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Changes in biota and sediment erodability following the placement of fine dredged material on upper intertidal shores of estuaries

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ABSTRACT: Fine sediments derived from maintenance dredging in estuarine and coastal zones may provide a potential resource for enhancing or creating intertidal habitats (i.e. 'beneficial use' schemes). This study investigates the temporal changes in biota and sediment erodability following the placement of fine dredge material (ca. 0.6 m depth) on the upper shore at 2 trial 'beneficial use schemes' in estuaries situated in Essex, UK. There was a rapid process of sediment consolidation and dewatering within 7 d, reaching bulk densities and water contents typical of intertidal sediments within 6 wk. This was accompanied by an increase in critical erosion velocity (U_{crit}) from 0.13 to 0.25 m s^{-1} (equivalent to a bed shear stress of 0.04 and 0.12 Pa) and a reduction in sediment erosion by 2 orders of magnitude. There was evidence of marked spatial (inter-site) and temporal variation in sediment stability which correlated with changes in the abundance of key species. The temporal changes in sediment erodability reflected the nature of benthic assemblages established during the recovery period (up to 19 mo). There were statistically significant correlations between microphytobenthos chl *a*, extracellular polymeric substances (EPS) and U_{crit}, and between total abundance of tube building/dwelling polychaetes and oligochaetes (minus *Hediste diversicolor*) and mass of sediment eroded at 0.3 m s^{-1}. The annual salt marsh plant *Salicornia europaea* was also found to reduce sediment erodability by reducing near-bed flows by up to 90%, as well as increasing U_{crit}. These biota represented ecosystem engineers with a functional role as bio-stabilisers. There were also significant correlations between U_{crit} and the abundance of *H. diversicolor* and *Corophium volutator*, and between sediment mass eroded at 0.3 m s^{-1} and *H. diversicolor* and *Hydrobia ulvae*. These biota represented ecosystem engineers with a functional role as bio-destabilisers. Most of the recorded correlations were consistent with previous species-specific flume studies establishing density-dependent effects on sediment erodability, thus indicating cause-effect relationships.

KEY WORDS: Estuaries · Sediment · Dredged material · Erodability · Biota · Current velocity · Temporal variability · Essex

INTRODUCTION

Dredging and disposal of dredged material constitutes one of the most important issues in coastal zone management in the UK and Europe. In the UK, approximately 40 million t of dredged material are disposed annually to estuarine and offshore sites (Bolam et al. 2003). This is carried out under license by the Department of Environment, Food and Rural Affairs (Defra) under the 1985 Food and Environment Protection Act (FEPA II). Typically, ~80% of the material arises from 'maintenance dredging' (i.e. typically fine sediments) and the remainder from 'capital dredging' (i.e. coarser sediments). In recent years there has been a shift in emphasis from 'disposal' at sea to a more managed 'beneficial use' of such material (Bolam et al. 2003).

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However, prior to dredging and use in recharge schemes, the Centre for Environment, Fisheries and Aquaculture Science (CEFAS) tests the sediments for metals, tributyl tin (TBT) and organic contaminants, and only relatively uncontaminated sediments are considered for recharge. The potential advantages of 'beneficial use' schemes include:

(1) Flood and coastal defence. Natural sea defences (i.e. mudflats and salt marshes) form an important but vulnerable part of our sea defence; they provide an essential 'ecosystem function', namely to dissipate tidal and wave energy. In the UK, salt marshes and mudflats are eroding at an unprecedented rate, particularly in the southeast (~40 ha yr^{-1}; Hughes & Paramor 2004). In areas with inadequate sediment supply this high rate of loss is likely to continue, especially with predicted increases in the rate of sea level rise. The presence of salt marshes and mudflats in front of seawalls (hard defences) helps to prevent erosion at the base of the wall and thus reduces construction costs by £ 3500 m^{-1} (Environment Agency figures from Hughes & Paramor 2004). The use of maintenance dredge material to protect or create salt marshes is viewed as a cost-effective way of protecting seawalls.

(2) Sediment cell maintenance. In tidal estuaries there tends to be a net balance between the amount of material eroded and deposited. Such a balance may be disturbed when an estuary is dredged, and continuous removal may lead to erosion of tidal flats and salt marshes. The 'in estuary' placement of dredged material during 'beneficial use' schemes minimises the perturbations to an estuary's sediment cell maintenance.

(3) Habitat conservation/enhancement. There is increasing concern regarding the continuing loss of intertidal habitats, including mudflats and salt marshes, due to erosion and reclamation. These habitats are productive systems that support important species with valuable 'ecosystem functions' (including sediment stabilisation, sea defences, fisheries, nutrient recycling) and, in addition, provide breeding and food resources for migrant birds. Therefore, the 'beneficial use' of fine dredged material in protecting, enhancing or creating intertidal habitats (mudflats and salt marshes) is being assessed.

At present in the UK, the beneficial placement of material from maintenance dredging is limited to small-scale trials in the intertidal zone (Bolam & Whomersley 2003). This is due to (1) concerns over the subsequent movement of material as a result of tidal currents and wave action and potential interference with other uses/users of the area and (2) the lack of knowledge of the rate of recovery and how this is affected by factors such as timing, rate and depth of recharge and the properties of the dredged material. The present study uses small-scale recharge sites that are well protected by bunds and old breached sea defences. Therefore, the impact of wave action is likely to be minimal compared to more exposed intertidal mudflats with a considerably greater fetch length.

Several studies have examined the recolonization of invertebrate macrofauna following the placement of dredged material on intertidal mudflats (Ray 2000, Bolam & Whomersley 2003, 2005, Bolam et al. 2004), or the creation of mudflats with earthmoving equipment (Evans et al. 1998). These studies have shown that recovery, in terms of taxa richness, abundance and species composition, may take from 6 to 24 mo depending on the size of the area, the timing of the scheme and the properties of the sediment.

Following the placement of fine dredged sediment on an intertidal shore, there are likely to be marked temporal changes in the stability of the newly deposited sediment. This will largely be a function of the changes in biological and physical properties of the sediment, which are a consequence of the consolidation/de-watering processes, tidal elevation/emersion time, and the settlement, recruitment and immigration of specific intertidal biota. Recent research has shown that key species act as ecosystem engineers, providing an important ecosystem function to the intertidal zone (Widdows & Brinsley 2002). These ecosystem engineers influence estuarine sediment dynamics by modifying sediment stability, bed roughness and near-bed hydrodynamics. The overall objective of the present study is to quantify the changes in the erodability of fine dredged sediment placed on the upper shore at small-scale trial sites, and to relate these to temporal changes in physical and biological properties of the sediment over a period of 19 mo. At present, it is unknown whether the fine dredged material placed on intertidal mudflats is potentially vulnerable to erosion and whether it is likely to remain at the site. It is also unknown to what extent and over what timescales key biota may colonise and stabilise or destabilise the sediments.

This multi-disciplinary study involved site specific measurements of changes in the sediment properties (physical, chemical, biological) and hydrodynamics over a timescale of 19 mo following sediment placement. Any statistical relationships between sediment erodability and the physical, chemical and biological properties of the sediment were interpreted in the light of established cause-effect relationships.

MATERIALS AND METHODS

Description of the field campaigns and sites. There were 3 field campaigns (September 2001, April 2002 and October 2002) to investigate temporal changes in the biological and physical properties of sediment at 4 sites in estuaries situated in Essex, southeastern UK. Descriptions of the sites are presented in Table 1. There were 2 experimental 'shore recharge sites' (Titchmarsh Marina and Westwick Marina), where relatively uncontaminated fine dredged material was pumped onto the upper intertidal shore immediately in front of an existing salt marsh. Due to the lack of spatial or temporal continuity at the recharge sites, following the smothering of the original habitat, there is no appropriate 'reference site' for such studies (Bolam & Whomersley 2003). Therefore, any changes in sediment erodability at the sites were analysed as a function of the prevailing environmental conditions (i.e. physical and biological sediment properties) and compared with a representative intertidal mudflat. The intertidal mudflat at Maldon, which had similar physical and biological properties to the recharge sites, served as a representative mudflat and provided a measure of the natural temporal changes that occurred over the study period. The Titchmarsh Marina recharge site differed from the other 2 sites in that the annual saltmarsh plant *Salicornia europaea* grew at the margins of the newly deposited sediment. As a consequence, a 'managed realignment' site (Orplands) with extensive areas of *S. europaea* was studied as a fourth site.

Field measurements. On each sampling occasion, surface sediment cores (n = 3; 5 cm diameter; 1 cm depth) were collected and analysed for physical sediment properties. Grain size was measured using a laser diffraction particle size analyser (Beckman Coulter LS 230). Measurements also included bulk density (mass of wet sediment/volume of wet sediment), water content (% water) and a measure of particulate organic matter (POM; by loss on ignition at 450°C). Biological sediment properties, such as microphytobenthos biomass and extracellular polymeric substances (EPS), were measured in terms of chl *a* and colloidal carbohydrate content. These were measured in the top 2 mm of sediment, collected by means of cut plastic syringes of 12.5 mm diameter (3 replicate cores). Chl *a* was analysed by HPLC following acetone extraction (Lucas et al. 2000), and EPS by the phenol-sulphuric acid method (Underwood et al. 1995). Macrofauna was collected using 6 cores (11.3 cm internal diameter or 0.01 m^2), sampled to a depth of 15 cm and retained on a 0.5 mm mesh sieve. Fauna was fixed with formaldehyde (10% final solution), sorted, identified and counted. Macroalgae and salt marsh plants were quantified in terms of individuals, biomass or stems m^{-2}. Sediment heights and slopes were calculated from shore profiles obtained using a laser levelling device with reference to mean high water spring tides.

Table 1. Intertidal sampling sites in Essex, southeastern UK

Site	Location	Type	Description
Titchmarsh Marina, Hamford Water	51° 51.763′ N 01° 15.133′ E	Recharge site	Solid clay bunded area on upper shore (~0.02 km^2). Recharged with fine dredged sediment (to a depth of 0.65 to 0.8 m) from marina between March and mid-April 2001. Gradual ramp profile up to mean high water spring (MHWS) tidal level (slope 1°) with *Salicornia europaea* and *Spartina anglica* growing on the margins near MHWS and in front of sea dyke. Macroalga, *Enteromorpha* spp., was growing in the lowest parts that retained surface water at low tide. *Enteromorpha* spp. formed an algal mat (>50% of surface coverage) on 10 to 15% of the bunded area. Enclosed area with a maximum fetch of 150 m
Westwick Marina, River Crouch	51° 38.692′ N 00° 39.61′ E	Recharge site	Relatively small experimental recharge site (~250 m^2). The end of a muddy channel in a saltmarsh system was bunded with woven willow wicker and recharged with fine dredge material from nearby Westwick marina at the end of July 2001 (to a depth of 0.5 to 0.6 m). After recharging the site, the sediment level (slope 1.7°) was ~35 cm below the existing saltmarsh (supporting populations of *Spartina anglica*, *Salicornia europaea*, *Inula crithmoides* and *Suaeda maritime* on consolidated mud). Protected area with maximum fetch of 15 m
Maldon, Blackwater estuary	51° 43.433′ N 00° 42.067′ E	Intertidal mudflat	Intertidal mudflat (400 m wide, slope 2.3°) with similar physical and biological properties to the experimental recharge sites (see Tables 2 & 3). Mudflat exposed to a maximum fetch of 1 km from the easterly winds
Orplands, Blackwater estuary	51° 43.3′ N 00° 52.133′ E	Managed realignment site	Previously reclaimed farmland flooded by opening up man-made defences. Colonised by *Salicornia europaea*

Tidal cycles of near-bed current velocities (at depth z = 0.05 m above the bed), bed shear stresses (turbulent kinetic energy [TKE] method), water depth and suspended sediment concentrations were measured using a Sontek/Yellow Springs Instruments (YSI) micro acoustic Doppler velocimeter (ADV Field), with an optical back-scatter sensor (OBS-3) coupled to a Hydra system. Plymouth Marine Laboratory (PML)'s mini-rig, incorporating an x-y electromagnetic (EM) current meter (5.5 cm spherical head; Valeport 800), an OBS-1, and a depth pressure sensor (Druck PDCR 940, 0 to 7 bar absolute) coupled to a microprocessor controller-logger and power supply, was also deployed. Both the Sontek ADV/Hydra and PML's mini-rig were programmed to record for 3 min every 12 min. The OBS sensors were calibrated against water samples with resuspended sediment obtained from flume experiments run with sediment collected from the appropriate site. These were filtered onto pre-weighed glass fibre filters (GFC), washed with distilled water, dried at 90°C and re-weighed together with blank filters.

Gross sediment deposition rates (g m^{-2} tidal $cycle^{-1}$) were measured at the salt marsh sites at Westwick and Maldon. Ten Petri dishes (90 mm diameter, 14 mm depth) were placed on the sediment surface within the salt marsh approximately 1 m from the front edge, anchored by means of metal wires, and initially filled with filtered seawater. After 1 high spring tide, the sediment within the dish was collected and washed into a container. These samples were then filtered onto pre-weighed GFC filters, washed with distilled water, dried at 90°C and re-weighed together with blank filters. The gross deposition rates were compared with the net deposition rates calculated from changes in suspended sediment concentration (SSC) and water depth, measured every 12 min over the tidal cycle using the Sontek Hydra.

Site specific flume studies of sediment erodability. PML's portable annular flumes (Widdows et al. 1998a) were used to quantify benthic-pelagic exchanges in relation to important physical and biological factors (i.e. current velocity, sediment properties and benthic biota). Two identical flumes were run simultaneously, thus providing replicates for each site or experimental condition. The flume had a 0.64 m outer and 0.44 m inner diameter, creating a 0.1 m annular channel. It had a total bed area of 0.17 m^2 and a maximum volume of 50 l. Current speeds were created by a rotating annular drive plate (without paddles) and increased stepwise from 0.05 to 0.45 m s^{-1} in 0.05 m s^{-1} steps, each with a duration of 20 min. Vertical profiles in current velocities were measured with either an electromagnetic current meter (Valeport Model 800-175, September 2001) or a Sontek laboratory micro ADV (April and October 2002). Changes in SSC were monitored with an OBS-3 (D & A Instruments) calibrated against water samples taken for gravimetric analysis. The relationship between depth-averaged current velocity (U; m s^{-1}) and bed shear stress (τ_0; Pa) for smooth cohesive mud, measured using a Laboratory Sontek/YSI micro ADV (Pope et al. 2006), was described by the following equation:

$$\tau_0{:} = 0.4702U^3 + 1.152U^2 + 0.1553U$$
$$r^2 = 0.99$$

Sediment erodability was quantified in terms of critical erosion velocity (U_{crit}), mass of sediment eroded (g m^{-2}) and erosion rate (g m^{-2} s^{-1}). The methods of calculating these measures of sediment erodability are described in detail by Widdows et al. (1998a,b). U_{crit} reflects the Type 1b erosion of bed sediment rather than the initial resuspension of flocs or Type 1a erosion (Amos et al. 1992).

Short-term changes in the stability of the fine dredged sediment were measured at intervals (4 h, 24 h and 7 d) after placement in the annular flume, and after longer-term placement on the upper shore (between 2 and 19 mo). Sediment stability at field sites was determined by sampling undisturbed sediment, with its natural benthic community, by means of stainless steel quadrant box cores (4 cores forming an annulus of 64 cm outer and 44 cm inner diameter) designed to fit precisely into the flumes. These cores were first pushed into the sediment to a depth of 7 cm and then dug out, allowing base plates to be inserted at the bottom of the cores. This enabled the cores to be lifted, and bands to be placed around them to retain the base and sediment during transportation to the CEFAS Laboratory, Burnham-on-Crouch, Essex. The quadrant box cores were carefully inserted into the annular flumes and the stainless steel boxes were removed, leaving the sediment and base plates in the flume. The sediment blocks were then carefully pushed together to fill any small gaps, and the remaining space was filled with a slice of sediment from an additional core. After inserting the sediment cores in the flume, a sheet of 'bubble wrap' the size and shape of the annulus was carefully placed on the sediment, and seawater was gently pumped onto the sheet, which then gradually floated off without disturbing the sediment surface. Erosion experiments were carried out within 1 h of filling the flumes with seawater at the ambient temperature and salinity. Recent studies (Widdows et al. 2000a, Widdows et al. 2006) have confirmed that there are no significant differences between *in situ* flume measurements and laboratory-based flume measurements using undisturbed cored sediments.

The relationships between physical, chemical and biological sediment properties were analysed using a Pearson product moment correlation coefficient. Significant differences between site and time for the species abundance data were tested using ANOVA, after transformation (square root) and testing for homogeneity of variances (statistical analyses performed using Minitab v13). The data were also analysed by principal component analysis (PCA) using PRIMER v6 (PRIMER-E; Clarke & Gorley 1999). PCA was only used for sites and times where all parameters were measured (i.e. complete data sets).

RESULTS

Temporal and spatial changes in physical properties of sediment

The temporal and spatial changes in physical and biological properties of sediments at the different sites are presented in Tables 2 & 3. At the 2 recharge sites, bulk density increased and water content decreased with time after deposition, primarily due to the consolidation of the sediment and the de-watering process. Consequently, there was a significant negative correlation ($r = -0.976$; $p < 0.001$) between bulk density and % water (Table 4). At the Westwick site there was a significant logarithmic decline in water content from 86 to 61%, which reached a near steady state within 6 wk of placement on the shore:

$$\%\ \text{water in sediment} = -4.4079 \ln(d) + 79.336$$
$$(r^2 = 0.98)$$

Similarly, the water content of the Titchmarsh sediment declined from 90 to 57% during the consolidation phase. Accompanying this consolidation process was an increase in sediment stability, particularly at the Westwick recharge site, with U_{crit} increasing from 0.13 to 0.25 m s^{-1} during a 6 wk period. This increase in U_{crit} resulted in a marked decrease in the sediment mass eroded. For example, in response to an applied current velocity of 0.3 m s^{-1} (equivalent to an applied bed shear stress of 0.163 Pa), sediment erosion decreased by 2 orders of magnitude (from 3250 g m^{-2} to 31 g m^{-2}).

Temporal changes in mass of sediment eroded (g m^{-2}) in the annular flume in response to increasing current speed (m s^{-1}) at the Titchmarsh and Westwick sites are illustrated in Fig. 1. Following the initial and rapid phase of consolidation that was accompanied by increased sediment stability (i.e. within 7 d), there was evidence of a significant temporal change in sediment erodability over the 19 mo period. The 2 recharged sites behaved differently, but these changes were not related to physical properties of the sediment. The Titchmarsh site showed increased sediment erodability in the spring (April 2002) compared with the autumn (September 2001 and October 2002). In con-

Table 2. Temporal changes in physical properties of sediments. Data on bulk density, water content, loss on ignition, silt (<63 µm) are mean ± SE (n = 3); data on critical erosion velocity (U_{crit}) and mass eroded are mean ± semi-range (n = 2); t_0: time zero

	Bulk density (g wet wt cm^{-3})	Water content (%)	Loss on ignition (%)	Silt (%)	U_{crit} (m s^{-1})	Mass eroded at 0.3 m s^{-1} (g m^{-2})
Titchmarsh						
t_0 + 4 h	1.03 ± 0.00	90.6 ± 1.0	6.7 ± 0.4	95.3 ± 1.3	0.13	3000
t_0 + 24 h	1.08 ± 0.01	85.8 ± 0.9	5.3 ± 0.04		0.15	780
t_0 + 7 d	1.11 ± 0.00	85.2 ± 0.7	4.5 ± 0.02			
Sep 2001	1.33 ± 0.01	57.4 ± 0.3	4.2 ± 0.1	94.0 ± 0.7	0.17	106
Apr 2002	1.41 ± 0.01	51.4 ± 0.2	4.4 ± 0.1	91.0 ± 0.5	0.145 ± 0.005	389 ± 31
Oct 2002	1.32 ± 0.01	59.3 ± 1.0	5.8 ± 0.2	90.9 ± 1.2	0.19 ± 0.0	60 ± 9
Titchmarsh + *Salicornia*						
Apr 2002	1.27 ± 0.02	44.7 ± 0.4	4.2 ± 0.1	72.3 ± 2.9	0.16 ± 0.01	372 ± 10
Oct 2002	1.38 ± 0.00	51.8 ± 0.3	4.7 ± 0.1	87.2 ± 0.3	0.29 ± 0.01	0
Westwick						
t_0 + 4 h	1.07 ± 0.02	86.3 ± 0.5	5.2 ± 0.0	86.7 ± 1.3	0.13 ± 0.0	3250 ± 250
t_0 + 24 h	1.15 ± 0.00	79.7 ± 0.2	4.7 ± 0.02		0.14 ± 0.0	1516 ± 26
t_0 + 7 d	1.19 ± 0.01	72.6 ± 0.6	4.2 ± 0.02		0.16 ± 0.0	216 ± 24
Sep 2001	1.29 ± 0.00	61.5 ± 0.3	5.7 ± 0.1	83.4 ± 2.9	0.245 ± 0.005	31 ± 10
Apr 2002	1.27 ± 0.02	60.9 ± 1.8	8.0 ± 0.1	90.1 ± 0.3	0.23 ± 0.0	26 ± 0
Oct 2002	1.25 ± 0.00	66.7 ± 0.5	8.4 ± 0.0	84.3 ± 1.1	0.11 ± 0.0	163 ± 20
Maldon						
Apr 2002	1.19 ± 0.01	73.7 ± 0.9	9.1 ± 0.1	94.9 ± 0.5	0.175 ± 0.025	75 ± 25
Oct 2002	1.24 ± 0.00	68.6 ± 0.4	5.9 ± 0.1	92.3 ± 0.7	0.13 ± 0.0	60 ± 7
Orplands						
Sep 2001	1.22 ± 0.02	66.2 ± 1.3	9.5 ± 0.8	84.7 ± 1.4	0.315 ± 0.005	5 ± 0

Table 3. Temporal changes in biological properties of sediments (mean ± SE)

	Chl *a* (mg m^{-2})	Carbohydrates (mg m^{-2})	Macrofauna (ind. m^{-2}): *Hydrobia ulvae*	*Corophium volutator*	Polychaetes + oligochaetes: Total without *Hediste diversicolor*	*Hediste diversicolor*	*Salicornia europaea*: Density (stems m^{-2})	Stem height (cm)
Titchmarsh								
Sep 2001			4589 ± 352	0	2844 ± 954	689 ± 179		
Apr 2002	35.9 ± 0.3	354 ± 28	8400 ± 1537	0	355 ± 89	3733 ± 176		
Oct 2002	35.2 ± 1.3	193 ± 7	9711 ± 1744	0	1977 ± 1265	467 ± 267		
Titchmarsh + *S. europaea*								
Apr 2002	46.3 ± 4.8	794 ± 199	335 ± 87				2827 ± 17	1 to 2
Oct 2002	105.4 ± 9.2	764 ± 110	1487 ± 171				1365	15
Westwick								
Sep 2001	118.7 ± 8.9	1140 ± 215	20 ± 10	0	11455 ± 4300	65 ± 51		
Apr 2002	51.6 ± 3.2	731 ± 48	0	2400 ± 1161	12700 ± 700	554 ± 319		
Oct 2002	18.4 ± 0.7	97 ± 1	547 ± 67	12189 ± 4254	9300 ± 876	4133 ± 1811		
Maldon								
Apr 2002	93.5 ± 8.7	1357 ± 215	223 ± 12	70 ± 30				
Oct 2002	25.5 ± 0.6	108 ± 9	47 ± 10	14480 ± 362	13305 ± 963	2005 ± 95		
Orplands + *S. europaea*								
Sep 2001			1050 ± 65				4252	20

trast, the Westwick site had a low sediment erodability in the spring (April 2002) and higher erodability in the autumn (October 2002). Measurements of sediment stability (i.e. U_{crit} and sediment mass eroded) taken more than 1 mo after sediment placement at Westwick and Titchmarsh were found to be within the range recorded at the Maldon site (Table 2).

There were differences among the sites for other physical properties, with the Titchmarsh site having the highest bulk density and lowest % water content (Table 2). However, sediment grain size analysis showed that all sites had a relatively high silt content (i.e. >72% of particles <63 µm) with 12.2 to 16.4% of particles in the clay fraction (<2 µm). The organic content, based on 'loss on ignition', was lowest at Titchmarsh.

Temporal and spatial changes in biological properties of sediment

A Pearson correlation matrix that incorporates sediment erodability and physical and biological sediment properties is presented in Table 4. There are several statistically significant correlations that reflect either sediment stabilisation (high U_{crit}, low mass eroded) or sediment destabilisation (low U_{crit}, high mass eroded). There were significant positive correlations between U_{crit} and both chl *a* and EPS ($p < 0.005$). This is consistent with previous field and laboratory studies showing that microphytobenthic biofilms increase sediment cohesiveness and, as a result, sediment stability (Sutherland et al. 1998a,b, Widdows et al. 2000a, 2004). The Essex sites, with the exception of Titchmarsh, typically showed a higher sediment chl *a* and carbohydrate (EPS) content in the spring (April 2002) compared with the autumn (October 2002), reflecting the spring peak in microphytobenthos biomass (Table 3). However, at the Westwick site, highest values of chl *a* (119 mg m^{-2}) and EPS content (1140 mg m^{-2}) were observed in September 2001, 6 wk after the deposition of dredged sediment. The presence of a well developed microphytobenthos biofilm at this time was probably associated with the increased availability of nutrients from the recently disturbed and deposited sediments, combined with the low density of grazers which had not yet migrated or recruited into the sediment.

At the margins of the Titchmarsh site, where *Salicornia europaea* was able to grow, the chl *a* content (105 mg m^{-2}) was higher in the surface sediments in October 2002 than in April 2002, and higher in comparison with other sites at this time of year. This was due to the presence of short filamentous macroalgae (e.g. *Enteromorpha* spp.) on the sediment surface. In contrast to other sites (where sediments were dominated by microphytobenthos) there was no associated increase in carbohydrate content.

The 2 recharged sites had very different benthic macrofauna assemblages. Only 4 macrofauna taxa, out of a total of 16 to 19, showed significant correlations with physical and biological sediment properties (Table 4). The Titchmarsh site had an order of magnitude higher density of the small snail *Hydrobia ulvae*

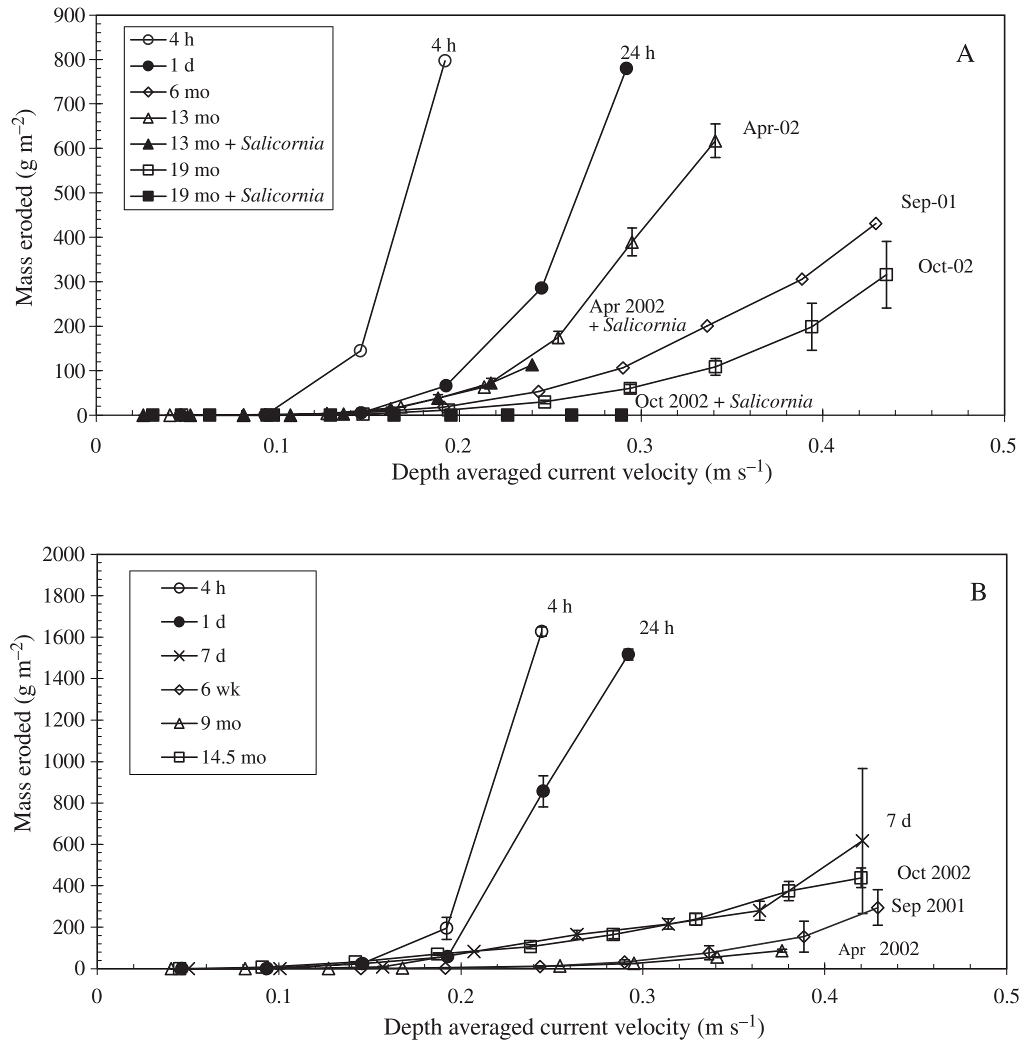

Fig. 1. (A) Titchmarsh recharge site. Sediment mass eroded (g m^{-2}) versus depth-averaged current velocity for dredged sediments deposited at the site. Flume erosion measurements made at intervals from 4 h to 19 mo. In April 2002 and October 2002, erosion measurements also included sediments from margins of the recharge site with *Salicornia europaea*. (B) Westwick recharge site. Sediment mass eroded (g m^{-2}) versus depth averaged current velocity for dredged sediments deposited at the site. Flume erosion measurements made at intervals from 4 h to 14.5 mo. Mean ± semi-range (n = 2)

(ANOVA, $p < 0.001$) and of the polychaete *Hediste diversicolor* (at least in September 2001 and April 2002; $p < 0.01$), whereas Westwick had a markedly higher abundance of 'small annelids' (polychaetes and oligochaetes, not including *H. diversicolor*; $p < 0.001$), and the crustacean *Corophium volutator* ($p < 0.01$). The benthic assemblages at the Westwick recharge site were similar to the Maldon site in October 2002 (ANOVA, no significant difference). At Westwick there was a steady and significant increase in *H. diversicolor* ($p < 0.03$) and *C. volutator* ($p < 0.007$) following the placement of sediment, while at Titchmarsh there was a peak in *H. diversicolor* in April 2002 ($p < 0.001$) and a total absence of *C. volutator* on all sampling occasions.

There was a significant negative correlation between U_{crit} and the density of *Corophium volutator* (Table 4; $p < 0.009$), which reflected lower U_{crit} values at Westwick and Maldon when *C. volutator* densities were high in October 2002 (Figs. 1B & 2). *Corophium volutator* and *Hediste diversicolor* graze the microphytobenthos, and this trophic interaction is consistent with the negative relationship between these 2 species and U_{crit}, chl *a* and EPS (Table 4). There was also a significant positive correlation between sediment erodability (mass eroded) and *H. diversicolor* and *Hydrobia ulvae* densities ($p < 0.002$ and $p < 0.009$), suggesting that these species bioturbate and disturb more than the very surface sediment.

Table 4. Correlation matrix showing significant correlations (r value, with p value below; p values < 0.05) in **bold** between physical and biological properties of sediment at the 3 main sites (Titchmarsh, Westwick and Maldon) in September 2001, April 2002 and October 2002. MassErod: mass eroded, Chloro: chl *a*, EPS: extracellular polymeric substances, % <63 µm: % silt <63 µm, BulkDen: bulk density, *Hydrobia*: *Hydrobia ulvae*, *Hediste*: *Hediste diversicolor*, *Corophium*: *Corophium volutator*, TotalAnn: total small annelids excluding *H. diversicolor*

	MassErod	U_{crit}	Chloro	EPS	% <63 µm	% Water	BulkDen	*Hydrobia*	*Hediste*	*Corophium*
U_{crit}	−0.528 **0.010**									
Chloro	−0.364 0.137	0.670 **0.004**								
EPS	−0.348 0.157	0.704 **0.002**	0.913 **0.000**							
% <63 µm	0.354 0.137	−0.259 0.284	−0.123 0.650	−0.223 0.407						
% Water	−0.105 0.650	0.104 0.671	0.065 0.796	0.097 0.702	0.658 **0.002**					
BulkDen	0.187 0.418	−0.266 0.271	−0.120 0.636	−0.152 0.548	−0.620 **0.005**	−0.976 **0.000**				
Hydrobia	0.555 **0.009**	−0.145 0.554	−0.330 0.181	−0.423 0.080	0.239 0.324	−0.317 0.162	0.349 0.121			
Hediste	0.762 **0.002**	−0.852 **0.000**	−0.635 **0.026**	−0.630 **0.028**	0.315 0.294	0.015 0.962	0.167 0.587	0.062 0.834		
Corophium	−0.081 0.775	−0.645 **0.009**	−0.573 **0.032**	−0.594 **0.025**	0.286 0.301	0.450 0.092	−0.401 0.138	−0.450 0.080	0.508 0.063	
TotalAnn	−0.645 **0.017**	0.201 0.510	0.285 0.370	0.346 0.270	−0.171 0.576	0.759 **0.003**	−0.862 **0.000**	−0.935 **0.000**	−0.169 0.643	0.573 **0.032**

The density of 'total small annelids' (i.e. not including *Hediste diversicolor*) when grouped together showed a negative correlation (Table 4) with sediment mass eroded ($p < 0.02$). This suggests that these tube building worms contribute to sediment stability. The dominant species in this assemblage were: *Streblospio shrubsoli*, *Tharyx* spp., *Manayunkia aestuarina*, *Paranais litoralis*, *Tubificoides benedii* and *Tubificoides pseudogaster*. There was also a significant positive relationship between 'total small annelid' abundance and % water content ($p = 0.003$), and an inverse relationship with bulk density ($p < 0.001$). Such relationships are consistent with tubes and burrows increasing the water content.

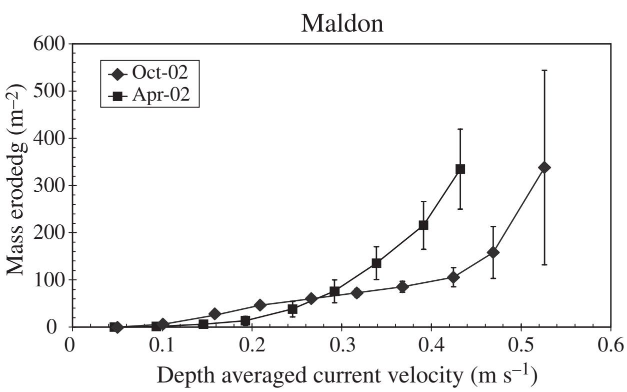

Fig. 2. Sediment mass eroded (g m^{-2}) versus depth averaged current velocity (m s^{-1}) for April 2002 and October 2002. Atypical erosion profile in October 2002 reflects resuspension of recently deposited sediment following a storm event. Mean ± semi-range (n = 2)

PCA supports the basic findings derived from the Pearson correlation analysis (Table 4), and serves to illustrate the temporal and spatial changes in the biological and physical properties of the sediment at the 3 main sites (Titchmarsh, Westwick and Maldon) (Fig. 3). Both recharge sites were very different from each other and the Maldon site in September 2001 (Time 1) and April 2002 (Time 2), but gradually converged towards the conditions at Maldon, the representative intertidal mudflat (without addition of sediment). In October 2002 (Time 3; Fig. 3) the conditions at Westwick were close to those at Maldon, whereas at Titchmarsh, although conditions were converging towards those at Maldon and Westwick, the site was still very different. In October 2002, Titchmarsh was characterised by high values of bulk density and *Hydrobia ulvae*, in contrast with Westwick and Maldon. In September 2001 and April 2002, the Westwick site had relatively high U_{crit}, chl *a* and EPS, which are known to

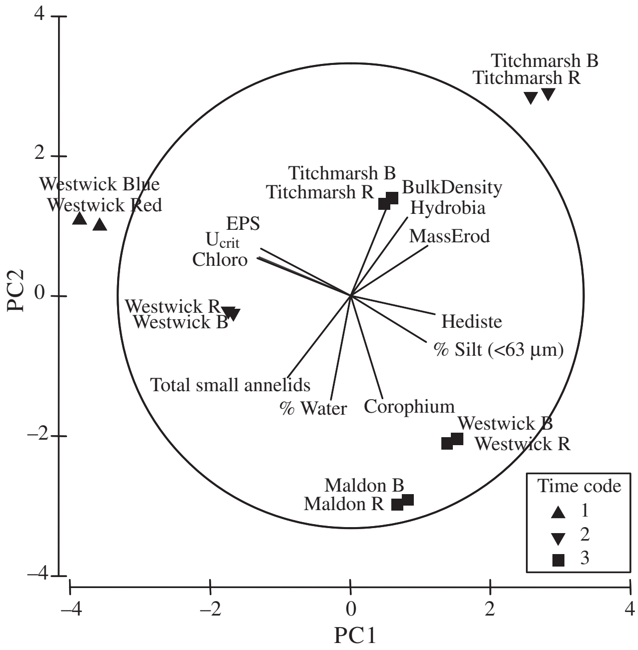

Fig. 3. Principal component analysis (PCA) of physical and biological sediment properties at 2 recharge sites (Titchmarsh and Westwick) and at undisturbed intertidal mudflat at Maldon. PC1 explains 45% of variation, which together with PC2 explains 85% of the variation. Time 1: September 2001, Time 2: April 2002, Time 3: October 2002. Blue (B) and Red (R) denote the replicate flume measurements (n = 2) over time at the different sites. Refer to Table 4 for full definitions of variables

be highly correlated. Then, in October 2002, there was further convergence of the Westwick site to the Maldon site, reflecting the high densities of *Corophium volutator* at these 2 sites.

Both the composition of the benthic community and the general topography of the sites had a major influence on the type of vegetation able to grow on the newly deposited surface sediments. The Titchmarsh site was a 0.02 km^2 protected/bunded area recharged with fine sediments. It had a gradual ramp profile up to the MHWS level and this enabled *Salicornia europaea* and *Spartina anglica* to grow around the margins, near the mean high water spring (MHWS) tidal level and in front of the sea dyke. Sediment erodability at sites with *S. europaea* was significantly lower in the autumn (September 2001 and October 2002) due to the growth of the annual salt marsh plant (height of 15 cm at margins of the Titchmarsh and 20 cm at Orplands). The canopy of *S. europaea* reduced the near-bed flows by up to 90% (depending on density and stem height; Fig. 4) and this minimised sediment resuspension, particularly in October when stem height was maximal (Fig. 1A). However, in the spring (April 2002) there was little difference between the Titchmarsh mud with or without *S. europaea* (Table 1, Fig. 1A) due to the early stage of growth (stem height 1 to 2 cm). At this stage of growth, any reduction in flow was accompanied by increased turbulence and bed shear stress (J. Widdows & N. D. Pope unpubl. data). An additional factor contributing towards the markedly reduced erodability at Titchmarsh in October was the presence of the algae *Enteromorpha* spp. that grew on the sediment as short filaments amongst the stems of the salt marsh plants (*S. europaea* and *S. anglica*).

Current speeds and SSC at the sites

Current speeds, SSC and water depth were measured over spring tidal cycles at Maldon (mudflat and salt marsh), Westwick (mudflat, salt marsh and main drainage channels) and Orplands (*Salicornia europaea* salt marsh). No current or SSC data were obtained from the bunded Titchmarsh site, but the tidal currents were likely to be low (i.e. <0.07 m s^{-1}) and similar to the Westwick recharge site. A summary of the maximum current speeds, maximum and minimum SSC values, and overall trend in SSC during inundation by spring tides is presented in Table 5. The salt marshes and recharge sites at Westwick, Orplands and Maldon only flood for 2 to 2.5 h during high water spring tides, when the water depth ranges from 0.2 to 1.2 m. At this stage of the tidal cycle, either side of slack water, the currents are low (<0.07 m s^{-1}). In contrast to the recharge sites, the Maldon mudflat is inundated for ~4 h during neap and spring tides, at depths of 1.5 and 2.3 m respectively, and is subjected to stronger currents. It also has 1 to 2 orders of magnitude greater fetch length and is therefore more exposed to wave action (Table 1).

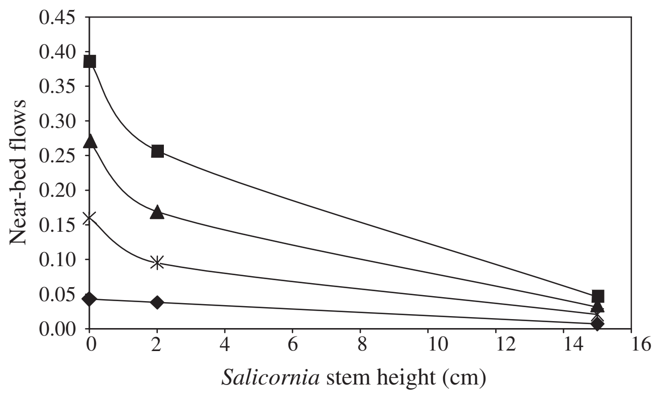

Fig. 4. Impact of *Salicornia europaea* stems on near-bed flows (m s^{-1}) 1 cm above the bed and at 4 flume speeds (◆: 8, ⁎: 25, ▲: 40, ■: 58 rpm). 0 stem height: smooth Titchmarsh sediment without *S. europaea*, 2 cm stem height: April 2002, 15 cm stem height: October 2002

Table 5. Summary of water currents and suspended sediment concentrations (SSC) during single tidal cycles at spring tides (1 neap tide at Maldon). Water depth, SSC and current velocity measured by Sontek ADV + Hydra and mini-rig every 12 min throughout tidal cycle. F: flood tide, E: ebb tide, Gross: measured gross deposition rates (mean ± 95% CI) based on sediment collected in Petri dishes, Net: estimated net deposition per tidal cycle, based on sum of changes in SSC during each 12 min period × mean volume of water above bed. 'Net' assumes that reductions in SSC reflect deposition of sediment on the upper mudflat and salt marsh, and increases in SSC reflect sediment resuspension

	Site	Depth (m)	Date (dd/mm/yr)	Submergence period (%)	Max. current speed ($m\ s^{-1}$)	SSC range ($mg\ l^{-1}$)	Deposition ($g\ m^{-2}\ tide^{-1}$) Net	Gross
Westwick	Recharged site	0.2	10/9/01	12	0.05	12.2 to 7.1	0.5	2.4 ± 0.57
	Drainage channel	1.0	12/9/01	28	0.65 (E)	95 to 170[c]		
Orplands	*Salicornia europaea*	0.2	8/9/01	12	0.07 (E)	47 to 15	4.6	
Westwick	Salt marsh	0.9	28/4/02	16	0.03 (F)	27 to 18	5.5	
	Recharged site	1.2	29/4/02[b]	16	0.04 (F)	49 to 13	29.8	32.1 ± 1.24
	Drainage channel	2.2	27/4/02	36	0.42 (E)	15 to 92[c]		
Maldon	Mudflat	2.3	29/4/02[b]	24	0.38 (F)	500 to 50	734	451 ± 32.0
	Mudflat	2.3	8/10/02	32	0.19 (F)	190 to 45	135	
	Salt marsh	1.0	9/10/02	12	0.05	110 to 52	44	47.1 ± 2.25
	Mudflat	1.5[a]	16/10/02	32	0.10 (F)	170 to 35		

[a]Neap tide
[b]Winds and heavy rain during previous 12 h resulted in high SSC and deposition values
[c] Erosion and net efflux on ebb

The recordings from the Sontek and mini-rig showed that upper-shore water flows at Maldon are complex, and are influenced by shore-normal channels as well as a salt marsh cliff face which directs and increases the flow in a long-shore direction. Maximum current velocities across the Maldon mudflat on spring flood tides were 0.38 $m\ s^{-1}$ on 29 April 2002 (Fig. 5), immediately after a period of windy conditions, and 0.19 $m\ s^{-1}$ in October 2002. These flows were significantly above the critical erosion velocities (U_{crit}) of 0.175 $m\ s^{-1}$ (measured on 25 April 2002, before any physical disturbance of sediments by the wind/wave action) and 0.13 $m\ s^{-1}$ (October 2002; Table 2). This was in contrast to the neap tides, where maximum currents of 0.11 $m\ s^{-1}$ were below the measured critical erosion velocities. Consequently, under calm conditions, tidal currents will not induce erosion on the neap tides; however, there will be significant sediment erosion on the spring tides at the Maldon mudflat.

During periods of high slack water over the Westwick recharge site and the salt marshes at Westwick and Maldon, flows were low (<0.07 $m\ s^{-1}$) and there was a significant and steady decline in SSC in the water column, as suspended sediment was deposited on the bed (Fig. 6, Table 5). The net deposition rates at the various sites were calculated from changes in SSC and water depth over a single tidal cycle, and ranged from 4.6 to 734 $g\ m^{-2}$. These values were in close agreement ($r^2 = 0.99$) with the gross deposition rates measured in Petri dishes on the salt marsh (ranging from 2.4 to 451 $g\ m^{-2}$; Table 5). The amount deposited at each site will be dependent on the depth of water, the period of immersion, and the sediment supply from offshore or the lower shore, as reflected by SSC in the overlying water column at the beginning of the flood tide. The SSC at both the Maldon and the Westwick sites was significantly higher on 29 April 2002 (Table 5) due to wind/wave activity during the 12 h prior to measurement.

These elevated SSC resulted in the salt marsh experiencing a deposition rate 5 times higher than that under calm conditions. The amount of sedimentation at a site is also dependent on the location (fetch and degree of exposure, offshore sediment supply). This is demonstrated in the present study by the fact that the Maldon site has a 24-fold higher deposition rate than the more enclosed Westwick site, regardless of weather conditions (Table 5). The consistently higher SSC at the Maldon site reflected its location near the main estuary channel and the more exposed nature of this site (i.e. 2 orders of magnitude greater fetch) compared to the sheltered Westwick site, which was behind an old breached sea wall. The higher deposition on the Westwick recharge site compared to the Westwick salt marsh was due primarily to water depth. When the recharge site was covered the SSC was 49 $mg\ l^{-1}$, and this declined to 29 $mg\ l^{-1}$ by the time the more elevated salt marsh was inundated (Fig. 6).

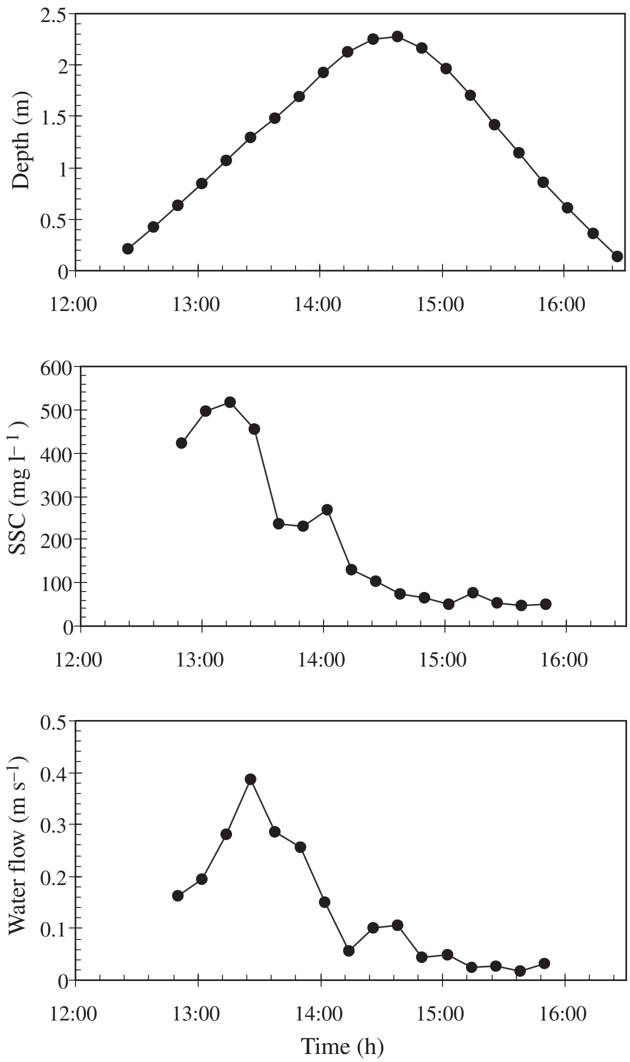

Fig. 5. Time course of depth (m), suspended sediment concentration (SSC; mg l^{-1}) and current velocity (m s^{-1}) measurements on upper shore mudflat over a tidal cycle at Maldon (reference) site

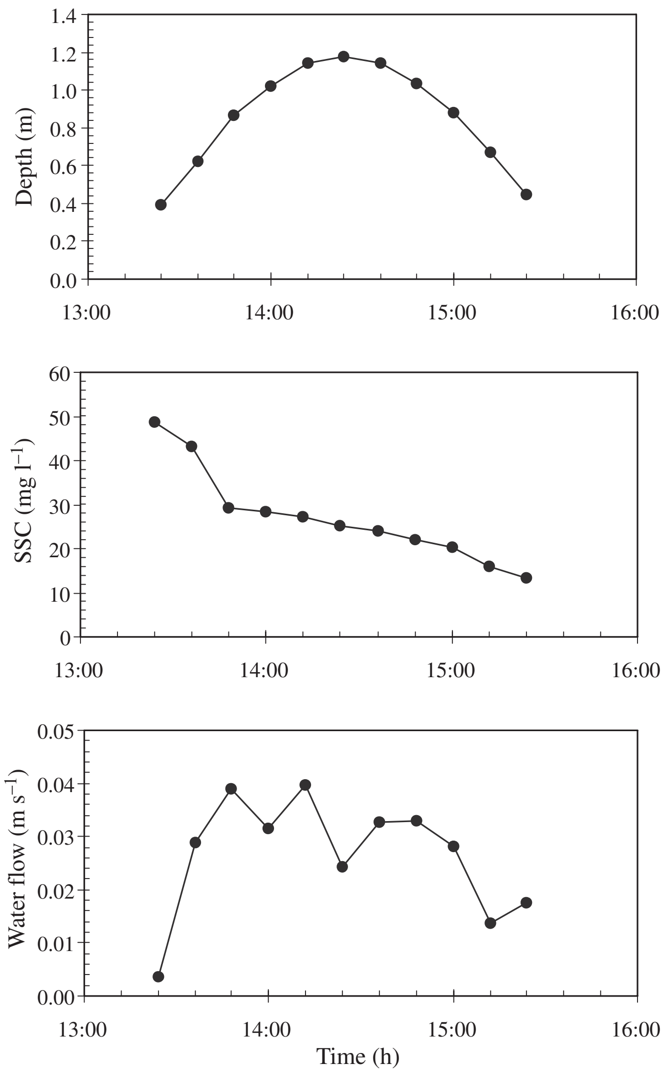

Fig. 6. Time course of depth (m), SSC (mg l^{-1}) and current velocity (m s^{-1}) measurements on salt marsh over a tidal cycle at Westwick site

DISCUSSION

Factors determining temporal and spatial changes in sediment properties

Temporal and spatial changes in the biological and physical properties, and therefore the stability, of newly deposited sediment on the upper shore will be dependent on a number of inter-related factors. These include: (1) the length of time after placement of dredged sediment on the site, which will influence consolidation/de-watering processes and the recruitment/settlement of biota; (2) the topography and tidal elevation of a specific site will affect the prevailing tidal currents, inundation/air exposure time, and the potential sediment stability and its suitability for specific intertidal biota; and (3) the location of the site within an estuary; this will affect the salinity regime and the nature of the sediment (grain size, organic content), both of which will determine benthic community structure and function. Only the first aspect has been examined in this study, with the potential impact of the other factors being minimised through selection of appropriate study sites. However, due to the lack of spatial or temporal continuity at the recharge sites following the smothering of the original habitat, any changes in sediment erodability at the sites were analysed in relation to temporal and spatial changes in the physical and biological sediment properties.

Temporal and spatial changes in sediment erodability

Sediment erodability was quantified in the annular flumes, and the results show marked temporal variation in U_{crit} and sediment mass eroded. Following the pumping and deposition of the fluidised dredged sediment onto the upper shore, there was a rapid process of self-weight consolidation and de-watering within 7 d, reaching a near steady-state within 6 wk. The bed increased in density and strength as a result of the compression and expulsion of pore water between the flocs and grains (Whitehouse et al. 2000). Measurements of the Westwick dredged sediment showed that there was a highly significant linear relationship between % water content and log time over a period of 6 wk, and this was accompanied by a marked increase in sediment stability (i.e. 2-fold increase in U_{crit} and 2 orders of magnitude reduction in sediment mass eroded). After the initial consolidation phase, governed by physical processes, there was evidence of significant temporal variation in sediment erodability at the 2 recharge sites over the period of study (up to 19 mo), and it was possible to relate this to biological processes.

After the initial 7 d, there was no evidence of a significant relationship between the physical sediment properties (grain size, % water content, bulk density) and sediment erodability (Table 4). This is consistent with the results of previous flume studies on fine sediment shores in the outer Humber and Westerschelde estuaries (Widdows et al. 1998b, 2000a,b, 2004), as are the significant correlations that reflect biota-sediment stability interactions and trophic interactions. The correlations are consistent with the recognised functional role of key biota which act as bio-stabilisers (e.g. the microphytobenthos *Salicornia europaea*), bio-destabilisers (e.g. *Hydrobia ulvae*, *Corophium volutator*), or grazers.

The microphytobenthos, or algal biofilm, is recognised as a major factor that influences the stability of intertidal cohesive sediments forming estuarine mudflats (Paterson & Black 1999). This enhanced cohesiveness is the result of algal cells secreting EPS. In the present study, there was a significant positive correlation between U_{crit} and chl *a* and EPS content (Table 4, Fig. 3), reflecting the role of the microphytobenthos in enhancing sediment stability. This is consistent with numerous earlier field studies that show a positive correlation between chl *a* content/EPS and sediment stability (Paterson 1989, Underwood & Paterson 1993, Sutherland et al. 1998a, de -Brouwer et al. 2000, Riethmüller et al. 2000, Andersen 2001, Widdows et al. 2004), and laboratory flume studies confirm that this correlation represents a cause-effect relationship (Sutherland et al. 1998b).

There were also highly significant correlations between sediment erodability and the abundances of specific macrofauna. Many of these observations are consistent with previous field and laboratory studies highlighting key species that act as 'ecosystem engineers' (Widdows & Brinsley 2002). For example, there was a positive correlation between *Hydrobia ulvae* and sediment mass eroded (Table 4, Fig. 3), which suggests that the bioturbatory activity (i.e. crawling, grazing and burrowing) of this small snail significantly destabilises the surface sediment. Similar relationships between the abundance of *H. ulvae* and sediment erodability have been recorded in the field (Andersen et al. 2002) and in the laboratory (Orvain & Sauriau 2002, Orvain et al. 2004).

Both *Hediste diversicolor* and *Corophium volutator* showed significant inverse relationships between their abundance and U_{crit}, chl *a* and EPS content (Table 4, Fig. 3). This reflects their feeding behaviour, which involves disturbing the superficial sediments while grazing on microphytobenthos. Consequently, these grazers physically disturb the sediments as well as reducing the density of algae and the function of algal biofilms as bio-stabilisers. Laboratory studies by de Deckere et al. (2000) demonstrated enhanced sediment resuspension with increasing density of *C. volutator*. *Hediste diversicolor* has also been shown to increase sediment erodability in recent flume studies by S. Fernandes & P. Sobral (unpubl. data).

The sediments at Westwick and Maldon were populated by high densities of small sedentary polychaetes and oligochaetes. Table 4 & Fig. 3 show the significant negative relationship between the mass of sediment eroded and the abundance of 'total small annelids', which indicates that collectively they have a stabilising effect on the sediment. The mechanism of enhanced sediment stabilisation by many of these species is probably due to the high density of mucus-lined burrows. The construction of discrete tubes of sediment grains affects the hydrodynamics at high tube densities by creating 'skimming flow', and thus protects the bed from turbulence (Rhoads & Boyer 1982).

Salt marsh plants, such as *Spartina anglica* and *Salicornia europaea*, did not colonise the soft mud at either the Westwick recharge site or the Maldon mudflat due to a combination of factors (i.e. lower tidal elevation and lower sediment stability than is preferred, and the presence of grazers). The tidal elevation (submergence time at Westwick recharge site: 20%; at Maldon: 32%) meant that the sediment water content was high and that there was insufficient consolidation of the mud for plant anchorage. The tidal elevation also meant that these sites had high densities of grazers such as *Corophium volutator* and *Hediste diversicolor*, which are known to feed on the seeds of salt marsh plants and therefore inhibit marsh development (Gerdol & Hughes 1994, Paramor & Hughes 2004).

The biologically induced temporal changes in sediment erodability at the 3 main study sites reflected changes in the abundance and species composition of macrofauna, due to recolonization of the newly deposited sediment. Bolam & Whomersley (2003, 2005) quantified the temporal changes in benthic assemblages at the Titchmarsh and Westwick recharge sites and concluded that recovery of several univariate parameters relative to the reference situation occurred within 3 mo of sediment placement on the small-scale recharge site, and that the rate of recovery was dependent on the size of the defaunated area. However, the macrofauna species composition failed to converge towards that of the 'reference sites' due to natural spatial heterogeneity between recharge and reference sites. Bolam & Whomersley (2003, 2005) also demonstrated that post-juvenile immigration was the predominant colonisation mechanism, with some planktonic recruitment if placement occurred in the spring/early summer. There was no evidence to suggest that macrofauna surviving direct transfer from the dredged site, or migrating vertically through the layer of deposited sediment, were important colonisation mechanisms.

The development and seasonal growth of the microphytobenthos *Enteromorpha* spp. and *Salicornia europaea* on the newly deposited sediment at the recharged sites reflects the transport of cells, spores and seeds from surrounding mudflats and salt marshes. The development of a vegetative mat or canopy by *Enteromorpha* spp. and *S. europaea* plays an important role in protecting and stabilising the bed by reducing near-bed flows, and thus sediment resuspension, by up to 90% for the highest filament/stem densities (Romano et al. 2003, J. Widdows & N. D. Pope unpubl. data). Therefore, they represent important ecosystem engineers on the upper intertidal shore by acting as bio-stabilisers.

Sediment resuspension in relation to tidal currents at the field sites

Recorded current velocities over the Maldon mudflat on spring tides were significantly higher than measured U_{crit}, thus indicating periods of sediment erosion during the tidal cycle. There was a significant correlation ($r^2 = 0.72$) between SSC and current velocity at Maldon in April, when the peak flows were 0.38 m s^{-1} and in excess of the U_{crit} (0.18 m s^{-1}) for a significant part of the tidal cycle (50 min). However, in October, there was no significant relationship, and the maximum flows were only 0.06 m s^{-1} above U_{crit}. Highest currents and SSC occurred on the flood tide and steadily declined throughout the inundation period (Table 5, Fig. 5). This indicates significant sediment erosion on the flooding spring tide and deposition at high slack water during spring and neap tides on the Maldon mudflat and salt marsh under calm weather conditions (Table 5).

Measured current velocities over the salt marsh sites and recharged mudflats were always <0.07 m s^{-1} and were therefore below the measured $U_{crit,}$ and also below the typical critical threshold for deposition of fine sediment (~0.13 m s^{-1} or τ_0 = 0.04 Pa; Bale et al. 2002, Ciutat et al. in press). Recorded current velocities were even lower than the U_{crit} values for the recently deposited sediments at the recharge sites (i.e. 0.13 to 0.15 m s^{-1} at 4 h and 24 h after sedimentation; Table 2), thus indicating that there was little potential erosion of newly deposited sediment at the recharge sites, at least during calm weather. During inundation of the recharged Westwick site and salt marshes at Westwick and Orplands there was evidence of a small but steady decline in SSC, suggesting some net deposition of suspended sediment on the upper shores at spring tides. Estimates of net sediment deposited per tidal cycle, based on fluctuation in SSC, were in close agreement with the measured gross deposition rates on the salt marsh (Table 5). However, the SSC and estimated sediment deposition rates at Westwick were lower than those at the Maldon site (mudflat and salt marsh) on spring and neap tides (Table 5). This indicated that sediment supply was very limited at Westwick compared to Maldon.

Despite the relatively low tidal current speeds on the upper shore (i.e. below the threshold of erosion), recent studies have shown that waves can be an important factor in shallow and relatively sheltered estuaries (Christie et al. 1999, Houwing 2000, J. Widdows, N. D. Pope, M. D. Brinsley unpubl. data). Waves can induce a ~4-fold increase in the total bed shear stress during storm conditions, resulting in the removal of 2 cm of sediment from the upper shore (J. Widdows, N. D. Pope, M. D. Brinsley unpubl. data). The present study shows a 2 to 2.5-fold increase in SSC at the Westwick and Maldon sites after a period of increased wind/wave activity. Despite storms and wave action being recognised as important factors responsible for increasing bed shear stress and sediment resuspension on mudflats, this does not appear to be a major factor modifying the underlying physical or biological properties at the more exposed Maldon site. Both Westwick and Maldon, representing the extremes of exposure to wind/wave action with a difference of 2 orders of magnitude in fetch (Table 1), had similar benthic assemblages and sediment erodability at the end of the study (Fig. 3). This suggests that although intermittent storm events can resuspend and remove surface sediment, they appear to have little effect on the long-term physical and biological properties of these mudflats. Recent studies have shown that mudflats, salt marshes and recharged sites may also be vulnerable to erosion by intense rainfall during low tide (Paterson et al. 2000, J. Widdows & M. D. Brinsley unpubl. data). Clearly, the relative impact of intermittent events (rain and waves) and persistent events (tidal spring-neap cycles) on the intertidal zone requires further study, particularly in the context of climate change (increased storminess) and its consequences for natural coastal defences.

Results of the present study have some important implications for the management/licensing of sites for beneficial use schemes. The preferred timing of the dredged sediment placement depends on the degree of exposure to wind-wave action at the particular site, the resultant tidal elevation and likely nature of colonisation by biota. Dredged sediment placed on the shore in the autumn and winter is more likely to suffer erosion before adequate stabilisation by physical and biological processes. Clearly, it is important to select a calm weather period for the sediment recharge in order to minimise erosion during the first 7 d of dewatering and consolidation. However, if not eroded, the sediment will become sufficiently consolidated by the spring to allow colonisation by bio-stabilisers such as *Salicornia europaea* and *Enteromorpha* spp. In contrast, sediment placed on a recharge site during the spring will tend to avoid resuspension by the winter storms, but will miss the opportunity for colonisation by bio-stabilisers that prefer more consolidated sediments (e.g. salt marsh plants and macroalgae) until the following spring. Results from the Westwick site, which was recharged in late July, showed that bio-stabilising microphytobenthos was able to colonise recently deposited sediments rapidly, and appeared to benefit from the slower colonisation rate by grazers at this time of year. The grazers did not reach their peak abundance until the following year. More information is needed on physical and biological changes following placement of sediment on mudflats at different times of the year.

CONCLUSIONS

This study has shown that the erodability of newly placed dredged sediments was initially governed by physical aspects of the sediment. The water content was high, and sediment erosion induced by currents was 2 orders of magnitude higher than that of natural sediment. However, this initial phase only lasted for a few days as the sediment quickly consolidated. After approximately 7 d, the erodability was comparable to natural sediments and the critical erosion threshold and mass of sediment eroded were mainly determined by biotic factors (biofilms and bioturbation). There was evidence of marked spatial (inter-site) and temporal variation in sediment stability, which was related to changes in the abundance of key species acting as ecosystem engineers in the intertidal zone. This study has significantly advanced our understanding of the physical and biological processes contributing to sediment stability following placement of dredged material during beneficial use schemes.

Acknowledgements. This study is an output of the DECODE (Determination of the Ecological Consequences of Dredged Material Emplacement) group, established in November 2000. The main aim of DECODE is to harmonise efforts and increase information transfer between researchers concerned with the beneficial use of fine grained dredged material for environmental enhancement, habitat sustainability and flood defence (for further information see www.cefas.co.uk/decode/default.htm). This study was funded in part by the Department for Environment, Food and Rural Affairs (Defra: project codes AE0259 & AE0231) and the Natural Environment Research Council Core Research Programme at Plymouth Marine Laboratory.

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