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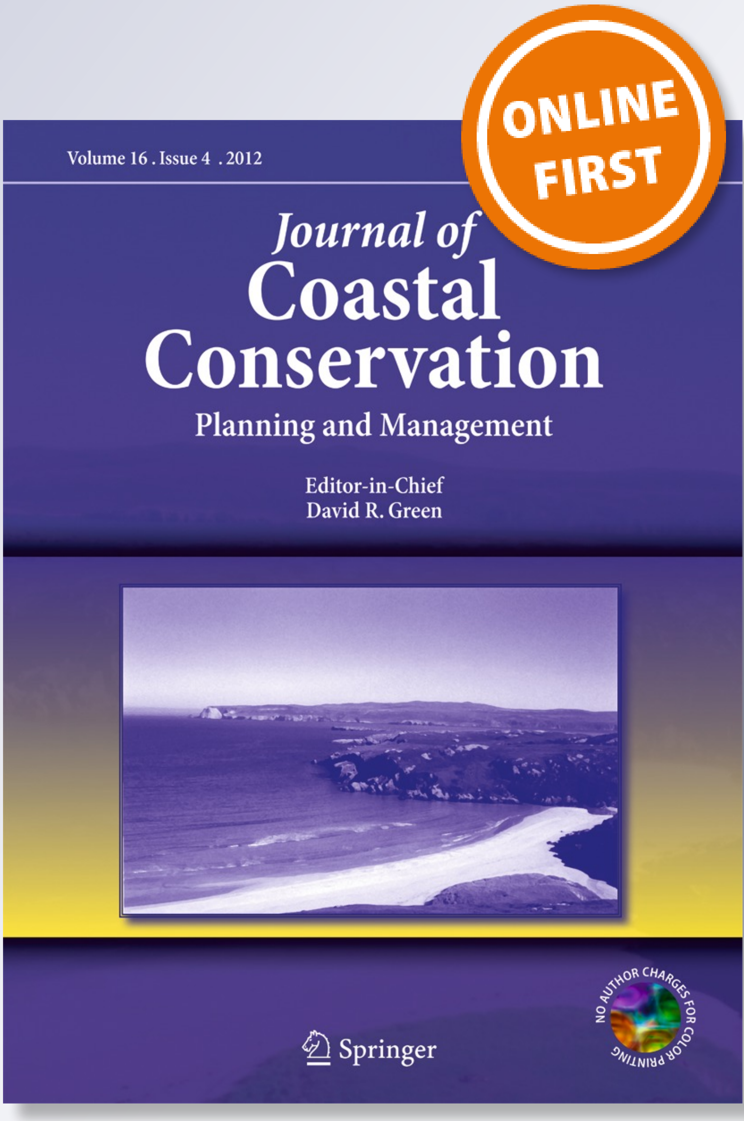

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Abstract It is increasingly recognised that interactions between geomorphological and biotic processes control the functioning of many ecosystem types as described e.g. by the ecological theory of ecosystem engineering. Consequently, the need for specific bio-geomorphological research methods is growing recently. Much research on bio-geomorphological processes is done in coastal marshes. These areas provide clear examples of ecosystem engineering as well as other bio-geomorphological processes: Marsh vegetation slows down tidal currents and hence stimulates the process of sedimentation, while vice versa, the sedimentation controls ecological processes like vegetation succession. This review is meant to give insights in the various available methods to measure sedimentation, with special attention to their suitability to quantify bio-geomorphological interactions. The choice of method used to measure sedimentation is important to obtain the correct parameters to understand the biogeomorphology of tidal salt marshes. This review, therefore, aims to be a tool for decision making regarding the processes to be measured and the methods to be used. We, subdivide the methods into those measuring suspended sediment concentration (A), sediment deposition (B), accretion (C) and surface-elevation change (D). With this review, we would like to further encourage interdisciplinary studies in the fields of ecology and geomorphology.

Keywords Accretion · Elevation change · Estuary · Salt marsh · Sediment deposition · Suspended sediment

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Introduction

The ability of plants or animals to directly or indirectly alter their own physical environment was already recognized by Darwin in the 19th century in his studies on earthworms (see Butler and Sawyer 2012). More recently, this phenomenon has been extensively described within the context of the ecological theory of ecosystem engineering (Jones et al. 1994), highlighting that certain organisms can modify their physical environment, and that these habitat modifications can have a feedback effect on the performance of the organism. For example, sea grasses or salt marsh vegetation directly trap fine sediments by slowing down water currents (e.g. Bouma et al. 2005), while beavers indirectly influence their environment by building dams (e.g. Wright et al. 2002). In both these examples of ecosystem engineering, the habitat modification has a positive feedback effect on the organism. More recently, geomorphologists also highlighted

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the role of mutual feedbacks between organisms and their geomorphological environment in the evolution of landforms and landscapes, and this has increased the number of recent studies in the field of biogeomorphology (Darby 2010; Murray et al. 2008; Reinhardt et al. 2010; Corenblit et al. 2011) or zoogeomorphology (Viles et al. 2008; Statzner 2012). However, there are still many questions unanswered about the connection between the physical environment, and the ecology and evolution of species as summarised by Corenblit et al. (2011). Additionally, the idea to include ecosystem engineers in practical solutions for ecosystem restoration and the provision of ecosystem services has been proposed (Byers et al. 2006). However, objective monitoring and assessment of such solutions are still limited and further studies are needed on the application of ecosystem engineers (Borsje et al. 2011).

Studies connecting geomorphological and ecological processes are often performed by researchers with either an ecological or a geomorphological background. However, this interdisciplinary field requires geomorphologists to understand ecology (Corenblit et al. 2011) and vice versa. Ecologists need to gain knowledge of geomorphology to measure, for example, the effect of ecological processes on sedimentation in tidal areas. However, sharing of knowledge on methods and techniques rarely occurs between the disciplines (Reinhardt et al. 2010). We aim to amend this state of affairs with this review.

Tidal marshes

The main focus of this review lies on coastal salt marshes, which can be found throughout the world along coasts that experience low wave action and sufficient fine sediment supply (Bakker et al. 1993). Sedimentation processes in intertidal areas and their link to ecological processes have lately been the object of many studies (e.g. Borsje et al. 2011; Eklöf et al. 2011; Mermillod-Blondin 2011). One important question in coastal ecology is how coastal marshes will cope with climatic changes, such as enhanced sea-level rise (SLR) (e.g. Temmerman et al. 2004a; Kirwan et al. 2010). A key mechanism that governs the ability of salt-marsh ecosystems to maintain elevation with rising sea level (e.g. Kirwan et al. 2010) and their long-term evolution (e.g. Olff et al. 1997) is sedimentation. Sedimentation in marshes is enhanced by the presence of vegetation (Bakker et al. 1993), which may slow down currents (Bouma et al. 2005; Möller 2006; Temmerman et al. 2012) (Fig. 1), diminish the impact of storm surges (Costanza et al. 2008; Wamsley et al. 2010), and counteract coastal erosion (Gedan et al. 2011). Vegetation succession is mainly driven by the elevation of the marsh and nutrient input through sediment (Olff et al. 1997; Reed 1989), which increases with increased sedimentation, leading to a positive feedback loop. This vegetation-sedimentation feedback enabled salt-marshes in the past to cope with SLR.

Vegetation-sedimentation feedbacks, however, are only one of many potentially important interactions. The main external controls of sedimentation are sea level (hydroperiod) and sediment supply, which is strongly related to the suspended sediment concentration (SSC) (Fig. 1), but the internal interactions between physical and biological features of coastal zones are also of great importance. Surface roots and algae may positively influence sedimentation (McKee et al. 2007). Additionally, the accumulation of biomass can play an influential role in accretion processes (Culberson et al. 2004). On the other hand, bioturbation is often found to have a negative effect on salt marshes by causing erosion (e.g. Davidson and de Rivera 2010). However, the possible positive effects such as sediment mixing (Hippensteel 2005) and soil aeration (Daleo et al. 2007) should not be overlooked. There is even evidence that burrowing crabs caused both erosion and accumulation of sediments in their burrows within the same marsh, but in different zones (Escapa et al. 2008). Finally, human impacts such as ditching or management practices (e.g. livestock grazing) can alter processes related to hydrodynamics, vegetation composition, sedimentation and erosion. Wide regions of the European Wadden Sea coast, for example, have been traditionally grazed by livestock since 600 BC (Esselink et al. 2000), but the impact of these animals on sedimentation processes and compaction is rarely studied. All these different geomorphological and ecological dynamics have been separately investigated in a large number of studies using a variety of methods to measure different processes. Integrating the knowledge gained from these studies will help us understand the complex interplay between biotic and abiotic factors and sedimentation, which is important for protecting these ecosystems (Corenblit et al. 2011).

Aim of the review

The increasing research focus on ecosystem engineering and bio-geomorphological interactions in tidal marshes has motivated the writing of this overview. It is meant to enable researchers to choose an appropriate method for investigating the interactions between sedimentation and ecological processes. This review gives an inventory of the available methods to quantify sedimentation, vertical accretion and erosion processes in coastal marshes for researchers from different fields. Table 1 lists the characteristics of these methods, including important references. In order to further improve the quality of such measurements, we discuss the possibilities offered by and limitations of these various methods and suggest possible combinations of different methods. All methods have specific advantages as well as disadvantages. Often, such disadvantages result from the method influencing or disturbing some part of the

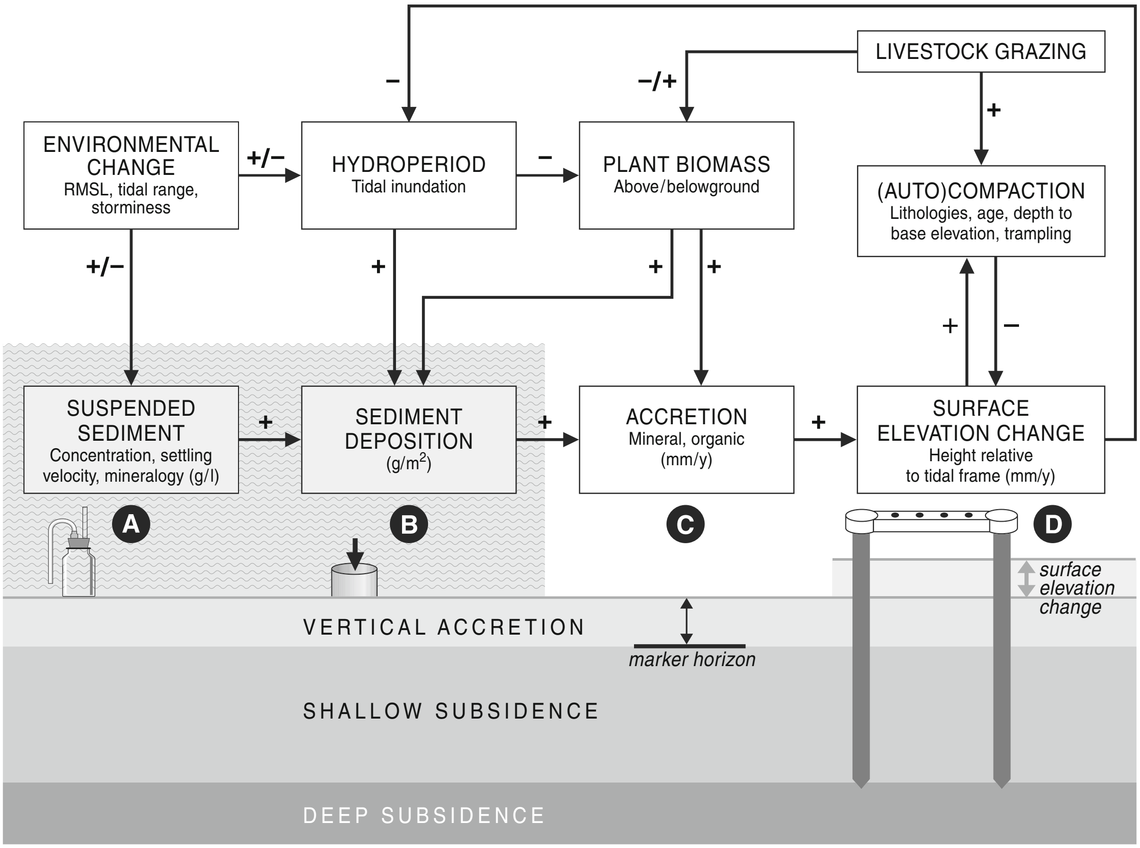


Fig. 1 Factors affecting sedimentation processes in coastal marshes after Allen (2000) and Cahoon et al. (2002b). The letters **A**, **B**, **C**, and **D** indicate the subsections of this review

sedimentation processes, or interfering with exactly the ecological process that is of interest in bio-geomorphological studies. Thus, depending on the aim of a study, a method should be chosen that measures the correct aspect of sedimentation, and which interferes least with the processes of interest. If, for example, the effect of vegetation structure on accretion rate is under investigation, a method that leaves the vegetation intact should be chosen.

Definition of terms

In this review, a wide variety of methods is analyzed. These methods are divided into four categories according to the process they are addressing (Fig. 1): the measurement of suspended sediment concentration (A in Fig. 1), sediment deposition (B), vertical accretion (C) and surface-elevation change (D). Below, the meaning of these four processes is first defined.

The terminology used in this paper is adapted from Cahoon et al. (1995) and Van Wijnen and Bakker (2001) but we supplement it with the term, suspended sediment concentration (SSC) (A in Fig. 1). SSC (in g/l) is used to describe the dry mass of sediment that is suspended in a defined volume of water. This refers to water running in creeks or flooding the marsh surface. The process of sediment particles settling out of the water column onto the marsh surface is called sediment deposition (B in Fig. 1) or simply sedimentation (in g/m^2, i.e. dry mass of sediment deposited/surface area of the marsh surface). In contrast to sediment deposition, we define accretion (C in Fig. 1) as the vertical increase in surface elevation (in mm) relative to a specific layer of the soil. This vertical accretion combines deposition (B) and erosion of sediments, as well as the accumulation of dead biomass, such as roots. If the surface of the marsh is measured with respect to a fixed benchmark, we refer to surface-elevation change (D in Fig. 1) (in mm). In the literature, rates of accretion (C) and surface-elevation change (D) are often expressed per time unit (mostly per year), and referred to as accretion rate (mm/year) and rate of surface-elevation change (mm/year). The distinction between accretion and surface-elevation change is not always clear in literature. However, in this paper, we use the term surface-elevation change (D) only if the measurements are compared to a fixed bench mark of known elevation with respect to an ordnance datum. Both accretion and surface-elevation change include a certain amount of subsidence, which is sometimes also called settlement (Kaye and Barghoorn 1964) or autocompaction (Cahoon et al. 1995; Bartholdy et

Table 1 The methods with references to the section and paragraph in this review, the unit in which they measure sedimentation, an important reference, time resolution, pre- or post event, level of cost, level of labour intensity, estimated accuracy, estimated precision, advantages, and disadvantages

Chapter	Paragraph	Unit	Reference	Time resolution	Pre/post event	Cost	Labour	Estimated accuracy	Estimated precision	Advantages	Disadvantages
2.A SSC	2.2 Bottle method	g/l	Temmerman et al. 2003a, b	1 tide-several weeks	Pre	Low	Medium	Low, increased with volume	Low (weather)	Important as model parameter	No information on accretion
	2.3 OBS/SAS	g/l	Thomas and Ridd 2004	Weeks–months	Pre	High	Low	Depends on equipment and calibration	High (instrument can stay on same location)	Semi continuous data	Cleaning of surface to get accurate measurements
	2.4 LISST/ ADCP	g/l	Fugate and Friedrichs 2002	Weeks–months	Pre	High	Low	Depends on equipment and calibration	High (instrument can stay on same location)	Semi continuous data	Cleaning of surface to get accurate measurements
3.B sediment deposition	3.2 Filters	g/m^2	Reed 1989; Culberson et al. 2004	1 tide	Pre	Low	Medium	Tend to underestimate because of wash-out of sediment	High; same location Low; seasonal changes	Repeatability; easy to measure organic matter	Disturbance by livestock
	3.3 Open cylinders	g/m^2	Hargrave and Burns 1979; Bloesch and Burns 1980	Weeks–months	Pre	Low	Medium	Tend to overestimate	High; same location Low; seasonal changes	Collection of sediment	Edge disturbing waterflow
	3.4 Flat surface traps	g/m^2	Steiger et al. 2003; Pasternack and Brush 1998	2 weeks–1 year	Pre	Low	Medium	Tend to underestimate because of wash-out of sediment	High; same location Low; seasonal changes	No edge disturbing water flow	Removal of vegetation
4.C accretion (short term)	4.2 Marker Horizons	mm/y	Steiger et al. 2003; Van Wijnen and Bakker 2001	Months–years	Pre	Low	Low	~5.0–10.0 mm (soil corer)	High; same location but tend to disappear over time	No influence on hydraulics, sampling success assessed directly	Autocompaction, bioturbation, redistribution of material, loss of material by sampling
	4.3 Sedimentation plates	mm/y	Watson 2008; French and Burningham 2003	Months–20 years	Pre	Medium	Medium	~1.5 mm (high)	High; same location	Not destroyed by sampling, can show shallow compaction	Disturbance of hydraulics and roots, uncertainty of plates staying level, burrowing animals
	4.5 'Erosion' pins	mm/y	Saynor et al. 1994	Months–decades	Pre	Low	Medium	~1 cm; BUT disturbance by pin itself increases error	High; same location	Easy to handle	Over/underestimation due to turbulences. Possibly unstable
5.C accretion (long term)	5.2 Paleo-environmental	mm/y	Lefsky et al. 2002	Decades (50 year)	Post	Medium	Medium	Intermediate	Low (cannot take the same core twice)	Data on vegetation included	Historical maps/photographs necessary
	5.3 Caesium (Cs)	mm/y	Callaway et al. 1996; DeLaune et al. 2003; Milan et al. 1995	Decades (50 year)	Post	High	High	Intermediate; 137Cs moves through soil, average over many years	Low (cannot take the same core twice)	Can be combined with ^{210}Pb	Bioturbation, leaching, uncertainty of local peaks
	5.4 Lead (Pb)	mm/y	Appleby and Oldfield 1978; Walling and He 1999	Decades (100–150 year)	Post	High	High	Intermediate; includes calculations with error propagation	Low (cannot take the same core twice)	Can be combined with ^{137}Cs	Bioturbation, leaching
	5.5 OSL	mm/y	Ollerhead et al. 1994; Madsen and Murray 2009	Decades–centuries	Post	High	High	Intermediate	Low (cannot take the same core twice)	Long time period	Inadequate sensitivity of the detector, thermal transfer, incomplete resetting of the material, handling of samples in red-light conditions
	5.6 Radiocarbon	mm/y	Bowman 1990	Decades–centuries	Post	High	Medium	Low	Low	Longest time period	Large error in age calculation

Table 1 (continued)

Chapter	Paragraph	Unit	Reference	Time resolution	Pre/post event	Cost	Labour	Estimated accuracy	Estimated precision	Advantages	Disadvantages
6.C/D hybrid methods	6.2 SET	mm/y	Cahoon et al. 2000, 2002a, b	Months–decades	Pre	Low	Medium	1.5 mm (high)	High (same location)	Very precise	Set-up bit more complicated than SEB
	6.2 SEB	mm/y	Van Wijnen and Bakker 2001	Months–decades	Pre	Low	Medium	1.5 mm (high)	High (same location)	Very precise, easy in minerogenic marshes	Disturbance at installation
7.D surface elevation change	7.2 levelling	mm/y	NN.	Decades	Pre	Low	Medium	0.5–1 cm (high)	Difficult to measure exact same location	Many measurements in short time span	Difficult to cover large areas
	7.3 Airborne LIDAR	mm/y	Nilsson 1996; Lefsky et al. 2002	Decades	Pre	High	Medium	10–15 cm (low)	Difficult to match with references in field	Covers large areas	Need to correct for vegetation cover
	7.3 Ground based LiDAR	mm/y	Huang and Bradford 1990; Nagihara et al. 2004	Decades	Pre	High	Medium	High	High, detailed 3D map	Detailed 3D map	Many scans to get accurate map

al. 2010). Subsidence can be classified into shallow and deep subsidence (Cahoon et al. 1995). Shallow subsidence is the decrease of the marsh surface elevation due to sediment compaction in the top layer of the soil, e.g. by shrinkage of silt, clay or peat deposits due to drying, and decomposition of subsurface organic material. Deep subsidence also includes every form of subsidence that the instrument itself is subject to, such as tectonic and isostatic processes. Therefore, methods that measure vertical accretion (C) generally include the effects of only shallow subsidence, whereas measurements of surface-elevation change (D) include both shallow and deep subsidence in this review. In the case of vertical accretion measurements, the depth of the boundary between shallow and deep subsidence depends on the specific methodology. Thus, the precise definition of the distinction between shallow and deep subsidence varies between methods.

We divide the methods summarised in this review according to the process they are addressing, following the highlighted sections in Fig. 1. The distinction between methods which measure accretion (C) and surface-elevation change (D) is sometimes difficult. In this review, we thus add a chapter in which we summarise a hybrid method which can measure both processes ("Hybrid methods (C/D)"). The methods in each chapter are then further sorted by the timescale they encompass (temporal resolution) (Fig. 2, Table 1).

The accuracy of a method is defined here as how close the measured value is to the real value, in contrast to the precision. The precision is defined as how similar results are to each other if the same measurement is performed repeatedly, preferably at the same time and position. Being able to identify the exact same location for repeated measures of accretion, for example, plays an important role in the

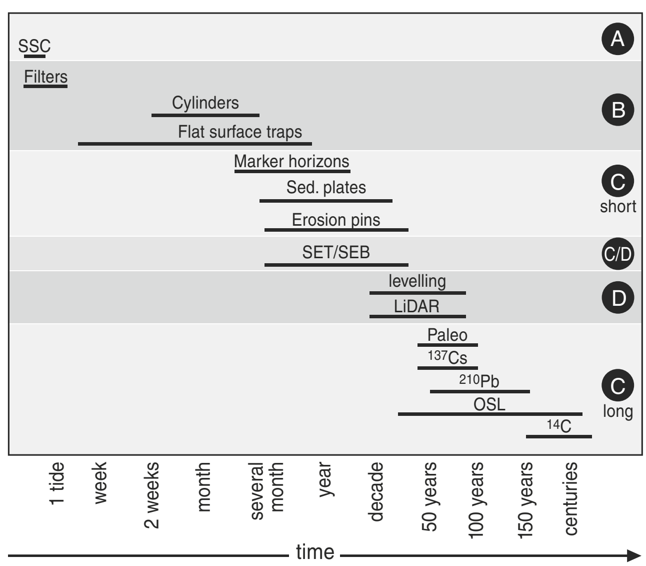


Fig. 2 Overview of methods described in this review divided according to the measured processes and their timescales

method's precision. Both the accuracy and precision of each method is given in Table 1. A relative estimation of both accuracy and precision was made if no value was available from the literature.

Time

In this review, the term, temporal resolution, is defined as the possible time period covered by the method, which ranges between shortest and longest possible application time (Fig. 2). The user should be aware that seasonal effects can influence the measured outcome when choosing a method and measuring period. For example, a higher flooding frequency during the winter season can affect the sediment deposition rate (Temmerman et al. 2005) or the seasonal process of shrinking and swelling of sediments can affect the measured accretion rate (Cahoon et al. 1995). Furthermore, it is important to make the distinction between pre- and post-event methods to measure accretion (C). Pre-event methods, on the one hand, need to be installed in the field before the event of interest takes place and are thus often only useful for relatively short-term investigations. Post-event methods, on the other hand, make use of already existing marker horizons (such as ^{137}Cs) and thus enable researchers to make assessments over longer time periods.

Space

The spatial resolution is the surface area covered by the method. In most cases, the spatial resolution is rather low and only includes the direct vicinity of the measurement (e.g. the surface of a sediment trap [cm^2]). Methods which combine low cost and low labour usually make it possible to perform a high number of measurements at different locations, thus enabling measurement of spatial variations in sedimentation processes. Only a very few methods, such as remote sensing, are able to cover larger areas per se.

Physical disturbances

Furthermore, methods should be chosen with respect to their resilience to disturbance. For example, poles and other structures are known to be disturbed by drift-ice. Other well-known causes for disturbance are the grazing activity of wild or domestic animals, such as grazing geese, sheep or cattle (Dijkema et al. 2005), or bioturbation by a wide number of animals such as crabs (Davidson and de Rivera 2010), small Crustaceans (Schrama et al. 2012), lugworms (van Wesenbeeck et al. 2007), water voles (Kuijper and Bakker 2012), and geese grubbing for belowground parts of the vegetation (Esselink et al. 1997). With respect to grazing, the exclusion of animals (e.g. by fences) is an option in only some cases to protect equipment from damage. Grazing animals are an important part of the ecosystem and if they are excluded, their effect on sedimentation (e.g. through grazing, causing reduction of vegetation structure and perhaps less sediment trapping) is not measured. This means that by locally excluding animals, the accretion rate may be altered and thus measurements inside exclosures would not represent the situation outside exclosures (Esselink and Chang 2010). With respect to bioturbation, both natural and artificial marker horizons can be destroyed or mixed by this process. Bioturbation was found to have a high impact in North-American marshes (e.g. Talley et al. 2001), but negligible effects in some, but not all (Wolters et al. 2005), European marshes (De Groot et al. 2011a, b).

Minerogenic vs. organogenic marshes

Finally, we find some differences in the applicability of methods to minerogenic vs. organogenic marshes. Minerogenic marshes are characterized by a dominance of mineral sediment input that is supplied from suspension in the inundating water. A local organic component of sediment comes from the vegetation on the marsh platform but to a lesser extent. When sea level is stable or falls, in response to century- or millennium-scale fluctuations in sea level, the organic sediment component becomes more dominant and minerogenic marshes may transform into organogenic ones (Allen 2000). At present, very few organogenic peat marshes occur in Europe, except in the Baltic Sea region (Dijkema 1987, 1990). In contrast, the East coast of North America features large areas of coastal peat marshes (Niering 1997). The rate of subsidence is normally higher in organogenic marshes as organogenic sediments are compacted or decomposed to a greater extent than minerogenic sediments. This review focuses on methods suitable for minerogenic marshes, although several may also be applied to organogenic marshes.

Suspended sediment concentration (A)

Introduction

The SSC in water inundating coastal marshes determines the amount of sediment that can potentially be deposited on a marsh. The measurement of SSC alone in a marsh system, however, does not hold any information about the actual deposition rate, as the suspended sediment can be part of either erosion or deposition processes. The SSC often varies both at large scales (e.g. between marshes) and within a single marsh (Reed 1989; Allen and Duffy 1998; Allen 2000; Temmerman et al. 2003a). Measured in g/l or kg/m^3, low SSC values are an indication that the marsh is not likely to accrete over the long term through mineral sediment deposition, although organogenic accretion may be dominant. Including grain size analysis may yield more information on settling velocity and, consequently, sediment deposition rate.

There are two major reasons to measure SSC mentioned in literature. Firstly, it is measured to determine sediment fluxes and budget (Dankers et al. 1984; Asjes and Dankers 1994). Secondly, it is used as a parameter to improve sedimentation models (Temmerman et al. 2003a, 2005; French 2006; Kirwan et al. 2010). When used to calculate sediment fluxes and budgets, SSC measurements are taken at the beginning and end of tidal cycles, often at the mouth of a creek (e.g. Dankers et al. 1984; Asjes and Dankers 1994; Van Duin et al. 1997; Temmerman et al. 2003a). Temporal variations in SSC may be associated with tidal range, inundation height, turbulence and (seasonal) weather conditions, such as storm events (e.g. Osborne and Greenwood 1992; Asjes and Dankers 1994; Allen 2000; Voulgaris and Meyers 2004; Temmerman et al. 2004a). The difference in SSC between the in- and outgoing water is assumed to have been deposited on the marsh surface (Dankers et al. 1984; Reed 1988; Brown et al. 2009). However, measuring at the creek does not give any information about sediment input via inundation from the marsh edge (Leonard and Luther 1995; Van Proosdij et al. 2006), or the spatial distribution of the sediment over the marsh. When SSC is used as a parameter in models, it can be measured at one location (as an indication of available sediment), multiple locations (for use in spatial models; Temmerman et al. 2005), or at different heights in the water column (to monitor decrease of SSC during tides; Schuerch et al. 2012a).

Taking SSC into account can enhance our knowledge about the ability of ecosystem engineers to affect their environment and vice versa. For example, increased SSC levels in the Wadden Sea are thought to have contributed to the disappearance of sea grasses, thus inhibiting their re-establishment (Van Der Heide et al. 2007; Eriksson et al. 2010).

Bottle method and automated sampling

SSC can be measured by taking water samples with bottles manually (Wattayakorn et al. 1990), with semi-automated samplers such as siphon samplers (Gregory and Walling 1971; Grazcyk et al. 2000) (Fig. 3a), or with fully automated samplers (Temmerman et al. 2003b). The collected water samples are usually filtered, dried and weighed (Dankers et al. 1984; Temmerman et al. 2003a), which requires substantial additional work in the laboratory. These methods are widely used on salt marshes (Gregory and Walling 1971).

Manual sampling with bottles (i.e. scooping up sea water) is generally associated with a higher uncertainty, because of the possibility of locally disturbing the sediment layer during the sampling process. This can be slightly improved by taking larger sampling volumes.

Using a siphon sampler is a cheap method for measuring SSC. A semi-automated method, it requires emptying of the bottles after every inundation event. Siphon samplers allow for the measurement of SSC at various depths and times during the flooding phase of inundation (Fig. 3a) (Reed et al. 1999; Grazcyk et al. 2000). This provides valuable information for model parameter estimation because SSC is not homogeneously distributed throughout the creek (French et al. 1995; Grazcyk et al. 2000; Temmerman et al. 2005). However, a siphon sampler cannot collect water samples during ebb tide and the sampler itself may substantially disturb the local hydro- and morphodynamics.

Automated samplers are able to take several samples during a tide. They are more expensive to implement than the other two methods, but they do not need to be emptied every tide (Temmerman et al. 2005), and it is possible to sample during ebb tide. Like the other methods, it is likely to disturb the local hydrodynamics.

Optical back scatter/sediment accumulation sensor

Optical Back Scatter (OBS) sensors and/or turbidity sensors also measure SSC (French 2000; Ridd et al. 2001; Thomas and Ridd 2004; Downing 2006). They were initially developed for near-shore use (Downing et al. 1981), but are now also used in tidal creeks and on marsh surfaces (Leonard and Luther 1995; Leonard et al. 1995; Davidson-Arnott et al. 2002). Usually, OBS-sensors are used to continuously measure SSC for up to several months. These continuous measurements allow for detailed information to be collected on SSC changes during tide inundations at one specific location. For information about the instrument mechanisms see www.campbellsci.ca/Download/LitNote_obsbasics.pdf.

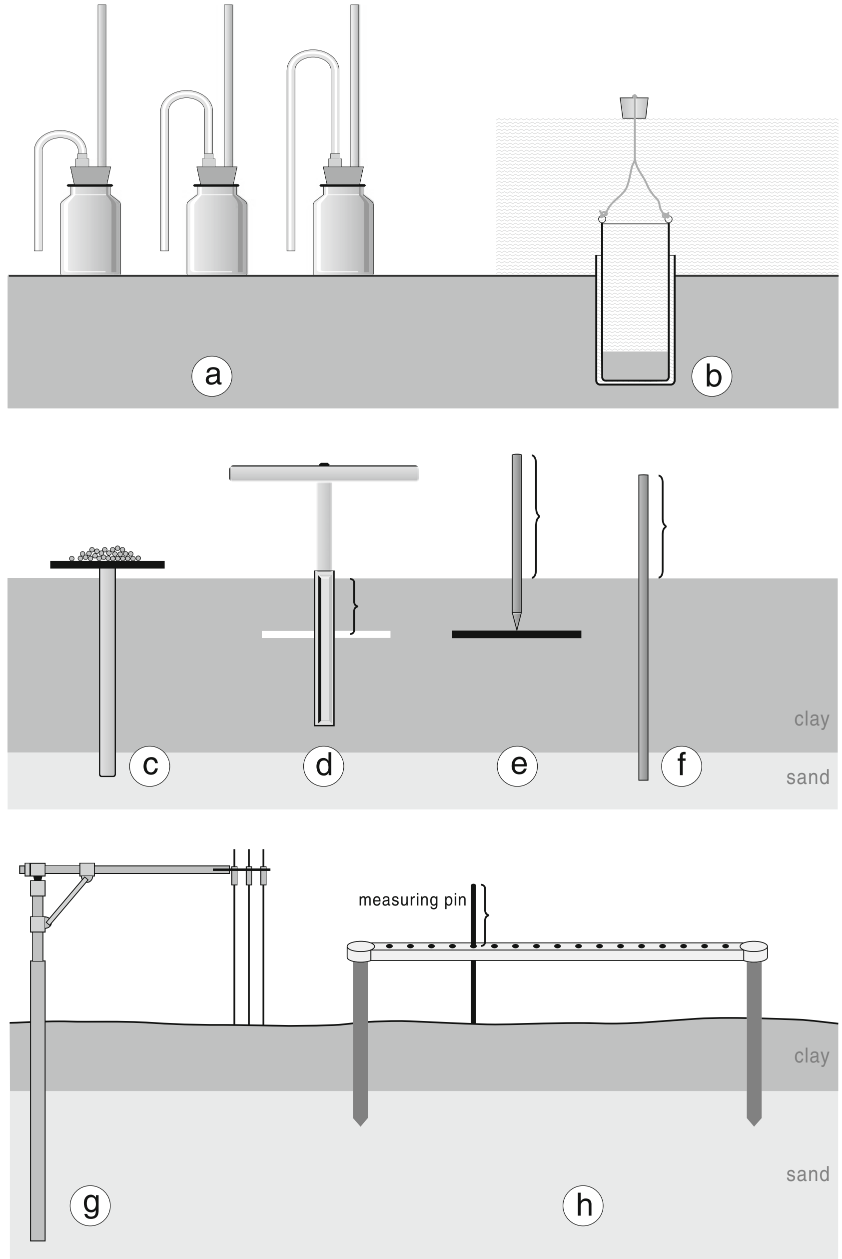
Due to site-specific differences regarding particle properties (such as flocculation and reflectivity), the accuracy of the measurements depends on the proper calibration of the sensor signal against the SSC (Downing and Beach 1989; Kineke and Sternberg 1992; Bunt et al. 1999). When OBS is used on the marsh surface, vegetation has to be removed in order to not disturb the optical signal of the instrument.

The OBS-sensor is the precursor of the Sediment Accumulation Sensor (SAS) (Thomas et al. 2002; Thomas and Ridd 2004), which works in the same way. The initial equipment costs for such OBS or SAS sensors are high. However, these sensors are able to run autonomously for several months, depending on data logger frequency and battery life.

LISST and ADCP

Other automated methods for SSC measurements in fluvial environments include Laser In Situ Scattering and

Fig. 3 Schematic illustration of equipment for **a** bottles, **b** open cylinder, **c** flat surface trap, **d** marker horizon, **e** sedimentation plate, **f** 'erosion' pin, **g** SET, and **h** SEB



Transmissometry (LISST) (Agrawal and Pottsmith 1994, 2000; Fugate and Friedrichs 2002), Acoustic Doppler Current Profiler (ADCP) (Kaneko and Koterayama 1988; Kaneko et al. 1990) and the Acoustic Doppler Velocimeter (ADV). In contrast to OBS-sensors, for which calibration is often difficult, LISST was developed for off-shore studies to provide more accurate measurements (Fugate and Friedrichs 2002). Whereas OBS and LISST use light reflection on suspended sediment particles, ADCP and ADV are based on sound reflection (Kaneko and Koterayama 1988; Fugate and Friedrichs 2002). All of these methods need to be calibrated against SSC measured from water samples (Gartner and Cheng 2001). Although these methods have not yet been applied to tidal marshes, they are feasible candidates for use in tidal marsh research. Fugate and Friedrichs (2002) give a detailed review on OBS, LISST and ADV.

Sediment deposition (B)

Introduction

In contrast to SSC, which only measures potential sedimentation, the methods reviewed in this section measure actual sedimentation of particles in g/m^2 or g/m^2/time. The downward flux of sediment is not only important for studying sedimentation processes on marshes and their reaction to SLR, but can also help to understand the dispersal and deposition of other materials in tidal marshes, such as seeds (Wolters et al. 2004), nutrients (Zhang and Mitsch 2007), silica (Struyf et al. 2007), and organic matter (Costantini et al. 2009). A wide variety of devices, mostly referred to as sediment traps, are used. These traps range from simple plywood boards or tiles (Steiger et al. 2003) to automated sampling devices (Zuniga et al. 2008). The trapped sediment is collected and dried to quantify its weight (Grant et al. 1997; Braskerud 2001). One of the advantages of sediment traps is that the trapped sediment can be used for further analyses such as grain size and chemical or mineralogical characteristics. Especially important to bio-geomorphological studies, sediment traps allow determination of the nutrient content of deposited sediments, and subsequently, relationships with ecological processes such as vegetation productivity. By enabling evaluation of the restoration site's functioning with respect to deposition of sediment and sufficient seed dispersal, sediment traps are very useful in monitoring marsh restoration projects (Wolters et al. 2004) and determining whether goals such as vegetation composition will be met. One problem with these methods is that they do not take into account particles deposited on the vegetation, which may settle on the ground at a later time.

Filters

In this method, filter papers or membranes of a known weight are placed on the sediment surface. The sediment from the inundating water accumulates on the filter during one or a few tides, after which the filter is removed. The accumulated sediment is measured by drying and weighing the collected filters (e.g. Reed 1989; Temmerman et al. 2003a; Culberson et al. 2004). Plastic discs or Petri-dishes are usually used underneath the filter to prevent soil particles from adhering to the underside of the filter (Culberson et al. 2004).

Advantages of this method include low cost and easy repeatability (Table 1). In addition, the organic matter in the deposited material can be easily determined through combustion of the (ash-free paper) filters in 550 °C after drying and weighing them (Reed 1989; Culberson et al. 2004). It is advisable to pre-weigh the dried filters before placing them in the field, in order to correct for any measuring errors that may occur through the drying process, particularly if deposition rates are very low.

Open cylinders

Cylindrical sediment traps consist of containers buried in the ground with the opening level to the soil surface. The variety of constructions ranges from bottles (Deicke et al. 2007), various plastic materials such as PVC pipes (Grant et al. 1997) to glass jars (Jordan and Valiela 1983). Differences also exist between simple cylinders (Grant et al. 1997) and conical traps with a funnel-shaped opening (Bloesch and Burns 1980). A slightly more complicated design was used by Braskerud (2001) (Fig. 3b).

A sediment trap design that combines elements of flat devices and cylindrical traps is described by Temmerman et al. (2003a, 2005): a very flat cylindrical trap with a rim of only a few mm high. The traps are attached to the soil surface using steel claws or a plastic rod running through the middle of the trap (Temmerman et al. 2003a). A buoyant cover, which is lifted by the tide, protects the deposited material from splashing by rain during low tide.

An important consideration when choosing a trap design is the aspect ratio (height/diameter), which strongly influences the turbulence around and within the trap (Hargrave and Burns 1979; Bale 1998). Both trap diameter and aspect ratio should be scaled according to the expected flow conditions and sediment load (Baker et al. 1988). For an extensive discussion on the practical aspects of trap design, consult Bloesch and Burns (1980), Hakanson et al. (1989), and Hargrave and Burns (1979).

Even though cylindrical traps are considered to be the best tool to measure downward settling fluxes within reasonable error limits (Bloesch and Burns 1980) by some authors, they can overestimate sediment flux compared to flat surface traps ("Flat surface traps") (Kozerski and Leuschner 1999). In cylinders, deposition takes place in the absence of significant turbulences and in decreased flow velocities caused by bottom shear, which allows sediment particles to settle faster in the cylinder than they would outside. Depending on trap construction, the method is relatively cheap, but retrieving the traps and drying the samples is labour intensive (Table 1).

Flat surface traps

Flat sediment traps are positioned directly on the soil surface. Simple versions consist of flat plates (Pinay et al. 1995), plywood boards (Mansikkaniemi 1985), clay roof tiles (Brunet et al. 1994) or plastic discs with a roughened upper surface to prevent sediment wash-off (Kleiss 1996). Total deposition rates on surfaces of different roughness were not found to be significantly different by Steiger et al. (2003). Nevertheless, some studies used Astroturf mats (Goodson et al. 2003; Steiger et al. 2003; Deicke et al. 2007) to mimic the effect of vegetation. A slightly more complicated method is described by Pasternack and Brush (1998), who anchored the trap in the soil with an aluminium rod (Fig. 3c). An alternative design of a flat sediment trap is discussed by Kozerski and Leuschner (1999). They introduce another more complex device with a lid construction, which also serves as an alternative to cylindrical sediment traps.

Flat traps are sensitive to sediment being washed away by rain or tides, but the major technical problem is sediment loss during retrieval (Gardner 1980; Kozerski and Leuschner 1999). A further drawback is that the surface of the trap is generally smoother and less adhesive than the marsh surface, affecting the sedimentation and resuspension process. On the other hand, a big advantage of these traps is that they have no edge to disturb the water flow in contrast to cylindrical traps (Table 1).

Accretion (short term) (C)

Introduction

Many studies investigating whether coastal marshes are able to cope with SLR have measured accretion. Therefore, a wide variety of methods exists for measuring accretion rates in mm/yr. We discuss them in the following two sections and classify them according to their timescale, which range from months to decades (short term, this section) or decades to centuries (long term, "Accretion (long-term) (C)") (Fig. 2).

In contrast to earlier described methods, accretion measurements include erosion as well as sedimentation processes of both mineral and organic material. The accumulation of dead plant material, such as roots, can contribute substantially to the yearly elevational rise in marshes. In addition, the fraction of organic material in the soil influences characteristics such as soil moisture content, which in an important factor for plants. The process of accretion also buries seeds. Therefore, studies addressing the availability of seeds as a food source for animals or for the formation of a seed bank might benefit from assessing accretion rates. Additionally, erosion processes are more likely to occur in sparsely vegetated parts of the marsh, such as the pioneer zone. Thus, measuring (short-term) accretion could be used to assess the vegetation-sedimentation interactions constraining establishment of pioneer marsh vegetation.

Marker horizons

A marker horizon serves as a reference layer within the soil, against which accumulation of mineral and organic sediment can be measured using a soil corer (Cahoon and Turner 1989; Cahoon et al. 1995; Van Wijnen and Bakker 2001; Krauss et al. 2010) (Fig. 3d). This measurement is taken with a soil corer and often repeated at regular time intervals, such as every year. A marker horizon generally consists of degradable material, as opposed to plates ("Sedimentation plates"). Some coastal marshes have a natural marker horizon formed by an underlying sand or gravel layer. In such cases, the base elevation of the sandy (or gravel) surface can be used as reference framework if the time of marsh initiation is known (Olff et al. 1997; Van Wijnen and Bakker 1999; De Groot et al. 2011a).

Artificial marker horizons have been constructed from different materials, which are applied to the soil surface, such as red sand (Nielsen 1935), aluminium glitter (Stumpf 1983), red tennis court gravel (Van Wijnen and Bakker 2001), white clay (Baumann et al. 1984), stable rare-earth elements (REE) (Knaus and Vangent 1989), sand (Stoddart et al. 1989; French and Spencer 1993; Nielsen and Nielsen 2002) and feldspar (Cahoon et al. 1995; Krauss et al. 2010). Material effectiveness depends on flooding frequency, wave activity and the retrieval time frame of the marker. Applying the marker material after clipping the vegetation is advisable in cases of dense vegetation (Van Wijnen and Bakker 2001). However, this procedure might change flow conditions and subsequent accumulation of organic material in the first year. For less dense vegetation types such as *Phragmites* sp. or *Scirpus* sp., it is possible to place the marker horizon on the soil surface in between plant stems in order to minimise disturbance of the vegetation canopy (e.g. Temmerman et al. 2004b). Thus, if the research aim is to investigate the effects of vegetation canopy structure on accretion, the marker horizon technique may be a suitable method. However, the level of vegetation disturbance accrued during placement of the horizon depends on the specific vegetation type, and the vegetation canopy may need some time to recover after the placement. The marker horizon area should be marked by sticks, or belowground metal pins or plates that can be found with a metal detector. A new marker layer can be added to the surface every couple of years to minimize the effects of autocompaction on measurements. This should be done, however, in adjacent plots or by using different colours for each horizon to avoid confusing the separate markers. If autocompaction is the focus of the study, stacked layers of markers could be used to assess autocompaction rates.

Several problems hindering accurate recovery are associated with using marker horizons, such as bioturbation (Krauss et al. 2010), redistribution of the marker layer by severe floods (Steiger et al. 2003), and mixing with darker organic or inorganic material (Cahoon and Turner 1989). Furthermore, the necessary coring at repeated intervals removes marker material, resulting in sampling inaccuracy after several years (Stoddart et al. 1989). Lowering the frequency of coring is not a good option, because of the aforementioned autocompaction. Therefore, the size, intended lifetime of the marker, and the number of cores taken per measurement should be considered together. The costs depend mainly on the choice of material, but are generally low to medium (Cahoon and Turner 1989). In addition, sampling effort is not very high and sampling success can be assessed directly in the field (Cahoon and Turner 1989).

Sedimentation plates

In the sedimentation plate method, the marker horizon consists of a firm plate made of metal (Watson 2008; Stokes et al. 2010) or plastic (e.g. nylon or Perspex (French and Burningham 2003)) (Fig. 3e). The plate is buried in the soil just below the rooting zone under a carefully extracted block of marsh turf, which is then placed back on top of the sedimentation plate (French and Burningham 2003). Thus, vegetation disturbance is kept to a minimum. An alternative method (preferable for organogenic marshes) is to dig a hole and carefully push the plate horizontally into the sediment from the side. Small holes drilled into the plate reduce the influence of the plate on drainage conditions and plant rooting. The plates should be placed in a perfectly horizontal position to allow for reliable repeated measurements. After burial, the plates need to settle for at least 1 month before the first measurement can be taken (Stokes et al. 2010). To measure sediment accretion, a thin metal pin is pushed into the sediment until it hits the plate, and its length above the sediment is determined. As with marker horizons, metal plates can be found back by using a metal detector or marking them with sticks. For plastic plates, it is advisable to use metal pins to mark the position of the plate.

The costs for this method strongly depend on plate material but are estimated to be intermediate. The amount of labour involved is also intermediate and comparable to that involved with marker horizons. Errors may occur when plates fail to stay level or are disturbed by burrowing animals such as water voles.

'Erosion' pin

The 'erosion' pin method is usually used in dynamic areas such as dunes and beaches (Saynor et al. 1994; Edeso et al. 1999; Saynor and Erskine 2006; Hancock et al. 2010; Veihe et al. 2011), but can also be applied to salt marshes. With this method, accretion is measured similarly to the Sedimentation Erosion Table (SET) technique (Cahoon and Lynch 1997, "Surface elevation table (SET) and sedimentation erosion bar (SEB)"). In this method, pins of stainless steel or glass fibre are driven into the ground, leaving a small part aboveground (Stokes et al. 2010; Hancock et al. 2010). The length of the pin above the soil is measured repeatedly (Fig. 3f). Thus, the exact location can be re-assessed. However, a major problem of this method is that the pin itself can be unstable. Furthermore, the pin changes flow velocity and turbulence (Veihe et al. 2011), which may lead to scouring of sediment directly around the pin and therefore to an overestimation of erosion. The measurement error associated with this method was found to be about 1 mm (Edeso et al. 1999; Hancock et al. 2010). Method durability varies from months to years (Sirvent et al. 1997; Hancock et al. 2010).

Accretion (long-term) (C)

Introduction

The methods in this section address the measurement of accretion rates over timescales ranging from decades to centuries (Fig. 2) and are used to answer questions on how environmental changes (natural and induced by humans) influence sedimentation processes over the long term. Using these methods, it is possible to increase knowledge on how systems might react to similar changes in the future when combined with, for example, aerial photographs, vegetation maps, and historical records of land-use change. For example, the influence of storm surges on sedimentation rates can be assessed after the events took place (post-event method) using a combination of the ^{137}Cs- and ^{210}Pb-dating method (Bellucci et al. 2007; Schuerch et al. 2012b). As the frequency of storms is predicted to increase, this connection will be very important in determining whether marshes can survive SLR. Other very important factors influencing marsh accretion over the long-term include various human activities. For example, dredging leads to changes in sedimentation rates on salt marshes along the river Schelde (Dyer et al. 2002). Another factor likely to affect accretion rates is grazing by livestock, which is a very common practice both for agricultural and nature conservation purposes in Europe, as the animals modify the vegetation structure and may add to compaction by trampling.

In contrast to the measurement techniques discussed in the previous section, most of the long-term accretion methods can only be employed if erosion during the investigated time span is assumed to be negligible. This is because measurements are made post-event and rely on the accurate estimation of the time period during which a reference layer has been buried. This limitation usually restricts the application of these methods to parts of the marsh that were already present during the time period of interest. Additionally, most methods mentioned in this section can be disturbed by bioturbation as they all depend on the availability of marker horizons or on clear sedimentation layers.

Paleo-environmental method

The paleo-environmental method links historical vegetation data to natural marker horizons consisting of vegetation remains. This technique can be applied to studying marsh accretion rates over timescales up to millennia (e.g. Brush 1989; Neumann et al. 2007; Tanaka et al. 2011) but it is also suitable for shorter time scales such as decades (Temmerman et al. 2003a). Over long timescales, ecosystems generally show a change in plant communities due to succession or land-use change. When conditions are favourable, plant remains such as pollen, roots, or peat are preserved in the soil.

The method consists of taking soil cores, slicing them into layers and identifying the vegetation remains in each layer. The depth of transition layers between plant communities is then related to the date of these changes obtained from historical information (Temmerman et al. 2003a) or dating techniques (Orson et al. 1998; Dobrowolski et al. 2012). Such historical information may include aerial photographs or vegetation maps that document when these transitions between successional stages or land uses took place.

Firstly, this method requires historical data on plant communities and/or land use and, secondly, the right circumstances (e.g. anoxia), for preservation of recognizable plant remains. Furthermore, the number and clarity of successional changes define the time resolution of this method because only a change in species composition results in a datable marker horizon. Note that this method cannot untangle accretion from autocompaction processes but is relatively cheap if labour intensive.

Caesium dating (^{137}Cs)

This method takes advantage of existing marker horizons formed from ^{137}Cs, which is an anthropogenic radionuclide. ^{137}Cs is deposited onto the soil surface after being released to the atmosphere as a product of nuclear accidents. Peaks in the historical release of ^{137}Cs into the atmosphere lead to peaks in ^{137}Cs activity concentrations in the soil profiles of sedimentary environments with relatively constant accretion rates, such as tidal marshes. Nuclear incidents contributing significantly to caesium deposition vary between regions. In Northern Europe, caesium deposition is dominated by atomic bomb tests conducted in the early 1960s and the Chernobyl accident in 1986, leading to two distinct ^{137}Cs horizons (Fig. 4) (Bellucci et al. 2007). Some authors have also found a minor peak at 1974–1977 in cores from the North and Baltic Sea, originating from an incidental release at the Sellafield nuclear reprocessing site (Kunzendorf 1998; Andersen et al. 2000). Chinese and French nuclear bomb tests conducted in 1973 created a significant peak in China (Wang et al. 2008). In America, the deepest occurrence of ^{137}Cs in the soil is assigned to the beginning of atomic bomb testing in 1954 and the highest activity to the peak of these tests in 1963 (DeLaune et al. 2003). The Fukushima accident of 2011 may form a new horizon in Asian marshes.

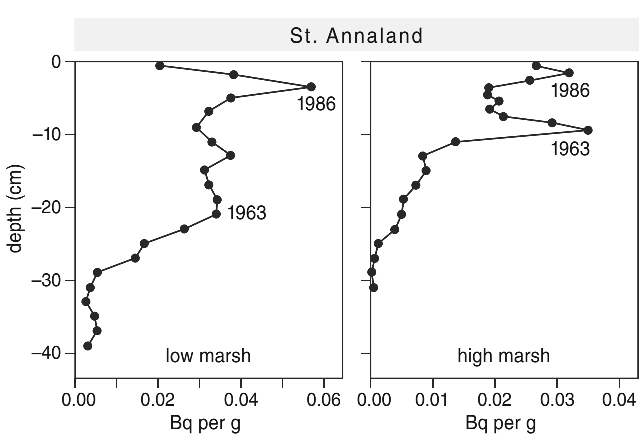


Fig. 4 ^{137}Cs profiles of cores from St. Annaland Marsh, Netherlands (Callaway et al. 1996)

Samples are taken by pushing a tube or cylinder into the soil (Callaway et al. 1996). During and after this procedure, it is important to measure the difference between the rim of the tube and the soil surface inside and outside the tube to quantify the compaction that occurred during the sampling process (Milan et al. 1995; Callaway et al. 1996). Additionally, the length of the core should also be measured before and after extrusion (Callaway et al. 1996), or the core tube (e.g. a PVC tube) should be cut open along its entire length when retrieving the sample from the core tube. The soil core is then cut into slices of 1 cm or greater, and dried (Callaway et al. 1996). The levels of ^{137}Cs are measured by gamma spectrometry and have a detection limit of 0.2 Bq kg^{-1} (Zwolsman et al. 1993; Turner et al. 2001; Bellucci et al. 2007).

The activity of ^{137}Cs in the soil has been shown to increase with smaller grain sizes and higher organic matter content (Kirchner and Ehlers 1998). In case of larger variations in these factors, normalizing this effect should be considered. Therefore, grain size and the organic carbon content should be measured in the same soil cores used to sample caesium.

Interpretation of the ^{137}Cs profile is not always straight forward. For example, the Chernobyl accident is sometimes difficult to distinguish due to regional differences in the fallout, which result from precipitation patterns and additional minor peaks such as Sellafield (Nikulina 2008). Therefore, additional validation of the age with an independent method is advisable, such as ^{210}Pb (discussed below), historic aerial photographs (Schuerch et al. 2012b), and/or the paleo-environmental method (“Paleo-environmental method”). Dating of the layer based on the ^{137}Cs peak may further be complicated by bioturbation and the downward migration of ^{137}Cs (Milan et al. 1995). Finally, ^{137}Cs has a half-life time of 30.5 years so that layers become more difficult to clearly distinguish over time.

Lead dating (^{210}Pb)

The radionuclide, ^{210}Pb, is a product of the natural decay series of ^{238}U, which consists of a chain of isotopes including ^{226}Ra (Walling and He 1999). The ^{210}Pb found in the soil can result from two origins (Fig. 5). Firstly, ‘supported ^{210}Pb’ is continually produced locally from the decay of ^{238}U via the long-lived radioisotope ^{226}Ra and the short-lived ^{222}Rn within the soil, which is in equilibrium with

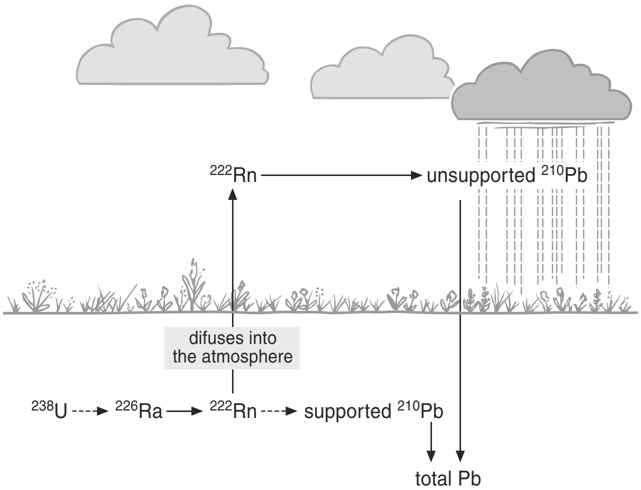


Fig. 5 Formation and pathways of ^{210}Pb until deposition onto the soil surface (Walling and He 1999)

^{226}Ra. Secondly, 'unsupported or excess ^{210}Pb' is deposited from the atmosphere, because parts of the highly mobile ^{222}Rn escape into the atmosphere, where it further decays to ^{210}Pb. Therefore, the unsupported ^{210}Pb also contributes to the total ^{210}Pb inventory in the soil. Thus, unsupported ^{210}Pb is calculated by subtracting the supported ^{210}Pb from the total ^{210}Pb (Fig. 5) (Walling and He 1999), which in turn is measured in combination with ^{226}Ra, for dating purposes. Due to its radioactive decay (half-life time of 22.3 years) the unsupported ^{210}Pb activity concentration in each layer in the soil declines exponentially with its age (Fig. 5) (Appleby and Oldfield 1978; Gelen et al. 2003). This exponential decline can be used to calculate the age of the sediment at different depths from which the sedimentation rate is derived.

Sampling and sample processing require the same steps as described for ^{137}Cs measurements. In fact, ^{210}Pb and ^{226}Ra are usually measured simultaneously with ^{137}Cs from the same sample. There are several approaches to measure the activity of ^{210}Pb. The most widely-used method is to directly determine the activity of ^{210}Pb via counting gamma-emissions using a gamma-ray spectrometer (often a Germanium detector) (He and Walling 1996; Walling and He 1999; Nie et al. 2001; Wang et al. 2008). Before measuring, samples should be hermetically sealed and stored for 20–30 days to allow equilibrium to establish between ^{226}Ra, ^{222}Rn and shorter-lived ^{222}Rn daughters (He and Walling 1996; Gelen et al. 2003; Wang et al. 2008). Indirect methods consist of measuring alpha-emissions of ^{201}Po (Frignani and Langone 1991; Zwolsman et al. 1993; Bellucci et al. 2007) or ^{210}Bi (Tsai and Chung 1989; Applequist 1975; Chung and Chang 1995) instead of ^{210}Pb itself.

After measuring ^{210}Pb activity, an appropriate model should be applied to calculate the date of sedimentation of the respective layer and the related accretion rates. These models are based on different assumptions in connection with ^{210}Pb and its deposition. The Constant Initial Concentration model (CIC) is based on the assumption that accumulation rates of both unsupported ^{210}Pb and suspended sediment are constant over time at the sampling location (Appleby and Oldfield 1978; Andersen et al. 2000; Gelen et al. 2003). From this assumption, it follows that the activity of unsupported ^{210}Pb decreases exponentially with depth in accordance with its half-life time of 22.3 years (Fig. 6) (Roman et al. 1997; Andersen et al. 2000; Gelen et al. 2003). Thus, the constant accretion rate can be determined from the slope of the least-square regression of the exponential decrease of unsupported ^{210}Pb with depth (Fig. 6) (Robbins and Edgington 1975; Goldberg et al. 1977; Zwolsman et al. 1993).

In contrast to the CIC model, the 'Constant flux' or 'Constant rate of supply' model (CRS) assumes that the influx of the isotope ^{210}Pb has been constant over time, but accretion rate has not (Bellucci et al. 2007). The CRS model allows for calculation of sediment age in soil profiles even when there is evidence of rapidly changing accretion rates (Pennington et al. 1976; Appleby and Oldfield 1978). Other studies have tried to improve this approach by calculating age/depth profiles and accretion rates, but these are not yet widely used (He and Walling 1996; Nie et al. 2001). Several studies have compared the validity of these models (Appleby and Oldfield 1978; Gelen et al. 2003; Wang et al. 2008).

Overall, the ^{210}Pb method is widely applied in marine and coastal research. Disadvantages of the method include the possibilities of physical disturbance of the sediments (e.g. erosive events or periods), bioturbation, leaching, and the uncertainties surrounding model selection.

Optically stimulated luminescence dating (OSL)

In the time since Huntley et al. (1985) used the OSL method to date sediments, studies have shown that younger

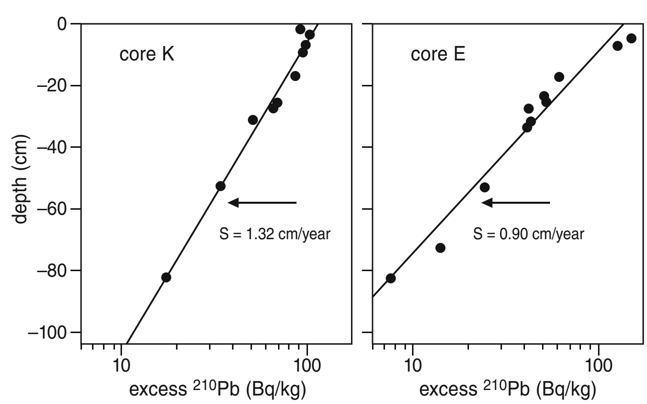


Fig. 6 Relation between depth and ^{210}Pb, which is used to calculate accretion rates (slope) (Zwolsman et al. 1993)

(<60 year) sediments can also be successfully dated with this method (Ollerhead et al. 1994; Madsen et al. 2005). A review of these studies is given in Madsen and Murray (2009). This method uses the phenomenon that natural radiation within the soil affects the crystalline structure of some minerals, such as quartz and feldspar. Energy from natural radiation is stored in imperfections of the mineral structure (Aitken 1985; Madsen and Murray 2009), and can be released through emission of photons, using heat (thermo luminescence) or light (optically stimulated luminescence) as a stimulus (Murray and Wintle 2000, 2003; Madsen and Murray 2009). The amount of stored energy is used to determine when sediments were last exposed to light, i.e. the moment that they were buried under younger sediment deposits. Consequently, it is important that samples are kept in the dark (Reimann et al. 2010) and handled under subdued red light conditions (Pietsch 2009). In addition, specific detectors (including gamma-ray detectors) are necessary to measure total energy absorbed per unit mass (the dose) and the rate of energy absorption (dose rate) in order to calculate burial time. Further detailed information about the principles of OSL can be found in Aitken (1985), Duller (2004), Lian and Roberts (2006), and Duller and Wintle (2012).

In their review, Madsen and Murray (2009) examine the validity of OSL results by assessing several coastal and marine studies (including one salt marsh) where OSL results were compared to an independent method (e.g. ^{210}Pb method). No evidence for systematic over- or underestimation was found, so that the OSL method can be characterised as reliable. An excellent agreement between OSL and ^{210}Pb methods was for example found in the younger part of a core (<60 year) in fine-grained intertidal sediments in the Danish Wadden Sea (Fig. 7) (Madsen et al. 2005).

For further information about technical problems associated with the OSL method, see Madsen and Murray (2009). Drawbacks of the method include very time-consuming laboratory work and the need for specialized equipment, making this method relatively expensive. Additionally, this method is sensitive to bioturbation. During bioturbation, stored energy contained within exposed sediments is completely re-set before soil is reincorporated in the soil column (Madsen et al. 2011) so that accretion rates suddenly seem to increase in upper layers of the soil core. Madsen et al. (2011) identified this sudden increase in calculated accretion rate as the border of bioturbation in the soil and consequently used it as a way to assess bioturbation rates.

Radiocarbon dating (^{14}C)

The method using ^{14}C to date organic material is well known in archaeology. It is sometimes used to determine changes in sea-level by dating basal salt-marsh peats (e.g. González and Törnqvist 2009; Yu et al. 2012). It is also possible to determine long-term sedimentation rates from freshwater (Toledo and Bush 2008) or marine ecosystems (Shaw and Ceman 1999; Watson 2004; Parker et al. 2008; Sabatier et al. 2012) using ^{14}C dating.

During photosynthesis, plants fix atmospheric CO_2 thus incorporating the ratio of ^{14}C/^{12}C isotopes found in the atmosphere at that time. The ^{14}C isotope in organic material decays at a specific rate (half-life of 5,730±40 years), which makes it possible to calculate the age of a sample. The amount of ^{14}C in the atmosphere was not always constant therefore ages must be calibrated to calendar years (Blaauw 2010).

Possible materials suitable for ^{14}C analyses include carbonate shell samples (Sabatier et al. 2012) or fossil plant remains (Shaw and Ceman 1999; Watson 2004), such as basal peats (González and Törnqvist 2009; Yu et al. 2012). For a summary of soil dating using radiocarbon analysis, see Scharpenseel and Schiffman (1977). In the past, ^{14}C-measurements used to be performed by counting the radioactive decay of single carbon atoms by gas proportional counting or liquid scintillation counting. Nowadays, most samples are analysed using accelerator mass spectronomy (AMS) (e.g.

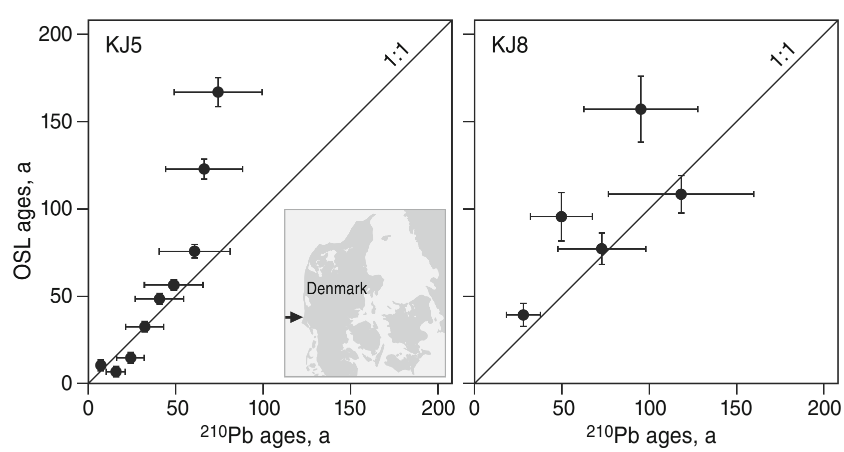


Fig. 7 Comparisons between ages determined using OSL and ^{210}Pb in two sediment cores recovered from the intertidal zone in the northernmost part of the Wadden Sea (Madsen and Murray 2009)

Watson 2004), which allows direct counting of the ^{14}C-atoms instead of indirect assessment from radioactive decay. Details of the method can be found in Bowman (1990) and Goslar and Czernik (2000).

Sedimentation rates estimated from ^{14}C are rough measurements, as they sometimes encompass a small number of measuring point suspended over a long timescale (*ca.* 300–62.000 years). For example, approximately 2,800 years are represented by five measuring points in a study of how human activity, such as deforestation, influenced sedimentation rates in the UK (Parker et al. 2008). Additionally, disturbance can lead to the contamination of samples with younger material, which in turn leads to misinterpretation of sample age. Nevertheless, this method can still provide valuable insights into changes in sedimentation rates over very long timescales (Bowman 1990).

Hybrid methods (C/D)

Introduction

Hybrid methods can be used to measure either accretion (C) or surface-elevation change (D), depending on how they are applied. Technically, these methods measure accretion (C) but can be used for surface-elevation change (D) when they are measured in relation to a recent fixed ordnance datum, thus incorporating deep subsidence processes.

Surface elevation table (SET) and sedimentation erosion bar (SEB)

The Surface Elevation Table (SET) was developed from the Sedimentation Erosion Table (Schoot and De Jong 1988), which was introduced by Boumans and Day (1993) (Fig. 3g). The SET was first applied in the Mississippi river delta by Cahoon et al. (2000). In order to enable SET measurements, a benchmark pole is inserted into the ground until it reaches a stable horizon, such as a sand layer. During actual measurements, the benchmark is used to attach a portable metal arm, with a horizontal metal plate at the end containing nine holes (Cahoon et al. 2002a)(http://www.pwrc.usgs.gov/set/). Metal pins are carefully put through the holes until they touch the soil surface. The length of the pin above the plate is measured to determine the relative surface-elevation change (Boumans and Day 1993; Cahoon et al. 2000). In order to stabilise the benchmark, the Rod SET (RSET) was introduced (Cahoon et al. 2002b).

The Sedimentation Erosion Bar (SEB, Fig. 3h) is further based on the principles of the SET (Van Duin et al. 1997; Van Wijnen and Bakker 2001; Van Duin et al. 2007) but the equipment setup is slightly modified. The setup consists of two horizontally aligned poles, and during measurements, a 2 m-long bar with 17 holes is placed on the poles. Some studies use three poles in a triangle formation to increase the amount of measuring points per station (Van Wijnen and Bakker 2001). There are several minor variations in the construction of the SEB (Daborn et al. 1991; Perillo et al. 2003). In comparison to the SET, this method is less costly and usually applied in Northern Europe on minerogenic salt marshes with low tidal ranges, fine grained sediment, and low accretion rates. In contrast to organogenic marshes, deep compaction is negligible in these marshes.

Both methods are able to measure the exact same points repeatedly, thus increasing the accuracy of the time series to about 1.5 mm vertically (Van Duin et al. 1997; Cahoon et al. 2002a). Installation of the benchmarks is labour intensive, and additionally in areas with high accretion rates, the poles need to be replaced often to prevent burial by fresh sediment. However, the actual measurement process is relatively fast. Installation of SET/SEB can disturb local sediments, which may decrease accuracy at the start of the experiment. For SET, these disturbances can be minimised by using a walkway during both installation and measurements (Cahoon et al. 2002a, b). During SEB installation, however, disturbances will be more difficult to minimise because the measurement points are relatively closer to the poles than for SET.

The SET/SEB-methods were initially designed to measure net surface-elevation change (Cahoon et al. 2000, 2002a, b). However, using SET/SEB to measure surface-elevation change requires that the poles are regularly recalibrated with respect to ordnance datum, or that subsidence of the poles can be ruled out. When poles are not connected to an ordnance datum, the method then simply measures accretion rate.

Surface-elevation change (D)

Introduction

Surface-elevation change is often measured in studies on the influence of SLR on coastal zones. Additionally, it is an important factor used to explain changes in characteristics of coastal marsh ecosystems that depend on inundation frequency, such as vegetation composition. The basic principle of the method is to measure surface elevation with respect to ordnance datum repeatedly, in order to calculate surface-elevation change. Surface-elevation change is measured in mm/yr, like accretion rate, but includes the effect of both shallow and deep subsidence and should always be related to a given ordnance datum. This section only includes methods for which these requirements are met.

Levelling

Levelling is a method used to calculate the elevation change of an area by repeatedly measuring the elevation of the same points (Olff et al. 1997; Esselink et al. 1998; De Groot et al. 2011b). This is done using conventional topographical surveying equipment, such as laser level (e.g. Parkhurst 1928) and total station theodolite (Keim et al. 1999; Lavine et al. 2003). Other techniques include Real Time Kinematic (RTK) and Differential GPS (DGPS), which makes use of satellite GPS signals. Although the latter two techniques are often used in land surveys, their vertical resolution is not always accurate enough for tidal marshes. For example, the vertical accuracy of both RTK and DGPS (several cm) is much less than that of the total station (5 mm).

Relative elevation measurements need to be linked to a fixed benchmark, which is calibrated to an ordnance datum, in order to calculate accurate values of absolute elevation (Keim et al. 1999). Accuracy of the measurements can be estimated by the closure error, i.e. the difference in measured elevation between the same point at the beginning and end of the survey. This error can depend on instrument quality, and weather conditions affecting sight and instrument stability (e.g. wind). The level of labour necessary for this method depends on the type of instrument used and proximity to a benchmark. Furthermore, in the case of repeated measurements, the precision of the method is low.

LiDAR

LiDAR (Light Detection And Ranging) is a remote sensing method based on laser scanning to measure the distance between objects. In this section, we discuss airborne LiDAR and ground-based LiDAR. Ground-based LiDAR consists of 3D laser scanners that emit optical arrays to capture the topography of an area (Huang and Bradford 1990; Fan 1998; Nagihara et al. 2004). Trees and other objects can sometimes obstruct the view of the lasers. To counteract this, multiple scans of an area are commonly used (Nagihara et al. 2004). In general, the merging of multiple scans results in an integrated, detailed and very accurate 3D-map of an area with a resolution of several mm (Nagihara et al. 2004).

Airborne LiDAR is applied from an airplane or helicopter (Nilsson 1996). An advantage of LiDAR is the large geographical area that can be covered in just one flight. Also, the possibility to assess biodiversity in the air (Tuner et al. 2003) may be the main reason to use this method in some studies. However, the presence of vegetation affects the quality of the results. Techniques for correcting for the presence of vegetation have improved over the years, and include using a collinear green wavelength (Nilsson 1996; Lefsky et al. 2002), adding markers (Nilsson 1996), and measuring canopy height in the field. The vertical accuracy is in the order of *ca.* 10–15 cm (Glenn et al. 2006, 2011), which renders this method less suitable for measuring short-term elevation changes. Frequently, salt-marsh researchers may have access to airborne LiDAR data, which were collected by management authorities (e.g. http://www.csc.noaa.gov/digitalcoast/data/click/index.html). In such cases, it is important to check horizontal and vertical resolution, as these may vary considerably between surveys. Additionally, even though the spatial resolution may be high, the temporal resolution is often not accurate enough for biogeomorphological research (Reinhardt et al. 2010).

Discussion

Aim

The wide range of methods available to measure sedimentation processes in interdisciplinary biogeomorphological research of tidal marshes has been summarised in this review. The discussion provides guidelines to help choose the method that is best suited for a specific research question. These guidelines are intended to improve biogeomorphological research on, for example, ecosystem engineers and their application to restoration of ecosystems and their services (Byers et al. 2006).

First of all, the process under consideration has to be identified, which then significantly constrains the number of suitable methods. Secondly, temporal and/or spatial scales need to be determined. Thirdly, more practical considerations for the choice of method are considered: minerogenic vs. organogenic marsh, types of external disturbance expected i.e. livestock, drift-ice or even tourists. A combination of different methods may also help to unravel the interplay between different abiotic and biotic influences on sedimentation processes. Unravelling such interactions is the aim of many bio-geomorphological research studies. We will give several examples of how a combination of methods may improve the knowledge and understanding of coastal ecosystems.

Processes

The first step in choosing an appropriate method, for investigating a specific research question about sedimentation processes, is defining the most relevant process. Does the study focus on SSC (A), sediment deposition (B), accretion (C) or surface-elevation change (D) (Fig. 1)? The distinction between accretion and elevation change can be subtle but important to make. Accretion, on the one hand, can be measured as the additional accreted sediment on top of a marker horizon (mm/yr). This may include, for example, accumulation of organic material, but excludes the influence of deep subsidence or autocompaction underneath the marker horizon (Fig. 1). Thus, if the future trajectory of marsh development is to be predicted, it is especially important to distinguish accretion (C) from surface-elevation change (D) in marshes that are strongly influenced by deep subsidence. If measurements of accretion alone are used, the net change of elevation may be strongly overestimated.

A combination of at least some of the different processes described here should be included if data are collected as input for dynamic models predicting future marsh development. Predictions of these models may also be more robust if they are based on data that were collected by using a combination of different methods. For example, basing dynamic models of coastal sedimentation on physical descriptions of suspended sediment settling and including SSC as a parameter may yield better results than more empirically-based models (Kirwan et al. 2010; Fagherazzi et al. 2011).

Time

The temporal scale is very important in biogeomorphology. For example, an ecosystem engineer's impact may be greatly affected by the scale of its activity over time (Jones et al. 1994). Therefore, depending on the ecosystem engineer, the appropriate time period to be considered can differ significantly, thus affecting the choice of an appropriate method. This choice may further depend on the time that is available for measurements, thus determining whether it is better to use a pre- or post-event method. Pre-event methods may enable the immediate understanding of patterns over short (ecological) time scales (i.e. individual tides to a few years), such as seasonal availability of SSC (e.g. Asjes and Dankers 1994). In contrast, post-event methods are useful in investigating prolonged temporal patterns of sediment deposition relative to ambient conditions, such as the influence of storm frequencies on accretion rates (Schuerch et al. 2012b; Bellucci et al. 2007).

Many long-term techniques are post-event and make use of existing markers in the soil. Temporal resolution for these post-event methods often depends on the deposition rate in the study area. The longest time periods are covered by the ^{14}C-method (300–60.000 year; Bird et al. 1999; Turney et al. 2001, 2006) and the OSL-method (up to 100.000 year; Huntley et al. 1985). Caution is needed when interpreting the results obtained using long-term methods since processes, such as autocompaction (Bartholdy et al. 2010), may considerably influence the measurements. In general, deeper sediments are more compacted. Therefore, accretion rates measured in the first years of measurement may be systematically higher than those of later years (Cahoon et al. 1995). As a result, the estimated average accretion rate becomes smaller with increasing depth of a marker in the soil profile (Fig. 8). Recently, methods have been developed to correct for compaction (Williams 2003; Bartholdy et al. 2010). These corrections can help to evaluate the efficiency of marker horizons. Nevertheless, soil properties that influence compaction can be affected by many factors, such as vegetation type, bioturbators (Schrama et al. 2012) and trampling by livestock.

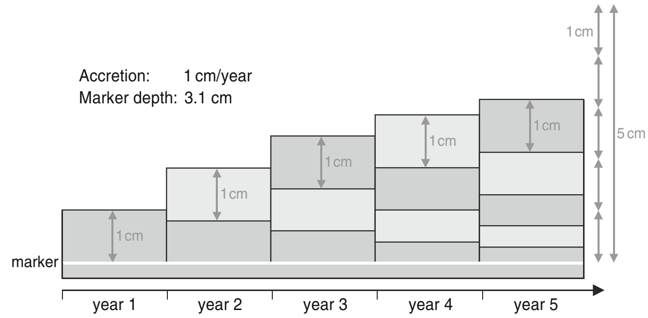


Fig. 8 Schematic illustration of the effects of autocompaction. The actual accretion rate is 1 cm y^{-1}. If measured 5 years after application, the marker horizons is found at a depth of 3.1 cm, because of the autocompaction of deeper layers. This would result in a calculated accretion rate of 0.6 cm y^{-1} and leads to an underestimation of yearly accretion rate

Using post- and pre-event methods in combination (e.g. ^{137}Cs and SET/SEB) can help to resolve how sedimentation processes have changed over time. This was demonstrated by a 15-year study on the Peazemerlannen in the Netherlands, where an underestimation of yearly accretion was identified in the first 2 years (Van Duin et al. 2007). The underestimation was linked to lack of sediment import by winter storms during these first 2 years (Bakker et al. 2002) (Fig. 9).

Space

Ecosystem engineers can have localized impacts, such as mussel beds stabilizing sediment (Eriksson et al. 2010), but their engineering can also affect large areas, such as the disappearance of sea grass is thought to have affected the

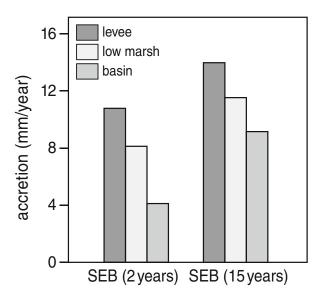


Fig. 9 Comparison of accretion rate measurements using SEB measurements for 2 years and 15 years. Measurements were done on levees, in the low marsh, and in basins at the Peazemerlannen salt marsh, the Netherlands. The 2 year measurements represent a period without major winter storms, which leads to an underestimation if general conclusions are drawn over short timescales

turbidity of the entire Dutch Wadden Sea (Van Katwijk et al. 2000; Eklöf et al. 2011). Furthermore, landscape features, such as proximity to tidal channels, influence the biogeomorphological development of an area over the long term (French et al. 1995; Temmerman et al. 2003a). Therefore, the spatial resolution of a method is important to many research questions. Many of the described methods do not have large spatial resolutions per se, or do not have the necessary vertical accuracy needed for some questions. The spatial resolution of most methods can be increased by spreading the stations over a larger area instead of putting placing them in clusters (De Groot et al. 2011b). Increasing the accuracy, however, is somewhat more difficult. For example, airborne LiDAR covers large spatial scales, but is not very accurate (ca. 10–15 cm). Airborne LiDAR, therefore, is mostly suitable for creating digital elevation maps or digital terrain models, which can be used for various (spatial) analyses (Petzold et al. 1999; Keim et al. 1999).

Geostatistical models can be applied for spatial interpolation of point measurements when combining methods. In many cases, one method may be difficult or expensive to measure (e.g. OBS-sensor), but the measurements maybe correlated with a second, simpler method (e.g. sediment trap) or a method with a large spatial resolution (e.g. LiDAR). Using the spatial correlation between OBS and sediment traps, for example, may increase the spatial coverage value of the OBS-sensor (Diggle and Ribeiro 2007).

Minerogenic vs. organogenic tidal marshes

The differences between minerogenic and organogenic marshes are important when choosing and applying methods to measure sedimentation processes. On minerogenic marshes, mineral sediments are generally the most important contributor to marsh accretion (Allen 1990, 2000). On organogenic marshes, however, organic deposition, originating from dead plant material such as roots (Niering 1997), is the most important contributor.

The significance of different processes (Fig. 1) may be different in these two types of marches. In organogenic marshes, applying SSC (A) and sediment deposition (B) methods may not be relevant, because of the low mineral input. Combining accretion (C) and surface-elevation change (D) methods to quantify shallow subsidence rates may be especially useful in these organogenic marshes. Shallow subsidence may be more important in organogenic than minerogenic marshes, since organic particles are decomposed and compacted to a higher degree than mineral particles. To address this problem, some studies use a combination of marker horizons and SET (Cahoon et al. 2000; Day et al. 2011). Additionally, due to the differences between organogenic- and minerogenic marshes, method adaptations are sometimes advisable. For example, it is not possible to bury a sedimentation plate in an organogenic marsh by extracting a piece of marsh turf, because organogenic marsh soil is less stable. Also due to the relative instability of organic soil, Cahoon et al. (2002a, b) used wooden walkways to prevent disturbances during measurements.

Physical disturbance

When measuring sediment processes, we encounter two different types of disturbance. Firstly, the process of interest itself can be disturbed by the equipment or the act of measuring. Secondly, the accuracy of the measurements can be negatively affected by external physical disturbances of the equipment (e.g. livestock or drift-ice).

The shape of the device used to measure sediment deposition can often affect wind and wave activity, especially near the end of tidal submersion. For example, the shape of the sediment trap can have a very large influence on water flow velocity and direction, which can lead to an overestimation of settled sediment (Hargrave and Burns 1979; Bale 1998). To avoid this problem, a flat device may be a better option. However, many flat devices increase the probability of losing sediment during recovery (Gardner 1980; Kozerski and Leuschner 1999). Another example of disturbance exerted by measuring equipment is the possible influence of sedimentation plates on soil hydrological processes and on the production of roots. When these processes are relevant to the aims of the study, using an SEB/SET might be a better option.

It is advisable to consider the possible effects of equipment disturbance in the research area before installation. Many instruments are easily disturbed or destroyed by extreme events such as severe floods (marker horizons may then be eroded just after installation), drift ice (poles of SEB) or mowing. Poles are also known to attract both wild- and domestic animals (Dijkema et al. 2005), as well as people. This activity may influence the measurements through trampling effects, for example, which was shown in the Peazemerlannen, the Netherlands. Here, the underestimation of accretion increased by 20 % due to livestock grazing (unpublished data). Therefore, the use of inconspicuous methods, such as sedimentation plates, may be preferable in such areas. Furthermore, bioturbation can disturb both natural and artificial marker horizons, as well as the layering of sediments, leading to errors in accretion measurements. If the only bioturbators are small animals, such as the Crustacean, Orchestia gammarellus (Schrama et al. 2012), common sedimentation plates may be a good alternative to artificial marker horizons. If the burrowing animals are bigger, such as the water vole (Kuijper and Bakker 2012), they may also disturb the plates and it is advisable to use hybrid methods.

Biogeomorphology on tidal marshes

The increased interest of ecologists in the interplay between biotic and abiotic factors has illuminated other aspects of geomorphological methods. In ecological and geomorphological studies, it is often realized that the biota are influenced by the geomorphology and vice versa. These interactions may be easy to recognize over short time scales, such as how plants trap sediments and then are affected by nutrients from these sediments. On longer time scales that also encompass evolutionary processes, the influence of the biology on the development of geomorphology is less recognised and/or less understood than the influence of geomorphology on biology (Corenblit et al. 2011). Research on short-term interactions can nevertheless enhance the understanding of long-term interactions.

The methods discussed in this review focus on geomorphological processes. However, the methods can also be used to assess biological processes on tidal marshes. For example, the dispersal of seeds and other propagules can be studied with sediment traps, as these seeds are caught together with the sediment. However, these traps are not always appropriate when studying the effect of ecosystem engineers, such as *Spartina anglica*, as they may underestimate the sediment deposited on the plant and the protection *S. anglica* confers on deposited sediments against re-suspension. Additionally, sediment deposition measurements do not measure the accumulation of dead biomass. To study these kinds of vegetation-sedimentation interactions, buried plates or SET/SEB may be better options as they disturb the interactions between vegetation and sediment less.

Another interesting but largely unstudied factor is livestock grazing, especially in European marshes. Grazing may have a major effect on vegetation-sedimentation dynamics on tidal marshes. On the one hand, grazing livestock may influence accretion rates directly through compaction of the soil. On the other hand, there are various indirect ways in which livestock may affect sedimentation processes. For example, livestock are known to reduce vegetation density in marshes (e.g. Berg et al. 1997), which may in turn alter flow conditions of inundating water and lead to differences in sediment deposition rates and patterns. Additionally, grazing may increase the accumulation of biomass in the soil, as plants which experience grazing are known to produce more roots (Esselink et al. 1998). However, soil conditions maybe become more anoxic for decomposers through compaction and thus grazing may actually reduce the decomposition of biomass (Schrama et al. 2013). In this way, grazing may stimulate organogenic accretion in marshes but this negative effect might be counterbalanced to a certain extent by the positive effect of bioturbation (Schrama et al. 2012). To assess the impact of grazing on marshes, it is thus important to unravel the potentially confounding factors associated with 1) changes in vegetation structure, by measuring sedimentation in grazed and ungrazed vegetation, from those associated with 2) changes resulting from compaction by trampling, by monitoring accretion and shallow subsidence in grazed and ungrazed marshes. It is not advisable to use methods that influence vegetation structure to assess influence on sedimentation processes. To install sediment traps, for example, it is necessary to clip the local vegetation to obtain a bare surface and thus sediment traps are not useful to study the effect of vegetation structure. To do this, we advise the use of SEB in combination with marker horizons, for example, or plates in grazed and ungrazed areas to assess both accretion and shallow compaction. By additionally taking soil samples close to the plot, it is also possible to gain extra information on differences between grazed and ungrazed marshes in soil texture, porosity, and organic matter content.

These examples show how the interplay between abiotic and biotic components of marshes, such as seen in ecosystem engineers, can be studied using a variety of methods to measure sedimentation processes or by applying combinations of these methods. However, ecosystem engineers and biogeomorphological interactions are present in many other ecosystems. For example, sea grasses in the Wadden Sea have been shown to maintain their own preferred environment of fine sediments on intertidal flats (Van Katwijk et al. 2000; Eklöf et al. 2011). Similarly in a lagoon in South-Africa, organic matter content of the sediment was highly influenced by the interaction between reefs, crabs and hydrodynamics (Bruschetti et al. 2011). These are just two examples, and much is still unknown about how these ecosystems function. In the future, more research is needed to understand sedimentation processes and the abiotic and biotic factors influencing them, especially in these times of global change when these ecosystems are threatened by SLR. The guidelines for application, for the methods presented in this review, are probably also valid for research in other ecosystem types with vegetation-sedimentation interactions, such as river floodplains, dunes, and aquatic systems. Insight into these vegetation-sedimentation interactions will hopefully enable researchers to find solutions to protect ecosystems by recruiting the ability of ecosystem engineers to modify their own environment.

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