Journal of Sea Research 181 (2022) 102177

Contents lists available at ScienceDirect

Journal of Sea Research

journal homepage: www.elsevier.com/locate/seares

Sand-capping stabilizes muddy sediment and improves benthic light conditions in eutrophic estuaries: Laboratory verification and the potential for recovery of eelgrass (*Zostera marina*)

Mogens R. Flindt [a,*], Nele S. Oncken [a,b], Kadri Kuusemäe [a,c], Troels Lange [a], Nicolaj Aaskoven [a,e], Sonja Winter [a], Ana I. Sousa [a,d], Erik K. Rasmussen [c], Paula Canal-Verges [a], Rod M. Connolly [f], Erik Kristensen [a]

[a] *Department of Biology, University of Southern Denmark (SDU), Campusvej 55, dk-5230 Odense M, Denmark*
[b] *German Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany*
[c] *DHI, Agern alle 5, dk-2970, Hørsholm, Denmark*
[d] *CESAM – Centre for Environmental and Marine Studies, Department of Biology, University of Aveiro, Campus Universitário de Santiago, 3810-193 Aveiro, Portugal*
[e] *Environmental Protection Agency, Ministry of Environment and Food, Denmark*
[f] *Coastal and Marine Research Centre, Australian Rivers Institute, School of Environment and Science, Gold Coast campus, Griffith University, Queensland 4222, Australia*

ARTICLE INFO

Keywords:
Marine restoration
Eelgrass recovery
Sediment resuspension
Turbidity
Environmental conditions

ABSTRACT

Decades of eutrophication have increased water turbidity in Danish estuaries and led to light limitation of eelgrass (*Zostera marina*) growth. Former eelgrass areas are now denuded and consist of organic-rich muddy sediment with frequent resuspension events that maintain a high turbidity state. In addition, low anchoring capacity of eelgrass in the soft organic-rich sediments has contributed to eelgrass loss. When navigation channels in Danish estuaries are dredged, large amounts (~100.000 m^3) of sandy sediment are shipped to remote dumping sites. Instead, we suggest that the dredged sand is used to consolidate adjacent muddy areas. We demonstrate in the present laboratory study that capping of fluid muddy sediment with 10 cm of sand is feasible without any vertical mixing and that this marine restoration approach potentially can lower the magnitude and frequency of resuspension events. Erosion of suspended solids change from 5 g m^{-2} min^{-1} above muddy sediment as to about 0.2 g m^{-2} min^{-1} after sand-capping, implying that the application of sand can improve light conditions. Moreover, since erosion thresholds increase from about 10–12 cm s^{-1} for mud to 40 cm s^{-1} for sand-capped mud the anchoring capacity of rooted vegetation is increased. However, the full potential of sand-capping to facilitate restoration of otherwise lost eelgrass habitats requires verification by large-scale field experiments.

1. Introduction

Eelgrass (*Zostera marina* L.) is the most common seagrass on the northern hemisphere, but it has declined substantially along European and North American coasts in recent decades (Orth et al., 2006a; Waycott et al., 2009; Boström et al., 2014) due to anthropogenically driven eutrophication (Short et al., 2011; de los Santos et al., 2019). Eelgrass has suffered from physical stress, reduced light climate and lowered anchoring capacity in organic enriched sediments primarily due to increased competition and organic deposition from macroalgae, epiphytes and phytoplankton (Flindt et al., 2004; Hauxwell and Valiela, 2004; Greve et al., 2005). Substantial efforts have in recent years been devoted to combat anthropogenic pressures and facilitate seagrass recovery (Greening and Janicki, 2006; Petersen et al., 2009; Marion and Orth, 2010; van Katwijk et al., 2016). However, natural restoration has been less successful than predicted despite a marked improvement in water quality (Greening and Janicki, 2006; van der Heide et al., 2007; Valdemarsen et al., 2010; Flindt et al., 2016). Given the present consensus that eelgrass recovery is required to achieve “good ecological conditions” in shallow estuaries (McGlathery et al., 2012), a detailed understanding of key processes affecting the recovery is urgently needed.

* Corresponding author.
E-mail address: mrf@biology.sdu.dk (M.R. Flindt).

https://doi.org/10.1016/j.seares.2022.102177
Received 7 October 2021; Received in revised form 6 January 2022; Accepted 15 February 2022
Available online 18 February 2022

Eelgrass cover in Odense Fjord, Denmark, declined by 90% from 1983 to 2000, and has remained low over the last 20 years. Valdemarsen et al. (2010, 2011) surveyed the growth and losses of seedlings in Odense Fjord as a proxy for the recovery process. They found that physical stress from waves, ballistic impact from drifting macroalgae and bioturbation by lugworms (*Arenicola marina*) was responsible for substantial seedling loss in shallow sandy areas. However, large areas of Odense Fjord have organic-rich and physically unstable sediments caused by eutrophication in the past (Valdemarsen et al., 2014). The high bed mobility in these deeper muddy areas prevents seedling establishment through resuspension-driven light limitation and low sediment anchoring capacity. Thus, sediments in severely impacted areas with organic content of up to 20% has completely lost the ability to support eelgrass, as plants are uprooted and shaded at even low free-stream velocities in the overlying water (Flindt et al., 2016; Lillebø et al., 2011). Even after an extended period of reduced nutrient inputs, the sediments remain organically enriched with frequent resuspension events. Muddy areas with such sub-optimal sediment conditions for eelgrass today cover about 40% of Odense Fjord, and without intervention it will take natural processes several decades to recover sediment stability in the fjord (Valdemarsen et al., 2014).

Although Odense Fjord is a micro-tidal estuary, physical stress is common due to strong winds. Wind speeds exceeding 9 m s^{-1} occur frequently, leading to substantial sediment mobility and consequently a need for yearly restoration of navigation channels by dredging. Harbour authorities report that they remove up to 100.000 m^3 of sandy material after stormy winters. The sand is dredged, loaded to barges and shipped to distant dumping sites. These activities are expensive in labour, shipping, dredging equipment and fuel. It would be a win-win situation, if the material instead is used to consolidate muddy areas by capping activities. Capping with a 10 cm thick sand layer can potentially consolidate muddy sediment and reduce the magnitude and frequency of resuspension. Thus, if unpolluted sand can be acquired from the dredging activities, local capping works will be less costly in labour and shipping/dredging, and have lower CO_2 emissions as well.

Sand-capping has previously been attempted in harbours to dampen the dispersion of sediment borne pollutants. Industrial activities resulted in massive deposits of contaminated sediments in some USA harbours and waterways and sand-capping was identified as a cost-effective technique for on-site remediation (Zeman and Patterson, 1997; Mohan et al., 2000). Sand-capping has also been applied as an effective technique to decrease nutrient release from lake sediments (Kim et al., 2007; Jiao et al., 2020). The feasibility of the sand-capping technique for these purposes is based on geotechnical assessment of sediment holding capacity and stability analyses. The outcome of these analyses also provide evidence for sand-capping as a successful restoration approach to improve ecological conditions in estuarine waters.

The aim of this study was to verify experimentally that sand-capping has potential as a new large-scale restoration approach to stabilize the seabed and improve ecological conditions in eutrophic muddy estuaries, which ultimately may promote seagrass restoration. Our hypotheses are that 1) capping of fluid mud with a 10 cm sand layer is possible without any vertical mixing; 2) sand-capping of mud lowers the magnitude of sediment resuspension. The present study using Odense Fjord sediment should be considered a laboratory test of the ecosystem services provided by this remediating tool (erosion control and water quality improvement). The individual processes are tested and assessed as a prerequisite and preparation for the full Odense Fjord ecosystem study reported by Oncken et al. (2022).

2. Materials & methods

2.1. Study location

Odense Fjord (2.2 m average water depth and 0.3 m tidal amplitude) is divided into a 17 km^2 inner and a 46 km^2 outer part (Fig. 1). The shallow inner fjord (0.8 m average depth) is impacted by freshwater discharge from Odense River, while the outer fjord has a more variable

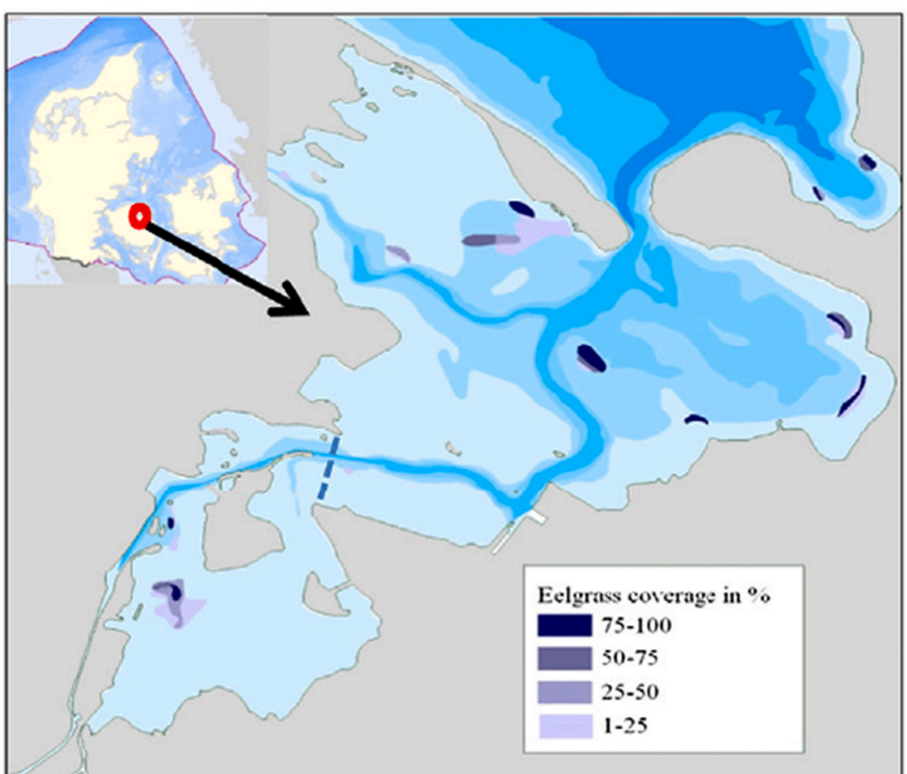

Fig. 1. Map of Odense Fjord with the current eelgrass distribution indicated. The dashed line indicates the boundary between the inner and outer part of the system.

bathymetry (2.7 m average depth) and connects to the open sea (Kattegat) through a narrow opening in the northeast (Fig. 1). Depending on freshwater input and exchange with Kattegat, the salinity varies from 5 to 17 and 15 to 25 in the inner and outer fjord, respectively (Petersen et al., 2009). Odense Fjord has a relatively large catchment area (1046 km^2) providing a substantial nutrient loading primarily due to agricultural runoff. Prior to 1990 the fjord received 2500 t N yr^{-1} and 300 t P yr^{-1}, but after the implementation of several water action plans the nutrient loading has been reduced to the present levels of 1500–2000 t N yr^{-1} and 50–70 t P yr^{-1} (Petersen et al., 2009). This has improved the water quality, diminished growth of opportunistic macroalgae and increased coverage of widgeon grass (*Ruppia maritima*) in the shallow inner fjord. Nevertheless, Odense Fjord does still not comply with the European Water Framework Directive (EU WFD) requirements with respect to eelgrass (*Z. marina*) depth limit, phytoplankton chlorophyll *a* and nutrient concentrations (Petersen et al., 2009). In the reference condition (*i.e.* around year 1900; Ostenfeld, 1908), eelgrass had a depth limit of about 5.5 m and covered substantial areas of Odense Fjord (~50–60%), while the depth limit today is below 2.5 m and only 2% of the estuary is covered by eelgrass patches (Timmermann et al., 2020). The EU WFD Water Management Plan targets a depth distribution for eelgrass of about 4.1 m in Odense Fjord, corresponding to 75% of the eelgrass depth limit in the reference state. Eelgrass has not shown signs of recovery in Odense Fjord – in neither shallow nor deeper areas (unpublished data from the National Monitoring Program) – indicating that light availability is one of the stressors affecting eelgrass distribution in the system (Kuusemäe et al., 2016; Flindt et al., 2016). The combined action by multiple stress factors maintains the estuary in poor to moderate ecological condition. Particularly the organic-rich conditions in large parts of the fjord prevent proper consolidation of the surface sediments that are prone to frequent resuspension events (Canal-Vergés et al., 2010; Kuusemäe et al., 2016).

2.2. Experiment 1: mixing and consolidation after sand-capping of muddy sediments

Sand-capping was mimicked by establishing sediment cores in transparent acrylic chambers ($\phi = 12.5$ cm, height $= 80$ cm, $n = 5$) containing 6 classes of muddy sediment from Odense Fjord to a depth of 25 cm and filled with seawater (salinity of 20). The 6 classes of mud with organic content of about 2, 4, 6, 8, 10 and 16% LOI (loss of ignition) were selected after an initial field survey of sediment water content (WC) and organic content (LOI) at about 100 sampling locations in the fjord. Sediment for the survey was sampled in 5 cm i.d. core tubes, either by hand in shallow water or using piston corers from a boat in deeper water. The upper 2 cm of the sediment was used for WC and LOI determination as described below. Subsequently, the mud classes selected for the experiment were sampled using a sediment dredge from the research vessel Liv II. The sampled mud was forced through a 1 mm mesh without adding water to remove larger particles and benthic fauna before further use.

Passive mixing and consolidation of sediments by gravity were assessed after allowing 10 cm of coarse beach sand (median grain size ~200 μm) to settle on top of the 25 cm deep muddy sediment cores. Cores were sacrificed two weeks after sand-capping for determination of WC, LOI and grain size profiles by sectioning them into 1 cm intervals. Sediment WC was determined as weight loss of wet sediment after drying (24 h, 100 °C) and LOI by combustion of dry sediment (5 h, 520 °C). Sediment granulometry was determined using a Malvern Mastersizer 3000 Particle Size Analyzer. The medium grain size was calculated from the φ distribution of volume size fractions (Bale and Kenny, 2005).

2.3. Experiment 2: flume test of changes in benthic light intensity after sand-capping

Annular flumes (Lundkvist et al., 2007; Neumeier et al., 2007; Kristensen et al., 2013) were used to determine erosion thresholds and benthic light intensity before and after sand-capping of the 6 classes of muddy sediment. Each flume consisted of two transparent acrylic plastic tubes with different diameter (40.6 and 50 cm) fixed onto an acrylic base creating a 4.2 cm wide annulus. The basal area of the channel was 669 cm^2 and given the height of 36 cm, it contained a volume of 24.1 L. The water current in the channel was controlled by an AC-servo motor with an integrated engine driver (MAC motor). The MAC motor was interfaced to a data logging PC. All data were stored by acquisition software that regulated the MAC motor output from voltage to engine rounds per minute (RPM). The MAC motor was attached to the lid of the flume and connected to six equidistantly placed rotating paddles. The MAC motor RPM was calibrated against free-stream current velocity (m s^{-1}) by visually tracking neutrally buoyant particles in the water column. Velocity measurements carried out at various RPM provided the following empirical relationship: Velocity = RPM x 0.0011. Two sampling ports located 15 cm above the base on opposite sides of the outer channel wall of the flume were used for water sampling and turbidity measurements. The turbidity port was equipped with a SeaPoint Turbidity Meter (STM) that detected backscattered light from suspended sediment particles at 880 nm. The STM was interfaced to the data logging PC with continuous logging at a frequency of 1 Hz.

Tests of erosion threshold for each muddy sediment class was first performed in three flumes ($n = 3$) with a mud layer of 10 cm (18 flumes in total) and subsequently the erosion trials were repeated with a sand-cap layer of about 10 cm on top of the mud. The sediment consolidated for 24 h under experimental conditions in estuarine water (temperature: 14 °C; salinity: 18) before erosion trials, while the flume was maintained under a constant free-stream current velocity (u) well below the critical erosion threshold (about 0.02 m s^{-1}). The flume water was aerated to avoid oxygen depletion, but gently enough to prevent sediment disturbance. During erosion trials, the sediment was subjected to increasing current velocities in incremental steps of 5 cm s^{-1} with 15 min duration, *i.e.* the time required to reach a steady state concentration of suspended solids (Fig. 2). Increments continued until a suspended solid concentration (SSC) of 0.5–1.0 g L^{-1} was achieved, or the turbidity signal was saturated. The critical erosion threshold (u_c) was defined as the current velocity where a significant increase in turbidity appeared during the stepwise velocity increments. Water samples were collected at every velocity step (after 15 min) for determination of SSC (g L^{-1}). Sampled water was replaced continuously with ambient estuarine water to avoid water level changes in the flumes. SSC was determined as the dry material recovered after filtering through pre-weighted GF/C filters and related to the corresponding turbidity (NTU) output to establish a calibration curve. Erosion rate (E, g m^{-2} min^{-1}) was calculated from the point at which the erosion threshold was reached for each velocity increment. Thus $E = V^*\Delta SSC/A/\Delta t$, where V is water volume in the flume (L), ΔSSC is the increase in SSC (g L^{-1}) during the time step Δt (min) and A is flume area (m^2).

Samples taken from the flume trials at the end of each velocity increment were used to determine the relationship between suspended solids (SSC), free stream velocity and Lambert-Beer's coefficient (k). For this purpose, the light attenuation coefficient of suspended mud (LOI = 8.0%) and sand (LOI = 0.4%) was measured by suspending each sediment type ($n = 3$) at stepwise increasing concentrations into a transparent acrylic column ($\phi = 30$ cm, h = 200 cm) prefilled with estuarine water (temperature: 14 °C; salinity: 18). A constant concentration of SSC per step was insured by two pumps with inlets at the bottom and outlets at surface of the water column. The light intensity was monitored using a LI-COR Data Logger (LI-1000) placed 50 cm above the bottom. Light attenuation through the water column was calculated using Lambert-Beer's equation: $L_d = L_0 * e^{(-k*d)}$, where L_d is the light intensity at depth

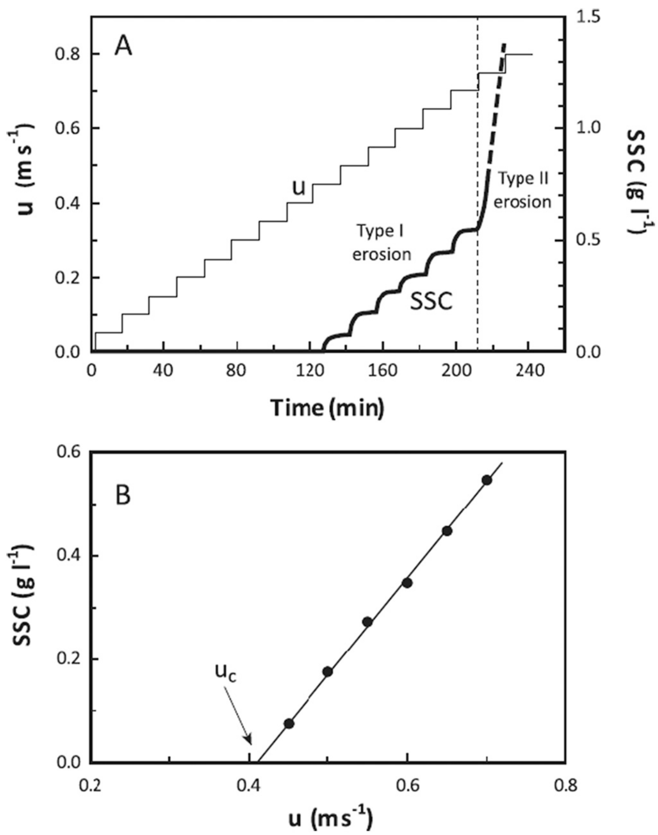

Fig. 2. (A) The outcome of a typical erosion cycle with 0.05 m s^{-1} increments in current velocity. The thin line represents free-stream current velocity (u). The thick line represents the stepwise increase in suspended sediment concentration (SSC) within the Type I erosion zone and the more erratic and unpredictable pattern (dashed part) within the Type II erosion zone. The vertical dashed line separates Type I and Type II erosion. (B) Regression used to estimate the erosion threshold. The critical current velocity (u_c) was estimated as the zero SSC intercept from a regression of measured SSC against u.

d, L_0 is the surface light intensity, and k is the light attenuation coefficient.

2.4. Statistical analyses

Non-linear data was logarithmically or exponentially transformed followed by Pearson correlation statistics to test the relationship between various sediment parameters: loss on ignition (LOI) *vs* water content (WC); consolidation *vs* WC and LOI; erosion threshold *vs* LOI and erosion rate *vs* LOI. The significance level for correlations (α) was 0.05 and the statistical analyses were performed using the SAS procedure Proc NLIN.

3. Results

3.1. Experiment 1: mixing and consolidation after sand-capping of muddy sediments

The sediment survey in Odense Fjord disclosed a range of sediment types from sand to highly organic mud that provided a significant power function between LOI and WC:

$$WC = 23.7*LOI^{0.45}\ (r^2 = 0.87, p < 0.01)$$

Median grain size of the muddy sediment classes ranged from 187 μm in 2.4% LOI low-organic mud (WC of 40%) to 59 μm in 16.2% LOI high-organic mud (WC of 83%). The sand material used in experiment 1 was well sorted with median grain size of 193–220 μm, LOI of 0.3–0.5% and WC of 18–21% (Table 1).

Consolidation/compaction of the different mud classes after sand-capping was in proportion to WC and LOI, and ranged from 3.9% in the low-organic to 13–14% in the high-organic mud, leading to the following significant linear correlations:

$$\text{Consolidation}_{WC} = 0.24*WC–6.3\ (r^2 = 0.81, p < 0.05)$$

$$\text{Consolidation}_{LOI} = 0.66*LOI + 4.9\ (r^2 = 0.73, P < 0.05)$$

The most pronounced impact of sand-capping was expected for mud classes with the smallest grain size. Thus, initial WC and LOI of the richest mud were 4 and 50 times, respectively, higher than those of the applied sand (Fig. 3). These differences were still apparent at the end of the experiment where sand and mud layers remained clearly separated with a narrow vertical mixing zone of 1–2 cm. However, the shape of mixing zones varied among replicates and extended from 9 to 11 cm depth as evident from the high standard deviations. Nevertheless, preservation of the initial characteristics of both sand and mud together with the maintenance of a rather narrow mixing zone after sand-capping, demonstrates that the heavy sand did not sink into the lighter muddy sediment in any of the tested mud classes (Table 1; Fig. 3).

Table 1
Sediment characteristics from sand-capping and flume experiments with erosion thresholds and erosion rates using sediment from 6 muddy stations in Odense Fjord. Organic matter (LOI) and water content (WC) are shown for the applied sand, the mud-sand mixing zone and the mud. Depth extension of the mixing zone and the consolidation/compression of the different muddy sediments are indicated. Values are given as average ± SD.

	Mud class					
Sediment profile data	2% LOI	4% LOI	6% LOI	8% LOI	10% LOI	16% LOI
LOI sand (%)	0.3 ± 0.0	0.3 ± 0.1	0.4 ± 0.1	0.3 ± 0.0	0.5 ± 0.1	0.3 ± 0.0
LOI mixing zone (%)	1.6 ± 0.8	2.3 ± 1.3	4.0 ± 2.4	5.3 ± 2.2	5.3 ± 2.8	7.7 ± 4.6
LOI mud (%)	2.4 ± 0.1	4.1 ± 0.6	6.1 ± 0.7	8.0 ± 0.6	10.3 ± 0.5	16.2 ± 0.2
Depth of LOI mixing zone (cm)	2	2	1	2	2	2
WC sand (%)	18.1 ± 0.5	18 ± 0.36	17.0 ± 0.5	20.0 ± 0.7	21.4 ± 0.8	19.4 ± 0.1
WC mixing zone (%)	31.0 ± 12.7	35 ± 13.33	36.3 ± 18.6	44.4 ± 17.2	47.0 ± 19.7	48.1 ± 13.9
WC mud (%)	38.4 ± 3.9	51 ± 1.92	57.2 ± 2.6	63.1 ± 1.3	69.3 ± 2.1	73.4 ± 4.7
Depth of WC mixing zone (cm)	2	2	1	2	2	2
Median grain size sand (μm)	193	220	197	187	208	221
Median grain size mud (μm)	187	108	84	101	76	59
Consolidation of mud (%)	3.9 ± 0.2	6.4 ± 0.3	14.4 ± 0.4	8.3 ± 0. 5	13.2 ± 0.6	14.1 ± 0.3
Flume exp. Data						
Erosion threshold Sand (cm s^{-1})	40 ± 1.9	37 ± 0.8	38 ± 0.9	40 ± 1.2	40 ± 0.3	40 ± 0.7
Erosion threshold Mud (cm s^{-1})	34 ± 3.3	25 ± 3.1	22 ± 2.6	21 ± 3.7	14 ± 3.6	12 ± 1.9
Settling time Sand (hours)	0.2 ± 0.1	0.2 ± 0.1	0.2 ± 0.0	0.2 ± 0.0	0.18 ± 0.02	0.2 ± 0.0
Settling time Mud (hours)	0.9 ± 0.2	1.1 ± 0.3	1.6 ± 0.5	1.9 ± 0.5	2.41 ± 0.35	5.1 ± 1.3

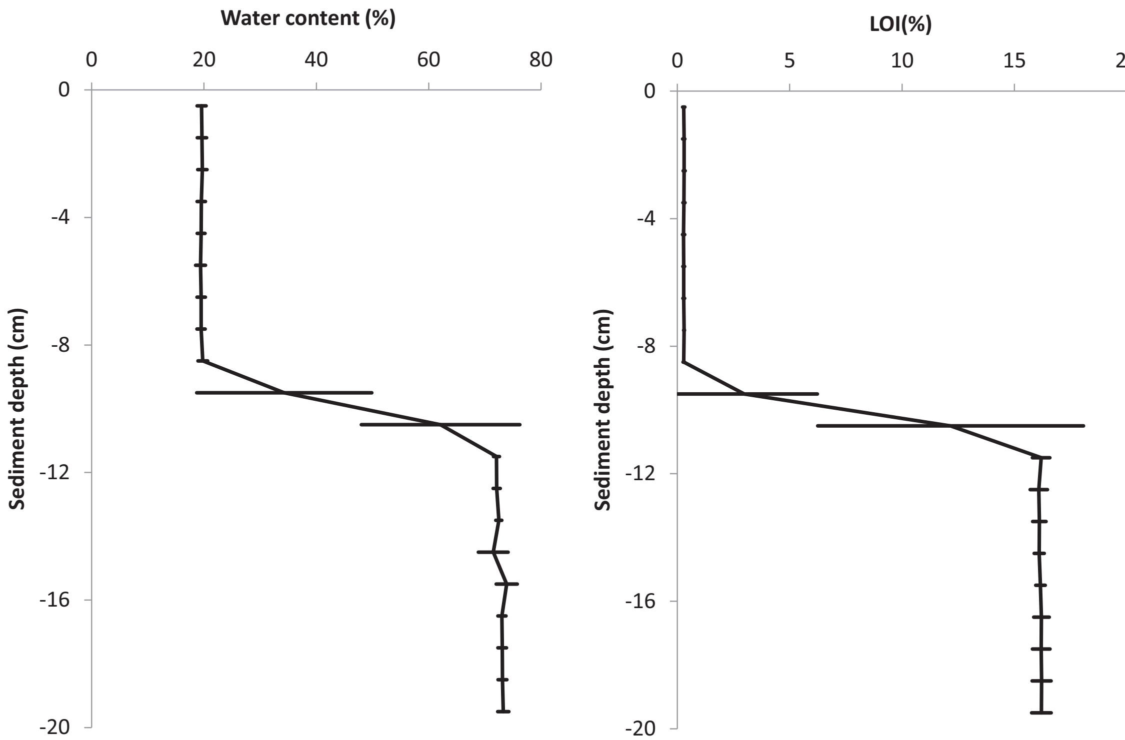

Fig. 3. Final vertical profiles of water content (left) and LOI (loss on ignition) (right) after sand-capping of 16.2% LOI muddy sediment. The horizontal bars represent the standard deviations at each depth ($n = 5$). Results of the other mud classes are summarized in Table 1.

3.2. Experiment 2: flume test of changes in benthic light intensity by sand-capping

An example of the erosion threshold results from the flume with mud (16.2% LOI) alone and after sand-capping of the mud is shown in Fig. 4, while the results from all mud types without and with sand-capping are presented in Table 1. Erosion of mud in the example with 16.2% LOI initiated at a free stream velocity (U) as low as 0.12 m s^{-1} and increased rapidly until the turbidity logger was saturated at a velocity of 0.50 m s^{-1} (Fig. 4). The increase in turbidity per velocity increment generally varied between 0.12 and 0.25 g SSC l^{-1}. The sand-capped mud, on the other hand, first started eroding at 0.40 m s^{-1} and increased with constant turbidity steps of about 0.03 g SSC l^{-1} until at least 0.70 m s^{-1}. The rapid erosion of muddy sediment was evident as elevated turbidity (> 0.1 g SSC l^{-1}) already at a free stream velocity of 0.2 m s^{-1}, while the turbidity of the sand-capped mud always stayed low (< 0.1 g SSC l^{-1}). The most pronounced difference was evident at 0.5 m s^{-1} of free stream velocity, where the turbidity in the mud alone and sand-capped mud was about 0.7 g SSC l^{-1} and 0.05 g SSC l^{-1}, respectively.

The flume assays showed distinct erosion thresholds in all trials (Table 1). For the trials with muddy sediments before sand-capping, the erosion thresholds were inversely related in an exponential pattern to the organic content (Fig. 5). Erosion rates of mud, on the other hand, increased linearly with the LOI content according to: Erosion rate = 0.29 * LOI – 0.09 ($r^2 = 0.81$; $p < 0.05$), albeit with considerable variation among replicates. The erosion threshold in all sand-capped treatments was similar at 0.37 to 0.40 m s^{-1} with low erosion rates ranging from 0.18 to 0.24 g SSC m^{-2} min^{-1} and was independent of the underlying mud composition (Table 1).

The experimentally derived light attenuation coefficient (k) of 0.092 ± 0.039 m^{-1} for the 16.2% LOI mud and 0.057 ± 0.024 m^{-1} for sand provided distinctly different light attenuations in the water column as a

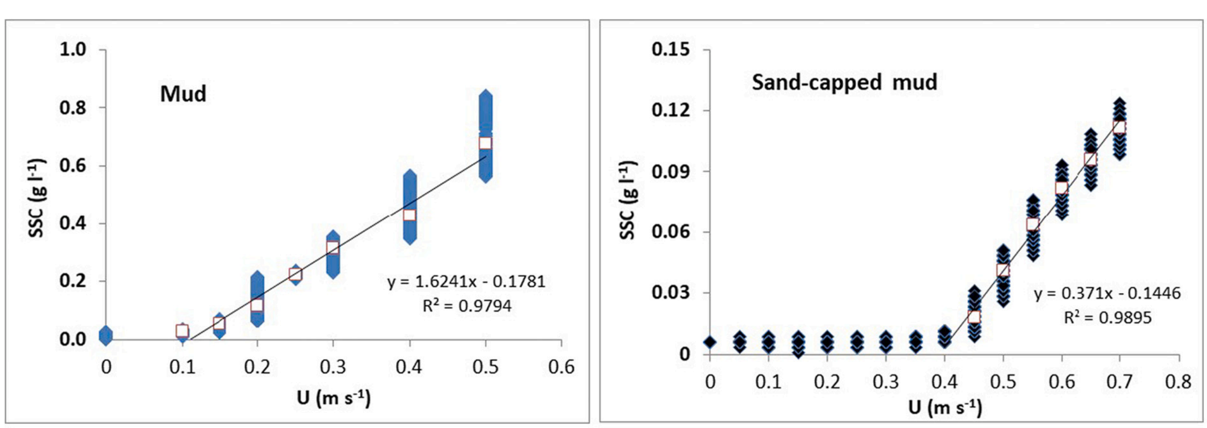

Fig. 4. Example of the erosion threshold (u_c) for 16.2% LOI muddy sediment before (left) and after (right) sand-capping. The regression statistics on both graphs are based on the average value of SSC at each increment. Results of all trials are summarized in Table 1.

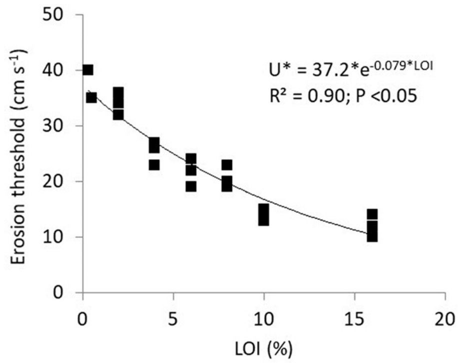

Fig. 5. Sediment erosion threshold correlated to the sediment LOI (loss on ignition). The points are the individual measurements erosion thresholds, and the regression line represents the exponential function shown by the equation.

function of current velocity in the flume (Fig. 6). Light penetrated much deeper in water overlying sand than mud at all current velocities.

4. Discussion

The erosion thresholds of sediments in Odense Fjord vary considerably and are highly dependent on the organic matter and water content (Table 1; Fig. 5). However, the muddy conditions in large parts of the fjord prevent proper consolidation of the surface sediments that experience frequent resuspension events (Kuusemäe et al., 2016; Flindt et al., 2016). Similar critical erosion thresholds and muddy conditions were found by Amos et al. (2004) and Lundkvist et al. (2007) for sediments in Venice Lagoon. The present experiments, that simulate sand-capping of muddy sediments from Odense Fjord, clearly show considerable mud stabilization by a persistent sand layer with little vertical mixing into the underlying mud, even in the most fluid organic-rich sediments (Fig. 3). Accordingly, muddy sediments can be capped with sand regardless of their fluidity and thus increase the overall erosion threshold. Oncken et al. (2022) recently confirmed in a large-scale field study that a sand-cap applied to Odense Fjord sediment remains stable with no vertical mixing for at least one year. Accordingly, the prevailing physical (*e.g.* currents) and biological (*e.g.* reworking by infauna) conditions in Odense Fjord have apparently no impact on the effectiveness and longevity of the sand-cap. Sand-capping is therefore a promising tool to alleviate the negative consequences of organic enrichment in estuaries by preventing sediment erosion, reducing turbidity and improving water quality. The approach may also prevent uprooting of plants, which often occur at very low water current velocities when the sediments are organic-rich. The low-organic sand applied in the present experiments must be appropriate for the purpose, since sediment WC and LOI should be below 40% and 2–3%, respectively, to support seedling performance of eelgrass at current velocity thresholds of up to 50 cm s^{-1} (Lillebø et al., 2011).

The significant exponential relationship between sediment organic matter content and erosion thresholds (Fig. 5) provides an approach to determine the type of sand needed for appropriate consolidation of muddy sediments. Using coarser sand with lower organic content than applied in the present experiment may increase the erosion threshold even further than observed here (>40 cm s^{-1}). It must be noted, though, that the applied flume setup only generates laminar currents as a proxy for the physical force added to the sediment and does not simulate true wave exposure. Thus, 3D hydrodynamic model simulations have demonstrated a high frequency of sediment resuspension due to wave action (Kuusemäe et al., 2016). Further experiments are therefore required to elucidate the impact of such pulsing wave pressure on sediments capped with different types of sand.

By extrapolating the eroded SSC mass from the flume study with muddy sediment to a water column as deep as 4 m, it is evident that even at very low current velocities the light attenuation in the water column is substantial (Fig. 6). Thus, for current velocities of 30 cm s^{-1} in water overlying 16% LOI sediment, the light intensity at a depth of 0.4 m is just 20% of that at the surface. Simulations of water overlying sand-capped sediment showed that this dampened light intensity is first reached at a depth of about 4 m. However, the relatively high light attenuation coefficient even for sand was unexpected and most probably caused by light absorbance due to traces of organic matter (LOI = 0.4%) coating on the sand grains. Nevertheless, the light attenuation with depth was much higher in water overlying mud than sand and increased dramatically with current velocity. These results substantiate the potential of sand-capping for improving light intensity and penetration depth in an otherwise turbid estuary like Odense Fjord. The large-scale study of Oncken et al. (2022) confirmed that sand-capping of ~1 ha muddy

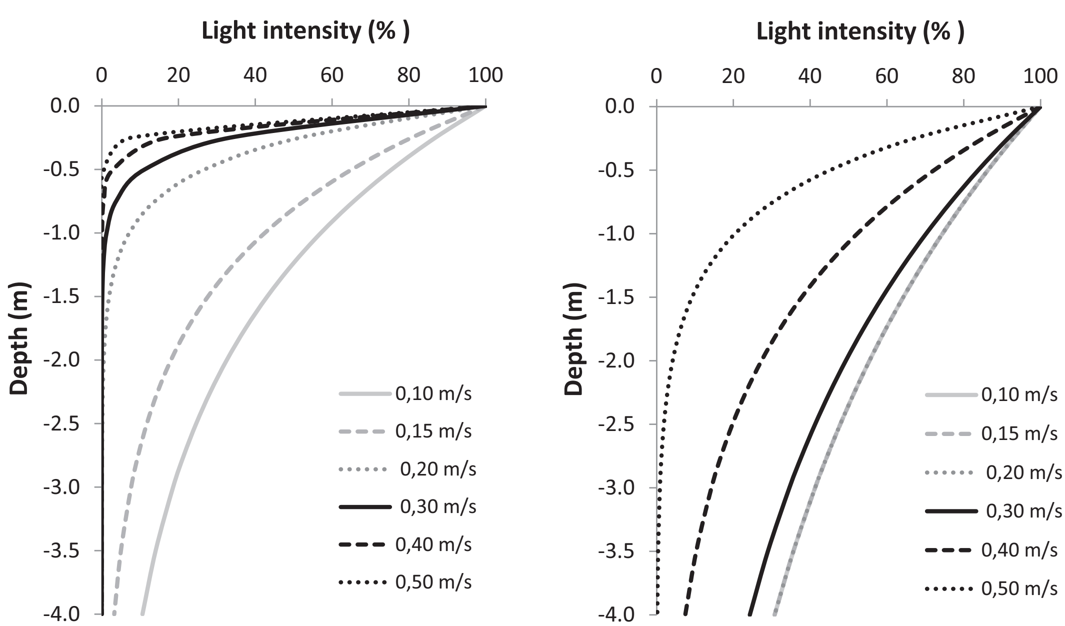

Fig. 6. Vertical light attenuation in the water column overlying 16.2% LOI muddy sediment before (left) and after (right) sand-capping. Note that for sand, 0.10, 0.15 and 0.20 m s^{-1} lines are on top of each other.

sediment in Odense Fjord increased the light intensity by up to 22% at 2 m water depth.

Sand-capping can therefore potentially provide support for eelgrass growth in deep areas. Lee et al. (2007) reported that eelgrass has zero net production (compensation irradiance) at a light intensity of 85 μE m^{-2} s^{-1}, and Orth et al. (2006b) found a saturated production at 485 μE m^{-2} s^{-1}. For comparison, field tests in Odense Fjord revealed that positive net growth of eelgrass seedlings required average benthic light intensities >200 μE m^{-2} s^{-1} (Flindt et al., 2016). The higher light threshold observed in Odense Fjord is partly caused by elevated turbidity in the near-bottom 20–30 cm of the water column, as typically observed over organic-rich sediments (Kenworthy et al., 2014). Using the threshold of 200 μE m^{-2} s^{-1} as a growth-season average, we document the service provided by sand-capping compared to the present condition with untreated muddy areas (Fig. 6). At muddy sites, light only supports eelgrass recovery at low current velocities (< 15 cm s^{-1}) and only down to a depth of 1.75 m. Erosion thresholds increase to about 40 cm s^{-1} after sand-capping, and eelgrass recovery may be possible down to 3.5 m or more with current velocities ≤30 cm s^{-1}. Accordingly, past eelgrass transplantations in muddy areas of Odense Fjord have failed at 2.5 m depth (Lange, unpublished). Petersen et al. (2021) stressed that frequent resuspension and low anchoring capacity of eelgrass caused by organic-rich sediments is a general threat to the success of eelgrass transplantations, not only in Odense Fjord, but in most Danish coastal waters. Thus, widespread use of sand-capping could possibly provide a greater area of benthic habitat that is suitable for eelgrass growth than previously anticipated.

It should be mentioned that the present results do not include dynamic changes in sediment biostability caused by benthic diatoms. These can, under optimal light conditions (>10 μE m^{-2} s^{-1}), more than double the erosion threshold of muddy sediments (Paterson et al., 2000; Quaresma et al., 2004; Lundkvist et al., 2007). This may potentially diminish the difference in light conditions between muddy and sandy areas. However, most estuarine areas have several destabilizing forces that disturb the diatom biostability of muddy areas: 1) Bedload transport of scouring macroalgae may occur at low current velocities (Flindt et al., 2004; Flindt et al., 2007; Canal-Vergés et al., 2010); 2) Grazing on benthic diatoms by benthic fauna like *Hydrobia ulva* (Kristensen et al., 2013); 3) Particle reworking by infauna like the polychaete *Hediste diversicolor* (Widdows et al., 2009).

McGlathery et al. (2012) found that eelgrass must cover about 20% of an area before the bed itself improves light condition by preventing resuspension, and at 50% coverage the turbidity is reduced to 1/3. As the eelgrass coverage in many Danish estuaries, like Odense Fjord, is below 2%, this eelgrass ecosystem service (*e.g.* turbidity reduction) is not provided. Furthermore, Odense Fjord has today lost about 40% of the sandy areas that previously supported eelgrass growth. Sand-capping may be the solution to alleviate problems with the expanding mud deposits and high turbidity. The improved light climate and increased anchoring capacity following sand-capping will enhance growth of eelgrass, but it is still uncertain how widespread sand-capping of muddy areas in Odense Fjord should be before turbidity improves on an ecosystem scale. For this purpose, a modelling scenario has revealed that sand-capping of about 100 ha muddy sediment is required to obtain a significant large-scale improvement of benthic light conditions (Bruhn et al., 2020). Although, large sand volumes for sand-capping activities can be acquired from the dredging of the many navigation channels, it is a logistically challenge to transplant eelgrass at such large scale. Instead, we suggest a patchwork of smaller transplantations that can support natural eelgrass recovery either through seed-based or vegetative expansion. By this approach, we expect that sand-capping can increase eelgrass coverage and improve the associated ecosystem services by retaining nutrients, reducing water turbidity and diminishing phytoplankton production. This will probably lead to further improvement of the benthic light climate and positive feedback mechanisms are initiated. The companion paper of Oncken et al. (2022) corroborates that *in situ* sand-capping at a scale of 1–2 ha stabilizes muddy sediments and improves light conditions. However, more work on even larger scales combined with eelgrass transplantation is required to verify these trends and elucidate any unforeseen challenges.

Declaration of Competing Interest

The authors have no conflicts of interest to declare for the submitted study. All co-authors have seen and agree with the contents of the manuscript and there is no financial interest to report. We certify that the submission is original work and is not under review at any other publication.

Acknowledgements

This project was funded by the Danish Strategic Science Foundation (grant # 09-063190/DSF) and by the Ministry of the Environment and Food (grant # 33010-NIFA-16-650). AIS was funded by Portuguese national funds through the FCT - Foundation for Science and Technology, I.P., under the project CEECIND/00962/2017 and the Post-Doc grant SFRH/BPD/79537/2011. AIS also thanks FCT/MCTES for the financial support to CESAM (UIDP/50017/2020+ UIDB/50017/2020).

References

Amos, C., Cappucci, S., Bergamasco, A., Umgiesser, G., Bonardi, M., Cloutier, D., Flindt, M.R., De Nat, L., Cristante, S., 2004. The stability of tidal flats in Venice Lagoon – the results of in situ measurements using two benthic annular flumes. J. Mar. Syst. 51, 211–241.

Bale, A.J., Kenny, A.J., 2005. Sediment analysis and seabed characterization. In: Eleftheriou, A., McIntyre, A. (Eds.), Methods for the Study of Marine Benthos. Blackwell Science, Oxford.

Boström, C., Baden, S., Bockelmann, A.-C., Dromph, K., Fredriksen, S., Gustafsson, C., Krause-Jensen, D., Möller, T., Nielsen, S.L., Olesen, B., Olsen, J., Pihl, L., Rinde, E., 2014. Distribution, structure and function of Nordic eelgrass (*Zostera marina*) ecosystems: implications for coastal management and conservation. Aquat. Conserv. Mar. Freshwat. Ecosyst. 24, 410–434.

Bruhn, A., Flindt, M.R., Hasler, B., Krause-Jensen, D., Larsen, M.M., Maar, M., Petersen, J.K., Timmermann, K., 2020. Marine virkemidler – beskrivelse af virkemidlernes effekter og status for vidensgrundlag. In: Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi, 126. - Videnskabelig rapport nr. 368. http://dce2.au.dk/pub/SR368.pdf.

Canal-Vergés, P., Kristensen, E., Vendel, M., Flindt, M.R., 2010. Resuspension created by bedload transport of macroalgae: implications for ecosystem functioning. Hydrobiologia 649, 69–76.

de los Santos, C.B., Krause-Jensen, D., Alcoverro, T., Marbà, N., Duarte, C.M., Van Katwijk, M.M., Pérez, M., Romero, J., Sánchez-Lizaso, J.L., Roca, G., Jankowska, E., Pérez-Lloréns, J.L., Fournier, J., Montefalcone, M., Pergent, G., Ruiz, J.M., Cabaço, S., Cook, K., Wilkes, R.J., Moy, F.E., Trayter, G.M.-R., Arañó, X.S., de Jong, D.J., Fernández-Torquemada, Y., Auby, I., Vergara, J.J., Santos, R., 2019. Recent trend reversal for declining European seagrass meadows. Nat. Commun. 10, 1–8.

Flindt, M.R., Neto, J., Amos, C.L., Pardal, M.A., Bergamasco, A., Pedersen, C.B., Andersen, F.Ø., 2004. Plant bound nutrient transport mass transport in estuaries and lagoons. In: Nielsen, S.L., Banta, G.T., Pedersen, M.F. (Eds.), Estuarine nutrient cycling: The influence of primary producers. Aquatic Ecology Book Series, vol 2. Springer, Dordrecht.

Flindt, M.R., Pedersen, C.B., Amos, C.L., Levy, A., Bergamasco, A., Friend, P.L., 2007. Transport, sloughing and settling rates of estuarine macrophytes: mechanisms and ecological implications. Cont. Shelf Res. 27, 1096–1103.

Flindt, M.R., Rasmussen, E.K., Valdemarsen, T., Erichsen, A., Kaas, H., Canal-Vergés, P., 2016. Using a GIS-tool to evaluate potential eelgrass reestablishment in estuaries. Ecol. Model. 338, 122–134.

Greening, H., Janicki, A., 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. Environ. Manag. 38, 163–178.

Greve, T.M., Krause-Jensen, D., Rasmussen, M.B., Christensen, P.B., 2005. Means of rapid eelgrass (*Zostera marina* L) recolonisation in former dieback areas. Aquat. Bot. 82, 143–156.

Hauxwell, J., Valiela, I., 2004. Effects of nutrient loading on shallow seagrass dominated coastal systems: patterns and processes. In: Nielsen, S., Banta, G., Pedersen, M. (Eds.), Estuarine nutrient cycling: The influence of primary producers. Aquatic Ecology Book Series, vol 2. Springer, Dordrecht.

Jiao, Y., Xu, L., Li, Q., Gu, S., 2020. Thin-layer fine-sand capping of polluted sediments decreases nutrients in overlying water of Wuhan Donghu Lake in China. Environ. Sci. Pollut. Res. 27, 7156–7165.

Kenworthy, W.J., Gallegos, C.L., Costello, C., Field, D., di Carlo, G., 2014. Dependence of eelgrass (*Zostera marina*) light requirements on sediment organic matter in

Massachusetts coastal bays: implications for remediation and restoration. Mar. Pollut. Bull. 83, 446–457.

Kim, G., Jeong, W., Choi, S., Khim, J., 2007. Sand capping for controlling phosphorus release from lake sediments. Environ. Technol. 28, 381–389.

Kristensen, E., Neto, J.M., Lundkvist, M., Frederiksen, L., Pardal, M.A., Valdemarsen, T., Flindt, M.R., 2013. Influence of benthic macro-invertebrates on the erodability of estuarine cohesive sediments: density- and biomass-specific responses. Estuar. Coast. Shelf Sci. 142, 32–40.

Kuusemäe, K., Rasmussen, E.K., Vergés, P.C., Flindt, M.R., 2016. Modelling stressors on the eelgrass recovery process in two Danish estuaries. Ecol. Model. 333, 11–42.

Lee, K.S., Park, S.R., Kim, Y.K., 2007. Effects of irradiance, temperature, and nutrients on growth dynamics of seagrasses: a review. J. Exp. Mar. Biol. Ecol. 350, 144–175.

Lillebø, A.I., Flindt, M.R., Cardoso, P.G., Leston, S., Dolbeth, M., Pereira, M.E., Duarte, A. C., Pardal, M.A., 2011. Restoration of seagrass community to reverse eutrophication in estuaries. In: Eric, W., Donald, M. (Eds.), Treatise on Estuarine and Coastal Science. Academic Press, Waltham, pp. 151–164.

Lundkvist, M., Grue, M., Friend, P., Flindt, M.R., 2007. The relative contributions of physical and microbiological factors to cohesive sediment stability. Cont. Shelf Res. 27, 1143–1152.

Marion, S.R., Orth, R.J., 2010. Factors influencing seedling establishment rates in *Zostera marina* and their implications for seagrass restoration. Restor. Ecol. 18, 549–559.

McGlathery, K.J., Reynolds, L.K., Cole, L.W., Orth, R.J., Marion, S.R., Schwarzschild, A., 2012. Recovery trajectories during state change from bare sediment to eelgrass dominance. Mar. Ecol. Prog. Ser. 448, 209–221.

Mohan, R.K., Brown, M.P., Barnes, C.R., 2000. Design criteria and theoretical basis for capping contaminated marine sediments. Appl. Ocean Res. 22, 85–93.

Neumeier, U., Friend, P.L., Gangelhof, U., Lunding, J., Lundkvist, M., Bergamasco, A., Amos, C.L., Flindt, M.R., 2007. The influence of fish feed pellets on the stability of seabed sediment, a laboratory flume investigation. Estuar. Coast. Shelf Sci. 75, 347–357.

Oncken, N.S., Lange, T., Kristensen, E., Quintana, C.O., Steinfurth, R.C., Flindt, M.R., 2022. Sand-capping – a large-scale approach to restore organic-enriched estuarine sediments. Mar. Environ. Res. 173, 105534.

Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K. L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M., Williams, S.L., 2006a. A global crisis for seagrass ecosystems. Bioscience 56, 987–996.

Orth, R.J., Luckenbach, M.L., Marion, S.R., Moore, K.A., Wilcox, D.J., 2006b. Seagrass recovery in the Delmarva coastal bays, USA. Aquat. Bot. 84, 26–36.

Ostenfeld, C.H., 1908. Ålegræssets (*Zostera marina*'s) udbredelse i vore farvande. In: CGJ, Petersen (Ed.), Beretning til Landbrugsministeriet fra den danske biologiske station, vol XVI. Centraltrykkeriet, København, pp. 1–61 (in Danish).

Paterson, D.M., Tolhurst, T.J., Kelly, J.A., Honeywill, C., de Deckere, E.M.G.T., Huet, V., Shayler, S.A., Black, K.S., de Brouwer, J., Davidson, I., 2000. Variations in sediment properties, Skeffling mudflats, Humber estuary, UK. Cont. Shelf Res. 20, 1373–1396.

Petersen, J.D., Rask, N., Madsen, H.B., Jørgensen, O.T., Petersen, S.E., Nielsen, S.V.K., Pedersen, C.B., Jensen, M.H., 2009. Odense pilot river bas implementation of the EU water framework directive in a shallow eutrophic estuary (Odense Fjord Denmark) its upstream catchment. Hydrobiologia 629, 71–89.

Petersen, J.K., Timmermann, K., Bruhn, A., Rasmussen, M.B., Boderskov, T., Schou, H.J., Erichsen, A.C., Thomsen, M., Holbach, A., Tjørnløv, R.S., Lange, T., Canal-Vergés, P., Flindt, M.R., 2021. Marine virkemidler: Potentialer og barrierer (Danish). DTU Aqua. DTU Aqua-report No. 385–2021.

Quaresma, V.D., Amos, C.L., Flindt, M.R., 2004. Influences of biological activity and consolidation time on laboratory cohesive beds. J. Sediment. Res. 74, 184–190.

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S., Bujang, J.S., Calumpong, H.P., Carruthers, T.J.B., Coles, R.G., Dennison, W.C., Erftemeijer, P.L.A., Fortes, M.D., Freeman, A.S., Jagtap, T.G., Kamal, A.H.M., Kendrick, G.A., Kenworthy, W.J., La Nafie, Y.A., Nasution, I.M., Orth, R.J., Prathep, A., Sanciangco, J.C., Tussenbroek, B.V., Vergara, S.G., Waycott, M., Zieman, J.C., 2011. Extinction risk assessment of the world's seagrass species. Biol. Conserv. 144, 1961–1971.

Timmermann, K., Christensen, J.P.A., Erichsen, A., 2020. Referenceværdier og grænseværdier for ålegræsdybdegrænser til brug for vandområdeplanerne. Aarhus Universitet, DCE – Danish Center for Environment and Energy.

Valdemarsen, T., Canal-Vergés, P., Kristensen, E., Holmer, M., Kristiansen, M.D., Flindt, M.R., 2010. Vulnerability of *Zostera marina* seedlings to physical stress. Mar. Ecol. Prog. Ser. 418, 119–130.

Valdemarsen, T., Wendelboe, K., Egelund, J.T., Kristensen, E., Flindt, M.R., 2011. Burial of seeds and seedlings by the lugworm *Arenicola marina* hampers eelgrass (*Zostera marina*) recovery. J. Exp. Mar. Biol. Ecol. 410, 45–52.

Valdemarsen, T., Quintana, C.O., Kristensen, E., Flindt, M.R., 2014. Recovery of organic-enriched sediments through microbial degradation: implications for eutrophic estuaries. Mar. Ecol. Prog. Ser. 503, 41–58.

van der Heide, T., van Nes, E.H., Geerling, G.W., Smolders, A.J.P., Bouma, T.J., van Katwijk, M.M., 2007. Positive feedbacks in seagrass ecosystems: implications for success. Conserv. Restoration Ecosys. 10, 1311–1322.

van Katwijk, M.M., Thorhaug, A., Marbà, N., Orth, R.J., Duarte, C.M., Kendrick, G.A., Althuizen, I.H.J., Balestri, E., Bernard, G., Cambridge, M.L., Cunha, A., Durance, C., Giesen, W., Han, Q., Hosokawa, S., Kiswara, W., Komatsu, T., Lardicci, C., Lee, K.-S., Meinesz, A., Nakaoka, M., O'Brien, K.R., Paling, E.I., Pickerell, C., Ransijn, A.M.A., Verduin, J.J., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. J. Appl. Ecol. 53, 567–578.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. U. S. A. 106, 12377–12381.

Widdows, J., Brinsley, M.D., Pope, N.D., 2009. Effect of *Nereis diversicolor* density on the erodability of estuarine sediment. Mar. Ecol. Prog. Ser. 378, 135–143.

Zeman, A.J., Patterson, T.S., 1997. Preliminary results of demonstration capping project in Hamilton harbour. Water Quality Res. 32, 439–452.