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Review on processes and management of saltmarshes across Great Britain

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Abstract

Centuries of coastal development has led to the loss of saltmarsh extent worldwide. As marshes are shrinking, scientific understanding of marsh expansion and erosion processes is growing. Coastal managers are also recognising the importance of marshes for flood protection, carbon sequestration, and pollutant filtering. Considerable effort is now being made to conserve saltmarshes. However, the rapid integration of science in policy remains an obstacle for ensuring successful conservation outcomes. This review explores how advances in the understanding of coastal dynamics, and the evolution of coastal management thinking, are shaping saltmarsh conservation policy in Great Britain. Saltmarsh management has shifted from reclamation, to protection, to restoration throughout the 20th and 21st centuries as calls for nature conservation grew and the importance of ecosystems in coastal erosion risk management became apparent. Studies have revealed that marshes cycle between expansion and erosion phases as part of their natural evolution, governed by processes acting across a range of spatial and temporal scales. Understanding which processes drive long-term marsh change provides an opportunity for coastal managers to undertake targeted intervention for positive conservation outcomes. The inherently dynamic nature of marshes also raises significant challenges in forecasting the long-term value provided by a given marsh. Challenges remain in the monitoring and management of sediment supply and transport, and the effective engagement with stakeholders during habitat protection and creation schemes, which are key to achieving marsh conservation goals.

1. Introduction

Saltmarshes mark the transition from land to sea on sheltered temperate coastlines around the globe, situated between Mean High Water Neap and the Highest Astronomical Tide (Balke et al., 2016). Localised interactions between water movement, sediment flow, and plant growth collectively build marshes, forming unique landscapes of salt tolerant plants perforated by bifurcating channels. Across Great Britain, saltmarsh extent has been considerably reduced in size by land claim for agriculture and industrial development since the Roman Empire. There remains approximately 49,500 hectares of saltmarsh in Scotland, England and Wales (Haynes, 2016; Phelan et al., 2011) (Fig. 1); nearly half the extent which existed 2000 years ago (Doody, 2004).

As saltmarshes were being lost to land conversion, descriptive studies on the composition and distribution of saltmarsh plants across Britain were also being completed (BFBI, 2020). These studies provided a foundation for the development of ecological theory (Tansley, 1947) and coastal geomorphology (Yapp et al., 1916). Saltmarshes are now recognised for their importance in harbouring biodiverse landscapes (Veldhuis et al., 2019), sequestering carbon to mitigate against climate change (Ford et al., 2019), protecting infrastructure from coastal flooding (Möller et al., 2014), filtering pollutants entering the estuarine environment (Enya et al., 2019), and providing important spaces for human wellbeing (Rendón et al., 2019). Continued degradation in marsh extent risks undermining the ecosystem services they sustain, at considerable social and financial cost (Barbier et al., 2011).

Policy towards coastal management has since moved away from land claim for economic growth in favour of marsh conservation for sustaining ecosystem services (Foster et al., 2013). However, there are considerable environmental and societal challenges in achieving protection and restoration

objectives. Already-diminished marshes are being exposed to new threats such as elevated rates of sea level rise, anthropogenic changes to sediment flux, and increased severity of coastal flooding (Gedan et al., 2009). Population growth at the coast is creating conflicting goals between nature conservation and economic development (Corlett, 1978; Doody, 2004). Changes in the environment and policy are changing the way in which the coast has been managed for generations, undermining sentiments of aesthetic, risk, and cultural value held by local communities (Nordstrom, 2014). In extreme cases, entire settlements could be decommissioned as the cost of protection against flooding is deemed too high (Buser, 2020). Swift adoption of current knowledge on socio-ecological system processes into management practice is needed for positive management outcomes (McFadden, 2007). Yet, incorporation of new science into management practice has historically been slow and incomplete, especially in the context of coastal flood risk management (Jarvis et al., 2015).

This review aims to assess how successful current knowledge on saltmarsh erosion and expansion processes is being integrated into conservation policy, using Great Britain (i.e. Scotland, England and Wales) as a case study. The review first presents a synthesis of current knowledge on the processes driving marsh expansion and erosion patterns. Multiple scales are considered, from the interaction of individual plants to wider coastal morphodynamic processes. Whilst there have already been several reviews on marsh evolution (e.g. Reed et al., 2018; Robins et al., 2016; Schuerch et al., 2014; Townend et al., 2011), the focus here is on how processes determining marsh areal extent change can be incorporated into conservation management. The review then explores how attitudes and policy towards marsh conservation have evolved since the Roman Empire, to assess how understanding of coastal morphodynamics has been integrated into management policy. Because this review focuses on marsh extent change, advances in policy directed at ecosystem health (e.g. changes in grazing management; Mason et al., 2019) are only considered in passing. Finally, a summary is presented which highlights where science and policy integration has resulted in successful conservation outcomes, how the British case study can inform marsh conservation policy and practice elsewhere around the globe, and where gaps remain in securing the long-term resilience of saltmarshes.

2. Processes

2.1. Local scale

Saltmarsh expansion is initiated through the colonisation of bare tidal flats by pioneer plant species. Expansion can occur from horizontally growing roots of already-established marshes, or from rhizome fragments and seedlings which deposit onto tidal flat surfaces (Silinski et al., 2016; Wolters et al., 2008). The annual plant *Salicornia europaea agg.* and perennial *Sporobolus anglicus* and *Puccinellia maritima* plants are the dominant pioneer species in Great Britain. All colonise through seed production, whilst *Sporobolus* and *Puccinellia* also expand radially through clonal growth.

2.1.1. Seeds and shoots

Seed dispersal by pioneer plants is highly localised (Zhu et al., 2014), with the majority of seeds being deposited onto tidal flats nearest to established marshes (Erfanzadeh et al., 2010). Successful seedling establishment depends on whether a series of disturbance-free periods, known as ‘windows of opportunity’, become available throughout the growing season (Balke et al., 2014). During these windows of opportunity, seedlings must develop a sufficient root length to avoid being carried away by the next high tide, and to tolerate sediment smothering and erosion from the action of waves and currents (Cao et al., 2018). Windows of opportunity is greatest in years when the number of extreme flood and storm events are low (Balke et al., 2014). The amount of sediment deposition and erosion on the tidal flat increases upon moving down the shore, therefore seedlings are more likely to establish nearest the land (Bouma et al., 2016).

The duration of windows of opportunity is also affected by a range of biophysical processes. Polychaetes, amphipods, and molluscs bioturbate the tidal flat and directly consume or indirectly dislodge seedlings (Gerdol and Hughes, 1993; van Wesenbeeck et al., 2007; Widdows and Brinsley,

2002). Soil anoxia, nutrient and salinity levels, and inundation stress can all affect seedling growth rates (Davy et al., 2001; Hacker and Bertness, 1999; Marani et al., 2006). Macroalgal turfs and microorganism secretions stabilise and trap sediments, which aid in seedling retention, and help regulate temperature and inundation stress (Malarkey et al., 2015; van Hulzen et al., 2007; Widdows and Brinsley, 2002). Excessive macroalgal growth can, however, negatively impact plants by smothering especially in eutrophicated systems (Newton and Thornber, 2013). The same processes which affect seedling survival also affect shoot growth from clonal expansion. However, shoot connection with established marshes increases clonal pioneers' resistance to environmental stress, allowing shoot-connected plants to establish lower on the intertidal (Silinski et al., 2016).

As shoot stiffness, density, and length increases, the plants present a barrier to flood and ebb tides. Water flow decelerates between shoots, and accelerates around plant patches (Bouma et al., 2013). For the clonal-growing and densely vegetated *Sporobolus* and *Puccinellia* species, sediment captured by the plants elevate the marsh surface (Li and Yang, 2009) and enrich the soil with nutrients. A positive feedback is initiated whereby sediment capture promotes the ability of plants to trap more sediment (van Hulzen et al., 2007). However, the positive feedback can be short-lived. Water flow diverted around the plants causes scouring (van Wesenbeeck et al., 2008). Scouring can dislodge plants, and gully formation around tussocks restricts clonal expansion and exposes the tussock to wave and current action (Balke et al., 2012). The individually growing and sparsely distributed *Salicornia* species have a much smaller effect on displacing water flow (Bouma et al., 2013). Only larger plants bend more easily in high flows, which can result in self-scouring and uprooting (Friess et al., 2012). The low capacity of *Salicornia* to modulate tidal elevations may help individuals recolonise the area over and over again (Bouma et al., 2013).

2.1.2. Patches and tussocks

Whether or not plants survive and thrive on the intertidal is strongly influenced by cooperation (facilitation, mutualism, and commensalism) and competition between plant patches and between other organisms. The presence of positive species interactions tends increases with increasing physical stress, following the stress-gradient hypothesis (He et al., 2013). In anoxic soils, oxygen shunted and leaked through roots diffuses into the substrate and becomes available for neighbour plants (Bertness, 1991). Shading by neighbouring plants reduces evaporation and salt accumulation in the soil (Bertness and Yeh, 1994). Bioturbating molluscs can increase water storage and reduce salinity levels in intertidal sediments, protecting plants from drought stress (Angelini et al., 2016). The presence of multiple patches collectively reduce tidal energy higher up the shore through friction (Friess et al., 2012). Collectively, positive species interactions facilitate the establishment of other marsh species, and the formation of closed swards intersected by channel networks (Temmerman et al., 2007). Several examples of other positive species interactions are given in Renzi et al. (2019). As conditions become more benign, however, competition for resources can limit plant growth, increasing the vulnerability of plant patches to scour and dislodgement (Duggan-Edwards et al., 2020). Pioneer vegetation is more readily exposed to herbivory and bioturbation by invertebrates, especially crab species which play an important role in defining the lower limit of vegetation colonisation onto tidal flats (Alberti et al., 2015).

A pattern emerges whereby established tidal flats and established saltmarshes are separated by an inherently unstable region, which continually cycles between plant establishment and erosion through short-term positive and long-term negative feedbacks (Wang and Temmerman, 2013; Fig. 2). Whether plant-plant and plant-animal interactions are positive or negative is influenced by season (Schulze et al., 2019), sediment type (Schwarz et al., 2015), life stage (Bouma et al., 2005, 2009), and hydrological regime (Duggan-Edwards et al. 2020).

2.1.3. Swards

Dense vegetation of established swards can efficiently trap sediment resuspended from the fronting tidal flat (Schuerch et al., 2019). As the marsh gains elevation, the surface is inundated for shorter periods of time, and so rates of sedimentation decline exponentially towards the elevation of the Highest

Astronomical Tide (D'Alpaos, 2011; Vandenbruwaene et al., 2011). Coarse grains are the first sediments to settle out of suspension during over-marsh tides (Temmerman et al., 2004) so that only fine sediments tend to reach the high marsh. Better edaphic conditions in the high marsh also improves plant growth and the rate of leaf and root litter production. Accumulation of fine sediment and plant detritus increases the fraction of organic matter content in the ground upon moving from the pioneer to the high marsh zone (French, 2006). The increasing mass of accumulated material on the marsh surface causes shallow autocompaction of the soil, reducing the rate of vertical growth (French, 2006). Erosion resistance of the soil is increased through soil binding by root growth, increased soil cohesiveness from organic matter enrichment, and soil compaction (Ford et al., 2016). As marshes mature, plant growth is improved through reductions in soil anoxia and associated phytotoxicity through root water uptake (Dacey and Howes, 1984), oxygen diffusion via plant tissues (Pezeshki, 2001), and the establishment of permanently aerated soil layers (Marani et al., 2006). Higher elevation marshes allow less inundation-tolerant plants to outcompete halophytic species, eventually giving way to terrestrial species which mark the landward limit of marsh extent (Allen, 2000). The zonation pattern of plant communities is strongly influenced by soil salinity and anoxia levels, both of which decrease as marsh elevation increases (Veldhuis et al., 2019). Transitions between plant zones are often abrupt, characterised by small differences in marsh elevation. It has been hypothesised that plant communities have the capacity to moderate rates of below-ground organic matter accumulation in order to modify marsh elevations to suit their optimal growth ranges. Shifts in plant communities represents the crossing of thresholds in plant competitive ability and stress tolerance, so that plant communities exist as a series of multiple stable states across the marsh (Marani et al., 2013).

2.1.4. Cyclicity

As marsh surface elevations gradually increase over time, tidal flats fluctuate between accretion and erosion in response to daily and seasonal variation in wave and current forcing (Bouma et al., 2016). Over time, differences in elevation between the vegetated marsh surface and the unvegetated tidal flat accentuate. Beyond a threshold elevation difference, wave and tidal scouring causes erosion at the marsh edge, giving rise to a 'marsh cliff' (Mariotti and Fagherazzi, 2010). Cliffs are a focus for wave impact, which drives lateral marsh erosion through a sequence of slump block formation and basal sediment removal (Francalanci et al., 2013). The rate of marsh cliff erosion is typically an order of magnitude faster than the vertical accretion rates that led to the marsh cliff becoming unstable (van der Wal et al., 2008). Sediment released by lateral marsh erosion is deposited immediately in front of the marsh cliff, facilitating new colonisation of pioneer plants even as the mature marsh erodes (van de Koppel et al., 2005; Fig. 3). Cyclical marsh erosion and expansion patterns have been documented across Britain (Greensmith and Tucker, 1965; Haslett and Allen, 2014; Singh Chauhan, 2009; Yapp et al., 1916).

2.2. Landscape scale

Cyclical marsh expansion and erosion patterns are defined by the dynamics of the tidal landscape. Geology, climate, soft-sediment transport by water flow, and anthropogenic activity exert long-term and large-scale controls over the rates of marsh change and whether erosion or expansion predominate.

2.2.1. Geology

The geological form of the coastline provides an overarching control on where saltmarshes can establish. Gradually sloping topographies, flooded valleys, and topographic depressions at the coast provide sheltered areas for the accumulation of sediments and formation of 'open coast' or 'estuarine' tidal flats and saltmarshes (Burningham, 2008; Pye and Blott, 2014). Whilst geological change tends to occur gradually over millennia, catastrophic events such as earthquakes can rapidly alter where saltmarshes develop along the coast (Orchard et al., 2020).

2.2.2. Temperate and precipitation

Natural and anthropogenically-accelerated climate change is altering temperature and precipitation patterns along latitudinal gradients of the globe. Along tropical-temperate boundaries, mangrove forests are outcompeting saltmarsh plants as temperatures and rainfall rates rise (Gabler et al., 2017). At northern latitudes, *Sporobolus* species are expanding their range and potentially increasing the size of marshes as larger areas of tidal flat become suitable for colonisation (Kirwan et al., 2009). Climate change is also expected to change the timing, frequency and magnitude of extreme weather events, altering estuarine processes and saltmarsh health (Robins et al., 2016). Across Europe, the timings of river floods and plant growing seasons are overlapping. Seedlings are therefore more likely to be uprooted, undermining the long-term survival of vegetated systems (Balke and Nilsson, 2019).

2.2.3. Changing sea levels

Rapid deglaciation at the end of the Last Glacial Maximum (19 and 8 kyr BP) caused a rapid rise in eustatic sea level. Rates of sea level rise gradually reduced by the start of the Holocene (Clark et al., 2009). Removal of the ice sheet mass also caused a rebound of the deformed plate across northern Britain and subsidence about the rising land in southern Britain (Bradley et al., 2011). During the Holocene, relative sea level fell across Scotland and rose in southern England and Wales (Horton et al., 2018). Sea level rise has accelerated once again in response to climate change, and sea level is predicted to rise by 0.29-1.15 m in London and 0.01-0.90 m in Edinburgh by 2100 after taking vertical movement of the land into account (Palmer et al., 2018).

Where sea levels have fallen, high marsh zones are replaced by terrestrial vegetation, and marshes are able to expand laterally onto emerging tidal flats. Prolonged flooding of marsh surfaces during sea level rise increases the time that sediments can settle out of the water column. This positive feedback provides a high degree of resilience to the vertical marsh surface against sea level rise (Kirwan et al., 2016). Vertical accretion is further enhanced by the lateral erosion of the marsh edge (Hopkinson et al., 2018). Sea level rise increases the inundation frequency above tidal flats, promoting both sediment scour by wind waves and tidal currents (Green and Coco, 2014), and the formation of marsh cliffs (Bouma et al., 2016). If there are no physical barriers preventing the hinterland from being flooded, waves and currents also transport eroded sediments inland into the expanding accommodation space allowing marshes to migrate landwards (Schuerch et al., 2018). Where physical barriers inhibit landward migration, marshes are only able to sustain their current extent if sediment supply onto the intertidal offsets the increase in water depth (Ganju et al., 2017). The long-term survival of marshes unable to migrate inland is unlikely. There are few reports of marshes older than 6,000 years, suggesting the deposits are continually reworked as marshes migrate landward and seaward with variations in sea level at geological timescales (Fagherazzi, 2013).

2.2.4. Sediment supply and transport

Sediments reach the intertidal environment primary from river catchment erosion, nearshore marine sources, soft-cliff erosion, and by the reworking of intertidal deposits (Fagherazzi et al., 2013). Prevailing wave action and asymmetric tides modify the foreshore, creating convex or concave profiles as average bed shear stresses equilibrate along the intertidal. Large tidal ranges with little wave action tend to create convex-shaped foreshores which can accommodate wide saltmarshes. Wind-wave exposed foreshores develop concave profiles where marsh development is limited (Hu et al., 2015b). Changes in the prevailing wave and current climate, driven by cyclical variation in climate phenomena (such as the North Atlantic Oscillation and 18.6 year nodal tidal cycle; Haigh et al., 2020), can alter foreshore profiles and hence the sizes that marshes can attain (Leonardi et al., 2016). Sediment composition of coastlines also shifts from fine to coarse deposits as hydrodynamic energy increases. Macrotidal saltmarshes tend to be sand-dominated, whilst microtidal environments are silt- and clay-dominated. Within estuaries, particle size is related to the dominant sediment source. Silt and clay alluvial deposits dominate nearest the estuary head. Large clay and silt particles make up deposits in the river- and tide-influenced regions of the middle estuary. Nearest the estuary mouth, sediments are mainly marine sands (Magar, 2016). The sediment composition of saltmarshes similarly reflects the particle size and source upon moving down the estuary (Jaffé et al., 2001). Exchange of sediment

between the estuary and nearshore environments is regulated by tidal asymmetry. Differences in the duration of flood and ebb tides along the length of estuaries either import or export sediment (Dronkers, 1986; Townend et al., 2007). The accumulation of sediment in a flood-dominant estuary presents a barrier to future incoming tides, slowing the water current. At a critical threshold, the incoming tide is slower than the ebb tide, and tidal asymmetry shifts towards ebb-dominance. The ebb-dominated tide exports sediment until the situation is once again reversed (Brown and Davies, 2010). Flood-ebb cycles have been identified in estuaries throughout Great Britain (Blott et al., 2006; Brown and Davies, 2010; Jago, 1980; Moore et al., 2009; van der Wal et al., 2002). Tidal channels are the conduits by which sediment enters and exists estuaries. Uneven flood and ebb tidal currents case tidal channels to migrate (Li et al., 2008), and river flood events can cause rapid channel avulsion (Braudrick et al., 2009). Both slow and rapid channel migration can cause trigger erosion or stimulate expansion of saltmarshes (Hood, 2010). In extreme cases, tidal channel migration can result in the total loss of a saltmarsh (Pringle, 1995).

Storm surges also play an important role in sediment transport. If coinciding with a high tide, storms can have a profound effect on the foreshore, driving rapid marsh expansion or erosion (de Groot et al., 2011; Fagherazzi and Wiberg, 2009; Tonelli et al., 2010). Storms can deposit large amounts of sediment onto the marsh surface can smother vegetation, causing die-back (Pringle, 1995). However, storms are rarely strong enough to remove vegetated marsh surfaces (Möller et al., 2014). Sediment deposition by storms may in fact be important for marshes to keep pace with sea level rise (de Groot et al., 2011). Storm intensity in southern Britain increased between 1900 and 1960, linked to negative phases in the North Atlantic Oscillation (NAO) atmospheric variability phenomenon (Hurrell, 1995). A switch to positive NAO since the 1960s has resulted in reduced storm surge intensity (Haigh et al., 2010). However, the overall role of storm surges in long-term marsh evolution appears to matter less than changes in prevailing conditions, since storms are by their nature, rare events (Leonardi et al., 2016). Peaks in the positive NAO also coincided with peak tides in the 18.6-year lunar nodal cycle, which contribute to the occurrence of extreme sea levels (Haigh et al., 2011). Increased water depths during high tide increases the exposure of marshes and tidal flats to wave and current erosion, and has been linked to accelerated declines in marsh extent in southern England during the mid-20th century (van der Wal and Pye, 2004; Ladd et al., 2019).

2.2.5. Herbivory and bioturbation

Holocene marshes in Europe were naturally grazed by native herbivores including wild cattle and deer (Allen, 2000). During the Anthropocene, natural grazers were largely replaced by livestock such as sheep and cattle, for agricultural or conservation purposes (Nolte et al., 2013). Despite causing defoliation, soil trampling, and suppressed biodiversity (Ford et al., 2013; Mason et al., 2019), long-term livestock herbivory appears to have little effect on rates of vertical marsh accretion (Elschot et al., 2013) and can even enhance erosion resistance through soil compaction (Pagès et al., 2019). Rapid and intensive herbivory can occur when grazer populations rise as a result of trophic cascades or during migration events (Silliman et al., 2013; Sharp and Angelini, 2021). The defoliation of large areas of saltmarsh exposes plants to salinity stress, fungal infection, and surficial erosion (Freitas et al., 2016). Consumer-driven die-off may trigger long-term marsh loss (Williams and Johnson, 2021), though negative feedbacks have also been observed whereby grazer populations decline due to the lack of vegetation, allowing new plant colonisation in reworked soils (Altieri et al., 2013). Consumer-driven die-offs have been reported in the U.S. and Arctic regions, though appear to be less common or underreported phenomenon across Europe (Hughes and Paramor, 2004).

2.3. Saltmarsh patterns

Human habitation and development along the coast has had a profound effect on coastal dynamics. Saltmarshes in particular have been severely impacted by land and marine use change (Carpenter and Pye, 1996; Gedan et al., 2009; Kirby, 2013). Land reclamation is among the primary cause of saltmarsh loss around the globe (Rodrigues et al., 2017) and represents the single largest reason for marsh extent decline across Great Britain (Allen, 1997). Land reclamation for development and agriculture began in

the Roman Empire along the Severn Estuary (Allen, 1997) and Essex-Kent coastline (Spurrell, 1885). Land reclamation accelerated between the 16th and 18th centuries. The Wash alone has lost nearly 500 ha of marsh to reclamation since Saxon times, and now represents nearly 40% of the former area (Kestner, 1962). Extensive land claim along the major British estuaries, including the Ribble Estuary, estuaries along the Essex and Kent coasts, the Severn Estuary, and the Wash, total in excess of 91,000 ha. Almost all marshes now have some form of embankment as their landward limit (Davidson et al., 1991). Construction of embankments for land claim has also had a legacy effect on local tidal prisms, influencing coastal morphodynamics for centuries (Jongepier et al., 2015). In the Wash for example, embankment construction enhanced sediment deposition. Further reclamation was done in a stepwise fashion once marshes had matured (Kestner, 1975).

Land reclamation was accelerated by the discovery of the *Spartina townsendii* hybrid in the Solent in 1870. *Spartina townsendii* is a hybrid between the North American *Spartina alterniflora* and the native *Spartina maritima* cordgrass. Chromosome doubling meant that S. *townsendii* became a fertile species, recorded as *Spartina anglica* (later *Sporobolus anglicus*; Peterson et al., 2014). *Sporobolus* has a higher tolerance to inundation stress than either parent species, and can therefore colonise lower on the intertidal. *Sporobolus* was planted across Britain in order to raise tidal flat elevations and facilitate rapid land claim (Goodman et al., 1959). After one hundred years of intentional and natural planting, *Sporobolus* covered 12,000 hectares of coastline (Ranwell, 1967), and extended to the Scottish border, limited by sandy sediments and lower mean temperatures (Lacambra et al., 2004).

Since the 1850s, the areal extent of marshes across Britain has increased in the north and declined in the south (Ladd et al., 2019). A number of intertidal habitat creation schemes have been completed in order to compensate for the losses in marsh extent, especially along the southern England coast, via the de-embankment of historically reclaimed saltmarshes (Wolters et al., 2005). Nearly 3,000 ha of new intertidal habitat has been added to the British coast since the 1990s (Fig. 4). However, the rates of natural and artificial marsh expansion are not yet enough to offset national losses (Phelan et al., 2011).

3. Management

3.1. Reclamation

The first law pertaining to the management of the coast, and saltmarshes by extension, was the Sewers Act of 1532 (Richardson, 1919). The Act established local commissioners to survey, maintain, and repair embankments. The Act remained largely unchanged for 600 years, until the 1906 Royal Commission on Coastal Erosion was tasked with developing a new national coastal flood management framework. The Commission was formed to tackle the ever-increasing risk of infrastructure damage by erosion, as coastlines were being steadily developed throughout the Industrial Revolution. The Commission identified that saltmarshes could protect sea defences against erosion, and recommended that *Sporobolus* should be planted to enhance flood protection. The Commission also identified that building sea defences and dredging activity could alter coastal morphodynamic processes, potentially inhibiting sediment transport to the coast and increasing erosion risk. The Commission stopped short of calling for a halt on saltmarsh reclamation, since the economic gains of land claim far outweighed the economic losses caused by embankment breaches. In light of the Commission's findings, the Land Drainage Act was passed in 1930 to better manage and maintain sea defence infrastructure (Nature, 1937).

3.2. Protection

The first major national policy on nature conservation was the 1949 National Parks and Access to the Countryside Act (Smith, 1947). Surveys of the British landscape, gathered during the Second World War, identified areas of land deemed important for natural heritage conservation. Areas were designated as Local and National Nature Reserves, Sites of Special Scientific Interest (SSSIs), Areas of Outstanding Natural Beauty, and National Scenic Areas. Each designation confers varying degrees of protection against land development using a statutory framework for rejecting planning applications.

Sections of saltmarsh were designated for protection, mostly for the preservation of bird habitat (Davidson et al., 1991).

Several national, European, and international treaties on protecting migrating birds were introduced during the 70s and 80s including Biosphere reserves (MAP project), Ramsar sites (Ramsar Convention), and Special Protection Areas (EU Birds Directive). These designations provided indirect protection for saltmarshes as important habitats for birds, and raised awareness on the importance of large-scale conservation of wetlands. The end of the 1970s was also an important turning point in saltmarsh management. The proposed enclosure of Gedney Drove End marsh in the Wash was rejected on the grounds of environmental conservation, marking the end of large-scale land enclosure in Britain (Doody, 2004).

3.3. Restoration

In 1989, the first national inventory of saltmarsh extent and plant composition was published (Burd, 1989). The first saltmarsh habitat recreation scheme was also underway at Northey Island, Blackwater Estuary in Essex (EN, 1994). The scheme involved the realignment of sea defences and the rewetting of formerly reclaimed land. The aim of the project was ostensibly to alleviate wave forcing on surrounding embankments, manage flooding caused by failing embankments, and create new saltmarsh habitat for conservation and coastal erosion management. Following the successes of the managed realignment scheme at Northey Island, a landmark meeting held at London Zoo in 1993 saw the adoption managed realignment as a viable way to restore saltmarshes and protect against coastal flooding nationally (Birks, 1993). The London Zoo meeting marks an important turning point in extending the saltmarsh management remit to restoration, not only conservation.

In 1992, the Rio Convention established a framework for enabling global biodiversity conservation. The UK consequently published a series of Biodiversity Action Plans (BAPs) from 1994 onwards, compelling action towards understanding drivers, aiding recovery, and regular reporting on the condition of threatened species and habitats (UK BP, 2007, 2006). Targets were initially set to maintain saltmarsh extent at a 1992 baseline by creating 60 ha of new habitat each year (Covey and Laffoley, 2002) and reports on the causes of saltmarsh expansion and erosion were prepared (Carpenter and Pye, 1996). In the same year, The EU Habitats Directive expanded protection enshrined in the Birds Directive, and listed several saltmarsh habitats for targeted conservation. The Directive led to the establishment of the Natura 2000 network for Special Areas of Conservation (SAC) sites across Europe (Foster et al., 2013) alongside commitments in the monitoring of ecosystem extent and health under the 2000 EU Water Framework Directive (Best et al., 2007).

The EU-LIFE funded 2003 'Living with the Sea' project was established to support the improvement of coastal Natura 2000 sites across Britain. 'Living with the Sea' sought to develop a strategic approach to record and predict coastal habitat change over decadal timescales, and to identify specific on-site measures to enable the recreation and restoration of inherently dynamic coastal habitats. A key output of the project was the concept of Coastal Habitat Management Plans (CHaMPs). CHaMPs extend policies of flood risk management to the natural environment, and incorporate estuarine and coastal-wide processes to predict localised habitat dynamics. CHaMPs emphasise: (i) the need to work with natural dynamic change rather than take steps to resist it, and; (ii) that flood and coastal defence requirements could work in harmony with the preservation of coastal habitats over the long-term, utilising the natural flood protection afforded by coastal habitats like saltmarshes (Duffy, 2003).

Coastal management policy was, once again, under revision at a national level during the 1990s. Shoreline Management Plans (SMPs), first published in 1993, were non-statutory policy documents which provided coastal defence management advice based on local reviews of coastal geomorphic processes and coastal erosion and flooding risk. The remit of each SMP was defined by the transport pathways and boundaries of sediment flux, known as littoral 'sediment cells'. The main objective of SMPs was to predict the likely evolution of the coast within a sediment cell over three epochs (2025, 2055 and 2100). Four strategic coastal defence options were presented within each epoch: 'hold'

existing defences, 'retreat' by realigning defences further up the coast, 'advance' by reclaiming intertidal land, or 'do nothing' and allow coastal processes to naturally proceed (Cooper et al., 2002). A second generation of SMPs was published in 2006, which better incorporated likely impacts of climate change, new scientific research on coastal processes, inclusion of stakeholders in the planning process, and objectives of other relevant plans including BAPs and CHaMPs (French et al., 2016). Findings from several national programmes including EMPHASYS Consortium (EC, 2000), FutureCoast (Burgess et al., 2004), EUROSION (RIKZ et al. 2004), and Foresight Future Flooding Assessment (Evans et al., 2004) informed the next generation of SMPs. The 2002 UK Climate Impacts Programme (updated under the UK Climate Predictions in 2009 and 2018) has continued to provide national-level predictions of climate scenarios which inform coastal planning (Palmer et al., 2018).

Central to the SMP guidance has been an advocation of adaptive and sustainable coastal risk management. Emergence of the 'ecosystem services' concept (MEA, 2003; UK NEA, 2011) has helped communicate the value of natural ecosystems, such as saltmarshes, in mitigating against coastal flooding (Zhu et al., 2020) whilst simultaneously providing a range of direct and indirect benefits to human well-being (Table 1). Saltmarshes are being increasingly incorporated into coastal erosion risk management, especially in the context of 'do nothing' and 'retreat' options of the SMP guidance. More traditional 'hard engineering' approaches for coastal erosion risk management are now being combined with habitat restoration schemes (Zhu et al., 2020). These 'Nature-based Solutions' extend the life of otherwise short-term (often 40 years) and reactive solutions, which cannot adapt to local and regional geomorphological change (Ballinger and Dodds, 2020). SMPs have become central to the government's approach in managing coastal erosion risk sustainably, and are being more readily adopted in land use planning policy (Ballinger and Dodds, 2020).

Coastal habitat creation schemes have expanded considerably in scope and size across Britain over the last decade, especially in the south where marshes have been declining in extent (Ladd et al., 2019). The five largest schemes were carried out since 2006, collectively adding 1,843 ha of intertidal habitat to the British coast (Fig. 4). Large-scale habitat creation schemes have been pioneered by charities including the Royal Society for the Protection of Birds, the National Trust, The Wildlife Trusts, and the Wetlands and Wildfowls Trust. Government agencies are now committing to similarly extensive habitat creation schemes. The ReMeMaRe project aims to expand national marsh extent by 15% over the next 25 years (REACH, 2020). Diverse methods are being developed, tested and applied for achieving habitat creation including managed realignment, regulated tidal exchange, beneficial use of dredged sediment, geo-texturing and planting, and allowing coastlines to evolve naturally (Vandenbruwaene et al., 2011; Macgregor and van Dijk, 2014; Spearman et al., 2014). Data on the monitoring, forecasting, and sharing of coastal processes has become widely available and freely accessible (e.g. CCO, 2020), and recognition is growing in the importance of inclusive governance for achieving positive saltmarsh management outcomes (Alexander et al., 2019; Ballinger and Dodds, 2020; McKinley et al., 2018).

4. Linking processes and management

In recognition of advances in scientific understanding, saltmarsh management policy in Britain has undergone a series of transformations throughout the 20th and 21st century (Fig. 5). For generations, the doctrine of coastal management was to advance policies of saltmarsh reclamation and the maintenance of embankments for land gain and economic growth. After 1949, concerns over environmental degradation led to key saltmarsh sites being protected by statute; first indirectly as bird habitat and later directly by habitats legislation stemming from the 1992 Rio Convention. The cessation of major marsh reclamation for schemes after 1978 marks another important turning point, ending a practice which was the main cause for marsh loss nationally. 1993 saw the acceptance of managed realignment as a means to recreate saltmarshes, and the adoption of coastal morphodynamics and adaptive management into coastal erosion risk policy. The convergence between saltmarsh restoration and coastal flooding policy enabled saltmarshes to form a key component of the coastal flood risk agenda, fuelled by the advancement of the ecosystem services concept and the recognition that saltmarshes offer an addition to traditional hard-defence strategies. Saltmarsh restoration project are now growing in scope and size.

However, issues remain with regard to incorporating saltmarsh morphodynamics and social factors into coastal erosion management plans.

Patterns of saltmarsh expansion and erosion are determined by the interaction between hydrology, sediment dynamics, biogeomorphology, and human intervention. The drivers of marsh change described above affect saltmarshes across a range of spatial and temporal scales (Fig. 6). For example, the impact of a single wave on the marsh edge is immediate and localised, whilst changes in the wave climate due to the North Atlantic Oscillation and 18.6 nodal tidal cycles alter expansion and erosion trends over entire regions for decades (Kraus et al., 1991).

Several studies have now identified that interaction between plant growth, water movement, and sediment flux can create cyclical dynamics of ecosystem establishment and collapse through short-term positive and long-term negative feedbacks (Corenblit et al., 2011; Viles, 2019). Cycles of saltmarsh expansion and erosion are apparent across scales (Fig. 6; blue shading). Recognising the scale at which natural and human processes influence coastal morphodynamics provides an opportunity to predict how marshes will likely evolve, and to identify where targeted intervention could lead to positive conservation management outcomes. Cowell and Thom (1994) define four spatio-temporal scale classifications for interpreting coastal morphodynamics (Fig. 6). Human impact tends to have the most immediate effect at the 'event' and 'engineering scales' (i.e. marsh to regional scales over days to centuries). This recognition has begun to shape modern coastal erosion risk management, perhaps most notably in the identification of littoral sediment cells as part of shoreline management planning within which engineering processes can be predicted (French et al., 2016).

Despite the progress made towards saltmarsh conservation and restoration throughout the mid 20th century, marsh extent has continued to decline in south and south-eastern Britain (Ladd et al., 2019). Saltmarsh creation schemes are, at present, of insufficient scale to compensate national losses (Boorman and Hazelden, 2017) and issues persist in translating advances in coastal process understanding into coastal erosion risk management strategy (Alexander et al., 2019; Ballinger and Dodds, 2020; McKinley et al., 2018). The following section uses the current understanding of saltmarsh dynamics and the present situation of management practice described above to highlight key insights to help shaping future marsh extent research and management.

4.1. Foreshore dynamics

Coastal development has severely reduced the capacity for marshes globally to migrate landward, truncating marshes between sea defences and rising sea levels (a phenomenon known as 'coastal squeeze'). Sea level rise is frequently cited as a key threat for saltmarsh survival in Great Britain (Adnitt et al., 2007; Cooper et al., 2001; Doody, 2004; Pontee, 2011; Wolters et al., 2005). There is, at present, considerable debate as to whether the interaction between plant sediment trapping, organic matter incorporation, and increased inundation can sustain vertical accretion rates relative to future sea level rise rates (Crosby et al., 2016; Horton et al., 2018; Kirwan et al., 2016; Schuerch et al., 2018; Spencer et al., 2016). Evidence indicates that vertical marsh accretion is keeping pace with sea level rise (Ladd et al., 2019) and will continue to do so provided the supply and transport of sediment is sufficient to compensate for future sea level rise rate increases (Kirwan, 2010).

Variation in sediment supply and hydrological forcing along seaward marsh edges are more likely to drive change in marsh extent than sea level rise over the decadal timescale. For example, the sandy marshes along Morecambe Bay, northwest England, tend to erode rapidly in response to storms and channel migrations, and recover quickly from sediments supplied by large tidal flats and delivered by large tidal ranges (Pringle, 1995). A net-positive sediment supply is driving marsh expansion across northern Britain (Ladd et al., 2019). In contrast, organic-rich saltmarshes along southern England erode more slowly under the microtidal environment, however limited sediment supply is likely insufficient to offset wave erosion (Ford et al., 2016; van der Wal and Pye, 2004).

The trend of marsh decline in southern England will likely continue if sediment supply is further reduced. Dredged material is routinely disposed of offshore, which potentially diminishes the sediment pool available to marshes and tidal flats. Beneficial use of dredged material has been shown to promote saltmarsh growth and will likely become an important process for maintaining marsh resilience where managed realignment is not possible (e.g. Spearman et al., 2014). Modification of structures such as groynes and dams which impede the movement of sediment, and reducing exposure of deposits to boat wakes, have all been shown to promote coastal sediment sustainability (Cooper and Pontee, 2006; Fagherazzi et al., 2013; Houser, 2010). Improvements in the regular monitoring of extent change and wave climate are also helping to identify where marshes are at risk of erosion (Burd, 1989; CCO, 2020; Haynes, 2016; MCCIP, 2018; Phelan et al., 2011). However, efforts should be directed towards better monitoring of sediment flux and tidal flat extent, which are crucial in determining the fate of saltmarshes (Marani et al., 2010). There is limited evidence that tidal flats are becoming narrower and steeper across Britain (Davidson et al., 1991; Pontee, 2011; Taylor et al., 2004) and could therefore undermine future habitat creation schemes. Recent developments of inexpensive methods to monitor habitat extent, hydrodynamics and sediment dynamics (Hu et al., 2020, 2015a; Laengner et al., 2019; Murray et al., 2019) should be capitalised upon by coastal managers in order to better forecasting future marsh change and the likely success of restoration schemes.

4.2. Scale-dependent and cyclical dynamics

Cycles of expansion and decline are an inherent phenomenon of saltmarsh evolution. Cycles occur across a range of spatial and temporal scales (Fig. 6) and thus a single observation of either marsh erosion or expansion at any point in time is unlikely to inform longer-term extent change trends. Long-term monitoring and numerical modelling can help disentangle decadal to centennial trends from seasonal cycles in marsh expansion and erosion (Laengner et al., 2019).

Cyclical marsh dynamics present a challenge to often rigid coastal management plans, which depend on stable marshes to maintain ecosystem services such as carbon sequestration, coastal flood protection, and aesthetic value. Indeed, coastal management plans often set targets to maintain marsh extent at defined levels, with little room for cyclical dynamics (e.g. BAP targets to preserve marsh extent at a 1992 baseline (Covey and Laffoley, 2002). The cost of maintaining saltmarshes in a given condition is high and often unsustainable. For example, the extent of marshes along the mainland Wadden sea have traditionally been maintained through a network of sedimentation fields and ditching, which frequently require the landscaping after storm events to preserve their ‘unnatural’ configuration (Esselink et al., 2017). Regular monitoring of marsh change can identify which parts of a coastline is likely to be dynamic (cycling frequently between marsh establishment and loss) or stable (remaining as a saltmarsh decades). ‘Geomorphic behaviour’ maps can be produced to aid coastal planning and develop a more realistic valuation of the decadal to centennial ecosystem service value of a given saltmarsh (Ladd, 2018).

4.3. Stakeholder engagement

To achieve conservation management goals, it is important that stakeholders recognise the value of saltmarshes for nature and people (McKinley et al., 2020). The ecosystem services framework has provided a language to capture and communicate the value of the natural environment (MEA, 2003), and has led to positive conservation outcomes around the world (Potschin et al., 2016). The slow adoption of ecosystem services into framing coastal governance issues is seen as a barrier to achieving conservation targets (McKinley et al., 2018). However, the ecosystem services concept does raise moral questions around whether ecological functions should be viewed as economic assets, which can be monetised and traded. Financialisation of natural processes could harm conservation goals (Silvertown, 2015). Extensions to the ecosystem services framework have been proposed, such as the ‘Nature’s Contribution to People’ (NCP) concept (Díaz et al., 2018). NCP emphasises the role of culture in defining links between people and nature rather than economic value. Cultural services are considered discrete from regulating and provisioning services under the current Ecosystem Services framework (Table 1). By incorporating indigenous people and local practitioner knowledge, worldviews, interests,

and values, NPC moves away from economic-based models. The growing body of work recognising intangible values of saltmarshes could underscore better ways of sustainably managing the coast (Rendón et al., 2019).

Habitat creation schemes, including managed and unmanaged realignment, restoration planting, and regulated tidal exchange, will likely play an important role in future saltmarsh restoration and coastal erosion risk management schemes across Britain. Current management plans indicate that 10% of the English and Welsh coastline will require realignment by 2030, and 15% by 2060 (Esteves and Williams, 2015). However, coastal realignment is controversial, and often perceived as "surrendering land to the sea" (Myatt et al., 2003). The success of habitat creation schemes is dependent on inclusive and equitable engagement with stakeholders, especially those most at risk of coastal flooding and who stand to benefit the most from marsh creation schemes (Alexander et al., 2019). Failure to engage effectively with stakeholders can lead to political deadlock and community activism which delay and even derail habitat creation schemes (Buser, 2020). When done correctly, early stakeholder engagement can streamline even the largest of restoration projects (Scott et al., 2015). Engagement is needed to communicate the multi-faceted benefits that saltmarsh creation provides, in order to move away from traditional reliance on 'hard' flood defences (Ballinger and Dodds, 2020).

4.4. Upscaling processes and management

Use of Great Britain as a case study offers insights for coastal managers worldwide on the processes and management of saltmarshes. When considering natural marsh establishment and erosion dynamics, pioneer plants share common traits in their capacity to alter sediment and water flows. The plants engineer the tidal landscape and initiate marsh establishment (Corenblit et al., 2015). The pioneer genera described in this review are distributed globally either naturally or by invasion (Riddin et al., 2016; Li et al., 2020; Maebara et al., 2020), or are functionally similar to other pioneer species (Viswanathan et al., 2020). Combining restoration practices with knowledge on the positive interactions common between pioneer species could increase the success of marsh restoration projects around the world (Silliman et al., 2015). For large-scale controls on marsh dynamics, sediment flux to the coast and exposure to hydrological forcing are key factors in determining how marshes evolve (Argow et al., 2011; Kirwan and Megonigal, 2013; Balke et al., 2016; Leonardi et al., 2016). Sea level rise and climate change are already universally recognised as common threats to marsh resilience (McKinley et al., 2020), requiring common agreements to curb CO_2 emissions and prevent the destruction of wetlands.

Nevertheless, large geographical differences also exist in the drivers of marsh change. For example, land reclamation has been a leading cause of marsh loss across Europe. Embankments prevent the natural landward migration of marshes under sea level rise. In contrast, large stretches of the Atlantic U.S. coast remain open, and marshes are expected to replace terrestrial vegetation as sea levels rise (Kirwan and Gedan, 2019). Large-scale vegetation die-off from consumer-driven fronts and drought conditions have only been reported in U.S. marshes (Hughes et al., 2012; Silliman et al., 2013; Marsh et al., 2016), underscoring differences in the role of climate and plant-animal interactions around the globe. Differences in the perceptions of the value of marshes also differ globally. In Europe and Asia, marshes are generally regarded as important for agriculture. In the U.S., marshes are more highly valued for wild resource gathering. In Australia, marshes are valued as refugia for native fauna and flora, with conservation efforts directed at preventing invasive species from establishing (McKinley et al., 2020). Such differences in attitude reflect differences in culture. Recognising regional variation in the processes and perceptions of marsh change can streamline the science-policy-practice journey, enabling effective coastal management (Jarvis et al., 2015).

5. Summary

Saltmarsh management in Great Britain has changed fundamentally over the last 70 years. Saltmarshes are no longer being exploited for land claim, and are instead being protected and restored at a national level. Reasons for the shift in management thinking are linked to societal concerns about environmental degradation after WWII (especially for bird habitat in the context of saltmarshes), and the need for

preserving the numerous ecosystem services that saltmarshes sustain. Marshes now form part of the national coastal flood risk management strategy, and ambitious targets have been set to reverse the national declines in marsh extent by the end of the decade. Scientific understanding of the patterns and processes of marsh change have underpinned the transformation in coastal management policy. Effective monitoring of both saltmarsh extent change and the key processes responsible for marsh loss have enabled a strategic approach for deciding where to focus conservation efforts. Characterisation of coastal units into sediment cells (under the Shoreline Management Planning framework) is enabling managers to predict how coastlines will likely evolve over decadal and centennial timescales, presenting an opportunity for the large-scale roll-out of saltmarsh habitat creation schemes. Important gaps need to be addressed if salt marshes are to be preserved in the long-term: (i) tidal flat extent change and sediment supply monitoring is needed in order to anticipate where marshes are vulnerable to net marsh loss under sea level rise; (ii) forecasting of the long-term ecosystem service value of saltmarshes should incorporate cyclical marsh dynamics which operate over varying spatio-temporal scales, and; (iii) greater effort needs to be made to integrate stakeholders in management decisions, so that differences in the processes and perceptions of marsh change are recognised, allowing the rapid roll-out of effective protection and restoration schemes.

Declaration of Competing Interest

No conflict of interest.

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Table 1. Ecosystem services provided by saltmarshes. Adapted from McKinley et al. (2018).

Regulating	Provisioning	Cultural
Climate regulation (carbon sequestration)	Domesticated livestock	Recreation and tourism
Hazard regulation (coastal flood defence and erosion prevention)	Wild fish and food	Aesthetic value and inspiration
Waste breakdown and detoxification	Wild game	Religious or spiritual, cultural, natural heritage, and media
Soil quality regulation		Education and ecological knowledge
Pollination		Enfranchisement and neighbourhood development
Disease and pest regulation		Physical and mental health, security, and freedom
Wild species diversity		

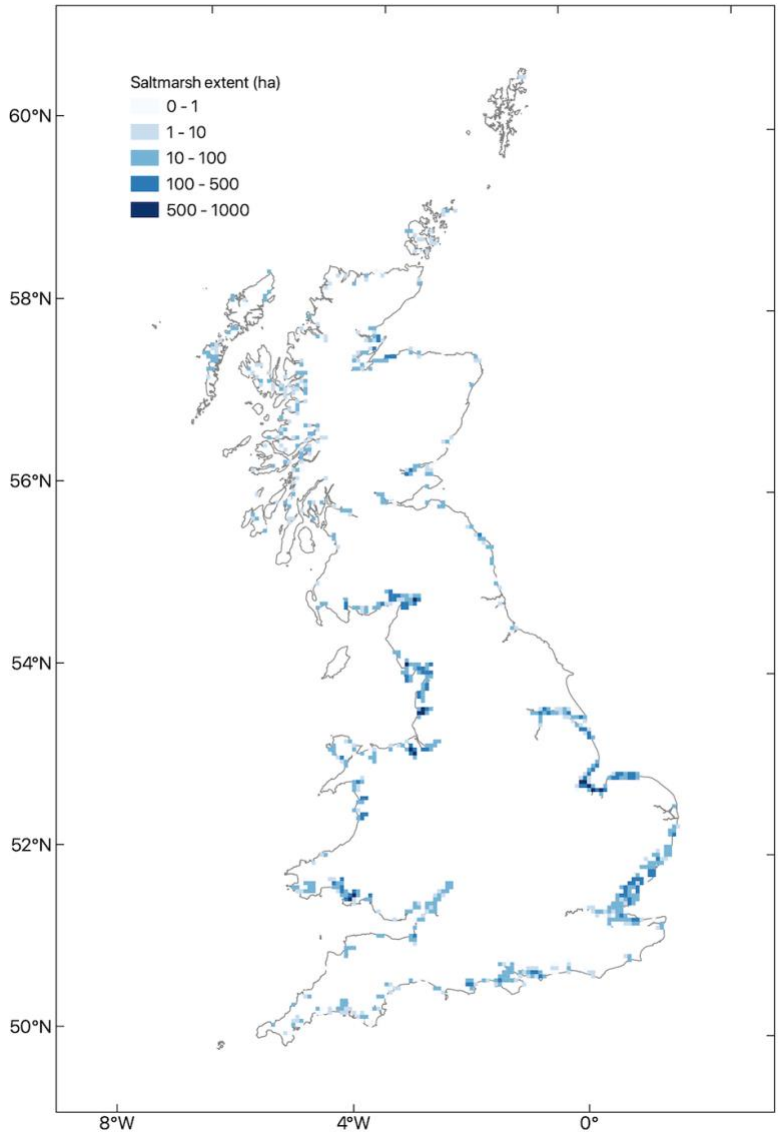

Fig. 1. Saltmarsh extent across Great Britain within 5 km^2 cells. Adapted from Phelan et al. (2011) and Haynes (2016).

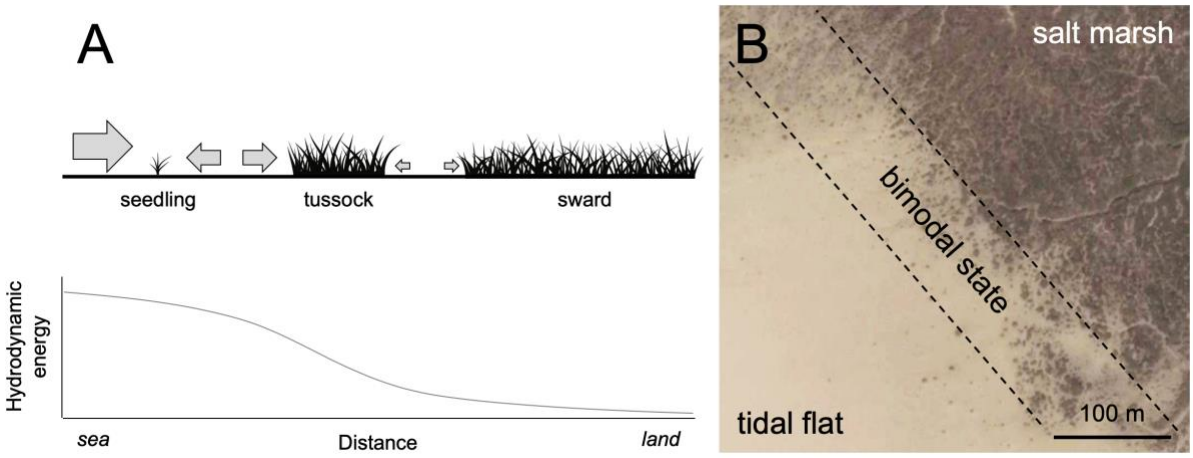

Fig. 2 A Representation of pioneer plant development on the tidal flat. Hydrodynamic energy causes scour at the edges of plant patches (grey arrows) and decreases with distance along the shore due to friction between the incoming tide and plant structure. Decreased hydrodynamic energy allows other plants to colonise and tussocks to form closed swards (adapted from Friess et al., 2012). **B** Stable tidal flats and saltmarshes establish at the bottom

and top of the shore respectively, whereas the interface between both persists in a bimodal state, cycling between vegetated and bare ground.

Fig. 3. Photograph of new vegetation establishment in front of a relic saltmarsh cliff from Rhymney Wharf, Severn Estuary, south Wales.

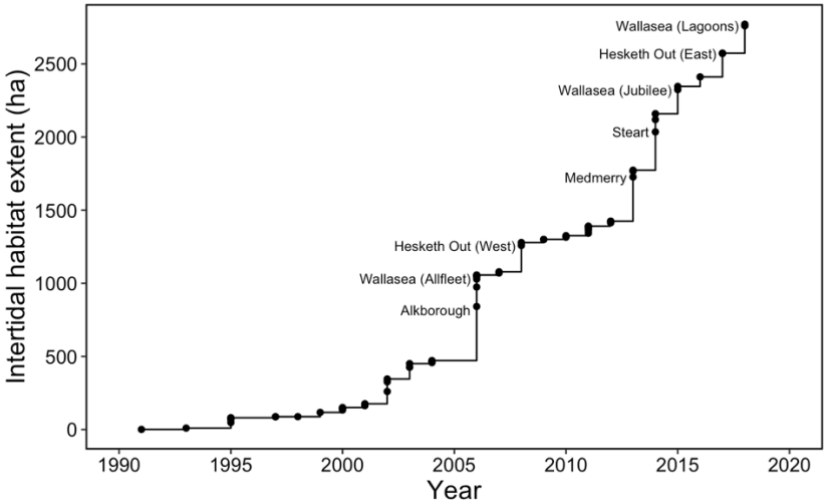

Fig. 4. Cumulative addition of intertidal habitat created through managed and unmanaged realignment, regulated tidal exchange, and beneficial use of dredged sediment across Great Britain. The five largest schemes are identified. Values do not differentiate between saltmarsh, lagoon, mudflat, reedbed, and transitional grassland habitat. Adapted from ABPmer (2020).

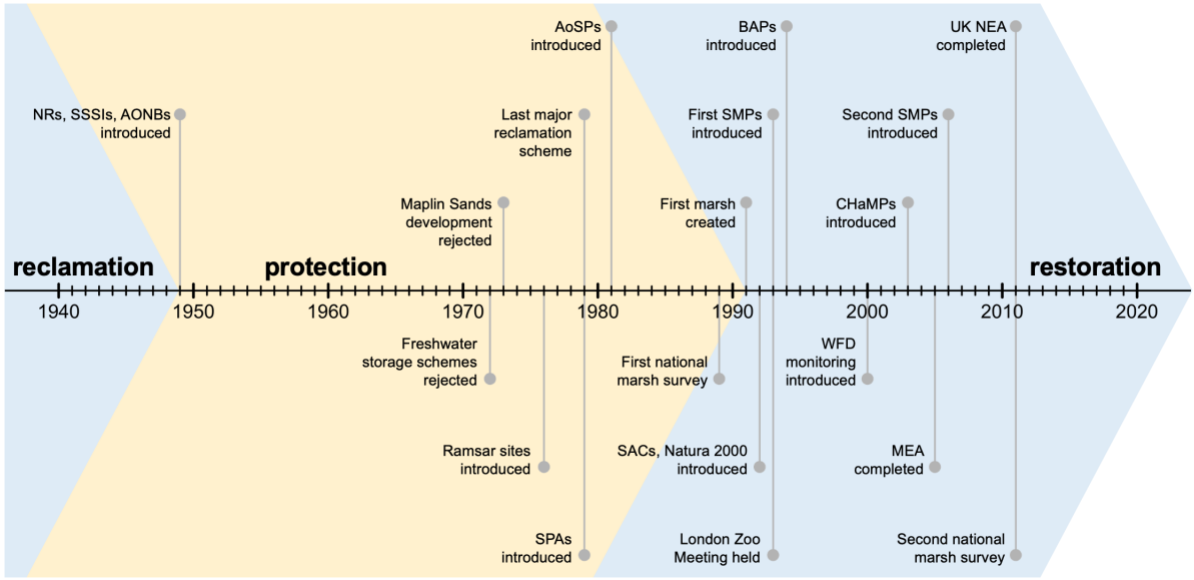

Fig. 5. Timeline representing key events which have shaped saltmarsh conservation practice in Great Britain. Events can be grouped into three overarching categories: reclamation, protection, and restoration (NRs: Nature Reserves; SSSIs: Sites of Special Scientific Interest; AONBs: Areas of Outstanding Natural Beauty; SPAs: Special Protection Areas; AoSPs: Areas of Special Protection; SACs: Special Areas of Conservation; SMPs: Shoreline Management Plans; BAPs: Biodiversity Action Plans; WFD: Water Framework Directive; CHaMPs: Coastal Habitat Management Plans; MEA: Millennium Ecosystem Assessment; NEA: National Ecosystem Assessment).

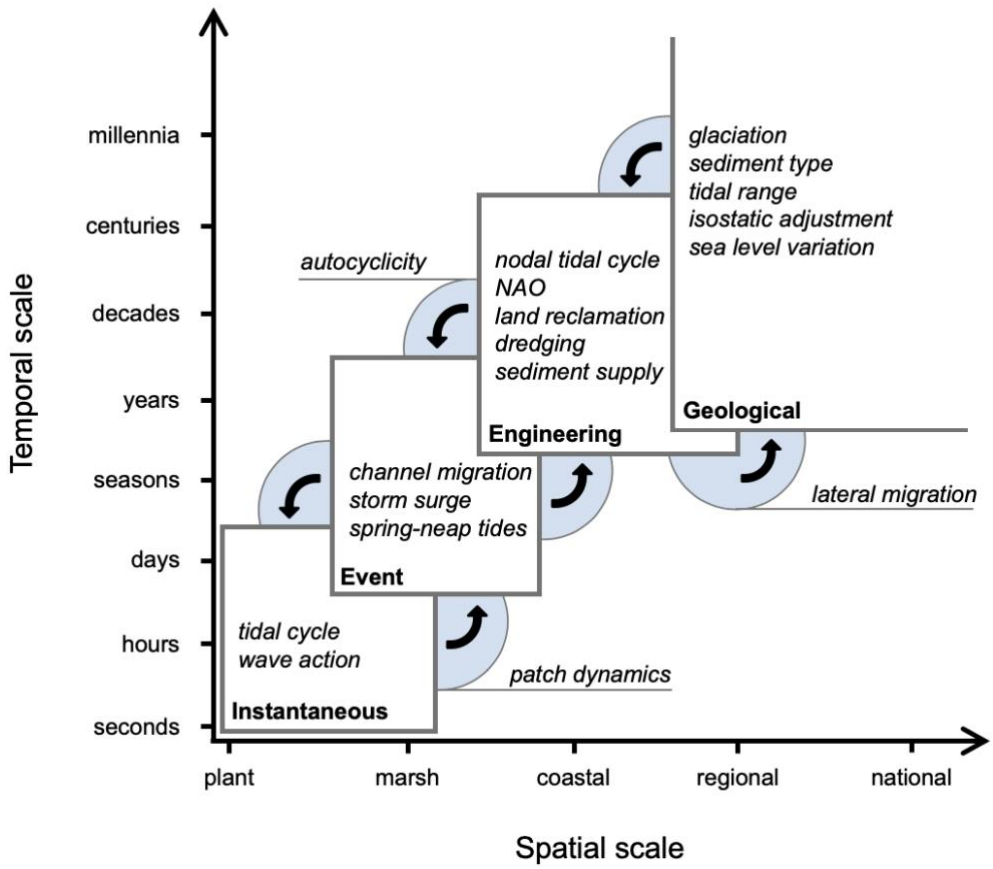

Fig. 6. Key drivers responsible for saltmarsh expansion and erosion in Great Britain which operate over different spatial and temporal scales. Drivers are clustered into four discrete scales using Cowell and Thom's (1994) classification scheme. Interaction between physical and biotic drivers produce three characteristic cyclical marsh expansion and erosion phenomena across scales (blue shading and arrows): 'Patch dynamics' represents the

cyclical process between pioneer plant patch formation, which induces scouring around the plant patch causing erosion (van Wesenbeeck et al., 2008); 'Autocyclicity' represents the gradual gain in marsh elevation, which exposes of the marsh edge to lateral erosion (Bouma et al., 2016) and the cyclical import and export into estuarine environments during flood- and ebb-dominant phases of tidal asymmetry (Brown and Davies, 2010); 'Lateral migration' represents the landward or seaward migration of marshes with rise and fall of mean sea level and resulting sediment transport (Fagherazzi, 2013). Adapted from Kraus et al. (1991), Cowell and Thom (1994) and Corenblit et al. (2011) (NAO: North Atlantic Oscillation).