SHORT COMMUNICATION

Blue Carbon in Coastal *Phragmites* Wetlands Along the Southern Baltic Sea

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Abstract

Coastal wetlands are important for carbon (C) storage and sequestration. Still, there are large knowledge gaps concerning the amount of "blue carbon" in coastal wetlands dominated by common reed (*Phragmites australis*). We quantified carbon stocks at the southern Baltic Sea coast at six representative *Phragmites* wetland sites at the Darss-Zingst-Bodden Chain (DZBC) and the Strelasund, which include different categories of adjacent land use (arable land, woodland, pasture, urban), topography (totally flat to undulating), and geographical restrictions (dyking). Sediment samples were taken to a depth of 1 m, in line with the IPCC guidelines, and total carbon concentrations and bulk densities were measured in 10 cm intervals. The sites stored, on average, 17.4 kg C m^{-2} with large variability between sites, ranging from 1.76 to 88.6 kg C m^{-2}. The estimated average is generally in good agreement with carbon stocks reported for tidal salt marshes, mangroves, and seagrass meadows. According to our estimation, based on widths of the reed belts and carbon stocks at the sampled sites, approximately 264,600 t of blue carbon could be stored in the coastal reed belts along the DZBC, a typical lagoon system of the southern Baltic Sea. Our study underlines the importance of these unique ecotones between land and sea for storage and sequestration of blue carbon. Since *Phragmites* is also a common (sometimes invasive) species along other large brackish water basins, such as the Black Sea or Chesapeake Bay, these estimates can be used for improved precision of modeling blue carbon budgets.

Keywords Brackish wetland · Saltmarsh · *Phragmites australis* · Darss-Zingst-Bodden Chain · Regional C stock inventory · Sediment carbon stocks

Introduction

The interface between aquatic and terrestrial ecosystems is often formed by coastal wetlands (Jurasinski et al. 2018). These highly productive ecotones perform several ecosystem functions and services, e.g., nutrient regulation, dissipating wave energy, providing habitat for flora and fauna, or sequestering atmospheric carbon dioxide (CO_2) (Reddy and DeLaune 2008; Laffoley and Grimsditch 2009; Kirwan and Megonigal 2013; Garbutt et al. 2017; Karstens et al. 2019; Heckwolf et al. 2021; Gilby et al. 2021). Carbon (C) stored in coastal wetlands, such as mangroves, seagrass meadows, salt marshes, and brackish reed wetlands, is referred to as "blue carbon" (Nellemann et al. 2009; McLeod et al. 2011; Pendleton et al. 2012; Serrano et al. 2019). The total world-wide area of coastal wetlands is small compared with other ecosystem types, and consequently, the total amount of C stored in coastal wetlands is relatively small (10–25 Pg compared to 1500 Pg in soils worldwide; Pendleton et al. 2012; Duarte et al. 2013; Serrano et al. 2019). However, carbon

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stored in coastal (and other) wetlands has a disproportionate importance, because it is stored over a longer time scale compared with most terrestrial soils (Serrano et al. 2019; Hopkinson et al. 2019). In coastal wetlands, C sequestration rates (on average 200 g C m^{-2} year^{-1}) are very high compared with other ecosystem types (e.g., 20 g m^{-2} year^{-1} in inland wetlands and 5 g m^{-2} year^{-1} in forest ecosystems) (Ouyang and Lee 2014). These high C sequestration rates are due to high biological productivity and the low rates of decomposition in waterlogged wetland substrates (McLeod et al. 2011; Ouyang and Lee 2014).

Coastal wetlands are strongly endangered by human activities, such as urbanization, aquaculture, or excessive nutrient input resulting in eutrophication (Deegan et al. 2012; Hopkinson et al. 2012; Kirwan and Megonigal 2013; Schuerch et al. 2018; Newton et al. 2020; O'Connor et al. 2020; Gilby et al. 2021). If coastal wetlands are lost, a large part of the stored carbon may be released back into the atmosphere, exacerbating the rise of atmospheric CO_2 concentrations (Pendleton et al. 2012; Bu et al. 2015). In order to evaluate the potential carbon release of wetland sites, knowledge about the carbon stocks of different types of wetlands is required. However, the overall global amount of carbon stocks is only known with a large margin of uncertainty (Duarte et al. 2013; Serrano et al. 2019). On the one hand, this is due to the high spatial variability of C stocks (Craft 2007; McLeod et al. 2011), since local carbon density is governed by a large variety of factors (e.g., salinity, nutrient status, sediment supply, climate, species composition, tidal range). On the other hand, the total area covered by coastal wetlands is uncertain, especially for salt marshes (McLeod et al. 2011). Although more recent estimates give a more precise value of 54,950 km^2 covered by salt marsh (Mcowen et al. 2017; Hopkinson et al. 2019), many coastal wetland areas in the southern Baltic Sea region seem to be not included in this value, likely because they do not fit well into the salt marsh category.

Large parts of the Baltic Sea coast are lined with coastal wetlands dominated by common reed (*Phragmites australis* (Cav.) Trin. ex Steud.); this holds true especially for the inner, protected coast within lagoons and estuaries (Dijkema 1990; Karsten et al. 2003; Selig et al. 2007; Meriste et al. 2012; Altartouri et al. 2014). *Phragmites australis* is a cosmopolitan species occurring worldwide, sometimes considered being an invasive species, for instance in North America (Emery-Butcher et al. 2020; Blossey et al. 2020). Reed wetlands are common along other large brackish water basins, such as the Black Sea (Sangiorgio et al. 2008) or Chesapeake Bay (e.g., Rice et al. 2000; Chambers et al. 2008). However, compared to other coastal wetland types, data about carbon storage in *Phragmites* wetlands worldwide is scarce. Here, we quantify C stocks in different *Phragmites* wetland sites in two lagoon systems of the southern Baltic Sea, as an example of a brackish marginal sea, and estimate the total amount of blue carbon stored along a representative Baltic Sea lagoon, based on wetland widths.

Material and Methods

Study Area

Our study sites are located along the Darss-Zingst-Bodden Chain (DZBC) and in the Strelasund (Fig. 1), two sheltered, brackish lagoon systems in the federal state of Mecklenburg-Vorpommern (MV), Germany, which are representative for inner coastal waters. These inner coasts are almost entirely lined with wetlands of varying widths (commonly in the range 10–100 m), which are dominated by common reed (*Phragmites australis* (Cav.) Trin. ex Steud.). The DZBC covers a total area of about 200 km^2 and a coast length of 194 km (Karsten et al. 2003; Lampe et al. 2007), while the Strelasund covers an area of 64 km^2 and its coastal length is 92 km (Reinecke 2005). The Strelasund has salinities of 8–9 PSU, whereas salinities in the DZBC increase from west to east, with 0–3 PSU in the innermost lagoon (Saaler Bodden) to 7–10 PSU in the outermost part (Grabow) (Selig et al. 2007).

Sampling and Laboratory Analyses

Sediments were sampled at six sites in three zones of the *Phragmites* wetlands from the sediment surface to 100 cm depth, divided into 10 cm depth intervals. A closed stainless steel sediment corer with 7 cm inner diameter (Hydro-Bios, Kiel, Germany) was used. Samples were taken in three zones: (1) at the terrestrial wetland edge (= supra-littoral, only rarely inundated), (2) in the transitional zone (= eu-littoral, intermittently inundated), (3) at the littoral wetland edge (= sub-littoral, permanently inundated) (see also Karstens et al. 2016a, b). These three zones were clearly discernible from each other in the field, based on elevation and water depth. To account for the different widths of the *Phragmites* wetlands, four replicate locations in each zone were sampled at Dabitz, where the reed belt was widest, whereas at the other five sites, three locations in each zone were sampled. All replicate samples were taken at similar elevations within the zones. These sampling spots within each zone were 3–5 m apart from each other. Each soil core was subdivided equally into 10 subsamples representing each 10 cm of the soil core length and therefore a specific depth interval. The subsamples were immediately bagged and transported to the laboratory. These fresh soil samples were weighed in the laboratory, and then oven-dried for 24 h at 105 °C and thereafter weighed again to determine dry mass. Oven-drying at 105 °C is standard procedure in soil
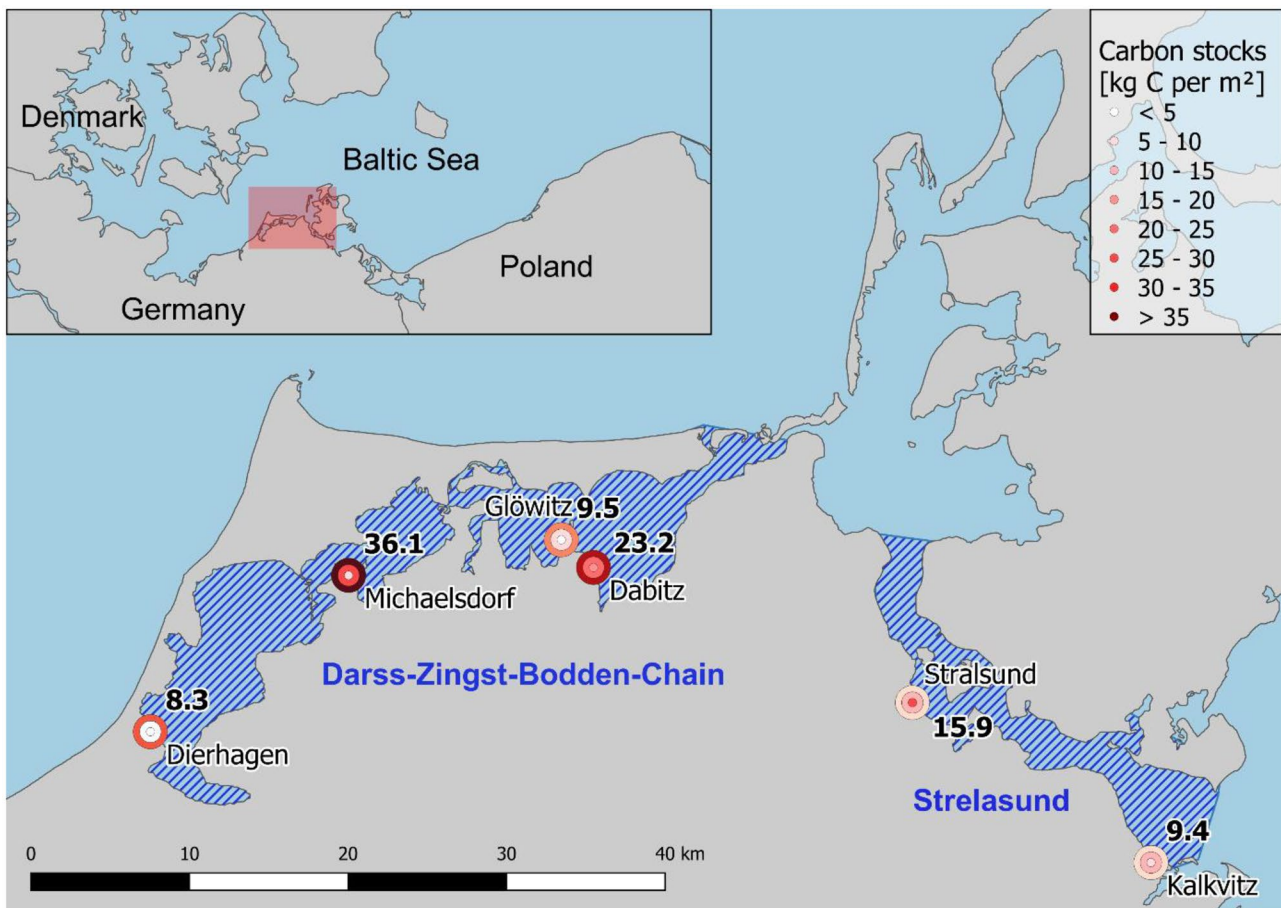
Fig. 1 Location of the sampling sites and carbon stocks kg C m^{-2} of the six sampling sites. Lagoons (Darss-Zingst-Bodden Chain and Strelasund) are displayed in blue stripes. C stocks are depicted in the circles: Inner circle represents the carbon stocks in the littoral zone, outer circle the terrestrial zone, and in between the transitional zone. Mean carbon stocks averaged over all three wetland zones are displayed as numbers. QGIS

science in Germany (DIN EN ISO 11461 2014; DIN ISO 11465 1993) but also internationally (e.g., Gardner 1986) and studies determining soil organic carbon have advocated drying temperatures of 105 °C (Torn et al. 1997) to remove all pore water from the soil and therefore to determine soil water content and soil dry mass. When drying soils for determination of soil organic carbon, it was sometimes recommended to dry at 60 °C, to prevent volatilization of organic matter (e.g., Howard et al. 2014). However, comparative studies suggest that the loss of organic matter at drying temperatures of 105 °C compared to 60 °C is usually negligible (e.g., Samuelson et al. 2006; Dettmann et al. 2021).

The samples did not contain large rhizomes; however, small adventitious roots remained in the samples. Each sample was ground with mortar and pestle, and then, the complete material was passed through a 2-mm sieve for further analysis. Total C concentration was measured by combustion in a CNS Analyzer (Vario Max, Elementar, Germany). The samples were treated with drops of 10% HCl (chloric acid)

solution to test for calcite. No calcite was detected during these tests, which suggests that all the analyzed carbon is organic. However, since we tested only with single drops on each sample, heterogeneous patches of calcite within the samples may have remained undetected. The bulk density (BD) was determined as dry sample mass divided by sample volume. Total soil carbon stocks per m^2 were calculated by multiplying, for each sampled soil layer of 10-cm thickness, the BD with the C concentration and 0.1 m (i.e., the thickness of the sampled soil layers). Then, carbon stocks for the 10-cm layers were cumulated for all sampled depths to obtain the carbon stocks per m^2 and 0–1 m soil depth.

Data Analyses

Data for bulk density, C concentration, soil C stock, and plant biomass were tested for normal distribution using the Kolmogorov–Smirnov test. The statistical significance of differences for specific depths among the three wetland

zones and the six sites was evaluated using analysis of variance (ANOVA) and Tukey's post hoc test for pairwise comparisons (using the software IBM SPSS, v. 20.0). Aerial images from 2016 provided by the federal office for geoinformation were used to derive an estimate of the mean width of the reed belts along the DZBC in order to approximate the wetland area. We multiplied the average carbon stock with the approximate wetland area to estimate the blue carbon potential of the reed belts at the DZBC and beyond.

Results

Although bulk density was highly variable among the sites, depths, and wetland zones (with an overall range between 0.13 and 2.01 g cm^{-3}), there are some general patterns (Fig. 2): for all sites except Kalkvitz, the highest values of bulk density were observed in the littoral zone, and the lowest in the terrestrial wetland edge. Carbon (C) concentrations (in mass %) spanned a broad range between 0.004 and about 19.9% (Fig. 2). C concentrations were in general highest in the terrestrial zone and lowest in the littoral zone. Carbon stocks per 10-cm layer (Fig. 2C) were also in general highest in the terrestrial zone, with values up to 15.4 kg C m^{-2} 10 cm^{-1} in Michaelsdorf, and maximum values of 8.3 kg C m^{-2} 10 cm^{-1} at the other sites. In contrast to C concentrations, carbon stocks showed no clear trend with depth, since they were determined both by BD and C concentration. These exhibited mostly contrasting depth trends (Figs. 2 and 3) which canceled each other out. However, for the four sampling sites in the DZBC (Dabitz, Dierhagen, Glöwitz, Michaelsdorf), carbon stocks in the upper 20 cm were higher in the terrestrial wetland edge compared to the other zones.

Cumulated carbon stocks for 100 cm depth covered a wide range between 1.76 and 88.6 kg C m^{-2}, depending on the site and reed belt zone (Fig. 3 Table 1). Averaged over the three reed belt zones, the differences between sites covered a range between 8.3 (Dierhagen) and 36.1 kg C m^{-2} (Michaelsdorf). The average C stock for all sites and all zones was 17.4 kg C m^{-2} with a large standard deviation of 16.7. Cumulated carbon stocks were on average highest in the terrestrial zone (29.9 kg C m^{-2}), and lowest in the littoral zone (10.6 kg C m^{-2}) (Table 1). However, for the two sites in the Strelasund, this relation was reversed. Analysis of aerial images for the DZBC yielded a mean width of the reed belt of 70 m, and thus a total area of the reed belt of 13.5 km^2, considering the coast length of 194 km. Extrapolating this to the total inner coastline of MV with a length of 1568 km, up to 109.8 km^2 could potentially be covered by brackish reeds.

Discussion

Carbon Stocks in Coastal *Phragmites* Wetlands

The basis for global assessments of carbon stocks are studies on regional and local scales, followed by mapping and modeling on larger scales. However, there is a lack of studies addressing blue carbon along the Baltic Sea. Here, we provide much-needed data on blue carbon stored in these important ecotones between land and sea. Overall, average C stocks in the reed belt sediments were 17.4 kg C m^{-2}. This is higher than C stocks of seagrass meadows in the Baltic Sea (0.23 kg C m^{-2} for the Puck Bay in Poland (Jankowska et al. 2016), 0.63 kg C m^{-2} for Finnish sites, and 4.3 kg C m^{-2} for sites in Denmark (Röhr et al. 2016)), but within the lower range of C stocks reported for tidal salt marshes worldwide. Global averages for C stocks in tidal salt marshes are 25 kg C m^{-2} according to Pendleton et al. (2012), whereas Duarte et al. (2013) give a value of 16.2 kg C m^{-2} (albeit with a very high standard deviation of 25.9 kg C m^{-2}). Similarly, Chmura et al. (2003), based on an analysis of 26 saltmarsh sites worldwide, report average C stocks of 19.5 kg C m^{-2}, although their estimate is only valid for depths of 0–50 cm (carbon density of 0.039 g cm^{-3}).

Compared to vegetation dominated by other salt marsh plants, like *Spartina maritima* or *Juncus gerardii*, studies addressing soil C stocks in *Phragmites*-dominated coastal wetlands are rare, not only for the Baltic Sea (Callaway et al. 1996), but also on a global scale (Anisfeld et al. 1999; González-Alcaraz et al. 2012; Ouyang and Lee 2014; Byun et al. 2019). For *Phragmites*-dominated salt marsh sediments in South Korea, Byun et al. (2019) report C stocks in the sediments of 15–20 kg C m^{-2} (to 1 m depth), and Callaway et al. (1996) found C stocks of 20–30 kg C m^{-2} (to 50 cm depth) in *Phragmites* wetlands of the Odra and Vistula estuaries (Poland). *Phragmites* has a very high biomass production that could lead to substantial soil C storage (Brix et al. 2001; Engloner 2009; Song et al. 2015). Organic C in the sediments is not derived solely from aboveground litter, but also from roots and rhizomes (Rooth et al. 2003). Rooting depth of *Phragmites* is with up to 1.5 m higher compared to other wetland species (Mozdzer et al. 2016; Lissner and Schierup 1997). Sedimentation of allochthonous material also impacts carbon storage. While sediment deposition patterns in *Spartina* and *Phragmites* wetlands are comparable (Leonard et al. 2002), higher sediment accretion rates in *Phragmites* compared with *Typha* wetlands have been described (Rooth et al. 2003). Gu et al. (2020) reported that invasive *Phragmites* increased blue carbon stocks in an estuarine marsh in Quebec, Canada, previously dominated by *Spartina patens*.
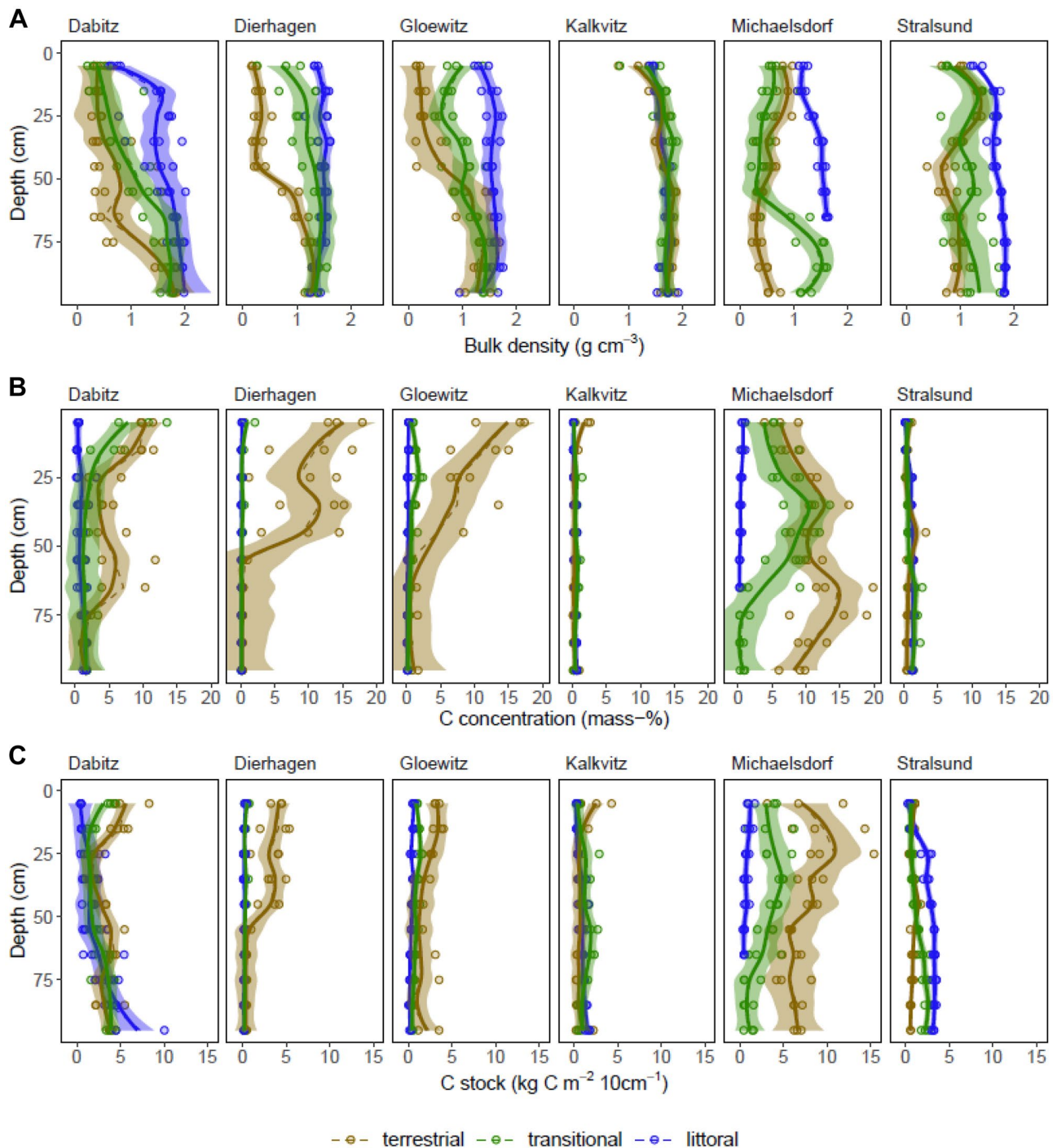
Fig. 2 Depth profiles of bulk density (upper panel **A**), C concentration (middle panel **B**), and C stock (lower panel **C**) for the six sampling sites and the three reed belt zones (see legend below plots). "R", package ggplot

Total Carbon Estimate for *Phragmites* Wetlands in the Darss-Zingst-Bodden Chain

For an assessment of regional C stocks of coastal wetlands, both data about the spatial variability of local C stocks and the area covered by coastal wetlands are required. In

Europe, data about the total areal coverage of coastal wetlands are until now incomplete: for instance, Dijkema (1987) estimated the total salt marsh area of Europe at 2302 km^2, whereas Sterr (2008) gives an area of coastal wetlands of about 1800 km^2 for the German portion of the Baltic coast. McOwen et al. (2017), in their global map of saltmarshes,
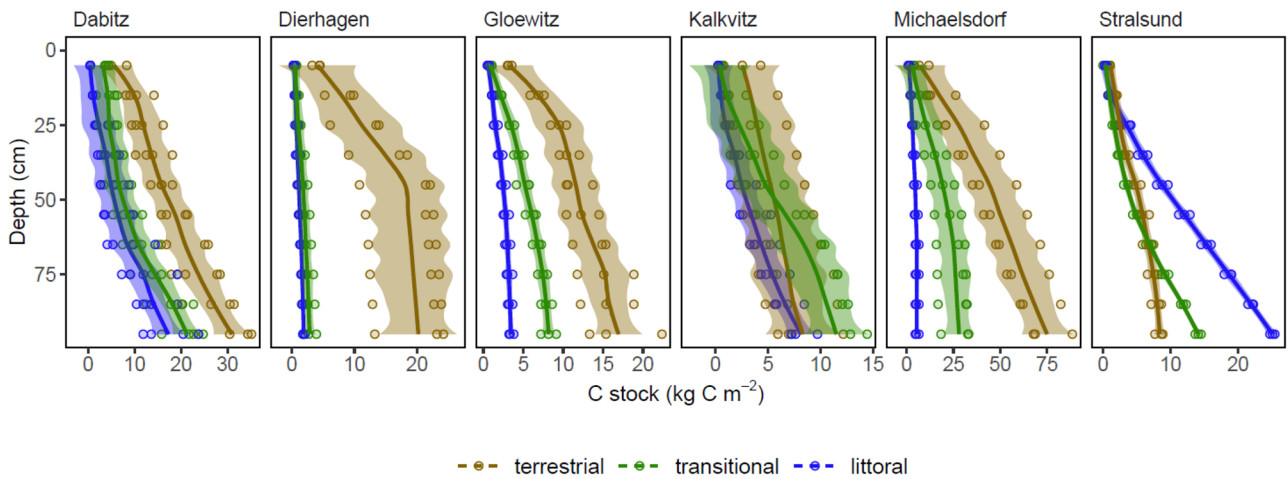
Fig. 3 Cumulated carbon stock vs depth for the six sampling sites and the three reed belt zones. Note that scaling of the *x*-axis differs among the sites. "R", package ggplot

calculate an area of 4512 km^2 for Europe (without Russia). However, for the German part of the Baltic Sea, only small parts of the coastal wetlands are included. With an average of 19.6 kg C m^{-2} and wetland width of 70 m, this leads to an amount of 264,600 t blue carbon stored in *Phragmites* wetlands along the coast of the DZBC. Considering the total length of protected inner coastline in MV with 1568 km, an estimated reed belt area of 109.8 km^2 and an average of 17.4 kg C m^{-2} for MV, the blue carbon potential could be as high as 1.9 million tons of carbon. This number includes only brackish reed wetlands and not yet salt meadows or coastal peatlands, which are also abundant along the Baltic coast (Jurasinski et al. 2018).

The high variation in C stocks between the six sites of our study (ranging from 8.3 kg C m^{-2} at Dierhagen to 36.1 kg C m^{-2} at Michaelsdorf) is consistent with high variability of C stocks in salt marshes worldwide (Serrano et al. 2019). Carbon storage in our study differed not only significantly between sites but also between the wetland zones. In general, C stocks are highest in the terrestrial zone, where biomass production and thus accumulation of litter are highest (Karstens et al. 2016a, b). Studies worldwide have shown that the amount of blue carbon is governed by a large variety of factors, for instance salinity (Osland et al. 2018), nutrient status, sediment supply and sediment texture (Kelleway et al. 2016), climate (Osland et al. 2018), plant species (Ouyang

Table 1 Information on sampling sites and summary of mean C stocks (kg C m^{-2})

	Dabitz	Dierhagen	Glöwitz	Kalkvitz	Michaelsdorf	Stralsund
Location	DZBC 54°22'08"N 12°48'15"E	DZBC 54°17'19"N 12°22'01"E	DZBC 54°23'08"N 12°46'28"E	Strelasund 54°11'01"N 13°19'36"E	DZBC 54°22'18"N 12°33'59"E	Strelasund 54°16'56"N 13°06'20"E
Adjacent land use	Arable land	Pasture	Arable land	Woodland	Pasture	Urban
Dyking	No	No	No	Yes	Yes	No
Width of reed belt	> 100 m	30–50 m	40–60 m	50–100 m	30–40 m	60–80 m
Mean salinity	7 PSU	< 3 PSU	5.5 PSU	8.5 PSU	3 PSU	8.5 PSU
Overall sediment carbon stocks 0–100 cm	23.2 (7.5)[A]	8.3 (9.5)[B]	9.5 (6.4)[B]	9.4 (3.1)[B]	36.1 (31.6)[A]	15.9 (7.4)[A,B]
Terrestrial zone	31.0 (5.2)[a]	20.2 (6.1)[a]	17.0 (4.8)[a]	8.4 (3.3)[a]	75.0 (11.8)[a]	8.4 (0.6)[a]
Transition zone	21.3 (3.8)[b]	2.8 (1.0)[b]	8.2 (1.0)[b]	11.5 (3.8)[a]	28.0 (8.4)[b]	14.1 (0.4)[b]
Littoral zone	17.4 (5.6)[b]	1.9 (0.2)[b]	3.4 (0.3)[c]	8.2 (1.3)[a]	5.4 (1.0)[c]	25.1 (0.4)[c]

Different indices within this row denote significantly different carbon stocks. Different indices within one column denote significantly different carbon stocks

Indices (a, b, c) in the rows "Terrestrial zone," "Transition zone," and "Littoral zone" denote significant differences in carbon stocks ($p < 0.05$) between wetland zones within a specific site. Indices (A, B) in the row "Overall sediment carbon stocks" denote significant differences in carbon stocks ($p < 0.05$) between sites

and Lee 2014), tidal range (Ouyang and Lee 2014; Serrano et al. 2019), sea level rise (Rogers et al. 2019), or management practices (O'Connor et al. 2020). In most coastal wetlands, C storage, however, is not governed by one single of these factors, but rather by multiple biotic and abiotic influences as well as the history of the sea and landscape (Rogers et al. 2019; Serrano et al. 2019) and it is used by humans. These factors need further investigations to better understand variability of carbon storage and sequestration in coastal *Phragmites* wetlands.

Conclusions

Our current estimation of blue carbon stored in the DZBC is 264,600 t in the *Phragmites* wetlands alone. Given the presence of further large carbon stocks in brackish grasslands and the sediments of the lagoon itself as well as the high spatial variation of stocks in the reed belts, more research is clearly needed. Coastal wetlands dominated by *Phragmites* are very common along the Baltic Sea, and a large part of the Baltic coast features lagoons, bays, and estuaries which show similarities to the DZBC, but also differences. However, as indicated in the "Discussion" section, information about blue carbon in *Phragmites* wetlands along the Baltic coast is virtually absent. Therefore, more research in other regions of the Baltic sea is needed, to gain a better understanding of the role of the Baltic Sea for blue carbon storage, and to better understand which factors determine carbon stocks and sequestration in this marginal sea. Based on our analysis alone, we can already state that the *Phragmites* wetlands along the southern Baltic Sea coast are certainly important carbon stores that we should aim to better understand regarding their stability and possible development in the future.

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