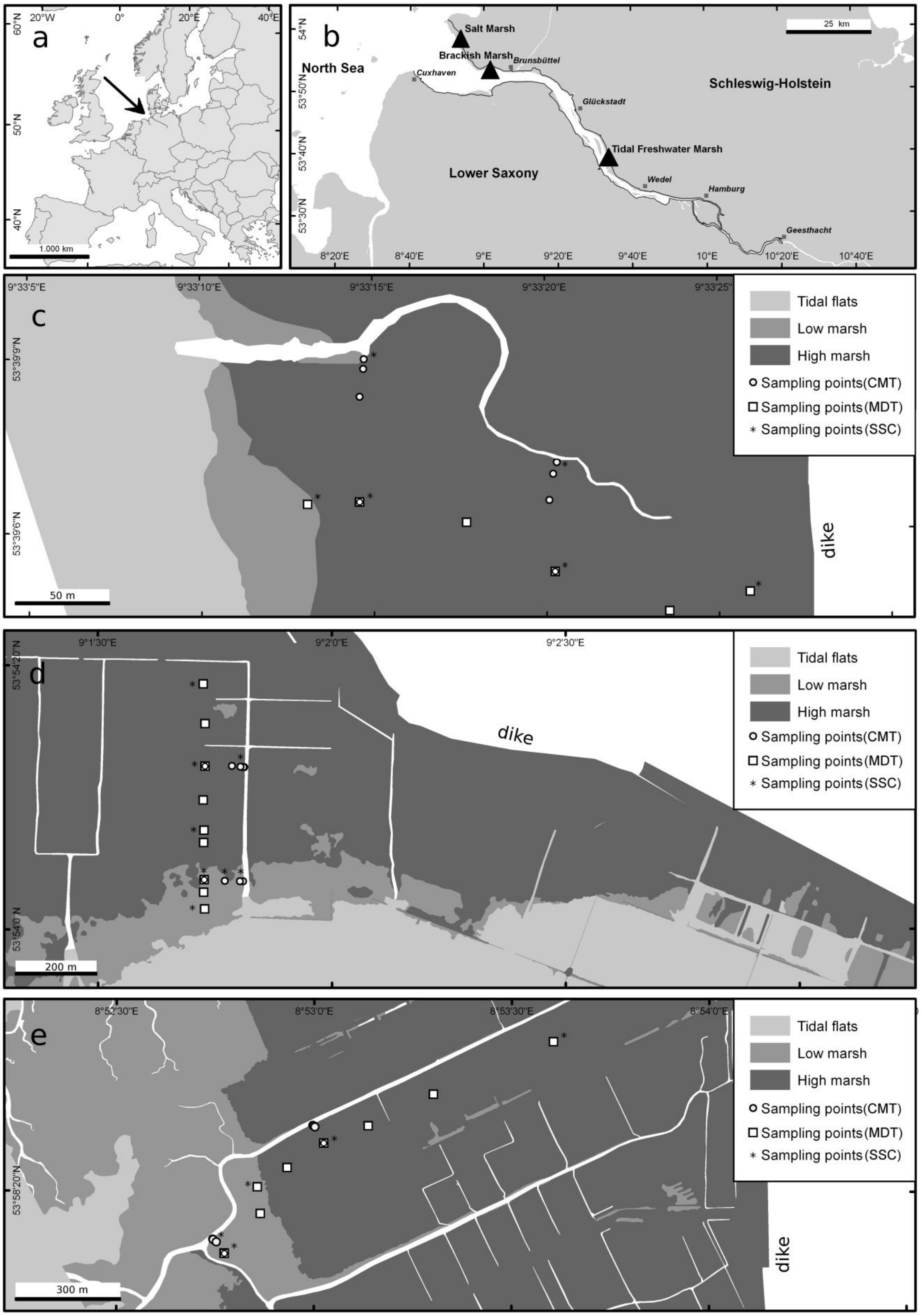


Fig. 3.1 (a) Location of the Elbe Estuary at the German coast of the North Sea, **(b)** Location of the three studied marshes, represented by black triangles, **(c)** Tidal freshwater marsh, **(d)** Brackish marsh, **(e)** Salt marsh with vegetation zones and arrangement of measuring points at marsh-dike-transect (MDT) and creek-marsh-transects (CMT). Additionally, suspended-sediment concentration (SSC) sampling points were marked with an asterisk.

3.2 Methods

3.2.1 Study Area

The Elbe Estuary (53° 40' N, 9° 31' E) in northern Germany is the largest estuary at the German coast. It is characterized by a semi-diurnal and meso- to macrotidal regime (Davies 1964), with wide tidal flats in some areas. The average discharge rate at the entry to the North Sea is 860 $m^3 s^{-1}$ (Bergemann 2006). Mean tidal range varies from 2.9 m at the mouth near Cuxhaven, increases to 3.6 m at Hamburg-St. Pauli and decreases to 2.5 m (BSH 2010) at the upstream border of the estuary (weir at Geesthacht), which is 142 km away from the mouth. Anthropogenic activities like channel deepening and dredging, sand extraction, and the construction of dikes have highly influenced the Elbe Estuary. Embankments between 1896/1905 and 1981/1982 caused a marsh area reduction of approximately 65 % between the city of Hamburg and Cuxhaven (ARGE Elbe 1984). Additionally, dredging altered the width of the shipping lane and minimum water depth from 4.5 m (1843) to 14.5 m (1999) below the mean low-water line. These anthropogenic impacts induced a gradual increase of the tidal amplitude from 1.8 to 3.6 m in Hamburg within the last 150 years (Bergemann 2006, Kerner 2007) and caused changes in tidal current with a much faster flood than ebb current.

In 2011, tidal marshes covered an area of 75 km^2 and adjacent tidal flats extended over 187 km^2 between the city of Hamburg and the mouth of the estuary. Tidal marsh vegetation composition along the Elbe Estuary is mostly affected by soil salinity (Kötter 1961), which is highest at the mouth of the estuary and decreases upstream, creating distinct marsh types along the estuarine salinity gradient (salt marshes, brackish marshes, tidal freshwater marshes). Differences in elevation and inundation (frequency, duration, height) cause distinct vegetation zones from tidal flats via low and mid to high marshes (Engels and Jensen 2009).

3.2.2 Study Sites

We selected study sites from three different salinity zones according to the vegetation composition (see Engels and Jensen 2009), namely one tidal freshwater, one brackish and one salt marsh (Fig. 3.1), to investigate sediment-deposition rates (SDR). The sites were situated at slow-flow sections with large bare tidal flats (salt and brackish marsh) in front of the seaward marsh edges or at a side-channel (tidal freshwater marsh). In this study, we classified low marshes (-low), and aggregated mid and high marshes together as high marshes (-high). All study sites had similar creek systems with one major creek and additionally only silted up minor creeks. Furthermore, no management (farming, grazing or mowing) was carried out on the studied marshes and no artificial embankments structures were present at the marsh edges.

The studied tidal freshwater marsh (TFM, Haseldorf, 53° 39' 1" N, 9° 33' 13" E, Fig. 3.1c) is located south of the river Pinnau in the nature conservation area „Haseldorfer Binnenelbe mit Elbvorland“. Mean salinity of creek water during the study was 0.5 (own continuous conductivity measurements), measured with a water gauge (Schlumberger Water Services, Delft, Netherlands: CTD-Diver) and data were conversed into the practical salinity unit (psu) after Bergemann (2005). Tidal amplitude and mean high water (MHW) were 3.1 and 1.8 m above NHN (German standard reverence level), respectively (BSH 2010). A muddy tidal flat bordered the 350-m wide marsh. We estimated that 20 % of the total marsh area was situated below MHW. In TFM-low, the foremost 15 m consisted of stands of *Schoenoplectus lacustris* (no measurements were conducted in this area), followed by a 35 m wide zone with *Phragmites australis* and *Caltha palustris*. TFM-high was dominated by *P. australis*. Close to the dike, vegetation at TFM changed into willow shrubs.

A site south of the port of Neufeld served as brackish marsh (BM, Neufeld, 53° 54' 4" N, 9° 1' 41" E, Fig. 3.1d). Here, average creek water salinity was 5.5. Tidal amplitude was 2.9 m, with a MHW of 1.6 m above NHN (BSH 2010). The 750-m wide marsh is bounded by a tidal flat of 1,600 m; roughly 20 % of the total marsh area was lower than MHW. The foremost parts of BM-low were dominated by *Cotula coronopifolia*. Our measurements in BM-low were carried out in a 125-m wide *Bolboschoenus maritimus* stand. Following, dense stands of *Phragmites australis* in the first 300 m and patches of *P. australis* and *Elymus athericus* alternated in the uppermost 200 m of BM-high.

The salt marsh site (SM, Dieksanderkoog, 53° 58' 9" N, 8° 53' 23" E, Fig. 3.1e) is located in the Schleswig-Holstein Wadden Sea National Park with an average creek water salinity of 18.7. Mean tidal amplitude next to the site was 2.9 m, with a MHW of 1.5 m above NHN (BSH 2010). A tidal flat of 8,000 m bordered the 2,000 m wide marsh. Approximately 10% of the total marsh area was located below MHW, but sediment deposition measurements were only performed at elevations above MHW, because of accessibility restrictions. *Salicornia europaea* agg., *Spartina anglica* and *Puccinellia maritima* characterized SM-low. The high marsh was dominated by *P. maritima* and *Aster tripolium* in the lower parts and dense stands of *E. athericus* and *Festuca rubra* in higher parts.

3.2.3 Sediment Deposition Measurements and Processing

At each study site we investigated the bi-weekly (14 ± 1 days, full neap-spring sequence) sediment deposition pattern along three transects to study spatial heterogeneity (see French and Spencer 1993, Temmerman et al. 2003a) depending on elevation and distance to sediment source (main creek, marsh edge, Fig. 3.1c–e). One long transect (marsh-edge-dike transect (MDT)) was installed starting at the marsh edge and continuing to the dike, parallel to the main creek of the study site. Perpendicular to the MDT, two shorter transects (creek-marsh transect (CMT)) were established. In total, we measured sediment deposition at 39 sampling points (SM, 12; BM, 15; and TFM, 12) with two traps at each sampling point (distance 0.5 m). The distance between adjacent sampling points of the MDT was increasing with increasing distance to the marsh edge (SM, 75–350 m; BM, 25–100 m; and TFM, 25–75 m). The traps at the CMT were also placed with increasing distance to the creek (2.0, 5.0, 15.0 m). The last sampling points of these CMT were located on the MDT (SM, 70 m; BM, 90 m; and TFM, 80 m). Each sampling point was assigned to a marsh zone according to plant species composition (low marsh, high marsh).

The circular sediment traps were made of plastic pots with an internal diameter of 18.9 cm and a rim of 3 cm. Traps were placed on the marsh surface to reduce lateral bed load transport of sediments into the traps (Neubauer et al. 2002). We used a floatable lid to protect the collected sediment in the trap from heavy rain events (Temmerman et al. 2003a). A plastic stick (length, 100–120 cm) was used to fix the trap to the marsh surface, which further allowed the lid to move up while the trap was inundated. When the water table moved down, the lid dropped down and sediment was sheltered in the trap. The sediment traps were emptied and replaced during ebb bi-weekly between March 2010 and March 2011. The trapped sediment was rinsed with water and a brush into a plastic bag and brought to the laboratory. To separate occurring larger organic particles (plant remains, insects, and seashells), sediment samples were processed by wet sieving (mesh size of 630 μm) in the laboratory. Rinsed sediment samples were collected and left to rest for three days. Thereafter, samples were dried until constant weight at 105°C. Finally, values were converted into sediment-deposition rate [g m^{-2}] per 14 days. Accessibility restrictions from April to June 2010 at TFM allowed only one sampling per month because of the breeding period of the protected Bluethroat (*Luscinia svecica*). In total, 1320 sediment samples (TFM, n = 572, BM, n = 514, and SM: n = 234) were collected.

3.2.4 Determining Accretion Rate

To determine accretion rates from SDR, soil profiles (TFM, 3; BM, 3; and SM, 3) were surveyed in summer 2010 and soil horizons were identified (Ad-hoc-AG Boden 2005). From each soil horizon undisturbed soil cores (100 cm^3) were collected (Hansen et al., in preparation) and bulk density [g m^{-3}] was calculated by dividing the mass of oven-dry soil by the core volume. Proportionately, weighted bulk density of the upper 50 cm (cf., Nyman et al. 1990, Callaway et al. 2012) was used to convert sediment-deposition rate [g cm^{-2} $year^{-1}$] to accretion rates [cm $year^{-1}$] (Eq. 1). Below 50 cm no significant increase of bulk density (BD) could be determined. In addition, in many cases a high ground water table prevented an undisturbed sampling of deeper horizons.

$$AR\ [cm\ yr^{-1}] = SDR\ [g\ cm^{-2}\ yr^{-1}]\ /\ BD\ [g\ cm^{-3}] \quad (1)$$

3.2.5 Determining Sediment Characteristics

The following additional sediment analyses should allow comparisons of the study areas to other marshes. We processed homogenized subsamples of the fresh deposited sediment for organic and mineral content (grain size). Total organic carbon (TOC) of marsh sediments at the Elbe Estuary is closely correlated with loss-on-ignition (LOI, Miehlich et al. 1997) at 550°C. We combusted 95 samples (SM, n = 20; BM, n = 38; and TFM, n = 37) for 3 h to calculate TOC (Eq. 2, Miehlich et al. 1997). Additionally, grain size distributions of 169 sediment samples (SM, n = 27; BM: n = 69; and TFM, n = 73) were determined by a laser diffraction sensor (Sympatec HELOS/KF-Magic, Clausthal-Zellerfeld, Germany), for which the measuring size ranged from 18 to 3,500 μm. In this study, grain sizes <63 μm are defined as mud. Our sand fraction only contains medium and fine sand (63 - 630 μm). No particles greater than 630 μm were found in the deposited sediment samples.

$$TOC\ [\%] = 0.42 \times LOI\ (550°C)\ [\%] \quad (2)$$

3.2.6 Topography and Inundation

At each study site, we installed a water pressure gauge (Schlumberger Water Services, Delft, Netherlands: CTD-Diver and CeraDiver resolution, 0.2 cm H_2O, maximal accuracy, 2.5 cm H_2O) in a minor creek, close to the main creek. Coordinates of all sampling points were determined with a GPS and distances to the nearest creek and the marsh edge were measured in a geographic information system (GIS) in geo-referenced aerial photographs. Elevation of all sampling points in relation to NHN and the gauge was measured with an optical Trimble-Station (Sunnyvale, USA). In combination with an additional air pressure gauge (Schlumberger Water Service: Baro-Diver resolution, 0.1 cm H_2O, maximal accuracy, 2.5 cm H_2O), inundation parameters (height, duration, and frequency) for each sampling point and tide were calculated. The temporal resolution of the gauges was 5 min.

3.2.7 Suspended-Sediment Concentration

SSC in the flooding water above the marsh surface was collected directly at 19 sampling points (SM, 5; BM, 8; and TFM: 6), situated close (0.5 m) to the sediment traps. Water samples were taken 3 cm above marsh surface with plastic bottles (adapted from Temmerman et al. 2003a) with a tube of 30 mm, an inlet opening of 5 mm, an air let out and a volume of 580 ml (inflow rate: 0.5 l min^{-1}). Only the first inundation event during the bi-weekly sampling period that submerged the bottle was collected with this technique. To analyze the relation between maximum inundation height and SSC, we calculated the time of this first inundation individually for each sample. Water samples were returned to the laboratory and were shaken before taking a sub-sample of 200 ml, which was vacuum-filtrated through pre-weighted 0.45 µm cellulose nitrate membrane filter (Sartorius Stedim Biotech). Afterward, samples were dried at 60°C for 4 h until constant weight to determine SSC in milligrams per liter.

3.2.8 Aboveground Plant Biomass

To quantify a possible effect of aboveground plant biomass (in the following described as biomass) on SDR, e.g., by reducing flow velocities, and retaining fresh deposited sediment (Brueske and Barret 1994), only the biomass of regularly submerged vegetation layers was analyzed. Between March 2010 and March 2011, the average inundation height at the study sites was 0.27 m (± 0.05 m SD). Therefore, we harvested plants from marsh ground surface to 50 cm. Biomass was measured within 0.25 m² quadrates (TFM, 12; BM, 13; and SM, 12), next to each sampling point at the end of the growing season in September 2010 and after winter in February 2011. The biomass was washed with tab water to eliminate adherent sediment particles. Finally, biomass was dried in pre-weighted perforated plastic bags (Cryovac ® bags) for 3 days at 60°C to determine dry weight [g m^{-2}].

3.2.9 Statistical Analysis

Individual flooding events were considered to calculate the cumulative inundation time (I_{TIME}) and frequency (I_{FREQ}) for each sampling point during each spring-neap cycle. Maximum inundation height (I_{MAX}) of all events during each spring-neap cycle was additionally recorded. Furthermore, we aggregated bi-weekly SDR and individual inundation parameters on a seasonal scale to identify differences between seasons. In the following, we defined the seasons as spring (20th March to 20th June), summer (21th June to 22th September), fall (23th September to 21th December), and winter (22th December to 19th March).

Box-Cox transformation (Box and Cox 1964, Osborne 2010) was used to normalize SDR. Natural logarithm (ln) was found to be the best normalizing transformation. The same procedure was used for SSC values, where log(10) transformation was found to be most appropriate. The data were evaluated for homogeneity of variances (Levene). Differences in SDR, SSC, I_{TIME}, I_{FREQ}, I_{MAX}, and biomass between marsh type (TFM, BM, and SM), marsh zone (low and high) and season were calculated with three-factorial analyses of variance (ANOVAs) and Tukey-tests. Seasonal calculations of the ANOVA for biomass represent only summer and winter. In addition, two-factorial ANOVAs (marsh type and marsh zone) and Tukey-tests were computed for calculated accretion rate, mud content, and TOC.

Multiple regression analysis was used to investigate effects of the explanatory variables SSC, distance to nearest creek (D_C), distance to marsh edge (D_M), I_{TIME}, and I_{MAX} on the bi-weekly SDR for each of the study sites. Analyses were carried out using the stepwise forward option. In this procedure, explanatory variables were stepwise added, starting with the variable that explained most variance in SDR. Variables with a too low contribution to the explained variance were not included in the model. Biomass was not used as an explanatory variable in multiple regression analysis, because of missing data for most of the bi-weekly measurements. I_{FREQ} strongly correlated with I_{TIME} and was therefore excluded from multiple regression models. All statistical analyses were done with STATISTICA 9.1 (StatSoft Inc.).

3.3 Results

3.3.1 Temporal and Spatial Variations in SDR and Its Predictor Variables

Mean bi-weekly SDR differed between marsh types and was significantly higher at low marshes than at high marshes (Table 3.1, Fig. 3.2). SDR were found to be significantly affected by the interaction between marsh type and marsh zone and between marsh type and season, too. At BM and SM, SDR was more than twice as high during fall and winter than in spring and summer. Less temporal variability of SDR was found at TFM (Fig. 3.2). Overall, SDR of TFM-low (6,916 ± 37 g m^{-2}) and SM-low (6,738 ± 1,696 g m^{-2}) were similar. At BM-low, SDR was more than two times higher (17,451 ± 3,992 g m^{-2}). SDR decreased by 65 % from low to high marsh at TFM, by 84 % in BM and by 82 % in SM.

SSC was highly variable and ranged from 13 to 13,447 mg l^{-1} during the sampling period. At low marshes, highest SSC was found at BM (987 ± 333 mg l^{-1}) and lowest at TFM (145 ± 73 mg l^{-1}, Fig. 3.3a). SSC significantly differed between marsh types and was significantly higher at low than at high marsh (Table 3.1). The significant interaction between marsh type and marsh zone indicated that differences in SSC between low and high marsh were less pronounced in TFM and SM than in BM. SSC was found to be significantly different between seasons with highest values during fall and winter (Fig. 3.3b). Differences between marsh types were less pronounced in spring and summer than in fall and winter (significant marsh type × season interaction, Table 3.1).

Table 3.1 *F*- and *p*-values resulting from multi-factorial ANOVAs (not included (n.i.)) for marsh types (tidal freshwater, brackish, and salt), marsh zones (low and high) and season (spring, summer, fall, and winter): bi-weekly ln sediment deposition rate (lnSDR), bi-weekly log(10) suspended sediment concentration (logSSC), bi-weekly inundation frequency (I_{Freq}), bi-weekly inundation maximum (I_{MAX}), bi-weekly inundation time (I_{TIME}), calculated annual accretion rate (AR), aboveground biomass from 0 to 50 cm ($Biomass_{50cm}$; for this analysis the factor *Season* represents only summer and winter), Mud content, Total organic content (TOC), and significance levels.

	Response variable																	
	lnSDR		*logSSC*		I_{FREQ}		I_{MAX}		I_{TIME}		AR		$Biomass_{50cm}$		Grain size		TOC	
Factor	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Marsh type	111.6	***	8.6	**	13.9	***	66.2	***	389.4	***	11.3	***	2.0	n.s.	12.2	***	7.1	**
Marsh zone	218.7	***	5.7	*	1.1	n.s.	1.8	n.s.	167.0	***	32.4	***	7.7	**	1.9	n.s.	4.9	*
Season	15.0	***	17.8	***	0.4	n.s.	16.7	***	8.7	***	n.i.	n.i.	0.5	n.s.	n.i.	n.i.	n.i.	n.i.
Marsh type × Marsh zone	16.6	***	4.0	*	1.0	n.s.	0.7	n.s.	10.4	***	6.8	**	0.4	n.s.	3.5	*	0.7	n.s.
Marsh type × Season	6.2	***	6.9	***	0.4	n.s.	7.9	***	7.8	***	n.i.	n.i.	0.3	n.s.	n.i.	n.i.	n.i.	n.i.
Marsh zone × Season	0.8	n.s.	1.4	n.s.	0.1	n.s.	1.4	n.s.	0.3	n.s.	n.i.	n.i.	7.5	**	n.i.	n.i.	n.i.	n.i.
Marsh type × Marsh zone × Season	1.1	n.s.	1.6	n.s	0.1	n.s.	1.7	n.s.	1.8	n.s.	n.i.	n.i.	4.9	*	n.i.	n.i.	n.i.	n.i.

n.s. = not significant; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Inundation frequency significantly decreased from TFM to SM and from low to high marsh (Fig. 3.4a). No seasonal differences in inundation frequency were found (Table 3.1, Fig. 3.4b). I_{TIME} was up to three and five times higher at TFM than at BM and SM (Fig. 3.4c). Low marshes showed significantly longer inundations than high marshes. Furthermore, significantly different I_{TIME} were found between seasons (Table 3.1, Fig. 3.4d) with up to 40 % longer inundations during fall and winter than during spring and summer. Maximum inundation height was significantly different between marsh types with up to twice as high values at TFM compared with BM and SM. Differences in inundation height between seasons were significant with highest inundations during fall and winter at all marsh types (Table 3.1, Fig. 3.4e, f).

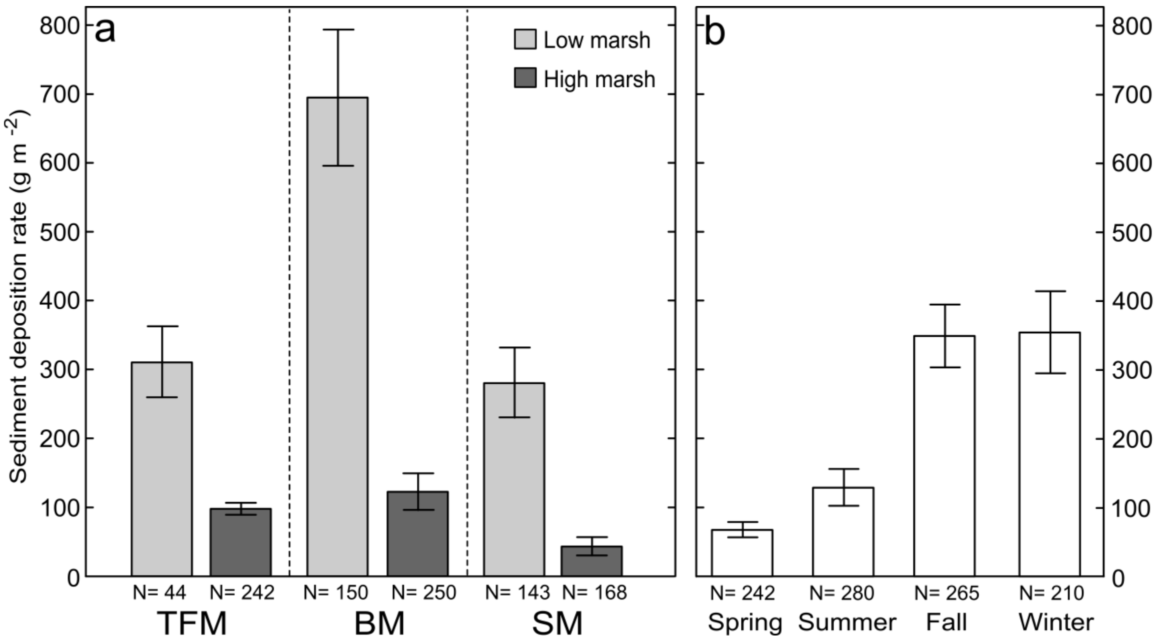


Fig. 3.2 Means (± SE) [g m-2] of bi-weekly sediment-deposition rates at tidal freshwater (TFM), brackish (BM) and salt marsh (SM) of the Elbe Estuary, measured with sedimentation traps, divided into low and high marsh **(a)** and season **(b)**.

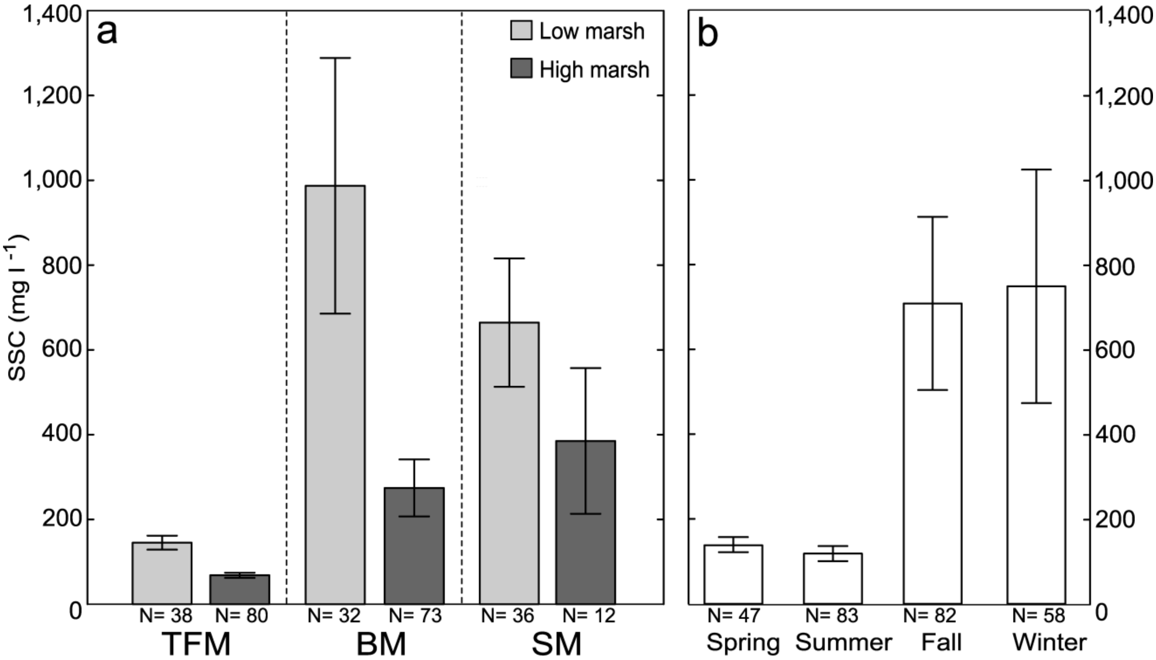


Fig. 3.3 (a) Means (± SE) of suspended-sediment concentration (SSC) [mg l^{-1}] during bi-weekly sampling period measured in low and high tidal freshwater (TFM), brackish (BM) and salt marshes (SM) at the Elbe Estuary. **(b)** Mean (± SE) SSC [mg l^{-1}] per season (spring 2010–winter 2010/2011) of all three marshes. SSC was measured in the inundation water 3 cm above the marsh surface close to installed sediment traps.

Significant differences in biomass of the lowest 50 cm were found between marsh zones while no differences between marsh types were distinguished (Table 3.1, Fig. 3.5). In summer, highest biomass was found at BM (low, 934 ± 63 g m^{-2} and high, 718 ± 79 g m^{-2}). Winter biomass was higher in high marshes of TFM and BM than in low marshes of the respective sites. The interaction between marsh zone, marsh type and season was found to be significant, too (Table 3.1). At TFM-high and BM-high, increases in biomass from summer to winter by 61 and 94 % were observed. By contrast, winter biomass at TFM-low and BM-low decreased up to 50 % compared with summer. At SM, no major changes in biomass distribution were found between summer and winter. Regression analyses between biomass and seasonal SDR for summer showed no significant relation at TFM ($R^2 = 0.07$, p = n.s.) and BM ($R^2 = 0.26$, p = n.s.), but a significant decline of SDR with increasing biomass at SM ($R^2 = 0.7$, $p < 0.001$).

Overall, highest bi-weekly SDR were found close to tidal creeks and the marsh edge (Fig. 3.6a–f). All transects showed decreases of SDR with increasing distance to the sediment source. In the marsh interior, SDR varied between 26 and 60 g m^{-2} at TFM (Fig. 3.6d), between 42 to 67 g m^{-2} in BM (Fig. 3.6e) and between 21 and 49 g m^{-2} in SM (Fig. 3.6f).

An increase in elevation with increasing distance to the creek or the marsh edge was determined for all transects. Only at SM, elevation of the CMT decreased with increasing distance to the creek (Fig. 3.6c). The lowest elevated CMT was situated at BM (between 1.68 and 1.87 m above NHN), the highest at SM (between 1.88 and 2.1 m above NHN). Topography of the MDT at TFM (between 1.69 and 1.96 m above NHN) was comparable with BM (between 1.66 and 1.97 m above NHN). Highest MDT elevation was found at SM (between 1.88 and 2.37 m above NHN).

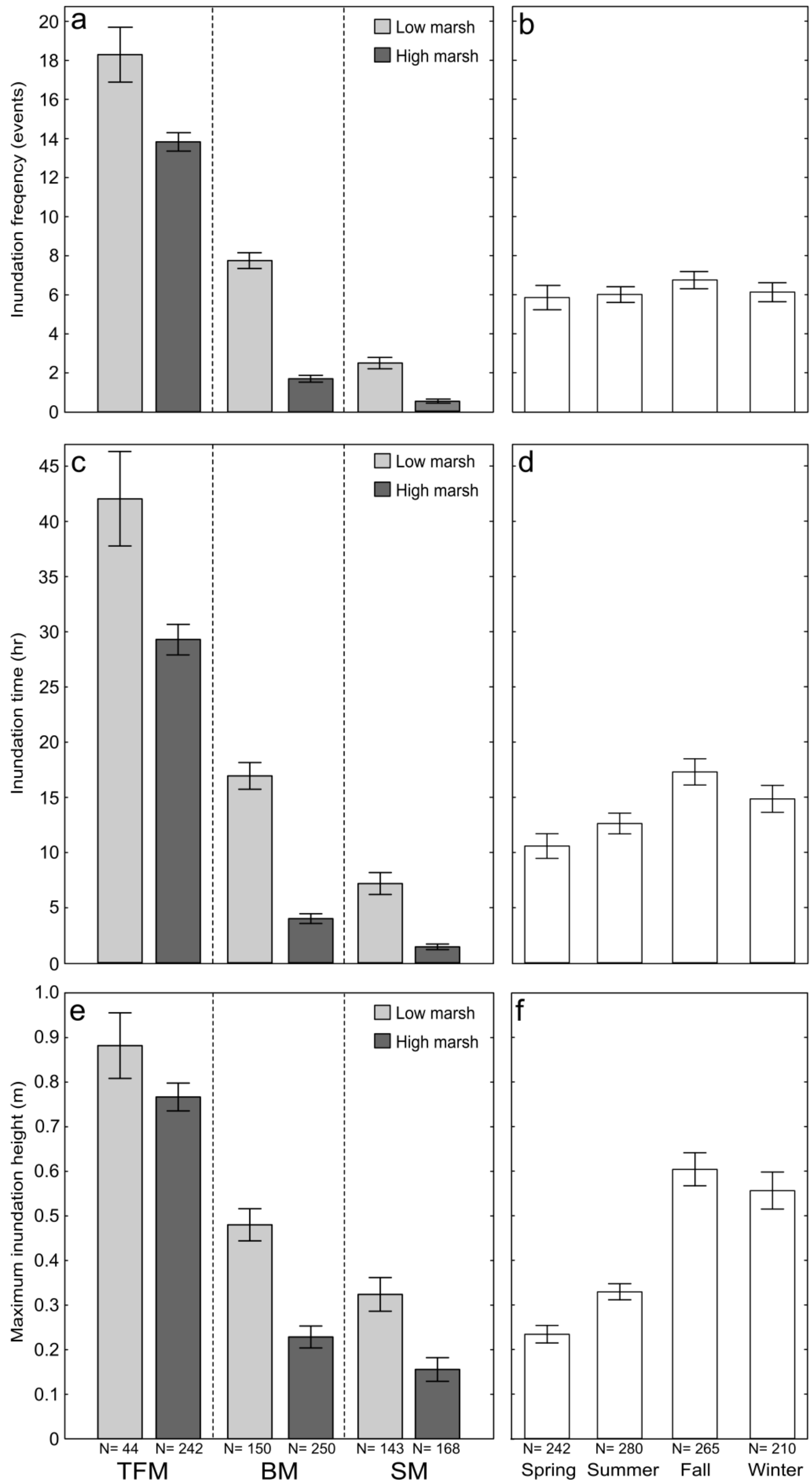


Fig. 3.4 (a) Mean (± SE) bi-weekly inundation frequency measured in low and high tidal freshwater (TFM), brackish (BM), and salt marshes (SM) at the Elbe Estuary, **(b)** Mean (± SE) inundation frequency per season (spring 2010 to winter 2010/2011) of all three marshes, **(c)** Mean (± SE) bi-weekly inundation time [hours], **(d)** Mean (± SE) inundation time [hours] per season, **(e)** Mean (± SE) bi-weekly maximum inundation height [m], **(f)** Mean (± SE) maximum inundation height [m] per season

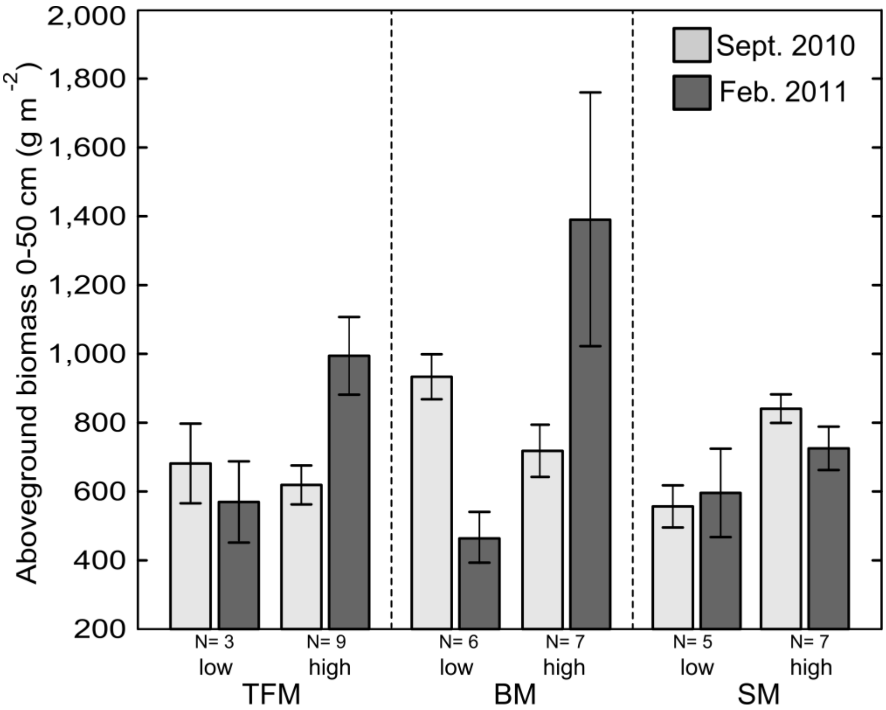


Fig. 3.5 Mean (± SE) aboveground biomass [g m^{-2}] of lowest 50 cm (live and dead biomass) measured in low and high tidal freshwater (TFM), brackish (BM), and salt marshes (SM) at the Elbe Estuary, divided into measurements in September 2010 and in February 2011.

At BM at 100 m distance from the creek, the same SDR was measured as at TFM at < 10 m and at SM at 25 m distance to the creek. These sampling points, however, were all situated at the same elevation.

3.3.2 Sediment Characterization

Bulk density increased from TFM (0.7 g m^{-3} for both marsh zones), to BM (0.86 and 0.98 g m^{-3}, BM-low and BM-high, respectively) to SM (1.17 and 1.03 g m^{-3}, SM-low and SM-high, respectively).

Mud content of the trapped sediment differed significantly between marsh types with high mud contents in TFM and BM (mean, 85–90 %) and lower values in SM (mean, 77–81 %, Tables 3.1, and 3.2). However, the interaction between marsh type and marsh zone was found to be significant, too. In TFM-low and BM-low, significant higher mud contents were present than in their respective high marshes, whereas no distinct difference in mud content between SM-low and SM-high was visible. Overall, the high amount of fine particles indicated low mean flow velocities during inundations (Table 3.2).

Mean TOC of the trapped sediment varied between 4.1 and 6.8 % (Table 3.2). It significantly decreased from TFM to SM and increased from low to high marshes (Table 3.1).

3.3.3 Importance of Predictor Variables for SDR

The multiple regression models relating bi-weekly SDR for each of the differentiated marsh types to the measured explanatory variables included at least four variables; 74, 79 and 71 %

of the bi-weekly variability in SDR were explained for TFM, BM, and SM, respectively (Table 3.3). At all marsh types, SSC was the most important predictor (R^2 = 0.5–0.67) of SDR. Including inundation variables (I_{TIME} and I_{MAX}) in the model, explained additional 8–9 % of the variability in SDR. The importance of D_C for explaining patterns in SDR differed between marsh types from 4 % (TFM and BM) to 11 % at SM. In combination with the other variables, D_M was found to be important only at TFM and BM.

3.3.4 Accretion Rates at Marshes of the Elbe Estuary

Highest calculated annual accretion rate was found at BM-low (20.3 ± 4.7 mm), which was two to more than three times higher than at TFM-low (9.9 ± 0.05 mm) and SM-low (5.8 ± 1.5 mm). Annual accretion rate was significantly higher at low compared to high marsh (Table 3.1, Fig. 3.7). At high marshes, accretion rates ranged from 1.1 ± 0.2 mm year^{-1} at SM, over 3.1 ± 0.6 mm year^{-1} at TFM to 3.8 ± 0.6 mm year^{-1} at BM (Fig. 3.7). Furthermore, the interaction between marsh type and marsh zone was found to be significant (Table 3.1).

3.4 Discussion

3.4.1 Spatial and Temporal Variation in Sediment-deposition Rates and Its Predictors

This study clearly demonstrates that predictor variables known to affect sedimentation deposition rates in estuarine marshes (e.g., SSC, flooding regime, and biomass) and SDR itself differed significantly both along the salinity and elevation gradient, and between seasons. This spatial and temporal variability thus needs to be considered when analyzing and modeling SDR in estuarine marshes.

SSC in the water column of flooding water of estuarine marshes is extremely variable in space and time. Variation in SSC is related to physical properties like freshwater discharge, tidal characteristics, wind regime, and water temperature, to biological activity and to re-suspension processes at the marsh or tidal flats surface (e.g., Leonard et al. 1995, Fettweis et al. 1998, Allen 2000, Ruhl and Schoellhamer 2004, Temmerman et al. 2005a, Bartholomä et al. 2009). We found highest SSC in flooding water above brackish marshes, which is probably related to the location of the estuarine turbidity maximum of the Elbe (see Kappenberg and Grabemann 2001, Bergemann 2005). Furthermore, SSC values were higher in low than in high marsh zones, which indicate continuous settling of sediment particles during “upmarsh” water movement (French and Spencer 1993, Christiansen et al. 2000, van Wijnen and Bakker 2001).

Finally, SSC was three to four times higher in fall and winter than in spring and summer, which is comparable with seasonal disparities found by Fettweis et al. (1998). This is most likely due to the combined effect of higher hydraulic forcing during the stormy season with higher inundations (Neumeier and Amos 2006, van Proosdij et al. 2006a) and simultaneously lower sediment stability in the intertidal flats in front of the estuarine marshes due to low benthic algae abundances (e.g., Underwood and Paterson 1993, Austen et al. 1999). As a consequence, sediment re-suspension above tidal flat surfaces might be higher in autumn and winter, which might lead to highest SSC values during these seasons.

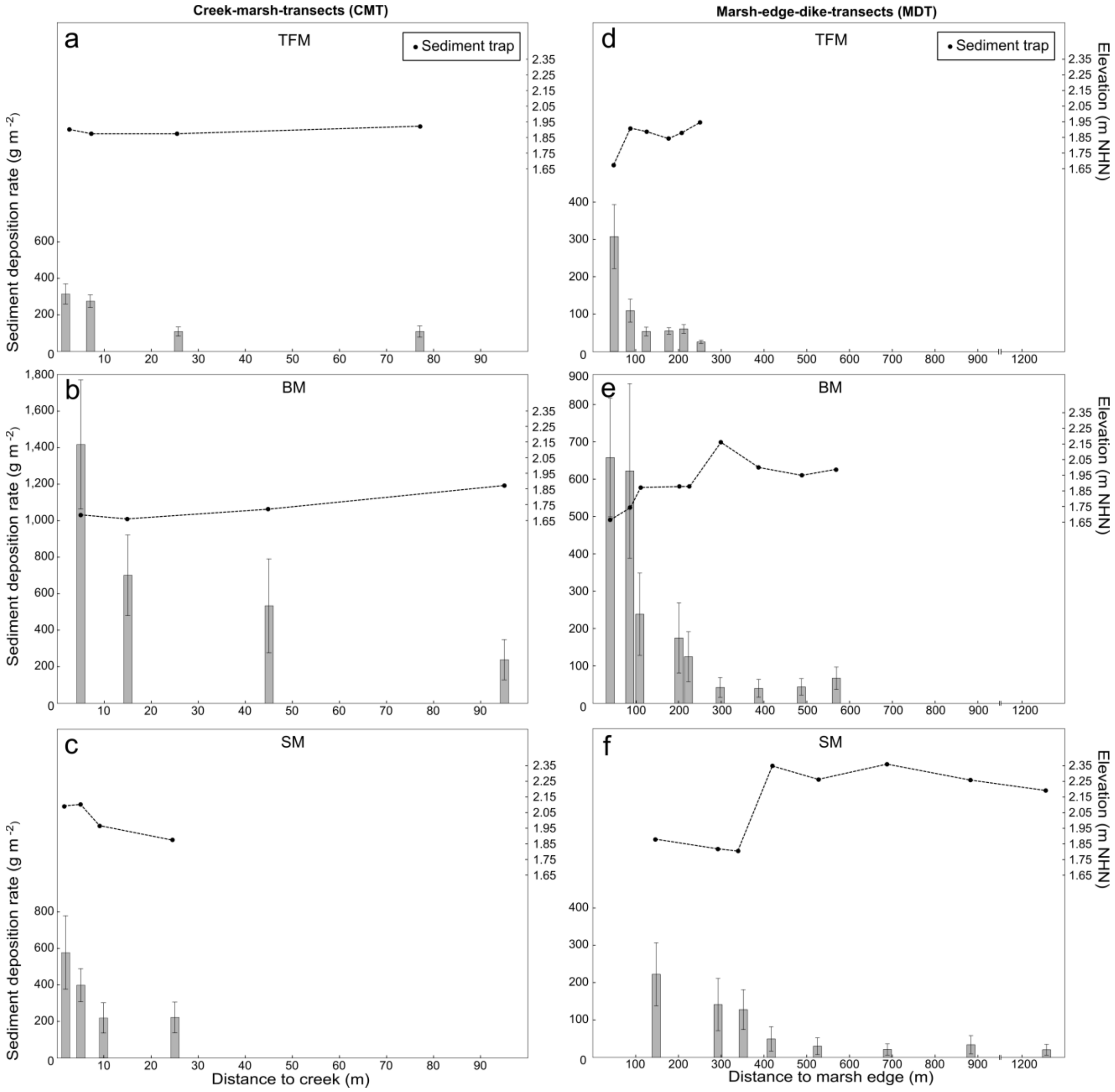


Fig. 3.6 Bi-weekly sediment-deposition rates (mean ± SE, left y-axis) and marsh elevation (black dots with dotted line, right y-axis) and distance to the closest creek **(a–c)** or to marsh edge **(d–f)** of tidal freshwater (TFM), brackish (BM) and salt marshes (SM) at the Elbe Estuary. Note the differences in left y- axis (sediment-deposition rate) and x-axis (distance) for creek and marsh transects.

Table 3.2 Mean ± SE total organic carbon (TOC [%], n_{TFM} = 37, n_{BM} = 38, and n_{SM} = 20) and percentage sand (630–63 µm) and mud (< 63 µm) (n_{TFM} = 73, $_{nBM}$ = 69, and n_{SM} = 27) for tidal freshwater (TFM), brackish (BM) and salt marsh (SM), subdivided into marsh zones (low = low marsh and high = high marsh).

	TFM			**BM**			**SM**		
	TOC	**Sand**	**Mud**	**TOC**	**Sand**	**Mud**	**TOC**	**Sand**	**Mud**
low	5.4 ± 0.2	11.0 ± 0.5	89.0 ± 0.5	4.7 ± 0.3	10.2 ± 0.6	89.8 ± 0.6	4.1 ± 0.5	22.9 ± 3.5	77.1 ± 3.5
high	6.8 ± 0.2	15.5 ± 0.7	84.5 ± 0.7	5.2 ± 0.5	15.1 ± 2.3	84.9 ± 2.3	4.8 ± 1.5	19.0 ± 4.8	81.0 ± 4.8

Table 3.3 Results of multiple regression analysis of bi-weekly sediment-deposition rate [ln g m-2] based on predictor variables; R^2 values resulting from forward stepwise multiple regression analysis of 25 bi-weekly (spring-neap) sediment deposition measurements at three marsh types, with 42 sampling points in total.

	Variable								
Marsh type	SSC [$\log_{10}$ mg]	D_C [m]	D_M [m]	I_{TIME} [hours]	I_{MAX} [m]	Total R^2	*p*	*F*	*n* (valid/total)
TFM	0.58	0.04	0.02	0.08	0.01	0.74	< 0.001	67.9	125/286
BM	0.67	0.04		0.08	0.01	0.79	< 0.001	123.0	137/400
SM	0.50	0.11	0.02	0.01	0.07	0.71	< 0.001	20.1	50/311

Five predictor variables were considered (1) SSC (suspended-sediment concentration), (2) D_C (distance to creek), (3) D_M (distance to marsh edge), (4) I_{TIME} (inundation time), and (5) I_{MAX} (maximal inundation height); $y_{TFM} = 2.131 + 1.163 \times SSC + 0.02 \times I_{TIME} - 0.012 \times D_C - 0.004 \times D_M + 0.489 \times I_{MAX}$; $y_{BM} = 1.707 + 1.206 \times SSC + 0.05 \times I_{TIME} - 0.01 \times D_C + 0.582 \times I_{MAX}$; and $y_{SM} = 1.695 + 1.277 \times SSC + 1.686 \times I_{MAX} - 0.015 \times D_C - 0.001 \times D_M + 0.006 \times I_{TIME}$

Higher water levels and flooding time in estuaries bordering the German Bight are mainly affected by storms (Schulte-Rentrop and Rudolph 2013). In agreement with this general pattern, two to three times higher maximum inundation heights in autumn and in winter and highest inundation frequencies in this period were found in the estuarine marshes of the Elbe Estuary. Along the estuarine salinity gradient, highest inundation frequencies and heights were recorded in the tidal freshwater marsh. Here, slightly lower elevations of the marsh platform and higher tidal amplitude compared with the brackish and the saltwater stretch of the Elbe (Fickert and Strotmann 2007) might have contributed to this result. The higher inundation frequencies and heights in low marsh zones compared with high marshes are obviously related to elevation and have been previously shown by Chmura et al. (2001) and Temmerman et al. (2003a) for salt and tidal freshwater marshes.

Vegetation biomass is another factor found to influence sediment-deposition rates. The biomass was expected to differ between study sites, because in tidal marsh ecosystems, lower salinity stress under freshwater compared with saline conditions leads to higher productivity of the vegetation (Gough et al. 1994, Baldwin and Mendelssohn 1998). Highest total biomass in TFM and lowest in SM was also found at our study sites (Hansen et al. in preparation). Furthermore, a stimulation of biomass production by higher tidal influences was reported for salt and tidal freshwater marshes (e.g., Odum 1988), which might be the reason for increased biomass in some low compared with high marsh communities. However, for considering biomass as a predictor for SDR, it is essential to account only for those portions of the vegetation which are regularly flooded. In this study, it was demonstrated that biomass up to a height of 50 cm did not differ between tidal freshwater, brackish, and salt marshes of the Elbe. Simple regression analysis did not reveal a (positive) relation between the biomass of the regularly flooded vegetation layers and SDR. In agreement to this, Boorman et al. (1998) and Temmerman et al. (2003a) neither found an

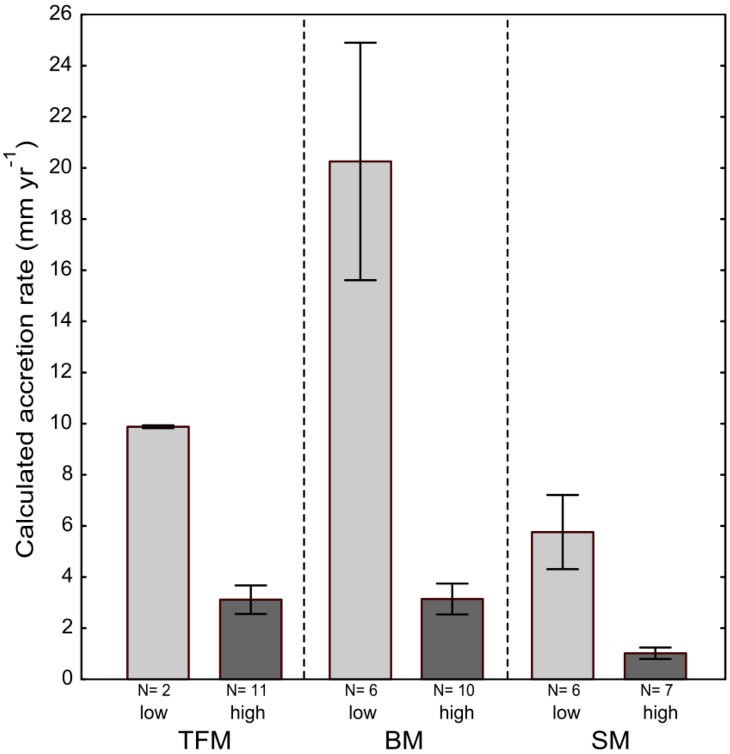


Fig. 3.7 Mean (± SE) annual accretion rates [mm year^{-1}] calculated from sediment-deposition rates and soil bulk density at tidal freshwater (TFM), brackish (BM), and salt marshes (SM) of the Elbe Estuary, divided into low and high marsh.

effect of biomass on SDR in estuarine marshes. By contrast, Bakker et al. (1993), Leonard et al. (1995), and Leonard and Croft (2006) all report a positive relation between biomass and SDR.

This is the first study that recorded SDR in estuarine marshes on a high spatial and high temporal (bi-weekly) scale over the entire salinity gradient. At the same time, the pattern in SDR resembled those found for its major predictors. One of the most striking results of this study is that SDR in brackish marshes were found to be two to three times higher than in salt or in tidal freshwater marshes. Furthermore, much higher SDR were found in low compared with high marshes: Here, both a higher inundation frequency and height, and a higher SSC are likely explanations for this pattern, which had been shown previously by van Proosdij et al. (2006b). During fall and winter, SDR were approximately twice as high as in spring and summer. Again, higher inundation frequencies, maximum inundation heights, and SSC values (Temmerman et al. 2003a) are possible reasons accounting for this effect. Seasonal variations in SDR were also shown by Neubauer et al. (2002) for a TFM in Chesapeake Bay, where, however, highest values occurred in spring and summer and lowest during fall and winter.

3.4.2 Relative Importance of Different Predictor Variables for Sediment-Deposition Rates

The strength of the similarity between the spatiotemporal pattern of SDR and its major predictors already suggests that estimating SDR from the variability of these predictors might be a promising approach. Considering the spatiotemporal variations in predictor variables and the distance to the sediment source, our multiple regression models were able to explain up to almost 80 % of the variability in SDR in TFM, BM, and SM. A similar good model performance was also found by Temmerman et al. (2003a 2005a) for SDR in TFM and SM of the Scheldt estuary.

In our study, the most important predictor in the multiple regression models for the variation of SDR was SSC. A strong relation between SSC and SDR in estuarine marshes was previously reported for the Scheldt by Temmerman et al. (2003a). Especially in systems with high spatial variability in SSC, it is important to include SSC in studies on SDR in tidal marshes. In our regression models, inundation parameters (I_{TIME}, and I_{MAX}) added another

8 to 9 % of explained variance in SDR. According to our findings, predicting SDR from inundation parameters alone might only be possible for small areas: Considering different marsh types along the estuarine salinity gradient, differences in SSC overrides effects of variation in inundation. Overall, SDR in estuarine marshes is affected by several predictor variables and their complex interactions.

In contrast to our study, Cahoon and Reed (1995) and Leonard (1997) showed that I_{TIME} was the most important predictor for SDR in studied estuarine marshes in North America. Despite longest, highest, and most frequent inundations in TFM, however, we found lowest SDR here. Esselink et al. (1998), Temmerman et al. (2003a) and van Proosdij (2006b) recorded highest SDR in areas close to the sediment source (marsh edge, creek). This is in agreement with our results of an exponential decrease of SDR with increasing distances to the sediment source.

3.4.3 Sea-Level Rise and Accretion Rates in Estuarine Marshes

At all study sites, calculated accretion rates in low marshes (5.8–20.3 mm year^{-1}) were higher than current sea-level rise with about 3.6 ± 0.7 mm year^{-1} (1971–2008) in the German Bight (Wahl et al. 2011). By contrast, TFM-high and SM-high (1.1–3.1 mm year^{-1}) might be vulnerable due to insufficient deposits of fresh sediments and accordingly resulting low accretion rates. Partly, high marshes might regress into low marshes and decrease in extent, if landward migration of marshes is limited by artificial infrastructures like dikes.

Relative changes in sea-level rise also include large-scale glacial isostatic adjustments up to 1.6 mm a^{-1} (IKÜS 2009) as elastic aftereffects of the last Ice Age (Vink et al. 2007). Additionally, local small scale shallow subsidence processes (see Cahoon et al. 1995), due to, e.g., sediment compaction, soil shrinkage, and biomass decomposition (Cahoon 2006) can occur.

To assess influence of subsurface processes on elevation change the SET-MH method, a combination of the sedimentation-elevation table and marker horizon methods, was proposed (Cahoon et al. 1995, Cahoon 2006). However, while high compaction and mineralization rates in organogenic marshes lead to a considerable subsidence, this effect is much smaller in minerogenic marshes (French and Burningham 2003, Nolte et al. 2013a). This is in line with other studies focusing on primarily minerogenic marshes, which showed only insignificant amounts of autocompaction (Allen 1990, French 1993). Bartholdy et al. (2010) verified these results and showed a significant decrease of bulk density with increasing organic carbon. These results for minerogenic marshes are comparable with our analyses of high bulk densities (0.7–1.17 g m^{-3}) and low TOC (4–7 %) of freshly deposited sediments. According to our information, no research on shallow subsidence for the Elbe Estuary is available. However, calculated accretion rates by bulk density might be affected by seasonal differences of soil autocompaction due to soil moisture, and organic content. Sampling of bulk densities were conducted in summer with low I_{TIME}, compared with other seasons, and with resulting low levels of soil water content. It can therefore be supposed that bulk densities were rather overestimated than underestimated, resulting in higher calculated accretion rates. Neubauer et al. (2002) identified a mineralization rate up to 30 % of freshly deposited organic matter in North American TFM. Transferred to our marshes with a substantially lower TOC than the North American marshes from the mentioned study, only an insignificant effect of mineralization on total accretion rates is expected. To exclude shallow subsidence processes (see Cahoon et al. 1995), such as top soil sediment compaction, biomass mineralization or drying processes (Nolte et al. 2013a), in-depth fixed SETs or SEBs measurements could be added to near surface methods like the applied trap measurements.

In general, increasing water tables induced by sea-level rise will change inundation parameters in tidal marshes, which are expected to adjust toward a new equilibrium (Morris et al. 2002) if accretion rates are able to compensate sea-level rise. Gönnert et al. (2009) specified a sea-level rise of 0.40–0.80 m until 2100 (approximately 4.4–8.9 mm $year^{-1}$) for the German Bight. In the Elbe Estuary, accretion rates in estuarine low marshes seem to be sufficient to keep pace with moderate rates of sea-level rise. If storm activity increases in the future (see IPCC 2007), this might lead to an increase of sediment deposition at the Elbe Estuary, as these storms and storm surges were important predictors of sediment-deposition rate on our study sites, especially for higher elevated marshes. High amounts of the total annual deposited sediment (TFM, 13–39 %; BM, 30–71 %; and SM, 17–95 %; data not shown) originated from two bi-weekly sampling periods in fall 2010 and winter 2010/2011 during which storms (1.5–2.5 m above MHW) occurred. The frequency of two extreme events during our 1 year study period was almost 60 % lower compared with the mean number of storms per year of the last decade (4.9 ± 2.9, see Appendix, Table 3.4). This low storm frequency might explain relatively low SDR in high marshes during our study. In addition, no storm surge events (> 2.5 m above MHW) occurred, which again is rather atypical compared with the last decade (0.7 ± 0.7, see Appendix, Table 3.4). In future, especially high marshes might benefit from an increase of storm activity, which could attenuate the possible regressive succession of high into low marshes.

Presently, SSC seems to be higher than required to adjust the marsh surface to sea-level rise at our study sites (see Kirwan et al. 2010). In addition, future increases in wind and wave energy in winter (see IPCC 2007) might cause higher erosion of sediments in tidal flats in front of the marshes enhancing SSC above marsh platforms and increasing accretion rates at tidal marshes. However, salt water intrusion could cause upstream shifts of marsh types in estuaries and alter the spatial distribution of the turbidity zone in the Elbe Estuary. In addition, TFM might be negatively affected by a reduced freshwater discharge of rivers (Neubauer and Craft 2009). In the Elbe Estuary, the potential area of TFM is restricted by dikes, the harbor of Hamburg and by a weir. Overall, TFM-high with low SDR and highest susceptibility to increased salinity might suffer most from effects caused by climate change.

3.5 Conclusions

We demonstrated the importance of considering spatial and temporal variations in SDR and its predictors over the entire estuarine salinity gradient. According to our findings, SDR and the resulting calculated accretion rates in low marshes of the Elbe Estuary seem to be sufficient to compensate moderate levels of sea-level rise. Variation in biomass among marsh types and marsh zones did not play a decisive role for SDR. Overall, multiple regression models seem to be well-suited to explain the variability in SDR. In future, multivariate models accounting for temporal and spatial variation in predictor variables should be used to predict SDR in estuarine marshes across regional and local gradients.

3.6 Acknowledgments

The authors would like to thank all the students and colleagues; especially Melanie Griem, Theresa Martens and Janina Schnoor for their engagement in collecting and handling the sediment samples, Alena Lucht for collecting and analyzing the aboveground biomass, Sebastian Lindhorst and colleagues from the Institute of Geology and Paleontology (University of Hamburg) for processing the grain size analysis. We thank Martin Stock from the Schleswig-Holstein State Agency for Coastal Defence, National Park and Marine Conservation, the nature conservation authority of the District Pinneberg (Schleswig-Holstein), especially Uwe Helbing and Bernd-Ulrich Netz for administrative decision to work in the conservation areas. The work was funded by the Federal Ministry for Education and Research within the research project KLIMZUG-NORD and supported by the Estuary and Wetland Research Graduate School Hamburg (ESTRADE) as member of the State Excellence Initiative (LExI) funded by the Hamburg Science and Research Foundation. Additionally, we thank the handling editor and the two anonymous reviewers for their valuable comments.

3.7 Appendix

Table 3.4 Overview about annual storms and storm surges from 1990–2011 (mean ± SD for decades (1990–1999; 2000–2009), sum of these extreme events as well as occurring storms and storm surges during the investigation period (Mar 2010–Mar 2011), and mean water height during storms and storm surges above MHW; data from http://www.portal-tideelbe.de for gauge Stadersand; stream, 654 km; MHW = 1.77 mNHN.

year	Number of storm events (water table 1.5–2.5 m above MHW)	Mean maximum inundation height above MHW (cm) during storm	Number of storm surge (water table > 2.5 m above MHW)	Mean maximum inundation height above MHW (cm) during storm surge	Total number of extreme events	Mean maximum inundation height above MHW (cm) during extreme event
1990	13	180	5	303	18	214
1991	7	185	0		7	185
1992	4	167	0		4	167
1993	13	201	1	339	14	211
1994	6	202	1	375	7	227
1995	7	184	1	364	8	207
1996	3	189	0		3	189
1997	4	182	0		4	182
1998	8	169	0		8	169
1999	4	171	3	325	7	237
2000	4	202	1	299	5	221
2001	1	170	0		1	170
2002	5	184	1	297	6	203
2003	2	168	0		2	168
2004	6	168	1	253	7	180
2005	8	177	0		8	177
2006	4	161	1	250	5	178
2007	8	175	2	300	10	200
2008	2	171	1	267	3	203
2009	2	179	0		2	179
2010	1	209	0		1	209
2011	7	182	0		7	182
15 Mar 2010 – 14 Mar 2011	**2**	**207**	**0**		**2**	**207**
Annual means for 1990–1999	6.9 ± 3.6	183 ± 12	1.1 ± 1.7	341 ± 29	8.0 ± 4.7	198 ± 24
Annual means for 2000–-2009	4.2 ± 2.5	175 ± 11	0.7 ± 0.7	277 ± 23	4.9 ± 2.9	187 ± 17


Chapter four

4 Performance of Different Sediment Traps Under Controlled Inundation and Sediment Supply Regimes

Christian Butzeck, Andrew H. Baldwin, Gary Felton, Stefanie Nolte, Kai Jensen

Manuscript

4.1 Introduction

Accelerated sea-level rise is threatening coastal wetlands worldwide (Craft et al. 2009, Morris et al. 2002, Neubauer and Craft 2009, Stralberg et al. 2011). The marsh's ability to maintain a positive surface elevation in relation to sea-level depends on sediment-deposition rates (SDR). Sediment deposition includes gravity-based surface processes of organic and inorganic particle deposition during inundations (Allen 2000, Temmerman et al. 2005a, Bartholomä et al. 2009, Nolte et al. 2013a). In addition, subsurface accumulation of dead biomass (Bricker-Urso et al. 1989), local shallow subsidence processes (Cahoon et al. 1995), as well as large scale glacial isostatic adjustments of the last Ice Age (Vink et al. 2007) affect marsh surface elevation.

Sediment deposition in tidal marshes is not a continuous process (Reed 1989), but highly variable in space and time. This high spatial and temporal variability is affected by various interacting factors, including distance to the sediment source (Esselink et al. 1998, Temmerman et al. 2003a), variability of suspended-sediment concentration (Fettweis et al. 1998, Butzeck et al. 2014), inundation time, height and frequency (Cahoon et al. 1995, Leonard 1997, Allen and Duffy 1998), seasonal dependency of water levels (van Proosdij et al. 2006a), and alterations of flow hydrodynamics by aboveground plant biomass (Leonard and Luther 1995).

Sediment-deposition rates and resulting rates of surface elevation change in times of sea-level rise (SLR) have recently been investigated in large number of studies (e.g., van Koningsveld et al. 2008, Craft et al. 2009, Kirwan and Megonigal 2013), although using a wide variety of methods. These methods differ for example in terms of study period; some methods cover single tidal events, while others encompass spring-neap cycles to months, seasons, or sampling periods up to several centuries. A detailed review of sediment-deposition and accretion measuring methods for different temporal and spatial scales was recently conducted by Nolte et al. (2013a). This review also discusses several studies comparing the efficiency of different sediment traps to measure sediment deposition. However, most of these studies were conducted with regard to SDR in shallow lakes or rivers (Bloesch and Burns 1980, Kozerski and Leuschner 1999) so there is still a lack of knowledge concerning the performance of different sediment traps used in intertidal areas. The hydrodynamics of intertidal systems, which greatly affect sediment trap efficiency (Swart and Zimmerman 2009), might not be comparable to river systems. Therefore, trapping efficiency of different trap designs need to be evaluated for intertidal systems. Trapping efficiency could differ between traps, because a rim, for example, can prevent natural lateral relocation processes (Temmerman et al. 2003a). Alternatively, a sediment trap with a surface that is completely level with the soil, i.e., a flat tile, might be vulnerable to washout of sediment by heavy rain events (Steiger et al. 2003).

The aim of this study was to compare the trapping efficiency of frequently used sediment traps under controlled experimental conditions in a flume. All trap types studied here are primarily used for short-term (tidal to bi-weekly) investigations. In our experiment, we measured the trapping efficiency of four different sediment-trap types during simulated tidal inundations. Results were additionally analyzed with respect to the distance to the sediment source, and different suspended-sediment concentrations of the flooding water.

4.2 Materials and Methods

Measurements were conducted at the Department of Environmental Science and Technology, University of Maryland (College Park, MD, USA). We used a self-contained glass sided tilting re-circulating flume (Armfield Ltd., Ringwood, UK, Fig.). The flume consists of a 7.3 m long, 0.3 m wide, and 0.45 m high rectangular channel. The flat inner bottom of the flume was completely covered with a soft and flexible artificial grass floor mat (The New England Turf Store, Canton, MA, USA, stem length: 43 mm) to simulate the friction of a moderately-grazed tidal marsh vegetation. The floor mat was attached to the flume bottom with waterproof Velcro® tape. Small patches were cut out of the mat at the sampling points for the measuring equipment. We tested four different sediment-deposition measuring methods with sediment traps (ST), including two different flat surface methods, namely, ceramic tiles (e.g., Pasternack and Brush 1998), circular AstroTurf® floor mats (e.g., Lamberg and Walling 1987) and two different set-ups of circular traps with and without a floatable lid (Temmerman et al. 2003a, Butzeck et al. 2014, see Fig. 4.2, Table 4.1).

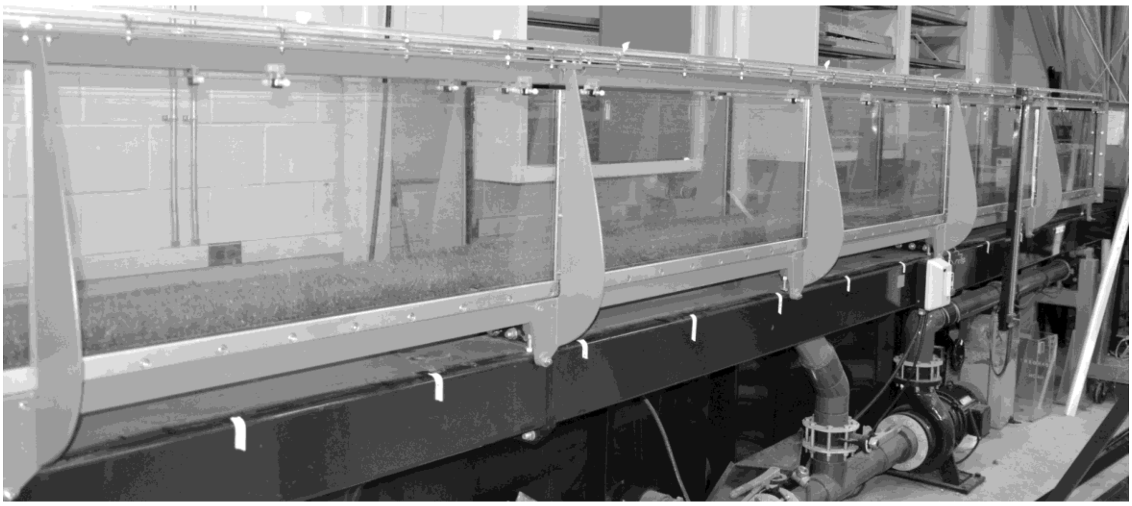


Fig. 4.1 View of the flume.

4.2.1 Set-up and Test Procedure

We installed the ST in a distance of 0.5, 2.0 and 6.0 m from the inlet opening of the flume (Fig. 4.3). The circular traps and floor mats were attached with Velcro® tape to the flume bottom to prevent sediment re-suspension. In addition, we measured the suspended-sediment concentration (SSC) by taking water samples at two locations (0.5 and 6.0 m) directly behind the ST. These samples were taken 3 cm above the flume surface with plastic bottles (adapted from Temmerman et al. 2003a, Butzeck et al. 2014) with a tube of 30 mm, an inlet opening of 5 mm, an air outlet and a volume of 580 ml (inflow rate: 0.5 l min^{-1}). A SSC-dummy bottle was placed behind the middle ST to attain identical conditions regarding potential turbulences and velocity disturbances in sedimentation patterns for all sampling points (Fig. 4.3). The SSC-dummy was not replaced or emptied during the experiment.

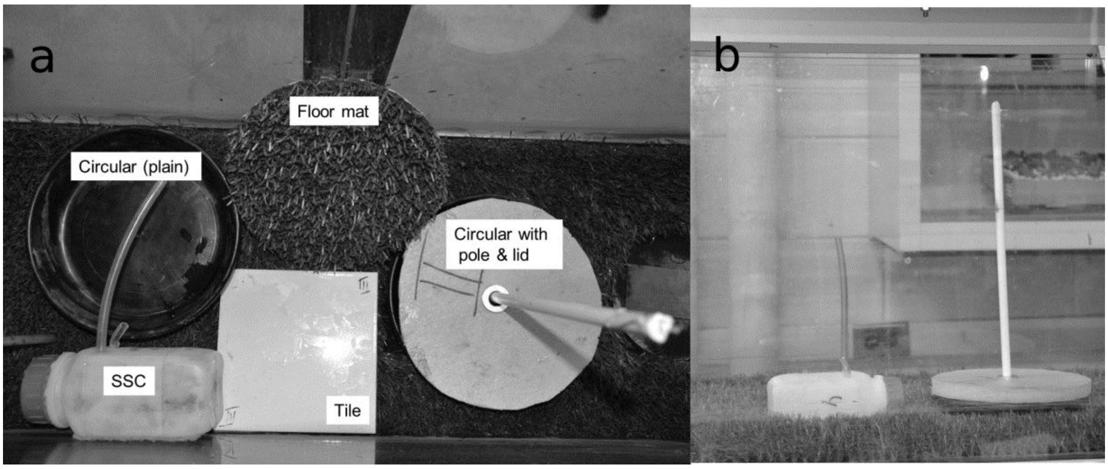


Fig. 4.2 (a) Types of sediment traps used to measure sediment-deposition rate and plastic bottles used to measure suspended-sediment concentration ($SSC_{Initial}$, SSC_{Slack}) placed on the flume bottom. **(b)** Flume bottom covered with an artificial floor mat, arrangement of SSC-bottle (left) and circular sediment trap with pole and lid (right).

Table 4.1 Area [cm²], size and specific features of the different sediment trap types.

Trap type	Surface Area [cm²]	size	specific feature
Circular trap (with lid)	280.55	18.9 cm (inside diameter)	3 cm high rim
Circular trap (plain)	280.55	18.9 cm (inside diameter)	3 cm high rim
Floor mat	314.16	20.0 cm (diameter)	at bottom level, stem length: 20 mm
Ceramic tile	232.26	15.24 x 15.24 cm	at bottom level

Table 4.2 Class fractions of the sediment samples used in the flume study.

ASTM Sieve No	Size [μm]	Class fraction [%]
40	425	< 0.01
50	300	0.04
60	250	0.03
80	180	0.04
100	150	0.02
140	106	0.06
200	75	0.17
270	53	11.43
Pan	< 53	88.21

The sediment used for this study consisted of over 99 % of clay and fine silt (Table 4.2) and was collected from an oligohaline marsh at the Nanticoke estuary (Maryland, USA). Sediment was pre-filtered with 1.18 mm (ASTM No. 16) and 425 μm (ASTM No. 40) sieves to remove large organic particles. A high (*SSC-high*: ~100 mg l^{-1}) and a low sediment supply scenario (*SSC-low*: ~65 mg l^{-1}) were simulated. We mixed the harvested sediment with a defined quantity of water in a bucket, and installed an air-pump, which was connected to an air pipe, on the bottom of the bucket. The pump provided a constant movement of the sediment-water mixture to prevent the sediment from settling inside the bucket. We added the sediment-water mixture to the flume via a pipe at the inlet opening of the flume using *Bernoulli's principle*. The outlet of the flume was closed during the entire experiment to simulate inundation heights of 15 cm above surface. Inundation heights were measured at Trap 1 (see Fig. 4.3). The timing of the tidal inflow-simulation was between 8 and 11 minutes. Water samples ($SSC_{Initial}$) were taken directly after the inundating water submerged the inlet opening of the SSC bottle. We then stopped the inflow of the water and the discharge of the sediment water-mixture after reaching the inundation peak point at

15 cm. Simultaneously, the two SSC-bottles were replaced to obtain two additional SSC-samples (SSC_{Slack}) from the outflowing water. The outflow of the water (ebb) occurred over the inlet opening of the flume, comparable to natural systems. Total inundation time of one run of our tidal simulation lasted between 37 ± 2 min (short inundation runs) and 61 ± 2 min (long inundation runs). Simulations of long inundation runs with *SSC-low*, and short inundation with *SSC-high* were performed, using one trap type per run. Ten runs with each of the four trap types of the *SSC-low* and the *SSC-high* simulations were conducted. The sequences within *SSC-low* and *SSC-high,* in which the trap types were tested, were randomized.

Table 4.3 *F* and *p* values resulting from the three-factorial ANOVA for effects of trap types (circular trap with pole and lid, circular trap without lid, floor mat, and ceramic tile), distance to the origin of the flume (0.5, 2.0, and 6.0 m), sediment supply (low and high) on sediment-deposition rate.

	Sediment-deposition rate	
Factor	*F*	*p*
Trap type	61.5	***
Distance	212.7	***
Sediment supply	273.3	***
Trap type × Distance	0.4	*n.s.*
Distance × Sediment supply	13.0	***
Trap type × Sediment supply	1.3	*n.s.*
Trap type × Distance × Sediment supply	0.2	*n.s.*

n.s. not significant, ***$p < 0.001$

After each run, we rinsed the sediment from the ST with distilled water into aluminum boxes and dried the samples at 105 °C until they reached a constant weight. Values were then converted into SDR [g m^{-2}] per tidal inundation. SSC-samples were shaken before taking a subsample of 200 ml, which was vacuum-filtrated through pre-weighed 0.45 µm glass fiber filters (Whatsman™). Afterward, SSC-samples were dried at 60 °C for 4 h until a constant weight was reached to determine SSC in milligrams per liter.

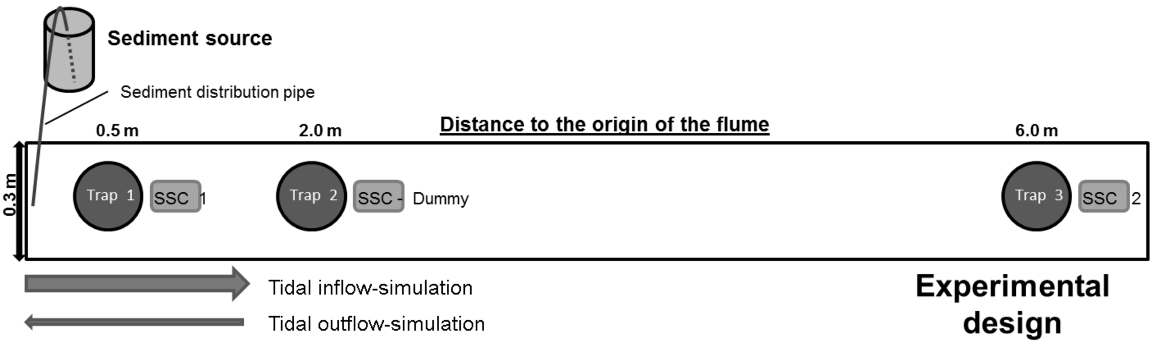


Fig. 4.3 Drawing of flume channel and arrangement of sediment traps and suspended sediment (SSC)-bottles (top view) and flow direction during tidal simulations. We placed a dummy SSC-bottle behind the middle sediment-trap to attain identical conditions regarding potential turbulences and velocity disturbances in sedimentation pattern for all sampling points.

4.2.2 Statistical Analysis

Data met the assumptions of normality and homogeneity of variance. To analyze differences in SDR between ST-types we used a three-factorial ANOVA including ST-type, distance to the inlet of the flume, and sediment supply (*SSC-low*/*SSC-high*) as factors. If a significant effect was detected, we performed pairwise comparisons using Bonferroni post-hoc tests. Additionally, spatial and temporal variations in SDR were explored using a series of simple linear regression analysis. We analyzed the relationships between $SSC_{Initial}$ and SDR, between SSC_{Slack} and SDR, and between $SSC_{Initial}$ and SSC_{Slack}. These regression analyses were performed separately for the two distances to the origin of the flume. All statistical analyses were done with STATISTICA 10 (StatSoft Inc. 2010).

4.3 Results

4.3.1 Differences in Sediment-Deposition Rates Between Sediment-Trap Types

Mean SDR significantly differed between ST-types used in our study (Fig 4.4, Table 4.3). We found the highest SDR in circular traps. SDR in circular traps with a floatable lid were slightly, but not significantly lower (7 %) than SDR in circular traps without a floatable lid. ST made of floor mats differed significantly from both tiles and circular traps, but not from circular traps with a lid (Fig 4.4). The lowest SDR were found on tiles, which significantly differed from all other ST-types. In total, the SDR of tiles were 31 % lower than the SDR of floor mats, and 43 to 47 % lower than the SDR of circular traps with lid and plain circular traps, respectively.

4.3.2 Spatial and Temporal Variation in SDR

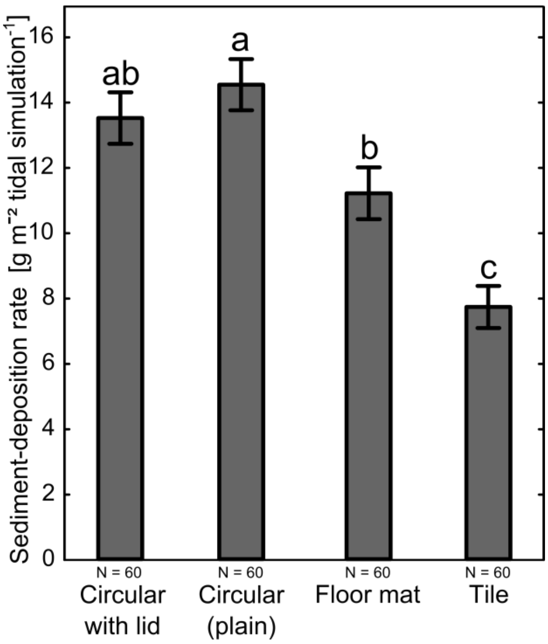


Fig. 4.4 Means (±SE) of sediment-deposition rates [g m^{-2}] of different sediment traps. Small letters denote statistical differences between sediment-trap types ($p < 0.05$). Runs with different trap types were conducted in a randomized sequence.

Results revealed strong effects of distance to the inlet of the flume (Fig. 4.5, Table 4.3), and sediment supply (Fig. 4.5a compared to Fig. **4.5**b, Table 4.3) on SDR. All ST-types showed a highly significant decrease in SDR with increasing distance from the inlet of the flume (Fig. 4.5, Table 4.3), although the reduction was greater under high than under low sediment supply rates (significant distance × sediment supply rate, Table 4.3).

A higher sediment supply (*SSC-high*, Fig. 4.5b) resulted in a significantly higher SDR, compared to the *SSC-low* scenario (Table 4.3). The effect of sediment supply was not uniform, but it varied significantly with distance. During higher sediment supply, the percent decreases of SDR with distance were slightly higher. No significant interaction between trap types and distance was found.

$SSC_{Initial}$ was slightly higher than SSC_{Slack} (see Table 4.4.), but did not significantly differ (Table 4.5). SDR was significantly correlated with $SSC_{Initial}$ and SSC_{Slack} (Table 4.5). We observed a clear decrease in SSC with increasing distance from the inlet of the flume. However, correlations of SSC and SDR were more pronounced close to the origin of the flume at 0.5 m ($SSC_{Initial}$: $R^2 = 0.51$, SSC_{Slack}: $R^2 = 0.56$), than in a distance of 6.0 m ($SSC_{Initial}$: $R^2 = 0.16$, SSC_{Slack}: $R^2 = 0.1$, Table 4.5).

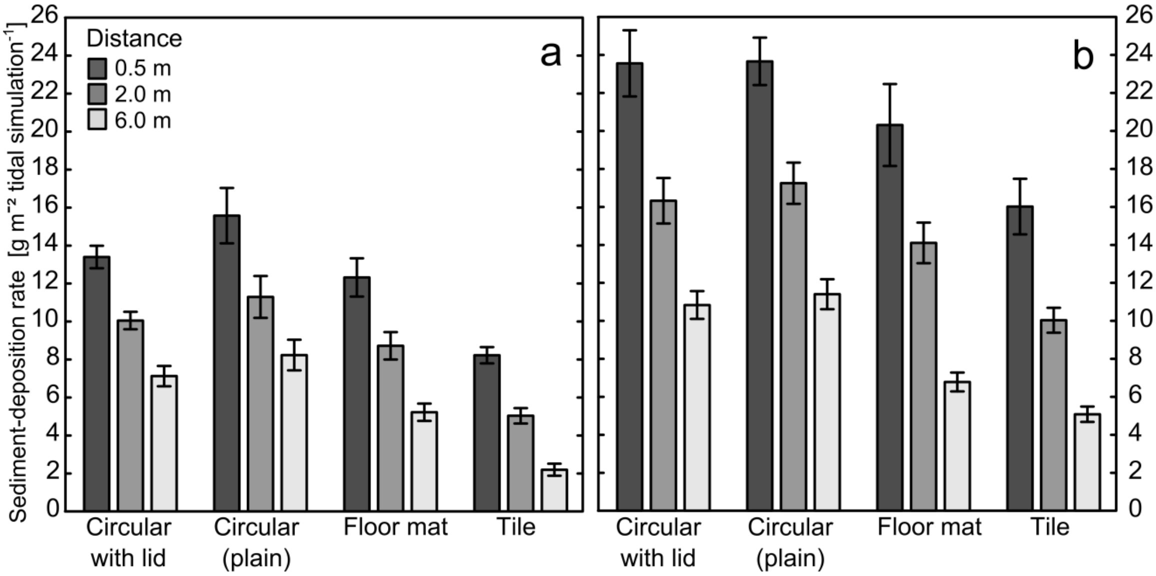


Fig. 4.5 Means (± SE) of sediment-deposition rates [g m^{-2}] of different sediment traps, depending on distance to the origin of the flume, during **(a)** low suspended-sediment concentrations and **(b)** high suspended-sediment concentrations. Each bar represents 10 measurements.

Table 4.4 Means ± SE, minimum and maximum values of suspended-sediment concentrations [mg l^{-1}], measured in a distance of 0.5 and 6.0 m from the origin of the flume, at the beginning ($SSC_{Initial}$) and end (SSC_{Slack}) of the inflow flood simulations.

		$SSC_{Initial}$			SSC_{Slack}		
Distance [m]	N	Mean ± SE	Min	Max	Mean ± SE	Min	Max
0.5	83	75 ± 4	28	148	72 ± 4	30	155
6.0	83	49 ± 2	23	93	44 ± 2	21	129

4.4 Discussion

4.4.1 Trapping Efficiency of Different Sediment Traps

A range of commonly used sediment trap designs differed significantly in their trapping efficiency. The strong differences between circular sediment traps (high efficiency) and both flat surface ST (low efficiency) methods found in this study were remarkable. Therefore, studies using different types of ST may not be directly comparable.

Table 4.5 Results of linear regression analysis between sediment-deposition rate (SDR), suspended-sediment concentration at the beginning of the inundation ($SSC_{Initial}$), and end of the inflow tidal simulation (SSC_{Slack}) in a distance of 0.5 and 6.0 m from the inlet of the flume.

Regression Analysis		
	R^2	p
0.5 m		
SDR × $SSC_{Initial}$	0.51	***
SDR × SSC_{Slack}	0.56	***
6.0 m		
SDR × $SSC_{Initial}$	0.16	***
SDR × SSC_{Slack}	0.10	**
$SSC_{Initial}$ × SSC_{Slack}	0.76	***

**$p < 0.01$, ***$p < 0.001$

During our study, deposited sediment in both circular ST-types was around 20 % higher than the deposited sediment found on the floor mat ST, and around 45 % higher when comparing circular ST-types with tiles. This difference between the circular ST (with a rim) and the flat surface ST might indicate re-suspension and/or lateral sediment-transport processes. This might occur on different scales depending on ST-type. Some studies have found that collected sediment trapped by flat surface traps is sensitive to washing off by rain and partly by tides (Gardner 1980, Kozerski and Leuschner 1999). However, during a shallow water study conducted by Mansikkaniemi (1985), no significant differences between smooth flat ST with and without attached floor mats were found. Contrastingly, Steiger et al. (2003) suggested the usage of floor mat ST for riparian sedimentation studies, due to the benefit of a rough surface simulating surrounding vegetation, and providing an easy handling during collecting and processing. However, it can be expected that the rim of the circular ST prevents trapped sediment from lateral dispersal to the surrounding surface (Neubauer et al. 2002), but also prevents a relocation of sediment from the surrounding surface into this ST-type. Both Reed et al. (1999) and Temmerman et al. (2003a) found no or only a marginal amount of re-suspension of fresh deposited sediment from circular ST. Bloesch and Burns (1980) stated that besides flow velocity and viscosity, the exact geometry of circular ST (ratio of height to diameter) affects the amount of re-suspension. Furthermore, our results show that circular ST featured higher deposition rates than flat tiles, which might be caused by reduced bottom shear stress (Kozerski and Leuschner 1999), while simultaneously the rim of the circular trap induced local flow acceleration which increases deposition rates (see Butman et al. 1986).

As might be expected, no significant difference between circular ST with and without a lid was found. In field studies during low tides, a floatable lid protects trapped sediment from splashing out by heavy rain events (Temmerman et al. 2003a). We can assume that the slightly higher SDR found in circular ST without a lid, might be partly explained by sediment adhered below the lid or at the pole which is holding the lid.

4.4.2 Spatial Features in SDR Between Sediment Trap-Types and Sediment Supply

Significantly decreases of SDR with increasing distance to the inflow of the flume and in dependence of sediment supply was found for all ST-types. In field studies, spatial effects of distance to the sediment source, like marsh edge and nearest creek on SDR (e.g., Esselink et al. 1998, Temmerman et al. 2003a, Butzeck et al. 2014), as well as the relationship between variations in sediment supply, which are caused by season and salinity zone (e.g., Fettweis et al. 1998, Temmerman et al. 2003a, Butzeck et al. 2014) has been described by several authors.

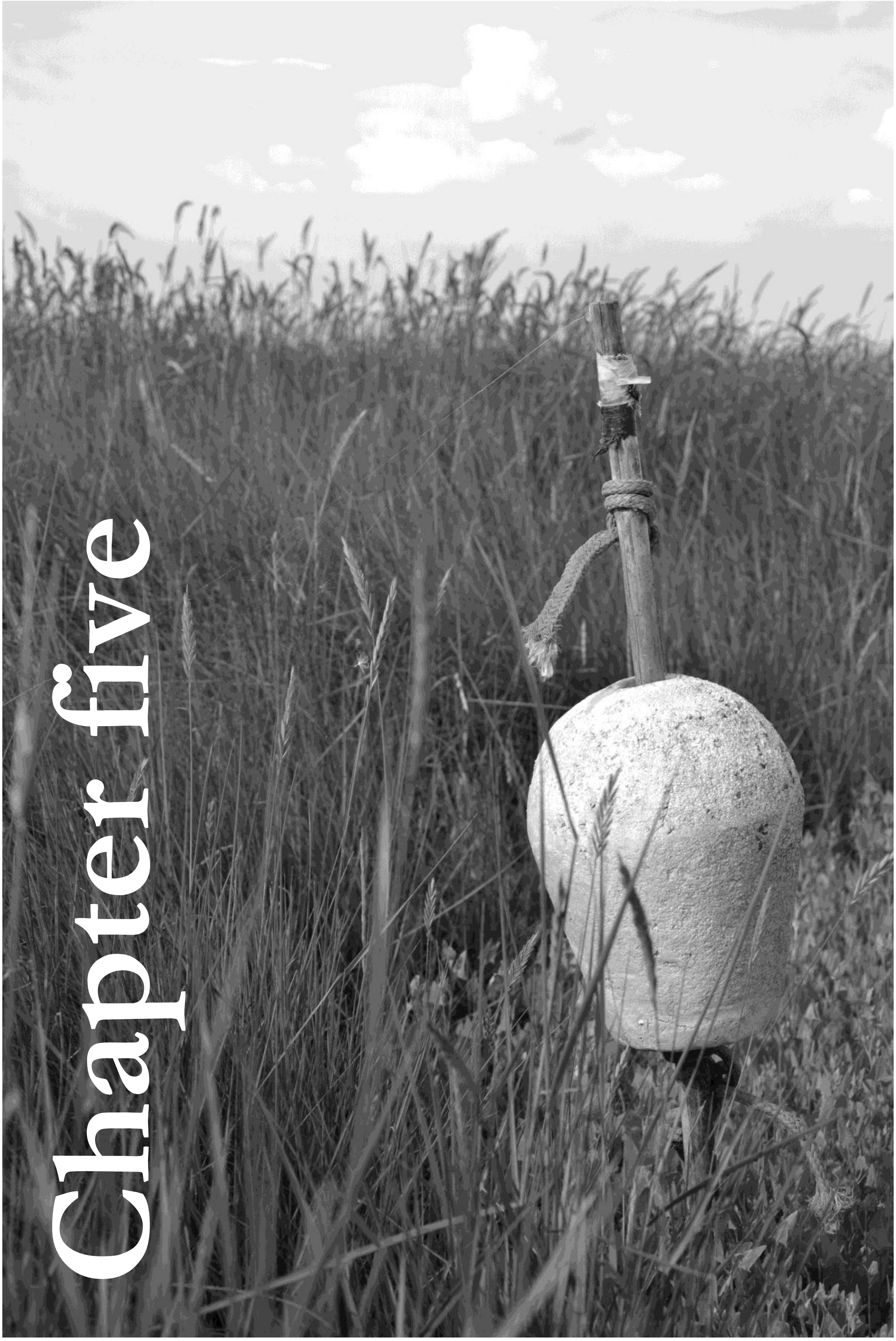
We expected that wave reflection at the end of the flume would promote turbulence of the suspended sediment and thus lower SDR. We calculated an initial mean flow velocity of approximately 0.2 m s^{-1}. This mean flow velocity was similar to that used in flume studies by Bouma et al. (2007), but about twice as high as flow velocity measured by Leonard and Luther (1995) within the marsh canopy, and in a flume study by Kozerski and Leuschner (1999). Changes of (horizontal) flow velocities during the experiment with increasing water-level were not measureable with the available equipment.

The reflections at the end of the flume might explain the higher decrease in SDR of flat surface in comparison to the circular ST-types measured at this part of the flume, in particular for the tile ST by wash-off of sediment. Kleiss (1996) used flat surface ST with a rough upper surface to minimize re-suspension of deposited sediments, whereas Steiger et al. (2003) did not find significant differences between flat surface ST with varying roughness. In addition, the smooth and low friction surface of tile ST increases the probability of losing sediment during collection (Gardner 1980, Kozerski and Leuschner 1999).

4.5 Conclusions

Differences in trapping efficiency impede the comparability of commonly used sediment trap types. Thus, a standardization in equipment for sediment measurements in intertidal habitats would be necessary. As a next step, we recommend field studies to compare different sediment trap types simultaneously under different inundation regimes, flow velocities, as well as different marsh types (mineral and organic). Short-term measurements of sediment-deposition rates with sedimentation traps like those compared in this study are especially useful for analyzing spatio-temporal variation in SDR and in their predictors. When calculating estimates of accretion rates from SDR, the bulk-density of the soil (see Butzeck et al. 2014) must also be analyzed. More reliable methods for estimating longer-term accretion rates in intertidal habitats include surface-elevation tables (Boumans and Day 1993), rod surface-elevation tables (Cahoon et al. 2002), marker horizons (French and Spencer 1993) or sedimentation-erosion bars (van Wijnen and Bakker 2001).

Chapter five



5 Synthesis

5.1 Key Findings

This thesis studied spatial and temporal sediment-deposition and vegetation patterns along elevational and salinity gradients, as well as the underlying environmental factors that influence marsh development. The thesis combined investigations of (i) the long-term (1980–2010) successional patterns of estuarine tidal marshes along a 97 stream-km long stretch with (ii) short-term (spring-neap cycle) sediment-deposition patterns in tidal freshwater, brackish and salt marshes of a high temporal and spatial resolution within the Elbe Estuary. To increase the global comparability of studies on sediment deposition patterns in estuarine marshes, an experiment on the trapping efficiency of different sediment traps in a flume was also included in the thesis. The key findings of the study include:

1) Tidal marsh area within the Elbe Estuary increased by 2 % from 1980 to 2010, but changes were unequally distributed between elevational and salinity zones. Salt and brackish high marshes increased by 22 and 4 %, respectively, whereas tidal freshwater high marshes decreased by 4 %. Low marshes decreased in all salinity zones between 4 and 30 %.

2) Tidal flats and high marshes showed a high persistence of 82 to 97 % between 1980 and 2010, whereas only 19 to 28 % of low marshes of 1980 persisted in 2010. In salt and brackish marshes more than two-thirds of low marshes expanded into high marshes (progressive succession), while in tidal freshwater low marshes, almost half of the 1980 low marshes developed into tidal flats (regressive succession).

3) Distance to the navigation channel was the major factor determining succession in salt and brackish marshes; the closer the distance to the channel, the higher the risk of regressive succession. In tidal freshwater marshes, river bank situation, changes of mean high water, and distance to the navigation channel were identified as main factors for marsh succession.

4) In the zone of tidal freshwater marshes, anthropogenic impacts by channel engineering caused strong decreases of mean low water and increases of mean high water between 1980 and 2010. It is quite likely that these interferences negatively modified marsh distribution, increased regressive succession, and thus, limited the quality of tidal freshwater marshes.

5) Bi-weekly sediment-deposition rates differed between tidal freshwater, brackish and salt marshes and were significantly higher (65–84 %) in low marshes than in high marshes. Sediment-deposition rates were highest in brackish low marshes and between 51 and 71 % lower in the low tidal freshwater and the salt marsh, respectively.

6) Highest suspended-sediment concentrations and longest inundations were found during fall and winter. Flooding duration and frequency was highest in tidal freshwater and lowest in salt marshes.

7) Decreasing sediment-deposition rates with increasing distances from the sedimentation source were recorded in all three marsh types.

8) Multiple regression models were able to explain between 71 and 79 % of variation in sediment-deposition patterns in tidal freshwater, brackish, and salt marshes. Suspended-sediment concentration was found to be the most important model predictor factor.

9) Assessing the possible stability of tidal marshes under projected accelerated sea-level rise and predicting future tidal marsh development needs to be based on results obtained for different spatial and temporal scales. Short-term investigations of sediment-deposition rates, which were conducted in slow-flow sections, showed that sediment-deposition rates in tidal low marshes of the Elbe Estuary generally seem to be sufficient to compensate moderate rates of sea-level rise. High marshes might be vulnerable due to insufficient input of sediment, and may regress into low marshes. The investigation on long-term dynamics of estuarine intertidal habitats in contrast showed a decrease of tidal low marshes.

10) Trapping performance of different sediment-trap types differed significantly in a flume study under controlled conditions. Highest sediment-deposition rates were found for circular trap types, which were 20 to 45 % higher compared to floor mat and tile sediment-trap types.

5.2 Sediment Deposition – A Matter of Scale and Measuring Methods

The results of this study demonstrate the importance of considering spatial and temporal factors for sediment-deposition rates along estuarine salinity and elevation gradients. I can clearly show that in estuarine marshes, changes in sediment-deposition rates on a short-term period and resulting successional pathways over a long-term period are strongly connected with various factors. Anthropogenic disturbances on different scales may alter hydro-dynamic patterns and must be considered during data interpretation.

The results of the field study on short term sediment-deposition rates presented in **chapter 3** show that in study sites with comparable conditions (e.g., located at slow-flow sections, range of distances to nearest creek and marsh edge, and creek size), variations in suspended-sediment concentration, inundations, and distance to the sediment source (marsh edge, creek) could be used to predict sediment-deposition rates in estuarine marshes. Variation in suspended-sediment concentration is related to highly variable physical factors such as freshwater discharge, tidal characteristics, wind regime, and water temperature, as well as biological activity and re-suspension processes at the marsh or tidal flats surface (e.g., Leonard et al. 1995, Fettweis et al. 1998, Allen 2000, Ruhl and Schoellhamer 2004, Temmerman et al. 2005, Bartholomä et al. 2009). Highest sediment-deposition rates in brackish marshes were related to the location of the estuarine turbidity maximum of the Elbe (see Kappenberg and Grabemann 2001, Bergemann 2005). Furthermore, suspended-sediment concentrations and sediment-deposition rates were higher in low than in high marsh zones. This pattern indicates continuous settling of sediment particles during "upmarsh" water movement, lower sediment- deposition rates with increasing distance from the sediment source, shorter inundation durations, lesser inundation frequencies, and lower inundation heights (French and Spencer 1993, Christiansen et al. 2000, van Wijnen and Bakker 2001). The higher inundation frequencies and heights in low marsh zones compared with high marshes are clearly related to elevation and have been previously shown by Chmura et al. (2001) and Temmerman et al. (2003a) for tidal marshes. Besides differences between elevational and salinity zones, strong seasonal differences were found. Highest suspended-sediment concentrations during fall and winter may be correlated with increasing inundation parameters, higher storm activity (van Proosdij et al. 2006a), and lower sediment stability of tidal flats due to a decrease of benthic diatoms abundance with decreasing temperatures (Underwood and Paterson 1993, Austen et al. 1999). In addition, several authors identified biomass as a factor affecting sediment-deposition and/or accretion rates in tidal marshes by belowground organic enrichment, particle capture by stems and leafs, and enhanced settling due to turbulent kinetic energy reduction (e.g., Leonard and Croft 2006, Neumeier and Amos 2006, Mudd et al. 2009, 2010). Furthermore, lower salinity stress under freshwater compared with saline conditions leads to higher productivity of the vegetation (Gough et al. 1994, Baldwin and Mendelssohn 1998) and might further increase sediment-deposition rates. However, no correlation between seasonal biomass and sediment-deposition rates were found in our study, which might be due to a higher relevance of other factors mentioned before.

In contrast to the short-term investigation (**chapter 3**), the long-term investigation conducted in **chapter 2** allowed a comprehensive overview of the historical development of intertidal habitats of the Elbe Estuary between 1980 and 2010. The use and comparison of different vegetation maps allowed indirect conclusions on sediment-deposition rates from changes of elevational zones (progressive or regressive succession). Here, in consequence of the large scale and long-term approach, varying environmental factors without a high temporal variability have to be used to explain successional pathways. To determine changes and to identify the direction of succession, more or less constant factors such as the distance to the navigation channel, as well as variable factors like changes of mean low and mean high water levels between the periods were considered. Overall, tidal marshes of the Elbe Estuary increased by 2 % from 1980 to 2010, which indicates that sediment input exceeded export. Increases of sediment deposition can be indirectly concluded from vegetation changes. Progressive and regressive succession was unequally distributed between elevational and salinity zones. Analyses showed the major importance of distance to navigation channel for marsh succession in salt and brackish marshes. Here, physical forces such as higher flow velocities (Leonard and Croft 2006), and wave activity (Temmerman et al. 2003a) promote regressive succession much stronger in areas situated closer to the navigation channel. These forces are more pronounced over tidal flats and decrease landwards from the vegetated marsh edge (Temmerman et al. 2005). Progressive succession in salt and brackish marshes prevailed in areas further away from the navigation channel. In tidal freshwater marshes, regressive succession predominated over all elevational zones, which might be explained by the generally lower distance to the navigation channel, compared to salt and brackish marshes. Tidal marshes of the Elbe Estuary were notably affected by channel engineering, which caused a decrease of mean low water and an increase in mean high water between 1980 and 2010. This change of tidal amplitude was especially pronounced in tidal freshwater marshes, which may have negatively modified the distribution and quality of marshes. In contrast to salt and brackish high marshes, decreases of tidal freshwater high marshes might be due to the interplay between shorter distances to the navigation channel and increasing tidal amplitude.

Chapter 2 and **chapter 3** illustrate the strong spatio-temporal dependence between the variation in environmental factors and anthropogenic impacts versus successional pathways and sediment-deposition rates. Results of the short-term field study on sediment-deposition rates at the Elbe Estuary (**chapter 3**) show that especially tidal high marshes seem to have adequate sediment-deposition rates to compensate moderate rates of sea-level rise (see chapter 5.3). In contrast, investigations of the long-term dynamics of intertidal habitats of the Elbe Estuary between 1980 and 2010 (**chapter 2**) show an increased risk of the development from low marshes into tidal flats with decreasing distance to the navigation channel. Therefore, it might be difficult to extrapolate the results from a small-scale study to an entire area, if hydro-morphologic variations, which are a characteristic feature for estuarine environments, were not fully covered (e.g., slow flow versus high flow velocity sections, natural elevated marsh edge versus marsh edge with enrockments, estuarine marshes with and without (grazing) management). In addition, even when comparing sediment-deposition rates on the same temporal and spatial scale, results significantly differed between different trap types (**chapter 4**). Disparities between different sediment-trap types might be caused by trap geometry (Bloesch and Burns 1980), differences in bottom shear stress (Kozerski and Leuschner 1999), trap dependent accelerations of local water flows (Butman et al. 1986), different amounts of lateral dispersal of trapped sediment to the surrounding surface (Neubauer et al. 2002), or washing off by rain or tides (Gardner 1980, Kozerski and Leuschner 1999). Therefore, sediment deposition rates obtained in studies using different trap types might not be directly comparable. In the future, a standardization of measuring equipment would largely increase the possibility of comparing results of different studies.

5.3 Anthropogenic Alterations – Impacts on Tidal Marsh Dynamics and Stability

In general, knowledge about large-scale changes of elevational zones allows conclusions about the sufficiency of sediment-deposition rates in relation to sea-level rise. However, regressive succession might indicate insufficient deposition rates (Cahoon and Reed 1995) or erosional processes (Allen 2000, van Proosdij et al. 2006a). Deposition rates that exceed increases of sea-level rise promote progressive succession (Allen 1990). In the Elbe Estuary, anthropogenic activities have highly influenced hydro-dynamics for centuries, resulting in changes in the dynamics of tidal marshes. The construction of dikes started approximately 1,000 years ago. Dike construction restricted tidal flooding to the non-diked marshes. The building of new dikes after the catastrophic storm surge of 1962 further reduced the area of tidal marshes by approximately 65 % between the city of Hamburg and Cuxhaven (ARGE Elbe 1984). In addition to diking, deepening and dredging altered the depth of the shipping lane from a minimum water depth of 4.5 m in 1843 to 14.5 m below the mean low-water line along the Elbe Estuary in 1999. This induced a gradual increase of the tidal amplitude from 1.8 to 3.6 m in Hamburg over the past 150 years, and as a consequence, changes in tidal currents have occurred: Today, the flood current is much faster than the ebb current leading to the so-called "tidal pumping" phenomenon (see Bergemann 2006, Kerner 2007). In the estuarine stretch close to the mouth where salt marshes develop, the increases of mean high and mean low water (see **chapter 2**) corresponded fairly well with current mean sea-level rise in the North Sea of 3.6 ± 0.7 mm year^{-1} (see Wahl et al. 2011). A similar relationship would be expected further upstream in the brackish and freshwater zone, but anthropogenic alterations, especially channel deepening, broadening, and straightening (see Arbeitsgemeinschaft für die Reinhaltung der Elbe 2007) changed the hydro-dynamic conditions. Increases in tidal amplitude and anthropogenic impacts were previously found to be related to regressive succession of tidal marshes (Cox et al. 2003). At the Elbe Estuary, hydrologic conditions in the stretch in which tidal freshwater marshes occur were notably altered. Here, a strong increase of mean high water and in contrast to the situation at the mouth of the Estuary and in the North Sea, a decrease of mean low water was found. My results indicated a strong impact of these anthropogenic alterations on marsh persistence. In tidal freshwater marshes with generally low distances to the navigation channel, decreases of mean low water and increases of flow velocities seem to be related to regressive succession (Kappenberg and Grabemann 2001), possibly as these hydrodynamic alterations enhance erosional processes of marshes and tidal flats.

Progressive succession and high marsh expansion of salt and brackish marshes were correlated with increases in mean high water levels. Similar results were previously described by Olff et al. (1997). Increases of mean high water prolong inundations, which increase the opportunity of suspended sediment in the flooding water to deposit on the marsh surface, resulting in an increase in elevation.

Annual dredging amounts from the navigation channel between Hamburg and Cuxhaven increased from around 11 million m^3 a^{-1} in 1979 to 18 million m^3 a^{-1} in 2007, although mean annual discharge rates remained constant (approximatively 800 m^3 s^{-1}; Hamburg Port Authority 2008). The increase of dredging amounts might be due to both a significant increase of suspended-sediment concentration within the water column of the Elbe Estuary and the previous mentioned "tidal-pumping", both of which are related to the last two main channel engineering projects in the late 1970s and 1999/2000. Therefore, tidal marshes indirectly benefit from these anthropogenic induced increases of suspended-sediment concentration. Suspended sediments mainly originated from broad areas of tidal flats at the mouth of Elbe, with a net export of approximately 100 million m^3 of sediments recorded within the last 30 years (Hamburg Port Authority 2008). Future increases in wind and wave energy (see IPCC 2007) may increase the erosion of tidal flats and further enhance suspended-sediment concentrations. However, there is not an unlimited sediment supply from tidal flats. Erosion and potential loss of tidal flats might strongly affect tidal marsh survival with regard to sea-level rise in the long run. In addition, insufficient sediment supply due to tidal flat erosion might lead to a submergence of tidal marshes (e.g., Nyman et al. 1990, Ward et al. 1998), and missing or limited tidal flats will enhance the vulnerability of tidal marshes during storms, causing damage to the vegetation structure, which stabilize surface sediments and diminish lateral erosion (Allen 2000, Adam 2002, Wolters et al. 2005). Finally, an increase of the erosion of the marsh edge can be expected (van de Koppel et al. 2005). These possible impacts might reduce the storm buffering ability of tidal marshes and affect cyclic marsh development. On the other hand, increases of storm activity and storm surge levels (Church et al. 2013) were especially important for high marsh accretion rates (see Schuerch et al. 2012). Therefore, it is difficult to determine the impact of accelerated sea-level rise on tidal marshes and tidal flats.

In estuaries, increases of sea-level will also cause salt water intrusion upstream, which would lead to a shift in the distribution of salt, brackish and tidal freshwater marshes, alter vegetation cover and species richness (Baldwin and Mendelssohn 1998). Simultaneously, the turbidity zone might also shift upstream with increasing salinity and alter sediment-deposition patterns. Climatic projections for the region of Elbe Estuary predict a decrease of the amount of summer precipitation (MPI-H 2006, Rechid et al. 2014), which would reduce freshwater discharge from rivers (Neubauer and Craft 2009), strengthen upstream salt water intrusion in estuaries during the peak of the vegetation period, and negatively affect the occurring species, particularly in tidal freshwater marshes. Overall, tidal freshwater marshes with highest susceptibility to increased salinity may suffer the most from effects caused by climate change and sea-level rise.

5.4 Future Challenges and Research Questions

The results of this study help to unravel the different spatio-temporal scales which are important to understanding marsh development. This study showed the benefit of using different approaches and investigation methods to assess marsh stability. Up to now, studies along salinity gradients, comparing sedimentation and vegetation dynamics between tidal freshwater, brackish and salt marshes were almost completely lacking. In the course of climate change and accelerated sea-level rise (see Church et al. 2013), substantial changes of inundation and salinity features might occur and alter the distribution of estuarine marshes along the salinity and elevational gradient. However, there is a lack of knowledge concerning the effects of multiple and interacting factors for sedimentation and vegetation dynamics in tidal marshes. This lack of knowledge needs to be filled by other studies. In the future, more realistic models accounting for temporal and spatial variation in predictor factors should be used to predict sediment deposition in estuarine marshes across regional and local gradients (Kleiss 1996, Temmerman et al. 2003a).

Differences in trapping efficiency impede the comparability of commonly used sediment-trap types; therefore short-term sediment measuring equipment and methods in intertidal habitats should be standardized. Additional field studies should be conducted to compare different sediment-trap types simultaneously under different inundation regimes, flow velocities, as well as in different marsh types. Also, established longer-term methods measuring accretion rates like surface-elevation tables (Boumans and Day 1993), rod surface-elevation tables (Cahoon et al. 2002), marker horizons (French and Spencer 1993) or sedimentation-erosion bars (van Wijnen and Bakker 2001) should be validated with the available short-term methods for measuring sediment-deposition rates.

Anthropogenic channel engineering is likely to influence marsh succession and persistence. However, further studies are necessary to understand the underlying mechanisms and to predict future developments. Future large assessments of tidal marshes of the Elbe Estuary will fortunately not be constrained by methodological problems as the Trilateral Monitoring and Assessment Program (TMAP) of the Wadden Sea, as well as the river basin management plan of the EU Water Framework Directive (WFD) will provide maps on a common methodological basis. This will enable researchers to identify pathways of marsh succession at a high spatial and temporal resolution and can assist in spotting problematical developments of tidal marshes, particularly in relation to channel engineering, which is regularly conducted in many estuaries worldwide (see Cox et al. 2003, Blott et al. 2006, Stralberg et al. 2011, Li et al. 2013). However, it must be noted that maps only present a snap-shot of the tidal marshes at any given time. Growth conditions in tidal marshes differ slightly between years due to variation in climate and other factors (e.g., mean temperature, mean inundation features, disturbances during winter), which might influence the comparability of aerial photographs and/or maps from different periods. This becomes especially important in the pioneer zone (low marsh), where intra-annual fluctuations of vegetation were observed during the study (see Appendix of chapter 5.5).

In many estuaries, conflicts between environmental and economic demands prevail. The Elbe estuary working group (2012) released an "integrated management plan (IMP) for the Elbe estuary", which included all Natura 2000 nature conservation areas, management targets, measures and their implementation and threats. The plan was conducted to summarize and determine relevant environmental monitoring observations and to prepare adaptations to estuary ecosystems in view of climate change. The IMP is the first extensive plan for the Elbe Estuary, where various stakeholders from administrations, nongovernmental organization, residents, and economy cooperate closely with each other. The IMP serves thus as guideline for future developments, however unfortunately without legal obligation.

The constructions of dikes for flood protection and creation of agricultural land have dramatically reduced the area of estuarine marshes in Northwest-Europe during the last centuries (Meire et al. 2005, van Koningsveld et al. 2008, Temmerman et al. 2012). Nowadays, increasing sea-level, storm surge frequencies, storm surge levels (IPCC 2007), as well as anthropogenic impacts increase the tidal prism and the flooding risk in the upstream areas of estuaries. To overcome these negative developments and to preserve the unique ecosystem services of tidal marshes (storm and flood buffering, erosion control, nutrient cycling, filter for pollutants, and many more, see Mitsch and Gosselink 2000, Costanza et al. 1997, Costanza and Mageau 1999, Kirwan and Megonigal 2013), adaption strategies are required, and may include measures to increase the water storage volume to reduce the tidal amplitude (Vandenbruwaene et al. 2011). In the Elbe Estuary, two first small areas of dike relocations (Hahnöfersand, 105 ha, finished in 2005) were conducted and an additional project is planned for the next years (Kreetsand, 30 ha, intended to be finished in 2016). It must be stated, however, that these areas are comparatively small and that they would lower the tidal amplitude by only a centimeter. Therefore, additional and much larger de-embankment projects should be taken into consideration for the future.

5.5 Appendix

The appendix shows the intra-annual fluctuations of the low marsh vegetation adjacent to the tidal flats between April 2010 and March 2011, at the tidal freshwater marsh (A.1) and brackish marsh (A.2) at the Elbe Estuary. The arrangement of the panoramic pictures, from the top to the bottom:

A.1: Tidal freshwater marsh (Haseldorfer Binnenelbe, Germany)

April 2010, June 2010, September 2010 (pp 80)

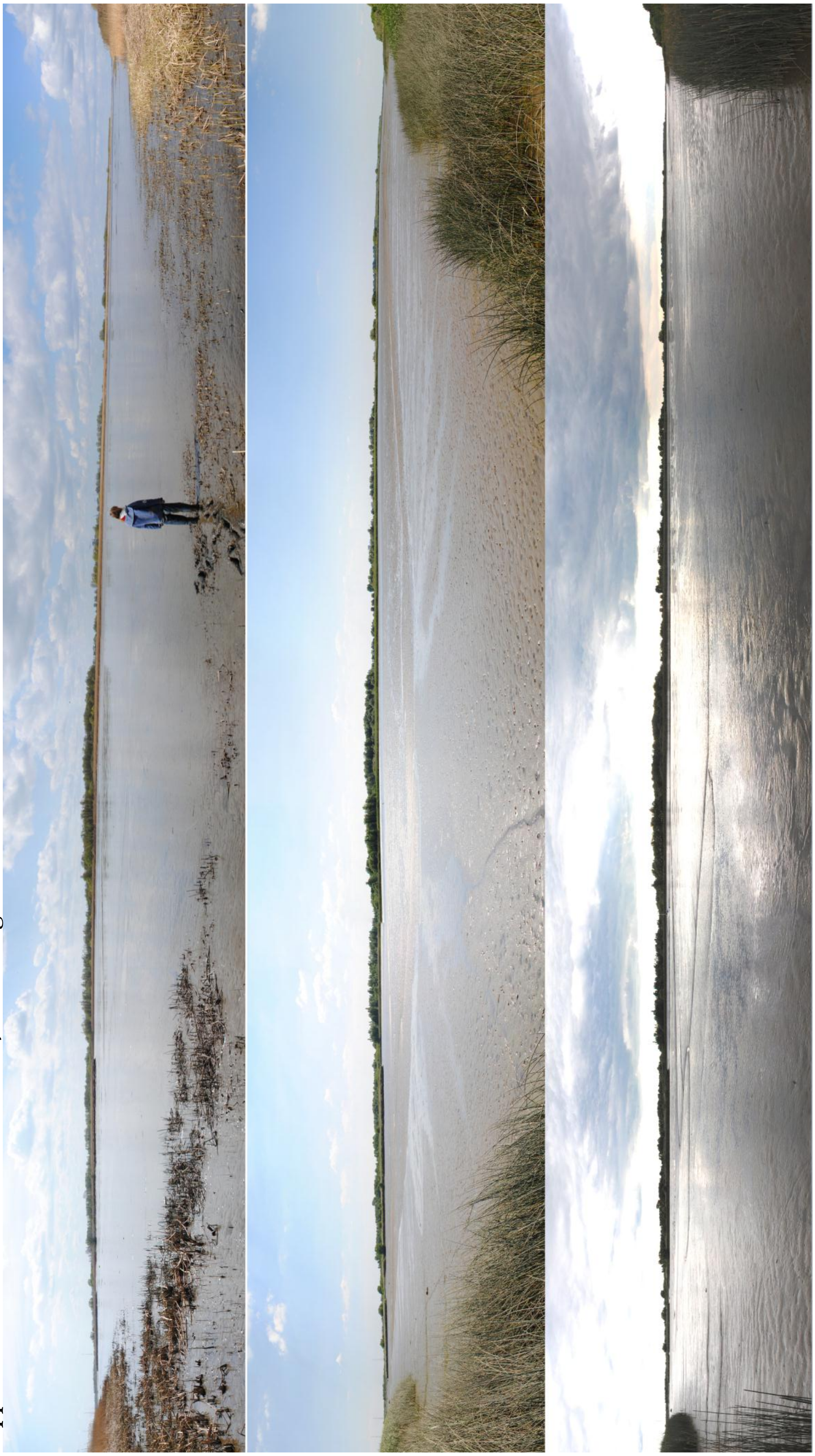
December 2010, January 2011, February 2011 (pp 81)

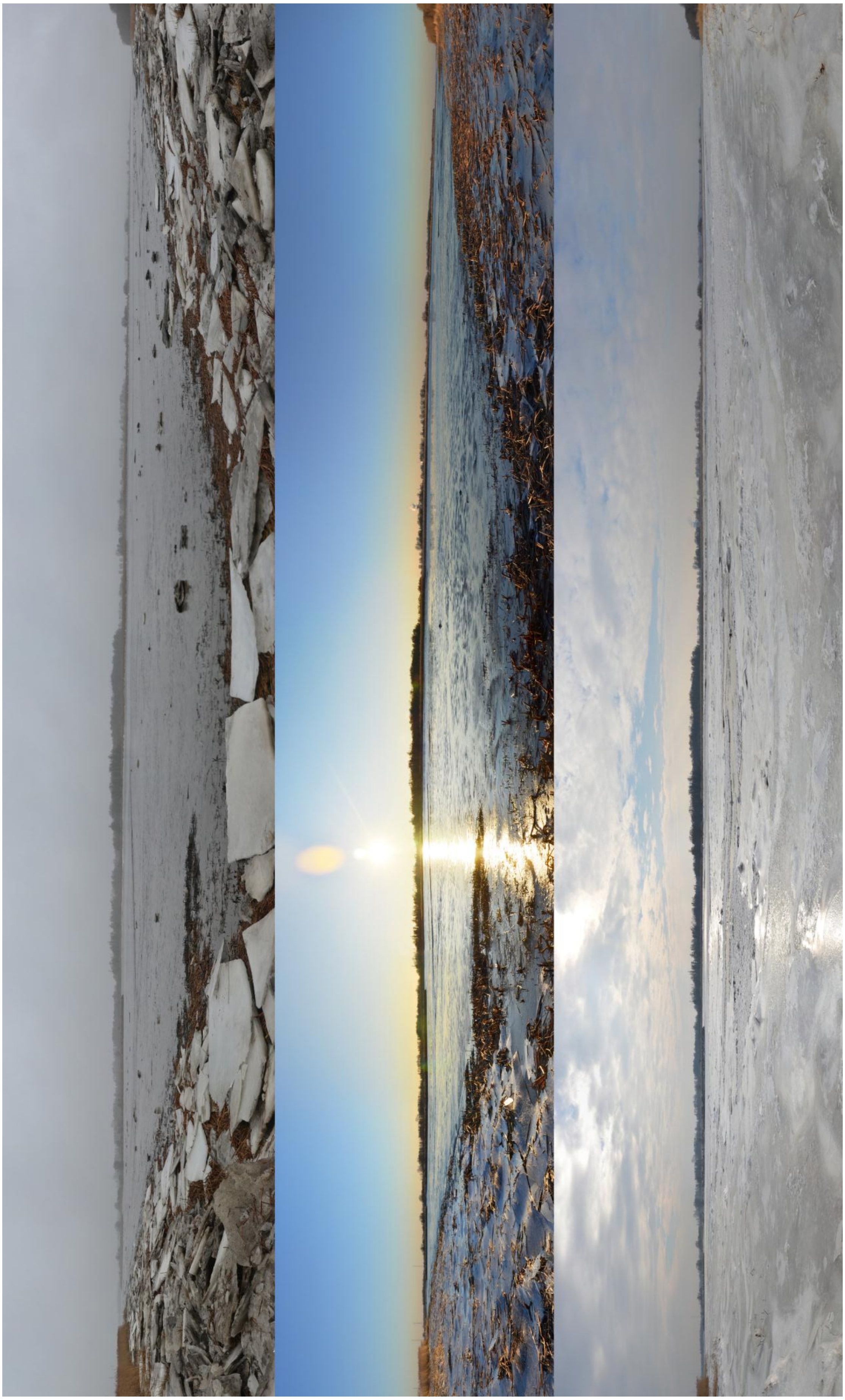
A.2: Brackish marsh (Neufeld, Germany)

April 2010, May 2010, June 2010, August 2010, September 2010 (pp 82)

October 2010, December 2010, January 2011, February 2011 (pp 83)

Appendix A.1: Tidal freshwater marsh, marsh edge





Appendix A.2: Brackish marsh, marsh edge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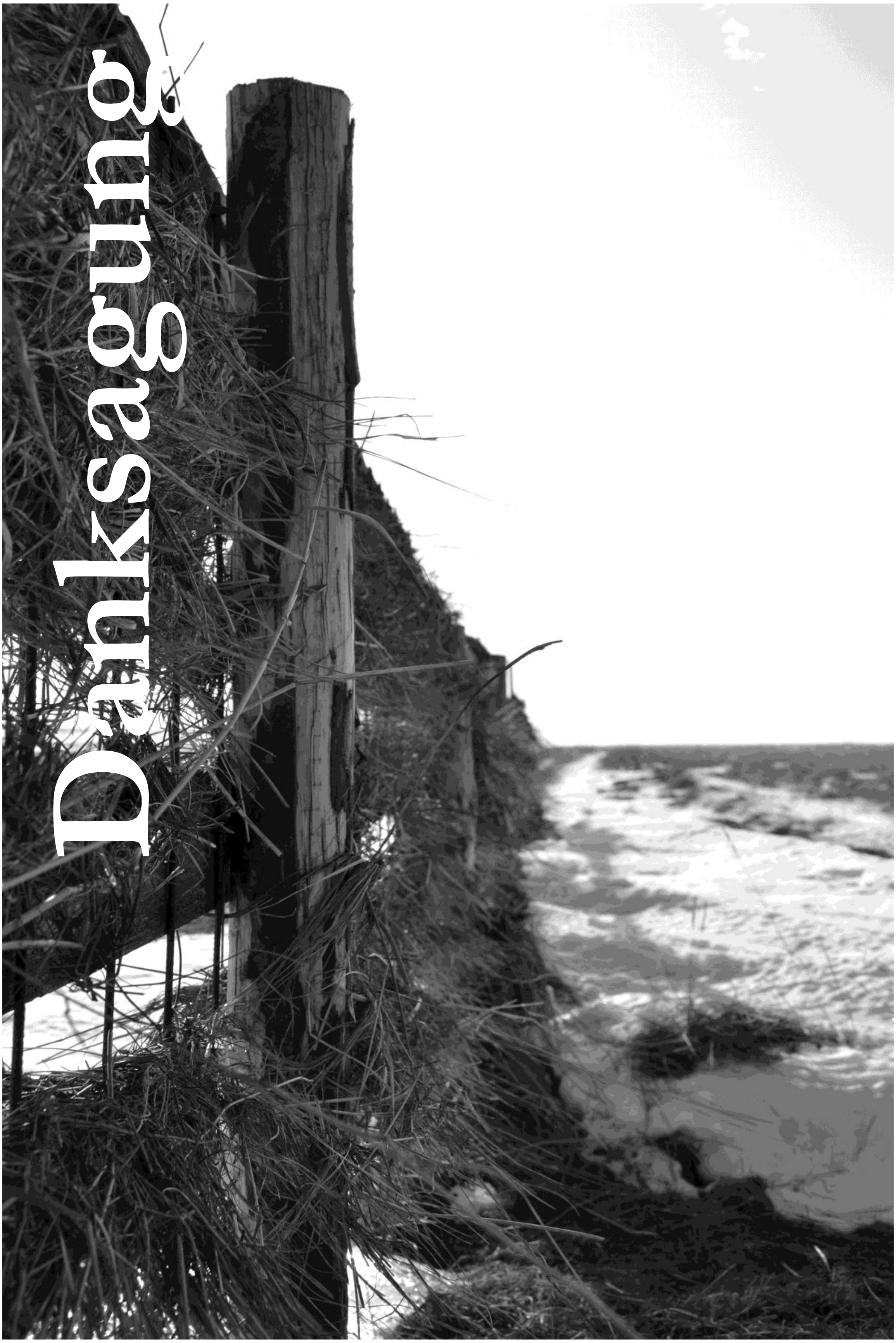
Danksagung



Danksagung – Acknowledgments

Viele Personen haben mich in den letzten Jahren bei meiner Promotion unterstützt. Ohne sie wäre die vorliegende Arbeit nicht in dem Umfang möglich gewesen. Ihnen allen gebührt mein herzlichster Dank.

Mein ganz besonderer Dank gilt Kai Jensen, der mich während meiner Doktorarbeit sehr engagiert betreut hat. Ich bin Dir sehr dankbar für das Vertrauen in meine Arbeit, deine kritischen Nachfragen, die stetige Unterstützung, nicht nur bei der Planung des Auslandsaufenthaltes und für die mir zur Umsetzung meiner Ideen gegebenen Freiräume.

Sehr herzlich möchte ich mich bei Annette Eschenbach für die Co-Betreuung und die Begutachtung meiner Arbeit sowie die Teilnahme an meinem ESTRADE-Panel bedanken. Ich habe Deine hilf- und aufschlussreichen Anmerkungen zu meiner Arbeit und den Manuskripten stets sehr gern aufgenommen.

Stefanie Nolte gilt ein besonderer Dank. Deine intensive Betreuung und stetige Motivation in den letzten Monaten haben diesen Zielspurt erst möglich gemacht.

Bei der gesamten AG Angewandte Pflanzenökologie und der AG Biodiversität der Nutzpflanzen möchte ich mich für das angenehme Arbeitsumfeld und ihre Unterstützung bedanken. Besonders danke ich Claudia Mählmann für ihre Hilfsbereitschaft in allen Lebens- und Formularlagen und ihre „schlagfertigen" Erwiderungen auf den ein oder anderen frechen Spruch. Detlef Böhm danke ich für die organisatorische Hilfe bei den Labortätigkeiten und der Bereitstellung eines immer gut gefüllten Werkzeugkastens. Marion Klötzl und Sigrid George danke ich für ihre Hilfsbereitschaft im Gewächshaus und ihr Durchhaltevermögen über 12 Monate lang den pikant-herben Geruch von schlammigen Wasserproben und trocknender Arbeitskleidung ertragen zu haben. Ich danke Kristin Ludewig und Franziska Rupprecht für die schöne Bürogemeinschaft, umfangreiche statistische Erklärungen und das ein oder andere Gesprächsthemen abseits der Doktorarbeit.

Bedanken möchte ich mich bei den (unzähligen) Leuten, die mich während der insgesamt 16-monatigen Feldkampagne tatkräftig unterstützt haben. Insbesondere bei Melly, die mich mehr als 10 Monate, alle 14 Tage im Gelände bei Temperaturen zwischen -15°C bis +35°C bei Nebel, Regen, Schneestürmen und Sonnenschein unterstützt hat. Du warst die einzige Person, die meinem Schritttempo folgen konnte! Des Weiteren gebührt ein großes Dankeschön den geländetauglichen Studentinnen und Studenten Alena L., Anna R., Anna S., Caroline T., Dennis S., Jana H., Janina S., Jasmin F., Jens D., Julia F., Kadda, Kim J., Kira K., Lotte K., Ludmilla P., Nadine B., Nikola L., Simon T., Theresa M., Wiebke M., die mich im Gelände und auch bei der Nachbereitung der Proben unterstützt haben. Ich denke ihr hattet alle während des effektiven „Workout" im schlickigen Schilf-Dschungel euren Spaß, auch wenn manche/r komischerweise nach nur einem Ausflug eine weitere Begleitung dankend abgelehnt haben.

Personelle Engpässe im Laufe der Untersuchungen und meine temporär eingeschränkte Mobilität im Winter 2009/2010 wurden zusätzlich hingebungsvoll von meinen Arbeitskollegen Frauke Müller, Katharina Schmidt, Kristin Ludewig, Sebastian Schmidt und Wiebke Schoenberg aufgefangen. Nochmals vielen Dank für Euren Eis- und Schneeeinsatz! Andreas Weiss, Jessica Tietke, Kerstin Hansen, Nina Kumbruck und Victoria Magath danke ich für den fachlichen Austausch und Begleitung zu meinen Untersuchungsflächen. Ebenso danke ich Cordula Stadter, Daniela Siedschlag, Martin Haseloff, Martin Stelzer, Stephan

Hollesch sowie meiner liebreizenden Schwester Juliane für die Unterstützung während ein paar Geländetagen und allen Ungenannten, die ich vergessen haben sollte. Melly und Wiebke M. danke ich zusätzlich für meine Rettung mittels Spanngurt aus dem Neufelder Priel. Ohne Euer beherztes Eingreifen hätte die Arbeitsgruppe eventuell eine neue Wathose benötigt und Ihr hättet alle Proben alleine zum Auto tragen müssen.

Ein besonderes Dankeschön gilt meiner Abschlusskandidatin Alena Lucht, die ich im Rahmen ihrer Bachelorarbeit mit betreuen durfte. Sie war mit Begeisterung und Engagement bei der Sache und hat wichtige Daten zur Biomasseverteilung im Elbe-Ästuar geliefert, die nicht nur in meine Arbeit eingeflossen sind.

Uwe Schröder von der Bundeanstalt für Gewässerkunde danke ich vielmals für die ausgezeichnete fachliche Zusammenarbeit bei unserem „Langzeitprojekt“. Ich verdanke Dir einen umfangreichen, praxisnahen Erkenntnisgewinn zur Entwicklung der Elbmarschen und fühle mich bestätigt, dass eine interdisziplinäre Zusammenarbeit und der Erfahrungsaustausch zwischen Behörden und Forschungseinrichtungen unabdingbar ist.

Bei Wiebke Schoenberg möchte ich mich sehr herzlich für den intensiven Austausch, für diverse organisatorisch-koordinierende Entlastungen im KLIMZUG-NORD Projekt und die Koordination des beständigen Kaffee- und Milchnachschubes bedanken. Meinen AG-Kollegen und KLIMZUG-NORD Mitstreitern Katharina Schmidt, Kristin Ludewig und Sebastian Schmidt danke ich besonders für die gemeinsame Zeit und den positiven Gedankenaustausch. Jetzt haben wir es alle geschafft!

Für die Teilnahme an interessanten Seminaren, Retreats und der Bezahlung des Hin- und Rückfluges in für den 3-monatigen Auslandsaufenthaltes bedanke ich mich bei der Graduiertenschule ESTRADE (Landesexzellenzinitiative, LExI) insbesondere für die ausgezeichnete Koordination durch Tobias Gebauer und Andrea Staudler. Bei Alexander Gröngröft bedanke ich mich für die Unterstützung und die Teilnahme an meinem Panel-Team.

Diese Doktorarbeit wurde im Rahmen des Verbundprojektes KLIMZUG-NORD angefertigt und vom Bundesministerium für Bildung und Forschung (BMBF) gefördert (Fördernummer: 01LR0805D). Ich danke Jürgen Becker und Arne von Maydell von der TU Tech für die Projektkoordination und allen Projektbeteiligten insbesondere aus der Querschnittsaufgabe Naturschutz und der Arbeitsgruppe Naturschutz und Klimawandel.

Weiterhin bedanke ich mich:

- bei den Praktikanten Anni, Andreas, Julian, Nadine, Nora, Reinhild, Tim, Thomas für ihre Hilfe im Rahmen ihrer Praktika
- bei Sebastian Lindhorst und Kollegen/innen vom Institut für Geologie der Universität Hamburg für die schnelle Bearbeitung meiner Proben zur Korngrößenbestimmung
- bei Jens Oldeland für eine umfangreiche Einführung in R und unsere „Drohnen-Fachgespräche“
- bei Martin Stock vom Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein für den guten fachlichen Austausch und die Datenbereitstellung

- bei Carl Carstensen und Michael Beverungen vom Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein für die Betretungsgenehmigungen im Dieksanderkoog
- bei Bernd-Ulrich Netz (Behörde für Stadtentwicklung und Umwelt, Hamburg) für die Bereitstellung von historischen Daten zur Gebietsentwicklung in den Süßwassermarschen der Elbe
- bei Uwe Helbing (Schutzgebietsbetreuer Naturschutzgebiet „Haseldorfer Binnenelbe mit Elbvorland"), Jörg Kastrup und Oliver Carstens (Untere Naturschutzbehörde Kreis Pinneberg) für die Betretungsgenehmigungen in der Haseldorfer Marsch
- beim „Neufelder Schäfer" Udo Engel für angeregten Diskussionen zur Entwicklung der Vordeichsflächen, das (Für und) Wider der fehlenden Beweidung der deichfernen Flächen und lustige Anekdoten zur Praxisferne mancher Wissenschaftler
- bei Ronny Beyer vom Wasser- und Schifffahrtsamt Hamburg für die umfassende Bereitstellung von digitalen Kartierungen und Luftbildern
- bei Sylvin Müller-Navarra vom Bundesamt für Seeschifffahrt und Hydrographie für die Aufbereitung von Hochwasserdaten
- bei der Generaldirektion Wasserstraßen und Schifffahrt -Außenstelle Nord- für die Bereitstellung von historischen Luftbildern der Schleswig-Holsteinischen Salzmarschen

Warmest thanks go to Andy Baldwin who supervised me and organized several things during my stay at the University of Maryland, USA. I am also very grateful to Jenny Allen, Leah Hope-Menzies Beckett, Metthea Yepsen, and Heike Markus-Michalczyk for several leisure activities, the assistance during the work at the flume channel and at the Nanticoke. I would love to visit the Nanticoke by (speed-)boat again!

I would particularly like to thank Jenny Allen for "last-minute" proofreading of chapters 1, 2, 4, and 5.

Die finalen Grußworte gehen an alle meine Freunde, die mich durch kleine oder größere Gesten im Laufe der letzten Jahre motivierend unterstützt haben und an meine Eltern Gabriele und Jörg, meine Schwester Juliane sowie meinen Großeltern Elly und Gerhard. Ohne Eure Unterstützung wäre ich nicht dort angekommen, wo ich heute bin!

Eidesstattliche Versicherung

Hiermit erkläre ich an Eides statt, dass ich die vorliegende Dissertationsschrift selbst verfasst und keine anderen als die angegebenen Quellen und Hilfsmittel benutzt habe.

Hamburg, 19.09.2014 Christian Butzeck

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16 September 2014

Re: Dissertation written by Christian Butzeck

To whom it may concern,

As a native English speaker and experienced proofreader, I do hereby confirm that the abovementioned dissertation “Tidal marshes of the Elbe Estuary-spatial and temporal dynamics of sedimentation and vegetation“ has been written in correct and concise English.

Kind regards

Jenny Allen