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Ecosystem services in coastal *Phragmites* wetlands at the southern Baltic Sea: Nutrient regulation, water purification and erosion control

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Contents

Summary

Coastal wetlands can provide a variety of ecosystem services such as protection against coastal erosion, pollutant buffering or nutrient regulation. Biological, physical and chemical processes in coastal wetlands need to be studied in order to understand how such services are supplied and might change under human influences. In this thesis often neglected coastal wetlands at the southern Baltic Sea are investigated with regard to regulating ecosystem services. The Baltic Sea region has become an important economic driver. Therefore, anthropogenic stresses on coastal ecosystems will continue to rise. Eutrophication remains a major environmental problem although much improvement was achieved through better sewage treatment. Coastal wetlands with their ability to buffer nutrients have the potential to counteract eutrophication. Large wetland areas of the southern Baltic coast are dominated by *Phragmites australis* (Cav.) Trin. ex Steud., a perennial grass which has the potential to act as a bio-engineer of its own environment. Regulating ecosystem services such as water purification, erosion control and nutrient dynamics were studied exemplarily in *Phragmites* wetlands at the Darss-Zingst Bodden Chain, a lagoon system at the southern Baltic Sea. The objectives were to analyze (1) how these regulating ecosystem services are supplied, (2) how they are interrelated, and (3) how physical, biological and chemical processes govern them.

Anthropogenic activities in the hinterland can have a large impact on coastal wetland sediments. Sediment characteristics are important as they influence the water purification capacity of wetlands by adsorbing and storing pollutants. Therefore, sediment composition at two *Phragmites* sites with different land uses was investigated: While one wetland is unconfined and directly borders arable fields, the other is confined landwards by a dyke with pastures in the hinterland. The research results show that influences from the sea on sediment compositions were minor compared to the influences from land. Heavy metal concentrations were significantly elevated in the wetland zone that borders directly an arable field where crop production with fertilizer application took place at least since the 1950ies.

Thus, the *Phragmites* wetland bordering cropland has served for decades as a ‘pollutant buffer’ and ‘water purification system’. However, under specific environmental conditions carrying capacities are already reached for some elements. In the temporarily anoxic basin zone of the wetland, redox-sensitive elements such as iron-phosphorus compounds can be released from sediments into the water. Research results indicate a threshold-type behavior in the relationships between oxygen saturation, water level and soluble reactive phosphorus in coastal wetland waters. During hydrodynamically calm conditions with stagnant waters, oxygen depletion can cause phosphorus release from sediments. However, if turbulent kinetic energy in the water rises again and oxygen conditions improve, a thin oxic micro-layer rich in iron will be sufficient to readsorb phosphorus quickly.

Whether a coastal wetland is unconfined or confined by a dyke did not only influence sediment composition, but also sediment trapping and thus erosion regulation. *Phragmites* is capable to accrete vertically if litter production is high and particles supplied from sea or land are captured in the reed stands. However, the results of this thesis regarding surface elevation changes show that none of the measuring locations in the dyked wetland could currently keep up with the local sea level rise. At the wetland fringe erosion prevailed and in the wetland interior the high organic matter content of the sediment increased compaction processes. Upward growth and wetland evolution of confined sites may be severely hampered by the suppression of sediment supply from land

Hence, it is important to identify the specific site conditions of a coastal wetland to know how regulating services are supplied and how they are interrelated. While provisioning services often compete with each other (e.g. crop vs biomass for energy), the regulating services investigated in this thesis can sustain and support each other. A wetland that traps particles and suppresses erosion serves well as an active ‘pollutant buffer’ storing excess nutrients or heavy metals in retained sediments.

Zusammenfassung

Küstenfeuchtgebiete stellen eine Vielzahl von Ökosystemleistungen wie beispielsweise Schutz gegen Küstenerosion, Nährstoffregulierung oder einen Puffer gegen Schadstoffe bereit. Biologische, physikalische und chemische Prozesse müssen detailliert untersucht werden, um zu verstehen wie solche Dienste funktionieren, wie sie interagieren und wie sie sich unter menschlichen Einflüssen verändern. Die Ostseeregion hat sich zu einem wichtigen Wirtschaftsraum entwickelt, wodurch die Belastung der Küstenökosysteme durch den Einfluss des Menschen verstärkt wird. Obwohl Verbesserungen im Bereich der Abwasserbehandlung erreicht wurden, bleibt Eutrophierung ein großes Umweltproblem. Küstenfeuchtgebiete mit ihrer Fähigkeit Nährstoffe zu puffern, haben das Potenzial der Eutrophierung entgegenzuwirken. Weite Teile der südlichen Ostseeküste werden von *Phragmites australis* (Cav.) Trin. ex Steud. dominiert, ein mehrjähriges Gras mit dem Potenzial als Bio-Ingenieur auf seine eigene Umgebung einzuwirken. Nährstoffregulierung, Wasserreinigung und Erosionsregulierung wurden exemplarisch in Schilfgürteln an der Darß-Zingster-Boddenkette, einem Lagunen-System der südlichen Ostsee, untersucht. Ziel war es zu analysieren (1) wie diese Regulierungsleistungen bereitgestellt werden, (2), wie sie miteinander agieren und wechselwirken, und (3) wie physikalische, biologische und chemische Prozesse sie steuern.

Anthropogene Aktivitäten im Hinterland haben einen großen Einfluss auf Sedimente von Küstenfeuchtgebieten. Sedimente können Schadstoffe adsorbieren und speichern und somit die Wasserreinigung in Küstenfeuchtgebieten unterstützen. Aus diesem Grund wurden Sedimentanalysen an zwei unterschiedlichen Schilfstandorten durchgeführt: Während der eine Schilfgürtel direkt an einen Acker angrenzt, ist der andere Schilfgürtel durch einen Deich landwärts von den angrenzenden Weiden getrennt. Die Forschungsergebnisse zeigen, dass die meerseitigen Einflüsse auf die Sedimenteigenschaften im Vergleich zu den landseitigen Einflüssen vernachlässigbar sind. In dem an den Acker grenzenden Feuchtgebiet waren die Schwermetallkonzentrationen signifikant erhöht. Der Acker wird seit über 50 Jahren

bewirtschaftet und gedüngt. Somit fungiert der Schilfgürtel seit Jahrzehnten als aktiver „Schadstoff-Puffer“. Unter bestimmten Umweltbedingungen ist die Tragfähigkeit für einige Stoffe jedoch bereits erreicht. In den temporär anoxischen Becken des Schilfgürtels können redox-sensitive Elemente, wie zum Beispiel Eisen-Phosphor-Verbindungen, freigesetzt werden. Hierzu zeigen die Forschungsergebnisse ein Schwellenwert-Verhalten zwischen Sauerstoffsättigung, Wasserstand und löslichen reaktiven Phosphor im Wasser. Während hydrodynamisch ruhigen Bedingungen führt Sauerstoffmangel zu einer Phosphorfreisetzung aus den Sedimenten. Steigt jedoch die turbulente kinetische Energie im Wasser wieder an, so verbessern sich die Sauerstoffbedingungen rasant und eine dünne oxische, eisenreiche Mikroschicht ist ausreichend um den gelösten Phosphor wieder zu adsorbieren.

Ob ein Küstenfeuchtgebiet eingedeicht ist oder nicht, beeinflusst nicht nur die Sedimentzusammensetzung, sondern auch die Fähigkeit Sedimente einzufangen und somit Erosionsprozessen entgegenzuwirken. Schilf ist in der Lage mit dem Meeresspiegelanstieg mitzuhalten solange Biomasse- und Streuproduktion ausreichend hoch sind und Partikel, die see- oder landseitig eingetragen werden, in den Schilfbeständen zurückgehalten werden. Die Forschungsergebnisse zeigen allerdings, dass die Veränderungen der Sedimentoberfläche an keinem der Messpunkte in dem eingedeichten Feuchtgebiet zurzeit mit dem lokalen Meeresspiegelanstieg mithalten können. Am seeseitigen Rand des Feuchtgebiets dominierte Erosion und im Inneren verstärkte der hohe Anteil an Organik im Sediment Verdichtungsprozesse. In eingedeichten Feuchtgebieten wo Sedimentzufuhr vom Land unterbunden wird, kann die Entwicklung von Küstenfeuchtgebieten in Gefahr sein.

Um zu verstehen, wie Regulierungsleistungen erbracht werden und wie sie miteinander in Beziehung stehen, ist es wichtig die spezifischen Standortbedingungen eines Küstenfeuchtgebiets zu identifizieren. Während Versorgungsleistungen oft miteinander konkurrieren (z.B. Lebensmittelproduktion vs. Biomasse zur Energiegewinnung), können Regulierungsleistungen sich gegenseitig unterstützen und verstärken. Ein

Küstenfeuchtgebiet, das Partikel einfängt und Erosion unterbindet, fungiert besser als aktiver ‚Schadstoffpuffer', der überschüssige Nährstoffe und Schwermetalle in den Sedimenten zurückhält.

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Abbreviations

ADV	Acoustic Doppler Velocimeter
CAP	European Common Agricultural Policy
DWD	German weather service
LOI	Loss-On-Ignition
m AMSL	meter Above Mean Sea Level
PCA	Principal Component Analysis
PSU	Practical Salinity Units
RR	Random Roughness
S_{max}	Langmuir Sorption Maximum
SET	Surface Elevation Table
SLR	Sea Level Rise
SRP	Soluble Reactive Phosphorus
T	Tortuosity
TKE	Turbulent Kinetic Energy
WSV	Federal Water and Shipping Administration

List of publications and author contributions

I. Karstens, S., Buczko, U., Glatzel, S., 2015. Phosphorus storage and mobilization in coastal *Phragmites* wetlands: Influence of local-scale hydrodynamics. Estuarine, Coastal and Shelf Science 164, 124–133.

➔ *Chapter 3 of this thesis*

Karstens, S.: Study design, field work, data analysis, writing, editing

Buczko, U.: Study design, field work, editing

Glatzel, S.: Study design, editing

II. Karstens, S., Schwark, F., Forster, S. Glatzel, S., Buczko, U., 2015. Sediment tracer tests to explore patterns of sediment transport in coastal reed beds – a case study from the Darss-Zingst Bodden Chain. Rostocker Meeresbiologische Beiträge 25, 41–57.

Karstens, S.: Study design, field work, data analysis, writing, editing

Schwark, F..: Study design, field work, data analysis, editing

Forster, S.: Study design, editing

Glatzel, S.: Study design, editing

Buczko, U.: Study design, editing

III. Karstens, S., Buczko, U., Jurasinski, G., Peticzka, R., Glatzel, S., 2016. Impact of adjacent land use on coastal wetland sediments. Science of the Total Environment 550, 337-348.

➔ *Chapter 2 of this thesis*

Karstens, S.: Study design, field work, data analysis, writing, editing

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Peticzka, R.: Data analysis, editing

Glatzel, S.: Study design, editing

IV. Karstens, S., Jurasinski, G., Glatzel, S., Buczko, U., 2016. Dynamics of surface elevation and microptopography in different zones of a coastal *Phragmites* wetland. Ecological Engineering 94, 152-163.

➔ *Chapter 4 of this thesis*

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Buczko, U.: Study design, editing

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Buczko, U.: Study design, field work, data analysis, writing, editing

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Glatzel, S.: Study design, field work, editing

Law, M.M.S.: field work

Xu, J.: field work

Lai, D.Y.F.: Study design, field work, editing

1. Introduction

1.1 Background

Coastal ecosystems are heavily impacted by human activities and belong to the most threatened ecosystems worldwide (Lotze et al. 2006; Worm et al. 2006; Halpern et al. 2008). Anthropogenic pressures are on the rise - including tourism, ship traffic, urbanization or aquaculture - and deterioration of coastal resources is increasing (Barbier et al. 2011). For the Baltic Sea the most serious problem is eutrophication (Schiewer and Schernewski 2004). Coastal wetlands have the potential to play a fundamental role as buffer and filter for nutrients, counteracting eutrophication. Coastal wetlands extend seawards to water depths where light penetration is still high enough to allow photosynthesis of benthic plants, while the landward edge of a coastal wetland is limited by the influence of salt water (Perillo et al. 2009). While marine research usually stops at the seaward edge of coastal wetlands, terrestrial research along all disciplines (e.g. soil science, biology, hydrology) often stops on the landward side of a coastal wetland. Consequently, the interface between marine and terrestrial ecosystems is under-studied. The research of this thesis took therefore place directly in coastal wetlands along the Darss-Zingst Bodden Chain, a lagoon system at the southern Baltic Sea. It is part of the broader research project “Baltic Coastal System Analysis” (BACOSA, a part or the research program FONA — Research for Sustainable Development; project number 03F0665A, for further information please see http://www.deutsche-kuestenforschung.de/bacosa-210.html).

1.2 Ecosystem services: How ecosystem functioning and ecological integrity contribute to human well-being

Various definitions for ecosystem services have been proposed, *inter alia* in the popular Millennium Ecosystem Assessment (MA 2003) or in the well-known TEEB study (The Economics of Ecosystems and Biodiversity; de Groot et al. 2010). Some authors even argue that a common and generally accepted classification system might not be possible because ecosystem services are too case-specific (e.g. Costanza 2008; Burkhard et al. 2012, 2014). For the BACOSA project as well as for this thesis the definition of Burkhard et al. (2012, page 2) was chosen: “Ecosystem services are the contributions of ecosystem structure and function – in combination with other inputs – to human well-being.” This definition matches the definition of the Millennium Ecosystem Assessment (MA 2003, page 3) quite well “Ecosystem services are the benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits.” However, in the approach of Burkhard et al. (2012), which was adopted for the BACOSA project, only three ecosystem service categories exist (regulating, provisioning, cultural services), while the supporting services as they still appeared in the Millennium Ecosystem Assessment were substituted by ecosystem functions. Ecosystem functions and ecological integrity (Figure 1-1) are the base for the system’s self-organization capacity and general capability to provide ecosystem services (Müller and Burkhard 2010; Kandziora et al. 2013). Hence a multitude of ecosystem properties and functions such as storage capacity for nutrients, energy and water or the biotic diversity influence the potential of an ecosystem to supply regulating, provisioning or cultural services (for a complete list of proposed indicators for ecological integrity see Kandziora et al. 2013).

While provisioning services refer to tangible products (e.g. crops, timber, fish) and cultural services refer to intangible products (e.g. recreation, landscape aesthetic), they have in common that both are widely acknowledged by humans (Kandziora et al. 2013). This acknowledgement and awareness are seldom the case for regulating services, although humans widely benefit from such services including natural hazard protection, water purification or erosion control (Barbier et al. 2011; Kandziora et al. 2013).

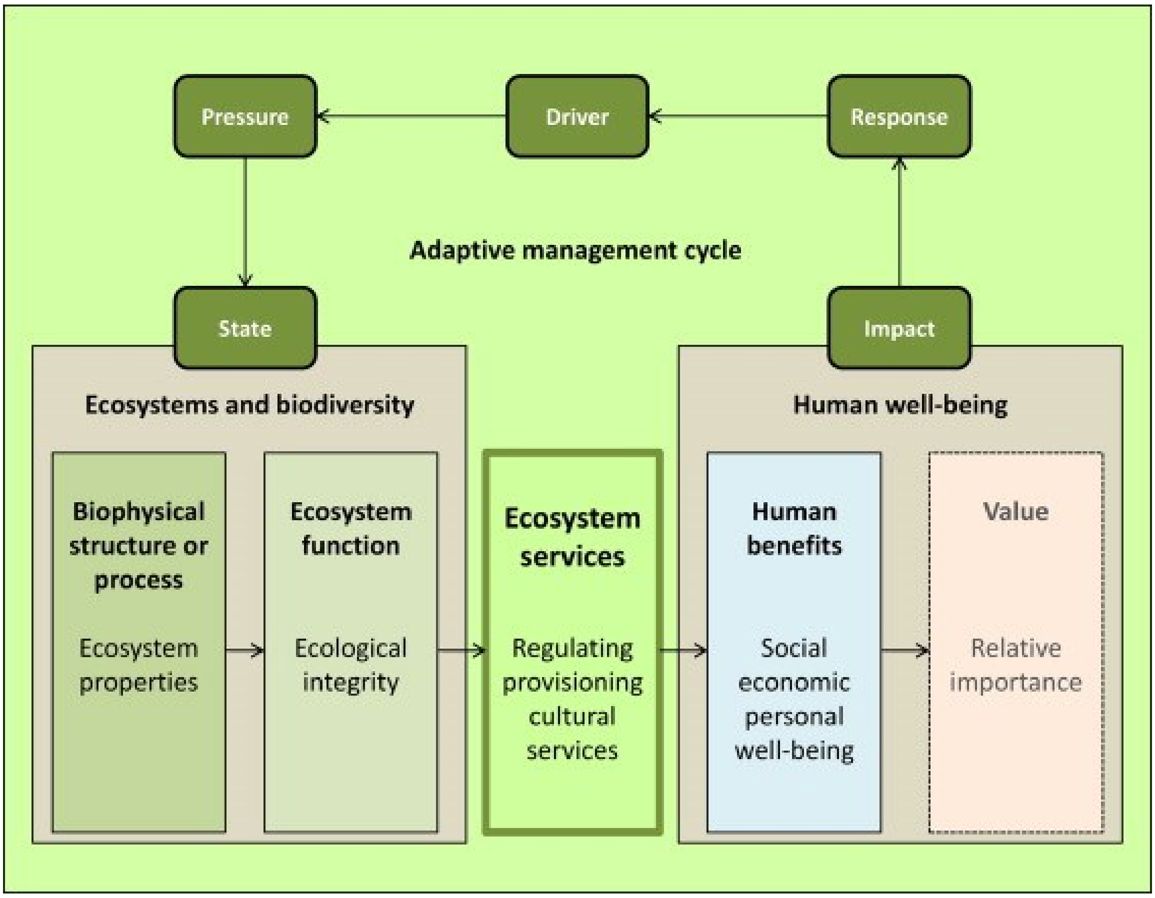

Figure 1-1: The 'ecosystem service cascade' as it is used within the BACOSA project and in this thesis (from Kandziora et al. 2013, p. 56)

1.3 Regulating services in coastal wetlands

In order to understand regulating services provided by coastal systems, the biophysical structure, processes and functions in these ecosystems have to be examined, because the ecological integrity strongly impacts regulating services (Burkhard et al. 2012). A peculiarity of coastal systems is that their ecosystem health and ability to provide services for human well-being is directly connected to the activities in the adjacent marine and terrestrial ecosystems – two systems that differ greatly from each other (Agardy et al. 2005). Anthropogenic activities in the hinterland such as agriculture can affect wetlands severely as they are 'critical transition zones' (Levin et al. 2001).

Salt marshes – to which coastal *Phragmites* wetlands belong – provide a particularly wide range of ecosystem services (Agardy et al. 2005; Barbier et al. 2011). The list of all proposed indicators by Kandziora et al. (2013) that represent regulating services has been complemented in Table 1-1 by columns describing the relevance for coastal wetlands including examples. The focus of this thesis will be on water purification, nutrient regulation and erosion control, since coastal wetlands are particularly important for the regulation of fluxes of nutrients, pollutants and particles (Levin et al. 2001). Water purification refers to the capacity of an ecosystem to filter and purify water from unwanted components, nutrient regulation means the capacity of an ecosystem to cycle nutrients such as phosphorus, and erosion control (also sometime listed as sediment or soil retention) is the capacity to prevent erosion processes and depends mainly on the structure of the vegetation cover and root network (de Groot et al. 2002; Costanza et al. 2008; Kandziora et al. 2013).

Coastal wetlands protect the hinterland from storm surges and coastal erosion by attenuating wave and wind energy (Davy et al. 2009; Morgan et al. 2009; Duarte et al. 2013). While the monetary value of coastal wetlands for general erosion regulation is still unknown (Barbier et al. 2011), Costanza et al. (2008) estimated that salt marshes reduced the costs of hurricane damage (hazard protection) in the USA by $8236 per hectare and year. Monetarization of the 'water purification service' revealed that marsh swamps in southern Louisiana (USA) economized $314-600 per hectare over traditional waste treatment (Breaux et al. 1995). However, more case studies that value regulating ecosystem services of coastal wetlands are often requested in the scientific literature with the argument that the failure to implement ecosystem service values in coastal management plans supports coastal degradation (Barbier et al. 2011).

Table 1-1: Regulating ecosystem services and definitions by Kandziora et al. 2013, complemented by a column giving examples for regulating services in coastal wetlands and their relevance (+/-/o scheme). Water purification, nutrient regulation and erosion regulation are highlighted in bold characters.

Regulating service	Definition	Relevance in coastal wetlands	Example of regulating service in coastal wetlands
Global climate regulation	Long-term storage of greenhouse gases in ecosystems	+	e.g. storage of 'blue carbon' in wetland sediments, McLeod et al. 2011
Local climate regulation	Changes in local climate components like wind, precipitation, temperature, radiation due to ecosystem properties.	+	e.g. wind attenuation, Karstens et al. 2015b
Air quality regulation	Capturing/filtering of dust, chemicals and gases.	o	
Water flow regulation	Maintaining of water cycle features (e.g. water storage and buffer, natural drainage, irrigation and drought prevention).	+	e.g. flood control, Mitsch and Gosselink 2000
Water purification	The capacity of an ecosystem to purify water, e.g. from sediments, pesticides, disease-causing microbes and pathogens.	+	e.g. buffer for pharmaceuticals, Andreu et al. 2016; buffer for heavy metals, Karstens et al. 2016a
Nutrient regulation	The capacity of an ecosystem to recycle nutrients, e.g. N, P.	+	e.g. nitrogen retention, Mitsch et al. 2005; phosphorus retention, Karstens et al. 2015a
Erosion regulation	Soil retention and the capacity to prevent and mitigate soil erosion and landslides.	+	e.g. sediment trapping to counteract erosion, Kathiresan 2003; soil stabilization, Barbier et al. 2011
Natural hazard protection	Protection and mitigation of floods, storms (hurricanes, typhoons . . .), fires and avalanches.	+	e.g. hurricane protection, Costanza et al. 2008; protection from storm surges and small tsunami waves, Gedan et al. 2011

1.3.1 Processes or services?

Kandziora et al. (2013, page 58) define regulating services as “benefits people obtain due to the regulation of natural processes such as water purification or erosion control”. Whether regulating services are final ecosystem services or only processes that lead to a service divides the research community. Boyd and Banzhaf (2007) argue that for example water purification is a function of specific land cover type which helps to achieve ‘clean water’. In this case clean water would be the service and water purification a process. Wallace (2007) has a similar approach where nutrient regulation or sediment retention are classified into ecosystem processes and food, water, energy etc. are ecosystem services. Nutrient cycling in wetlands is important to prevent eutrophication and oxygen shortages, and thus stress on aquatic species population – but Boys and Banzhaf (2007) regard it as a process and the final service are the aquatic populations. For practitioners it is much more complicated to measure and value processes than final outcomes (Boyd and Banzhaf 2007). In this thesis water purification, erosion control and nutrient regulation are defined as ecosystem services, in accordance with the classification systems of de Groot et al. (2002), Millennium Ecosystem Assessment (2005), Costanza et al. (2008) or Kandziora et al. (2013). Irrespective of the classification scheme applied, it should be kept in mind that even if nutrient regulation, erosion control or water purification would not be called 'services', they would still be valuable and positively affecting human welfare (Boyd and Banzhaf 2007; Fisher and Turner 2008).

1.3.2 Complexity and interrelationships between regulating services

Another salient feature of regulating services is that many of them are interrelated and interdependent (Kandziora et al. 2013). For example, erosion in coastal wetlands impacts climate regulation due to the release of 'blue carbon' from the sediments. Thus, the assessment of climate regulation is incomplete when erosion regulation is not taken into consideration. Theuerkauf et al. (2015) showed that carbon export from eroding saltmarsh fringes can exceed carbon accumulation. Saltmarsh carbon budgets that exclude such erosion processes, therefore, are incomplete. The same phenomena can be observed for nutrient regulation, where it matters whether a coastal wetland is expanding or eroding. In chapter 5.1 interrelations of regulating services in coastal wetlands will be discussed in detail.

However, linkages and interrelationships do not only occur between ecosystem services within wetlands, but also between the neighboring systems: terrestrial – wetland – marine.

Choices made on land or at sea can directly impact regulating services in coastal wetlands. Over 70% of the pollutants in coastal systems come from land-based sources and enter the coastal areas via rivers, from run-off or through atmospheric deposition (Agardy et al. 2005). According to Diaz and Rosenberg (2008, page 926) the key factor to tackle eutrophication and hypoxia "will be to keep fertilizers on the land and out of the sea. For agricultural systems in general, methods need to be developed that close the nutrient cycle from soil to crop and back to agricultural soil." As another example diversion of water for agriculture impedes the flow of fresh water and sediments to wetlands, leading to marsh subsidence in many regions (Cahoon et al. 1995; Vörösmarty and Meyback 1999). Still, the linkages between the neighboring ecosystems are often overlooked or not regarded as they are considered as too complex.

1.4 Spatial embedding: Coastal wetlands at the southern Baltic Sea

The Baltic Sea covers an area of over 400,000 km² and is a shallow sea with an average depth of 52 m (Schiewer and Schernewski 2004). It is a brackish sea with decreasing salinities from west to east and tides do not exist (Hupfer 2010). The total area draining to the Baltic Sea is approximately four times larger than the sea and accommodates about 85 million people, whereof 15 million live along a 10 km wide stripe of the coast (Schiewer and Schernewski 2004). The economic importance of the Baltic Sea region has increased markedly since the fall of the 'Iron Curtain' and the accession to the EU of several eastern countries, and consequently anthropogenic pressure on the Baltic Sea are still increasing (Schiewer and Schernewski 2004). Eutrophication and hypoxia remain serious environmental challenges in the Baltic Sea (Zillén et al. 2008; Voss et al. 2011). Also in the coastal zones of the Baltic Sea hypoxia has received growing attention. Conley et al. (2011) identified 115 sites along the Baltic Sea coast that have been experiencing hypoxia at least once since 1955. Agriculture in the Baltic Sea drainage basin is the main cause for nutrient inputs. Increased loads of nitrogen and phosphorus from agriculture to the Baltic Sea result from the separation of crop and animal production. The specialization of agriculture has resulted in intensive livestock farming which produces a surplus of nutrients and simultaneously a more specialized form of crop cultivation based on chemical fertilizers and pesticides (Granstedt 2000).

As a result of the last glacial period and subsequent postglacial processes the coast of the Baltic Sea is fragmented with many shallow water bodies in form of lagoons and fjords (Hupfer 2010). Shallow, semi-enclosed coastal systems are particularly prone to eutrophication and hypoxia due to reduced water exchange with the open sea and human pressures such as intensive agriculture in the hinterland or sewage discharges (Kautsky and Kautsky 2000; Newton et al. 2013). Nutrient and pollutant loads from point-like and diffuse sources have to pass these coastal waters with their wetlands before they reach the open sea (Schiewer and Schernewski 2004). The Darss-Zingst Bodden Chain with its vast wetlands, the study area of this thesis, is such a lagoon system with the potential to protect the open Baltic Sea from pollution. Beyond the seaward edge of coastal wetlands, the Darss-Zingst Bodden Chain has been intensively researched including a variety of studies on nutrient regulation (e.g. Nausch and Schlungbaum 1991; Schiewer 1998; Selig et al. 2006; Görs et al. 2007; Berthold and Schumann 2016). The same applies to the hinterland and terrestrial wetlands where ecosystem services have been investigated (e.g. Kruse et al. 2015; Stoll-Kleemann 2015; Kliesch et al. 2016). However, the coastal wetlands of the Darss-Zingst Bodden Chain that connect the terrestrial and aquatic ecosystems have been neglected so far (one exception: Voigtland 1983).

Coastal wetlands along the Baltic Sea are very heterogeneous with a wide range of species due to strong gradients in salinity, climate or water level fluctuations (Dijkema 1990). Wetland area along the Baltic Sea coasts is still unknown, even on the European scale data about coastal wetland distribution are only rough estimates (Buczko et al. 2016). Many Baltic coastal wetlands had traditionally been used for grazing, hay-making or were harvested for construction material. Since the decline of such activities due to economic reasons or nature protection goals, reed has replaced halophytes in many wetlands (Dijkema 1990; Köbbing et al. 2013). Consequently large coastal areas are dominated by *Phragmites australis* (Cav.) Trin. ex Steud. (Dijkema, 1990). While nutrient regulation, water purification or erosion regulation are fairly well studied in seagrass meadows, mangroves or fresh water wetlands, (e.g. Bowden 1987; Moore et al. 1994, Reddy et al. 1999; Moberg and Rönbeck 2003; Holmer et al. 2006; Deborde et al. 2008; Ewel et al. 2008; Delgard et al. 2013), very few studies have addressed these regulating services in coastal wetlands colonized by *Phragmites*.

1.5 *Phragmites*: A bio-engineer of its own environment

Phragmites australis (Cav.) Trin. ex Steud. is a perennial grass (family *Poaceae*) which can grow up to 4 m and overtops most other emergent macrophytes in wetlands such as *Typha*, *Scripus* or *Spartina* (Cronk et al. 2001). *Phragmites* has the potential to supply various ecosystem services such as sequestering nutrient or heavy metals, stabilizing soils, buffering wind and wave energy or providing habitat in urban or industrial areas where many plants would not thrive otherwise (Kiviat 2013; see Figure 1-2 for habitat in an urban coastal area). Reed stems are flexible and their 'bending stiffness' (Ostendorp 1995) enables the plant to cause high drag forces and attenuate waves (Möller et al. 2011). Furthermore *Phragmites* provides valuable products for direct human use such as construction material for roofing or fodder in summer (Köbbing et al. 2013). However, in order to provide the desired ecosystem services as effectively as possible *Phragmites* wetlands may need active interventions and management such as thinning, fragmentation or containment (Kiviat 2013).

Figure 1-2: *Phragmites australis* growing along and on top of an artificial stone dyke in an urbanized area of the Baltic Sea island Fehmarn, Germany.

Although *Phragmites* is principally a freshwater plant, it is well adapted to brackish water conditions because it is able to cope with a wide range of salinities (Karsten et al. 2003; Meriste et al. 2012; Altartouri et al. 2014). Along the coast of the Darss-Zingst Bodden Chain and other Bodden coasts of the southern Baltic Sea *Phragmites* is the dominant plant species (Voigtland 1983). During growth in spring and early summer, large amounts of nutrients are incorporated in the above-ground biomass. In autumn, the majority of nutrients is transported back into the rhizomes and stored below-ground during winter. The large rhizome network of the plant supports also shoreline stabilization (Ostendorp 1993). Another important feature of *Phragmites* plants are air conducting channels (aerenchyma) that allow the transport of oxygen from the upper parts of the plant into the root zone (Rodewald-Rudescu 1974; Haraguchi 2012). The resulting aeration of the sediment in the vicinity of the roots (rhizosphere) can lead to the formation of oxidized forms of iron and manganese

(Sundby et al. 2003). In the past decades many European wetlands have suffered from reed die-back and eutrophication seemed to be the key controlling factor (van der Putten 1997). However, the consequences of eutrophication on reed die-back were indirect: the formation of toxic byproducts of decomposing litter in anoxic environment impacted *Phragmites* wetland vitality (van der Putten 1997). High loadings of anaerobically digested sludge in very nutrient rich sites inhibit reed growth severely and lead to typical stress symptoms such as stunted growth or chlorotic leaves (Romero et al. 1999). In brackish *Phragmites* wetlands sulfide as a principal toxin may be responsible for reed stress (van der Putten 1997). Persistent concentrations of hydrogen sulfide above 5 mg/L can destroy reed stands within one year (Rodewald-Rudescu 1974).

1.6 Objectives and structure of this thesis

The objectives of this thesis are to analyze regulating ecosystem services in coastal wetlands: (1) how they are supplied, (2) how they are interrelated, and (3) how physical, biological and chemical processes impact regulating services. Here, the focus is on nutrient regulation, water purification and erosion control (see Table 1-1). In chapter 2 the often-mentioned role of wetlands as ‘pollutant buffers’ for macronutrients and heavy metals from the adjacent hinterland is examined. In chapter 3 we take a closer look at one particular macronutrient: phosphorus. We will see that coastal wetlands cannot function limitless as nutrient buffers and that the phosphorus cycle in these systems is very complex and dependent on several aspects, such as hydrodynamics and oxygen flows. In chapter 4 erosion regulation is the centerpiece and the question how ecosystems properties such as litter mass, organic matter content or water flow impact dynamics of surface elevation and microtopography changes. Each chapter has its own particular hypothesis and specific research questions, but they are united in their aim to portray, analyze and discuss regulating services supplied by coastal wetlands.

2. Impact of adjacent land use on coastal wetland sediments

Karstens, S., Buczko, U., Jurasinski, G., Peticzka, R., Glatzel, S., 2016. Impact of adjacent land use on coastal wetland sediments. Science of the Total Environment 550, 337-348.

Abstract

Coastal wetlands link terrestrial with marine ecosystems and are influenced from both land and sea. Therefore, they are ecotones with strong biogeochemical gradients. We analyzed sediment characteristics including macronutrients (C, N, P, K, Mg, Ca, S) and heavy metals (Mn, Fe, Cu, Zn, Al, Co, Cr, Ni) of two coastal wetlands dominated by *Phragmites australis* at the Darss-Zingst Bodden Chain, a lagoon system at the southern Baltic Sea, to identify the impact of adjacent land use and to distinguish between influences from land or sea. In the wetland directly adjacent to cropland (study site Dabitz) heavy metal concentrations were significantly elevated. Fertilizer application led to heavy metal accumulation in the sediments of the adjacent wetland zones. In contrast, at the other study site (Michaelsdorf), where the hinterland has been used as pasture, heavy metal concentrations were low. While the amount of macronutrients was also influenced by vegetation characteristics (e.g. carbon) or water chemistry (e.g. sulfate), the accumulation of heavy metals is regarded as purely anthropogenic influence. A principal component analysis (PCA) based on the sediment data showed that the wetland fringes of the two study sites are not distinguishable, neither in their macronutrient status nor in their concentrations of heavy metals, whereas the interior zones exhibit large differences in terms of heavy metal concentrations. This suggests that seaside influences are minor compared to influences from land. Altogether, heavy metal concentrations were still below national precautionary and action values. However, if we regard the macronutrient and heavy metal concentrations in the wetland fringes as the

natural background values, an accumulation of trace elements from agricultural production in the hinterland is apparent. Thus, coastal wetlands bordering croplands may function as effective pollutant buffers today, but the future development has to be monitored closely to avoid breakthroughs due to exceeded carrying capacities.

2.1 Introduction

Coastal wetlands are open structures with strong interactions along the land-water interface, linking terrestrial with marine ecosystems (Andreu et al. 2016), and represent ecotones in the core sense of the term. Coastal wetlands can provide a variety of ecosystem services that are fundamental for physical processes and biogeochemical cycling including sediment retention and protection against coastal erosion, habitats for fish or birds, raw material provisioning, pollutant buffering and nutrient regulation (Duarte et al. 2013; Karstens and Lukas 2014; Perillo et al. 2009; Reddy et al. 1999). The relative importance of these services for humans often depends on management decisions and the specific location of a coastal wetland.

Coastal wetland sediments are a mixture of material from various sources including terrestrial input via surface or groundwater flows, erosion of near-by coastal sites or adjacent land, atmospheric depositions or sedimentation of marine particles (Abi-Ghanem et al. 2009; Bao et al. 2015). Macronutrients may have natural origins (wetland vegetation or sea influence), whereas the accumulation of heavy metals in sediments can be regarded as a ‘finger print of human activity’ (Andreu et al. 2016). Heavy metal concentrations in the landward zones of wetlands may be largely driven by input from adjacent parts of land, especially when these are croplands because erosion tends to be stronger compared to permanent grasslands or forests (Pimentel and Kounang 1998). In aquatic environments, heavy metals can be a major threat due to their persistence, prevalence, potential toxicity and bioavailability (Boyd 2010; Marchand et al. 2011). Identifying the sources of heavy metals and evaluating of the influence of anthropogenic activities is difficult (Bayen 2012;

Wang et al. 2014). Agriculture is often mentioned as an important input source and fertilizers applied in agroecosystems can be a major source of heavy metals (Jiao et al. 2012). Some heavy metals included in fertilizers are essential for plant growth but toxic above critical concentrations (e.g. copper, zinc, manganese, iron), whereas others are always contaminants with no benefits for plant growth (e.g. chromium) (He et al. 2005). Contamination assessments and monitoring of heavy metal concentrations in coastal wetland sediments are essential, especially when anthropogenic pressures are on the rise (Andreu et al. 2016; Pascual-Aguilar et al. 2015), and coastal wetlands are 'the last line of defense' before pollutants reach adjacent waters.

In this study we analyze the sediment composition of different coastal wetlands including all macronutrients (C, N, P, K, Mg, Ca, S) as well as eight heavy metals (Mn, Fe, Cu, Zn, Al, Co, Cr, Ni) to answer the question, whether sediment characteristics of coastal wetlands dominated by *Phragmites australis* differ depending on adjacent land use. Two typical and representative sites with respect to land use, topography and hydraulic conditions in the southern Baltic Sea region were chosen. The wetlands were further subdivided into three zones in order to distinguish between influences from land or sea. We address how adjacent land use (pasture vs. cropland) impacts sediment composition and how influences from the land or sea dominate across the wetland zones. By doing this, we aim to differentiate between 'natural' and 'anthropogenic' influences. While the amount of macronutrients in sediments might be either influenced by vegetation characteristics (e.g. carbon) or water chemistry (e.g. sulfur), the accumulation of heavy metals in sediments is regarded as anthropogenic influence.

2.2 Research approach and methods

2.2.1 Study sites

The Darss-Zingst Bodden Chain is a lagoon system with four sub-basins at the southern Baltic Sea in Germany (Figure 2-1). It is a shallow water body with a mean water depth of 2 m. The only connection to the open Baltic Sea is a narrow outlet called Gellenstrom in the northeast (Schumann et al. 2006). Tides do not exist and water exchange with the Baltic Sea is induced meteorologically with inflow situations under strong and persistent northeasterly winds (Selig et al. 2007).

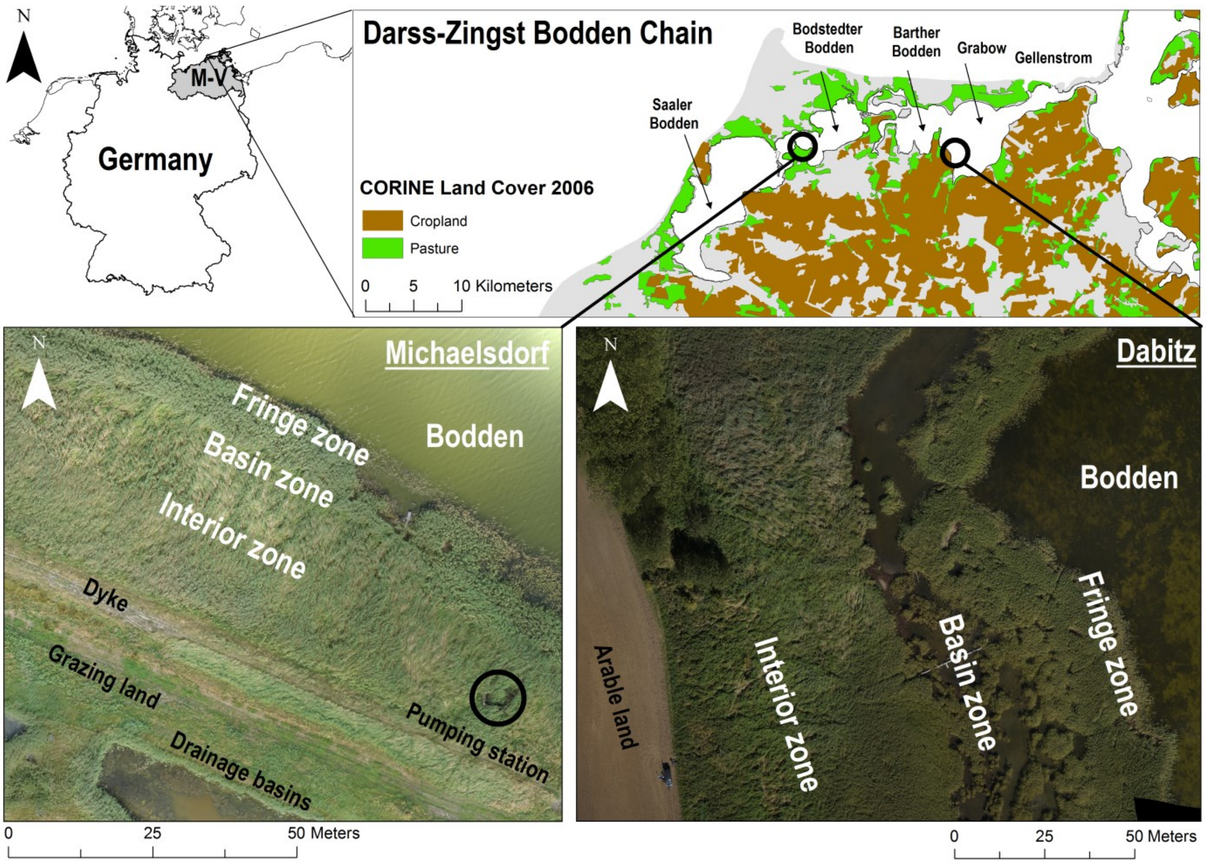

Figure 2-1: Study sites Michaelsdorf and Dabitz at the southern coast of the Darss-Zingst Bodden Chain. Areas used as pastures or croplands were extracted from the CORINE Land Cover Dataset of 2006. Aerial images of the study sites were taken in August 2015 with an unmanned aerial system flight.

The southern hinterland of the Darss-Zingst Bodden Chain is predominantly used for agriculture (Figure 2-1). Lowland areas are dyked and used as grassland, whereas areas with a more pronounced topography are usually not dyked and used as cropland. Consequently two different types of coastal wetlands can be differentiated: coastal wetlands bordering arable fields and coastal wetlands confined by a dyke landwards with pastures in the hinterland. At the coasts of the Darss-Zingst Bodden chain both wetland types are dominated by *Phragmites australis* (Cav) Trin. Ex Streudel (common reed).

In this study, one site of each type was investigated: the *Phragmites* wetland at Dabitz borders directly cropland, whereas the wetland at Michaelsdorf is 'squeezed' behind a dyke and the hinterland used as pasture for sheep (Figure 2-1). The distance between the study sites is about 15 km and consequently climatic conditions do not differ. Both sites are situated at the southern coast of the Bodden system. The sediment textures of the adjacent Bodden sediments are fine to medium sands (Bitschofsky et al. 2015). A detailed description of vegetation, water and sediment characteristics of the wetlands follows in the results section 2.3.1.

Tidal salt marshes are often divided into low, mid- and high marsh according to the influence of the tidal range (Packham and Willis 1997). Since there are no tides in the Darss-Zingst Bodden Chain the wetlands are subdivided into interior, basin and fringe zone, based on water level and hydraulic energy (Figure 2-1). This clasification was already proposed in the 1970ies for mangrove forests (Lugo and Snedaker 1974), and proved to be functional. Water level and hydraulic energy are higher in the fringe than in the basin zone, whereas the interior zone is rarely flooded (see Brinson 1993; Karstens et al. 2015a; Lugo et al. 1988).

2.2.2 Analysis of land use

Aerial images (1953-2013) provided by the government office for geoinformation, surveying and cadaster MV were used to analyze land use changes at the two study sites since the 1950ies. Semi-structured, face-to-face interviews were conducted with the farming company that manages the cropland at Dabitz and with the shepherds responsible for sheep grazing at

Michaelsdorf to improve our understanding of past and present land use activities. Questions were grouped into categories, but the order of questions was not pre-defined, and further questions could be added during the interviews to enhance flexibility and allow in-depth information (Hollway and Jefferson 2000; Hellferich 2009).

2.2.3 Sampling and analysis of sediment, water and vegetation

A total of 60 sediment samples at Dabitz and 48 samples at Michaelsdorf were collected between March 2014 and January 2015 in the three wetland zones using a stainless steel corer with 7 cm diameter (*Hydrobios*, *Kiel, Germany*). Sediment samples were separated into depth slices of 0-2 cm and 2-10 cm. We chose this subdivision, because the upper 2 cm best reflect influences of modern land use, while the lower sediment layer of 2-10 cm reflects the last decades. Sedimentation rates outside the wetlands range between 0.9-1.8 mm per year at the Grabow (Meyer et al. 2016, submitted) and 0.2-0.6 mm per year at the Barther Bodden (Müller 2002). A subset of sediment samples was dried at 105°C for 24 hours, sieved to <2 mm and subsequently grinded to prepare for further analysis. Sediment organic matter was determined gravimetrically by loss-on-ignition (LOI) in a muffle furnace at 550°C for 4h. Total carbon, nitrogen and sulfur contents were quantified by combustion in a CNS Analyzer (*Vario Max, Elementar, Germany*). A multi element analysis including total phosphorus, potassium, calcium, magnesium, iron, aluminum, manganese, zinc, copper, chromium, cobalt and nickel was carried out by inductively coupled plasma optical emission spectrometry after aqua regia digestion (*Optima 5300 DV, Perkin-Elmer, USA*) (Wuenscher et al. 2015).

Vegetation characteristics including stem density and aboveground biomass were recorded bimonthly between March 2014 and January 2015. Biomass sampling took place in 20x20 cm squares in each zone. In addition, in January 2015 we collected in each wetland zone litter samples by harvesting all on-ground litter in three 20×20 cm squares. Aboveground plant and litter material was dried at 60°C for 48 hours to weigh dry biomass

(Schieferstein 1997). Total carbon and nitrogen concentrations of the litter samples were quantified by combustion in a CNS Analyzer (*Vario Max, Elementar, Germany*). Total phosphorus concentrations were analyzed after aqua regia digestion in an ICP-MS (*Optima 5300 DV, Perkin-Elmer, USA*).

A total of 17 water samples at Dabitz and 10 water samples at Michaelsdorf were taken in the fringe zones between November 2013 and November 2014. Sulfate was measured photometrically after precipitation with barium. Salinity was measured in situ above the sediment bed using *Hach Lange* sensors. Suspended sediment concentrations were determined by filtration over pre-weighted GF/F filter (e.g. Deborde et al. 2007).

2.2.4 Data processing

All data analysis was performed and all figures were prepared using the open-source statistical software R (version 3.2.1). Pearson correlation coefficients were calculated for all sediment parameters. Beforehand, scatter plot matrices were used to identify visually if other correlations than linear correlations occur. Principal component analysis (PCA) was applied to investigate if we can characterize groups of individual samples. Correlated sediment variables were combined into factors and sources related to the factors were then identified (e.g. Bai et al. 2011; Bao et al. 2015). The R package FactoMineR was used for the PCA (http://factominer.free.fr/index.html).

2.3 Results

2.3.1 Wetland characteristics and land use

The study site Dabitz adjoins fields that have been used as cropland at least since the 1950ies: already the first available aerial image from 1953 shows that arable fields directly border the *Phragmites* wetland. The farming company that currently cultivates the field confirmed that arable farming was practiced since about 1945. Nowadays a crop rotation system including oil seed rape, wheat and barley is applied. Harvest takes place during summer and tillage in early autumn. The field lays fallow after harvest for 6-8 weeks. Aerial images show that this usually occurs around August. Herbicides and fungicides are applied twice a year, in autumn and spring. Fertilizers are only applied in spring between March and June depending on the crop. Since 2000 mineral phosphorus fertilizers are no longer applied, instead cow manure is used to cover the phosphorus and potassium demands of plants. Since the amount of applied manure does not satisfy the crop demand for nitrogen, mineral nitrogen fertilizers are applied additionally. Due to the relatively high elevation of the field and the sloping relief towards the coast, the field is not affected by flooding. During heavy rain events, erosion rills may form, giving evidence of periodic solid matter transport into the wetland. The soil type of the cropland is loamy sand. Soil pH varied with topography and was lowest with 6.7 in the sinks adjacent to the coastal wetland. Liming is done every 4-5 years.

The topography at the study site Michaelsdorf differs significantly from Dabitz and does not allow arable land use. The land is flat and the ground water gradient goes from sea to land (see Kliesch et al. 2016). Dyke construction took place in the 1970. The hinterland is intensively drained and used for sheep grazing since 1990. Besides sheep manure no additional fertilizers or other agricultural products are applied. The sheep graze between April and October and the herd of sheep rotates once an area is grazed (around 15 sheep per hectare). During the other six months of the year grazing is not allowed due to nature protection restrictions and the area is not used agriculturally during that time of the year.

Although both study sites are situated at the southern site of the lagoon system, they are influenced by water masses of two different sub-basins (Figure 2-1). Suspended sediment concentrations were comparable at both fringe sites (Michaelsdorf: 65±51 mg/L; Dabitz: 64±55 mg/L). Due to the higher influence of the Baltic Sea at the outermost Bodden salinity and sulfate concentrations were higher at Dabitz with 10.8±1.5 PSU and 497±134 mg/L sulfate compared to 8±1.4 PSU and 266±141 mg/L at Michaelsdorf.

At both wetlands, the vegetation is dominated by *Phragmites australis*. Biomass, stem density and litter mass were comparable between the interior and basin zones of the two study sites (Table 2-1). Maximum aboveground biomass occurred in late summer and was 8.2 kg/m² in the interior zone of Michaelsdorf and 8.4 kg/m² at Dabitz. For the fringe zones of the two study sites vegetation characteristics differed. In the fringe zone at Michaelsdorf stem density was significantly lower than at Dabitz and aboveground biomass remained below 2 kg/m² during the vegetation maximum. Almost no *Phragmites* stems survived the winter season and when litter sampling took place in January 2015, no litter was left on the sediment (Table 2-1).

Table 2-1: Mean aboveground biomass and stem density throughout the year ± standard deviation (n=21). Litter mass and CNP concentrations in January 2015 (n=9 for Dabitz and n=6 for Michaelsdorf).

	Dabitz			Michaelsdorf		
	Interior zone	Basin zone	Fringe zone	Interior zone	Basin zone	Fringe zone
Biomass [kg/m²]	3.9±3.2	3.4±2.8	2.1±1.3	3.8±2.9	2.6±2.4	0.7±0.7
Stem density [stems/m²]	375±154	407±119	353±87	408±200	422±151	184±82
Litter mass [g/m²]	853±23	553±135	394±233	1080±104	353±171	0
Carbon in litter [%]	42.4±0.3	43.3±0.5	38.8±2	44.6±0.2	44.8±0.2	-
Nitrogen in litter [%]	1.6±0.2	1.2±0.3	1.3±0.1	1.9±0.1	1.3±0.1	-
Phosphorus in litter [%]	0.011±0.001	0.009±0.004	0.008±0.001	0.013±0.001	0.005±0.001	-

2.3.2 Sediment properties in different wetland zones

The characteristics of collected sediment samples are summarized in Table 2-2. Sediment texture was comparable at the wetland interior and basin zone of both study sites with sandy loams and became coarser toward the fringe zones with fine to medium sands. The highest carbon and nitrogen concentrations occurred in the surface sediments of the interior zone at Michaelsdorf, where also litter mass was highest (Table 2-1). Sulfur concentrations were highest in the lower sediments of the basin zone at Dabitz. The other macronutrient concentrations were always higher in the surface sediments of the interior or basin zone at Dabitz compared to the fringe zones of the two study sites (Figure 2-2).

Mean heavy metal concentrations were highest in the interior wetland zone at Dabitz compared to all other zones of the two study sites except iron which is slightly higher in the surface sediments of the basin zone. The differences between the interior zones at Dabitz and Michaelsdorf were significant for all heavy metals except copper and zinc (Figure 2-3).

Table 2-2: Mean dry bulk density, organic matter (LOI: loss-on-ignition) and element concentrations ± standard deviations (n=128).

	Dabitz						Michaelsdorf					
	Interior 0-2 cm	Interior 2-10 cm	Basin 0-2 cm	Basin 2-10 cm	Fringe 0-2 cm	Fringe 2-10 cm	Interior 0-2 cm	Interior 2-10 cm	Basin 0-2 cm	Basin 2-10 cm	Fringe 0-2 cm	Fringe 2-10 cm
Density [g/cm³]	0.63 ±0.37	0.85 ±0.24	0.16 ±0.08	0.25 ±0.22	0.59 ±0.51	1.13 ±0.29	0.14 ±0.04	0.53 ±0.14	0.39 ±0.17	0.59 ±0.14	0.83 ±0.46	0.75 ±0.46
LOI [%]	21.05 ±14.12	12.04 ±7.11	48.34 ±13.28	36.82 ±14.42	9.86 ±7.36	1.29 ±0.75	65.48 ±10.8	8.31 ±3.95	23.12 ±4.88	22.1 ±2.46	4.82 ±4.57	9.18 ±16.67
C [%]	10.9 ±6.55	5.86 ±3.14	24.29 ±6.44	18.91 ±7.02	4.87 ±3.64	0.54 ±0.25	31.07 ±7.33	4.98 ±3.24	11.74 ±2.61	10.7 ±3.13	2.23 ±2.1	3.77 ±5.55
N [%]	0.97 ±0.52	0.58 ±0.26	1.84 ±0.44	1.36 ±0.48	0.45 ±0.29	0.07 ±0.04	2.49 ±0.5	0.42 ±0.29	0.99 ±0.2	0.89 ±0.23	0.27 ±0.2	0.45 ±0.52
S [%]	0.2 ±0.14	0.13 ±0.08	0.84 ±0.11	1.21 ±0.73	0.55 ±0.41	0.09 ±0.04	0.65 ±0.11	0.15 ±0.08	0.27 ±0.09	0.29 ±0.11	0.41 ±0.63	0.32 ±0.45
P [mg/kg]	1309 ±548	1187 ±568	1975 ±710	794 ±222	435 ±165	201 ±35	1422 ±245	274 ±119	765 ±154	480 ±114	151 ±49	118 ±71
K [mg/kg]	3191 ±1323	3216 ±1716	4262 ±974	3849 ±1216	1519 ±845	360 ±110	1331 ±260	559 ±132	1767 ±380	1893 ±423	412 ±216	422 ±279
Ca [mg/kg]	5963 ±2623	4489 ±1753	7194 ±1895	4817 ±1280	3732 ±2357	1419 ±518	7225 ±1720	1345 ±554	5104 ±1032	3376 ±589	3136 ±2607	2507 ±2685
Mg [mg/kg]	3717 ±1844	3569 ±2129	5992 ±841	5184 ±1573	2436 ±1316	603 ±137	3725 ±806	735 ±245	2717 ±564	2245 ±549	796 ±711	1088 ±1370
Mn [mg/kg]	1062 ±685	631 ±361	406 ±388	118 ±51	95 ±46	29 ±8	36 ±10	11 ±3	140 ±88	33 ±7	39 ±34	24 ±14
Fe [mg/kg]	18465 ±8319	21746 ±12024	26917 ±17640	12725 ±5131	6831 ±3646	1846 ±511	6849 ±1279	1601 ±383	17552 ±8711	5727 ±1153	4942 ±7175	3077 ±4102
Cu [mg/kg]	18 ±7	16 ±6	16 ±2	18 ±5	11 ±4	5 ±1	16 ±3	7 ±2	10 ±2	9 ±2	4 ±2	4 ±1
Zn [mg/kg]	97 ±48	74 ±38	70 ±18	55 ±17	45 ±18	20 ±8	69 ±20	17 ±11	46 ±21	35 ±16	15 ±9	16 ±9
Al [mg/kg]	11368 ±4227	14151 ±7817	6302 ±540	8123 ±2522	3208 ±1485	1159 ±349	2226 ±714	1596 ±323	4775 ±1139	5870 ±1339	1226 ±533	1492 ±786
Co [mg/kg]	7 ±3.1	6.8 ±3.1	4.5 ±2.3	3.5 ±1.1	2.3 ±1	0.9 ±0.3	1.2 ±0.2	0.6 ±0.1	2.1 ±0.5	1.4 ±0.3	1.4 ±1.3	1.4 ±1.4
Cr [mg/kg]	17.7 ±5.9	20.4 ±9.5	13.2 ±1.3	16.6 ±5.1	7.2 ±3.5	2.4 ±0.6	5.6 ±1.7	3.9 ±1	10.4 ±2.4	11.7 ±2.5	2.4 ±0.9	2.6 ±1.2
Ni [mg/kg]	13.1 ±5.6	12.7 ±6.2	12.6 ±1.6	12.4 ±3.7	7.9 ±5	2.4 ±0.8	5.3 ±0.9	2.3 ±0.8	7.2 ±1.2	6.3 ±1.1	2.8 ±2.3	3 ±3.6

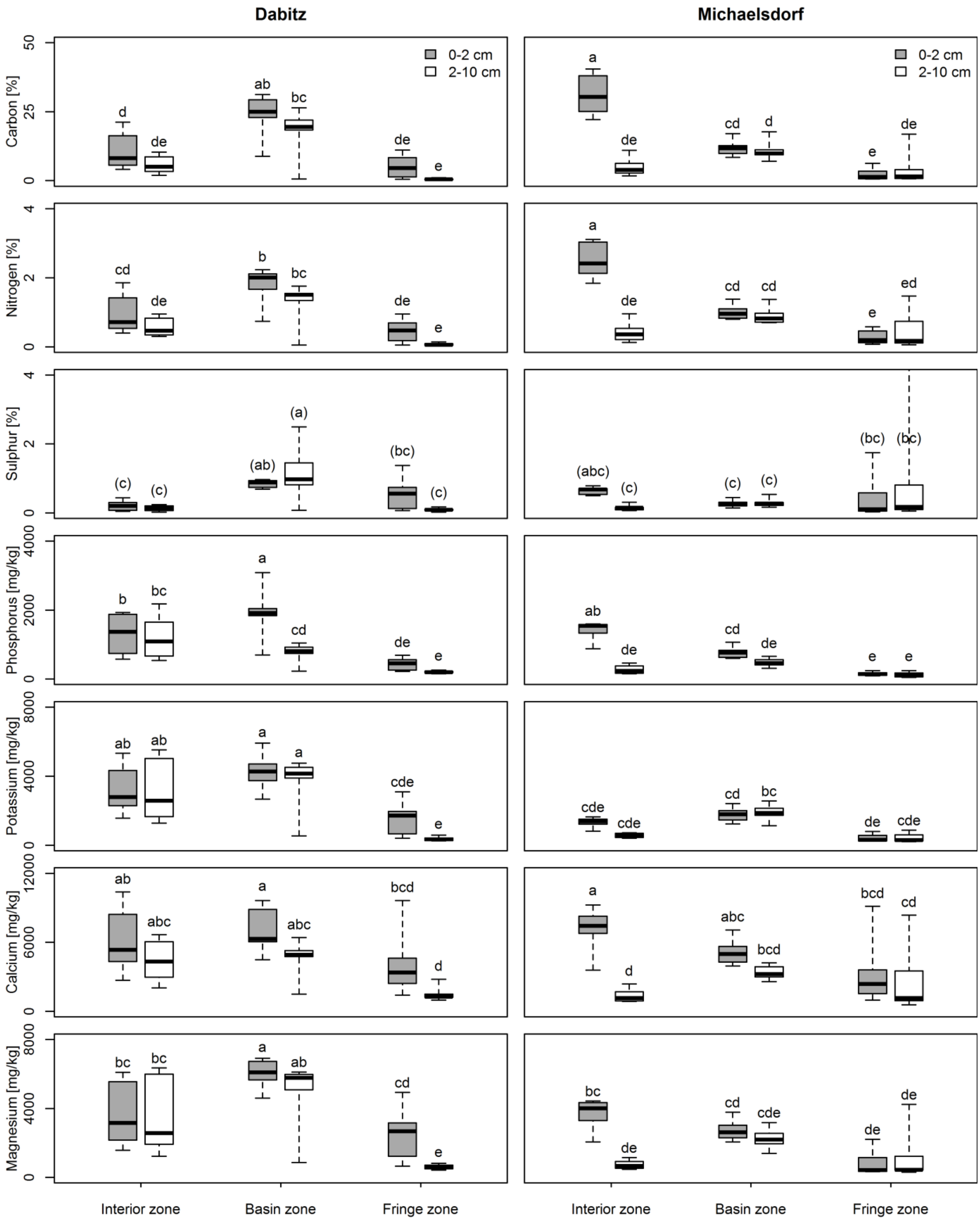

Figure 2-2: Macronutrients in coastal wetland sediments: Carbon, nitrogen, sulphur in [%], phosphorus, potassium, calcium and magnesium in [mg/kg]. The central mark of each box is the median; the edges display the 25th and 75th percentiles. The whiskers extend to the most extreme data points. Letters represent the results of post-hoc comparisons of group means with Tukey's honest significant differences test ($p < 0.05$) which were conducted between all zones of the two study sites (one outlier in sulfur concentrations was removed for testing).

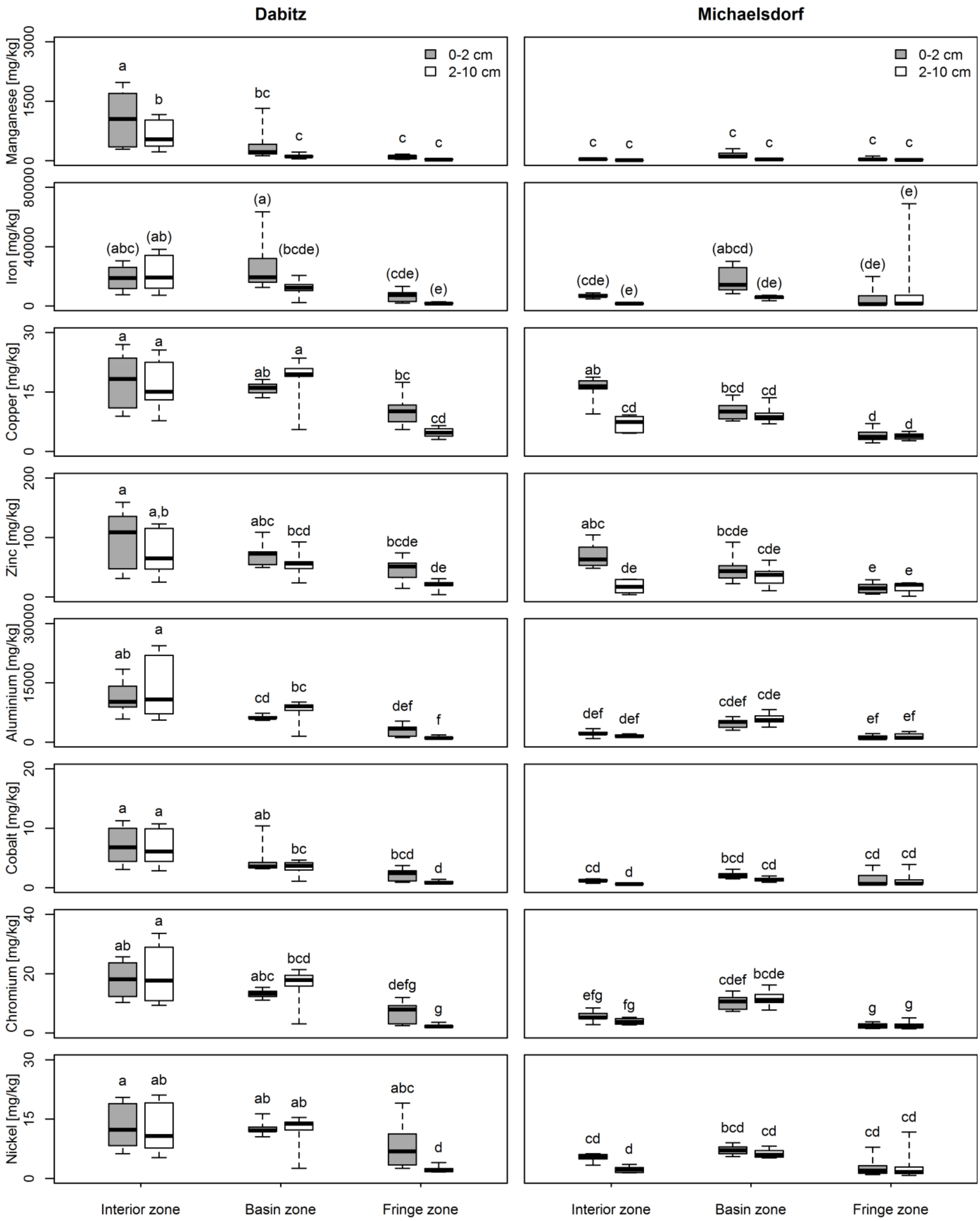

Figure 2-3: Heavy metal concentrations [mg/kg]: Manganese, iron, copper, zinc, aluminum, cobalt, chromium, nickel. Letters represent the results of post-hoc comparisons of group means with Tukey's honest significant differences test ($p < 0.05$) which were conducted between all zones of the two study sites (one outlier in iron concentrations was removed for testing).

The first two principal components of the PCA explain 78.8 % (0-2 cm) and 87.4 % (2-10 cm) of the total variance of the element concentrations (Figure 2-4). The first principal component covers mainly trace elements such as nickel, chromium or copper which also occur in agricultural fertilizers and indicate an ‘anthropogenic’ influence. The second principal component is associated with carbon, nitrogen and sulfur and reflects the ‘natural’ influence from vegetation and water characteristics (see discussion 2.4.1 and 2.4.2). The PCA for the surface sediments (0-2 cm) allows us to divide the factorial plane into four parts: (1) Dabitz interior zone: rich in heavy metals, but comparably poor in carbon, nitrogen and sulphur, (2) Michaelsdorf interior zone: rich in carbon, nitrogen and sulphur, but low heavy metal concentrations, (3) Dabitz basin zone: rich in macronutrients and some heavy metals, and (4) Michaelsdorf basin zone, Michaelsdorf fringe zone and Dabitz fringe zone: low heavy metal as well as low macronutrient concentrations (Figure 2-4 A, Table A-3 appendix). For the lower sediment layer (2-10 cm) the interior zone and the fringe zone at Michaelsdorf are not separated in the PCA (Figure 2-4).

None of the analyzed elements at either study site are significantly negatively correlated (Figure 2-5). At Dabitz, all heavy metals are significantly positively correlated with each other. This pattern is less pronounced at Michaelsdorf.

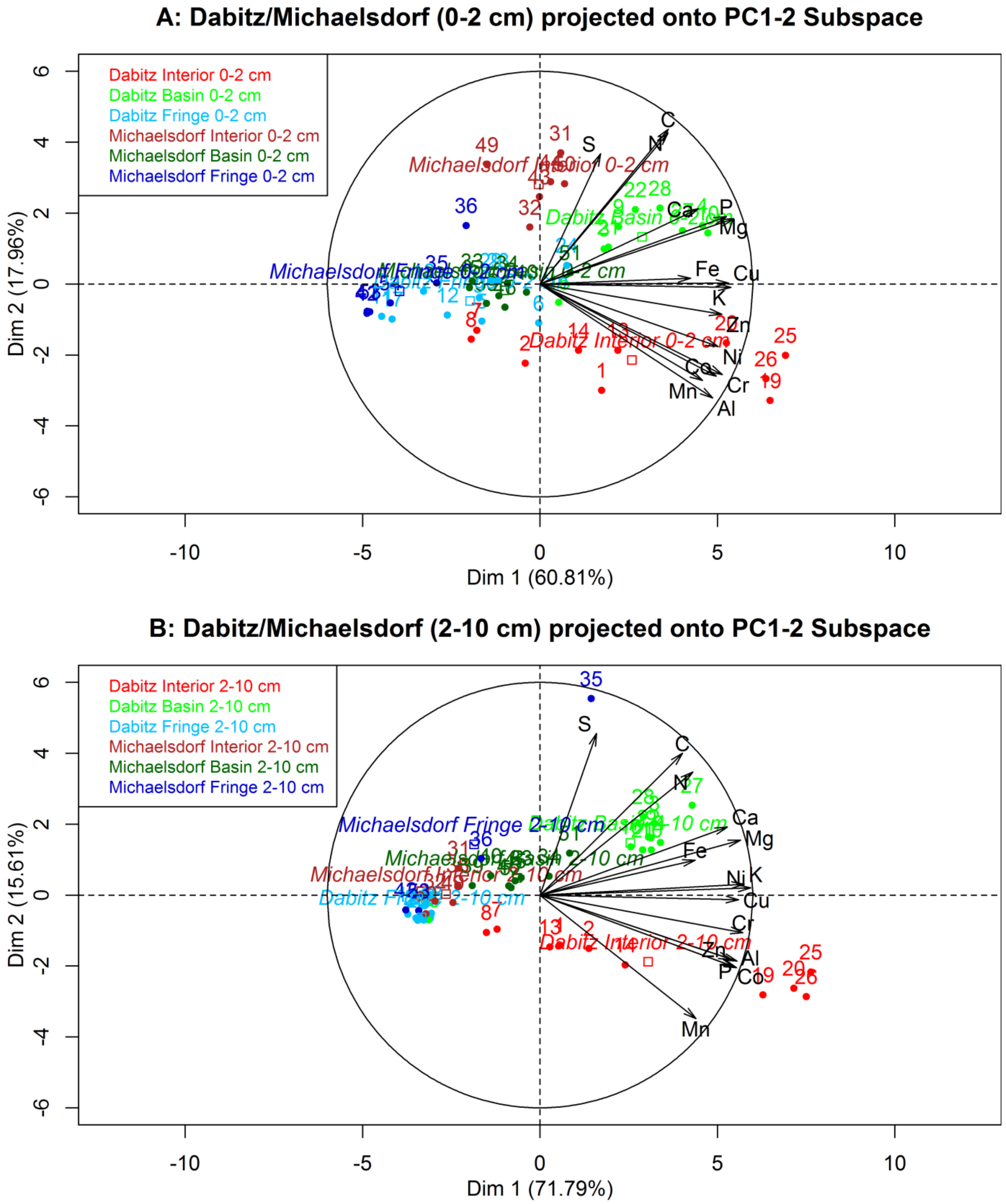

Figure 2-4: Principal component analysis for (A) the surface sediments (0-2 cm) and (B) the lower sediment layer (2-10 cm). For correlations between the first two dimensions and sediment variables see Table A-3 and Table A-4 in the appendix.

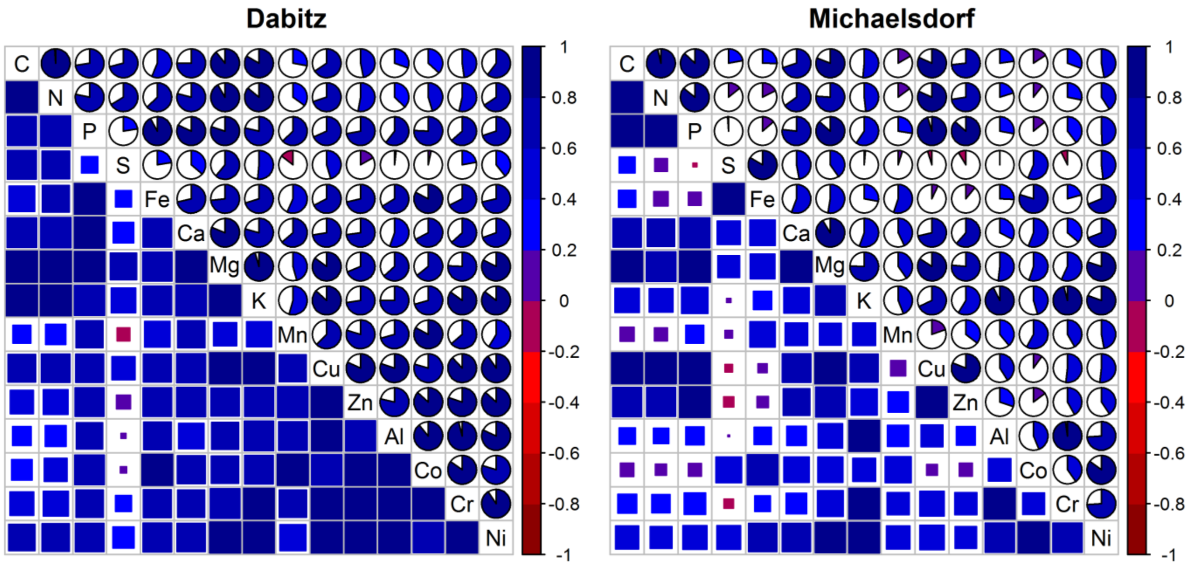

Figure 2-5: Graphical display of Pearson correlation matrix of element concentrations at Dabitz and Michaelsdorf. In the lower triangle the biggest squares represent the highest correlations. In the upper triangle a full ellipse stands for a correlation=1 and an empty ellipse=0. For correlations numbers and significance levels please refer to Table A-1 and Table A-2 in the appendix.

2.4 Discussion

2.4.1 'Natural' influences: Impact of vegetation and water on sediment composition

The PCA showed that the influence from the seaside is minor compared to the influence from the land: the wetland fringes are not distinguishable with respect to their sediment composition. The only exception was an outlier at Michaelsdorf with 7.9 % sulfur. In this sample also the iron concentrations were – with 68 862 mg/kg – tenfold higher than at the other sites. Iron and sulfur were significantly positively correlated at Michaelsdorf (Figure 2-5) and this outlier might have been driven by an iron-sulfide-compound. Besides this exception, sulfur concentrations were highest in the basin zone at Dabitz. Dabitz is situated relatively near to the Bodden outlet, and sulfate ions are supplied by saltwater inflow from the Baltic Sea through the Gellenstrom (Figure 2-1). Under anoxic conditions sulfate is reduced and can react with iron compounds and precipitate as FeS (e.g. Boström et al. 1988). Sediments in the fringe zone at Dabitz are generally well oxygenated due to higher turbulence and also the surface layer in the basin zone becomes anoxic only occasionally (Karstens et al. 2015a). The lower sulfur contents in the upper two centimeters in the basin zone and in the fringe zone compared to the second layer suggest a decreased efficiency of sulfate reduction in the lower sediments.

In the interior zone at Michaelsdorf, we measured the highest carbon, nitrogen and phosphorus stocks in the litter mass compared to all other zones of both study sites. This zone is well protected and rarely flooded. Therefore, litter can be directly incorporated into the sediment. The high carbon, nitrogen and phosphorus concentrations in the surface sediments support this hypothesis. In this zone, *Phragmites* affects the sediment compositions due to its high biomass production and litter abundance and functions therefore as a habitat-modifier (see also Rooth et al. 2003). High biomass production in combination with long residence times of litter material on the sediment floor can increase carbon accumulation rates 5-fold in *Phragmites australis* stands compared to *Spartina patens* stands (Windham 2001). At Dabitz, aboveground biomass and litter mass were slightly lower in the basin than in the interior zone; however carbon, nitrogen and phosphorus concentrations in the surface sediments were significantly higher in the basin zone (Figure 2-2). The basin zone is water-logged most of the year. Shallow inundation, alternating desiccation-flooding-cycles and damp conditions favor decomposition of *Phragmites* litter (Bedford 2005; Völlm and Tannenberger 2014). *Phragmites* is a strong source of carbon, nitrogen and phosphorus in both the interior and basin zone, but the hydrological conditions determine the further fate of those inputs. Additionally, the basin zone at Dabitz is an accumulation area for fine-grained particles rich in organic matter and phosphorus supplied from the Bodden (Karstens et al. 2015a). Although vegetation density and litter accumulation were higher in the fringe zone at Dabitz than at Michaelsdorf, the wetland edges differed only slightly and not significantly in their sediment composition (Figure 2-2, Figure 2-3). Neither vegetation nor water characteristics seem to be sufficiently different at the study sites to produce an overall difference in sediment composition of the wetland fringes (Figure 2-4).

2.4.2 'Anthropgenic influences': Impact of land use on sediment composition

At Dabitz, where the wetland directly borders arable land, heavy metal concentrations were significantly higher in the interior and in the basin zones compared to the fringe zone (Figure 2-3). Besides the six essential macronutrients (N, P, K, Ca, Mg, S), agricultural fertilizers often contain trace elements (Gupta et al. 2014). Therefore, agricultural production and fertilizers applied in the hinterland are often identified as one major input source of heavy metals into adjacent coastal wetlands (Wang et al. 2014). Among the analyzed heavy metals, iron, manganese, copper, zinc and nickel are essential for plant growth, but can be toxic to plants at high concentrations (He et al. 2005). However, other trace metals found frequently in agroecosystem are not essential for plant growth and are toxic for living organisms, including the analyzed chromium (He et al. 2005). The topography of the hinterland at Dabitz is undulating and the elevation decreases towards the perimeter of the Bodden. Therefore, erosion and particle transport from the arable land towards the adjacent wetland seems likely, especially when the cropland lays fallow for 6-8 weeks during summer or when erosion rills form after heavy rain events.

There were no significant differences in heavy metal concentrations in the interior zone at Dabitz between the surface sediments and the sediment layer below, except for manganese (Figure 2-3). This suggests that accumulation of trace elements started already in the past. The hinterland at Dabitz has been used for crop production for more than 60 years and agricultural activities continue to influence the adjacent coastal wetland. Especially phosphorus fertilizers can be a strong source of heavy metal input into agroecosystems (He et al. 2005). Kratz et al. (2015) analyzed trace elements in five different types of mineral phosphorus fertilizers sold in Germany. Most fertilizers contained high amounts of heavy metals, even if not declared as such on the product package (Kratz et al. 2015). Mean element concentrations in straight phosphorus fertilizers were 7289 mg iron, 2934 mg aluminum, 354 mg zinc, 138 mg manganese, 135 mg chromium, 43 mg copper and 24 mg nickel per kg fertilizer (Kratz et al. 2015). Compared to phosphorus fertilizers, nitrogen fertilizers contain only negligible amounts of heavy metals (Kördel et al. 2007). Since 2000, mineral phosphorus fertilizers are no longer applied at the study site Dabitz because the farming company shifted to cow manure. Cow manure contains a tenfold higher amount of zinc and copper and twice as much nickel as mineral phosphorus fertilizers (Kratz and Schnug 2005). Kratz and Schnug (2005) did not include manganese in their study; however German Chambers of Agriculture report higher manganese contents than zinc in cow manure (e.g. Landwirtschaftskammer NRW 2015). Consequently, the manganese input at Dabitz probably increased since the shift from mineral fertilizers to farm manure. Chromium concentrations are lower in cow manure than in mineral fertilizers (Kratz and Schnug 2005), thus we expect a decrease in chromium input. In accordance with that, mean manganese, zinc, copper and nickel concentrations were higher in the surface sediments of the interior zone, while chromium concentrations were lower (Figure 2-3).

Mean heavy metal concentrations in the fringe zone at Dabitz are significantly lower than in the interior and basin zones, indicating that particle transport from land to sea has been inhibited until now. This is in agreement with tracer test studies showing that dense *Phragmites* stands effectively suppress particle transport in the interior zone (Karstens et al. 2015b). However, redox-sensitive elements that accumulated due to agricultural activities may dissolve in the temporarily anoxic basin zone and be transported in solution into the adjacent Bodden water. The release of phosphorus under anoxic conditions in the basin zone at Dabitz has been discussed by Karstens et al. (2015a). For heavy metals, the mobility depends not only on total concentrations but also on sediment properties such as pH or amounts and forms of oxides (He et al. 2005). Depending on the surrounding environmental conditions and temporal dynamics coastal sediments can switch from sinks to sources for heavy metals (El Nemr et al. 2016; Rai 2009). In the basin zone at Dabitz, environmental conditions and especially the oxygen status change quickly, and consequently there is a potential risk of heavy metal release into the Bodden water.

At Michaelsdorf heavy metal concentrations in sediments were low with two exceptions: zinc and copper in the surface sediments of the interior zone. One possible explanation could be leaching from the rusted construction material of the pumping station (see Figure 2-1). The pumping station was installed back in 1970 and the material corrodes. Another possible explanation could be input of zinc and copper from sheep manure. In this case, manure is not applied additionally but left by the grazing sheep. Solid sheep manure contains naturally high amounts of zinc and copper (149 mg zinc and 15 mg copper per kg dry mass; Kördel et al. 2007). However, sheep grazing only takes place between April and October and the herd of sheep rotates. A point-sourced input of copper and zinc from rusting metal seems more realistic. At the moment the pumping station at Michaelsdorf is no longer in operation. However, repeated input of water from channels that drain the hinterland could cause trace element enrichment at a larger scale and contaminations beyond localized point-sources: It has been reported, that field drainage networks may transport pollutants from the hinterland to receiving surface water bodies (He et al. 2005; Hill and Robinson 2012).

Bao et al. (2015) summarized heavy metal concentrations including copper, zinc, chromium and nickel in coastal wetland sediments at 25 different study sites worldwide. The concentrations of copper, zinc, chromium and nickel at Dabitz do not exceed these values and are within the mid-range of reported concentrations. Only manganese concentrations in the surface sediments of the interior zone at Dabitz are – with average 1062 mg/kg – particularly high and above average values measured at other coastal wetlands (e.g. 733 mg/kg Bao et al. 2015; 1013 mg/kg Fernandes et al. 2011; 392 mg/kg Morillo et al. 2004; 127 mg/kg Wang et al. 2014). Heavy metal concentrations at Dabitz and Michaelsdorf are still below the precautionary and action values of the German Federal Soil Protection Ordinance (BBodSchV 1999). Also the potential ecological risk level for zinc, copper, nickel and chromium, developed by Hakanson (1980) for coastal sediments, is still low (<40). The risk index ranges from 'low' (<40) over 'moderate' and 'considerable' to 'high' and 'serious' (>320) and is based on the measured heavy metal concentrations, pre-industrial background values and a 'toxic-response' factor for the given substance (see also El Nemr et al. 2016 or Li et al. 2015 for ecological risk index method). However, if we assume that the concentrations in the fringe zones can be regarded as the natural background value, an enormous accumulation of heavy metals from agricultural production in the hinterland has to be assumed.

2.4.3 Management options inside coastal wetlands

Phragmites has been used traditionally in northern Germany for construction material (Köbbing et al. 2013). Also along the coast of the Darss-Zingst Bodden Chain, our aerial image analysis showed large areas where reed biomass was harvested in February or March but reliable and large scale data on the amount of harvest does not exist yet. However, several studies have proposed phytoremediation as a green technology to remove heavy metals from polluted soils (e.g. McIntyre 2003; Raskin et al. 1997; Rai 2009; Salt et al. 1995). It is regarded as an environmentally friendly and cost-effective technology, where the success depends mainly on plant characteristics: the vegetation has to produce large biomass in short time-frames, accumulation of heavy metals in shoots has to be high and harvest has to be easy (He et al. 2005). Southichak et al. (2006) proposed *Phragmites australis* as a biosorbent for the removal of heavy metals. *Phragmites* contains high amounts of lignin and cellulose which allow the adsorption of heavy metals even at low concentrations (Srivastava et al. 2001). Furthermore, *Phragmites* is very resistant to polluted environments and grows fast with high biomass production (Southichak et al. 2006). *Phragmites* harvest in northern Germany takes usually place in winter (Köbbing et al. 2013). To be most effective, harvest should take place when biomass and heavy metal content reach maximum values. This occurs for *Phragmites* during early autumn before senescence (Bragato et al. 2006). However, harvest in autumn contradicts with the objectives of nature protection in our study region (e.g. bird protection: arrival of cranes in early autumn). Furthermore, the removal of biomass would influence the capacity of the *Phragmites* wetland to suppress particle transport and could potentially accelerate erosion processes and, thus, transfer of heavy metals towards the fringe zones. Presumably, measures directly applied on the cropland might be more appropriate to prevent a transfer from heavy metals into the adjacent coastal wetlands.

2.5 Conclusions

This study demonstrated that land use activities adjacent to coastal wetlands can have a large impact on the sediment composition. Heavy metal concentrations are significantly elevated in the wetland zone that borders directly an arable field where crop production with fertilizer application took place at least since the 1950ies. In contrast to this, heavy metal concentrations are low in the coastal wetland that is confined by a dyke with sheep grazing in the hinterland. The only exceptions are comparably high zinc and copper concentrations in the surface sediments of the interior zone, possibly resulting from point-source leaching of the rusted pumping station at the study site. Influences from the sea on coastal wetland sediment compositions are minor compared to the influences from land: The two wetland fringes do not differ significantly, neither in their macronutrient status nor in their concentrations of heavy metals. While the anthropogenic activities in the hinterland seem to impact the heavy metal accumulation, 'natural' differences in vegetation and water characteristics in the different wetland zones likely influence carbon, nitrogen, phosphorus and sulphur concentrations. Coastal wetlands provide a variety of ecosystem functions and which of these are seen as services often depends on management decisions, including decisions about land use in the hinterland. At Dabitz, the *Phragmites* wetland had to become an active pollutant buffer. Until now heavy metals are accumulated and retained in the interior and basin zones, but future developments should be monitored closely to avoid breakthroughs due to exceeded carrying capacities. Further, the possibility of heavy metal release into the water in the temporarily anoxic basin zone should be investigated.

Acknowledgments

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3. Phosphorus storage and mobilization: Influence of local-scale hydrodynamics

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Abstract

Coastal *Phragmites* wetlands are at the interface between terrestrial and aquatic ecosystems and are important for nutrient retention and regulation. They can act both as nutrient sinks and sources for phosphorus, depending on environmental conditions, sediment properties as well as antecedent nutrient loadings and sorption capacities of the sediments. The Darss-Zingst Bodden Chain is a shallow lagoon system at the German Baltic coast with a long eutrophication history which is lined almost at its entire length by *Phragmites* wetlands. In order to elucidate under which conditions these wetlands act as sources or sinks for phosphorus, in-situ data of chemo-physical characteristics of water and sediment samples were combined with hydrodynamic measurements and laboratory experiments. A basin zone within the wetland serves as accumulation sinks for fine-grained particles rich in phosphorus, iron, manganese and organic matter. The *Phragmites* stems reduce the water flow and allow the accumulation of organic matter and minerals. Without turbulent mixing and downward flow the bottom water and the sediment surface lack replenishment of oxygen. The relationship between oxygen saturation in the water and soluble reactive phosphorus (SRP) concentrations showed a threshold-type behavior. Under oxygen deficiency soluble reactive phosphorus concentrations in the water were significantly higher than under aerated conditions (oxygen saturation < 10%: mean SRP = 0.22 mg/L vs. oxygen saturation > 10%:

mean SRP = 0.02 mg/L). During stagnant periods with low water level, low-turbulence and thus low-oxygen conditions phosphorus from the sediments is released. But the sediments are capable of becoming sinks again once oxygen is resupplied. When vertical downward flows are dominant in the water and turbulent kinetic energy rises, oxygen saturation increases. A thin oxic sediment surface layer rich in iron and manganese is able to readsorb phosphorus quickly as kinetic sorption experiments show, but due to its redox sensitivity the immobilization is - at least in part - only temporary. This study demonstrated that sediments in coastal *Phragmites* wetlands can switch their function from sink to source of soluble reactive phosphorus on a very short time-scale, depending on local-scale hydrodynamic conditions and the state of the oxic-anoxic sediment interface.

3.1 Introduction

Coastal wetlands are interfaces between terrestrial and aquatic ecosystems, receiving freshwater as well as saltwater input (Johnston et al. 1990; Templer et al. 1998; Reddy et al. 1999). They are associated with various ecosystem services including coastal protection and nutrient regulation (Mitsch et al. 2009; Duarte et al. 2013). While nitrogen is the key limiting nutrient in marine ecosystems, phosphorus is mostly limiting in freshwater ecosystems, wetlands and estuaries, and there, excessive concentrations of phosphorus are largely responsible for eutrophication (Corell 1998; Reddy et al. 1999). Coastal wetlands have the ability to transform, retain or release phosphorus and are therefore an important factor for regulating the nutrient status of the adjacent water body (Lai and Lam 2008).

The German coast of the southern Baltic Sea is characterized by lagoon systems featuring coastal wetlands dominated by *Phragmites australis* (Cav) Trin. Ex Streudel (common reed), for instance the Darss-Zingst Bodden Chain. Eutrophication of the Baltic coastal waters increased severely in the late 20th century (Kautsky et al. 1986; Schiewer 1998) and although in recent years nutrient input from agricultural activities decreased, the Darss-Zingst Bodden Chain is still strongly eutrophic (Selig et al. 2007). Eutrophication is driven by excessive nutrient concentrations leading to high phytoplankton production. This high productivity results in elevated organic matter degradation and respiration rates, with concomitant oxygen consumption, leading eventually to hypoxic or anoxic conditions in the bottom waters (Corell 1998). Low oxygen concentrations are further aggravated if calm and warm conditions prevail and mixing through waves, currents or turbulence is low (Kemp et al. 2009; Berg et al. 2003). Strong stratifications in the water column and long residence times in the water body increase the sensitivity of coastal waters to hypoxia (Howarth et al. 2011). A positive feedback can develop and reinforce eutrophication in case anoxic sediments release further nutrients which were previously adsorbed and bound in the sediments (Corell 1998). Especially for phosphorus, a feedback loop can develop in which nutrient release from sediments increases eutrophication, while eutrophication itself reinforces nutrient release. More organic material settles and more respiration intensifies the anoxic situation if oxygen transport is repressed (Corell 1998; Gomez et al. 1999; Selig et al. 2007).

The objectives of this paper are (1) to explore under which conditions high concentrations of soluble reactive phosphorus occur in the water column in two different wetland zones, (2) to understand phosphorus release and adsorption mechanisms by means of sorption isotherms, sorption kinetics and sediment characteristics, and (3) to elucidate the interactions between turbulent kinetic energy, fluctuations of vertical velocity, oxygen saturation and soluble reactive phosphorus.

3.2 Materials and methods

3.2.1 Site description

The Darss-Zingst Bodden Chain, a lagoon system in northeast Germany, is connected with the open Baltic Sea only by a narrow outlet in the northeast (Figure 3-1). Hence, water exchange between the open Baltic Sea and the Bodden is low and takes place sporadically in form of distinct events, mostly after strong and persistent northeasterly winds (Selig et al.

2007). The entire lagoon system with its four sub-basins covers an area of almost 200 km² and is a shallow water body with a mean water depth of 2 m (Karsten et al. 2003; Schubert et al. 2003). The boddens were formed after the Weichselian glaciation during the Litorina transgression in an area with glaciogenic basins and meltwater channels (Lampe 1990). Salinities decrease from east to west due to freshwater input from the rivers Recknitz and Barthe and the dominant outflow situation, from 0-3 PSU in the innermost lagoon (Saaler Bodden) to 7-10 PSU in the outermost lagoon (Grabow) (Selig et al. 2007).

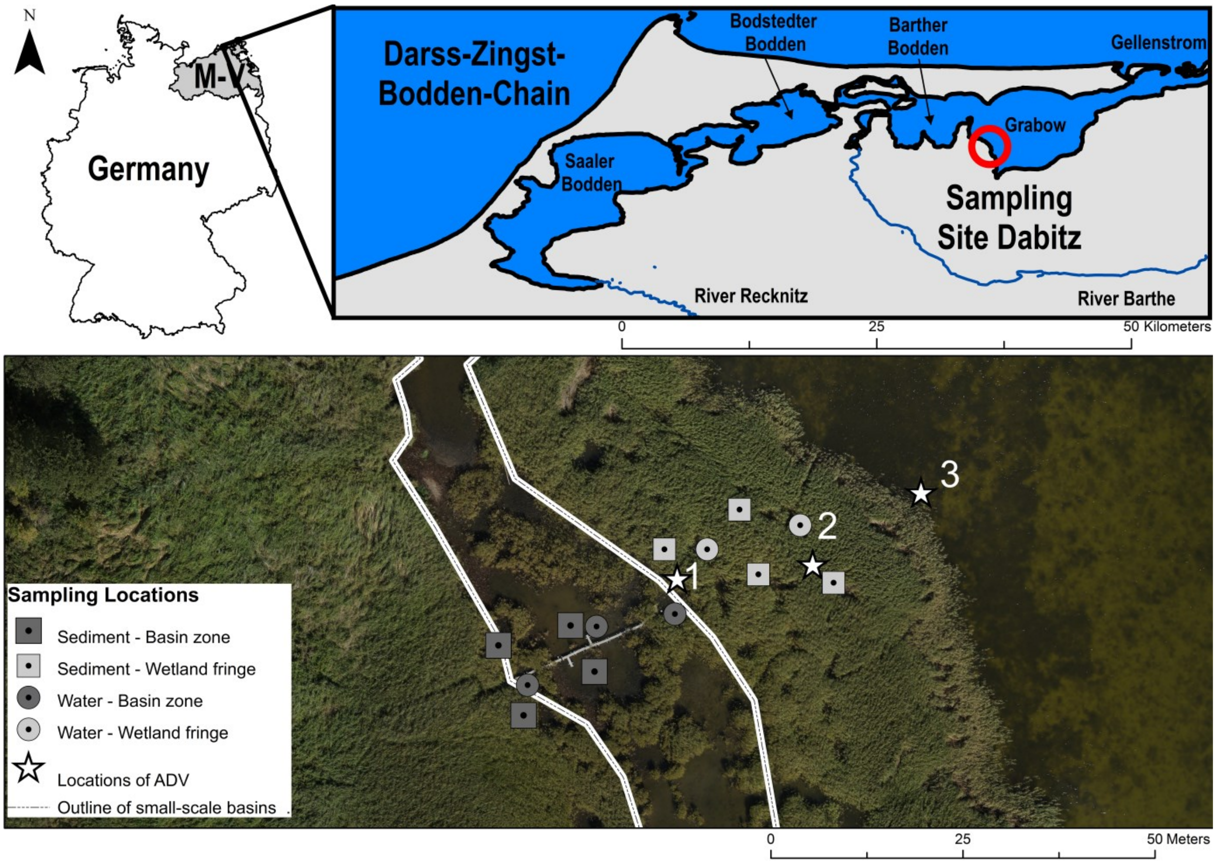

Figure 3-1: The Darss-Zingst Bodden Chain in Mecklenburg-Western Pomerania (MV), northeast Germany, is a lagoon system connected to the open Baltic Sea by a narrow outlet (Gellenstrom) in the north-east. The wetland at the sampling site *Dabitz* is 130 m wide and heterogeneous with small-scale basins (5-25 m width; elevation of -0.2 to 0.1 m above mean sea level) inside the wetland. The lower map shows the sediment and water sampling points as well as the locations where the acoustic Doppler velocimeter was deployed (1= basin zone inside the wetland; 2= wetland fringe; 3= outside the *Phragmites* wetland).

The Darss-Zingst Bodden Chain is lined with an almost continuous belt of *Phragmites* wetlands. In this study, water and sediment samples from a *Phragmites* wetland close to

Dabitz at the southwest littoral of the Grabow have been examined (Figure 3-1) The topography of this coastal *Phragmites* wetland is heterogeneous with a zone of small-scale basins of 5-25 m width. Such small-scale basin morphology is common within the wetlands along the Darss-Zingst Bodden Chain. Consequently, at the Dabitz site, two distinct zones can be discerned in the *Phragmites* wetland: (1) basin zone and (2) wetland fringe (Figure 3-1). These two zones exhibit different hydrological characteristics: in general water levels and turbulence are lower in the basin zone than in the wetland fringe. Mean water level in the basin zone was 18 cm compared to 34 cm in the wetland fringe (relative to ground level). Water level changes of several centimeters can occur during a week and depend on wind conditions. Already in the 1970ies the terms ‘basin’ and ‘fringe’ were proposed to classify mangrove forests (see Lugo and Snedaker 1974) and we adopted these terms for the *Phragmites* wetland. The hydrologic energy in fringe mangroves, that border large water bodies, is higher than in the basin mangroves that are further inland and less frequently flooded (Twilley et al. 1986; Lugo et al. 1988; Brinson et al. 1993). This also applies to our study site.

3.2.2 Methodological approach

It was expected that hydrodynamic processes have a larger influence on hydrochemical parameters and phosphorus dynamics in the basin zone compared with the fringe zone. Consequently, water and sediment samples were analyzed separately for both these zones. Also, measurements of turbulent kinetic energy were conducted separately in the basin zone and wetland fringe (Figure 3-1). Turbulence and fluctuations of the vertical velocity of the water influence the oxygen status of bottom waters and surface sediments (Berg et al. 2013). The redox status of the sediment surface has a decisive influence on the stability and amount of oxidized iron and manganese compounds within the sediment which have a large capacity to adsorb phosphorus from the water body. Thus, turbulence and fluctuations of the vertical velocity can be used as a proxy for the ability of the surface layer to adsorb phosphorus. Sorption experiments were conducted for the upper 10 cm of the basin zone sediments to evaluate the phosphorus sorption capacity under oxic conditions. Both equilibrium and kinetic sorption experiments were conducted, since it was expected that equilibrium conditions may not be attained in case of swift changes in the hydrodynamic and hydrochemical conditions. Sorption experiments were restricted to the temporarily anoxic basin zone because only there high concentrations of soluble reactive phosphorus occurred.

3.2.3 Water and sediment samples

Water samples were taken monthly between October 2013 and November 2014. Additional field experiments were carried out on a diurnal scale in June and July 2014. Soluble reactive phosphorus concentrations were determined by the molybdenum blue method (e.g. Murphy and Riley 1962, Lai and Lam 2008). Sulfate was measured photometrically after precipitation with barium. Redox potential, pH, conductivity, temperature and oxygen saturation were measured in situ above the sediment bed using *Hach Lange* sensors.

A total of 8 sediment cores were collected within the basin zone and the wetland fringe and separated into 0-2 cm and 2-10 cm slices. Redox potential (Eh) of the sediment in 10 cm depth was measured in situ with an ORP meter (*Hach Lange*). Subsamples of the sediments were dried at 105°C overnight and used for further physico-chemical analysis. A subset of sediments was wet-sieved to determine the mean particle size and mud content (% <63µm) (Selig et al. 2007). Sediment organic matter was determined gravimetrically by loss-on-ignition (LOI) in a muffle furnace at 550°C for 4h. Total carbon, nitrogen and sulfur contents were quantified by combustion in a CNS Analyzer (*Vario Max, Elementar*). A multi element analysis including total phosphorus, iron, aluminum, manganese and calcium was carried out by inductively coupled plasma optical emission spectrometry after aqua regia digestion (*Optima 5300 DV, Perkin-Elmer*).

3.2.4 Sorption experiments under aerobic conditions

Equilibrium sorption isotherms were assessed by batch incubation experiments (e.g. Slomp et al. 1998; Sundareshwar and Morris 1999; Pant and Reddy 2001). From three sediment cores from the basin zone, rhizomes and other plant material were removed and the samples were immediately homogenized. Sediments were neither dried nor sieved to prevent oxidation and changes in phosphorus sorption characteristics (Cyr et al. 2009; Lai and Lam 2009). For incubation under aerobic conditions, 2 g fresh (i.e. original water content) sediment was mixed with a 50 ml solution containing 0.01 M KCl and ten different P concentrations (0, 0.1, 1, 2.5, 5, 10, 25, 50, 100, 200 mg KH_2PO_4/L, pH 7). Later on the adsorbed amount of phosphorus was adjusted to the dry mass of sediment. A sediment:solution ratio of 1:25 was chosen to simulate the ratio of surface waters (~10-40 cm) and a thin oxic sediment layer (~0.5-1.5 cm). The centrifuge tubes were shaken for 24h at 20 ± 2 °C. Experiments were carried out in duplicates. After equilibrating for 24h the samples were filtered and the filtrate immediately analyzed for soluble reactive phosphorus. The difference between the initially added P concentration and the P concentration measured after 24h corresponds to the amount of P absorbed by the sediment.

Sorption data were fitted to the Langmuir model (eq. 1) to calculate the phosphorus sorption maximum.

$$S = S_{max} * \frac{k_L * C_{eq}}{1 + k_L * C_{eq}} - S_0 \qquad (1)$$

With S the amount of P adsorbed during the sorption experiments [mg P/kg], C_{eq} [mg/L] the equilibrium P concentration in the solution after 24 hours, k_L a bonding energy constant [L/mg], S_0 [mg P/kg] the native sorbed P and S_{max} [mg P/kg] the sorption maximum (see also Pant and Reddy 2001; Cyr et al. 2009; Lai and Lam 2009).

Prior to batch incubation experiments, kinetic sorption experiments of the same sediments were conducted to study the speed of adsorption processes. 2 g of fresh sediments were added to a series of 50 ml tubes containing a solution with 0.01M KCl and three different P concentrations (0.1, 1, 10 mg KH_2PO_4/L). The centrifuge tubes were shaken and samples were taken at nine different time intervals (5min, 15min, 30min, 1h, 5h, 15h, 24h, 48h, 7 days). For each time step and each concentration, centrifuge tubes were set up in triplicates.

3.2.5 Measurement of turbulent kinetic energy

Oxygen saturation of bottom waters and oxygen uptake by sediments is influenced by turbulence and fluctuations of the vertical velocity. Oxygen concentrations above the sediment bed increase when the vertical velocity is pointing down and vice versa (Berg et al. 2003). The fluctuating vertical velocities and turbulent kinetic energy were measured with a 3D acoustic Doppler velocimeter (ADV) (*Nortek - Vector*). The velocimeter measures three velocity components (x,y,z) based on an acoustic echo from particles moving through the cylindrical measuring volume which is located approximately 10 cm from the base of the three sensors. A data output rate of 16 Hz was chosen, separated into bursts with 4800 samples in 300 seconds to facilitate evaluations of the data later on. Turbulent kinetic energy was calculated using turbulent fluctuations and turbulent strength. As our ADV records a series of discrete data points we can use the Reynolds decomposition where an average part $\overline{u}$ is separated from the fluctuations u_i'. Turbulent fluctuations equal $u_i' = u_i - \overline{u}$ and the turbulent strength is $u_{rms} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(u_i')^2}$. Turbulent kinetic energy is therefore a representative of the intensity of variations from the mean velocity: $TKE = \frac{1}{2}(\overline{u_x'^2} + \overline{u_y'^2} + \overline{u_z'^2})$.

Several researchers have already used *Nortek's Vector* for calculations of turbulence, bed shear stress or vertical fluxes (see Berg et al. 2003; Pope et al. 2006; Salehi and Strom 2012). In order to explore the differences of turbulent kinetic energy between zones within and outside the wetland we moved the velocimeter every ~30 minutes to another location as only one device was available. We took care that wind speed and water level did not change during those measuring intervals making the measurements at different locations comparable.

Vegetation mapping including stem density and stem width data was conducted monthly between June and September 2014, separately in the basin and fringe zones. Climate data including wind speed, precipitation, soil and air temperature were recorded on an hourly basis throughout the year (weather station *DALOS 353 W*).

3.2.6 Statistical methods

The data were tested for normal distribution using the Kolmogorov-Smirnov test. If necessary, the data were transformed before parametric analysis using logarithmic transformation. ANOVA was used to test for differences in sediment characteristics according to location (basin, fringe) and depths (0-2 cm and 2-10 cm). *A priori*, the Levene's test provided information about the homogeneity of variances. Scatter plots were used to explore the relationships between different variables. In case of linear correlations the Pearson's product-moment correlation coefficient was calculated. In case of non-linearity but monotonic relationships Spearman's rank-order or Kendall's tau coefficient correlations were calculated. Statistical analysis, calculations and graphical representations were performed with Matlab R2013b or IBM SPSS Statistics 20.

3.3 Results

3.3.1 Concentrations of soluble reactive phosphorus in the water column

The mean soluble reactive phosphorus concentration in the basin zone averaged over the year and the three sampling locations was 0.12 mg/L, while the mean concentration at the two wetland fringe sites was 0.014 mg/L. The differences in concentrations of soluble reactive phosphorus were significant between the basin zone and the wetland fringe ($p<0.01$). While the soluble reactive phosphorus concentration in the basin zone reached values of up to 0.9 mg/L, the concentrations in the wetland fringe stayed below 0.1 mg/L. Also the water level and the oxygen saturation in the basin zone and in the wetland fringe were significantly different ($p<0.01$). Mean oxygen saturation at the basin site was 35 %, but 62 % at the wetland fringe. The water level was significantly lower at the basin sites with a mean value of 18 cm compared to 34 cm in the wetland fringe ($p<0.01$). Especially high concentrations of soluble reactive phosphorus were measured in the basin site for oxygen saturation below 10 %, water levels below 15 cm and negative redox potentials (Figure 3-2). Our results show a significant correlation between soluble reactive phosphorus concentration in the water column and oxygen saturation (Spearman's correlation coefficient, r_s= -0,764**), water level (r_s= -0,689**) and redox potential (r_s= -0,527**).

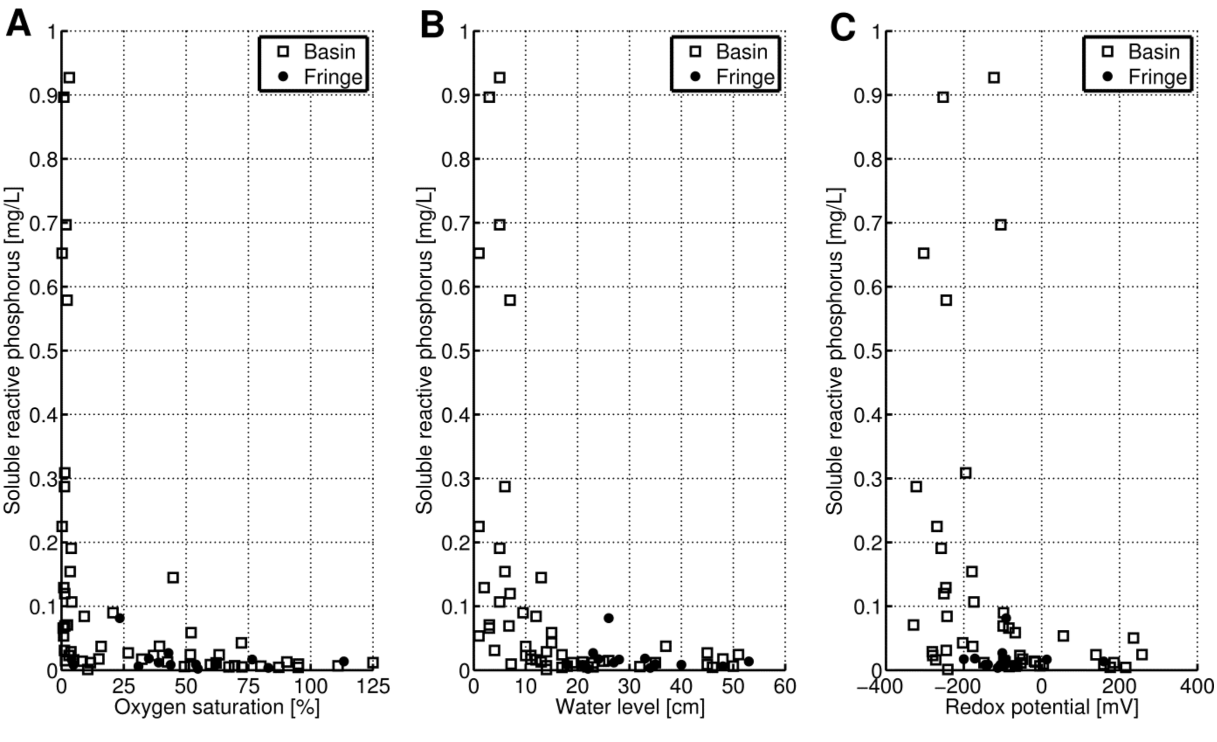

Figure 3-2: (A) Soluble reactive phosphorus concentration (SRP) vs. oxygen saturation of bottom waters (n=68). SRP concentrations increase sharply below a threshold oxygen saturation of 10%. (B) Water levels vs. SRP concentration (n=69). (C) Redox potential vs. SRP concentrations (n=67). The interplay between SRP and oxygen, SRP and water level, and SRP and redox potential indicates a threshold-type behavior. In the *Phragmites* wetland fringe, SRP concentration remained comparatively low throughout the sampling period.

During measurement days when water level was below 10 cm and the wind speed at 2 m height below 4 m/s, the oxygen saturation in the water never exceeded 10 %. When the water level dropped, the bottom water became first hypoxic and finally anoxic. At days with high oxygen saturation in the water column, a light brown thin layer was visible at the sediment surface (Figure 3-3). In contrast, when low oxygen conditions were present in the water column at the study site Dabitz, the lighter brown surface layer disappeared. The sediment below this layer, with a mean redox potential of - 201 ± 63 mV, showed further strong symptoms of anoxia such as a black color and a distinctive smell of hydrogen sulfide.

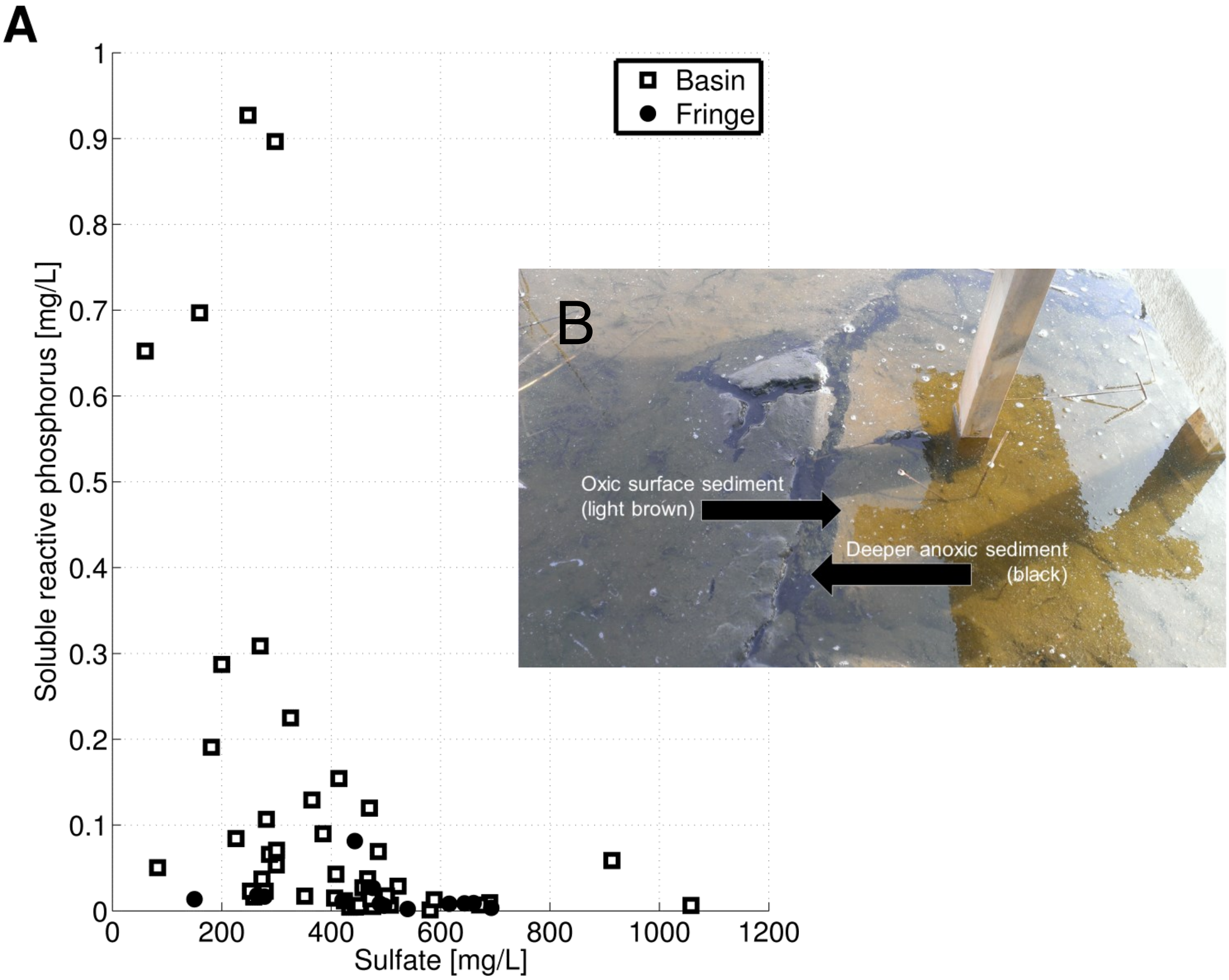

Figure 3-3: (A) Sulfate concentration vs. soluble reactive phosphorus (n= 64). (B) Photo of a thin, light brown, oxic surface layer during one measurement interval.

There was a significant inverse correlation between soluble reactive phosphorus and sulfate concentrations (r_s= -0,608**, Figure 3-3). Temperature seems to have less influence on phosphorus dynamics (Table 3-1) and days with high concentrations of soluble reactive phosphorus occurred in all seasons. Continuous time series with data logging of oxygen would be needed to elucidate the frequency, duration and seasonal pattern of such events.

Table 3-1: Median, minimum and maximum values of soluble reactive phosphorus (SRP), oxygen saturation, water level, conductivity, pH, temperature, and sulfate concentration in the bottom waters for the different wetland zones. Spearman's correlation coefficients r_s *for correlations with soluble reactive phosphorus concentrations are included. ** Correlation is significant at the 0.01 level (2-tailed). *Correlation is significant at the 0.05 level (2-tailed).*

	Both zones	Basin wetland	Fringe wetland
SRP [mg/L]			
Median	0.018	0.029	0.009
Min	0.001	0.004	0.001
Max	0.927	0.927	0.081
n	76	57	19
Oxygen Saturation [%]			
Median	43.4	26.8	54
Min	0.25	0.25	4
Max	131	125	131
r_s	- 0.764**	*- 0.778***	*- 0.507**
n	82	61	21
Redox Potential [mV]			
Median	- 94	- 89	- 95
Min	- 329	- 329	- 200
Max	258	258	160
r_s	- 0.527**	*- 0.618***	*- 0.054*
n	71	54	17
Water Level [cm]			
Median	20	14	33
Min	1	1	18
Max	57	53	57

r_s	- 0.689**	*- 0.746***	*- 0.122*
n	83	62	21
Conductivity [mS/cm]			
Median	13.9	13.9	13.8
Min	8.7	8.7	12.1
Max	18	18	14.7
r_s	- 0.155	*- 0.238*	*0.191*
n	72	55	*17*
pH			
Median	7.3	7.3	7.4
Min	6.8	6.8	7.1
Max	8.9	8.9	8.9
r_s	- 0.482**	*- 0.412***	*- 0.522**
n	74	54	*20*
Temperature [°C]			
Median	15.9	16.9	14.8
Min	0	0	3.7
Max	26.3	26.4	21.7
r_s	- 0.059	*- 0.288**	*0.198*
n	83	62	*21*
Sulfate [mg/L]			
Median	444	429	476
Min	60	60	150
Max	1058	1058	639
r_s	- 0.608**	*- 0.607***	*- 0.604**
n	65	50	*15*

3.3.2 Sediment characteristics, sorption isotherms and sorption kinetics

Total phosphorus contents in the sediment differ significantly between the sampling locations. The high phosphorus content and the large variability in the 0-2 cm slices of the basin sediments stand out (Figure 3-4). ANOVA shows that the distributions of phosphorus, iron, aluminum, calcium or manganese content differ across locations and depths ($p<0.05$).

The surface sediment layer of the basin zone is characterized by remarkably high phosphorus, iron, manganese, calcium, carbon and mud contents (Figure 3-5, Table 3-2).

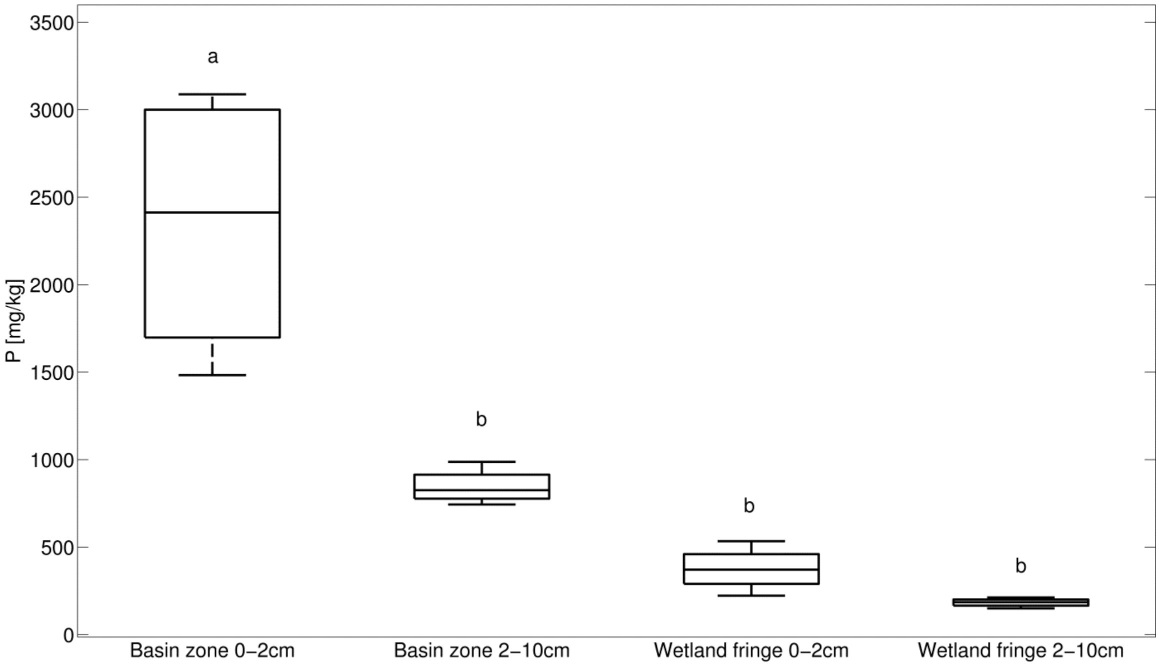

Figure 3-4: Boxplots of total phosphorus contents [mg/kg] in sediments for the two sampling locations "basin zone" and "wetland fringe" and the two depths (total n=16). The central mark in each box is the median; the edges display the 25th and 75th percentiles. The whiskers spread out to the most extreme data points. Letters 'a' and 'b' represent the results of Turkey's post-hoc comparisons of group means (p<0.05).

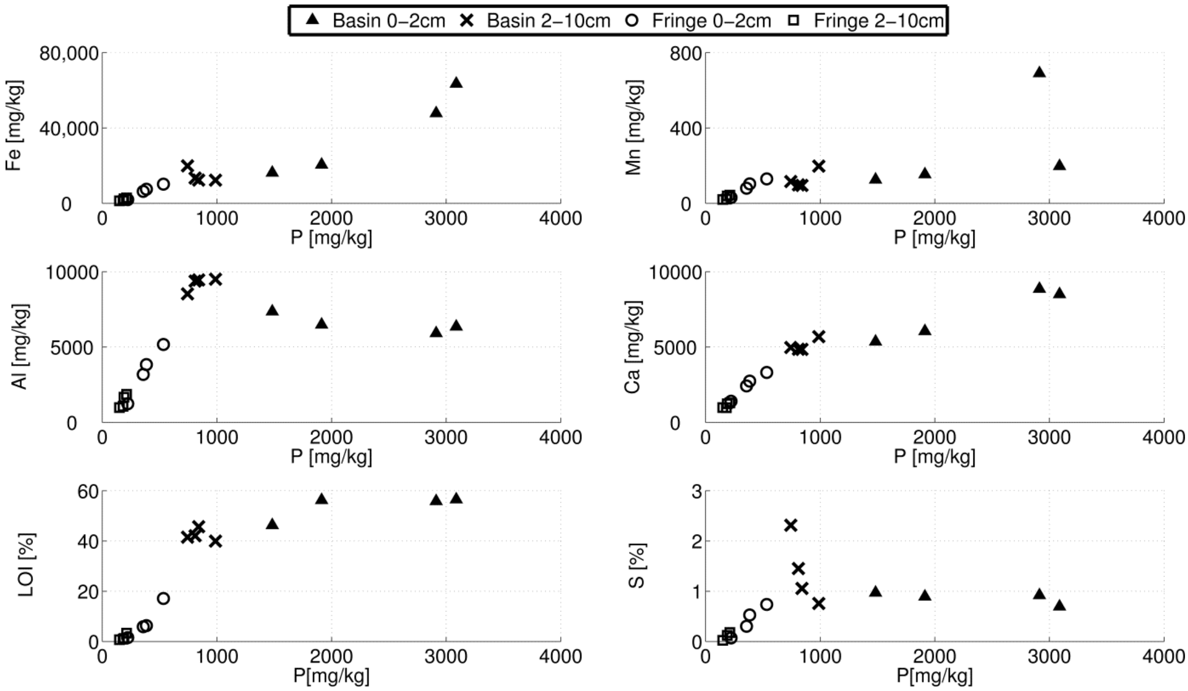

Figure 3-5: Total phosphorus content in sediment samples versus total iron, manganese, aluminum, calcium, organic matter content and sulfur (total n= 16; 4 per site and depth).

Phosphorus and iron concentrations in the sediment samples correlate significantly (Pearson correlation coefficient 0.964**). Significant positive correlations were also observed between phosphorus and manganese (0.901**) and calcium (0.983**). Organic matter and total carbon contents show a bimodal distribution which can be stratified into two normal distributions when split into the two different locations *basin* (LOI= 48 ± 7%) and *fringe* (LOI= 4.6 ± 5.5%). Kendall's tau coefficient for organic matter and phosphorus content regarding all sediment samples is 0.917**. In the basin sediments, total sulfur content is higher in 2-10 cm depth compared with the surface sediments (Figure 3-5, Table 3-2).

Table 3-2: Mean values of total phosphorus, total iron, total aluminum, total manganese, total carbon, total sulfur, organic matter (loss-on-ignition), mud content (grains size <63µm) and standard deviation. n= 16 (4 samples per subcategory), except mud content: n= 4.

	P [mg/kg]	Fe [mg/kg]	Al [mg/kg]	Mn [mg/kg]	Ca [mg/kg]	C [%]	S [%]	LOI [%]	Mud [%]
Basin 0-2cm	2349 ± 775	36976 ± 22519	6525 ± 604	292 ± 267	7192 ± 1746	25 ± 1.9	0.87 ± 0.12	54 ± 5	40
Basin 2-10cm	846 ± 103	14466 ± 3638	9210 ± 460	126 ± 48	5088 ± 402	20 ± 1.6	1.4 ± 0.68	42 ± 2.4	7.6
Fringe 0-2cm	375 ± 128	6474 ± 3432	3355 ± 1639	86 ± 42	2466 ± 800	3 ± 2.1	0.41 ± 0.29	7.7 ± 6.6	6.2
Fringe 2-10cm	183 ± 126	2021 ± 733	1395 ± 436	31 ± 12	1122 ± 166	0.7 ± 0.4	0.11 ± 0.08	1.7 ± 1.2	1.7

For the basin sediments, the Langmuir sorption maximum (S_{max}) of 15921 mg/kg as well as the bonding constant k are extremely high (Figure 3-6 A). Thus, under oxic conditions, the sorption capacity of the sediment for phosphorus is five times larger than the actual total phosphorus content. The kinetic sorption experiments showed that phosphorus sorption in the basin sediments is fast. As much as 65 % of the adsorption took already place within the first five minutes (Figure 3-6 B).

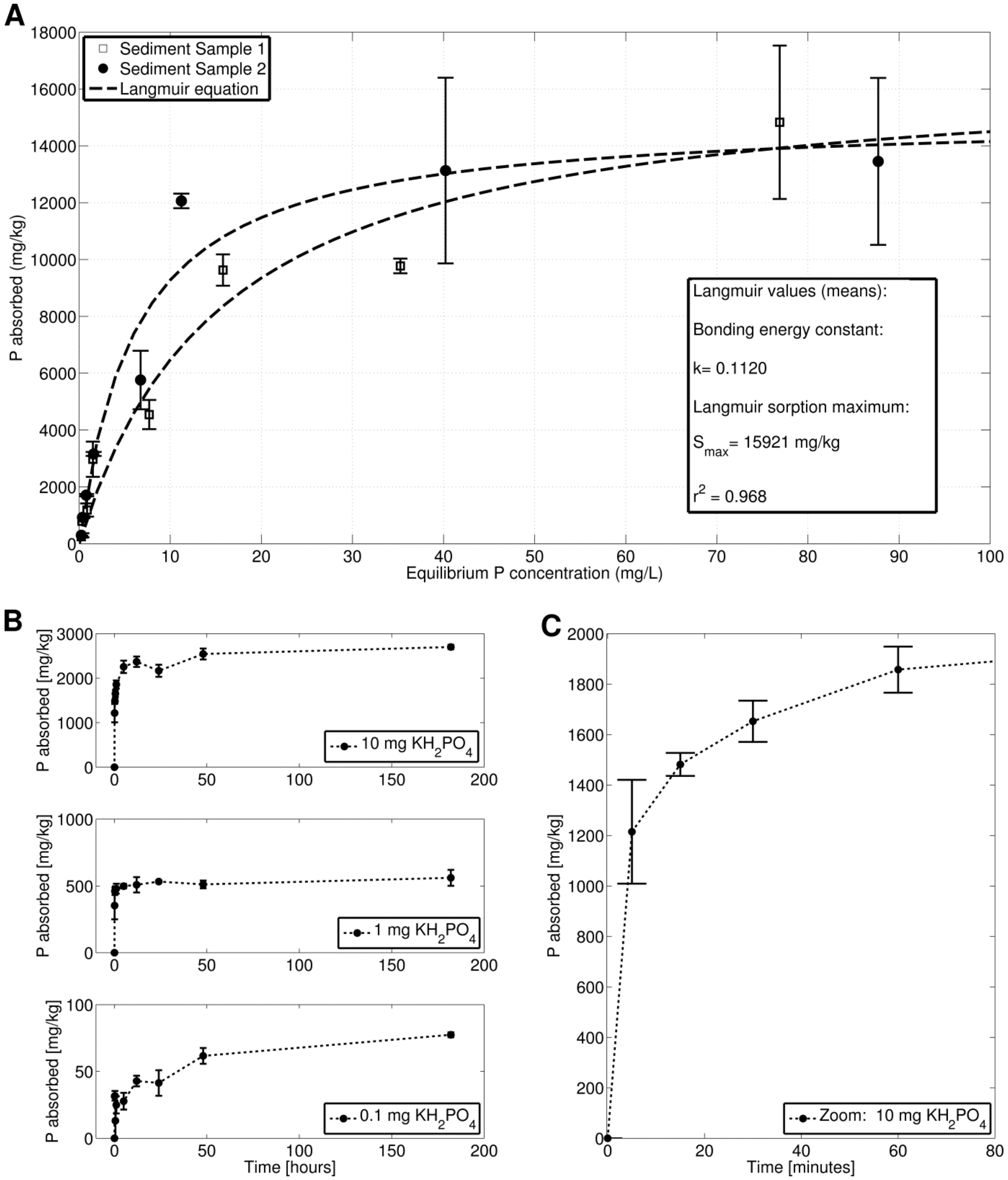

Figure 3-6: (A) Langmuir sorption isotherms for basin sediments. The Langmuir sorption maximum of 15,921 mg/kg is high and the sorption limit under oxic conditions by far not reached yet. (B) Sorption kinetics for the basin sediments with 10, 1 and 0.1 mg KH_2PO_4 (3 replicates). (C) Zoom into the first 60 minutes of adsorption for the 10 mg KH_2PO_4 curve. Already 65% of the adsorption took place after 5 minutes.

3.3.3 Local-scale hydrodynamics

Our turbulence measurements show significantly lower turbulent kinetic energy in the basin zone and the wetland fringe compared with the zone outside the *Phragmites* wetland (Figure 3-7). The mean turbulent kinetic energy outside the *Phragmites* wetland was 30 times larger than in the small-scale basins. Mean turbulent kinetic energy was in (1) the basin zone 0.045×10^{-3} m² s^{-2}, (2) the wetland fringe 0.18×10^{-3} m² s^{-2}, and (3) outside the *Phragmites* wetland 1.39×10^{-3} m² s^{-2}. Vegetation densities and stem widths increased from the fringe towards the inner part of the *Phragmites* wetland. In the wetland fringe stem densities were lower than in the vegetation surrounding the basin zone, with 381±72 and 469±113 stems per m², respectively. Mean stem width was 3.88±0.1 mm in the wetland fringe and 4.41±0.43 mm next to the basins.

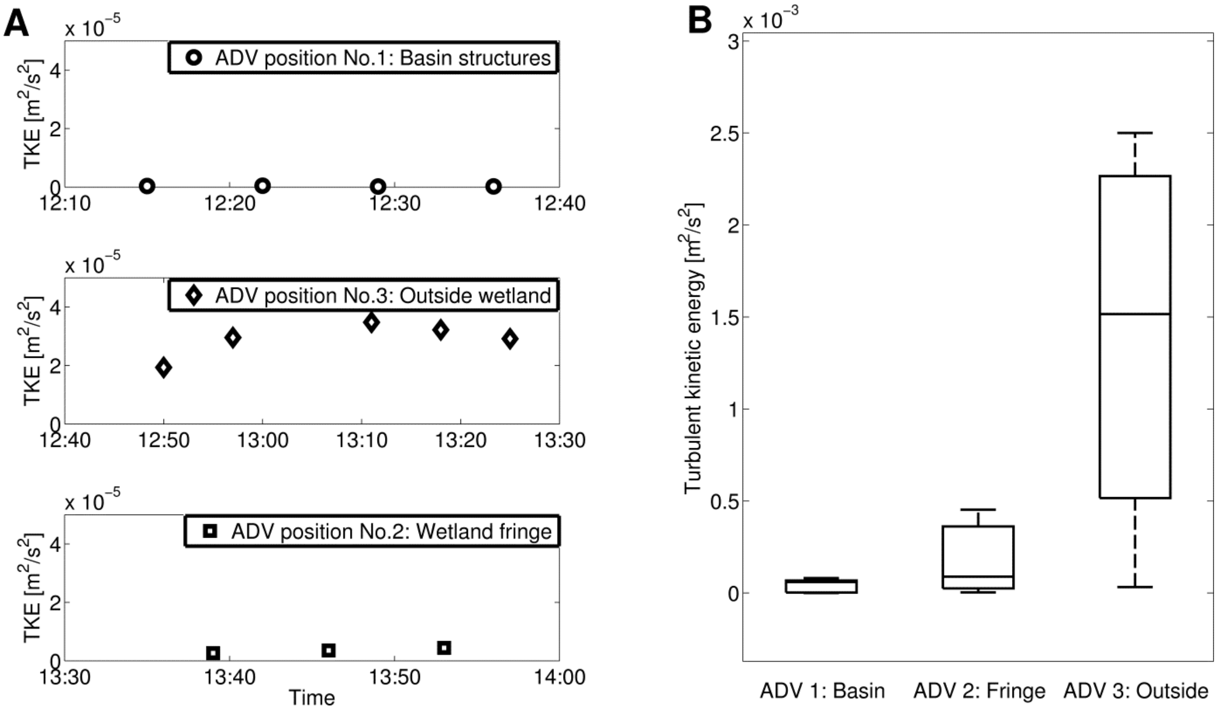

Figure 3-7: (A) Example for variations of turbulent kinetic energy inside and outside the *Phragmites* wetland. The wind speed (1.3 m/s) did not change over the measurement interval. (B) Box plots of turbulent kinetic energy in the basin zone, wetland fringe and outside the wetland. The central line in each box is the median; the edges display the 25th and 75th percentiles. The whiskers spread out to the most extreme data points. N= 180 bursts during four different days with 4800 samples per burst. Wind speed during measurement days did not exceed 2 Beaufort.

Field experiments demonstrated that if the turbulent kinetic energy in the water rose and a vertical downward flow was dominant, the oxygen saturation would also rise. Changes in oxygen impacted the phosphorus cycle. Adsorption of soluble reactive phosphorus under oxic conditions was fast in the basin zone and soluble reactive phosphorus decreased by 17-49 % within 10 hours on June 24th/25th and 41-55 % within 7 hours on July 8th. Concomitantly, we observed an increase of the turbulent kinetic energy and a dominant vertical downward flow (Figure 3-8). The oxygen saturation in the basins rose from 9±8 to 38±34 % during the first field experiment in June, while the soluble reactive phosphorus concentration decreased from 0.084±0.046 to 0.058±0.044 mg/L. The same phenomenon occurred during the second field experiment in July: oxygen saturation increased from 26±32 % to 84±31 % and phosphate decreased from 0.017±0.009 to 0.008±0.004 mg/L.

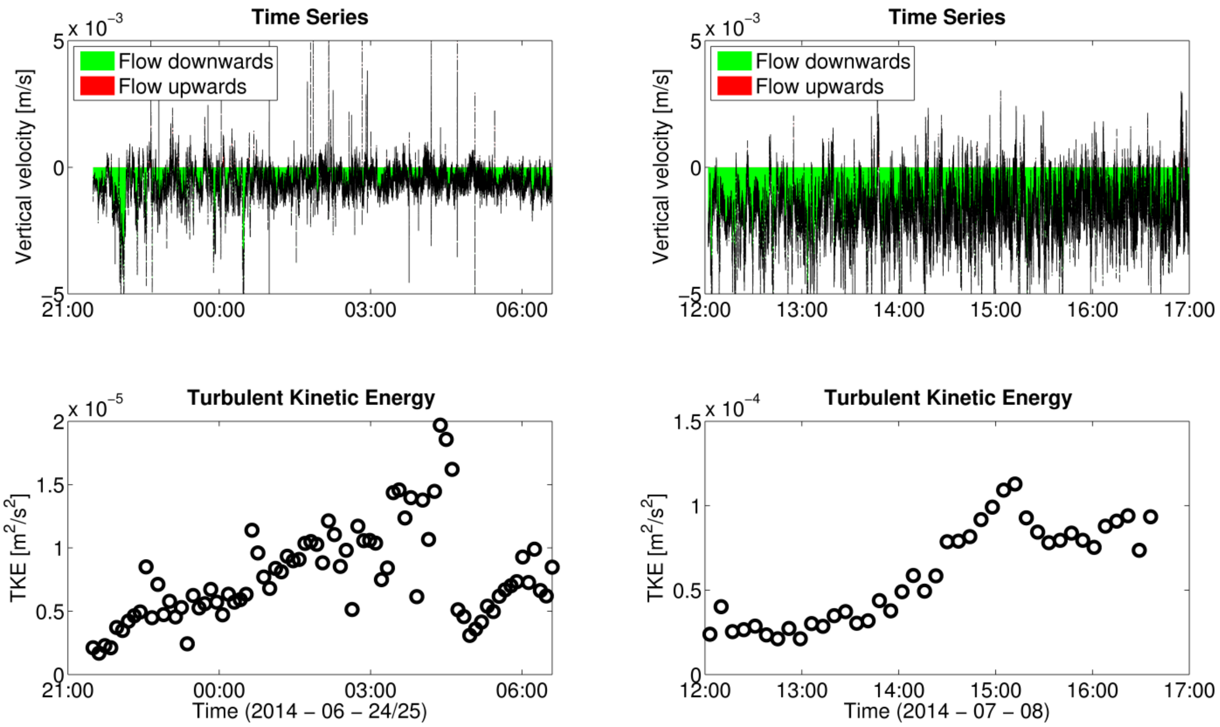

Figure 3-8: Time series of vertical velocity and turbulent kinetic energy during the 24th/25th of June and 8th of July 2014 (ADV Location 1, see Figure 3-1). While the turbulent kinetic energy rose over time and the vertical velocity pointed downward towards the sediments, the oxygen saturation of the bottom waters increased and the soluble reactive phosphorus concentration diminished.

3.4 Discussion

3.4.1 Phosphorus (im)mobilization under different oxic conditions

The results indicate that oxygen conditions in the water column and the sediment surface layer seem to be the most important factors in controlling phosphorus (im)mobilization at our study site. The relationship between oxygen and soluble reactive phosphorus in the water at our study site is not linear (c.f. Conley et al. 2002), rather there seems to be a steep inverse exponential function or an oxygen saturation threshold at 10 % below which phosphorus is released into the water column (Figure 3-2). It is well known that reductive dissolution of iron (hydr)oxides under oxygen deficiency conditions releases adsorbed phosphate anions (Mortimer 1941; Perillo et al. 2009; Puttonen et al. 2014). On the other hand, under oxic conditions dissolved phosphate is readily adsorbed to oxides and hydroxide of iron (Froelich 1988; Kemp et al. 2009; Howarth et al. 2011). Thus, oxygen controls phosphorus release and adsorption through its effect on the oxidation status of iron. As long as a thin oxic sediment layer rich in iron oxides exists, it can bind phosphorus and act as a trap. Under anoxic conditions the iron oxides in the micro-layer are reduced and dissolved, and phosphorus is desorbed into the surrounding pore- and surface waters (“oxygen-control model”; e.g. Caraco et al. 1989; Jensen et al. 1995; Slomp 2011).

Knowledge of the maximum sorption capacity of sediments is vital to understand sediment-water exchanges. Differences in phosphorus sorption isotherms are reflected in the composition of the sediment. Thus the maximum P-sorption capacity within the *Phragmites* wetland depends on the chemo-physical properties of the sediments. Phosphorus sorption in sediments is mainly on amorphous oxyhydroxides of iron and manganese, clays or carbonates (e.g. Boström et al. 1988; Cyr et al. 2009; Lai and Lam 2009). The total amount of iron, manganese and clay is much higher in the basin zone than in the wetland fringe (Table 3-2). Sorption isotherms and sorption kinetics show that the basin sediments are capable of adsorbing large amounts of phosphorus within short timeframes if oxygen supply is sufficient.

An amplifying mechanism of phosphorus release under anoxic conditions is the reaction of sulfate and iron. Sulfate ions are supplied by saltwater input. Under anoxic conditions sulfate is reduced and can react with iron compounds and precipitate as FeS, thus enabling phosphate to be released into the water column (see also Boström et al. 1988; Baldwin et al. 2000; Haraguchi 2012). The kinetic of FeS precipitation is fast. For instance, complete formation of iron sulfide under estuarine conditions and pH 8.5 were measured within 30 minutes by Pyzik and Sommer (1981). The thin oxic surface layer in the basin zone of the *Phragmites* wetland at our study site becomes only sometimes anoxic. The lower sulfur content in the upper two centimeters (0.87 ± 0.12 %) compared to the lower sediment (1.39 ± 0.68 %) might reflect the decreased efficiency of sulfate reduction in the upper sediment layer. Black-colored zones that are associated with iron sulfides often occur in fine-grained sediments of freshwater systems, estuaries and wetlands (Rickard and Morse 2005; Rydin et al. 2011*).* When low oxygen conditions were present in the water column at the study site Dabitz, the lighter brown surface layer disappeared. Redox processes at the sediment surface react fast (within hours) to oxygen pulses (Forster 1996). Similar to our findings, Lehtoranta et al. (2004) noted for the Neva estuary different colors of the oxidized surface layer and the sediment below. Furthermore, there was a significant inverse correlation between soluble reactive phosphorus and sulfate concentrations in the water at our study site (Figure 3-3), indicating a correlation between the reduction of sulfate and the release of soluble reactive phosphorus from the sediment.

3.4.2 The influence of hydrodynamics on the aeration status

Correll (1998) noted that in eutrophic systems shallow waters may become anoxic over a diurnal scale during warm and windless days. In our case, episodic anoxic events were also common during cold temperatures and associated with low water levels and stagnation. Sediments consume oxygen by aerobic decomposition of organic matter, respiration and oxidation of reduced compounds such as Fe^{2+}, H_2S, FeS, and FeS_2, NH_4^+ or Mn^{2+} (Berg et al. 2003). The oxygen uptake by sediments is influenced by transport processes in the overlying water. In general, molecular diffusion and advection are the two transport mechanisms in the water column, but turbulent advection is the dominant process above the diffusive boundary layer (Berg et al. 2003). If no additional oxygen is delivered frequently from the overlying water towards the sediment by turbulent motion, bottom waters and the sediment surface layer may become anoxic.

The results from the Dabitz site show that hydrodynamics may indeed play a major role in the phosphorus release and immobilization processes. Turbulence in the wetland fringe is higher than in the basin zone and accordingly the oxygen saturation and the soluble reactive phosphorus concentrations differ in the wetland zones (Table 3-1). The *Phragmites* stems effectively reduce the turbulent kinetic energy and the basin zone within the wetland serves as an accumulation sink for fine-grained particles rich in phosphorus, iron, manganese and organic matter (see Table 3-2 and Figure 3-7 B). On calm days, neither wind- nor current-induced turbulence transports sufficient amounts of oxygen downward. When the water level is low, the anoxic sediments consume all the oxygen, the oxidized surface layer vanishes and even the bottom waters become anoxic. Iron (hydro)oxides are reduced and the adsorbed phosphates are released. But changes in the oxygen saturation can occur on relatively short time scales, i.e. during few hours. Such short-time fluctuations of soluble reactive phosphorus may be caused by changes in the hydrological and meteorological conditions. Hypoxia in the basins at our study site Dabitz is episodic (see Kemp et al. 2009 for hypoxia classifications) and adsorption of soluble reactive phosphorus is fast once the oxygen status changes.

While wind-induced resuspension is often associated with increased diffusion of phosphorus, sediment disturbance through mechanical mixing and subsequent release of phosphorus into the water column (Boström et al. 1988), it appears to be the opposite at our study site. Wind induced turbulence caused oxygen transport downward leading to the development of an oxidized sediment surface layer. In a study by Horppila et al. (2013) the critical shear velocity for sediment resuspension for organic rich sediments covered by *Phragmites* is 0.0035-0.0055 m/s. This threshold was not exceeded during our measurements and therefore mixing and vertical oxygen transport were the dominant wind-induced processes. In case the critical shear velocity is crossed, either resuspended particles with high binding capacities are capable of immobilizing phosphorus in oxic waters or phosphorus is released from the particles depending on the equilibrium conditions in the surrounding surface waters.

Hypoxia is often related to eutrophication, while hydrodynamic forcing is considered as a factor modifying the ecosystem response to eutrophication (e.g. Conley et al. 2009). In our case, not only the ecosystem response is shaped by hydrodynamics, but hydrodynamics seem also to drive eutrophication: During stagnant periods with low water level, low-turbulence and thus low-oxygen conditions phosphorus from the sediments is released increasing the trophic status of the water.

3.5 Conclusions

A threshold-type behavior could be observed in the relationships between oxygen saturation, water level and soluble reactive phosphorus at our coastal wetland study site. If the water level is low, the deeper anoxic sediments rich in organic matter can consume all the oxygen of the surface layer and the water above, resulting in hyp- or anoxic bottom waters. Once oxygen is depleted, phosphorus is released from the sediments. In many cases hypoxia and anoxia are a result of increased nutrient input with enhanced phytoplankton production and resulting oxygen consumption. This study is an example of a contrary feedback mechanism. We show that oxygen shortages may not be caused by external nutrient input, but by hydrodynamic calm conditions with stagnant waters.

Sorption experiments under aerobic conditions showed that the sediments of the basin zone within the coastal *Phragmites* wetland are capable of immobilizing large amounts of phosphorus within short timeframes if oxygen supply is sufficient. The high sorption capacity is due to an organic-rich, fine-grained sediment with large amounts of iron compounds. These particular sediment characteristics result from the specific local-scale topography and vegetation patterns and the resulting hydrological conditions. The deeper areas inside the *Phragmites* wetland become accumulation basins. Once the oxidized surface layer vanishes, phosphorus is released into the water column. This typically happens during hydrodynamically calm days with stagnant waters. But when turbulent kinetic energy in the water rises, oxygen in bottom waters and sediment surface increases. A thin oxic micro-layer rich in iron and manganese is able to readsorb phosphorus quickly, but due to its redox sensitivity the immobilization is only temporary.

Acknowledgements

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4. Dynamics of surface elevation and microtopography

Karstens, S., Jurasinski, G., Glatzel, S., Buczko, U., 2016. Dynamics of surface elevation and microptopography in different zones of a coastal *Phragmites* wetland. Ecological Engineering 94, 152-163.

Abstract

Surface elevation and microtopography of coastal wetlands are important attributes with respect to the adaptation to sea level change. While the total increase in elevation determines if a wetland can keep up with sea level rise, microtopography impacts, *inter alia*, hydrological patterns, which again drive surface elevation changes. We analyzed the dynamics of surface morphology during the course of one year at five locations along a transect from land (interior zone) to sea (fringe zone) in a coastal *Phragmites* wetland at the Darss-Zingst Bodden Chain (southern Baltic Sea) using a modified Surface-Elevation Table. Landwards this wetland is confined by a dyke while the seaward boundary of the wetland is formed by a micro-cliff. A small strip between the interior and the fringe zone is characterized by small basins (basin zone). Surface elevation changes in the wetland interior are mainly driven by its own productivity. Litter accumulation by the dense *Phragmites* stands is high, but a combination of high organic sediments and water scarcity led to compaction and counteracted vertical accretion. While the surface elevations of the measurement location in the basin zone and one of the locations in the interior zone remained fairly stable (intra-annual changes < ±0.8 mm), the other location in the interior showed a drastic decrease in surface elevation (-3.3 cm year^{-1}). This large deviation may have been caused by the high spatial variability of sediment organic matter. The micro-cliff eroded constantly (-3.7 cm year^{-1}); its steepness and surface roughness increased making it more vulnerable to

the forces of the sea. The micro-cliff protects the wetland interior from flooding but at the same time suppresses allogenic sediment input. The simple approach to use freely available water level data to predict morphological changes in the fringe zone did not yield significant model results. However, the strong intra-annual variability in elevation change especially in the fringe zone showed that short-term changes may often mask long-term trends. Although we cannot predict long-term developments from our relatively short-term measurements, our results indicate that wetland evolution in such squeezed environments is threatened. In dyked wetlands sediment supply from the land is suppressed and lateral erosion at the seaward edge leads to marsh retreat. None of the measuring locations is currently keeping up with local sea level rise.

4.1 Introduction

The surface elevation of coastal wetlands is controlled by a variety of factors including storm activities, subsidence, tides, changes in local hydrology, sediment supply, biotic activities and biomass production (Boumans and Day 1993; Cahoon et al. 2011). Annual elevation change can vary considerably between less than 0.01 and 5 cm per year (Boumans and Day 1993). Coastal wetlands are to a certain degree able to adapt to rising sea levels due to a feedback mechanism of plant production and sediment trapping resulting in vertical accretion (Morris et al. 2002). Climate change scenarios predict a global rise in sea level of 0.2 to 0.6 m or more by 2100 (Nicholls et al. 2007). In order to enable the surface elevation of a coastal wetland to cope with sea level rise, it must accrete vertically at least at the same rate as the local sea level rises (Boumans and Day 1993; FitzGerald et al. 2008). If wetland surface evolution cannot keep up with the rising sea level and in addition human-made constructions or steep landward topography precludes landward migration, wetlands may be threatened and can be completely submerged (FitzGerald et al. 2008; Howe et al. 2009; Reed 1990).

Besides the average surface elevation, also the microtopography – at the scale of a single plant – is important for various ecosystem services. Microtopography in coastal wetlands has large influence on the hydrology, the physico-chemistry including soil nutrients, the habitat variability including vegetation patterns, and thus, the ecosystem functioning (e.g., Moser et al. 2007; Moser et al. 2009; Pollock et al. 1998). Small-scale variations in surface elevation are of importance for the temporal and spatial variability of flooding and water retention. Microtopographic heterogeneity can be caused by sedimentation, erosion, root growth, litter accumulation, animal activities, peat compaction or shrink-swell processes (Cahoon et al. 2011; Moser et al. 2007; Rogers et al. 2006; Vivian-Smith 1997). Measuring the microtopography of densely vegetated wetlands is challenging and many studies described it only qualitatively using descriptors such as hollows, flats or hummock (e.g. Bruland and Richardson 2005; Courtwright and Findlay 2011). Furthermore, most of the studies did not take into account the temporal development of microtopography.

In the Darss-Zingst Bodden Chain, a lagoon system located at the southern Baltic Sea coast, sea level changed at a pace of 0.7 mm per year during the 20^{th} century (Dietrich and Liebsch 2000; Lampe et al. 2007). However, during the last 20 years, sea level rise in the Bodden region increased markedly to about 2.5-3 mm per year (EEA 2014) and regional scenarios including isostatic movements and subsidence expect a sea level rise of at least 0.5 m within the next 100 years (Kliesch et al. 2016). Many of the wetlands bordering the lagoon are confined landwards by dykes. These human-made constructions cause “coastal squeeze” (Doody 2013), meaning that the wetland development is constricted by a fixed boundary landwards and rising sea levels on the other side. The landward boundary can be artificial, formed by anthropogenic barriers such as dykes, or natural, when steep slopes prevent landward migration of wetlands (Doody 2013; Schleupner 2008). Coastal squeeze is a growing concern and threat for wetland evolution in view of the worldwide acceleration of sea level rise (Maynard et al. 2011; Torio and Chmura 2013; Wolters et al. 2005).

Here, we analyze the intra-annual dynamics of surface elevation and microtopography along a wetland transect from land to sea with a Surface-Elevation Table (SET). The SET approach is suitable for densely vegetated environments and was successfully used in a variety of wetlands (e.g. Cahoon et al. 1995; Calvo-Cubero et al. 2013; Rooth and Stevenson 2000). We assume that changes in elevation and microtopography differ along a transect from land to sea. Associated with different wetland zones (fringe, basin, interior), the specific questions addressed in this study are: (1) Can the magnitude and duration of water level fluctuations in combination with wind data be used as a proxy for changes in surface elevation and microtopography at the wetland fringe? (2) Are litter mass and sediment organic matter content the most important factors determining elevation change inside the wetland?

4.2 Material and Methods

4.2.1 Site description

The Darss-Zingst Bodden Chain is a lagoon system at the southern Baltic Sea in Germany (Figure 4-1). It was formed after the Weichselian glaciation during the Litorina transgression (ca. 7000 BC – 0 AD) in an area with glaciogenic basins and meltwater channels (Lampe 1990). It is a shallow lagoon with a mean water depth of 2 m that covers an area of almost 200 km² (Karsten et al. 2003). Salinities increase from 1-3 PSU in the west to 8-12 PSU in the east due to freshwater input and the dominant outflow situation (Selig et al. 2007). Widespread, monodominant stands of *Phragmites australis* (Cav) Trin. Ex Streudel (Common reed) are characteristic for the Bodden coasts. The species can grow well under a large range of salinity conditions (Karsten et al. 2003). The study site Michaelsdorf is characterized by a *Phragmites* wetland that is confined at the landward side by a dyke construction. The land behind the dyke is intensively drained and agriculturally used for sheep grazing (Karstens et al. 2016 a).

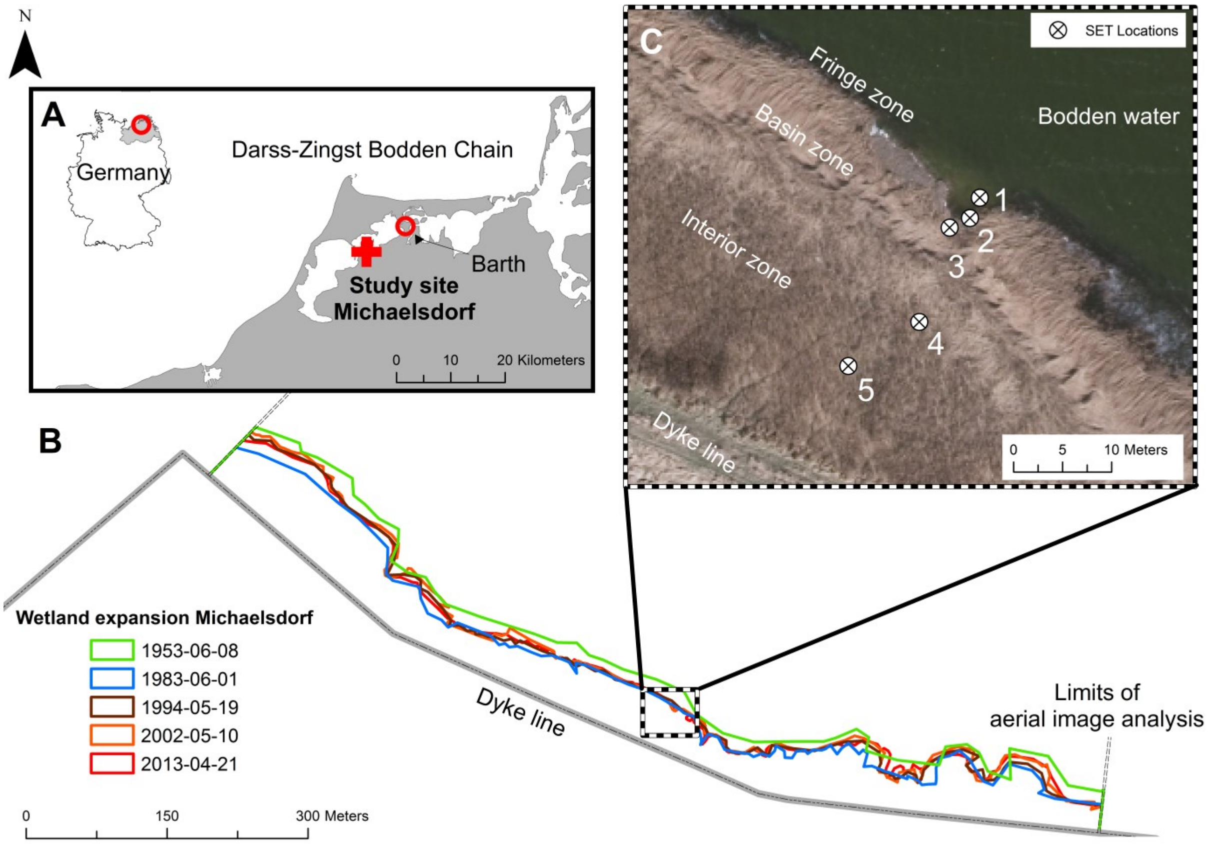

Figure 4-1: (A) Location of the Darss-Zingst Bodden Chain in northeast Germany. (B.) Development of the seaward boundary of the *Phragmites* wetland in Michaelsdorf between 1953 and 2013 derived from aerial images. (C) Five permanent measurement locations for the Surface-Elevation Table were installed in January 2014.

Tidal salt marshes are often divided into low, mid- and high marsh according to the influence of the tidal range (Packham and Willis 1997). Since tides do not exist in the Darss-Zingst Bodden Chain and water exchange with the Baltic Sea is induced meteorologically with inflow situations under strong and persistent northeasterly winds (Selig et al. 2007), the wetland was subdivided into interior, basin and fringe zone, based on water level and hydraulic energy. Surface elevation, microtopography, litter and sediment characteristics were analyzed in these three different wetland zones (Figure 4-1). The hydraulic energy in the constantly submersed fringe zone is high, whereas the basin zone is less frequently flooded and the interior zone is rarely flooded (e.g. Brinson 1993; Karstens et al. 2015a; Lugo et al. 1988). Five permanent measurement locations for the Surface-Elevation Table (SET) were installed in January 2014: SET 1 in the fringe zone, SET 2 covers a micro-cliff that marks the boundary of the fringe, SET 3 behind the cliff in the basin zone, and SET 4 and 5 in the interior zone (Figure 4-1). Already in 2013 a boardwalk was constructed at the study site to minimize disturbances during measurements.

4.2.2 Analysis of aerial images

Wetland expansion was analyzed using aerial images provided by the government office for geoinformation, surveying and cadaster MV for the years 1953, 1983, 1988, 1994, 1998, 2000, 2002, 2005, 2007, 2010 and 2013. To geometrically correct and adjust the aerial images prior to 2002, the aerial image from 2013 was chosen as reference map. Since the natural landscape and especially the coastline is changing in the study region, artificial man-made features, such as crossroads or dyke features, were used for the manual georeferencing process. To define the outer wetland boundary, the aerial images were interpreted by visual inspection. Elements of image interpretation were shape, pattern, tone and texture (Campbell 2002; Lillesand et al. 2008). Visual wetland edge detection with aerial imagery spanning a comparably wide range of years may be prone to bias induced by varying observers and/or varying conditions during shooting of the imagery. We took great care to minimize both sources of possible error: All imagery was analyzed by the lead author of the study and was recorded between the end of April and the beginning of June in the respective years, thus, covering a relatively short period of time. Further, the seaward boundary of wetlands that are dominated by reed plants is usually very good to identify throughout the seasons in aerial imagery, since the dead reed culms are persisting until the new shoots are growing relatively late in the vegetation season.

4.2.3 Sediment and litter analysis

Sediment samples were collected in the three wetland zones (see Figure 4-1). Sediment cores up to 1 m depth were taken as triplicates in each zone in September 2014. For the upper 30 cm a stainless steel corer with 7 cm diameter (*Hydrobios*, *Kiel, Germany*) was used and for 30-100 cm soil augers according to Dr. Pürckhauer (18 mm diameter, Ehlert&Partner, Niederkassel, Germany). Each core was subdivided into 10 cm segments. In addition, sediment surface samples (0-2 cm) were collected in duplicate bimonthly between March 2014 and January 2015. All samples were dried overnight at 105°C. Sediment organic matter was determined gravimetrically by loss-on-ignition (LOI) in a muffle furnace at 550°C for 4h. Total carbon content was measured by combustion in a CNS Analyzer (*Vario Max, Elementar, Germany*). A subset of sediments was wet-sieved to determine the median particle size, percentage of very coarse sand particles (>1 mm) and mud content (% <63 µm) (Selig et al., 2007).

In January 2015 we collected litter samples by harvesting all on-ground litter in three 20×20 cm squares in each wetland zone. However, in the fringe zone, the only zone with standing water at that time, no litter was available. Litter was dried at 60°C for 48 hours to weigh dry biomass. Total carbon content of the litter samples was quantified by combustion in a CNS Analyzer (*Vario Max, Elementar, Germany*).

4.2.4 Weather data and water level fluctuations

Hourly weather data were derived from the nearby DWD (German weather service) station in Barth (Figure 4-1). Water level data were provided by the WSV (Federal Waterways and Shipping Administration) from the official gauging station in Barth (temporal resolution: one minute). Water level fluctuations at the study site were compared with water level fluctuations at Barth using a water level logger (*Solinst*, *Georgetown, Canada*). The logger was installed at the study site between 12th of May 2015 and 1st of June 2015. The level logger data and those from the WSV station in Barth showed a close correlation (Pearson correlation coefficient r_p=0.92, p<0.01). Data from the station in Barth were analyzed for local extrema (second derivative) that can be potentially used as indicators for the intensity of water level fluctuations. Numbers of local minima and maxima of water level, and positive peaks with a prominence of at least 0.3 m were calculated with Matlab 2014b to filter out sharp water level changes. Prominence is a measure of independence and indicates how a single peak protrudes due to its intrinsic height and its position between neighboring peaks (Llobera 2001; MathWorks Documentation 2015). The prominence height [m] and the width at half-prominence [days] were calculated for every peak with a prominence larger than 0.3 m. This threshold was chosen because it represents approximately one third of the total water level range during the study period and the range of long term data from the station Barth between 2000 and 2010. Furthermore, we filtered the water level and wind data for those days where water level reached the upper part of the cliff structure and winds blew on-shore (NNE-ENE: 20-70°).

4.2.5 Surface elevation and microtopography

Surface elevation changes and microtopography were measured bimonthly from March 2014 to March 2015 at five different locations within the coastal wetland using a Surface-Elevation Table (SET) (Figure 4-1). The SET approach was developed by Boumans and Day (1993) for high precision measurements of small-scale changes in elevation in shallow coastal areas and has since been applied in various studies (e.g., Cahoon et al. 2011; Day et al. 2011; Morris et al. 2002). The SET consists of benchmark pipes permanently installed in the ground and a portable leveling device that can be attached to each benchmark pipe. For the benchmark pipes we used round stainless steel (4 cm diameter, 2 m length; EN 10278/h9) and each pipe was driven 1.5 m into the sediment with a sledgehammer. Our modified portable SET device consists of two aluminum plates (23 × 49 cm) with 128 holes (1 × 1 cm; 3 cm apart). The plates are installed exactly one below the other (16 cm distance) to guarantee that the pins are leveled vertically. A screw-slot system ensures that the portable device is always placed at the same position on the benchmark pipe (Figure 4-2). The horizontal position of the SETs was ensured by repeated measurements with a level. For each measurement, 60 aluminum pins were lowered to the sediment surface and the elevation was measured as the distance from the top plate to the top of the pin (see also Morris et al. 2002). Most studies use SETs with 36 pins (e.g. Rooth and Stevenson 2000;

Vandenbruwaene et al. 2011; Whelan et al. 2005). Since we aimed at assessing the microtopography and its changes we chose to use a plate with 128 holes. However, after some testing we recognized that 128 holes would be too time consuming. Therefore, we then used every second hole of the rectangular plates, leading to a total of 60 pins. Since it was difficult to detect the exact sediment surface in the fringe zone with standing water, the pins were dropped under comparable conditions from the same height unto the sandy sediment floor. 3-D visualization of the microtopography was performed with Matlab R2014b based on the 60 data points. Average surface elevation of each SET location is the mean of the 60 pin readings. Changes were calculated as the difference to the relative elevation of the first measurement in March 2014. All measurements were carried out from a boardwalk to minimize disturbances.

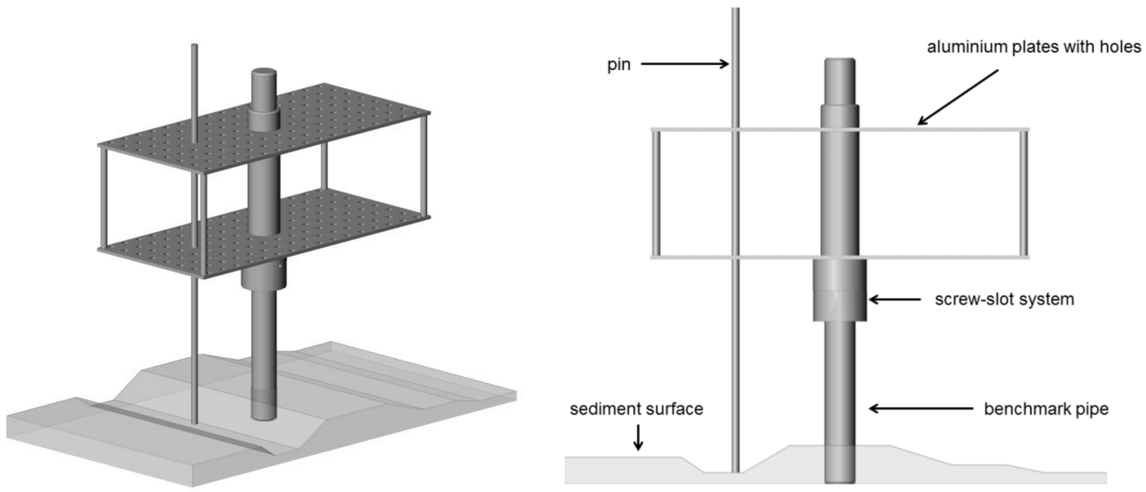

Figure 4-2: Modified Surface-Elevation Table with rectangular plates (technical drawing: Lukas Jobst).

To quantify the microtopography, two different indices were used: '*Random Roughness'* and '*Tortuosity*' (e.g. Kamphorst et al. 2000; Moser et al. 2007). Random Roughness is the standard deviation of all pin readings ($\sqrt{\frac{\sum(x_i-x_\mu}{n-1}}$). This index describes the height distribution well but does not allow any conclusions regarding the spatial distribution of higher or lower points (Kamphorst et al. 2007). Kamphorst et al. (2007) compared 11 indices and showed that random roughness is the most suitable indicator for water storage in local depressions.

To add a second microtopography index we chose the frequently used tortuosity. Tortuosity is the ratio of the summed point-to-point distances of the measured horizontal SET lines and the corresponding planar transect distance of 49 cm $((\sum\sqrt{((x_2-x_1)^2+(y_2-y_1)^2+(z_2-z_1)^2)})/49)$. Tortuosity is an indicator for sediment surface roughness and relief (Moser et al. 2007; Wolf et al. 2011).

4.3 Results

4.3.1 Wetland development

Analysis of the aerial images shows that the wetland extent at the study site decreased by about 25% after the dyke construction which took place in the early 1970ies (Figure 4-1, area in 1953: 67404 m²; area in 1983: 50425 m²). Between 1983 and 2000 the wetland expanded again seawards by 16 % and the analyzed wetland area reached 61461 m², followed by a slower retreat since 2000 (56191 m² in 2013). In 1953 the wetland edge was located up to 10 m further into the water than in 2013. In 2000, the peak of the recovery phase after dyke construction, the wetland edge was located 5 m further into the sea than in 2013.

4.3.2 Sediment and litter characteristics in different wetland zones

In the basin and interior zones dry soil bulk densities of the upper 2 cm were below 1 g cm^{-3} throughout the year (Figure 4-3 A). Organic matter content and mean carbon concentrations in the surface layer were highest in the interior and lowest in the fringe zone (Table 4-1). In contrast to the other zones, organic matter and dry bulk densities showed temporal fluctuations in the fringe zone (Figure 4-3 A). Median grain size and amount of coarse particles (> 1 mm) were highest in the fringe zone, whereas mud content (<63 µm) was highest in the wetland interior (Table 4-1).

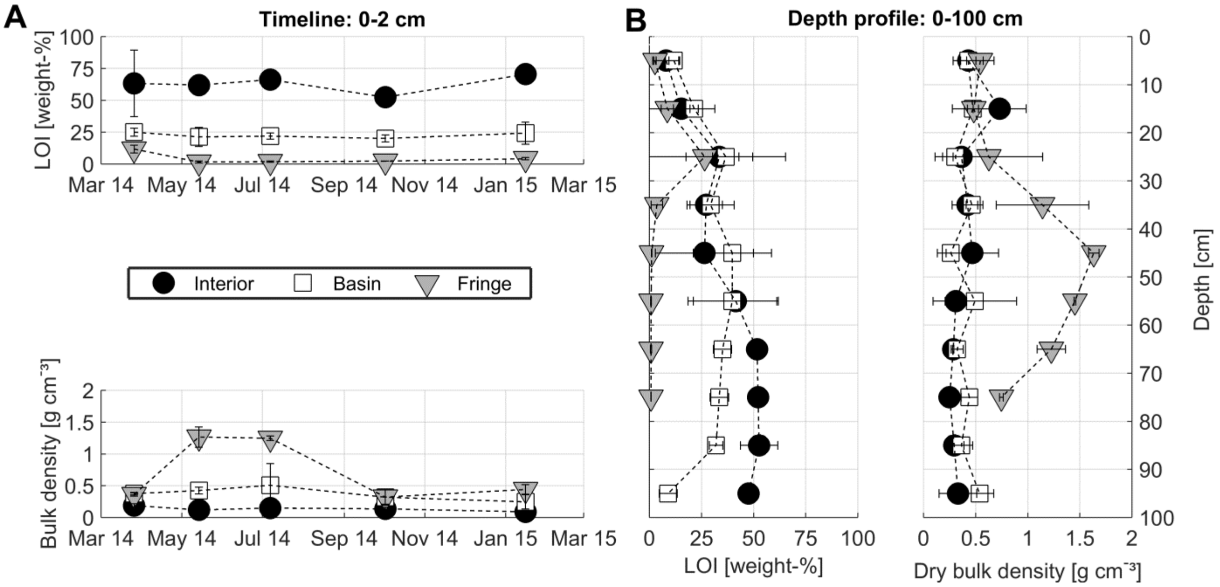

Figure 4-3: (A) Timeline of organic matter (LOI= loss on ignition) and dry soil bulk density for sediment surface layer (0-2 cm) in the interior, basin and the fringe zone. (B) Organic matter content and dry soil bulk density for sediment cores up to 1 m depth. Symbols represent mean values, error bars standard deviations.

In the sediment cores organic matter increased with depth inside the *Phragmites* wetland (Figure 4-3 B). While inside the wetland soil bulk density remained below 1 g cm^{-3}, in the fringe it increased below a depth of 25 cm.

Litter mass in the wetland interior was 1080±104 g m^{-2} in January 2015, whereas in the basin zone it was only 353±171 g m^{-2}. Although some individual *Phragmites* plants grew in the fringe zone (SET 1), no litter was left there on the sediment floor in January 2015. Carbon concentrations of the litter material did not differ between the interior (44.6±0.2 %) and the basin (44.8±0.2 %) zone.

Table 4-1: Sediment properties of the top soil (0-2 cm) in the different wetland zones. For dry bulk density, organic matter (loss-on-ignition) and carbon content annual means (n=10). For grain size characteristics n=1.

Zone	**Density**	**LOI**	**C**	**Median grain size**	**Very coarse sand**	**Mud**
	g/cm³	%	%	µm	% >1mm	% <63µm
Interior	0.13 ± 0.04	62.9 ± 11.3	31.1 ± 7.3	135	9	46
Basin	0.37 ± 0.16	22.5 ± 4.6	11.7 ± 2.6	147	13	35
Fringe	0.73 ± 0.46	4.5 ± 4.4	2.2 ± 2.1	588	33	19

4.3.3 Weather data and water level fluctuations

Mean wind speed was highest between November and January (Table 4-2), but storms (> 21 m s^{-1}) did not occur. Winds blew in an on-shore direction only during 30 days, whereof during 25 days the water level reached the top of the micro-cliff at the wetland edge (Table A-5 in the appendix). The mean temperature during the study period was higher than the long term average of 8.6°C (1981-2010; Climate Data Center DWD). A stable ice layer did not develop. Total precipitation between March 2014 and March 2015 (586 mm) was less than the long term mean of 657 mm (1981-2010; Climate Data Center DWD) and no heavy rain events (>20 mm within 6 hours according to DWD) occurred.

Table 4-2: Weather and hydrological data. *Intervals were structured according to SET measurements: A= 18th of March - 15th of May, B= 15th of May - 7th of July, C=7th of July - 3rd of September, D= 3rd of September - 17th of November 2014, E= 17th of November 2014 - 16th of January 2015, F= 16th of January 2015 - 27th of February 2015. ** 90°=east, 180°=south. * normalized to a 30-days period. ****Peaks= Number of local extremes with a prominence larger than 0.3 m were calculated for each measurement period and normalized to a 30-days period to guarantee comparability.**

Intervals*	**Mean temperature**	**Wind speed**	**Wind direction** **	**Monthly precipitation** ***	**Water level**			
	°C	m s^{-1}	°	mm	Min	Mean	Max	Peaks****
Total	10.4	4.36	185	50.1	-0.51	-0.07	0.31	9.7
A	8.6	4.49	178	43	-0,37	-0,08	0,24	8.3
B	15.9	3.81	192	33.8	-0,24	-0,09	0,26	11.5
C	17.2	4.1	169	68.1	-0,35	-0,05	0,23	6.8
D	10.5	3.59	163	28.2	-0,28	-0,04	0,19	7.7
E	3.3	5.64	202	101.6	-0,51	-0,15	0,22	12.7
F	0.9	4.36	217	27.5	-0,21	0,07	0,31	10.2

Mean water level at the WSV station in Barth during the whole study period was 0.07 m AMSL. SET 4 in the interior of the *Phragmites* wetland was flooded for only two days in January 2015 (Figure 4-4). Prominence heights and widths (half-prominence) did not differ significantly between the SET measurement intervals (p=0.36, p=0.57) (Figure 4-4). Days where the water level reached the top of the micro-cliff and winds blew on-shore occurred comparably often during the first (A) and third (C) SET measurement interval (Table A-5 appendix).

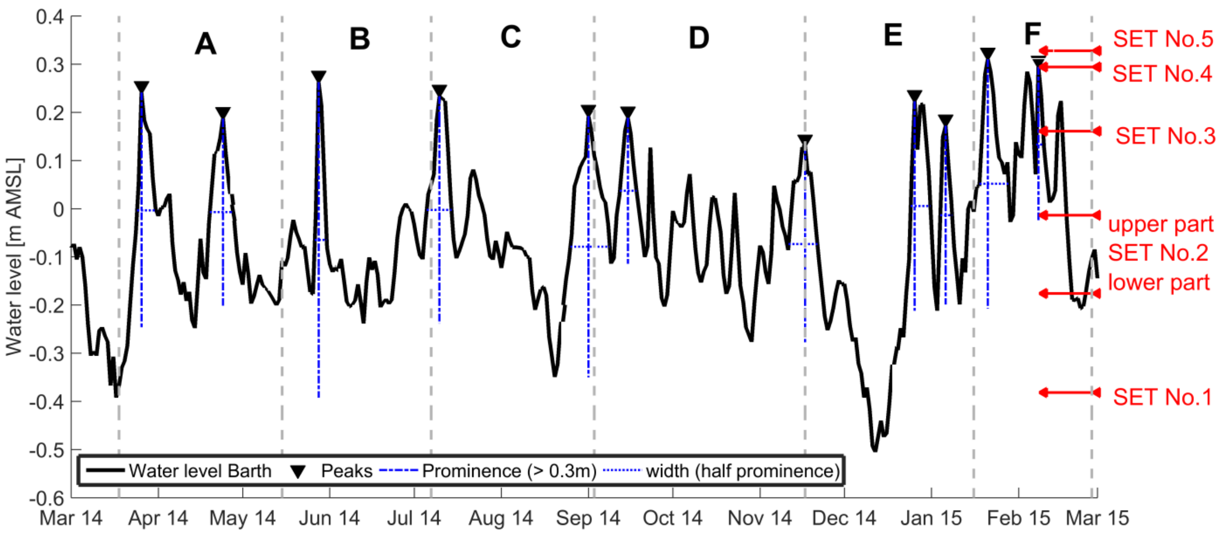

Figure 4-4: Water level (m above mean sea level) measured at the station Barth (black) and peaks with a prominence of at least 0.3 m. Red arrows on the right side show the elevation of the SET measurement locations. Vertical grey dotted lines indicate dates of SET measurements. Labels A-F represent measurement intervals (see Table 4-2).

4.3.4 Surface elevation changes in different wetland zones

The development of wetland elevation microtopography varied between the five locations. Intra-annual changes were larger compared to total annual changes for SET locations 1, 3 and 5 (see Figure 4-1 for locations).

The first SET-spot was located right in front of the *Phragmites* wetland at the fringe (Figure 4-1). A small-scale hollow structure (~ 460 cm²) was clearly visible when measurements started and remained (Figure 4-5 A), but overall the relief became more balanced and tortuosity decreased significantly (Pearson correlation coefficient r_p= -0.77, p=0.04). Random roughness was highest in March 2014 and lowest in January 2015 (Table 4-3), but there was no clear trend over the year (r_p= -0.34, p=0.4).

Table 4-3: Microtopography indices (T=Tortuosity and RR= Random Roughness) for all five SET locations over the study period March 2014 - March 2015.

	SET 1		SET 2		SET 3		SET 4		SET 5	
	T	RR	T	RR	T	RR	T	RR	T	RR
March	3.9±0.3	26.1	5.8±0.5	74.3	1.8±0.4	7.9	3.2±0.4	17.9	3.2±0.8	15.6
May	3.2±0.5	23	6.3±0.6	79.8	2.4±0.5	9.6	4.5±0.7	21.7	3.3±0.9	17.2
July	3.1±0.5	23.3	6.9±1.3	83	2± 0.2	9.2	4±0.7	19.9	3±0.6	12.5
September	2.8±0.4	24.	6.9±1	84.7	2.3± 0.3	10.8	3.9±0.7	21.1	2.6±0.4	12.1
November	3.2±0.8	25.4	5.6±0.4	82.5	1.6±0.2	7.5	3.1±0.5	19	3.8±0.8	17.2
January	2.5±0.4	21.9	6.2±0.8	87.4	2.1±0.2	9.8	3.7±1	20.3	3.9±0.6	18.5
February	2.83±0.6	23.7	5.8±0.5	86.9	2.2±0.4	10.6	3.8±0.8	17.6	4.1±1	18.3

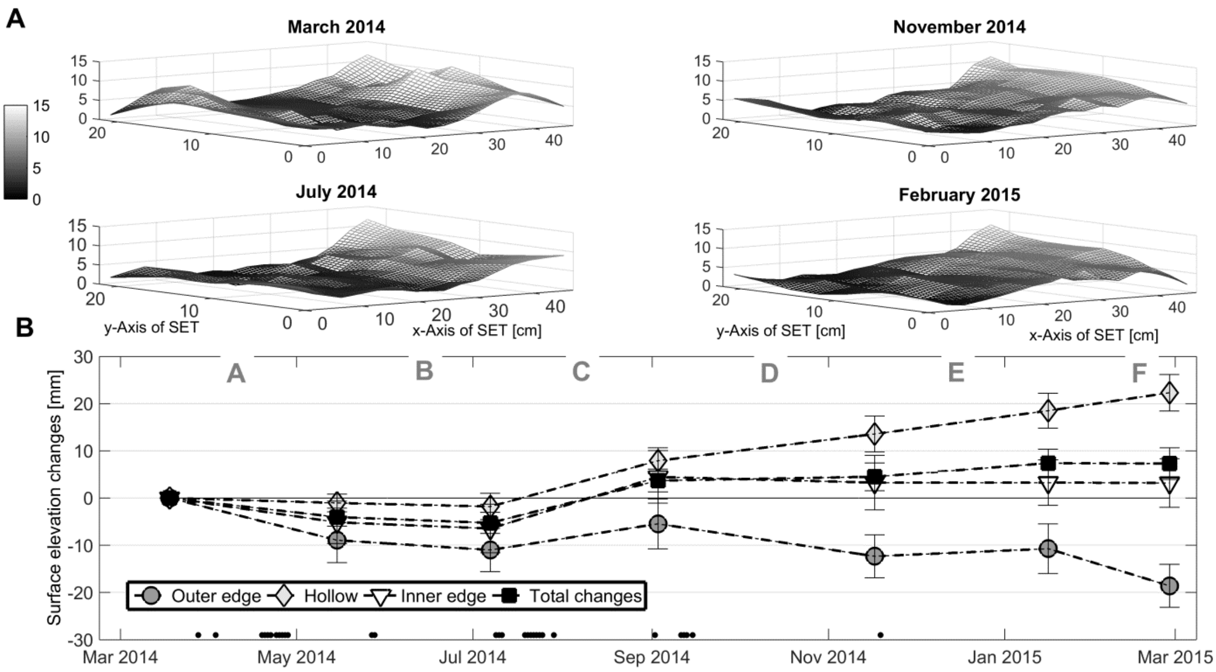

Figure 4-5: (A) 3-D model of microtopography of SET 1 for four different months. Relative elevation is displayed as the distance from the lowest point of the relief [cm]. (B) Surface elevation changes [mm] over time for SET 1. Symbols represent the means of the 60 pins and error bars the standard error. Black dots on the bottom of the timeline show days with wind blowing on-shore.

A differentiation of the SET array into “*outer edge*” (the side of the relief facing the open water and being mostly exposed), “*inner edge*” (the side pointing towards the wetland) and “*hollow*” showed that the surface elevation of the outer edge decreased over the study period, while the surface of the hollow in between the edges increased (Figure 4-5 B). The positive surface elevation trend for the hollow and the negative trend for the outer edge is clearly visible when summing up the changes of surface elevation for each measurement point of the relief (Figure 4-6).

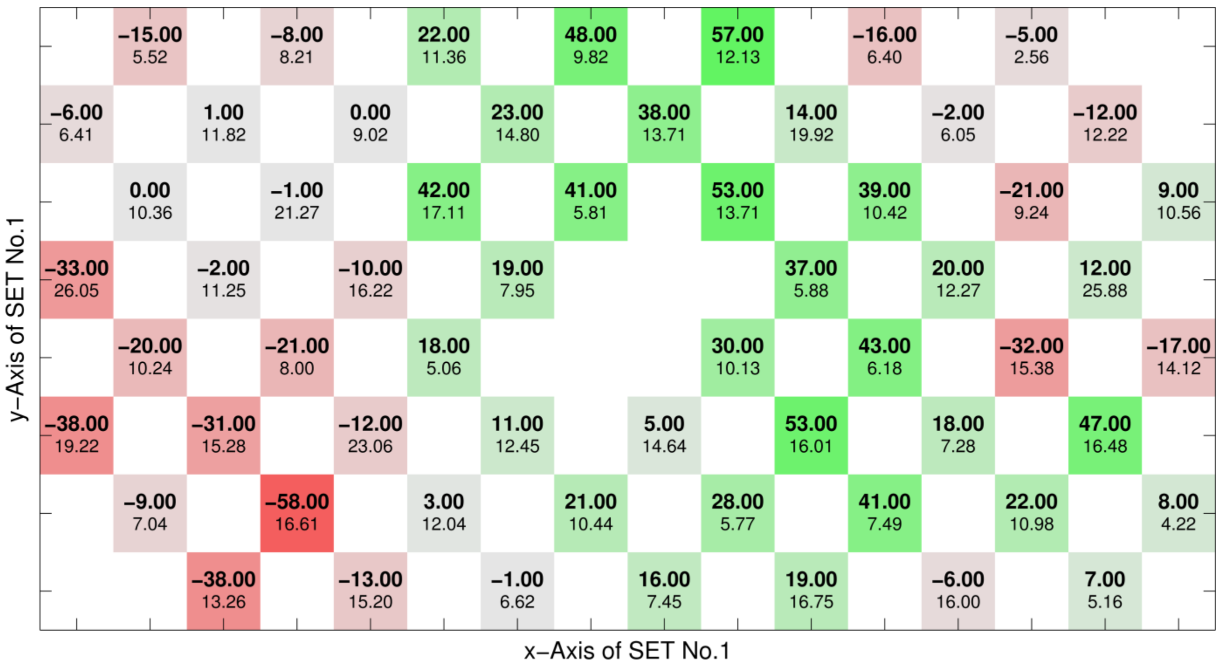

Figure 4-6: 3-D matrix of measurement results of SET 1. Colors indicate if surface elevation increased (green), decreased (red) or remained stable (grey). Bold numbers indicate the total changes [mm] in surface elevation between March 2014 and March 2015, and numbers below show the standard deviations [mm] of the elevation changes between the bi-monthly measurements during that period.

The second SET recorded elevation changes of the micro-cliff at the wetland fringe that marks the seaward boundary of the wetland. Such micro-cliffs are typical features in coastal wetlands and are exposed to waves and currents (e.g. Feagin et al. 2009; Möller et al. 2011; van der Wal and Pye 2004). The overall surface elevation trend for the micro-cliff was -3.7 cm year^{-1} (R^2=0.95), but with greater elevation decrease in the lower and middle part than in the upper part of the cliff structure (Figure 4-7 B). Total erosion was strongest during the first measurement interval (A), when winds blew comparably often in an on-shore direction (see Table A-5 in the appendix). However, on-shore winds were even stronger during the third interval (C), where total erosion was lowest. Only the lower part of the micro-cliff eroded distinctly during this period.

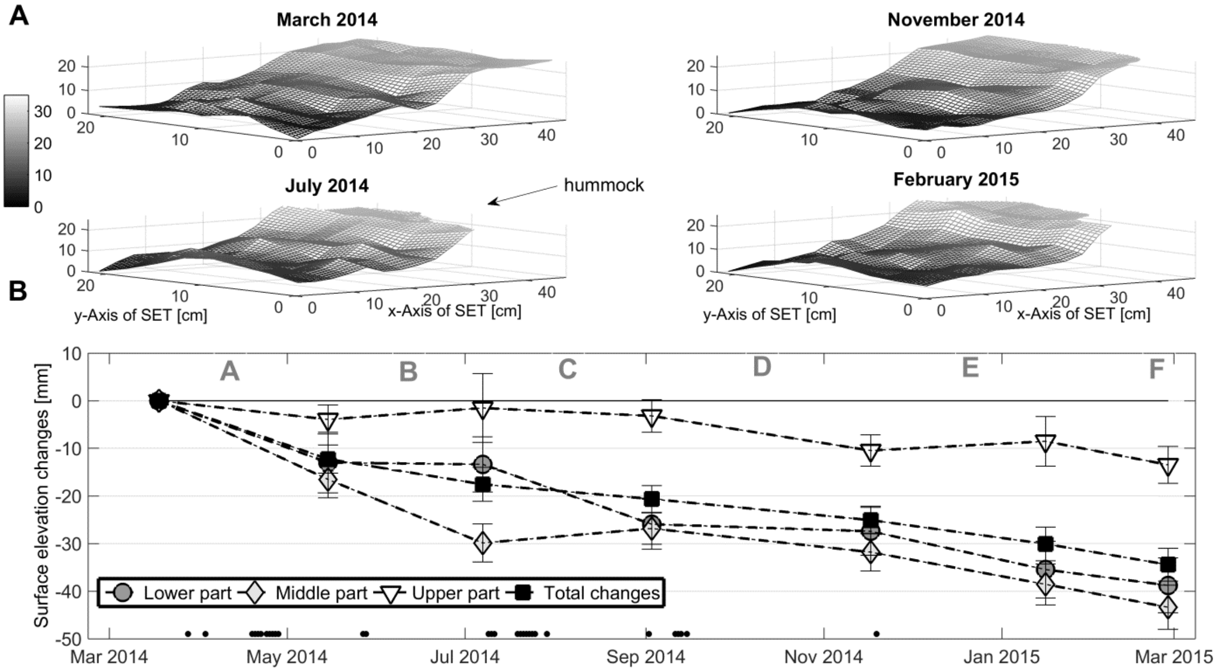

Figure 4-7: (A) 3-D models of microtopography [cm] and (B) surface elevation changes [mm] over time for SET 2. Black dots on the bottom of the timeline show days with wind blowing on-shore.

While *Phragmites* stems grew on top of the micro-cliff, the bottom part was bare. In July 2014 emerging *Phragmites* shoots created small individual hummocks at the top of the micro-cliff that were visible in the field and in the SET measurements (Figure 4-7 A). During this period tortuosity was also highest. Random roughness increased over the study period and was significantly correlated with the changes in elevation (r_p= -0.95, p=0.0001).

Directly behind the micro-cliff (SET 3) and in front of the dyke (SET 5) surface elevation changes were minor (< ±1 cm) during the one-year period (Figure 4-8). In March 2015 the mean surface elevation was back to the values of March 2014. At SET 3, soil elevation was highest in September 2014, the time of the vegetation maximum. In front of the dyke (SET 5) no such peak occurred. Random roughness and tortuosity were lowest for SET 3 compared to all other SET locations (Table 4-3).

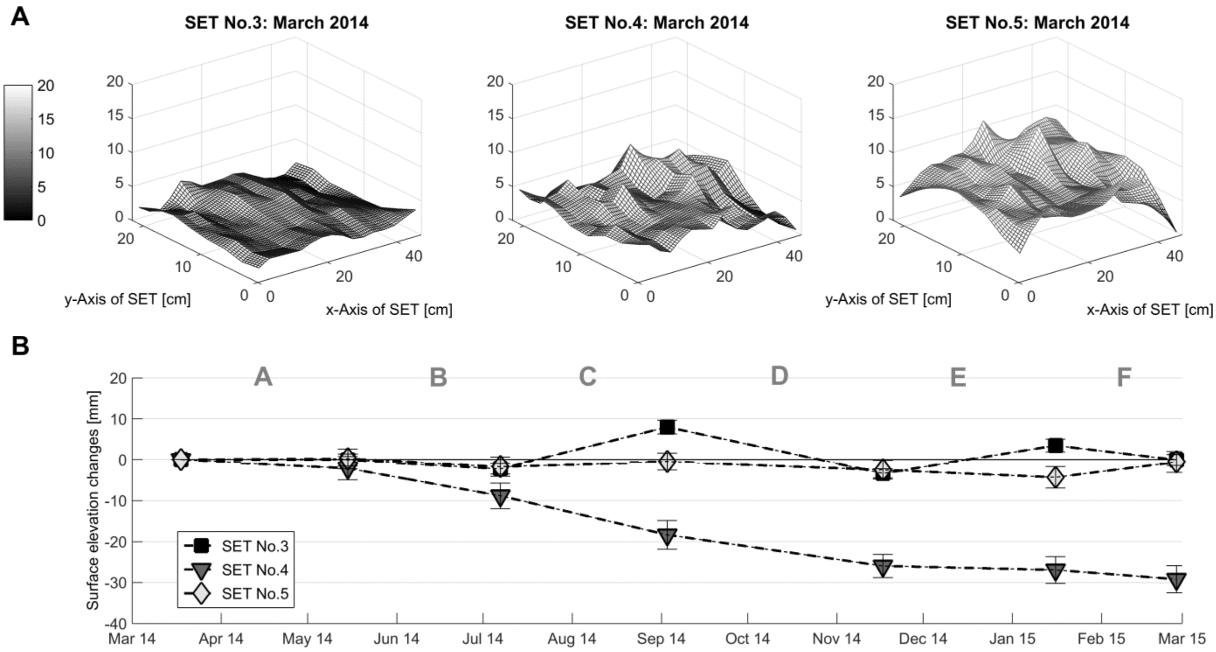

Figure 4-8: (A) 3-D models of micro-topographies [cm] for SET 3, 4 and 5 in March 2014 and (B) surface elevation changes [mm] over time.

SET 4 in the middle of the *Phragmites* wetland stands out because a continuous decrease of surface elevation was measured throughout the year (Figure 4-8), with a mean of -3.3 cm year^{-1} (R^2=0.96). The elevation decrease was strongest between May and November (slope gradient of SET 4= 0.16; Figure 4-8 B). There was a significant correlation between the decrease of mean surface elevation and average temperature during each measurement interval (r_p= 0.87, p=0.02, n=6). Surface elevation increased only at 8 out of 60 measurement points and standard deviations between the measurements were high (Figure 4-9).

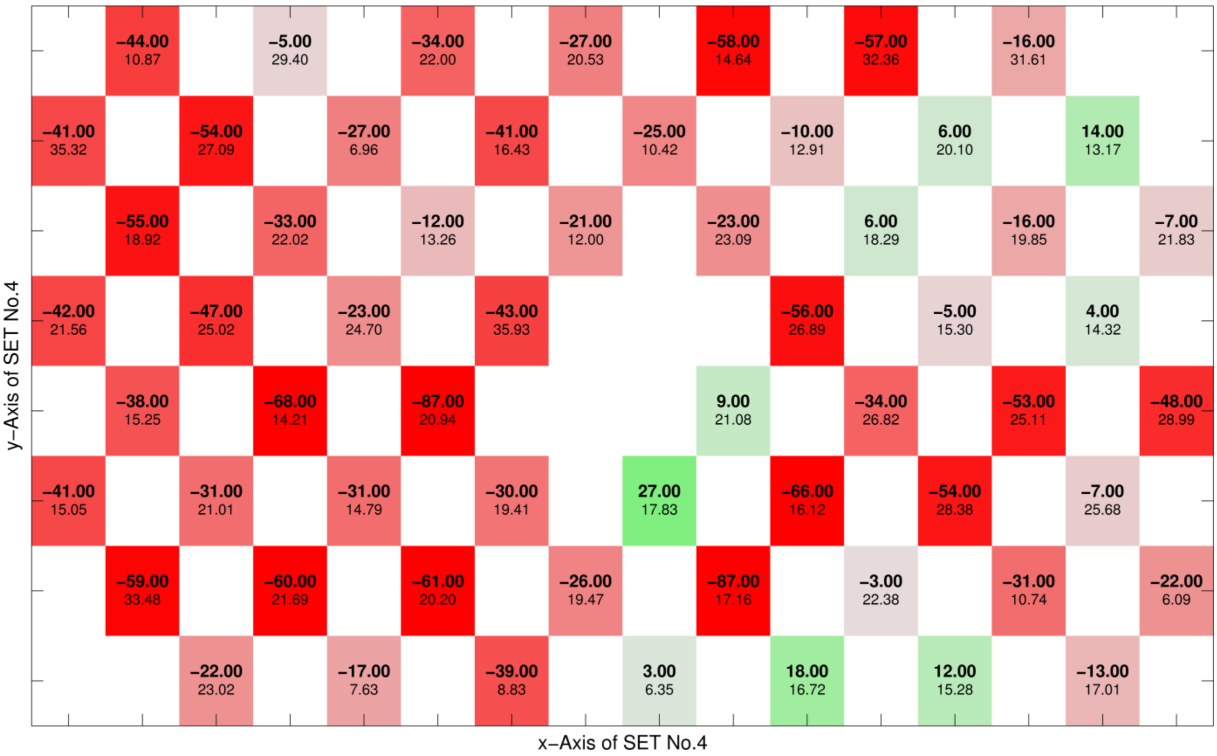

Figure 4-9: 3-D matrix of measurement results of SET 4. For explanations see Figure 4-6 caption.

4.4 Discussion

4.4.1 Elevation changes at the wetland fringe

The micro-cliff at SET 2 became steeper during the study period and overall surface elevation decreased with a rate of -3.7 cm year^{-1}. There are very few elevation change rates reported for marsh cliffs in the literature. However, our rates are comparable with a rate reported from a cliff-like *Spartina alterniflora* marsh edge on Galveston Island, Texas (~-5 cm year^{-1}, Feagin et al. 2009). In our data, total surface elevation decreased linearly and water level fluctuations did not differ significantly between the measurements. This suggests that the micro-cliff eroded constantly. Mass failures along the wetland scarp as described by Francalanci et al. (2013) and Bendoni et al. (2014) were not observed. Yet it seems rather likely that extreme events in the form of heavy storms (which did not occur during the study period) with wind-waves may lead to abrupt morphological changes. Already the presence of cliff-structures indicates that the wetland can be exposed to higher incident waves: Möller at al. (2011) interpreted the occurrence of cliffs in the transition zones of *Phragmites* wetlands as a morphological signal related to higher wave energy compared to sheltered sites. Random roughness and steepness of the micro-cliff at our study site increased, leading to a higher exposition seaward. Wind-waves play a key role in the lateral erosion of wetland edges and can induce mass failure processes (Bendoni et al. 2014). Furthermore, vertical water level changes due to their influence on wave energy impact marsh edge erosion (Möller and Spencer 2002; Singh Chauhan 2009). Initially scouring and the differential accretion between vegetated surface and bare sediment trigger cliff formation, but wave erosion becomes more important once the cliff height increases. This is due to the greater water depth in front of the cliff which reduces wave attenuation at the marsh edge (Singh Chauhan 2009). The combination of on-shore winds and water level heights that reach the upper part of the cliff and hold on for at least three consecutive days occurred particularly often during the first and third SET measurement interval (see Table A-5 in the appendix). Total erosion of the micro-cliff was strong during the first interval, but it was lowest during the third interval where on-shore winds blew even stronger. However, storm conditions which would generate higher incoming waves, did not occur during our study period and the relationship between high and long lasting water levels, on-shore wind and cliff erosion might be more pronounced under higher wind speed conditions.

The role of vegetation along the wetland edge is discussed controversially in the literature: On the one hand, there is evidence that the presence of vegetation along the wetland edge delays mass failures because the root system stabilizes the upper sediment layer and tension cracks are smaller (Francalanci et al. 2013). On the other hand, laboratory and field experiments by Feagin et al. (2009) showed that vegetation did not prevent erosion along cliff-like marsh edges and plants may even amplify erosion when wave-induced movements of roots protruding out of the cliff release sediment. In our study, we observed such root movements at the micro-cliff (Figure 4-10). However, vegetation can indirectly mitigate wetland edge erosion by modifying soil parameters on a long-term, and thus, may contribute to the adaption to gradual changes such as sea-level rise (Feagin et al. 2009). Vegetation can trap fine particles and in combination with plant detritus the sediment becomes less coarse and consequently more cohesive and resistant to erosion (Feagin et al. 2009).

Figure 4-10: Roots sticking out at the wetland edge at our study site Michaelsdorf. Root movement could potentially enhance erosion along the micro-cliff.

Although the micro-cliff at SET 2 eroded constantly, the surface elevation in front of the micro-cliff at SET 1 did increase only slightly with large intra-annual variations. Thus, the eroded material was not deposited in the direct vicinity but transported further. Möller and Spencer (2002) showed that wave attenuation is stronger at micro-cliff sites than at smooth marsh transitions, therefore micro-cliffs provide an effective protection for the marsh surface landwards (SET 3), but wave energy is increased in front of the micro-cliff (SET 1). The micro-relief in the fringe zone became smoother. This development was not continuous; rather a major disruption occurred between May and July, when overall surface elevations decreased. During this period bulk densities were higher and organic matter content lower

(Figure 4-3 A). Feagin et al. (2009) suggest that erosion rates may increase considerably with bulk densities > 0.9 g cm^{-3} (as observed in May and July). Higher bulk densities due to the presence of coarse particles make the sediment more vulnerable to erosion (Feagin et al. 2009). The percentage of very coarse sand was higher in the fringe zone than in the other wetland zones (Table 4-1). During this erosive period, on-shore winds blew only during two days, but comparably strong (Table A-5 in the appendix). Furthermore, the number of water level peaks was high between May and July (Table 4-2). However, we found no statistical proof that water level fluctuations are impacting the surface morphology in the fringe zone although surface elevation showed large intra-annual changes: Regressing elevation changes against the number of peaks, the height and width of peak prominence, mean water level as well as wind direction and speed, including interaction terms, did not yield significant models.

Coastal wetland retreat and changes in morphology are strongly influenced by the lateral erosion of wetland edges (Bendoni et al. 2014). Aerial image analysis showed that the wetland at Michaelsdorf is currently retreating. Under exposed conditions cliffed edges are not a sustainable morphological configuration and marsh retreat is probable (Möller and Spencer 2002). It would be of interest to continue monitoring the changes of the wetland edge and elucidate the role of possible storm events in the future.

4.4.2 Elevation changes in the basin and interior zones

SET 3 to 5 did not show a spatial gradient of surface elevation changes although they were arranged along a transect from the basin zone to the wetland interior. Whereas annual changes at SET 3 and 5 were minor, surface elevation decreased strongly over the year at SET 4. Therefore, and because development through time was comparable at SET 3 and 5, we first address these SET locations and then SET 4 in some detail.

Intra-annual surface elevation changes for SET 3 and 5 were higher than the net total changes over the whole measurement period (max: +8 mm, min: -4 mm). These fluctuations are comparable to intra-annual elevation changes in marsh sites covered by *Bolboschoenus maritimus* and *Puccinellia maritima* (+7mm; -2 mm) reported by Maynard et al. (2011). In *Phragmites* wetlands of Chesapeake Bay, Rooth and Stevenson (2000) found even higher elevational increases (up to +9.5 mm in six months), with *Phragmites* sites increasing stronger than *Spartina* sites. The authors explained these high rates with the high productivity of *Phragmites* and the absence of litter export in the mid-marsh regions. Approximately 50 % of the accreted material inside *Phragmites* wetlands may be organic (Rooth and Stevenson 2000), resulting from the litter input, while the fringe zone is affected by wave action and dead material is regularly exported. In mangrove forests, several studies found positive relationships between litter standing stocks and vertical accretion for the inland sites, while no relationship was found for sites bordering the open water (Twilley et al. 1986; Middleton and McKee 2001; Cahoon et al. 2006). Although mangroves are tree species dominated, we assumed a similar relationship between litter mass and elevation change in reed wetlands. Even though biomass and stem density are comparable at the basin and interior zone (Karstens et al. 2016), litter mass was significantly higher at SET 5 than SET 3, indicating that litter might be exported from the basin zone. Thus, we would expect larger elevation increase in the interior zone. However, the elevation changes did not differ between the two locations. It could be that more litter was incorporated into the sediment at SET 5, but compaction counteracted the vertical accretion (see below). Organic matter content in the interior and basin zones of the *Phragmites* wetland show a similar depth profile (Figure 4-3 B) which could reflect the historical development. According to our aerial imagery analysis, SET 3 was not situated directly behind the micro-cliff but 10 m further inland in the recent past (Figure 4-1). Because SET 3 was then rather at an interior location, litter production and input were very likely higher, which is reflected in higher organic matter contents of deeper sediment layers at SET 3.

Although SET 4 is situated between SET 3 and 5 in the interior zone, it showed a strongly different behavior with a drastic decrease in surface elevation. Since neither storms nor heavy rain events occurred during the study period, it is unlikely, that this decrease was driven be erosion. Even during storms heavy erosion at this part of the interior zone is rather unlikely since reed stands are an effective wind-buffer and suppress aeolian sediment transport (Karstens et al. 2015b). Thus, strong compaction is the more likely explanation for the decrease in surface elevation, which has been shown in previous work as well: Rogers et al. (2006) presented results, which suggest that elevation increases due to vertical accretion and litter incorporation into the sediment can be offset by decreases due to compaction. Compaction of organic matter and loss of porosity was also suggested by Cahoon et al. (1995) as one possible driver for decreases. According to Knott et al. (1987) the compressibility of salt marshes is directly related to their organic matter content. Since the top soil at SET 4 is highly organic (>50 %) and organic matter remains high down to 1 m depth, this supports the idea of compaction driven decrease of elevation. Drought related soil water losses can amplify compaction processes (Cahoon et al. 2011; Rogers et al. 2006). In our case, the study period was warmer and drier than the long term average of 657 mm and also previous years had less precipitation (study period 2014/03-2015/03: 586 mm; 2014: 551 mm; 2013: 574 mm; 2012: 547 mm). At SET 4 the surface elevation decreased by 23 mm during the warm and dry summer months. In several other coastal wetlands drought related lower water tables resulted in decreases of up to 20 mm (Cahoon et al. 2006). Such developments can be explained by a water storage mechanism known as dilation storage (Cahoon et al. 2006). When sediments dry, the volume shrinks and so does the surface elevation (Nuttle et al. 1990). Also the microtopography impacts the sediment water content and thus compaction processes: Depressions reduce runoff after precipitation. Random roughness as an indicator to predict maximum depressional storage was even higher at SET 4 than at SET 5 and ranged between 18-22 cm. According to Kamphorst et al. (2000) this corresponds to ~6 mm maximum depressional water storage capacity. However, random roughness does not have a spatial component and predictive models for depressional storage based on roughness can have uncertainties of up to 3 mm (Kamphorst et al. 2000). Another important mechanism controlling water storage can be evapotranspiration by vegetation (Cahoon et al. 2006). By removing soil water, evapotranspiration can lead to a collapse of pore spaces and daily changes in evapotranspiration can cause shrink-swell processes of salt marshes (Nuttle et al. 1990). Also Paquette et al. (2004) presented evidence that plant transpiration caused elevation changes in salt marshes. Evapotranspiration in *Phragmites* wetlands shows a seasonal trend (Zhou and Zhou 2009), and for SET 4 there was a significant negative correlation between temperature and elevation change. However, plant transpiration alone cannot explain why the surface elevation of SET 4 decreased continuously, whereas the elevation at SET 5 remained the same. As discussed above, the large spatial variability of organic matter in the top 50 cm of the sediment samples taken around SET 3, 4, and 5 (mean coefficient of variation= 49 %, Figure 4-3 B) may provide a possible explanation for the strong variation in surface development since organic matter is a strong driver of compaction. Therefore a small shift in location of the SET may have a strong influence on the results.

Vertical accretion has to be equal or greater than the sum of total subsidence and local sea level rise for a coastal wetland to thrive (Cahoon 2015). Local sea level rise is currently 2.5-3 mm per year in the Bodden region (EEA 2014) and none of the SET measurement locations can keep up with this rate. The wetland interior very likely depends on its own productivity for vertical accretion because considerable sediment supply from the lagoon as well as from land seems to be minor for different reasons: Sediment supply from the lagoon side is minor because the interior wetland is rarely flooded and sediment supply from land is suppressed by the dyke. In case of allogenic input deficiency, autogenic input is crucial for marsh survival with regard to rising sea levels (e.g. Calvo-Cubero et al. 2013; DeLaune and Pezeshki, 2003; Kirwan and Guntenspergen, 2012). In addition, due to the intensive drainage network behind the dyke the groundwater gradient at our study site goes from sea to land and not vice versa (Kliesch et al. 2016). It seems likely that this human-induced groundwater gradient promotes the lowering of the water table in the wetland. Infrequent flooding of the interior zone (Figure 4-4) leaves precipitation as the main water source and accretion to be solely driven by the input of organic matter via litter in this squeezed coastal wetland.

4.5 Conclusions

Phragmites is a bio-engineer of its own environment and capable to accrete vertically if plant production and thus litter production is high. Accretion can be further increased by the sedimentation of particles supplied from sea or land which are captured in the reed stands. However, none of the SET measuring locations can currently keep up with the local sea level rise. While the surface elevation of SET 3 and 5 remained stable but did not increase, SET 4 showed a drastic decrease in surface elevation. Also at the micro-cliff erosion prevails and the cliff is becoming steeper. Solely the surface of SET 1 in front of the micro-cliff is increasing, but also there intra-annual changes were large and the increase not sufficient. We could not use the magnitude and duration of water level fluctuations as a proxy for changes in surface elevation and microtopography in the fringe zone. Our data also does not provide evidence that the amount of litter mass determines elevation changes inside the wetland. However, sediment organic matter content acts as an important control on elevation changes in the interior zone because it strongly influences compaction. Short-term changes in coastal wetland surface elevation may often mask long-term trends. This was reflected in the strong intra-annual variability in some of our SET locations. Therefore, we do not only need long-term data sets of surface evolution, but also an understanding of intra-annual changes. Many of the *Phragmites* wetlands along the Darss-Zingst Bodden Chain are dyked and sediment supply from land is suppressed. Consequently, our results indicate that wetland evolution in such dyked environments is threatened (coastal squeeze), although it is problematic to extrapolate our relatively short-term surface elevation measurements to predict long-term developments.

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5. Synthesis

5.1 Interdependency and linkages between regulating services in coastal wetlands

Environmental problems in coastal wetlands are often complex due to their location between two contrasting systems: terrestrial and marine. It is not very expedient to address single threats in order to tackle the issues of coastal wetland degradation (Agardy et al. 2005). Wetland managers need to follow a holistic approach and should be prepared to implement actions across a wide array of sectors (e.g. agriculture, fisheries, industry, tourism, nature protection). In a detailed review, Kandziora et al. (2013) showed that ecosystem services are useful instruments for environmental management. However, the authors pointed out that to enhance the success of this concept special attention has to be given to ecosystem functions and mutual dependence between services. In the following I focus on phosphorus dynamics in coastal wetlands to portray how regulating services are interrelated and impact each other. Phosphorus is chosen exemplarily because contrary to marine systems where nitrogen is usually the limiting nutrient, phosphorus is mostly limiting in coastal wetlands and lagoons (Corell 1998; Reddy et al. 1999). Under optimal light and temperature conditions, the Darss-Zingst Bodden Chain is phosphorus limited (Berthold and Schumann 2016). To enable wetland managers to tackle eutrophication problems at the Darss-Zingst Bodden Chain or along the southern Baltic Sea in general, knowledge about phosphorus dynamics in wetlands and a more holistic picture of processes across the coastline from the landward to the seaward edge are vital. While phosphorus dynamics in *Phragmites* wetlands were discussed in detail in chapter 3, here I will focus on possible interactions between regulating services regarding the phosphorus cycle in coastal wetlands.

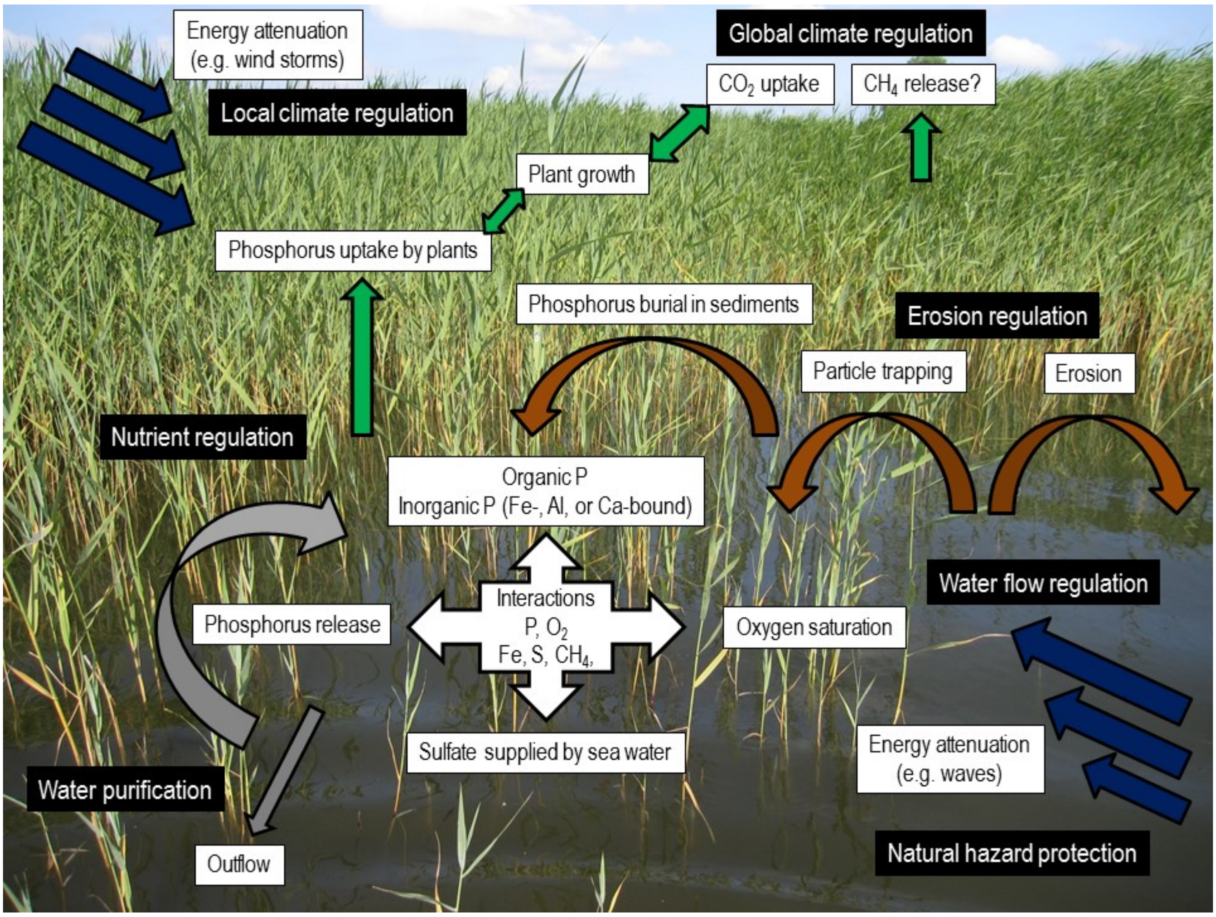

Figure 5-1: Phosphorus dynamics in coastal *Phragmites* wetlands and interactions with ecosystem services (Photo taken at study site Dabitz, Darss-Zingst Bodden Chain, Germany). Black boxes show regulating services that have the potential to impact the phosphorus cycle or interact with it. For a list of all regulating services please refer to Table 1-1.

Regulation of phosphorus flows in coastal wetlands is complex due to interactions between wetland plants, sediments and water, as well as with the adjoining terrestrial and aquatic ecosystems. Wetland plants are able to attenuate wind and water energy and to trap particles from land or sea (Duarte et al. 2013; Karstens et al. 2015b; Figure 5-1 ☞*local climate regulation, water flow regulation, erosion regulation*). Particles rich in nutrients or pollutants can be deposited and accumulated in the sediment. In case of nutrients, the vegetation can profit and phosphorus uptake by wetland plants can enhance vegetation growth, wetland expansion and carbon dioxide uptake (Figure 5-1 ☞*nutrient regulation*, *global climate regulation*). Oxygen in the water and sediments is an important driver for phosphorus storage or mobilization. Oxygen shortages may result from increased nutrient inputs from land or sea into the coastal wetland waters and can cause enhanced phytoplankton production which may lead then to elevated organic matter degradation and respiration rates (Corell 1998; Diaz and Rosenberg 2008). However, the order of processes can also be opposite and eutrophication can arise without external inputs: Because wetland plants reduce wave energy, water residence times can be long (Figure 5-1 ☞*water flow regulation, natural hazard protection*). During such hydrodynamically calm conditions when stagnant waters prevail, the downward transport of oxygen in the water is limited (Howarth et al. 2011; Karstens et al. 2015a). Redox-sensitive elements such as iron-phosphorus compounds may dissolve in the temporarily anoxic wetland waters and be transported in solution into adjacent waters. However, under oxic conditions these elements remain stable in the sediments and phosphorus can be adsorbed to iron oxides and hydroxides (Froelich 1988; Kemp et al. 2009; Howarth et al. 2011; Figure 5-1 ☞*water purification, nutrient regulation*). Thus, the location of the interface of oxic-anoxic conditions in the sediment determines whether phosphorus is stored in the sediments or released into the water. This interface does not only depend on hydrodynamic conditions (Karstens et al. 2015a), but also on oxygen transport by wetland plants. *Phragmites* possesses air conducting channels (aerenchyma) that enable the plant to transport oxygen to the anoxic sediment zone, where the radial oxygen loss along the wide-spread root system may impact the oxic-anoxic sediments interface (Rodewald-Rudescu 1974; Hebert and Morse 2003; Sundby et al. 2003). In the anoxic zone, electron donors for iron oxide reduction may be sulfides at the seaward edge and methane at the landward edge of the wetland (Caraco et al., 1990; Blomqvist et al. 2004). Consequently, the methane and phosphorus cycles are also indirectly coupled (Figure 5-1 ☞*global climate regulation, nutrient regulation)*. While in many terrestrial wetlands carbon sequestration is partially offset by methane emission from plant decomposition, methanogenesis can be inhibited by sulfates in coastal wetlands, thus reducing greenhouse gas emissions (Howe et al. 2009). However, Heyer and Berger (2000) showed that also the brackish coastal shoreline of the southern Baltic Sea can be at least periodically a hot spot for methane emissions depending on the availability of organic matter and its influence on the interrelationship between sulfate reducing bacteria and methanogenic bacteria. Autochthonous and allochthonous organic matter lead to oxygen consumption, increase of anoxic zones, sulfate reduction and sulfate consumption and thus to the termination of methanogenesis inhibition (Heyer and Berger 2000). Comparable to the quickly changing conditions of phosphorus adsorption or release from sediments described in chapter 3, also the methane emissions can stop within a few hours once water movement terminates the oxygen limitation (Heyer and Berger 2000).

This short discussion of some aspects of the phosphorus cycle made apparent how strongly regulating services in coastal wetlands are interrelated. While provisioning services often compete with each other (e.g. crop vs biomass for energy production), most regulating services can sustain each other and have a strong probability of mutual support (Kandziora et al. 2013). A healthy wetland is capable of trapping sediments and sequestering carbon at the same time as retaining heavy metals and excess nutrients as long as oxygen conditions are sufficient. The linkages between nutrient regulation, water purification and erosion control are particularly pronounced in coastal wetlands due to their location as ecotones between terrestrial and marine ecosystems. Water flow links the neighboring systems hydrologically and particles or dissolved substances can be transported by surface or subsurface flows (Reddy et al. 1999).

5.2 Impact of management decisions on regulating services in coastal wetlands

Human-made management decisions strongly influence ecosystem services. Some services might be supported while others are suppressed (Kandziora et al. 2013). This can happen on purpose or due to ignorance. In coastal wetlands a typical example is dyke construction where the adjoining wetland is purely regarded as additional natural buffer for flood protection, while other services such as nutrient regulation or water purification are inhibited.

This is the case at our study site Michaelsdorf where the dyke allows flood protection and sheep grazing in the hinterland, but uncouples nutrient dynamics in the terrestrial hinterland from the wetland (see chapter 2). Another example is reed harvest in wetlands along the Baltic Sea coast. Reed stems harvested in winter have traditionally been used as construction material in many Baltic countries (Köbbing et al. 2013). Roofs thatched with reed are characteristic for northern Germany and are very popular with tourists (Figure 5-2). Consequently reed harvest does not only supply a provisioning service with its stems for construction material, but supports also recreation and tourism values in the region, hence cultural services. Furthermore, harvested reed has the potential to remove pollutants and nutrients from the system (see phytoremediation in chapter 2.4.3). However, while reed harvest benefits provisioning and cultural services as well as nutrient regulation in some aspects, it could diminish erosion regulation. In Karstens et al. (2015b) me and my co-authors showed how dense *Phragmites* stands effectively suppress particle transport in the interior zone of the coastal wetland, even during strong winter storms. Winter harvest creates gaps in the densely vegetated wetlands and may thus enhance erosion processes during winter storms. A ‘greenbelt’ between the terrestrial hinterland and the coastal wetland without harvest should always remain to maintain the erosion regulating ecosystem service also immediately after cutting in wintertime. However, winter harvest could also have a positive impact on erosion regulation: Cutting can increase culm density and overall aboveground biomass production of *Phragmites* in the following vegetation period (Ostendorp 1999). Consequently harvest has the potential to be an overall positive human intervention, if the ‘greenbelt-rule’ is applied.

Figure 5-2: Typical reed thatched house in northern Germany.

As a third example, federal regulations on biotope protection elucidate how management decisions support some services in coastal wetlands while others are suppressed. Bodden waters in Mecklenburg-Vorpommern with its shore areas and reed beds with a minimum size of 100 m² are legally protected biotopes (LUNG 2003). For reed harvest or other intervention in the ecosystem, specific approvals from the federal conservation agencies are needed and biodiversity concerns should be particularly taken into account (LUNG 2003). Also for the BACOSA project species protection assessments and compatibility studies for the European habitat directive had to be carried out before research and field work could begin in the *Phragmites* wetlands. The aim of these regulations is the protection of species such as otter (*Lutrinae*), bittern (*Botaurinae*), raccoon dogs (*Caninae*) or red-necked grebe (*Podiceps*) (for a complete species list see LUNG 2003). Thus the Federal Act for the Protection of Nature (BNatSchG & LNatG M-V) supports biotic diversity which is an indicator for ecological integrity (Müller and Burkhard 2010; Kandziora et al. 2013). However, this ignores the fact that *Phragmites* wetlands along the southern Baltic Sea and especially along the Bodden waters have been heavily impacted by humans for decades. Nutrients and pollutants have accumulated in the sediments (see chapter 2), and the possibility of their release exists (see chapter 3). Hypoxic conditions occur during all seasons in the wetland waters and increased phosphorus fluxes from the sediment during hypoxia may drive eutrophication. Furthermore it is possible that nitrogen removal via denitrification and anaerobic ammonium oxidation is moderated during hypoxia (Conley et al. 2009). If any intervention in coastal *Phragmites* wetlands is prohibited, species protection might well be attained for the moment, but at the expense of nutrient regulation and water purification. Aerial image analysis showed that at the study site Dabitz and at many other locations along the Darss-Zingst Bodden Chain the *Phragmites* wetlands are becoming denser and water circulation becomes constrained. The creation of flooding corridors could improve the circulation of oxygen in the wetland waters and consequently prevent release of phosphorus or heavy metals from anoxic sediments (see chapter 3). However, this would be an active intervention in the protected habitat and is currently in conflict with the Federal Act for the Protection of Nature (BNatSchG & LNatG M-V). This suggests, that nature conservation would profit from a more holistic approach. Most of the mentioned species under the habitat protection for reeds depend on clean waters. Since the coastal wetlands along the Baltic Sea have been influenced by humans for decades, self-regulation might not work as in ‘untouched and natural’ systems. Therefore, federal regulations should be changed in order to allow sustainable harvest of reed plants more easily. Sustainable harvest shall take into consideration not only species protection (e.g. time of harvest due to bird breeding), but also the possibilities to remove nutrients or pollutants from the sediments (phytoremediation, see chapter 2.4.3) or to create ventilation channels and flooding corridors as mentioned above. Certainly impacts of harvest on other regulating services such as greenhouse gas emissions should also be considered prior to any decisions. According to Günther et al. (2015) reed harvest has no negative effect on greenhouse gas balances on a timescale of a few years, however the long-term effects are still under investigations and once results are available they should be incorporated into the sustainable harvest concept for coastal wetlands. A ‘greenbelt’ where no harvest takes place should be maintained between the hinterland and the coastal wetland to guarantee that erosion regulation offered by wetlands is not hampered.

5.3 Multifunctionality of coastal wetlands

Multifunctionality of landscapes is "the phenomenon that the landscape actually or potentially provides multiple material and immaterial ‘goods’ to satisfy social needs or meet social demands" (Barkman et al. 2004). The term of multifunctional landscapes gained popularity with the European Common Agricultural Policy (CAP) reform which shifted the focus from solely agricultural production towards the sustainable development of rural areas in Europe with consideration of social, environmental and economic concerns likewise (Wiggering et al. 2006). Multifunctionality is not spatially homogenous and the capability to supply a variety of services depends on biophysical and socio-economic conditions as well as interactions with adjacent landscapes (Wilemen et al. 2010). However, some authors argue that landscapes are per definition multifunctional as they are always a mixture of units with structural and functional heterogeneity (Mander et al. 2007; Vejre et al. 2007). The studied coastal wetlands in this thesis are also not homogenous: Water level and hydraulic energy are higher in the fringe than in the basin zone, whereas the interior zone is rarely flooded, and biomass decreases from land towards the sea (Karstens et al. 2015a; Karstens et al. 2016a). Consequently processes and resulting services in the three wetland zones differ. While heavy metals and excess nutrients from the adjacent cropland are retained in the sediments of the interior zone at the study site Dabitz, the possibility of their release exists in the temporarily anoxic basin zone. On the other hand *Phragmites* stems effectively reduce the turbulent kinetic energy in the fringe zone, allow the accumulation of fine-grained particles in the basin zone and support therefore the sediment retention service.

Multifunctional landscapes can provide regulating, provisioning or cultural services likewise (Wilemen et al. 2010). For commodity outputs such as crop yields markets and price values exist, while non-commodity outputs such as landscape aesthetics are public goods with no markets and no direct monetary value (Wiggering et al. 2006). The results of this thesis show that coastal wetlands provide a wide range of regulating services such as water purification, erosion control or nutrient regulation with no direct markets. However, the possibility of commodity outputs does also exist. A sustainable harvest concept could offer the possibility to use the harvested biomass for a variety of products, *inter alia* as insulation material for walls or as roofing material, as energy source (combustion, biogas, biofuel) or as fodder when harvested in summer (Köbbing et al. 2013). Landscapes that contain elements of sustainable land use and only an intermediate human intervention (such as olive groves or holm oak dehesas in the Mediterranean) are perceived by society as particularly valuable, aesthetically pleasant and multifunctional (Garcia-Llorente et al. 2012). Lovell and Jonhston (2008) argue that ecologists have not been involved sufficiently in the design of managed ecosystems and that the design of multifunctional landscapes shall be based on ecological principles. However, it seems as if it is the other way around for coastal wetlands at the Darss-Zingst Bodden Chain. Nature protection is the main target and reed beds are protected habitats. Economists would need to get involved to develop a sustainable harvest concept together with the ecologists. This thesis offers new knowledge on the functioning of coastal wetlands that can be integrated in a sustainable harvest concept.

In order to reach sustainable land use targets it is necessary to identify the social, environmental and economic landscapes functions within a participatory progress (Wiggering et al. 2003). Local communities are vital for the decision-making regarding multifunctional lands use because they are the ones who are able to identify local economic, social and ecological needs in their rural environment (Garcia-Llorente et al. 2012). In the case of the Darss-Zingst Bodden Chain, traditions and local identity are closely linked to wetland use. Houses with reed thatched roofs characterize the region (see Figure 5-2). Reed harvest and roof thatching carried out by local companies would benefit the region economically, while tourists would appreciate the continuation of the traditional northern housing style. One landscape can have different services for different people and appreciation depends on the viewpoint. Heilig (2007) gives the example of an alpine meadow which is valued by tourists for its beauty and recreation, by the landowner as a source of income and subsidies and by environmentalists for its flora and fauna. The same applies for coastal wetlands: While tourists might appreciate birds and other animals related to coastal wetlands such as wild boars, local communities might profit from reed harvest and environmentalist value the wetlands for their capacity to buffer pollutants, protect against floods or retain sediments.

5.4 Conclusions and outlook

Coastal wetlands such as salt marshes, mangroves and seagrasses are globally degrading due to anthropogenic pressures (Barbier et al. 2011). In the United States, where wetland development is closely monitored, about 325 km² of wetland area were lost each year between 2004 and 2009 (Dahl and Stedman 2013). Wetland loss or degradation leads to a decline of ecosystem functions and inherent ecosystem services such as water purification, erosion control or nutrient regulation. Along the Darss-Zingst Bodden Chain, the study area of this thesis, coastal wetlands are protected habitats, thus they are not affected by land use changes, but pressures exist and come more subtly and hidden. Although nutrient inputs via rivers were strongly reduced with the rise of the European Water Framework Directive, the lagoon is still highly eutrophic (Schiewer 1998; Berthold and Schumann 2016). For decades, the coastal wetlands have been serving as buffers for excess nutrients and pollutants from the agricultural hinterland (chapter 2, Karstens et al. 2016a). However, under certain environmental conditions carrying capacities are already reached for some elements. This is the case when surface sediments become anoxic due to stagnant waters and redox-sensitive elements such as phosphorus-iron compounds are released from the sediments into the water (chapter 3, Karstens et al. 2015a). Another aspect that threatens coastal wetland thriving along the Darss-Zingst Bodden Chain is not connected to direct pressures and is out of the scope of regional governance: sea level rise (chapter 4, Karstens et al. 2016b). *Phragmites* is a bio-engineer of its own environment and capable to accrete vertically to keep up with sea level rise if litter production and sediment trapping are high enough. However, this might not be the case in dyked environments where sediment supply from land is suppressed. In contrast, in ‘open’ systems without dykes where sediment supply from the hinterland is high, the *Phragmites* wetlands can cope better with sea-level rise.

Hence in order to know how regulating services are supplied, it is important to identify the specific site conditions of a coastal wetland. It has to be clear whether storage capacities for nutrients or pollutants have been reached, how the wetland and the adjacent hinterland have been used in the past, how freely water exchange can take place or if the wetland is expanding or retreating. All these information need to be gathered before making management decisions. Furthermore, interactions of regulating services have to be taken into account. Often decision makers consider only one branch of ecosystem services (regulating, provisioning, cultural) or even focus solely on one single service (e.g. 'blue carbon' storage debate in the course of global warming; bird protection within the EU birds directive). However, a salient feature of regulating services is that they can sustain and support each other. A wetland that traps particles and suppresses erosion serves well as an active 'pollutant buffer' storing excess nutrients or heavy metals in retained sediments. Harvest that aims to remove pollutants or flooding corridors that enhance the aeriation of wetland waters do not have to be in conflict with erosion regulation as long as a 'greenbelt' of reed with no harvest is maintained between the adjacent hinterland and the wetland.

This thesis offers important research results for a seriously under-studied area: the coastal wetlands at the southern Baltic Sea. A more holistic view on regulating services should be adopted by wetland managers and state agencies. It should be taken into consideration to use easily accessible variables like water level changes and oxygen saturation in the water to understand and predict nutrient dynamics. Furthermore, the development of a sustainable harvest concept for coastal wetlands could offer the possibility for better nutrient regulation and phytoremediation without neglecting erosion regulation.

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A. Appendices

Tabellen / Tables

Table A-1: Correlation matrix for element concentrations at study site Dabitz. Bold number= Correlation is significant at the 0.01 level (2-tailed). Grey background= Pearson correlation coefficient is higher than 0.7.

	C	N	P	S	Fe	Ca	Mg	K	Mn	Cu	Zn	Al	Co	Cr	Ni
C	1	0.99	0.73	0.71	0.56	0.75	0.89	0.84	0.28	0.66	0.47	0.3	0.37	0.48	0.6
N	0.99	1	0.78	0.66	0.63	0.79	0.92	0.86	0.35	0.7	0.53	0.37	0.45	0.54	0.65
P	0.73	0.78	1	0.23	0.92	0.82	0.8	0.79	0.63	0.68	0.72	0.6	0.76	0.64	0.7
S	0.71	0.66	0.23	1	0.22	0.36	0.61	0.52	-0.15	0.45	0.17	0.02	0.03	0.22	0.39
Fe	0.56	0.63	0.92	0.22	1	0.71	0.73	0.7	0.57	0.67	0.71	0.66	0.83	0.67	0.72
Ca	0.75	0.79	0.82	0.36	0.71	1	0.81	0.79	0.64	0.72	0.73	0.55	0.67	0.63	0.7
Mg	0.89	0.92	0.8	0.61	0.73	0.81	1	0.96	0.46	0.85	0.69	0.63	0.65	0.76	0.83
K	0.84	0.86	0.79	0.52	0.7	0.79	0.96	1	0.54	0.87	0.72	0.75	0.71	0.85	0.86
Mn	0.28	0.35	0.63	-0.15	0.57	0.64	0.46	0.54	1	0.62	0.79	0.71	0.83	0.64	0.59
Cu	0.66	0.7	0.68	0.45	0.67	0.72	0.85	0.87	0.62	1	0.82	0.8	0.79	0.88	0.9
Zn	0.47	0.53	0.72	0.17	0.71	0.73	0.69	0.72	0.79	0.82	1	0.79	0.87	0.8	0.87
Al	0.3	0.37	0.6	0.02	0.66	0.55	0.63	0.75	0.71	0.8	0.79	1	0.89	0.96	0.82
Co	0.37	0.45	0.76	0.03	0.83	0.67	0.65	0.71	0.83	0.79	0.87	0.89	1	0.85	0.8
Cr	0.48	0.54	0.64	0.22	0.67	0.63	0.76	0.85	0.64	0.88	0.8	0.96	0.85	1	0.91
Ni	0.6	0.65	0.7	0.39	0.72	0.7	0.83	0.86	0.59	0.9	0.87	0.82	0.8	0.91	1

Table A-2: Correlation matrix for element concentrations at study site Michaelsdorf. Bold number= Correlation is significant at the 0.01 level (2-tailed). Grey background= Pearson correlation coefficient is higher than 0.7.

	C	N	P	S	Fe	Ca	Mg	K	Mn	Cu	Zn	Al	Co	Cr	Ni
C	1	0.96	0.87	0.22	0.25	0.7	0.81	0.51	0.16	0.82	0.73	0.23	0.16	0.31	0.47
N	0.96	1	0.86	0.14	0.17	0.65	0.76	0.48	0.15	0.82	0.72	0.2	0.11	0.28	0.41
P	0.87	0.86	1	-0.01	0.14	0.77	0.86	0.59	0.28	0.95	0.86	0.28	0.14	0.39	0.49
S	0.22	0.14	-0.01	1	0.83	0.47	0.4	0.01	0.05	-0.05	-0.09	0	0.56	-0.07	0.48
Fe	0.25	0.17	0.14	0.83	1	0.56	0.53	0.28	0.55	0.07	0.11	0.26	0.8	0.21	0.69
Ca	0.7	0.65	0.77	0.47	0.56	1	0.91	0.56	0.43	0.71	0.61	0.33	0.56	0.36	0.69
Mg	0.81	0.76	0.86	0.4	0.53	0.91	1	0.75	0.4	0.84	0.77	0.52	0.55	0.57	0.8
K	0.51	0.48	0.59	0.01	0.28	0.56	0.75	1	0.44	0.68	0.59	0.93	0.45	0.96	0.81
Mn	0.16	0.15	0.28	0.05	0.55	0.43	0.4	0.44	1	0.2	0.36	0.39	0.58	0.42	0.47
Cu	0.82	0.82	0.95	-0.05	0.07	0.71	0.84	0.68	0.2	1	0.81	0.41	0.1	0.52	0.52
Zn	0.73	0.72	0.86	-0.09	0.11	0.61	0.77	0.59	0.36	0.81	1	0.3	0.15	0.42	0.4
Al	0.23	0.2	0.28	0	0.26	0.33	0.52	0.93	0.39	0.41	0.3	1	0.44	0.98	0.74
Co	0.16	0.11	0.14	0.56	0.8	0.56	0.55	0.45	0.58	0.1	0.15	0.44	1	0.4	0.85
Cr	0.31	0.28	0.39	-0.07	0.21	0.36	0.57	0.96	0.42	0.52	0.42	0.98	0.4	1	0.74
Ni	0.47	0.41	0.49	0.48	0.69	0.69	0.8	0.81	0.47	0.52	0.4	0.74	0.85	0.74	1

Table A-3: Principal component analysis for sediment samples 0-2 cm depth: Correlations between first two dimensions and sediment variables. Only variable were significance level is <0.05 are displayed. Results of *dimdesc* function of R package Factominer.

Dimension 1

	correlation	p.value
Mg	0,91	1,52E-19
K	0,9	3,04E-18
Cu	0,89	2,95E-17
P	0,87	2,18E-16
Cr	0,86	4,43E-15
Zn	0,85	6,45E-15
Ni	0,84	7,78E-14
Co	0,83	2,28E-13
Al	0,81	1,68E-12
Mn	0,76	1,91E-10
Ca	0,74	1,01E-09
Fe	0,71	1,36E-08
C	0,6	4,57E-06
N	0,6	5,03E-06
S	0,28	0,048

	R^2	p.value
Zones	0,63	2,01E-08

	Estimate	p.value
Dabitz Basin 0-2 cm	3,13	0,0011
Dabitz Interior 0-2 cm	2,84	0,0018
Dabitz Fringe 0-2 cm	-1,72	0,0203
Michaelsdorf Fringe 0-2 cm	-3,71	0,0004

Dimension 2

	correlation	p.value
C	0,73	3,22E-09
N	0,71	1,13E-08
S	0,61	2,95E-06
Ca	0,35	0,013
P	0,32	0,027
Mg	0,31	0,032
Ni	-0,29	0,041
Cr	-0,42	0,002
Co	-0,43	0,002
Mn	-0,45	0,001
Al	-0,54	7,42E-05

	R^2	p.value
Zones	0,85	1,53E-16

	Estimate	p.value
Michaelsdorf Interior 0-2 cm	2,62	7,68E-07
Dabitz Basin 0-2 cm	1,14	0,0063
Dabitz Interior 0-2 cm	-2,33	2,30E-07

Table A-4: Principal component analysis for sediment samples 2-10 cm depth: Correlations between first two dimensions and sediment variables. Results of *dimdesc* function of R package Factominer.

Dimension 1

	correlation	p.value
Ni	0,99	2,88E-38
K	0,96	1,10E-25
Cr	0,95	1,78E-24
Mg	0,94	2,16E-22
Cu	0,93	4,74E-21
P	0,92	4,64E-20
Al	0,92	7,19E-20
Co	0,91	5,91E-18
Zn	0,9	2,77E-17
Ca	0,88	8,23E-16
Mn	0,73	7,31E-09
Fe	0,73	9,51E-09
N	0,72	1,97E-08
C	0,67	3,61E-07

	R^2	p.value
Zones	0,63	1,10E-07

	Estimate	p.value
Dabitz Interior 2-10 cm	3,53	0,0006
Dabitz Basin 2-10 cm	3,03	0,0048
Dabitz Fringe 2-10 cm	-2,86	0,0004

Dimension 2

	correlation	p.value
S	0,76	9,08E-10
C	0,67	4,36E-07
N	0,58	2,52E-05
Ca	0,32	0,03
Zn	-0,3	0,041
P	-0,31	0,036
Co	-0,33	0,023
Al	-0,34	0,02
Mn	-0,58	2,28E-05

	R^2	p.value
Zones	0,66	2,18E-08

	Estimate	p.value
Dabitz Basin 2-10 cm	1,32	0,0003
Dabitz Interior 2-10 cm	-2,03	1,12E-06

Table A-5: Days with wind blowing on-shore (NEN-ENE: 20-70°). Wind speeds above the average (4.36 m s^{-1}) are highlighted, as well as water levels that reached the top of the micro-cliff (SET 2) and periods where the upper part of the micro-cliff was flooded for at least three consecutive days. Intervals were structured according to SET measurements: A= 18th of March - 15th of May, B= 15th of May - 7th of July, C=7th of July - 3rd of September, D= 3rd of September - 17th of November 2014, E= 17th of November 2014 - 16th of January 2015, F= 16th of January 2015 - 27th of February 2015.

Intervals	Date	Direction [°]	Speed [m s^{-1}]	Water level [m AMSL]
A	28-Mar-14	65	3,83	0,17
A	03-Apr-14	57	4,50	0,00
A	19-Apr-14	41	4,25	-0,03
A	20-Apr-14	61	5,76	0,04
A	21-Apr-14	69	5,23	0,11
A	22-Apr-14	66	4,58	0,12
A	24-Apr-14	63	3,70	0,19
A	25-Apr-14	63	3,75	0,11
A	26-Apr-14	62	3,63	0,06
A	27-Apr-14	66	4,35	0,01
A	28-Apr-14	63	3,07	-0,03
B	27-May-14	65	8,88	-0,01
B	28-May-14	67	9,80	0,26
C	09-Jul-14	63	7,28	0,18
C	10-Jul-14	48	5,89	0,23
C	11-Jul-14	50	6,21	0,23
C	19-Jul-14	65	3,50	-0,05
C	20-Jul-14	63	5,56	-0,01
C	21-Jul-14	69	7,42	0,06
C	22-Jul-14	47	5,07	0,08
C	23-Jul-14	38	4,80	0,06
C	24-Jul-14	32	4,22	0,03
C	25-Jul-14	57	5,11	0,01
C	29-Jul-14	62	3,43	-0,11
C	02-Sep-14	60	3,65	0,16
D	11-Sep-14	47	3,28	-0,01
D	12-Sep-14	33	4,85	0,03
D	13-Sep-14	29	4,67	0,10
D	15-Sep-14	69	6,46	0,19
E	19-Nov-14	70	3,70	0,08

Lebenslauf / Curriculum vitae

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Bildungsweg

10/2006– 10/2012	**Rheinische Friedrichs-Wilhelms Universität Bonn** Diplomgeographie mit den Nebenfächern Meteorologie und Physik
08/2008 – 07/2009	**Universität Michel-Montaigne Bordeaux, Frankreich** Erasmus-Studienjahr (Umweltwissenschaften und Meereskunde)
06/2006	**Abitur am Gymnasium Altenholz, Schleswig-Holstein** NC: 1,9 – Leistungskurse: Mathematik und Englisch
08/2003 – 06/2004	**Eagles Landing High School, Stockbridge, Georgia, USA** Einjähriger Schüleraustausch in der 11. Klasse

Praxis

02/2016 – 02/2017	**Elternzeit**
04/2013 – 04/2016	**Wissenschaftliche Mitarbeiterin an der Professur für Landschaftsökologie, Universität Rostock** Analyse von Nährstoffkreisläufen und Sedimentationsdynamiken in Küstenfeuchtgebieten im Projekt „Baltic Coastal System Analysis“
01/2013 - 03/2013	**Feldarbeiten im Natura 2000 Gebiet Għajn Tuffieħa, Malta** Verschiedene Tätigkeiten in der NGO *Gaia Foundation* (Küstenzonenmanagement)
02/2012 – 05/2012	**Forschungsarbeiten in West-Kalimantan, Indonesien** DFG-Projekt „Stadt, Land, Fluss: Eine politische Ökologie des Kapuas“ Diplomarbeit zu Umwelttransformationen im Kapuas-Delta
02/2011- 06/2011	**Praktikum am Leibniz-Institut für Ostseeforschung, Rostock** Forschungsarbeiten zu Klimawandelanpassung im Ostseeraum innerhalb des EU-Projekts „Baltadapt“
11/2010 – 01/2011	**Praktikum in der NGO Al Amal, Amzrou, Marokko** Installation von Tröpfchenbewässerungssystemen in Dattelpalm-hainen; Wiederaufforstung von Tamarisken; Sensibilisierung der Bevölkerung zu Themen der Umwelt und Gesundheit; Erstellung einer Internetseite
09/2009 – 07/2010	**Studentische Hilfskraft im Zentrum für Entwicklungsforschung, Bonn** GLOWA Volta Projekt: Analyse eines hydrologischen Netzwerks mit ArcGis; Übersetzungen (englisch/französisch); organisatorische Aufgaben

Selbstständigkeitserklärung / Declaration of independence

Ich erkläre, dass ich die hier vorgelegte Arbeit selbständig und ohne fremde Hilfe verfasst, andere als die von mir angegebenen Quellen und Hilfsmittel nicht benutzt und die den benutzten Werken wörtlich oder inhaltlich entnommenen Stellen als solche kenntlich gemacht habe.

Rostock, den 30.11.2016 Svenja Karstens

Thesen / Theses

Theses for the dissertation entitled:

Ecosystem Services in coastal *Phragmites* wetlands at the southern Baltic Sea: Nutrient regulation, water purification and erosion control

Submitted by Svenja Karstens

Rationale and objectives

- Coastal wetlands can provide a variety of ecosystem services. In order to understand how ecosystem services are supplied and might change under human influences the biological, physical and chemical processes need to be studied.
- The Baltic Sea region has become an important economic driver and anthropogenic stresses on the coasts continue to rise. Eutrophication and hypoxia remain major environmental challenges. Coastal wetlands with their ability to buffer nutrients have the potential to counteract eutrophication.
- Coastal wetlands are still under-studied. The research of this thesis took therefore place directly within coastal wetlands at the Darss-Zingst Bodden Chain, a lagoon system at the southern Baltic Sea.
- The objectives were to analyze (1) how regulating ecosystem services in coastal wetlands are supplied, (2) how they are interrelated, and (3) how physical, biological and chemical processes govern them. The focus was on following regulating services: erosion control, water purification and nutrient regulation.

Materials and Methods

- In order to investigate the role of wetlands as ‘pollutant buffers’ sediment samples in different wetland zones at two contrasting study sites were analyzed for their heavy metal (Mn, Fe, Cu, Zn, Al, Co, Cr, Ni) and macronutrient (C, N, P, K, Mg, Ca, S) concentrations.

- To gain insights into the processes of phosphorus (im)mobilization, laboratory experiments and field work were combined. Soluble reactive phosphorus concentrations, oxygen saturation, pH, redox potential, sulfate concentrations, conductivity and temperature were measured monthly in the wetland waters. Equilibrium sorption isotherms were assessed by batch incubation experiments to evaluate the phosphorus sorption capacity of sediments under oxic conditions. Additionally, turbulent kinetic energy was measured in the wetland waters because hydrodynamics influence the oxygen uptake by sediments.
- Changes of surface elevation and microtopography were measured bimonthly at five different locations within a dyked coastal wetland using a Surface-Elevation Table (SET). The SET consists of benchmark pipes permanently installed in the ground and a portable leveling device that can be attached to each benchmark pipe. Additionally, wetland expansion between 1953 and 2013 was analyzed.

Results

- Land use activities adjacent to coastal wetlands have a large impact on the sediment composition:
  - Heavy metal concentrations were significantly elevated in the wetland zone that borders directly an arable field where crop production with fertilizer application took place at least since the 1950ies.
  - In contrast to this, heavy metal concentrations were low in the coastal wetland that is confined by a dyke with sheep grazing in the hinterland.
  - Influences from the sea on sediment compositions were minor compared to the influences from land.
- The relationships between oxygen saturation, water level and soluble reactive phosphorus in the wetland waters showed a threshold-type behavior. Oxygen shortages may not be caused by external nutrient input, but by hydrodynamic calm conditions with stagnant waters.

- Sorption experiments under aerobic conditions confirmed that the sediments are capable of immobilizing large amounts of phosphorus within short timeframes if oxygen supply is sufficient. Once the oxidized sediment surface layer vanished, phosphorus was released into the water column.
- None of the SET-measuring locations in the dyked wetland could currently keep up with the local sea level rise. At the wetland fringe erosion prevailed and in the wetland interior the high sediment organic matter content enhanced compaction processes.

Conclusions

- In order to know how regulating services are supplied, it is important to identify the specific site conditions of a coastal wetland. It has to be clear whether storage capacities have been reached, how the wetland and the adjacent hinterland have been used in the past, how freely water exchange can take place or if the wetland is expanding or retreating.
- *Phragmites* wetlands at the Darss-Zingst Bodden Chain can be ‘water purification systems’ retaining excess macronutrients and heavy metals in the sediments. However, it is possible that redox-sensitive elements are released from the sediments in the temporarily anoxic basin zone. Future developments should be monitored closely to avoid breakthroughs due to exceeded carrying capacities.
- Wetland evolution in dyked environments could be threatened due to the suppression of sediment supply from land.
- Regulating services can sustain and support each other. A wetland that traps particles and suppresses erosion serves well as an active ‘pollutant buffer’ storing excess nutrients or heavy metals in retained sediments.
- Sustainable reed harvest that aims to remove pollutants or flooding corridors that enhance the aeration of wetland waters do not have to be in conflict with erosion regulation as long as a ‘greenbelt’ of reed with no harvest is maintained between the adjacent hinterland and the wetland.

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Wild boars at the Darss-Zingst Bodden Chain. Photo taken by Marion & Peter Karstens.