Impacts of climate change on coastal habitats, relevant to the coastal and marine environment around the UK

A. Burden [1], C. Smeaton [2], S. Angus [3], A. Garbutt [1], L. Jones [1], H.D. Lewis [4] and S.M. Rees [5]

[1] *Centre for Ecology & Hydrology, Environment Centre Wales, Deiniol Road, Bangor, Gwynedd, LL57 2UW, UK*
[2] *School of Geography & Sustainable Development, Irvine Building, University of St Andrews, North Street, St Andrews, KY16 9AL, UK*
[3] *Scottish Natural Heritage, Great Glen House, Leachkin Road, Inverness, IV3 8NW, UK*
[4] *Natural Resources Wales, Tŷ Cambria, 29 Newport Road, Cardiff, CF24 0TP, UK*
[5] *Coastal & Woodland Habitats Team, Natural England, Eastbrook, Shaftesbury Road, Cambridge, CB2 8DR, UK*

EXECUTIVE SUMMARY

Coastal habitats are at risk from both direct (temperature, rainfall), and indirect (sea-level rise, coastal erosion) impacts due to a changing climate. Beyond the environmental impacts and ensuing habitat loss, the changing climate will have a significant societal impact to coastal communities ranging from health to livelihoods, as well as the loss of important ecosystem services such as coastal defence – particularly relevant with predicted increase in storminess.

Vegetated coastal ecosystems sequester carbon – another 'ecosystem service' that could be disrupted due to climate change. There has been considerable recent attention to the potential role these habitats could play in climate mitigation, and also in transferring carbon across the land–sea interface. To understand the relative importance of these habitats within the global carbon cycle, coastal habitats need to be accounted for in national greenhouse gas inventories, and a true multidisciplinary catchment-to-coast approach to research is required.

Management options exist that can reduce the immediate impacts of climate change, such as managed realignment and sediment recharge. Fixed landward coastal defences are becoming unsustainable and creating 'coastal squeeze', highlighting the need to work with natural processes to recreate more-natural shorelines where possible.

Citation: Burden, A., Smeaton, C., Angus, S., Garbutt, A., Jones, L., Lewis H.D.and Rees. S.M. (2020) Impacts of climate change on coastal habitats relevant to the coastal and marine environment around the UK. *MCCIP Science Review 2020*, 228–255.

doi: 10.14465/2020.arc11.chb

Submitted: 04 2019
Published online: 15th January 2020.

1. INTRODUCTION

The coastline of the UK consists of many natural and semi-natural habitats, as well as urban areas. This report focusses on those habitats only found at the coast, which are not considered 'marine'. These are:

- Saltmarsh
- Machair
- Sand dunes
- Shingle
- Maritime cliff and slope.

Seagrass beds are at risk from the same climate pressure as these coastal habitats, but are not included in this report as they are considered shallow subtidal habitats and are discussed in an accompanying Report Card (q.v., Moore and Smale, 2020).

Coastal habitats in the UK provide many ecosystem services, such as flood defence, climate regulation, and tourism opportunities, which are all beneficial to society and the economy. They represent a zone of transition between the terrestrial and marine domain and are in a constant state of flux. Coastal processes are dependent on tides, waves, winds, flora, fauna, and sediment processes; they susceptible to and altered by climatic changes, whilst also vulnerable to, and often negatively affected by, human activities. In part, the exact effect of climate change on these habitats is unpredictable. However, broad predictions have been made to the pressures that are likely to cause change in the coastal zone. This Report Card focusses on the impact of climate change on coastal habitats of the UK, and presents the key challenges and emerging issues that need to be addressed.

Coastal climate change

Climate change is likely to have a severe impact on the UK coast by 2100. The UK's coastline is under multiple natural and anthropogenic pressures from

- local and global climate change;
- sea-level rise;
- changes in the frequency and intensity of storms;
- increases in precipitation;
- warmer oceans;
- pollution; and
- increases in natural hazards.

In turn, these pressures can lead to changes in coastal processes, habitat loss and degradation (which itself is a pressure on the coastal environment), and changing species distribution patterns due to changes in the climate envelope.

The double impact from habitat loss and from altered climate means coastal habitats are more sensitive to climate change than most terrestrial ecosystems.

The total rise in sea-level around the UK coast may exceed one metre by 2100 (UKCP, 2018). The frequency of intense storm events is expected to increase and lead to more coastal flooding. Temperatures are expected to rise, particularly in the south and east of the UK. Winter precipitation is likely to increase markedly on the northern and western UK coastline. Coastal erosion is also expected to increase, partly due to sea-level rise. Low-lying and soft-sediment coasts in the east of England will be most vulnerable as they are most easily eroded. The most-exposed locations and estuaries may be particularly vulnerable.

Climate and coastal-change impacts will be felt along the whole of the UK coast. Thirty million people live in urban coastal areas in the UK, and these threats will be felt particularly keenly in communities that rely on the coastal area for their economic and social wellbeing. Confronting existing challenges that affect man-made infrastructure and coastal ecosystems, such as shoreline erosion, coastal flooding, and water pollution, is a concern in many areas.

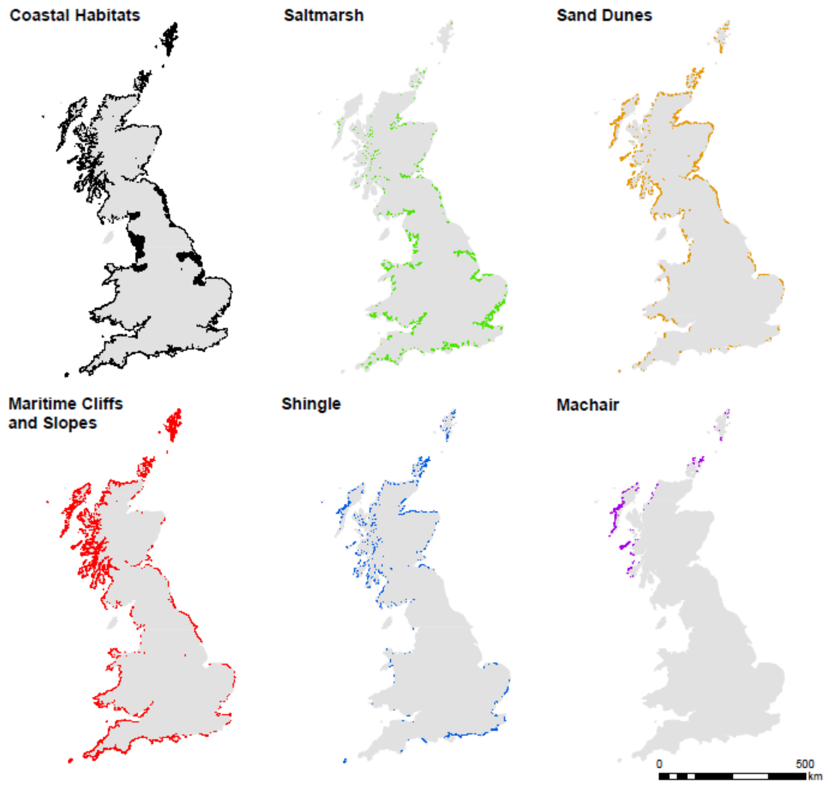


Figure 1: Location of coastal habitats in Great Britain. Habitat data for each region accessed from: The Habitat Map of Scotland, Priority Habitat Inventory (England), and Phase 1 Habitat Survey (Wales).

Table 1: Current estimated area of coastal margin habitats in the UK (hectares). Cliff extent measured in km length. (From Beaumont et al.*, 2014 Jones* et al.*, 2011; Haynes, 2016; Dargie and Duncan, 1999; Murdock* et al.*, 2014.)*

	Units	UK	Scotland	England	Wales
Saltmarsh	ha	44,102	5,840	32,462	5,800
Sand Dune	ha	58,298	38,300	11,897	8101
Machair	ha	11,680	11,680		
Shingle	ha	5802	1120	5023	109
Maritime Cliffs and Slopes	km	4060	1084	2455	522

2. WHAT IS ALREADY HAPPENING?

2.1 Saltmarsh

2.1.1 Description

Saltmarshes generally occur between mean high-water spring tides and mean high-water neap tides at temperate latitudes. The development of saltmarsh is largely controlled by physiography, where fine-grained sediments accumulate in relatively low-energy environments where wave action is limited. Consequently, salt-tolerant vegetation develops where there is an accumulation of mud in estuaries, inlets, behind barrier islands or spits, and occasionally via marine inundation of low-lying ground. Specialist ‘perched saltmarsh’ can also be found behind rocky outcrops or wave-cut platforms. Four physical factors – sediment supply, tidal regime, wind-wave climate, and the movement of relative sea-level – primarily govern the character and dynamic behaviour of saltmarshes (Boorman, 2003). The composition of saltmarsh flora and fauna is determined by complex interactions between frequency of tidal inundation, salinity, suspended sediment content and particle size, slope, and biotic factors (i.e. herbivory). In general, total species richness increases with elevation leading to a characteristic zonation of the vegetation (Doody, 2008). Transitions to mudflat occur at the seaward limit, while in the upper elevations of saltmarshes there may be further transitions to brackish or freshwater marsh, dune vegetation, or vegetation overlying shingle structures. The halophytic flora is relatively species poor, dominated by perennial grasses, rushes and dwarf shrubs. Annual species are poorly represented and restricted to the upper (terrestrial) and lower (mudflat) transition zones. Saltmarsh invertebrates are dominated by the high abundance of a few species and a high degree of adaptation to cope with the intertidal environment. Saltmarshes are important habitats for breeding, feeding, and roosting birds, many of them migratory. More recently there has been a growing recognition of the role coastal habitats play in sequestering and storing carbon (C) (Duarte *et al*., 2005; Nellemann *et al*., 2009). Globally it has been shown that saltmarsh can trap several orders of magnitude more C per area unit than the world’s forests (McLeod *et al*., 2011). Through the sequestration and capture of C these environments have the potential as a climate buffer preventing CO_2 from reaching the atmosphere (McLeod *et al*., 2011). This ecosystem service further adds value to these habitats and further increase the need to protect and preserve these environments.

2.1.2 Extent and regional pattern trends

Saltmarsh is widely distributed around the UK. The most extensive areas occur along estuaries in the counties of Hampshire, north Kent, Essex, Norfolk, Lincolnshire, and Lancashire (May and Hansom, 2003). The extent of saltmarsh habitat in the UK is estimated to be between 40,000 and 45,000 ha (Burd, 1989; Jones *et al*., 2011) with the five largest sites (Wash, Inner Solway, Morecambe Bay, Burry estuary, Dee estuary) accounting for one third of the UK total (Burd, 1989). The current extent of saltmarsh habitat is considerably less than in the past as, historically, large areas of saltmarsh were drained and cut off from the tide by sea defences to increase the area that could be used for agriculture or development (Morris *et al*., 2004). More recently, saltmarsh habitat has been claimed for activities such as port development, and sea-level rise also poses a threat through coastal squeeze – where the natural landward migration of saltmarshes in response to sea-level rise is restricted by sea defences (Blackwell *et al*., 2004; Adaptation Sub-Committee, 2013 – further discussed in Section 5.1.3). Losses also occur due to erosion, which takes a number of different forms, most commonly including the landward retreat of the seaward edge, either as a cliff or steep 'ramp', or an expanding internal dissection of the marsh by the widening creeks. Erosion predominantly affects lower marsh communities which are more vulnerable to wave action, although mid- and high-saltmarsh is susceptible to internal erosion through creek expansion.

There are many estimates of the extent of saltmarsh habitat loss. French (1997) estimated that globally, 25% of intertidal estuarine habitat has been lost due to land reclamation, and Barbier (2011) estimated 50% of the world's saltmarshes have been degraded or lost mainly due to habitat conversion (or destruction). On an annual basis the loss rate has been estimated at between 1% and 2% (Nottage and Robertson, 2005; Duarte *et al*., 2008). However, differences in methodologies between surveys can make it difficult to verify change over time. A recent study by Horton *et al. (*2018) showed a greater than 80% probability of saltmarsh retreat for the whole of Great Britain by 2100. The current major losses in saltmarsh extent in the UK are in the south-east of England. Between 1973 and 1998, over 1000 ha were lost (Cooper *et al*., 2001). In the Solent the total saltmarsh resource declined from 1700 ha to 1080 ha between the 1970s and 2001 (Baily and Pearson, 2007) with further losses in Poole Harbour (Born, 2005).

Restoration of saltmarsh, to mitigate historical and ongoing losses of saltmarsh habitat, has been gathering momentum since the early 1990s, mostly via managed realignment – the landward realignment of coastal defences and subsequent tidal inundation of reclaimed land. A total area of 2647 ha has been created since 1991 (to 2017: ABPmer, 2018) and there are long-term plans in England to realign 10% of the coastline by 2030, rising to 15% by 2060 (Adaptation Sub-Committee, 2013). There have also been many accidental breachings of sea walls during storm events, where repair has not been economically viable, however there is no central record of the areas involved. The largest saltmarsh restoration project in the UK is the RSPB Wallasea Island Wild Coast project on the Essex coast, which aims to transform nearly 800 ha of farmland back to wetland habitat, approximately 400 years after reclamation by the end of 2018, 321 ha of which will be saltmarsh habitat (ABPmer, 2018). Restoration of fringe saltmarsh is also starting to be considered as a natural solution to flood protection and wave attenuation along estuarine foreshores.

2.1.3 Processes (both natural and anthropogenic) likely to be affected by climate change

The primary effects of climate change on saltmarshes are sea-level rise and changes to storminess, temperature, and precipitation. These will all likely impact the areal extent, predominantly by interrupting sediment transport pathways (MCCIP, 2018). Land-use and inland catchment-management changes in freshwater systems (as well as changes to precipitation patterns) also affect flows and sediment supply to the coastal zone from river networks. Changes in seasonal extremes, increase in storminess, etc. both at the coast itself, and inland, will also affect timing, quantity and potentially source of sediment.

Sea-level rise will affect saltmarshes in different ways depending on local context. Saltmarshes are able to keep pace with sea-level rise as long as there is an adequate sediment supply to maintain vertical accretion. Therefore, marshes with both higher tidal ranges and suspended sediment loads will be more resilient. However, the lateral extent of marsh could be reduced as deeper water and larger waves cause erosion to the seaward edge, which could also be exacerbated by an increase in storminess. Landward migration of saltmarsh could compensate for these losses, but only in places without hard sea defences.

With changes in temperature, species composition is likely to change as climatic envelopes shift. For example, warmer temperatures could favour the spread of *Spartina anglica*, an invasive species which out-competes the native cordgrass (Loebl, 2006) producing a monoculture. As plant diversity has been linked to soil stability (Ford *et al*., 2016), a change such as this could also lead to increased erosion and loss of saltmarsh.

2.2 Machair

2.2.1 Description

Machair is an extreme form of calcareous dune grassland, restricted globally to the north and west of Scotland and the west of Ireland. The definition of the habitat is complex, involving coastal topography, vegetation, shell components, climate, land use, herbivory, and water table (Ritchie, 1976; Angus, 2004, 2006). The Annex I machair habitat or ‘machair grassland’ invariably occurs within a wider functional ‘machair system’ comprising beach, dune, machair grassland, marsh, and freshwater loch, with transitional ‘blackland’ as blown sand decreases in influence towards the inland, acid peatlands. Where the lochs are particularly low-lying, they can be flooded by seawater at high tide creating saline lagoons. Marshes within the machair system subject to marine influence are saltmarshes. Though machair has a high biodiversity, the habitat has developed in tandem with human settlement and anthropogenic influences are an inherent aspect of the habitat and its value. Several of the main machair areas have been extensively altered by drainage such as Sanday in Orkney (Rennie, 2006) and South Uist and Benbecula (Angus, 2018).

2.2.2 Extent and regional pattern trends

Identifying the extent of machair has involved first identifying machair systems then, using the national Sand Dune Vegetation Survey of Scotland (Dargie, 1999), allocating individual polygons to be drawn up for machair, first using an automated, modified version of the definition by Angus (2006) then by interrogation of individual polygons to refine this output. Though a polygon map now exists for the habitat, it is no more than a snapshot of aggregated surveys spanning the period 1985–1998, and such a map can only be indicative of the distribution of a highly dynamic habitat. The extent will vary in space and time in response to natural variations in climate and also land use. Using this method the total area of Annex I machair grassland in Scotland is 11,680 ha as measured in 2018.

2.2.3 Processes (both natural and anthropogenic) likely to be affected by climate change

Machair is likely to be affected by climate through changes in water management, precipitation, relative sea-level rise, and increased storminess. The issues primarily relate to water management are keeping seawater from overtopping the dune ridge, keeping seawater from contaminating the machair water table, and finally ensuring that precipitation can be discharged to the sea.

The coastlines of these islands are all low-lying, in some cases having an interior up to 1 m below the level of MHWS. Therefore, if the machair is overtopped by rising sea level, both the machair and a significant portion of the terrestrial environment could be displaced by other habitat, such as saltmarsh or sandflat. There are places where sea already enters the interior, notably via the estuary of the Howmore River in South Uist and via saline lagoons, some of which have onward connections to other lochs. Saline flooding is known to impact the water table by increasing salinity (Angus and Rennie, 2014), but the geographical extent of influence on the water table is unknown.

Serious storms, such as that of January 2005, have the capacity to overtop the dune ridge and flood the interior with sea water (Angus and Rennie, 2014). The kelp beds west of the Uists are 7 km wide and believed to have a significant attenuating effect on wave energy (Angus and Rennie, 2014). The existing severity and frequency of storms are thus likely to have an increasing impact as sea-level rise progresses and wave-energy increases. Similar kelp beds are known to exist off Tiree and parts of Orkney, but less is known about their role in coastal processes.

Much of the modern extent of machair in South Uist, Benbecula and, to a lesser extent, Tiree and western North Uist, was submerged beneath inland lochs until a drainage programme began in the 18th century. The drains discharge on the foreshore at low tide. However, sea-level has risen by as much as 279 mm since the drains were built, reducing not only the 'head' between inland waters and the sea, but reducing the period of the tidal cycle when such discharge is possible. Some of the drains are valved but others are not, and the latter are known to allow backflow of sea water at high tide. With winter precipitation likely to increase, perhaps significantly (Kay *et al.*, 2011), this reduced discharge capacity could prove problematic (Angus, 2018).

Comparison of two sets of precipitation figures covering 1961–1980 and 1981–2010 reveals a change in the seasonal distribution of rainfall, though the annual totals are similar. There are increases in spring and autumn, corresponding with ploughing and harvest respectively, with reduced rainfall during the summer growing period, which will be particularly problematic on the drier machairs (Angus, 2018).

2.3 Sand dunes

2.3.1 Description

Coastal sand dunes are formed from sand (0.2–2mm grain size) blown inland from the beach, which is colonised by vegetation (Packham and Willis, 1997). Typically, phases of mobility and natural coastal dynamics lead to a sequence of dune ridges, which increase in stability and age further away from the sea. Ridges are often separated by low-lying flat areas called 'swales'. Where these low-lying areas are in contact with the water table, dune wetlands form. The main vegetation types are dry dune grassland and dune slacks – a seasonal wetland, with dune heath on some acidic sites. Scrub and natural dune woodland are relatively sparse in the UK, although large areas of dune have been artificially forested with pine trees. Sand dunes support a high diversity of plant, insect and animal species, many of which are rare. They are particularly important for specialists dependent on bare sand or early successional habitats, including the natterjack toad, which requires early successional dune slacks for breeding, and the sand lizard, which requires open bare areas for basking and breeding burrows. Dune slacks have a high botanical diversity. Sand dunes are also important for geomorphological conservation. Many UK sites are notified as SSSIs/ASSIs for these interests and several are of international importance for active coastal processes.

2.3.2 Extent and regional pattern trends

Dune systems in the UK and Europe have shown large changes in the last few hundred years (Provoost *et al*., 2011), including habitat loss and changes in habitat quality (Jones *et al*., 2011). New evidence in the last five years from a re-survey of dune wetlands in England suggests that dune slacks are drying out. Overall there has been a 30% loss in the extent of dune slacks at the largest protected sites in England over the period 1990–2012. The remaining dune slack habitat has also shown a shift in species composition and in habitat extent from wetter to drier plant communities (Stratford *et al*., 2014). There is some regional differentiation to the patterns, with the greatest drying occurring in the south and west, whereas sites in the north and east appear to be less affected. Over a similar time period, a separate resurvey of sites in Scotland showed that dune vegetation seemed to be largely unaffected by climate change (Pakeman *et al*., 2015), re-inforcing the apparent spatial pattern of change across the UK. In Scotland, the observed changes were due primarily to succession (or management) rather than climate, and there were no apparent changes in the range of dune species. In the England resurvey, there was a concurrent increase in eutrophication of the dune slack vegetation, most likely driven by 'internal eutrophication', i.e. by increases in mineralisation rates and nitrogen turnover as a result of drying out of the wetlands (Stratford *et al*., 2014).

2.3.3 Processes (both natural and anthropogenic) likely to be affected by climate change

Climate change can affect coastal dunes in a number of ways. These include direct loss of habitat due to coastal erosion coupled with accelerated sea-level rise, and changes in the climate envelopes of dune-plant- and animal-species. These also include indirect effects through changes in underlying ecosystem processes such as soil mineralisation rates, plant productivity, soil moisture deficit, evapotranspiration, and the recharge to groundwater. These processes will affect competition between species, mediated via plant growth, they will affect soil development, and via influences on groundwater systems will affect the dune wetland communities.

Specific processes sensitive to climate change include the rate and direction of sand movement, which is governed by the wind climate, encompassing spatial and temporal variation in wind direction and wind speeds as well as rainfall. A short period of high winds during dry conditions can move more sand than longer durations of high winds during wet conditions. Over longer timescales, the amount and type of vegetation cover will also affect sand movement and sand capture by vegetation. Different plant species trap sand in different ways, resulting in different types of dune formation (Zarnetske *et al*., 2018).

The seasonal pattern of rainfall can affect both dry-dune- and wet-dune-slack vegetation. Soil moisture deficit and summer drought is likely to affect dry-dune vegetation. Studies in The Netherlands suggest a likely increase in the cover of drought-adapted mosses and lichens under climate change due to summer drought (Bartholomeus *et al*., 2012; Witte *et al*., 2012). Such Dutch sites are not too different from some of the UK east and south coast dunes. Greater winter rainfall appears to facilitate growth of scrub species like sea-buckthorn, while summer droughts will affect the species composition of dune slacks through lowering of the water table (Doody, 2013). In dune slacks, small shifts in water table can result in species change. An experimental study showed that a shift of 10 cm in the water-table regime resulted in competitive shifts in two species (Rhymes *et al*., 2018), while field survey evidence suggests 20 cm shifts in a four-year average water-table regime differentiate the main dune slack communities, and only 40 cm difference in regime separates the wettest from the driest dune-slack vegetation type (Curreli *et al*., 2013).

2.4 Shingle

2.4.1 Description

Shingle (also known as gravel/coarse clastic sediment) consist of sediments 2–200 mm in diameter. Sediment is supplied from offshore glacial deposits and cliff erosion, with longshore drift taking sediment into beaches, bays, spits and nesses. Shingle beaches occur in high wave-energy environments which sorts the particle size and influences the longer-term development of vegetation. Under moderate storm-wave energy, shingle is pushed up the beach, but in major storms much larger overtopping events can occur. The development of vegetation is therefore strongly linked to past and present processes.

Fringing beaches have ephemeral seasonal vegetation from seeds of mostly annual species deposited with tidal debris. Some of these communities are rare with a number of species restricted to the habitat. Above the reach of waves, the more extensive shingle structures, such as Dungeness, have more permanent perennial vegetation. The habitat type is complex, with several different elements reflecting surface topography, sediment-size variation, available organic matter, moisture conditions, and geographical position. Vegetation patterns are influenced by the underlying ridge structure that developed as the sediment was deposited by storm waves, resulting in a linear pattern of higher ridges and hollows. Vegetation colonises the ridges in a distinct, usually linear pattern following the ridge lines. Studies on key sites, including Dungeness (Ferry *et al.*, 1990), show this strong relationship between the geomorphology, topography and ecology. As a site evolves, pioneer plant communities establish on newly formed ridges to seaward, as well as developing into more diverse plant communities landwards. Vegetation communities are described in Rodwell (2000) and in more detail in Sneddon and Randall (1993).

Shingle structures can support breeding gulls, waders and terns. Diverse invertebrate communities are also found on coastal shingle, with some species restricted to shingle habitats (Shardlow, 2001). Specialised invertebrates occur on both vegetated and bare shingle, with some living deep in the matrix where humidity and temperature enable their survival (Low, 2005). Whilst many plants have adaptations to allow seed dispersal by the sea, for example buoyant seeds such as sea kale (*Crambe maritima*) (Sanyal and Decocq, 2015), there is a risk that the fragmented nature of shingle systems may reduce ability of species to migrate in response to climate change impacts.

2.4.2 Extent and regional pattern trends

This is a globally restricted coastal landform, with important locations in the UK. Ratcliffe (1977) estimated 30% of the English and Welsh coasts support fringing shingle beaches. May and Hansom (2003) suggest that 1040 km of the British coastline is formed of shingle structures: when added to those underlying sand beaches, this increases to 2900 km. The largest areas are in Scotland (Spey Bay/Culbin Bar), and in the north-west (Cumbria), south (Dorset to Kent) and south-east (Suffolk and Norfolk) of England. However, there are often smaller areas that provide important plant and animal habitats, such as the 'cheniers' associated with saltmarshes in the south and east of England and 'pocket beaches' or shingle barriers across bays which influence tidal inundation inland. Each location, no matter how small, is important because of the scarcity of this coastal landform. The general regional pattern of distribution has not changed since the previous MCCIP report cards, (Rees *et al*., 2010; Jones *et al*., 2013). The Welsh coast has a number of small sites. This habitat is poorly represented in Northern Ireland, where the key site is Ballyquintin in County Down. A small amount of shingle is present in the Isle of Man (F. Gell, pers. comm.)

Recent SNH work (Murdock *et al*., 2011, 2014) updated the extent of Scottish shingle habitat to 1120 ha, slightly more than previously estimated. In 2012, field validation took place of 1083 ha and the data provides an important reference point against which future changes can be assessed. The project also identified some northern variants of the habitat type, improved strandline vegetation definition and stressed the influence of the water table. All of these will be important for assessing impacts of climate change alongside other pressures. The SNH work was preceded by a similar exercise in England (Murdock *et al*., 2010). In contrast, the English area figure for the habitat was revised down to 4276 ha since the 1990s national inventory (Sneddon and Randall, 1994). This habitat is difficult to map due to its open vegetation and naturally dynamic nature, so these latest inventories have provided a clear method to assess future change.

2.4.3 Processes (both natural and anthropogenic) likely to be affected by climate change

There is a complex relationship between relative sea-level rise and the evolution of shingle/gravel barriers. The principle mechanism for barrier change is through wave and surge flows, primarily in extreme storm events that overwash the crest and transfer sediment from the beach face over the crest and down onto the back barrier slope. It has been postulated that there is a strong relationship between the rate of mean sea-level rise and landward movement of gravel barriers (Orford *et al*., 1995). Where an artificial profile or position is maintained for flood-risk management, the greater the potential breakdown and failure, as seen at Porlock in Somerset (Orford *et al*., 2001). Sediment supply and morphology of the landward environment, combined with past or current human modifications, are key factors which mean each site will have a different response to storm events. Breeding colonies of ringed plover (*Charadrius hiaticula*) on shingle-dominated foreshores are likely to be affected by rising sea levels, summer droughts, and habitat shifts, as are some plant species such as sea campion (*Silene uniflora*) and sea kale (*Crambe maritima*) as they lose suitable climate space under 3°C and 4.5°C temperature rise scenarios respectively.

Climate change could influence the way in which shingle structures contribute to reducing risk of flooding, potentially leading to changes in management responses. Gravel beaches slow the run-up of waves and absorb wave energy, and allow water percolation, thus providing the main flood-risk management benefit as opposed to just crest height. Rising sea levels could reduce natural inputs of marine-derived material which help maintain volume of shingle beaches. It is not clear if increased erosion of cliffs could provide a substitute source of similar size and geology, and constraints to longshore drift may also occur due to presence of coastal defences. Beach form may change as systems adjust to different conditions. In most cases there will be a landward movement in response to sea-level rise, and substantial re-working of the available sediment.

2.5 Maritime cliff and slope

2.5.1 Description

Maritime cliff and slope comprises any form of sloping through to vertical faces on the coastline where a break in slope is formed by failure and/or coastal erosion. On the seaward side, the cliff slope extends to the limit of the supralittoral zone. On the landward edge the boundary is less clear, but is often understood to include the zone affected by sea-spray salt deposition, typically ~50 m, but occasionally up to 500 m (Jones *et al*., 2011), although in practice agricultural land or infrastructure frequently occur closer to the cliff top than this, and the remaining strip of natural vegetation is considerably narrower. Coastal cliffs are broadly classified as ‘hard cliffs’ or ‘soft cliffs’, however, in reality, these may exist as mosaics or intermediate types (Natural England and RSPB, 2014). The vegetation of maritime cliff and slope varies with exposure to wind and salt spray, the lithological composition, soil depth and stability of the substrate, its water content, and on soft cliffs the time elapsed since the last slope failure. The result is a range of specialised vegetation communities, restricted to maritime cliff and slope and often exhibiting distinct zonation. The communities in the most exposed locations, in close proximity to the sea, are made up of highly adapted plant species that are salt tolerant and able to withstand the extreme conditions. Whereas further inland, maritime forms of grassland and heathland can develop as the effects of salt spray decline and the influence of other factors increase, such as soil depth and lithology. The vegetation of soft cliffs is more varied but where there are fresh exposures, these are often characterised by pioneer species of disturbed ground.

Hard cliffs are formed of rocks resistant to wave erosion and subaerial weathering, such as gneiss, basalt, granite, sandstone and limestone, but can also include softer rocks, such as chalk. Vertical or sub-vertical profiles are common since the restricted amount of debris produced by failure is easily removed by wave activity. Soft cliffs are characterised by less-resistant rocks like shales or unconsolidated materials, such as glacial till that produce large volumes of failure debris that is removed slowly by wave activity. Rates and patterns of erosion differ between hard and soft cliffs, with soft cliffs experiencing frequent or episodic failures; slumping and landslips, often driven by undercutting from wave action and groundwater seepage.

2.5.2 Extent and regional pattern trends

Approximately 4060 km of the UK coastline has been classified as ‘cliff’ (in reality hard rocky coast), with an estimated 1084 km in England, 2455 km in Scotland and 522 km in Wales (JNCC, 2013). In the UK, hard cliffs are widely distributed on more exposed coasts, dominating coastlines of the south-west and the south-east of England, and in more-resistant lithologies in north-west and south-west Wales, western and northern Scotland, and on the north coast of Northern Ireland. Shorter lengths or lower cliffs also occur extensively around the coasts, albeit with clustered distribution. Soft cliffs are more restricted to the east and central south coasts of England and to a lesser extent Cardigan Bay and north-west Wales. England and Wales are estimated to have lengths of 255 km and 101 km respectively. Of the 255 km, 80% of this is found in the seven counties Devon, Dorset, Humberside, Norfolk, Suffolk, Isle of Wight, and Yorkshire. Shorter lengths of soft rock cliffs occur in north-east Scotland on the Pennan coast and Nigg, and Northern Ireland. The UK holds a significant proportion of the soft cliff in north-western Europe (Whitehouse, 2007). Whilst it is assumed that the overall length of cliffs is stable, the narrow strip of cliff top vegetation is vulnerable to a number of pressures which are likely to be accentuated by climate, including cliff erosion on the seaward edge, and agricultural encroachment, and development on the landward edge, which are leading to loss and fragmentation of habitat.

2.5.3 Processes (both natural and anthropogenic) likely to be affected by climate change

Cliff profiles are highly variable given their control by both detailed structural architecture and lithology (May and Hansom, 2003), and with the geomorphological character of the hinterland. The complex interplay between atmospheric, terrestrial, and marine processes, and the controlling role of geology hinders the formulation of reliable models of coastal cliff response to climate-change effects (Masselink and Russell, 2013). Marine erosion is a critical natural function of both hard and soft cliffs, however climate change is likely to increase erosion rates through a number of pathways. Changes to the regularity and severity of storms and wave climate could alter patterns of undermining and the removal of basal sediments, and increase direct abrasive forces from wave and wind action. A study of erosion rates at two vulnerable cliffs in Cornwall during the most energetic winter (2013–2014) since 1948 recorded erosion rates at a factor three to five times larger than the long-term average (Earlie *et al.*, 2018).

Soft cliffs are dynamic in nature and erode rapidly; areas with the most-rapid rates of recession are on the south and east coasts of England. For example, Holderness cliff erosion is estimated to supply 3M m^3 a year of fine-grained sediment into the marine system, most of which is transported to the Lincolnshire coast and the Humber (HR Wallingford, 2002). It is very likely that currently eroding stretches of coast will experience increased erosion rates due to sea-level rise (Masselink and Russel, 2013), therefore these retreating coastlines are particularly vulnerable. Increased rainfall in the future may also lead to increased slope failure, particularly affecting the movement of groundwater in softer lithologies. High levels of rain have reactivated landslides on the Dorset coast at Lyme Regis and Cayton Bay, Yorkshire.

Building defences as part of coastal erosion risk management, siting of infrastructure such as railway lines at the toe of cliffs, and modification of drainage on the cliffs have led to the stabilisation of soft cliffs; constricting sediment movement and restricting the creation of new exposures with deleterious effects for invertebrates and pioneer plant communities characteristic of these open areas of disturbed ground. Unhindered dynamic processes, such as erosion and cliff failure and unimpeded drainage, are critical to soft cliffs retaining their invertebrate interest (Howe, 2015). Because the frequent failure of soft rock cliffs propagates inland to threaten human assets, such cliffs with no artificial coast protection are a rare resource in the British Isles and in Western Europe. Schemes to extend or replace coast protection are still being proposed, often in response to reactivation of landslides.

Soft cliff erosion is an important source of sediment for other coastal habitats, such as dunes, shingle, and saltmarsh. These dynamic coastal systems have the potential to be self-regulating in the face of rising sea levels (Natural

England and RSPB, 2014). However, sediment availability is a critical factor in enabling these habitats to adapt. Protection of the base of cliffs stops erosion, but prevents the introduction of eroded cliff material into the nearshore sediment system, which may also have a deleterious effect on downdrift beaches (Masselink and Russell, 2013). It is estimated that in the 100 years up to the 1990s, 860 km of coast protection works have been constructed to reduce erosion (Lee, 2001), reducing sediment input by an estimated 50%.

3. WHAT COULD HAPPEN IN THE FUTURE?

3.1 Saltmarsh

Further loss of saltmarsh habitat is likely in the near future. Relative sea-level rise will mean deeper waters and bigger waves will reach saltmarsh, causing erosion at the seaward edge. This eroded sediment is then deposited landwards– a process known as ‘roll over’ allowing the saltmarsh to accrete vertically (Pethick, 2006). However, in the UK, much of the extent of estuaries are bounded by artificial static sea defences, meaning that landward migration of habitat is unable to occur. This process is known as ‘coastal squeeze’. Other human activities at the coast, such as dredging, also potentially increase the vulnerability of marshes to climate change. This diminishes and disrupts the natural sediment supply which will slow down saltmarsh growth, further reducing its natural recovery capacity and resilience (MCCIP, 2018).

Some ongoing loss of habitat will be mitigated by the increased interest in restoration. However, research suggests that restoration of saltmarsh may not recreate habitat that functions, or provides ecosystem services equivalent to those from natural systems. The timescale for restored sites in the UK to attain equivalent soil C pools has been estimated as approximately 100 years (Burden *et al.*, 2013), whereas it can also take many decades for plant communities in restored marshes to resemble those of natural marshes, if indeed at all (Mossman *et al.*, 2012). Furthermore, as plant diversity has been linked to soil stability (Ford *et al.*, 2016), and species richness is known to be lower in restoration sites (Garbutt and Wolters, 2008) than natural saltmarshes, habitat to mitigate loss may prove to be less resilient in the face of changing climatic conditions such as increased wave energy.

3.2 Machair

Machair is arguably as much a socio-economic feature as an ecological one, and the two should be linked at policy level if effective conservation of the habitat is to be achieved (Angus, 2001). With much of the machair low-lying and thus subject to marine or freshwater flooding, the integrity of the higher dune ridge that separates the machair from the Atlantic seaboard, is critical. Where there is erosion, it can result in ‘roll over’ of the dune on to the machair, re-circulating sand within the wider system. It has been assumed (perhaps wrongly) that the landward movement of sand applies across the extent of the system involved, but while habitats are capable of rollover, land tenure is fixed. Land was allocated to crofts only after long periods of campaigning, and the attachment to land in the machair areas is exceptionally strong. As with many other coastal habitats, human response to climate change could be more problematic for the environment than the climate change itself, and it is essential that any adaptation is as well informed as it can be (Angus, 2018). There are also socio-economic influences that are critical to all the machair islands that may be external to the habitat but have the potential to affect (usually negatively) active crofting, such as transport provision, employment, and civil infrastructure, especially in areas experiencing declining and/or ageing population.

Dynamic Coast (www.dynamiccoast.com) has identified significant areas of change in beach systems in the machair islands. In the Western Isles of Scotland, the extent of erosion has reduced since 1970 relative to the historical period (1880s–1970), from 16% to 13%, while the extent of accretion has increased slightly from 11% to 12%. These figures should not be interpreted as balancing each other out as impacts vary from site to site and even within sites. Notably, average rate of retreat has quickened from the historical to the recent period (0.6 to 1.3 m per year) whilst accretion has fallen slightly from 0.9 to 0.8 m per year. Systems on islands such as Baile Sear and peninsulas such as Aird a'Mhòrain (both in North Uist) were particularly prone to erosion on their east coasts (Hansom *et al*., 2017). The situation in Tiree involved higher rates of existing and predicted erosion; the area of An Riof was particularly vulnerable, as there is an extensive area of very low-lying land inland of the eroding dune ridge of Tràigh Bhàgh (Fitton *et al*., 2017).

Machair has evolved over millennia in association with varying sea levels and human management, and has survived over this period in an area of extreme climate. It might be that the habitat as a whole (i.e. in the machair system sense) will prove resilient as it has in the past, but the role of people in this environment, and their response to change, is likely to be a pivotal aspect of machair's future: the habitat as a whole could well prove resilient, but the added value provided by human input is arguably more vulnerable.

2.3 Sand dunes

With respect to wind speeds, modelling experiments on French dunes suggest that more-frequent storms have less impact than overall increases in wind speed intensity, while net shifts in the dominant wind direction may alter rates of dune movement (Gabarrou *et al*., 2018). In dry dunes, increased summer drought is likely to result in soil-moisture limitation of growth of many vascular plants, leading to increases in cover of drought-adapted mosses and lichens. Fuzzy bioclimatic modelling in Denmark and Europe of 81 species, including many coastal species, suggested that roughly 75–85 % might show a decline in Denmark (at similar latitude to northern Britain) under different climate-change scenarios (Normand *et al*., 2007).. However, some species are likely to benefit. Species predicted to expand their range in Denmark, and by extension the UK, were *Beta vulgaris*, *Glaucium flavum*, *Salsola kali*, and *Sanguisorba minor*. Species predicted to decline in range included *Cochlearia officinalis*, *Fillipendula ulmaria*, *Honkenya peploides*, *Potentilla anserina* and *Salix repens* (Normand *et al*., 2007).

With respect to water tables, UK modelling suggests that dune water-tables may drop by over 1 m by 2080 (Clarke and Ayutthaya, 2010), while more-recent evidence demonstrates plant community shifts and changes in nitrogen cycling resulting from much smaller changes in water tables (Rhymes *et al*., 2016; 2018).

2.4 Shingle

Shingle habitats develop in highly dynamic situations, so in theory could adapt to rising sea levels as plants colonise re-worked shingle deposits. Sites might become smaller or take a different form, with some of the more-mature communities reverting to vegetation more typical of the seaward forms. Such shifts are part of the overall dynamics of the habitat type and have been identified at Dungeness as part of the vegetation sequence (Ferry et al., 1990). However, the majority of English sites have been modified by coastal management, and the ability for natural landward transgression is limited by sediment supply. The breakdown of barrier beaches is a key risk for many locations and there is limited understanding of how this process happens amongst flood risk management engineers. Lack of appreciation that barriers need to move will lead to increased risk of breakdown. Barriers that are allowed to roll back are more resistant to breaching, but this needs to be planned and there are implications for other wetland or terrestrial habitats behind them. Where it cannot adjust by moving landwards, the profile is likely to further erode and steepen, which can also in turn increase nearshore wave energy. The importance of the sediment supply and processes in the inter-tidal and sub-tidal areas also need to be taken into account for both large and small systems. Increases in wave heights could also be an issue (Masselink and Russel, 2013).

Risks to shingle aquifers from a combination of sea-level rise forcing saline intrusion and reduced summer rainfall, particularly in east and south England where some of the key shingle sites occur, could be problematic especially where these are used as public water supplies, as is the case at Dungeness, Kent (Denge gravel aquifer). Here, groundwater levels influence the vegetation of the shingle and open-water areas important for wintering birds. A balance between abstraction and recharge will help safeguard groundwater levels and prevent saline intrusion of the aquifer. The aquifer requires close monitoring and is subject to specific drought restrictions (Natural England, 2013). Shingle structures have limited surface water retention, and are strongly dependent on rainfall (Davy *et al*., 2001), therefore extended dry periods in spring could alter vegetation and the existing thin soils, thus reducing resilience to other forms of damage.

Jones *et al*. (2013) point out that the scarce nature of this habitat could lead to low recolonisation rates after disturbance. Changes in patterns of precipitation or temperature will affect vegetation composition. Water retention is poor and evapo-transpiration is likely to increase year round but particularly in autumn and summer, leading to greater impact of summer droughts. Shingle systems will therefore undergo long-term change as well as change in response to extreme events. The latter appear to be more frequent which means that planning for change – vital for the continued existence of this and other coastal habitats – must begin now so that necessary adaptations can be started.

2.5 Maritime cliff and slope

Under climate change, sea-level rise and changes to the wave climate (storminess and prevailing wave direction) will impact erosion rates (Masselink and Russel, 2013). On hard cliffs, rocky shore platforms may have more marine scour/wave attack at cliff foot due to beach lowering, and may ultimately be lost with sea-level rise, because they cannot accrete like soft sediments. Headlands form natural hard points and may promote changes in the shape of intervening bays and beaches. Both hard headlands and rocky shore platforms can also provide protection for other coastal habitats, such as perched saltmarsh, erosion of these structures would result in the decline and eventual loss of the habitats they protect. Land use such as intensive agriculture and development along cliff tops means the coastal slope habitats are often only found in a fragmented narrow band. Habitat loss and fragmentation due to a combination of coastal retreat and lack of space for cliff top habitats to rollback is a serious risk to both coastal slope vegetation and invertebrates reliant on cliff top habitat for both breeding and foraging. Over land the projected general trends of climate changes in the 21st century, predict warmer, wetter, winters and hotter, drier, summers (UKCP18) with greater increases in maximum summer temperatures over the southern UK compared to northern Scotland (UKCP18). Warmer temperatures could lead to changing patterns and distribution of vegetation and species. Warmer temperatures and increased disturbance may favour invasive species, for example the introduced alien Hottentot fig (*Carpobrotus edulis*) grows at 50 cm per year and smothers important native species on cliffs in southern England resulting in a change in species composition and a need for management (Frost, 1987). Movement of thermophilic species north with increasing temperatures could occur, however this will be dependent on habitat connectivity. Connectivity of the sea-cliff habitat is naturally restricted by the physical nature of the coast and is in itself affected by increases in cliff erosion rates driven by climate change. Reduced summer rainfall, which is most likely to be greatest in the south of England (UKCP18), could lead to increases in salinity in some coastal habitats leading to increases in salt-tolerant species (Mossman et al., 2015). Increased winter rainfall combined with milder winter temperatures could enable more competitive grasses to survive on the shallow clifftop soils (Natural England and RSPB, 2014). UK climate change projections show a pattern of larger increases in winter precipitation over southern and central England, and some coastal regions towards the end of the century (UKCP18).

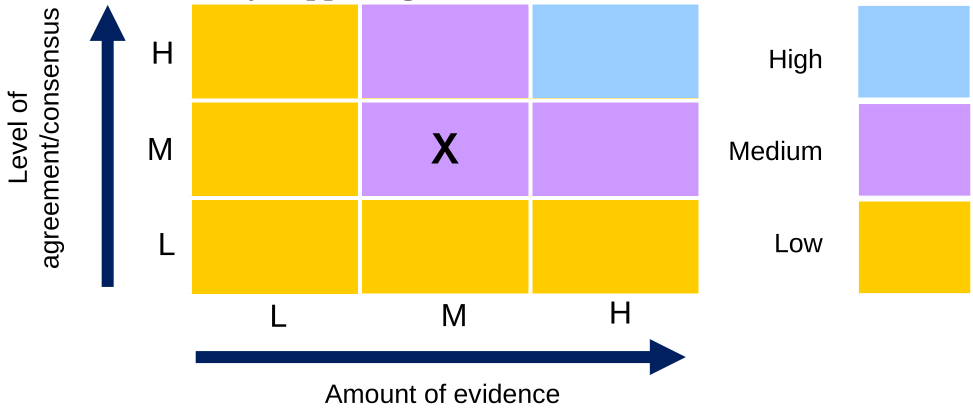
In soft cliffs, increased winter rainfall may promote greater risk of landslides. Old landslide complexes are likely to reactivate more rapidly than expected as groundwater pressure increases. The balance of bare ground to successional vegetation may be altered on soft cliffs, with potential loss of mosaics important for scarce invertebrates, but conversely may create greater areas of the new habitat necessary for early successional species. The increasing instability and elevated erosion rates driven by sea-level rise could lead to more demands for coast protection on the already depleted soft-cliff resource.

4. CONFIDENCE ASSESSMENT

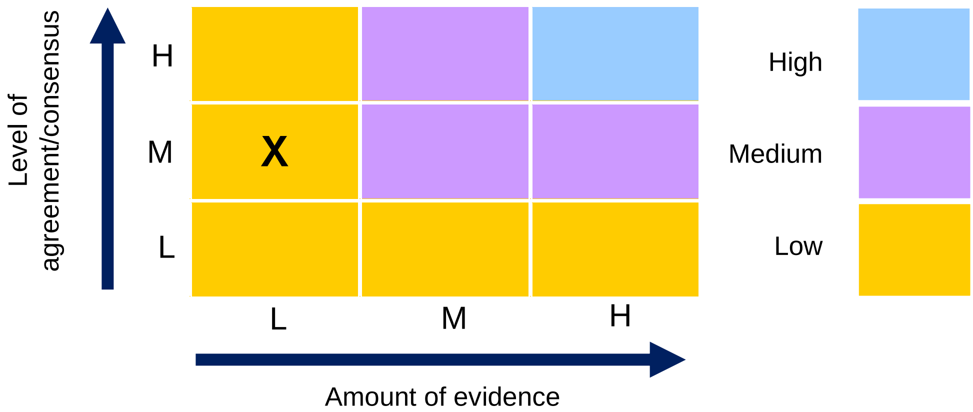
The levels of confidence assigned to the current and future impact of climate change on coastal habitats remains the same as previous reporting in 2010 and 2013. Though there is a great quantity and quality of data available it is still difficult to fully understand the impact of climate change across the diverse habitats of the UK.

Within saltmarsh uncertainties remain in respect to interaction between stressors and the natural erosion/accretion phases (Bouma *et al*., 2016). Impacts of climate change on geomorphology and plant ecology are well understood, but little is known about impacts on other functional groups like microbes and animals (Evin *et al*., 2002). Shingle systems are increasingly studied in terms of their response to storm events. There is high confidence that systems will change, but lower confidence in how they might change, due to the need to understand sediment budgets. There is less information on the vegetation changes driven by increased temperatures and shifts in rainfall patterns. However, the systems support naturally drought-tolerant vegetation. In terms of cliff erosion there is a medium to high confidence of the process governing soft cliff erosion, but this is in primarily in England where there is less data on hard cliffs. Predicting cliff retreat still remains difficult and is the largest technical hurdle in assessing the impacts of climate change on cliffs. As with shingle, research relating to climate change impacts on vegetation is low.

4.1 What is already happening



4.2 What could happen in the future?



5. KEY CHALLENGES AND EMERGING ISSUES

5.1 Challenges

5.1.1 Accounting for coastal habitats in national greenhouse gas inventories

The carbon sequestered in vegetated coastal ecosystems, such as saltmarshes, has been termed 'Blue Carbon' ('Blue C') and there has been considerable recent attention to the potential role it could play in climate mitigation. For example, in Scotland, Blue C is mentioned in the Climate Bill and plan up to 2032. In 2014, the Intergovernmental Panel on Climate Change (IPCC) published guidelines on how to include wetland drainage and rewetting activities in national greenhouse gas (GHG) inventories, including that from coastal wetlands. By reviewing the global literature, default emission factors have been proposed for drainage and rewetting as a 'Tier 1' approach, with the suggestion that country-specific ('Tier 2') defaults should be developed using empirical data. A further step would be to develop 'Tier 3' approaches using process models that take account of change over time and response to differing environmental factors. However, quantitative empirical data on rates of carbon accumulation following restoration or drainage of habitat in the UK remain scarce and presents a challenge to the scientific community. It has also been noted that little understanding currently exists on how coastal carbon accumulation rates will change with a changing climate (Chmura, 2011), with some evidence that the net impact of climate change (based on sea-level rise and increased temperatures) will likely increase carbon burial in the first half of the 21st century (Kirwan and Mudd, 2012), and slightly decrease in the second half of the century. There are global programmes working to mitigate climate change through the restoration and sustainable use of coastal and marine ecosystems, notably The Blue Carbon Initiative, and The International Partnership for Blue Carbon.

5.1.2 Linking the land to the sea: Carbon perspective

Coastal habitats that sit on the fringe of the terrestrial environment tend to be overlooked by both the terrestrial and marine research communities, but these systems at the land-ocean interface are important links between terrestrial and marine ecosystems. Globally there has been a recent concerted effort to understand the transfer of carbon from the terrestrial to marine environment (Cui *et al*., 2016; Smeaton and Austin, 2017). For example, saltmarsh is known to capture and bury terrestrial and marine-derived carbon (Chmura *et al*., 2003; Van de Broek *et al*., 2016), but these carbon-transfer studies treat coastal habitats as passive environments which purely facilitate the transfer of carbon across the land–ocean interface. Currently the NERC C-SIDE (Carbon Storage in Intertidal Environments) project seeks to quantify the terrestrial carbon contribution to saltmarsh in the UK and to better understand the role of saltmarsh in the global carbon cycle. Future changes in climate could potentially disturb these carbon dynamics by increasing the input of terrestrial derived carbon to the intertidal and marine environments, but the true consequences of a changing climate on carbon across the land-ocean interface is largely unknown. To understand the linkages between the terrestrial, intertidal, and marine environments, and the impact of a changing climate, a true multidisciplinary catchment to coast approach is required. The NERC-funded project LOCATE (Land Ocean Carbon Transfer) is currently researching the fate of terrestrial organic matter from land to sea with a particular focus on estuaries and coastal waters. An objective of this project is to build a new model of organic matter cycling in both marine and freshwaters to predict the future evolution of the land to sea carbon flux.

5.1.3 Coastal squeeze

Coastal systems are naturally dynamic and intrinsically resilient to change. For example, they can be self-regulating in response to sea-level rise, but only if both an adequate sediment supply and space for landward migration are maintained. However, natural coastal environments have been altered over many decades by the construction of hard coastal defences, with development on the coast not taking long-term stability into account (CCC, 2018). The inability of coastal systems to migrate inland due to artificial, static, sea defences is known as ‘coastal squeeze’.

The Committee on Climate Change (2018) has predicted that climate change will exacerbate the exposure of the English coast to flooding and erosion, and that the present approach to coastal management is unsustainable. Creating more-natural shorelines to restore the function of natural coastal processes, whilst promoting cultural acceptance of the dynamic nature of these habitats, is needed in the future.

5.1 Emerging issues

5.2.1 Managing and working with the natural environment: sand, sediment, and shingle recharge

Sediment recharge (using sediment derived from either dredging, or material ‘recycled’ from areas of accretion along the foreshore) is increasingly being used to counter erosion in coastal areas where sediment supply is limited. Sediment supply can be reduced where sea-level rise increases distance between offshore deposits and the coast, exacerbated by reduced inputs where coastlines are modified by hard engineering. By introducing sediments near the intertidal area to be re-deposited on the coast via natural processes, the aim is to maintain habitat, and in many instances also a standard of flood defence using a form of ‘soft engineering’ which works with coastal processes. Recharge is also used directly in restoration projects to raise the land surface level before reconnection to the tide (ABPmer, 2017). Large-scale shingle recharge, similar to the ‘Sand Engine’ in The Netherlands (primarily aimed at dune restoration, Stive, *et al*., 2013), is being mooted as an approach for parts of England’s south and east coasts that could work with natural processes and lead to more-natural function (Cowling, 2016). However, as shingle vegetation can only develop above the high tide mark and designs do not yet appear to take this into account, it is uncertain whether this could offset some of the likely climate-change related changes expected in shingle habitats.

The concept of working with natural processes is being used in sand-dune restoration activities both ongoing and planned as part of two large LIFE-funded projects in England and Wales. These projects aim to encourage natural dune dynamics to allow dune systems to self-regulate in response to climate change. There is acknowledgement that this procedure works better with large areas than small, fragmented interventions.

5.2.2 Natural capital

‘Natural capital’ refers to natural resources and the benefits that these resources provide. As of 2015, the UK natural capital was estimated to be £761 billion (Office of National Statistics, 2018) with 58% of this value being attributed to cultural and regulating ecosystem services (recreation, pollution removal, and carbon sequestration) – within which coastal habitats play a crucial role. One of the main areas of focus is the carbon sequestration of different environments. It is well known that saltmarsh is one the most efficient natural habitats at sequestering carbon (Duarte *et al*., 2005). The latest experimental carbon stock valuations (Office of National Statistics, 2016), which are still in development, do not include these habitats due partly to data limitations on the exact extent of coastal margin habitats, and the difficulty of reconciling those extents with the extent of other habitats. Previous economic valuation has suggested the carbon sequestration service that saltmarsh, sand dunes and machair will provide between 2000 and 2060 was in the region of £1 billion. It was also estimated that at current rates of natural and anthropogenic habitat loss £0.25 billion worth of carbon sequestration capability could be lost by 2060 (Beaumont *et al*., 2014). These economic valuations are built on complex but, in some cases, limited data which can be misunderstood and misused (i.e. the origins and annual fluxes of carbon into and out of saltmarsh remain poorly understood).

In the future we will need the quality and quantity of habitat data to significantly increase to allow a robust economic valuation to support policy interventions. Several ongoing and past projects (e.g. UKNEA, CBESS, COASTWEB) have applied natural capital valuation techniques to the coastal habitats, but there is still a significant need for environmental economists to work with the coastal and marine science communities to understand the complex nature of these habitats. Further, it is important to consider that a universal approach is not suitable to valuing the services coastal environments provide.

5.2.3 Assessing the social impact of climate change on coastal communities

The UK is a coastal nation with a population of approximately 30 million living in urban coastal areas, 40% of all manufacturing occurs on or near by the coast, 90% of all trade comes through coastal ports, and coastal tourism and recreation support the economy of many towns and regions. Yet very little research to date has been carried out on the potential social impacts of climate change on the UK coast. There are a multitude of potential social impacts ranging from health to livelihoods. Climate change is likely to negatively affect people’s health, particularly through a greater occurrence of extreme events such as flooding and heatwaves (Department of Health, 2008). It is also suggested that climate change will affect on coastal livelihoods (Zsamboky *et al*. 2011), particularly for those who depend on the coast for employment (e.g. in fishing and tourism). Areas that suffer from extreme flooding events or are considered to be at high risk may be affected economically due to reduction in housing values, development, and investment.

To tackle this, a multidisciplinary approach moving beyond the current natural/physical science viewpoint of climate change impacts is needed. Bringing together social, health, economic, and natural scientists will be the first steps to quantifying the social impact of climate change on coastal communities.

Acknowledgements

The authors would like to thank the wider team of colleagues that have helped support the production of this Report Card. CS participation with this report was supported by the C-SIDE project funded by Natural Environment Research Council (grant NE/R010846/1).

REFERENCES

ABPmer (2017) Using Dredge Sediment for Habitat Creation and Restoration. A cost benefit review. A summary of the techniques, costs and benefits associated with using fine dredge sediment to 'recharge' intertidal habitat. *ABPmer Internal White Paper*, Report No. R.2865.

ABPmer Online Marine Registry (2018) Database of international shoreline adaptation projects. Available online: www.omreg.net/

Adaptation Sub-Committee (2013) *Managing the Land in a Changing Climate*. Report to the Committee on Climate Change, https://www.theccc.org.uk/wp-content/uploads/2013/07/ASC-2013-Book-singles_2.pdf

Angus, S. (2001) The conservation of machair in Scotland: working with people. Coastal dune management: shared experience of European conservation practice [Houston, J.A., Edmondson, S.E. and Rooney, P.J. (eds.)]. Liverpool University Press, Liverpool, pp. 177–191.

Angus, S. (2004) De tha machair? Towards a machair definition. *Proceedings Vol.2, Littoral 2004. Delivering Sustainable Coasts: Connecting science and policy*, Cambridge Publications, pp. 552–558.

Angus, S. (2006) De tha machair? Towards a machair definition. *Sand Dune Machair*, Aberdeen Institute for Coastal Science & Management, Aberdeen, pp. 7–22.

Angus, S. (2018) Beyond the meta-ecosystem? The need for a multi-faceted approach to climate change planning on coastal wetlands: An example from South Uist, Scotland. *Ocean & Coastal Management*, **165**, 334–345.

Angus, S. and Rennie, A. (2014) An Ataireachd Aird: the Uist storm of January 2005. *Ocean & Coastal Management*, **94**, 22–29.

Baily, B., & Pearson, A. W. (2007). Change detection mapping and analysis of salt marsh areas of central southern England from Hurst Castle Spit to Pagham Harbour. *Journal of Coastal Research*, 1549–1564, https://doi.org/10.2112/05-0597.1

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. and Silliman, B.R. (2011) The value of estuarine and coastal ecosystem services. *Ecological Monographs*, **81**, 169–193.

Bartholomeus, R.P., Witte, J.P.M. and Runhaar, J. (2012) Drought stress and vegetation characteristics on sites with different slopes and orientations. *Ecohydrology*, **5**, 808–818.

Beaumont, N.J., Jones, L., Garbutt, A., Hansom, J.D. and Toberman, M. (2014) The value of carbon sequestration and storage in coastal habitats. *Estuarine, Coastal and Shelf Science*, **137**, 32–40.

Blackwell, M.S.A., Hogana, D.V. and Maltbya, E. (2004) The short-term impact of managed realignment on soil environmental variables and hydrology. *Estuarine, Coastal and Shelf Science*, **59**, 687–701.

Boorman, L.A. (2003) Saltmarsh review. An overview of coastal saltmarshes, their dynamic and sensitivity characteristics for conservation and management. *Joint Nature Conservation Committee Report* No. 334, JNCC, Peterborough.

Born. K. (2005) Predicting habitat change in Poole Harbour using aerial photography. In *The Ecology of Poole Harbour*, [Humphreys, J. and May, V. (eds.)]. Elsevier, London, pp. 239–253.

Bouma, T.J., van Belzen, J., Balke, T., van Dalen, J., Klaassen, P., Hartog, A.M., Callaghan, D.P., Hu, Z., Stive, M.J.F. and Temmerman, S. (2016). Short-term mudflat dynamics drive long-term cyclic salt marsh dynamics. *Limnology and Oceanography*, **61**, 2261–2275.

Burd, F. (1989) *The Saltmarsh Survey of Great Britain*. An Inventory of British Saltmarshes, Research and Survey in Nature Conservation No. 17, Nature Conservancy Council, Peterborough.

Burden, A., Garbutt, R.A., Evans, C.D., Jones, D.L. and Cooper, D.M. (2013) Carbon sequestration and biogeochemical cycling in a saltmarsh subject to coastal managed realignment. *Estuarine, Coastal and Shelf Science*, **120**, 12–20, doi: 10.1016/j.ecss.2013.01.014

CBESS (n.d.) Available online: https://synergy.st-andrews.ac.uk/cbess/

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R. and Lynch, J.C. (2003) Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, **17**, 1111, doi:10.1029/2002GB001917

Chmura, G. L. (2011) What do we need to assess the sustainability of the tidal saltmarsh carbon sink? *Ocean and Coastal Management*, **83**, 25–31, doi:10.1016/j.ocecoaman.2011.09.006

Clarke, D. and Ayutthaya, S.S.N. (2010) Predicted effects of climate change, vegetation and tree cover on dune slack habitats at Ainsdale on the Sefton Coast, UK. *Journal of Coastal Conservation*, **14**(2), 115–125.

COASTWEB (n.d.) Available online: https://www.pml.ac.uk/Research/Projects/CoastWEB

Committee on Climate Change (2018) Managing the Coast in a Changing Climate, Committee on Climate Change. Available online: https://www.theccc.org.uk/publication/managing-the-coast-in-a-changing-climate/

Cooper, N.J., Cooper, T. and Burd, F. (2001) 25 years of saltmarsh erosion in Essex: Implications for coastal defence and nature conservation. *Journal of Coastal Conservation*, **9**, 31–40.

Cowling, M. (2016) A Shingle 'Engine' for Slaughden? Suffolk Coast Forum (SCF), 3rd February 2016, The Crown Estate. Available online: http://www.greensuffolk.org/assets/Greenest-County/Water--Coast/Suffolk-Coast-Forum/SlaughdenShingleEngineSuffolkCFFeb2016.pdf

C-SIDE (n.d.) Available online: https://www.c-side.org/

Cui, X., Bianchi, T. S., Savage, C. and Smith, R.W. (2016) Organic carbon burial in fjords: Terrestrial versus marine inputs. *Earth and Planetary Science Letters*, **451**, 41–50, https://doi.org/10.1016/j.epsl.2016.07.003

Curreli A., Wallace H., Freeman C., Hollingham M., Stratford C., Johnson H. and Jones, L. (2013) Eco-hydrological requirements of dune slack vegetation and the implications of climate change. *Science of the Total Environment*, **443**, 910–919.

Dargie, T. and Duncan, K. (1999) The Sand Dune Vegetation Survey of Scotland. In *Scotland's Living Coastline* [Baxter, J.M. Duncan, K., Atkins, S. and Lees, G. (eds)], The Stationery Office, London.

Davy, A.J., Willis, A. J. and Beerling, D. J. (2001) The Plant Environment: Aspects of the Ecophysiology of Shingle Species. In *Ecology and Geomorphology of Coastal Shingle* [Packham, J.R., Randall, R.E., Barnes, R.S.K. and Neal, A. (eds)], Westbury Academic and Scientific Publishing, Otley.

Department of Health (2008) *Health Effects of Climate Change in the UK 2008*. An Update of the Department of Health report 2001/2002. Available online: https://pdfs.semanticscholar.org/8df9/e5f586521f5f35f1a903c3497c2093595d3f.pdf

Doody, J.P. (2008) *Saltmarsh Conservation, Management and Restoration*, Coastal Systems and Continental Margins Series, Springer, USA.

Doody, J.P. (2013) *Sand Dune Conservation, Management, and Restoration*, Springer, USA.

Dornbusch, U., Bradbury, A., Curtis, B. and Lane, G. (2010) *Beach management plans for mixed beaches: Review and ways forward*. Available online: https://www.se-coastalgroup.org.uk/wp-content/uploads/2013/10/Beach-management-plans-for-mixed-beaches.pdf

Duarte, C. M., Middelburg, J. J. and Caraco, N. (2005) Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*, **2**, 1–8, doi:10.5194/bg-2-1-2005

Duarte, C.M., Dennison, W.C., Orth, R.J. and Carruthers, T.J.B. (2008). The charisma of coastal ecosystems: Addressing the imbalance. *Estuaries and Coasts*, **31**, 233–238

Earlie, C., Masselink, G. and Russell, P. (2018) The role of beach morphology on coastal cliff erosion under extreme waves. *Earth Surface Processes and Landforms*, **43**, 1213–1228.

Evin, L.A. and Talley, T.S. (2002) Influences of vegetation and abiotic environmental factors on salt marsh invertebrates. In *Concepts and Controversies in Tidal Marsh Ecology*, Springer, pp. 661–707.

Ferry, B., Lodge, N. and Waters, S. (1990) Dungeness: A vegetation survey of a shingle beach. *Research and Survey in Nature Conservation*, No. 26, NCC, Peterborough.

Fitton, J.M., Rennie, A.F. and Hansom, J.D. (2017). Dynamic Coast – National Coastal Change Assessment: Cell 5 – Cape Wrath to the Mull of Kintyre, CRW2014/2. Available online: http://www.dynamiccoast.com/outputs.html

Ford, H., Garbutt, A., Ladd, C., Malarkey, J. and Skov, M.W. (2016) Soil stabilization linked to plant diversity and environmental context in coastal wetlands. *Journal of Vegetation Science*, **27**, 259–268, doi: 10.1111/jvs.12367

French, P.W. (1997) *Coastal and Estuarine Management*, Routledge, London, 268 pp.

Frost, L.C. (1987) The alien Hottentot fig (*Carpobrotis edulis*) in Britain – a threat to the native flora and its conservation control. University of Bristol Lizard Project. Available online: www.devon.gov.uk/bap-seacliffandslope.pdf

Gabarrou, S., Le Cozannet, G., Parteli, E.J., Pedreros, R., Guerber, E., Millescamps, B., Mallet, C. and Oliveros, C. (2018) Modelling the Retreat of a Coastal Dune under Changing Winds. *Journal of Coastal Research*, **85**(sp1), 166–170.

Garbutt, A. and Wolters, M. (2008) The natural regeneration of saltmarsh on formerly reclaimed land. *Applied Vegetation Science*, **11**, 335–344, doi.org/10.3170/2008-7-18451

Hansom, J.D., Fitton, J.M. and Rennie, A.F. (2017) Dynamic Coast – National Coastal Change Assessment: Cells 8 and 9 – The Western Isles, CRW2014/2. Available online: http://www.dynamiccoast.com/outputs.html

Haynes, T.A. (2016) Scottish saltmarsh survey national report. *Scottish Natural Heritage Commissioned Report*, No. 786. Available online: https://www.nature.scot/sites/default/files/2017-05/Publication%202016%20-%20SNH%20Commissioned%20Report%20786%20-%20Scottish%20saltmarsh%20survey%20national%20report%20%28A2215730%29.pdf

Horton, B.P., Shennan, I., Bradley, S.L., Cahill, N., Kirwan, M., Kopp, R.E. and Shaw, T.A. (2018) Predicting march vulnerability to sea-level rise using Holocene relative sea-level data. *Nature Communications*, **9**, 2687, doi: 10.1038/s41467-018-05080-0

Howe, M. (2015) Coastal soft cliff invertebrates are reliant upon dynamic coastal processes. *Journal of Coastal Conservation*, **19**(6), 809–820.

HR Wallingford (2002) *Southern North Sea Sediment Transport Study, Phase 2 Sediment Transport Report*. Report EX 4526 produced for Great Yarmouth Borough Council by CEFAS/UEA, Posford Haskoning and Dr Brian D'Olier.

IPCC (2014) 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. [Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds)]. IPCC, Switzerland.

Jones, M.L.M., Angus, S., Cooper, A., Doody, P., Everard, M., Garbutt, A., Gilchrist, P., Hansom, G., Nicholls, R., Pye, K., Ravenscroft, N., Rees, S., Rhind, P. and Whitehouse, A. (2011) Coastal margins. In *UK National Ecosystem Assessment. Understanding Nature's Value to Society*. Technical Report, UNEP-WCMC, Cambridge, pp. 411–457.

Jones, L., Garbutt, A., Hansom, J. and Angus, S. (2013) Impacts of climate change on coastal habitats. *MCCIP Science Review*, 167–179.

JNCC (2013) UK Article 17 EU Habitats Directive Third Report. Available online: http://jncc.defra.gov.uk/page-6387 H1210 http://jncc.defra.gov.uk/pdf/Article17Consult_20131010/H1210_UK.pdf and H1220 http://jncc.defra.gov.uk/pdf/Article17Consult_20131010/H1220_UK.pdf

Kay, A.L., Crooks, S.M., Davies, H.N. and Reynard, N.S. (2011) *An Assessment of the Vulnerability of Scotland's River Catchments and Coasts to the Impacts of Climate Change*. Work Package 1 Report, Centre for Ecology & Hydrology, Wallingford.

Kirwan, M.L. and Mudd, S.M. (2012) Response of salt-marsh carbon accumulation to climate change. *Nature*, **489**, 550–553, doi:10.1038/nature11440

Lee, M. (2001) Restoring biodiversity to soft cliffs. *English Nature Research Report*, 398. Available online: http://publications.naturalengland.org.uk/publication/65025

LOCATE (n.d.) Available online: http://www.locate.ac.uk/

Loebl, M., van Beusekom, E.E.J. and Reise, K. (2006) Is spread of the neophyte *Spartina anglica* recently enhanced by increased temperatures? *Aquatic Ecology*, **40**, 315–324.

Low, E.J. (2005) *Shingle Biodiversity and Habitat Disturbance*. www.sussex.ac.uk/geography/researchprojects/BAR/publish/shingle_bio-and-habitat_disturbace.pdf

Masselink, G. and Russel, P. (2013) Impacts of climate change on coastal erosion. *MCCIP Science Review*, 71–86.

May, V.J. and Hansom, J.D. (2003) *Coastal Geomorphology of Great Britain*. Geological Conservation Review series No. 28, Joint Nature Conservation Committee, Peterborough.

MCCIP (2018) *Climate Change and Marine Conservation: Saltmarsh* [Ladd, C., Skov, M., Lewis, H. and Leegwater, E. (eds)] MCCIP, Lowestoft, 8 pp., doi: 10.14465.2018.ccmco.005-smr

McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Bjork, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H. and Silliman, B.R. (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment*, **9**, 552–560.

Moore, P.J. and Smale, D.A. (2020) Impacts of climate change on shallow and shelf subtidal habitats, relevant to the coastal and marine environment around the UK. *MCCIP Science Review 2020*, 272–292.

Morris, R.K.A., Reach, I.S., Duffy, M.J., Collins, T.S. and Leafe, R.N. (2004) On the loss of saltmarshes in south-east England and the relationship with *Nereis diversicolor*. *Journal of Applied Ecology*, **41**, 787–91.

Mossman, H.L., Davy, A.J. and Grant, A. (2012) Does managed coastal realignment create saltmarshes with 'equivalent biological characteristics' to natural reference sites? *Journal of Applied Ecology*, **49**(6), 1446–1456.

Mossman, H.L., Grant, A. and Davy, A.J. (2015) Biodiversity climate change impacts Report Card. *Technical Paper 10: Implications of climate change for coastal and inter-tidal habitats in the UK*. https://nerc.ukri.org/research/partnerships/ride/lwec/report-cards/biodiversity-source10/

Murdock, A., Hill, C.T., Cox, J. and Randall, R.E. (2010) Development of an evidence base of the extent and quality of shingle habitats in England to improve targeting and delivery of the coastal vegetated shingle HAP. *Natural England Commissioned Report*, No. 054. Available online: http://publications.naturalengland.org.uk/publication/41015?category=43007

Murdock, A.P., Hill, C.T., Randall, R. and Cox, J. (2011) Inventory of coastal vegetated shingle in Scotland. *Scottish Natural Heritage Commissioned Report*, No. 423.

Murdock, A.P., Hill, C.T., Randall, R., Cox, J., Strachan, I., Gubbins, G., Booth, A, Milne, F., Smith, S.M. and Bealey, C. (2014) Inventory of coastal vegetated shingle in Scotland – field validation. *Scottish Natural Heritage Commissioned Report*, No. 739. Available online: https://www.nature.scot/snh-commissioned-report-739-inventory-coastal-vegetated-shingle-scotland-field-validation

Natural England (2013) NCA Profile: 123 Romney Marshes (NE499). Available online: http://publications.naturalengland.org.uk/publication/5701066775592960?category= 587130

Natural England and RSPB (2014) Climate Change Adaptation Manual. *Natural England Publication NE*, 546. Natural England, Unpublished Draft Habitat factsheet Maritime Cliff and Slope.

Nellemann, C., Corcoran, E., Duarte, C.M., Valdes, L., De Young, C., Fonseca, L. and Grimsditch, G. (Eds). (2009) *Blue Carbon*. A Rapid Response Assessment. United Nations Environment Programme, GRID-Arendal, https://gridarendal-website-live.s3.amazonaws.com/production/documents/:s_document/83/original/BlueCarbon_screen.pdf?1483646492

Normand, S., Svenning, J.C. and Skov, F. (2007) National and European perspectives on climate change sensitivity of the habitats directive characteristic plant species. *Journal for Nature Conservation*, **15**(1), 41–53.

Nottage, A.S. and Robertson, P.A. (2005) *The Saltmarsh Creation Handbook: A project manager's guide to the creation of saltmarsh and intertidal mudflat*. The RSPB, Sandy & CIWEM, London, UK.

Office of National Statistics (2016) UK Natural Capital: Experimental carbon stock accounts, preliminary estimates.
Available online: www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapital/experimentalcarbonstockaccountspreliminaryestimates

Office of National Statistics (2018) UK natural capital: Ecosystem service accounts, 1997 to 2015.
Available online: www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapital/ecosystemserviceaccounts1997to2015

Orford J.D., Carter R.W.G., McKenna J. and Jennings, S.C. (1995). The relationship between the rate of mesoscale sea-level rise and the retreat rate of swash-aligned gravel-dominated coastal barriers. *Marine Geology*, **124**(1–4), 177–186.

Orford, J.D., Jennings, S.C. and Forbes, D.L. (2001) Origin, development, reworking and breakdown of gravel-dominated coastal barriers and Altantic Canada: Future scenarios for the British Coast. In *Ecology & Geomorphology of Coastal Shingle*. [Packham, J.R., Randall, R.E. Barnes R.S.K. and Neal, A. (eds)]. Westbury Academic and Scientific Publishing, Otley.

Packham, J.R. and Willis, A.J. (1997) *Ecology of Dunes, Salt Marsh and Shingle*, Springer Science & Business Media, US, 335 pp.

Pakeman, R.J., Alexander, J., Beaton, J., Brooker, R., Cummins, R., Eastwood, A., Fielding, D., Fisher, J., Gore, S., Hewison, R. and Hooper, R. (2015) Species composition of coastal dune vegetation in Scotland has proved resistant to climate change over a third of a century. *Global Change Biology*, **21**(10), 3738–3747.

Pethick, J. (2006) *Review and Formalisation of Geomorphological Concepts and Approaches for Estuaries*. R&D Technical Report, FD2116/TR2, Defra, UK, http://www.estuary-guide.net/pdfs/FD2116_TR2.pdf

Provoost, S., Jones, M.L.M. and Edmondson, S.E. (2011) Changes in landscape and vegetation of coastal dunes in northwest Europe: a review. *Journal of Coastal Conservation*, **15**, 207–226.

Ratcliffe, D.A. (1977). *A Nature Conservation Review. Volumes 1 and 2*. Cambridge University Press, Cambridge.

Rees, S., Angus, S., Rhind, P. and Doody, J.P. (2010) Coastal Margin Habitats. MCCIP Annual Report Card 2010–11, *MCCIP Science Review*, 21 pp.

Rennie, A.F. (2006) *The Role of Sediment Supply and Sea-Level Changes on a Submerging Coast, Past Changes and Future Management Implications*. PhD Thesis, University of Glasgow.

Rhymes, J., Jones, L., Wallace, H., Jones, T.G., Dunn, C. and Fenner, N. (2016) Small changes in water levels and groundwater nutrients alter nitrogen and carbon processing in dune slack soils. *Soil Biology and Biochemistry*, **99**, 28–35.

Rhymes, J., Wallace, H., Tang, S.Y., Jones, T., Fenner, N. and Jones, L. (2018) Substantial uptake of atmospheric and groundwater nitrogen by dune slacks under different water table regimes. *Journal of Coastal Conservation*, **22**, 615–622.

Ritchie, W. (1976) The meaning and definition of machair. *Transactions of the Botanical Society of Edinburgh*, **42**, 431–440.

RSPB (n.d.) *Wallasea Island Wild Coast Project*. Available online: https://www.rspb.org.uk/reserves-and-events/reserves-a-z/wallasea-island-wild-coast-project/

Rodwell, J.S. (ed.) (2000) *British Plant Communities. Volume 5: Maritime Communities and Vegetation of Open Habitats*, Cambridge University Press, Cambridge, UK.

Sanyal, A. and Decocq, G. (2015) Biological Flora of the British Isles: *Crambe maritima*. *Journal of Ecology*, **103**, 769–788, http://onlinelibrary.wiley.com/doi/10.1111/1365-2745.12389/full

Shardlow, M.E.A. (2001) A review of the conservation importance of shingle habitats for invertebrates in the United Kingdom. In *Ecology and Geomorphology of Coastal Shingle* [Packham, J.R., Randall, R.E., Barnes R.S.K. and Neal, A. (eds)]. Westbury Academic and Scientific Publishing, Otley.

Sneddon, P.E. and Randall, R.E. (1993) *Coastal Vegetated Shingle Structures of Great Britain*. JNCC, Peterborough.

Sneddon, P.E. and Randall, R.E. (1994) *Coastal Vegetated Shingle Structures of Great Britain: Appendix 3 – England*. JNCC, Peterborough, http://jncc.defra.gov.uk/page-2644

Smeaton, C. and Austin, W.E.N. (2017) Sources, sinks, and subsidies: Terrestrial carbon storage in mid-latitude fjords. *Journal of Geophysical Research: Biogeosciences*, **122**, 2754–2768.

Stive, M.J.F., de Schipper, M.A., Luijendijk, A. P., Aarninkhof, S. G. J., van Gelder-Maas, C., van Thiel de Vries, J. S. M., de Vries, S., Henriquez, M., Marx, S. and Ranasinghe, R. (2013). A new alternative to saving our beaches from sea-level rise: The sand engine. *Journal of Coastal Research*, **29**(5), 1001–1008.

Stratford, C., Jones, L., Robins, N., Mountford, O., Amy, S., Peyton, J., Hulmes, L. Hulmes S., Jones, F., Redhead, J. and Dean, H. (2014) Survey and analysis of vegetation and hydrological change in english dune slack habitats. *Natural England Commissioned Report*, No. 153, http://publications.naturalengland.org.uk/file/5658059917492224

The Blue Carbon Initiative (n.d.) The International Partnership for Blue Carbon. Available online: https://bluecarbonpartnership.org/

UKCIP (2009) *Coastal Issues*. Available online: www.ukcip.org.uk/index.php?option=com_content&task=view&id=409&Itemid=451

UKCP (2018), UKCIP18 Marine Report. Available online: https://www.metoffice.gov.uk/binaries/content/assets/metofficegovuk/pdf/research/ukcp/ukcp18-marine-report-updated.pdf

UKNEA (n.d.) Available online: http://uknea.unep-wcmc.org/

Van de Broek, M., Temmerman, S., Merckx, R. and Govers, G. (2016) Controls on soil organic carbon stocks in tidal marshes along an estuarine salinity gradient. *Biogeosciences*, **13**, 6611–6624, https://doi.org/10.5194/bg-13-6611-2016

Whitehouse, A.T. (2007) *Managing Coastal Soft Cliffs for Invertebrates*. Buglife, https://cdn.buglife.org.uk/2019/07/Managing-Soft-Cliffs-for-Invertebrates_Summary.pdf

Witte, J.P.M., Runhaar, J., van Ek, R., van der Hoek, D.C.J., Bartholomeus, R.P., Batelaan, O., van Bodegom, P.M., Wassen, M.J. and van der Zee, S.E.A.T.M. (2012) An ecohydrological sketch of climate change impacts on water and natural ecosystems for the Netherlands: bridging the gap between science and society. *Hydrological Earth System Science*, **16**, 3945–3957.

Ruggiero, P., Hacker, S., Seabloom, E. and Zarnetske, P. (2018). The role of vegetation in determining dune morphology, exposure to sea-level rise, and storm-induced coastal hazards: a US Pacific Northwest perspective. In *Barrier Dynamics and Response to Changing Climate* Springer, Cham., pp. 337–361.

Zsamboky, M., Fernández-Bilbao, A., Smith, D., Knight, J. and Allan, J. (2011) *Impacts of Climate Change on Disadvantaged UK Coastal Communities*. Joseph Rowntree Foundation, www.jrf.org.uk/file/41289/download?token=lHnih1Q1&filetype=download