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Understanding system disturbance and ecosystem services in restored saltmarshes: Integrating physical and biogeochemical processes

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ABSTRACT

Coastal saltmarsh ecosystems occupy only a small percentage of Earth's land surface, yet contribute a wide range of ecosystem services that have significant global economic and societal value. These environments currently face significant challenges associated with climate change, sea level rise, development and water quality deterioration and are consequently the focus of a range of management schemes. Increasingly, soft engineering techniques such as managed realignment (MR) are being employed to restore and recreate these environments, driven primarily by the need for habitat (re) creation and sustainable coastal flood defence. Such restoration schemes also have the potential to provide additional ecosystem services including climate regulation and waste processing. However, these sites have frequently been physically impacted by their previous land use and there is a lack of understanding of how this 'disturbance' impacts the delivery of ecosystem services or of the complex linkages between ecological, physical and biogeochemical processes in restored systems. Through the exploration of current data this paper determines that hydrological, geomorphological and hydrodynamic functioning of restored sites may be significantly impaired with respects to natural 'undisturbed' systems and that links between morphology, sediment structure, hydrology and solute transfer are poorly understood. This has consequences for the delivery of seeds, the provision of abiotic conditions suitable for plant growth, the development of microhabitats and the cycling of nutrients/contaminants and may impact the delivery of ecosystem services including biodiversity, climate regulation and waste processing. This calls for a change in our approach to research in these environments with a need for integrated, interdisciplinary studies over a range of spatial and temporal scales incorporating both intensive and extensive research design.

1. Introduction

Coastal and estuarine environments are faced with numerous challenges, including over-population and economic development, climate change and sea level rise, and water quality deterioration (Kennish, 2002). As a result, the sustainable management of these environments at national and local scales must reconcile regulatory compliance with the demands of a wide range of stakeholders. In addition, considerable emphasis is now being placed on the economic and societal value of the natural functioning of global ecosystems in terms of the flows of materials and energy from natural resources that constitute 'ecosystem services' (Costanza et al., 1997; Millennium Ecosystem Assessment, 2005a; Jones et al., 2011). This is reflected in legislative and policy frameworks for managing and conserving aquatic ecosystems (European Parliament and the Council of the European Union, 1992, 2000; European Academies Science Advisory Council, 2009; Lawton et al., 2010) and the 'ecosystem services' approach provides a common framework for evaluating coastal management options and communicating their consequences to diverse stakeholder groups (Granek et al., 2009).

Although inter-tidal environments such as saltmarshes and mud flats occupy a small percentage (4%) of Earth's total land area, they deliver a wide range of ecosystem services that have significant global value and contribute to national economies (Barbier et al., 2011); in the UK this has been estimated at £48 billion or 3.46% of the UK's national income (Jones et al., 2011). Ecosystem services associated with estuarine and marsh ecosystems at the global scale include: provisioning services such as food, fuel and fibre; regulating services such as nutrient cycling, atmospheric and climate regulation, waste processing, disease regulation and flood hazard regulation; and cultural services such as recreation, amenity and

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aesthetical values (Millennium Ecosystem Assessment, 2005; EFTEC et al., 2006; Costanza et al., 2008; Granek et al., 2009). For saltmarshes, by far the most important benefits are sea defence, immobilisation of pollutants and the provision of rare and unique habitats which support both nursery grounds for fish, and breeding/feeding grounds for birds (Jones et al., 2011). Yet, up to 50% of saltmarshes worldwide have been degraded by human activity and this is likely to have significant impact on critical ecosystem services (Barbier et al., 2011) and as a consequence saltmarshes are frequently the subject of a range of management, restoration, remediation and rehabilitation strategies (Elliott et al., 2007).

An increasingly widespread coastal management approach across Europe and the USA that has the potential to restore saltmarshes and deliver these ecosystem services is Managed Realignment (MR), which is the deliberate removal of a coastal flood defence to allow the tidal inundation of a previously protected low-lying coastal area. Such schemes encompass a range of soft engineering techniques whereby floodwalls or embankments may be breached, removed or lowered (Rupp-Armstrong et al., 2008). Through engineered modifications, both the elevation and hydroperiod of these sites can be controlled by either excavation or sediment re-charge, or by using sluice gates and pumps to control cycles, rates and periods of tidal inundation to enhance the conditions required for specific habitat development (ABPmer, 2010). In Europe and the USA, there are at least 150 MR schemes (or similar) (ABPmer, 2009, 2010) driven by legislative requirements under the EU Habitats and Birds Directives (European Parliament and the Council of the European Union, 1992, 2009) and the Clean Water Act (Committee on Mitigating Wetland Losses, 2001) for habitat (re) creation for either conservation or compensation purposes (Rupp-Armstrong et al., 2008; ABPmer, 2010) and sustainable coastal flood defence. In addition to habitat restoration and coastal defence these schemes have the potential to offer additional ecosystem services including improvements to surface water quality through nutrient and contaminant storage and denitrification, and carbon sequestration (Williams and Orr, 2002; Andrews et al., 2006; Environment Agency, 2007; Shepherd et al., 2007) and therefore have the potential to help EU member states meet their obligations for improving chemical water quality and ecological status under the Water Framework Directive (European Parliament and the Council of the European Union, 2000; Environment Agency, 2010). Despite this, there has been little focus on quantifying the full range of ecosystem services delivered by saltmarsh restoration schemes. Although many restoration schemes are deemed successful and result in environmental enhancements (ABPmer, 2010), there is building evidence to suggest that in restored sites vegetation colonisation may often be poorer/slower than expected, that fewer microhabitats develop, and that restored sites may be less effective at sequestering organic carbon, with higher emissions of the greenhouse gases CO_2 and N_2O and high variability in denitrification rates (Kenny et al., 2004; Elsey-Quirk et al., 2009; Blackwell et al., 2010; Garbutt, 2010; Mossman et al., 2012). This indicates that these restoration sites may not be maximising the delivery of regulating services including nutrient cycling, atmospheric and climate regulation and waste processing, or provisioning services such as wild species diversity.

There is a pressing need for saltmarsh restoration as the maintenance of coastal defences becomes economically unviable and as more space is required for the accommodation of tidal floodwaters and habitat recreation, yet technical, financial and cultural constraints to the further provision of restoration and MR in particular persist (Parrott and Burningham, 2008). Demonstrating not only that current practices result in environmental enhancement, but also that they result in fully functioning ecosystems and maximise the delivery of the full range of potential ecosystem services is crucial in supporting the future expansion of MR and other restoration schemes. Fully functioning ecosystems must be underpinned by the effective rehabilitation and long-term sustainability of inextricably linked ecological, biogeochemical and physical processes (Viles et al., 2008). For restored coastal saltmarshes, associations between vegetation patterns and geomorphic characteristics are broadly understood and already considered within both design and monitoring protocols (e.g. Neckles et al., 2002; Callaway and Zelder, 2004). However, there remains very little understanding of the long-term physical (hydrogeomorphological and hydrodynamic) and biogeochemical functioning of restored sites, the interactions between physical, biogeochemical and biological processes (Townend, 2010) and the impacts this may have on ecosystem service delivery. In addition, many of these restored sites will have been subjected to external system impacts or physical ‘disturbances’ (cf. Viles et al., 2008) associated with their former land use (drainage, urbanisation or agriculture) and/or restoration technique and there is no understanding of how this might affect ecosystem functioning and the potential delivery of ecosystem services in these systems.

This paper first assesses the availability of physical and biogeochemical process data for restored saltmarshes on which we currently base our understanding of ecosystem functioning. Secondly, we examine the impact of disturbance on physical and biogeochemical processes and hence delivery of ecosystem services, focussing on the diversity and development of saltmarsh vegetation. Finally, we consider how such knowledge may initiate a step-change in our approaches to research (and potentially management) in these systems. This complements recent calls within the wider biogeomorphology literature for improved theoretical understanding of complex and non-linear relationships between ecological and geomorphological systems within a range of terrestrial and aquatic environments and over various spatio-temporal scales to inform practical environmental management (Viles et al., 2008; Reinhardt et al., 2010; Rice et al., 2010).

2. Data availability

Current understanding of both the functioning of restored saltmarshes and quantification of the ecosystem services that they deliver is derived from pre- and post-project monitoring data. However, the usefulness of these data is limited and although a wide range of monitoring variables are recommended (including sediment erosion/accretion, surface water flow and hydrodynamics, physical and chemical sediment characteristics, vegetation, birds and fish (Leggett et al., 2004; Environment Agency, 2010)) no standard protocols exist, making comparison of data at both local and regional scales difficult (Neckles et al., 2002). In addition, long-term monitoring data sets are rare due to cost and partly because restoration techniques, such as MR, are relatively new management practices. Consequently, although habitat development may take place quite quickly (e.g. Morgan and Short, 2002; Thom et al., 2002; Byers and Chmura, 2007) these sites are often considered immature in terms of the development of wider ecosystem functioning (Kentula, 2000). An additional obstacle to generating an improved understanding of system functioning is associated with a reluctance to focus on, and report, the less successful aspects of restoration. For example, in a review of MR monitoring activities across Europe (ABPmer, 2010) many projects were identified as either being moderately or highly successful with vegetation development identified as poorer than expected for just two out of 51 projects reviewed. However, a lack of detail on project effectiveness may partly result from the fact that any perceived ‘failure’ can compromise both future funding and stakeholder confidence, but it also reflects a general acceptance that schemes achieving any environmental enhancement have been ‘successful’ in some way, irrespective of whether enhancements are underpinned by the development of a fully functioning ecosystem that maximises the delivery of ecosystem services.

A further issue is a significant lack of data quantifying any physical and biogeochemical processes. For instance, the recent review of monitoring activities within Europe (ABPmer, 2010) indicates that the majority of effort focuses on vegetation structure and other ecological parameters such as birds and invertebrates (Table 1). Comparatively little attention is devoted to physical (hydrodynamic and hydrogeomorphological) and biogeochemical attributes. Furthermore, the majority of these monitoring data are reported in the grey literature and consequently are not readily available for comparison and wider dissemination. Table 1 also summarises the monitoring parameters reported by over 50 research publications, that directly examine the functioning or development of restoration sites following de-embankment and inundation. Our searches may not be exhaustive, but here we include any projects where floodwalls, dykes or embankments were managed (breached, removed or lowered) to allow the tidal flooding of previously-defended land, including projects where flooding was controlled by hydraulic structures e.g. sluice gates or pumps. As illustrated in Table 1, the main focus of over half of these studies was the development of habitat (marsh vegetation species abundance and diversity, often with a bulk indicator of sediment accretion or elevation) and other ecological indicators, with a far smaller number of studies explicitly exploring hydrogeomorphological processes, hydrodynamics or biogeochemical cycling (e.g. Symonds and Collins, 2007; Santin et al., 2009; Blackwell et al., 2010).

3. The influence of disturbance on physical and biogeochemical processes and implications for ecosystem service delivery

The establishment of saltmarsh habitat is considered a two-stage process requiring first, a suitable seed supply, and second,

Table 1
Availability of data on ecological, physical and biogeochemical parameters within MR sites within the grey and primary literature.

Type of parameters recorded	Number of sites for which monitoring data available (grey literature)*	Included within studies in the primary literature†
Ecological		
Vegetation	25	23
Birds	19	2
Invertebrates	17	5
Fish	7	1
Other ecological parameters	3	3
Physical		
Sediment accretion and/or erosion	18	10
Bathymetry (elevation) and hydrodynamics (e.g. current flow or water depth)	9	11
Sediment characteristics	4	4
Morphology	2	5
Biogeochemistry		
Porewater chemistry (salinity, redox, nutrients)	2	8
Sediment chemistry (nutrients, contaminants, organic matter)	3	5
GHG		1
No or limited monitoring	19	
Total no. of sites	51	

* Source: ABPmer, 2010.
† Literature search may not be exhaustive as a result of language and terminology.

conditions suitable for germination and seedling establishment (Erfanzadeh et al., 2010). As a result, studies of saltmarsh vegetation development frequently focus on site suitability in terms of the availability of target species seeds, including supply from the wider catchment and dispersal across the marsh, and the abiotic factors that may inhibit or promote germination, seedling development and plant establishment. Vegetation development is nested within the wider hydrogeomorphic and biogeochemical saltmarsh system (Fig. 1) and will predominantly be controlled by tidal inundation (hydroperiod), the delivery of suspended material to the site and the development of marsh surface morphology (Allen, 2000). However, when coastal saltmarshes are restored these physical parameters are subject to ‘disturbances’ (cf. Viles et al., 2008); also summarised in Fig. 1. Such disturbances include: (1) the return to a tidally influenced and potentially heavily modified and/or artificially controlled hydrology (for example controlled reduced tide schemes); (2) elevation or surface drainage modifications associated with the restoration scheme design and/or the pre-existing morphological features; and (3) changes to the sub-surface sediment structure/hydrology associated with the previous land reclamation (e.g. Hazelden and Boorman, 2001; Crooks et al., 2002; Ellis and Atherton, 2003). In turn, these physical disturbances may alter the biogeochemical processes within restored sites by influencing the rates and pathways of solute exchange and altering the redox status of the sediments. Externally, restoration may also influence the hydrodynamic regime of the wider estuarine system potentially influencing factors such as the supply of suspended material and the tidal prism (Cooper et al., 2004). Therefore, restored saltmarsh sites may have physical and biogeochemical parameters that differ significantly from their natural ‘undisturbed’ counterparts. These disturbances may hinder the development of fully functioning saltmarsh systems within restored sites, potentially reducing the efficacy with which ecosystem services, including those associated with plant species diversity, can be delivered. For example, it has been suggested that poorer than expected vegetation development over longer timescales in restored saltmarshes may be due to poor drainage and sediment anoxia associated with the altered sub-surface structure and hydrology of sediments (Crooks et al., 2002; Grismer et al., 2004; Montalto et al., 2007; Mossman et al., 2012). Similarly, a lack of morphological heterogeneity has been shown to result in the development of fewer microhabitats in restored sites (Elsey-Quirk et al., 2009).

The general philosophy underlying coastal wetland restoration involves the assumption that if a baseline condition is provided (i.e. restoration of tidal hydroperiod), then ‘nature will do the rest’. Whilst hydrochory has been shown to reduce the need for management intervention in the rehabilitation of other aquatic environments (e.g. riparian river habitat; Gurnell et al., 2006a,b), the situation may not be straightforward within restored saltmarsh sites as a result of the ‘disturbances’ identified above. Biodiversity and flood risk mitigation enhancements may be (and have been) achieved irrespective of this (ABPmer, 2010). However, if a return to fully functioning ecosystems is to be achieved, supporting the effective delivery of a wide range of ecosystem services, we need an improved understanding of physical and biogeochemical processes and their interrelationships within restored sites in order to underpin the design and implementation of restoration schemes. Such information will also improve our understanding of how coastal wetlands respond to disturbance. Thus, whilst short-term monitoring that focuses on saltmarsh vegetation species abundance and diversity can provide important insights into patterns of vegetation development in restored sites, such approaches may not be effective in identifying physical and biogeochemical constraints on the longer-term evolution of ecosystem functioning or in

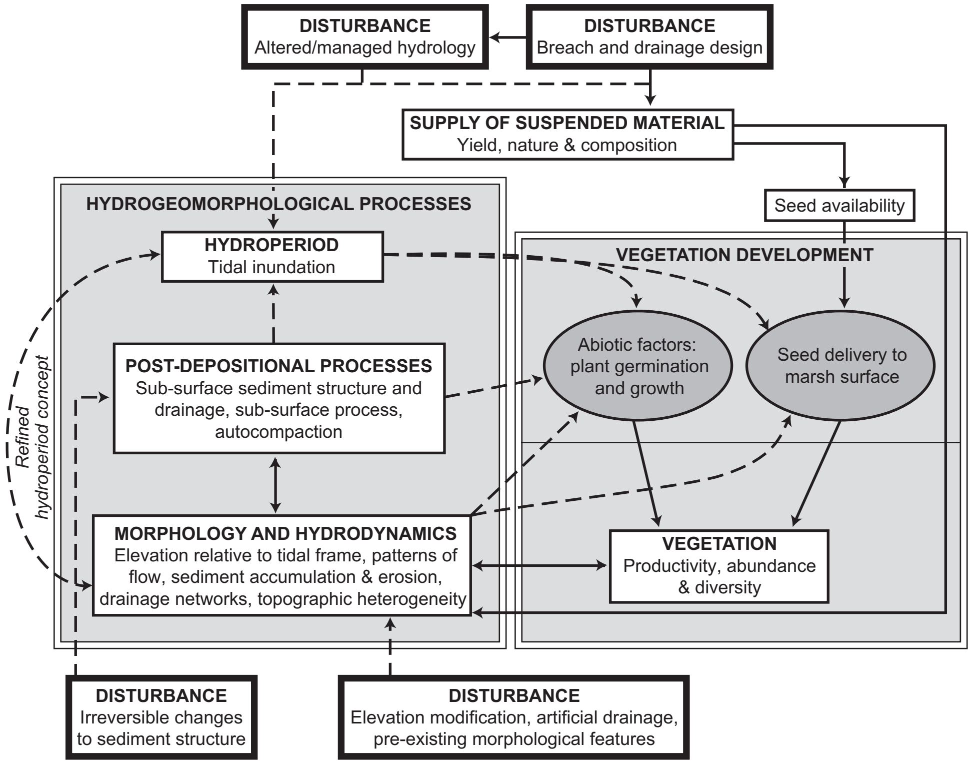

Fig. 1. The proposed composition and linkages of the saltmarsh system within an MR site, showing geomorphic influences of MR as system impacts and linkages between physical and biogeochemical processes and vegetation development. Dotted lines identify areas in which process understanding is most lacking (modified from Allen, 2000 and Viles et al., 2008).

understanding the links between physical, biogeochemical and ecological development (Hughes et al., 2009).

The following sections, supported by Fig. 1, explore existing understanding of the hydrogeomorphological, hydrodynamic, and biogeochemical changes that take place in restoration sites (cf. MR) following inundation and the potential impacts on vegetation development, identifying the main areas in which understanding is most lacking.

3.1. *Supply of suspended material*

Minerogenic and organic material is supplied to the saltmarsh surface in suspension predominantly via tidal flow. In the early stages of saltmarsh formation sediment accumulation is rapid and this decreases as the marsh height increases (Allen, 2000). This influences the relative height of the saltmarsh within the tidal frame and provided there is an adequate supply of suspended material, the saltmarsh will increase in height, keeping pace with sea level (Allen, 2000). In saltmarshes, rates and patterns of sediment accumulation/erosion are important for vegetation development. For instance, excess accumulation can result in seedling burial (Brown and Garbutt, 2004), whilst erosion may have a significant impact on seed retention (Merritt and Wohl, 2002; Wolters et al., 2005a, 2008; Chang et al., 2008).

Restored saltmarshes may not have received supplies of suspended material for many decades due to the presence of hard engineered coastal defences. Once these defences are breached or removed sites are inundated and they capture significant proportions of sediment input (Symonds and Collins, 2004), with sediment accumulating very rapidly following initial inundation dependent upon site hydrodynamics, elevation, topography and sediment supply (Brown and Garbutt, 2004; Cooper et al., 2004; Mazik et al., 2007; Rotman et al., 2008). Sediment accumulation rates are commonly monitored using simple techniques such as marker horizons and sedimentation stakes, pins or plates (e.g. Brown and Garbutt, 2004; Garbutt et al., 2006; Bowron et al., 2009; Howe et al., 2009; van Proosdij et al., 2010). Such approaches provide a general understanding of sediment budgets and broad-scale patterns and rates of sediment delivery (and hence potential for seed delivery) across the flooded area. In most MR schemes high rates of scour at the breach or in the main drainage channels can contribute significantly to sediment loads transported into the marsh (e.g. van Proosdij et al., 2010). The highest rates of sediment accumulation are often observed in relative proximity to the breach (Boyes and Mazik, 2004; Rawson et al., 2004; Rotman et al., 2008) and, in general terms, sedimentation rates decrease with distance from the breach and are highest where elevation is low.

More detailed understanding of the variability in sediment dynamics within restored sites is, however, limited by the temporally- and spatially-averaged nature of the methodologies currently used. Despite inherent technical challenges, a range of techniques for higher-resolution monitoring of sediment dynamics (e.g. suspended sediment concentrations and turbidity) in space and time have been applied within other aquatic environments and within saltmarsh systems under reference conditions (e.g. French et al., 1993; Reed et al., 1999), but generally not within restored sites. A small number of more detailed sedimentation studies in restored sites demonstrate that sediment accumulation patterns and sediment characteristics (e.g. size and sorting) are spatially highly heterogeneous (Takekawa et al., 2010). In addition, patterns of sediment supply are dynamic, changing with the morphological development of the marsh surface following inundation (Chang et al., 2001; Cornu and Sadro, 2002). Such morphological

Fig. 2. Embryo drainage channels quickly form on newly deposited sediments.

development will vary significantly over a range of temporal and spatial scales and will continue to develop as the MR sites mature. This creates a significant gap in both our current knowledge and monitoring approach and there is very little understanding of small-scale spatial and temporal patterns of sediment accumulation and subsequent impacts on vegetation development. In addition, there is no understanding of the relationship between sediment supply and seed delivery. These issues have direct implications for regulating ecosystem services, including flood storage and storage of contaminants, as well as indirectly influencing other provisioning and cultural services by potentially influencing vegetation development.

3.2. Hydrodynamics

Tidal water flow over a saltmarsh is highly complex in space and over time, controlled by the interaction of topography, tidal stage, wind, wave action and vegetation (Chang et al., 2007). Hydrodynamics exert strong controls over patterns of habitat development (Howe et al., 2009, 2010), not least because hydrochory is the main mechanism for seed dispersal in saltmarsh environments (Huiskes et al., 1995; Wilson and Traveset, 2000), but also because it is essential that the marsh surface receives flow for seed delivery and that rates of flow must be favourable for both seed transport and retention (Merritt and Wohl, 2002). In MR sites, where sediments have been previously drained, sub-surface seed banks will have been heavily degraded and are unlikely to be viable (Wolters et al., 2008). Consequently, vegetation development relies almost entirely upon an adequate external supply of target species seeds and, since seeds do not travel great distances, proximity to a local community species pool (Wolters et al., 2005a,b, 2008). As a result, low species abundance and diversity at MR sites have often been attributed to the lack of seed supply and/or seed dispersal traits (Wolters et al., 2005a, 2008).

Patterns and rates of tidal flows across a marsh surface are dynamic and since flows are controlled partly by morphology, will vary as the landscape features of the restored site (e.g. elevation, microtopography and geomorphology) develop (Chang et al., 2007; Torres and Styles, 2007). Tidal flow is distributed across natural saltmarsh surfaces via both creek networks and marsh edge flow, with the relative importance of marsh edge flow changing with marsh size, tidal height and elevation within the tidal frame (Temmerman et al., 2005). Within MR sites this is observed as the initial dominance of sheet flow (Watts et al., 2003; Symonds and Collins, 2007) followed by the (potentially rapid) development of embryo creek networks (Fig. 2), transporting floodwaters, sediments, nutrients and potentially seeds to the interior of the marsh (Cornu and Sadro, 2002; van Proosdij et al., 2010). Altered hydrodynamics in restored saltmarshes will therefore have direct implications for the delivery of provisioning services associated with saltmarsh vegetation development, as well as for regulating services such as storage of sediments, nutrients, contaminants and floodwaters.

Therefore, it is clear that both the spatial and temporal complexity of surface hydrodynamics exert a significant control on the delivery, transport and retention of seeds in saltmarsh environments. Yet, although surface flow is frequently modelled in the MR design process to ensure that sediment deposition, rather than erosion takes place (e.g. French, 2008), patterns and rates of surface flow on newly restored saltmarsh surfaces have not been measured and there have been no observations of how this may develop with time. In addition, there is a general lack of understanding of the relationships between seed retention and tidal flow velocity (Chang et al., 2008) and this lack of knowledge may prohibit us from understanding fully the controls on vegetation development, with consequences for the delivery of species diversity in saltmarsh restoration.

3.3. Morphology

Conceptual models for creek development on saltmarshes indicate that they are strongly influenced by pre-existing landscape features (French and Stoddard, 1992). In restored sites there may have been significant alteration to large-scale morphological features including the introduction of artificial drainage channels, elevation modification and surface features associated with the pre-reclamation land use e.g. plough lines from agriculture. The timescales over which saltmarsh creeks develop in restored sites, and the density and morphological characteristics of the creek network, are influenced by the tidal energy (Crooks et al., 2002; van Proosdij et al., 2010), sediment characteristics (including drainage properties; Crooks et al., 2002), marsh gradient (Cornu and Sadro, 2002) and the presence of pre-existing drainage channels (D'Alpaos et al., 2007). As a result, the successful development of creek networks with characteristics resembling semi-natural conditions is highly variable between restored sites and the geomorphology of drainage networks can resemble pre-MR conditions for many years and potentially remain a permanent feature (Storm et al., 1997; Bowron et al., 2009). The initial erosion and development of drainage channels can be very quick, with channels widening and deepening to accommodate increased tidal flow soon after breach (Williams and Orr, 2002; Symonds, 2006; D'Alpaos et al., 2007; Symonds and Collins, 2007; van Proosdij et al., 2010). Over longer timescales, drainage networks increase in both their density and complexity (Weishar et al., 2005; Bowron et al., 2009). As the creek systems develop, they become more effective; increasing drainage and tidal exchange and frequently enhancing ebb flows (within and outside the MR site), before reaching equilibrium (Williams and Orr, 2002; Watts et al., 2003; Rawson et al., 2004; Symonds and Collins, 2007). Although morphological features such as creeks can develop quite quickly, restored sites show less variation in topography than natural saltmarshes (Elsey-Quirk et al., 2009) and many sites remain poorly drained (e.g. Crooks et al., 2002). Reduced morphological complexity within restored marshes may be of particular significance to provisioning services associated with saltmarsh vegetation development, through influencing the structure and diversity of vegetation communities, whilst poor drainage may influence biogeochemical cycling within restored sites and hence impact carbon sequestration and climate regulation or denitrification and nutrient cycling.

3.4. Post-depositional processes and the sub-surface environment

Following deposition, saltmarsh sediments will become dewatered and compacted with burial, whilst particulate organic matter deposited on the saltmarsh surface, either associated with sediment supply or from *in situ* plant decay, will decompose, typically resulting in an increase in bulk density and a decrease in organic matter content with depth through the vertical sediment profile. Sub-surface hydrology in saltmarsh sediments is influenced by a range of factors including grain size, degree of compaction, topography, tidal pressure and the presence of sub-surface vertical (e.g. burrows and roots) and horizontal (e.g. sand lenses) features resulting in a high degree of spatial heterogeneity in both the physical and biogeochemical environment (Taillefert et al., 2007). Adequate drainage (surface and sub-surface) assists the removal of pollutants and metabolites and establishes a sub-surface unsaturated zone ensuring suitable conditions for seed germination and aerobic root growth (Ursino et al., 2004).

Saltmarsh restoration generally takes place on land that was historically coastal saltmarsh, but has since been embanked and drained, usually for agricultural purposes, often for significant periods of time (decades to centuries). The resultant sediments have been significantly (and irreversibly) altered from their natural state, and have distinctive pore geometries relating to root development and desiccation, low elevation due to clay shrinkage and oxidation of organic matter (Hazelden and Boorman, 2001; Crooks et al., 2002; Ellis and Atherton, 2003) and have frequently been compacted due to agricultural activity. This represents a significant disturbance to the saltmarsh sub-surface sedimentary environment (Fig. 1). It is onto these heavily altered, compacted relict land surfaces that fresh marine sediment accumulates once a site is breached. Generally, the newly deposited sediments have higher bulk density and moisture content, and lower porosity and shear strength than natural saltmarsh sediments (Crooks et al., 2002; Havens et al., 2002; Boorman et al., 2002; Watts et al., 2003; Kadiri et al., 2011) and most studies suggest that it will take decades for MR sites to develop sediment characteristics that resemble those of natural saltmarshes (Craft et al., 2002; Santin et al., 2009). The compacted relict land surfaces within MR sites may act as aquacludes (Crooks et al., 2002), resulting in water-logged surface sediment conditions and limiting the vertical movement of pore waters (Spencer et al., 2008) and resulting in both anoxia and high salinity in surface sediments. Seedling germination can be highly sensitive to hypersalinity and water-logging which results in a lack of dissolved oxygen in the root zone (Engels et al., 2011) and consequently these factors have been identified as being equally or more important than site elevation for controlling vegetation development on MR sites (Wolters et al., 2008; Smith et al., 2009; Erfanzadeh et al., 2010; Howe et al., 2010; Mossman et al., 2012). Thus, sub-surface processes have direct implications for the delivery of key regulating ecosystem services such as immobilisation of pollutants, as well as for provisioning services associated with saltmarsh vegetation development. Despite this, key physical characteristics of restored saltmarshes, such as sub-surface hydrology, sediment texture and structure, which control drainage and anoxia, have to date been largely overlooked (Silvestri et al., 2005).

3.5. Hydroperiod

The hydroperiod, defined as the length of time that a wetland is submerged (Mitch and Gosselink, 2007), is considered to be the most important factor for determining the saltmarsh plant species that may germinate and grow (e.g. Howe et al., 2010). Consequently, ensuring an adequate seed supply, and a hydroperiod similar to reference conditions, is often assumed to be sufficient for habitat development in restored saltmarshes (Garbutt and Wolters, 2008; Wolters et al., 2008). Hydroperiod is largely controlled by elevation, with elevation frequently a key modification in MR design (e.g. Howe et al., 2009). Therefore, as long as elevation permits, once the coastal defence is breached, MR sites are quickly flooded, restoring tidal flow, re-establishing hydrologic connectivity with the wider estuary and resulting in a significant increase to the tidal prism (Williams and Orr, 2002; Bowron et al., 2009). However, hydroperiod can be a far more complex concept, encompassing water depth and frequency of tidal inundations and can be extended to include sub-surface saturation (Ursino et al., 2004; Eaton and Yi, 2009). Considering this refined definition, the hydroperiod will also be controlled by geomorphic features and sub-surface sediment structure (discussed previously in Sections 3.1–3.5; Boswell and Olyphant, 2007; Fig. 1). Many studies have observed relationships between elevation and seed distribution in restored saltmarshes (e.g. Silvestri et al., 2005; Dausse et al., 2008; Smith et al., 2009), yet it is the interrelationships between the subtle, but complex, changes in morphology (and therefore water depth and surface drainage), tidal inundation, sub-surface sediment structure and supply of suspended material that will influence the dispersal of seeds in the saltmarsh environment (Elsey-Quirk et al., 2009) (Fig. 1) and hence potentially influence the delivery of a diverse saltmarsh flora.

3.6. Biogeochemistry

Vegetation development within saltmarshes is predominantly considered to take place via facilitated succession whereby pioneer zone vegetation accretes sediment by reducing flow velocities, encouraging the deposition of fine sediment and stabilising deposited sediments through root development and resulting in an increase in elevation (Hughes et al., 2009). In turn, this, in association with the nature of sediment and freshwater inputs and sediment drainage determine a range of abiotic sediment properties (e.g. salinity, nitrate concentrations and redox) that are important for colonisation by saltmarsh plant species and determining the extent and nature of plant cover (Adam, 1990) (Fig. 1). The sedimentary environment within MR sites, however, has physical characteristics that are very different from both the drained sediments beneath them and from natural saltmarsh sediments (e.g. Craft et al., 2002) and these differences, together with altered hydrology and surface morphological features will have a direct impact on the pathways and rates of solute exchange through the saltmarsh environment and consequently the biogeochemical cycling of both nutrients and contaminants and the abiotic conditions at the sediment surface.

With tidal inundation, sediments within restored saltmarshes may become either saturated or undergo cycles of wetting and drying depending on the site elevation, position in the tidal frame and on the management of tidal exchange for example in controlled reduced tide schemes (e.g. Beauchard et al., 2011). This can result in significant changes to pore water chemistry, salinity and dissolved oxygen concentrations. For instance, several studies note the solubilisation of Fe and Mn sediment phases and the release of nutrients and metals to the overlying water column (MacLeod et al., 1999; Boorman et al., 2002; Blackwell et al., 2004, 2010; Kolditz et al., 2009) suggesting that, in the short term at least, restoration can have a negative effect on overlying water quality. Yet, the understanding of the longer-term changes to sediment and pore water chemistry is poorly understood (e.g. Teuchies et al., 2012). Surface anoxia and limited pore water movement may also have significant effects on the wider ecosystem services that saltmarsh restoration can offer, in particular on regulating and supporting services associated with the biogeochemical cycling of nutrients, carbon and contaminants, which are driven by vertical redox stratification and the physical advection of pore water and hence are controlled by sediment structure and sub-surface hydrology (Huettal et al., 1998; Taillefert et al., 2007; Koretsky et al., 2009). For example, in natural saltmarshes heterogeneity in the rates and pathways of Fe reduction is controlled by the spatial variability and connectivity of sediment pore space and pore water flushing (Kostka et al., 2002; Koretsky et al., 2005). Research understanding the links between sub-surface sediment structure, hydrology and biogeochemistry in restored saltmarshes is significantly lacking, but preliminary studies suggest that denitrification rates are less temporally variable in MR saltmarsh sediments compared to natural saltmarsh sediments and that N cycling is less responsive to changes in floodwater chemistry than in natural saltmarsh sediments due to limited pore water movement (Blackwell et al., 2010). Furthermore, Santin et al. (2009) have demonstrated that restored saltmarshes are less effective at sequestering organic carbon due to the predominance of labile organic matter compared to natural wetlands, and a number of workers have reported higher emissions of both CO_2 and N_2O from MR sites, but the causes for this are unclear (Kenny et al., 2004; Blackwell et al., 2010; Garbutt, 2010).

Despite the potential implications for seedling germination, water quality, contaminant storage, nutrient cycling and global greenhouse gas emissions, biogeochemical cycling within MR sites has also received relatively little attention.

4. A new approach to research in disturbed coastal saltmarsh restoration

Due to legislative pressures for habitat recreation, sustainable coastal defence and water quality improvements, coastal restoration schemes, such as MR, are likely to continue. In addition, such schemes, whereby previously-defended low-lying coastal areas are returned to a tidally dominated hydrology, offer a proxy for how coastal systems may respond to unmanaged coastal flooding following sea level rise, storm surges and consequent defence failure. Yet, these systems have been subjected to a range of intense physical disturbances associated with their past land use that may impact their short- and long-term ecosystem functioning following restoration that are poorly understood. Such disturbance will impact the hydrological functioning of restored sites through changes to tidal inflows, outflows and storage, as well as impacting the transport rates and flow pathways of water in both the surface and sub-surface environment. The geomorphological and hydrodynamic functioning of such sites will also be impacted with changes to the composition, supply and internal distribution of minerogenic material to the marsh surface and the complexity (spatial and temporal) of the morphological features that develop. In turn, this will significantly influence the biogeochemical functioning of these sites including the delivery, cycling and removal of metabolites, nutrients (C, N, P, S) and contaminants with consequences for plant species abundance and diversity, climate regulation and water purification. This means that we do not have the knowledge and understanding to quantify fully the ecosystem services that may be provided by restored coastal systems.

There is also poor understanding of the interactions between physical, biogeochemical and ecological processes in restored saltmarshes. Little is known of the complexity of the relationship between developing surface morphology, sub-surface sediment structure and the influence that this may have on hydroperiod and the provision of abiotic conditions at the marsh surface suitable for seedling germination and plant growth. This raises the question of whether a more refined concept of hydroperiod is needed in order to understand the controls on plant abundance and diversity and to inform the design and management of coastal restoration schemes. Hydrodynamics are frequently modelled and monitored within restoration schemes, yet there is little understanding of the evolution of surface morphology (e.g. drainage networks and microtopography) and surface flow patterns at the spatial scales that may influence sediment, and potentially seed, delivery to the marsh surface with consequences for the microhabitats that develop. Finally, there is little understanding of the links between sub-surface sediment structure and hydrology, and the rates and pathways of solute (including gases) transport through the sub-surface sediment environment. Therefore, it is clear that further interdisciplinary research into hydrogeomorphological, hydrodynamic, biogeochemical and ecological linkages is required to inform the scientific basis of coastal restoration schemes such as MR.

Within the wider biogeomorphology and aquatic systems literature there has been increasing emphasis on the importance of integrating hydrogeomorphological and ecological research from micro- to landscape scales (e.g. Viles et al., 2008; Vaughan et al., 2009; Reinhardt et al., 2010; Rice et al., 2010). However, much of this work refers to freshwater environments, has been published within geomorphologically- and hydrologically-oriented journals (e.g. Gurnell et al., 2000; Darby, 2009) and has been criticised for having insufficient consideration of ecological processes (e.g. Lancaster and Downes, 2009). In contrast, a key problem within the MR literature appears to be a lack of explicit consideration of physical and biogeochemical processes and their role in vegetation development. This dichotomy may in part be due to the drivers of restoration, with many fluvial schemes focussing on the need for flood storage and conveyance, whilst for coastal restoration, habitat (re)creation has frequently been of primary importance.

Due to the disturbed nature of many restored saltmarshes, changes in ecosystem functioning are likely to reflect rehabilitation rather than a complete restoration to pre-reclamation or reference conditions (Bradshaw, 1996). As a result of the uncertainties associated with ecosystem responses to management efforts, adaptive management approaches have been advocated (e.g. Zedler and Kercher, 2005). This requires continued monitoring of system responses to management efforts and relies upon a complete understanding of ecosystem functioning for the restored system. Yet, the literature considered within the previous sections suggests that this understanding is lacking for restored saltmarsh sites.

The challenges associated with conducting interdisciplinary research at the interface of hydrology, geomorphology, biogeochemistry and ecology are significant as terminology, perceptions, methodological frameworks (including different spatio-temporal scales and dimensions), and approaches to explanation vary between disciplines (Richards, 1996; Bond, 2003). In the case of MR, there has been a tendency towards extensive approaches seeking broad trends in vegetation development and causal explanations associated with spatially and temporally averaged morphological and hydrological data. The potential contributions to process understanding that can be gained through intensive 'small-N' approaches (cf. Richards, 1996) typical of geomorphological studies (e.g. Takekawa et al., 2010) should be acknowledged more widely. Indeed, as with freshwater environments (Vaughan et al., 2009), a range of approaches, from small-scale mechanistic experimental research, through intensive process studies to broader scale spatial analysis and characterisation are required to develop a complete understanding of both the impacts of disturbance on the system and the operation of the developing saltmarsh following restoration. In addition to improved collaboration between scientific disciplines, there is also a need to integrate empirical scientific knowledge with application at planning, design and implementation levels, and it has been suggested that the ecosystem services approach may provide an effective framework for ensuring such collaborations (e.g. Reyers et al., 2010).

5. Conclusions

We argue that by settling simply for environmental enhancement, as opposed to the development of rehabilitated inter-tidal ecosystems that maximise the delivery of ecosystem services, current design, monitoring and assessment approaches may result in unrealised potential. This is particularly significant in light of the need for compliance with international legislative frameworks that emphasise the value of healthy, self-sustaining ecosystems. Furthermore, clear demonstration that we can maximise ecosystem services offered by coastal restoration schemes such as MR is likely to foster stronger public support and end-user confidence in the approach.

More holistic approaches to the appraisal of system functioning must necessarily incorporate physical (hydrogeomorphological and hydrodynamic) and biogeochemical processes together with ecological indicators and conventional morphological parameters. These are the areas in which knowledge and understanding are currently most lacking. In order to improve the scientific basis of restoration design, future research should, therefore, focus on: (1) acquisition and effective dissemination of fundamental baseline data for assessing the hydrogeomorphological, hydrodynamic and biogeochemical changes taking place in these disturbed systems; (2) quantification and characterisation of the spatial and temporal variability of these processes at a range of spatial scales; and (3) development of an improved interdisciplinary understanding of linkages between physical, biogeochemical and ecological processes in relation to the development of saltmarsh vegetation. Such research will necessitate improved collaboration between geomorphologists, biogeochemists and ecologists through interdisciplinary research programmes, and the development of closer relationships with stakeholders.

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