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Short Communication

Ecosystem engineering by annual intertidal seagrass beds: Sediment accretion and modification

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Abstract

Seagrasses are generally known as ecosystem engineers, as they reduce flow velocities in their canopies. In perennial subtidal meadows, this usually leads to increased net sedimentation rates and reduction of the grain size. The present study aims to describe the contribution of annual seagrass populations to these processes and elucidate the temporal dynamics. Sediment accretion and grain size modification were experimentally tested by transplanting seedlings of an annual intertidal eelgrass population to an unvegetated tidal flat. Within the planting units (79 shoots m^{-2}) 4.7 mm of sediment accreted, whereas in the most dense parts of these units (199 shoots m^{-2}) accretion amounted to 7.1 mm. The silt fraction (<63 μm) increased and the sand fraction (63–500 μm) decreased in the eelgrass beds, which provides evidence that higher silt content in seagrass beds is the result and not the cause of seagrass presence. Annual intertidal eelgrass beds significantly contribute to the immobilisation of sediment during the growing season with its magnitude depending on canopy density. During winter, the accumulated sediments were released again and could even induce additional erosion. Possible consequences of these sediment dynamics for the larger scale functioning of estuarine ecosystems are discussed.

Keywords: coastal management; ecosystem service; eelgrass; erosion; sedimentation; transplantation; Wadden Sea; *Zostera marina*

1. Introduction

Macrophytes are ecosystem engineers as they can modify their abiotic environment (Jones et al., 1997). For example, seagrasses are known to reduce hydrodynamic energy from currents and waves (Gambi et al., 1990; Bouma et al., 2005), increase sediment accretion (Gacia et al., 1999, 2003), alter sediment quality (Fonseca, 1996; Koch, 1999) and stabilize sediments (Fonseca, 1989). With respect to sediment dynamics, several studies documented sediment erosion following seagrass losses (e.g. Ramage and Schiel, 1999; de Falco et al., 2000) and erosion prevention by seagrass presence (Amos et al., 2004; Thompson et al., 2004; Adriano et al., 2005). At high wave energy locations, both these effects are usually low (van Keulen and Borowitzka, 2003; Paling et al., 2003). Regarding erosion prevention this was confirmed by Fonseca and Koehl (2006), who observed elevated levels of turbulence within small eelgrass patches to produce a surface with coarser sediment.

Evidence of sediment accumulation within seagrass meadows remains limited to perennial subtidal populations (e.g. Fonseca and Bell, 1998; Gacia et al., 1999; Koch, 2001) with, to our knowledge, no direct measurements available for annual intertidal populations. Hence it remains unresolved whether the commonly observed higher silt fraction in annual intertidal seagrass vegetations (Heiss et al., 2000; Granata et al., 2001) is induced by the presence of the seagrass or vice versa. In addition, the temporal dynamics of sediment

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accumulation may be expected to differ between perennial subtidal populations and annual intertidal populations, as the latter will not be present during winter.

As in many other areas in NW Europe, the perennial subtidal form of *Zostera marina* L. has disappeared from the Wadden Sea during the last century (Giesen et al., 1990; de Jonge et al., 1996). However, the annual intertidal form of eelgrass still exists in northern and eastern parts of the Wadden Sea and was recently reintroduced in the western part (van Katwijk et al., 1998, 2000; Bos and van Katwijk, 2007). Regarding our current lack of knowledge on how annual intertidal seagrasses affect sediments, the impact of large scale reintroductions is hard to predict. The present study aims to assess to what extent eelgrass beds contribute to sedimentation and sediment modification in intertidal habitats, thereby also elucidating the temporal dynamics.

2. Materials and methods

Eelgrass seedlings were collected at the tidal flat *Hond/Paap* in the Eems estuary in the eastern Dutch Wadden Sea on 10 June 2003 and were stored and transported at a mean temperature of 11 °C. On 11 June 2003, the seedlings were transplanted to the tidal flat *Balgzand* in the western Wadden Sea at +0.06 m (±0.02) mean sea level (MSL) about 500 m offshore. Three hexagonal-shaped planting units (diameter 1.8 m) with 37 seedlings at a density of 14 plants m^{-2} (Bos and van Katwijk, 2007) were paired with control units without plants at a distance of 7.5 m (Fig. 1). Replicates were 30 m apart. Number of surviving plants and shoot density were determined for each planting unit (PU) on 15 July, 27 August, 24 September and 27 October 2003.

A Stanley Compulevel was used to measure the relative height of the sediment (in mm) at the following dates: 17 June 2003, 10 September 2003 and 20 April 2004. This device measures elevations by detecting pressure differences in an enclosed fluid-filled hose-system (Fig. 1). In each pair of planting and control units, sediment heights were measured at 28 points by using a spatial grid (Fig. 1). This grid was obtained by temporarily placing two scaled rods at the sediment aside from the unit and perpendicularly moving a third scaled rod. Two small sticks were used to permanently mark the position of one of the rods. For each pair of experimental units, a 1-m PVC pipe was hammered into the sediment to serve as calibration point for the duration of the experiment (Fig. 1). Absolute height of the three calibration points was measured with differential GPS on 9 September 2003. The significance of observed differences between the unit pairs ($n = 3$) was calculated with a paired *T*-test. Normality was tested using the Kolmogorov–Smirnov test.

To study sediment characteristics, three 10-cm sediment cores with a diameter of 2.8 cm were collected per unit on 27 August 2003 and mixed before storage at −18 °C. The samples were freeze-dried and sieved (1 mm) to remove small pieces of shell and were analyzed with a Malvern Laser Particle Sizer. Sediment particles were categorized into a silt fraction (<63 µm) and a sand fraction (63–500 µm). Sediment characteristics were tested with both the paired *T*-test and the student *T*-test.

3. Results

Mean number of surviving eelgrass transplants slightly decreased from 78% (±12%) in July, the first month after

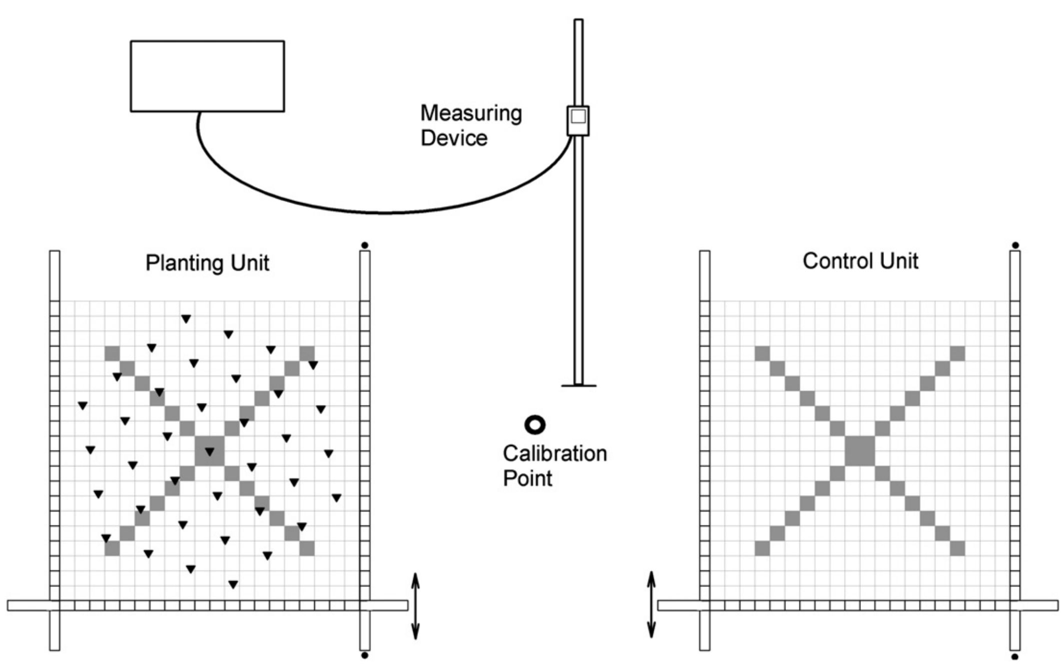

Fig. 1. A schematic overview of the measurements with a planting and control unit, the measuring device and the calibration point. Inverted triangles in the planting unit represent seagrass plants. Twenty-eight sediment heights were measured in each unit as represented by the grey squares. The measuring device consisted of a main unit, a fluid-filled hose and a display attached to a pole. This figure is not on scale.

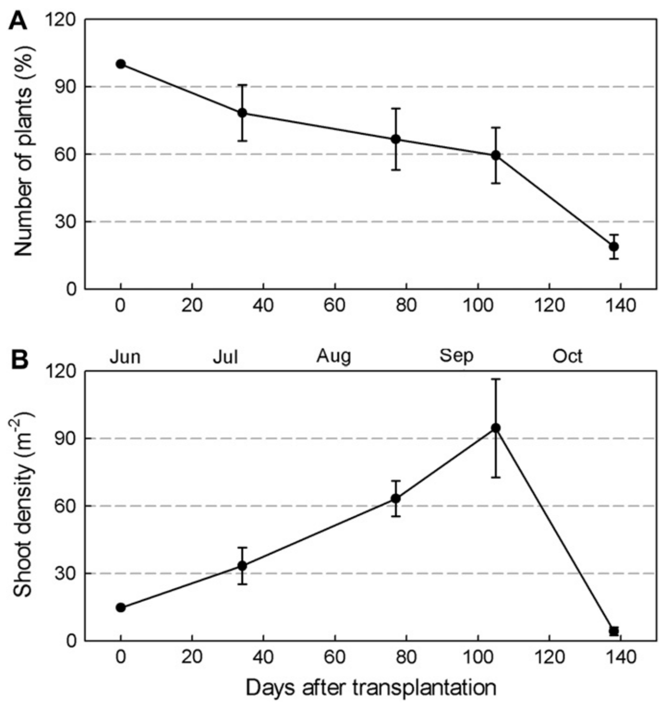

Fig. 2. Eelgrass characteristics after transplantation on 11 June 2003 (=day zero). (A) Number of surviving plants (%), (B) Eelgrass density (shoots m^{-2}).

transplantation, to 60% (±12%) in September (Fig. 2A). Surviving plants grew well, which resulted in a mean shoot density of 79 (±23) shoots m^{-2} in September (Fig. 2B). By that time, surviving plants were not evenly distributed over the planting units (PUs) and the shoot density was observed to be as high as 199 (±56) shoots m^{-2} in parts of the PUs. Most of the eelgrass' aboveground tissues disappeared in October as a result of the natural end-of-season die-off.

In September 2003, significantly more sediment had accreted (Paired T-test, $p < 0.01$) in the PUs than in the controls (Fig. 3). The accretion difference between the PUs and controls was 4.7 mm. In the dense eelgrass parts of the PUs, local accretion amounted to 7.1 mm. In early winter, when the aboveground parts of the eelgrass had largely disappeared, the locations with increased sediment level were still visible (Fig. 4). In spring 2004, significantly less sediment had accreted (Paired T-test, $p < 0.01$) in the PUs than in the controls (Fig. 3). By then the accretion difference between the PUs and controls was −6.2 mm. The accretion difference between the average PUs and those parts of the PU that had the highest eelgrass density during the previous summer had disappeared (Paired T-test, $p > 0.05$).

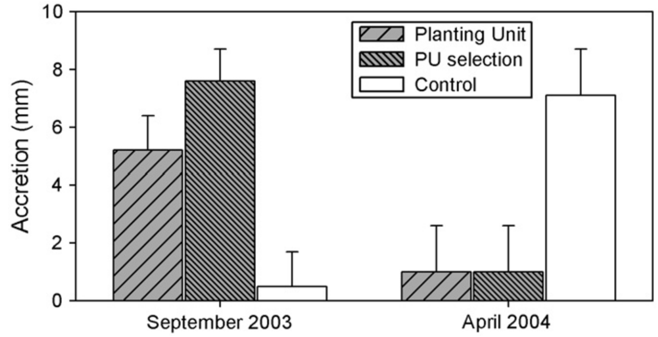

Fig. 3. Mean sediment accretion (mm) in planting units (PU), eelgrass covered selections of a PU and the control units in September 2003 (3 months after transplantation) and in April 2004 (10 months after transplantation).

Late summer, the silt fraction (<63 μm) was significantly higher in the planting units than in the controls (Table 1). In line with this, the sand fraction (63−500 μm) was found to have significantly decreased in the PUs in comparison to the control units. Organic carbon and C:N ratio were both found not to be significantly different between eelgrass and control units (Table 1).

4. Discussion

The present study revealed that annual, intertidal eelgrass patches can accrete 5−7 mm of sediment during the growing season in the Wadden Sea. Accretion rate of subtidal, perennial *Posidonia* beds was 2 mm per year in the Mediterranean Sea (Gacia and Duarte, 2001; Gacia et al., 2002). The relatively high rate of accretion in the present study may have been caused by the much larger suspended sediment concentrations in the Wadden Sea than in the Mediterranean Sea. Harlin et al. (1982) found 25 mm of sediment to be eroded following experimental removal of high density annual eelgrass plants (approximately 1000 shoots m^{-2}). This is in good agreement with our observation that sediment accretion rates increase upon planting density of seagrass meadows. In addition we observed the smaller sediment fractions to be higher in the eelgrass units than in the controls. This relationship was described earlier by Heiss et al. (2000) and Granata et al. (2001). The present study, however, in which seagrass was planted at an unvegetated tidal flat, is the first to prove that finer sediments within seagrass beds are the *result* and not the cause of seagrass presence. This ability of increasing the fine sediment content at a newly colonised location may increase the chance of survival of an eelgrass bed by increasing the nutrient availability (Koch, 2001) and by supporting germination (van Katwijk and Wijgergangs, 2004).

The patch size used in the present study, although it represents a major transplantation effort, remains relatively small when compared to the average size of natural seagrass meadows. Hence it is important to consider what present findings mean if the seagrass patches would be more extended. The observed sediment accretion, despite the small size of the seagrass patches, agrees well with sediment measurements on similar sized patches of artificial vegetations (Bouma et al., 2007) and perennial salt marsh species (Castellanos et al., 1994; van Hulzen et al., 2007). Studies on marsh species also showed that the sediment accumulation and the silt content increased with patch size (Castellanos et al., 1994; van Hulzen et al., 2007). Combined with the observation that reduction of hydrodynamic energy in both seagrass canopies as well as marsh vegetations requires some distance before reaching the maximal effect (Bouma et al., 2005, 2007;

Fig. 4. A planting unit as observed on 27 October 2003. The aboveground parts of the eelgrass plants largely disappeared and erosion of previously accreted sediment had started. However, the overall elevation of the patch is still clearly visible. The silty character of the high areas can still be observed by the lack of ripples.

Fonseca and Koehl, 2006), it is to be expected that more sediment per area would accumulate within larger seagrass patches. Thus, the observed effect is a minimum estimate of what may be expected in large-scale reintroductions.

Whereas perennial seagrass vegetations accumulate sediment, at least on a net yearly basis (e.g. Mateo et al., 1997), our study showed that annual populations release the accumulated sediments during winter. Remarkably, during winter, the formerly vegetated units showed significant erosion in comparison to the control units. We can only speculate about the possible cause of this observation (e.g., rhizomes and decaying seagrass plants may have attracted sediment destabilising fauna or have enhanced erosion by scouring). More interestingly, the summer accretion and winter release of sediments by annual seagrass beds may have implications for the functioning of estuaries as a whole, when regarded with respect to the sediment dynamics of salt marshes. Sediment input in salt marshes is largely controlled and limited by suspended sediment concentrations (van Proosdij et al., 2006). The sediment trapping capacity of salt marshes peaks in November (Neumeier, 2005), which coincides with the start of the sediment release from seagrass beds after shoot loss in autumn. The significance of such seagrass beds ‘feeding’ salt marshes with sediment should be further studied and tested, before drawing any conclusions on the importance of this process.

Table 1
Silt (<63 μm), sand (63–500 μm) fractions, organic C (%) and C:N ratio of sediment in planting units (PU) and control units on 27 August 2003

	Sediment fractions		Organic C (%)	C:N ratio
	<63	63 < x < 500		
PU	24.3 (2.7)	75.9 (2.7)	0.51 (0.07)	10.1 (0.27)
Control	20.1 (1.2)	80.1 (1.2)	0.50 (0.05)	10.2 (0.23)
p Value	0.014	0.046	0.559	0.224

5. Conclusion

Until now, it was unclear how annual seagrasses are able to modify their environment, and if the commonly observed higher silt fraction in annual intertidal seagrass vegetations is induced by the presence of the seagrass or vice versa. Our research demonstrates that annual eelgrass patches are: (1) able to enhance sediment accretion; and (2) increase the silt content of sediment during the growing season. The magnitude of accretion by annual seagrass was found to strongly depend on canopy density. Thus, during the growing season, annual seagrass beds affect sediment dynamics similarly as known from the well-studied perennial seagrasses. However, in contrast to perennial seagrasses, annual populations release the accumulated sediments during winter. The timing of this sediment release may make this sediment especially available for salt marshes, so that seagrass beds may be ‘feeding’ salt marshes with sediment. This, however, needs further investigation.

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