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Terrestrial biodiversity climate change impacts report card technical paper

10. Implications of climate change for coastal and inter-tidal habitats in the UK

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Executive summary

- Coastal habitats are complex, dynamic and interdependent. They are important in providing sea defences, areas for recreation, biodiversity and a range of other ecosystem services.

- Increased air- and sea-surface temperatures have resulted in changes in the distribution of marine and coastal species. Both warmer- and colder-water species are shifting northwards. However, warmer-water species are shifting northwards faster than colder-water species are retreating, resulting in changes in community composition. Changes in the abundance of keystone taxa can cause a cascade of responses, further altering community composition.

- Changes in the phenology of coastal species have been observed, with the rates of change in marine species being considerably greater than those in terrestrial and freshwater systems. Recent advances in the phenology of species have not all occurred at the same rate, in some cases resulting in mismatches of timing of annual cycles of animals and their food organisms.

- Changes in precipitation are likely to affect coastal habitats, but the projected increase in winter rainfall and decrease in summer rainfall will tend to have opposing effects; the net result of these is not known. High winter rainfall and milder winter temperatures may extend the growing season and lead to faster succession and dominance by taller competitive plant species. This will be exacerbated by anthropogenic nutrient enrichment. However, increasing frequency and severity of summer droughts may counteract the effects of nutrient enrichment and winter precipitation. Increased drought will have impacts on habitats that are highly dependent on the maintenance of hydrological regimes, such as machair lochs and dune slacks.

- Rising sea levels have been associated with the loss of coastal habitats. Predicted future rises will have significant impacts on coastal and intertidal habitats, including changing geomorphological processes, further habitat loss and increasing the vulnerability of infrastructure. However, coastal systems are dynamic and have the potential to adapt to rising sea levels, but only if there is an adequate supply of sediment to allow accretion *and* if there is landward space for the coast to roll-back into. Sea defences and other coastal management interrupt the movement of sediment between systems and prevent natural coastal realignment.

- Managed coastal realignment is beneficial because it offers the potential to create habitat and provide flood defence benefits. Inevitably, there will be conflict between the need to maintain intertidal and other coastal habitats (e.g. saltmarsh, mud flat and sand dune) by realignment, and the need to protect valuable inland coastal habitats, such as grazing marsh and saline lagoons.

- Future changes in coastal habitats are hard to predict because it is difficult to separate the impacts of rising sea levels from those of coastal management, including sea defences. Coastal zone management and adaptation, and the interactions with other climate drivers, nutrient deposition and habitat management, will have significant influence on the quantity, quality and location of future coastal habitats.

1. Coastal and intertidal habitats

Coastal habitats are diverse and vary in the extent to which they are shaped by physiographic processes, such as wave action, wind, tides and sediment availability, and the relative influence of terrestrial and marine environments, e.g. tidal inundation versus groundwater levels. Coastal systems usually comprise mosaics of habitats that are functionally interdependent: for instance, saltmarsh may form behind a barrier island or shingle ridge that itself may also support a dune system; or estuaries may include a range of habitats that ultimately depend on sediment supply from the catchment and the mixing of fresh and saline waters. Coastal grazing marsh is a man-made, largely freshwater habitat, occurring landward of intertidal and coastal habitats and protected from them by natural or man-made structures. Whilst grazing marsh and other coastal habitats are not strictly functionally interdependent, there are significant conflicts between protecting grazing marsh and allowing landward movement of coastal habitats.

This review considers both intertidal and coastal margin habitats and includes:

- Hard and soft rock maritime cliffs and slopes
- Shingle structures and beaches
- Machair
- Coastal sand dunes
- Intertidal rock, including rocky shores and chalk beds
- Intertidal sediments, including sand and mudflats
- Seagrass beds
- Coastal saltmarsh
- Coastal saline lagoons
- Coastal grazing marsh.

The UK has 17,381 km of coastline, of which 17% is currently eroding and 18% has defences and artificial beaches (Ramsbottom, Sayers & Panzeri 2012). The implications of climate change for coastal and intertidal habitats have been previously covered by Marine Climate Change Impacts Partnership (MCCIP) report card reviews by (Rees *et al.* 2010) and (Mieszkowska 2010), respectively. The current review includes updated information and incorporates both coastal margin and intertidal to consider the coast as an integrated unit – changes in intertidal habitats have important consequences for coastal margin habitats. The social, economic and biodiversity implications of climate change and sea level rise in coastal and floodplain areas have been assessed by Ramsbottom, Sayers & Panzeri (2012) and Brown *et al.* (2012).

Fifteen coastal and intertidal habitats have been identified as UK BAP priority habitats (Table 1) and 20 are listed in Annex 1 of the Habitats Directive, of which five are Priority habitats (Table 1). The UK has 'special responsibility' (high proportion of European habitat area) for seven habitats under the Habitats Directive, including coastal lagoons, grey dunes and dune slacks, estuaries, vegetated shingle banks, machair and vegetated sea cliffs (McLeod *et al.* 2005).

1.1   Hard and soft rock maritime cliffs and slopes

Maritime cliffs and slopes have steep or vertical faces on the coastline where a break in slope is formed by slippage and/or coastal erosion (JNCC 2007). Wave action is the most important coastal process maintaining/eroding sea cliffs. Changes in water depth and wave energy, and factors affecting long-shore drift and current movements, together with groundwater conditions, will determine the rate of sea cliff erosion. Local geology, climate and exposure to wind and salt spray define the floral and faunal communities. Communities in sheltered locations are more similar to those found inland, whereas those in exposed situations include more salt-tolerant species. Rock type is a significant factor defining the communities of cliffs and slopes (JNCC 2007). The vertical faces and steep slopes of hard rock cliffs can be colonized by lichens and lower plants, but ledges and crevices support higher plant communities. The vegetation of soft cliffs is characterised by

successional processes, with areas of recent slippage colonised by early successional communities and woodland potentially colonising stable areas (Hill *et al.* 2001). Frequent slumping and slippages provide important habitat for specialist invertebrates requiring soft substrates in open conditions; seepages and springs associated with soft cliffs are also important invertebrate habitats (Whitehouse 2007). Whilst different processes affect hard and soft cliffs, they are not entirely distinct; for example some hard rock types, such as chalk, may sporadically experience massive collapses (Hill *et al.* 2001). It is estimated that there are approximately 1100 km of maritime cliffs and slopes in England, with a further 2372 km in length in Scotland and 522 km in Wales (Rees *et al.* 2010). Hard cliffs are found around the exposed coasts of the UK, mainly in the west, and soft cliffs primarily on the east and south coasts of England.

1.2   Shingle structures and beaches

Shingle beaches occur in high wave energy environments, where shingle supply and mobility are important coastal processes. It is estimated that one third of the UK coastline is fringed by shingle beaches (Sneddon & Randall 1993), but very little of this area is stable enough to support perennial vegetation. There are only 5810 ha of vegetated shingle in the UK (Jones *et al.* 2011), with c. 40% occurring at a single site, Dungeness, S.E. England (JNCC 2007). Shingle foreshores are mobile and require a continuing supply of new material. Vegetated shingle structures develop when a sequence of foreshore beaches is deposited at the limit of high tide and more permanent ridges are formed as storm waves throw pebbles high up on the beach. The vegetation communities vary with the stability of the shingle, the amount of fine material amongst the shingle, and the hydrological regime. The seaward shingle is sparsely vegetated and colonized by pioneer species, tolerant of sea spray and some erosion. In the landward areas of the shingle, where conditions are more stable, grassland, heath and scrub communities can develop. Shingle supports a number of species largely confined to this habitat, as well as those also associated with sand dune and saltmarshes (Murdock *et al.* 2010).

1.3   Machair

Machair is a distinctive sand dune formation, where sand with high shell content is blown inland from coastal dunes and beaches onto low-lying coastal plains. Machair is only found in the north and west of Scotland and in western Ireland; there are 19,000 ha of machair in Scotland (Jones *et al.* 2011), comprising c.70% of the global resource (Pakeman 2012). Short-turf communities form that are typical of sandy, calcareous grasslands and these vary according to agricultural history, water-table fluctuation and sand supply (Maddock 2008). These species-rich grasslands are dependent on their continued use for low-intensity agriculture (Pakeman 2012). Few plant species are restricted to machair, but several pondweeds are strongly associated with machair lochs.

1.4   Coastal sand dunes

Coastal dunes form when sand grains are blown inland and are deposited in the lee of obstacles, such as pebbles or tidal debris, and where wind speeds are reduced by the geometry of the growing mounds. Specialised plants are able to colonize the mobile sands, encouraging further sand deposition. Dune ridges often migrate landward in the direction of the prevailing wind and, providing there is sufficient supply of sand, new ridges of dunes form to seaward, providing protection to the older dunes; increasing sand stability allows vegetation typical of later successional stages to colonize. Dune slacks are seasonally fluctuating wetland habitats, often in depressions between ridges that are important habitats for many scarce invertebrates. Dune systems are quintessentially dynamic and provide a diverse range of microhabitats. However, disruption to sediment availability and transport, particularly by coastal defences, can lead to erosion. Dunes occur widely around the coast of the UK, totalling 71,000 ha (Jones *et al.* 2011), but the largest area is found in Scotland. Approximately 17% of dune area is estimated to have been lost between 1945-2010, and habitat loss or deterioration from development pressures, such as golf courses and residential areas, continues (Jones *et al.* 2011).

1.5   Seagrass beds

Seagrasses are marine flowering plants with long, narrow, ribbon-shaped leaves. Four species of seagrass occur in the UK, two species of tasselweeds (*Ruppia* spp.) and two species of eelgrass (*Zostera*) . Seagrass beds are sub- and inter-tidal habitats, occurring around the coast of the UK on soft sediments where there is protection from wave action, such as in inlets and estuaries. Seagrass beds can capture and stabilise sediments, although this is dependent on the size of the bed with small seagrass patches not trapping and retaining sediment (Wilke *et al.* 2012). Seagrass beds are important habitats for a range of species and are utilised as nursery grounds for fish and cephalopods (OSPAR Commission 2009). Increased turbidity, nutrient loading and disease are causing degradation of the seagrass beds (OSPAR Commission 2009). There are approximately 6000 ha of seagrass beds in the UK.

1.6   Intertidal rock

Intertidal rocky habitats usually occur on more exposed coasts around the UK, although some estuarine intertidal rock occurs in the north-west. The extent of the UK's intertidal rock resource is not known. Along the high energy coastline of the south-west and west, rocky shores are exposed to considerable wave energy. Intertidal chalk beds are a rare habitat in Europe and those in the south and east of England comprise over 50% of Europe's resource (Maddock 2008). The zonation of plants and animals on rocky shores is driven by the degree of tidal inundation, wave action and species interactions such as competition and predation (Little & Kitching 1996). A number of non-native species has established on intertidal rocky habitats, including the non-native Pacific oyster, *Crassostrea gigas* and a vigorously growing Japanese kelp, *Undaria pinnatifida* (Oakley 2007; Mieszkowska 2010). Rocky shores of all types are also vulnerable to changes in wave and tidal energy and, in some areas, by infilling (Maddock 2008).

1.7   Intertidal soft sediments

Intertidal mud and sandflats occur in estuaries and in coastal areas sheltered by inlets, bays and offshore islands or bars, where low wave energies allow the build-up of sediment. Sufficient accretion of sediment may raise the elevation in the tidal frame of mudflats and allow colonization by saltmarsh plants. Whether a coastline is prograding (marine regression) or retreating (marine transgression) is dependent on the balance between the rate of relative sea level rise and the supply of sediment. The sources of sediment can be fluvial (riverine) or marine in origin, with the dominance of these sources varying between systems. For example, fluvial sources dominate the sediment budget of the Fal Estuary (Ranwell 1974), but in contrast, the supply of sediment to the Ribble Estuary from the River Ribble are small and most of the accreting material is derived from reworking of glacial sediments in the Irish Sea (van der Wal, Pye & Neal 2002). Sediment can be prevented from accreting, or be re-suspended, by waves or wave-induced currents; in exposed areas, powerful waves can cause erosion of larger blocks of material. Intertidal sediments support assemblages of benthic invertebrates. Sediment grain size is an important determinant of the benthic community composition and changes in sediment type, as a result of altering coastal energies, may have important effects (Mieszkowska 2010). Mud and sandflats support internationally important wintering populations of waders.

1.8   Coastal saltmarsh

Coastal saltmarshes occur in the upper intertidal, usually between the levels of mean high water neap and mean high water spring tides (Adam 1990). Saltmarshes usually extend from the landward edge of mud or sandflats, where there has been sufficient accretion to allow the colonization by pioneer salt-tolerant (halophytic) plants, such as glassworts *Salicornia* spp. (Little 2000); such accretion of sediment usually occurs in locations sheltered from wave action, such as behind shingle bars and fringing estuaries. Salt marsh plant species are highly adapted, rarely being found in other habitats, and the distribution of saltmarsh plant communities depends on a number of environmental factors, the most important of which is elevation in relation to the tidal frame. The high marsh is subject to infrequent inundation and as a result salinity conditions are more variable

than on low marsh, with periods of low rainfall and evapotranspiration causing hypersalinity. There are currently 40,522 ha of salt marsh in England and Wales (Phelan, Shaw & Baylis 2011; area calculated from 2006-2009 aerial images). Soft sediments are highly erodible and changes to local hydrology and geomorphology, wave action and sediment supply can result in salt marsh loss.

1.9   Coastal saline lagoons

Coastal saline lagoons are areas of shallow, saline or brackish water, separated from the sea by sandbanks, shingle or, less frequently, rocks (Maddock 2008). While water is retained within the lagoon during periods of low tide, there is water exchange between the lagoon and the sea via percolation through the barrier or via overtopping. Lagoons are important for a range of specialist invertebrate species (McLeod *et al.* 2005), although the community occurring in each lagoon is dependent on the salinity and substrate. Saline lagoons are vulnerable to changes in salinity and water quality, and infilling due to rolling back of the coastline. There are approximately 5500 ha of saline lagoon in the UK (Jones *et al.* 2011).

1.10   Coastal grazing marsh

Coastal grazing marsh is seasonally or periodically inundated pasture with drainage ditches to maintain the water levels that contain brackish or fresh water. Areas are protected from tidal inundation by sea walls and regular tidal inundation does not occur, although rare flood events may occur due to overtopping. Most of the pasture is improved and species-poor. In contrast, the ditches are often very species-rich and contain many rare and scarce species (Drake *et al.* 2010). Over 300,000 ha of grazing marsh occur in the UK, but this includes floodplain grazing marsh and the area of coastal grazing marsh is not known (Maddock 2008); furthermore this area includes habitat with inappropriate hydrological management and/or low biodiversity interest – only 18% of England's coastal and floodplain grazing marsh was considered to have high biological value (Mountford *et al.* 1999). Coastal grazing marshes are vulnerable to conversion to arable usage, eutrophication and increases in salinity due to more frequent flooding events (Drake *et al.* 2010).

2.   Changes in sea level – 1900-2100

Sea level around the UK rose by about 1.4 mm yr^{-1} over the 20th century (Woodworth *et al.* 2009). Sea-levels are projected to continue to rise relative to land surface, particularly in the south and east of England where the land is subsiding. By 2095 relative sea level is projected to have increased in London by 21–68 cm and in Edinburgh by 7–54 cm (5th to 95th percentile, medium emissions scenario) (UKCP09; Jenkins *et al.* 2009). High temperatures are resulting in record melting of freshwater ice sheets and glaciers, e.g. the Greenland ice sheet (Harper *et al.* 2012). This could cause mean sea levels to rise by up to 2 m by 2100, but rises of this level are currently considered very unlikely (Jenkins *et al.* 2009), and pore spaces in the ice sheets may hold melt water, buffering (possibly for a number of decades) the effects of climate warming on sea level rise (Harper *et al.* 2012).

In contrast to previous projections, there is a prediction of a slight weakening in storminess. Increases in storm surge height in the southern North Sea are less likely than previously thought, and projected increases around the UK are small and in many places can be accounted for by natural variability. Seasonal mean and extreme waves are generally expected to increase slightly to the SW of the UK, reduce to the north of the UK and experience little change in the North Sea, although there are large uncertainties (Jenkins *et al.* 2009).

3.   Temperature-induced changes (e.g. range shifts, colonizations, phenology)

Increased air- and sea-surface temperatures have resulted in changes in the range sizes and distribution of a range of terrestrial, aquatic and marine organisms (Hickling *et al.* 2006; Hawkins *et

al. 2009). The majority of the range shifts in marine and coastal species have been observed in rocky inter- and sub-tidal species, where the availability of long-term data is greater. Warmer-water species ('southern' species) are shifting northwards; for example in recent decades the molluscs *Osilinus lineatus* and *Gibbula umbilicalis* have extended northwards and have increased in abundance (Mieszkowska *et al.* 2006). There is also moderate evidence (and moderate consensus) that cold water, northern, species have declined in abundance (Hawkins *et al.* 2009). However, the southern ('trailing') range margins of northern rocky shore species have changed less than the northern ('leading') margins of southern species (Helmuth *et al.* 2006). This may be because many northern species reach their southern limits further south in Europe than in the UK, or that there are more southern species overall (Hawkins *et al.* 2009). Marine ectothermic species tend to occupy the extent of latitudes within their thermal tolerance limits and therefore range shifts at both the trailing and leading edge are more predictable (Sunday, Bates & Dulvy 2012). In contrast, terrestrial species do not fully occupy the most southerly latitudes within their tolerance range, and therefore the trailing edge does not move north with changing climate in such a predictable manner (Sunday, Bates & Dulvy 2012).

Species will differ in the speed with which they colonize new areas following climate changes. Species with planktonic larval stages are relatively good dispersers, with dispersal distances of up to 100 km per generation (Gaines *et al.* 2007). In contrast, dispersal distances of seaweeds range from several metres to 10 km (Gaines *et al.* 2007). Propagules of saltmarsh plants are transported over short distances by tidal waters, but many of these species are also thought to be poor long-distance dispersers (Wolters *et al.* 2008). Dispersal of rare species of isolated coastal lagoons, such as sea anemones, may be very limited. Our understanding of the dispersal distances of invertebrate groups is generally very poor. However, the complex topography and heterogeneous geology of the UK coastline means that habitat patches are usually separated by long stretches of unsuitable habitat and this is likely to slow the dispersal of species northwards.

Where southern and northern species with similar niches co-occur, there has been a relative increase in the abundance of the southern species (Helmuth *et al.* 2006). *Coelopa pilipes*, a coastal strandline fly with a southern distribution, has expanded its UK range northwards and become more abundant; however, the similar, previously dominant, northern species has declined in abundance (Edward *et al.* 2007). Increasing temperatures have been demonstrated to provide *C. pilipes* with a competitive advantage (Edward *et al.* 2007). Coastal plants with northerly distributions, such as saltmarsh flat-sedge *Blysmus rufus*, Baltic rush *Juncus balticus*, curved sedge *Carex maritima*, oysterplant *Mertensia maritima* and the grass *Puccinellia distans* ssp. *borealis,* might be expected to suffer a contraction of range; conversely Mediterranean-Atlantic halophytic species in southern Britain might be expected to move north, including shrubby sea-blite *Suaeda vera*, golden samphire *Inula crithmoides*, perennial glasswort *Sarcocornia perennis*, sea heath *Frankenia laevis* and sea purslane *Atriplex portulacoides* (Preston, Pearman & Dines 2002). Sea lavender (*Limonium vulgare* and *L. humile*) is an important component of English saltmarshes that might extend its range northwards. So might common cord-grass *Spartina anglica*, which has been invasive in estuaries over the last century in southern Britain. Sea purslane is potentially the physiognomic dominant of saltmarshes and has been found to dominate some newly created managed realignments rapidly (Mossman, Davy & Grant 2012); expansion of this potentially dominant species may lead to a shift in community structure.

Changes in the abundance of keystone taxa can cause a cascade of responses, greatly altering community composition (Schiel, Steinbeck & Foster 2004). These changes are unpredictable and may result in changes in ecosystem functioning. For example, a northern limpet *Patella vulgata* has been shown to have a stronger role in controlling macroalgal abundance than the southern species, *P. depressa*. Increases in the abundance of *P. depressa* as the climate warms may result in a decrease in grazing control and changes in the abundance of macroalage, resulting in changes in net primary production and ecosystem functioning (Moore, Thompson & Hawkins 2007). Rocky shore grazer communities are more diverse in warmer southern waters than in the north of the UK (Southward, Hawkins & Burrows 1995); in contrast, fucoid algae diversity is greater in colder waters.

As southern species expand north, increases in the range of grazer feeding mechanisms, resulting from a more diverse grazer community, may reduce the abundance and diversity of fucoid algae (Hawkins *et al.* 2009). Reduction in algal cover could to lead to lower biodiversity and productivity (Hawkins *et al.* 2009), because fucoids provide shelter and protection to a range of other organisms and decomposing algae can aid productivity on nearby sediment beaches and strandlines (Kendall *et al.* 2004).

Warmer temperatures have also resulted in changes in the phenology of a range of species (Thackeray *et al.* 2010), with the rates of change in marine species being considerably greater than those in terrestrial and freshwater systems (Moore, Thompson & Hawkins 2011). The southern species of limpet *Patella depressa* has advanced the date of its reproductive development, lengthened its reproductive season and more of the population are now reproductively active (Moore, Thompson & Hawkins 2011). In contrast, *P. vulgata* has delayed the timing of its development, and there have been more years of reproductive failure and a reduction in the proportion of the population reaching advanced reproductive stages. Recent advances in the phenology of many species have not occurred at the same rate between trophic groups, with disparity increasing at higher trophic levels (Thackeray *et al.* 2010). For example, rising seawater temperatures have advanced the timing of spawning of the intertidal bivalve *Macoma balthica* but the timing of the phytoplankton bloom is not related to temperature, resulting in a mis-match in timing between the bivalve and its food source (Philippart *et al.* 2003).

4. Effects of changes in rainfall

Increases in winter rainfall and heavy rainfall events, and decreases in summer rainfall, are currently projected in the UKCP09 and other scenarios. However, there are a number of sources of uncertainty in these projections, for example the effects of changes in sea ice extent on the jet stream position (Rowell 2012). Nevertheless, it is likely that some changes in precipitation patterns will occur as a result of climate change.

Increased winter rainfall and heavy rainfall events will result in higher run-off into receiving water courses and estuaries. This will result in higher turbidity, higher nutrient loads, more frequent flushing of sewer outflows and higher inputs of other contaminants (Wilby *et al.* 2006; Callaway *et al.* 2012). It is currently estimated that more than 35 million m^3 y^{-1} of untreated sewage are discharged from stormwater overflows into the Thames (Defra 2007); increases in storm frequency will increase frequency of storm overflow, and increases in river water temperature will further reduce dissolved oxygen levels (Defra 2007). Eutrophication of coastal waters can result in harmful algal blooms and hypoxia. Although policies are now in place to reduce nutrient loading of water courses, concentrations remain high (Maier *et al.* 2009) and the effects of higher nutrient loading of coastal waters are likely to be exacerbated by higher sea temperatures, which can enhance algal blooms and further reduce dissolved oxygen. High turbidity may lead to a reduction in eelgrass (*Zostera*) beds. Increased winter rainfall and intensification of heavy rainfall events will result in higher freshwater flows into estuaries and more flooding events. Heavy rainfall can have effects on estuarine infaunal communities, but impacts are site-specific and short-lived (Ford, Anderson & Kelly 2007). However, intensification of flooding events can have significant effects on the structure and functioning of estuarine benthic communities (Cardoso *et al.* 2008). Coastal lagoons are particularly sensitive to changes in salinity and long periods of heavy rainfall and low evaporation, which cause reductions in salinity, could cause local extinction of specialist invertebrate species.

Warmer, drier summers will lead to low flow conditions in many rivers and estuaries. Low flow, or more extreme drought conditions, result in high water temperatures and low dissolved oxygen. Such conditions can have significant impacts on estuarine fauna. For example conditions in the River Thames during a major drought event led to an increase in abundance of shore crab *Carcinus maenas* and a decrease in brown shrimp *Crangon crangon* (Attrill & Power 2000); such changes in abundance have knock-on effects on estuarine food webs. High water temperatures and low

dissolved oxygen have resulted in a delay in the upstream migration and poor estuary survival of adult Atlantic salmon *Salmo salar*. Future summer climate may to lead to a shift to either earlier or later run timing; however, freshwater reaches of rivers in the south-west of England may become less suitable for salmon in the future (Solomon & Sambrook 2004). Drought will also result in higher estuarine salinity and more saltwater penetration upstream. This has significant impacts on brackish and freshwater assemblages (Attrill, Rundle & Thomas 1996). Low flow in some estuaries may be exacerbated by increases in water abstraction upstream.

Increased winter rainfall could lead to softening of the surface of cliffs and, coupled with potentially higher water tables, resulting in higher rates of cliff erosion (Greenwood & Orford 2008). In addition, increased winter rainfall may lead to more frequent summer landslips as a result of groundwater movement. Desiccation during prolonged periods of summer drought may lead to the formation of crumbly crusts of some cliff materials (e.g. clays), which are then easily removed by subsequent rainfall (Greenwood & Orford 2008). Reduced summer rainfall could lead to increases in salinity in some coastal habitats. In free-draining habitats, such as coastal cliffs, there may be an increase in the proportion of halophytic plants; growth of less salt-tolerant species may be limited. If the cliff is relatively stable, high winter rainfall and milder winter temperatures may extend the growing season and lead to increases in competitive plant species, particularly dominant grasses. This is exacerbated by anthropogenic nutrient enrichment; although there are distinctive plant communities associated with nutrient-rich sea-bird nesting cliffs (Rodwell 2000). Most coastal habitats, including dunes, vegetated shingle and saltmarshes, are nutrient-limited and increases in nutrient availability result in changes in community structure; empirical critical nutrient loads for coastal and marine habitats are estimated by Bobbink & Hettelingh (2011). On saltmarshes, nutrient levels above critical loads will result in an increase in late-successional species and dominance of graminoids, such as *Elytrigia atherica* (van Wijnen & Bakker 1999; Bobbink & Hettelingh 2011). Dunes may become more stable and densely vegetated, with an increase in tall grasses (Bobbink & Hettelingh 2011); this will reduce floral diversity (Jones *et al.* 2004) and make the habitat less suitable for thermophilic invertebrates requiring bare substrates and open swards (Howe, Knight & Clee 2010). However, increasing frequency and severity of summer droughts may counteract the effects of nutrient enrichment and winter precipitation, and the balance of these opposing effects is currently unknown. Water retention is particularly limited in shingle structures and extended dry periods in spring could affect their vegetation adversely (Davy, Willis & Beerling 2001).

Dune slack communities are characterized by seasonally fluctuating water tables; they may be seasonally flooded in winter but dry out summer, depending on local hydrology (Davy *et al.* 2006). They are important habitats for many scarce invertebrate species (Buglife 2004). Dune slack communities are highly dependent on the maintenance of their hydrological regime. Low summer rainfall, coastal erosion and the narrowing of the dune system may lead to drying of dune slacks. In contrast, higher winter rainfall and rising sea-levels may increase groundwater levels (Davy *et al.* 2006). The balance of these effects is likely to be site-specific and dependent on factors, such as wider coastal morphology, coastal defence and water abstraction. Even subtle differences in hydrology result in significant changes in dune slack vegetation (Davy *et al.* 2010). However, at some sites water tables may fall drastically, e.g. at Ainsdale, north-west England, falls in water tables of 1 m are predicted by 2100 (Clarke & Ayutthaya 2010).

Warmer, drier summers are predicted to increase visitor numbers to coastal habitats (Coombes, Jones & Sutherland 2009). This may lead to small declines in the dune vegetation cover and diversity, concentrated in areas close to site entrances. Breeding birds, such as ringed plover, may experience an increase in disturbance (Coombes, Jones & Sutherland 2008). However, the warmer conditions may promote participation in low impact activities, such as paddling and sunbathing, which may limit the pressure caused by additional visitor numbers (Coombes & Jones 2010).

5. Effects of sea level rise

Predicted rises in sea level will have significant impacts on coastal and intertidal habitats, including changing geomorphological processes, losses of habitat and increasing the vulnerability of infrastructure.

5.1 Effects of sea level rise on cliffs and rocky shores

Soft rock cliffs are particularly vulnerable to changes in sea level and wave action and the majority of the UK's soft cliffs are located in the south and east of England where relative sea level rise will be highest. Erosion or recession rates of soft cliffs can be very high, particularly on the south and east coast of England; for example, cliffs on the Suffolk coast had long-term (1883–2010) retreat rates of 3.5 m yr^{-1}, rising to 4.7 m yr^{-1} between 1993 and 2010 (Brooks, Spencer & Boreham 2012). Retreat rates of soft rock cliffs are dependent on the interaction of marine processes at the cliff base, internal hydrology and rainfall patterns. High water levels, including storm surges, and large waves cause cliff-base erosion; although there is currently no predicted increase in storm surge frequency or wave height in the southern North Sea. Wider, higher beaches that front cliffs reduce cliff retreat rates by dissipating more wave energy (Lee 2008). Relative sea-level rise is likely to result in narrower, shallower beaches and may increase the retreat of soft cliffs. Future variation in beach levels, resulting from long-term sediment availability will be an important determinant of cliff recession rates (Lee 2008). Soft cliff erosion rates are extremely varied but the majority of loss occurs in winter months (Greenwood & Orford 2008). Rainfall, particularly sequences of wet weather, can reduce material shear strength, triggering slippages and retreat (Brooks, Spencer & Boreham 2012). Projected increases in winter precipitation are likely to coincide with periods when cliff base erosion is high, potentially resulting in higher recession rates (Brooks & Spencer 2012).

Soft cliff erosion is, however, an important supply of sediment for other coastal habitats. The soft cliffs of Suffolk currently release 178,000 m^3 yr^{-1} of sediment (Brooks & Spencer 2012). Under higher rates of sea level rise, approximately 460 ha of soft cliff habitat in Suffolk could have been lost by 2095, resulting in sediment inputs to the North Sea of 300,000 m^3 yr^{-1}. Adequate sediment supply will be required by other coastal features to keep pace with rising sea levels; inputs from the accelerated eroding of cliffs may allow some areas of the coast to prograde (Dickson, Walkden & Hall 2007).

There are few studies of the impact of sea level rise on 'hard' cliffs and shores, although it is assumed that rocky substrates will erode more rapidly with higher sea levels because the deeper water reduces wave attenuation (Trenhaile 2011). Very resistant rocks experience less erosion; shoreward displacement will be determined by the slope of the land (Trenhaile 2011) and the potential for block failure, which will vary with geology. Whilst not currently predicted, changes in wave patterns and energy, and the intensity and frequency of storms would have the greatest effects on the erosion rates of hard rock cliffs (Trenhaile 2011).

There are few studies documenting the response of communities of intertidal rock to rising sea levels. Macrotidal rocky shores where the slope of the seashore continues uniformly well above the present level of high water or that are backed by eroding soft cliffs will be less affected by coastal squeeze, because the zones will be able to retreat (Kendall *et al.* 2004). However, as sea-level rises there may be a shift towards a greater availability of vertical rocky surfaces compared to horizontal, leading to major change in assemblage from the current barnacle and turf-forming algae, to coralline algae and grazing gastropods (Vaselli *et al.* 2008). Rocky shores backed by steep inclines, such as hard cliffs, will be unable to retreat and the rocky intertidal area will decrease (Jackson & McIlvenny 2011).

5.2 Effects of sea level rise on sedimentary coastlines

Almost two-thirds of the intertidal profiles in England and Wales have steepened over the past hundred years (Taylor, Murdock & Pontee 2004). Eighteen percent of the UK coastline is lined with

defences or artificial beaches, rising in the south and east of England (54% in south-east England and 100% in the East Midlands (Ramsbottom, Sayers & Panzeri 2012)). These defences prevent coastal habitats naturally migrating inland in the face of rising sea levels. Such coastal squeeze is particularly important in the south and east of the UK, since coasts in these areas are most heavily defended and relative sea level rises are greatest due to isostatic land subsidence. However, it may be that isostatic uplift is now being outpaced by rising sea levels in Scotland (Rennie & Hansom 2011).

Saltmarsh extent in England and Wales has recently been calculated at 40,522 ha (Phelan, Shaw & Baylis 2011). However, comparisons of this figure with previous estimates (e.g. Burd 1989) are problematic because of considerable differences in methodology. The net annual change in saltmarsh extent between 1989 and 2006-2009 has been estimated between +1 and -82 ha per year (Phelan, Shaw & Baylis 2011). There have been significant losses in intertidal habitat in some areas; for example, up to 50% of saltmarsh area along stretches of the south coast of England was lost between 1971 to 2001 (Baily & Pearson 2007) and between 17% and 59% lost in estuaries in the south east of England (Cooper, Cooper & Burd 2001). Saltmarsh loss has occurred through reclamation, erosion of the marsh frontage (i.e. creating a cliff edge) or through widening of existing channel (internal dissection). These losses have, in part, been attributed to rising sea levels, particularly rising high and extreme water levels (van der Wal & Pye 2004), and coastal squeeze (Morris *et al.* 2004). However, land claim and embankment construction cause increased tidal range and current velocities, which are contributing to coastal erosion (van der Wal & Pye 2004). There is also limited evidence that other factors, such as herbicide loads reducing diatom abundance and in turn sediment stability (Mason *et al.* 2003), are contributing to erosion. Upper intertidal zones, e.g. upper saltmarsh communities, are at most threat from coastal squeeze, but areas of these zones are already small (Burd 1989) because steeply sloping sea defences provide little area at suitable elevations. Upper saltmarsh and driftlines are particularly important for rare coastal invertebrates (Buglife 2004).

Predicted future losses in saltmarsh and mudflat are significant. For example, model simulations predict that a sea level rise of 30 cm would result in the loss of approximately 7% of the intertidal area in the Humber estuary; leading in turn to a 7% loss in the total biomass of macrobenthic invertebrates (Fujii & Raffaelli 2008). Modelling on a larger scale suggests that marshes with higher tidal ranges are less vulnerable (Simas, Nunes & Ferreira 2001). Coastal habitats in the UK are of international importance to wintering populations of waders and small losses of habitat could have major negative impacts (Durell *et al.* 2006). Loss of saltmarsh area will affect terrestrial invertebrate assemblages adversely, particularly those of the upper marsh that have previously suffered significant losses in habitat area (Irmler *et al.* 2002).

Erosion and accretion are dynamic natural processes that potentially allow coastlines to adjust to changes in sea level. Saltmarshes in the Greater Thames and Severn estuaries have accreted vertically at rates in line with sea level rise and have been able to maintain their position in the tidal frame (Allen & Duffy 1998; van der Wal & Pye 2004). In other areas, such as the Dee Estuary, saltmarsh has expanded because sediment accretion has been greater than relative sea level rise (Huckle, Marrs & Potter 2004). Vertical accretion and horizontal expansion of coastal habitats, including beaches, requires an adequate supply of sediment. It has been suggested that the most serious issue facing coastal habitats in the future may be the reduction in sediment availability (Orford & Pethick 2006). However, accurate assessment of both sediment inputs and demands is very challenging. An assessment of the southern North Sea indicated that there may be sufficient sediment availability to meet current and near-future demand (Orford & Pethick 2006), although increases in the rate of sea level rise will substantially alter the sediment balance.

Eroding coastlines are important sources of sediment and increases in the rate of soft cliff recession may make important contributions to overall sediment supplies, particularly in the south and east of England. Protection works to stabilize cliffs or slow the rate of erosion may protect the cliffs, but they can significantly reduce sediment inputs into the wider coastal system (Clayton 1989). Hard

defences, such as sea walls and revetments are designed to reduce coastal erosion, but they prevent the release of sediment into the system, interrupt longshore drift and can result in erosion elsewhere. The construction of hard defences in the Mersey estuary resulted in major changes in the morphology and sediment transport dynamics of the system and across a wider area of dune coastlines to the north (Blott *et al.* 2006). Beach nourishment schemes may be used to increase beach volume, width and height, and therefore capacity to dissipate wave energy (Pye & Blott 2006). However, the source of nourishment material is critical; dredging extraction license applications are subject to extensive scrutiny prior to permission, but dredging of sand and gravel from locations important for supply of sediment to coastal systems could result in negative sediment budgets and habitat erosion elsewhere.

There is considerable uncertainty regarding future changes in storm surge and extreme wave frequency. Prevailing wind and wave action are known to be an important factors in the erosion of coastal habitats (e.g. van der Wal & Pye 2004; Pontee 2011), and relatively small changes in the direction and intensity of wave action may have considerable effects. However, there is insufficient understanding of how this might change in the future. Large storm events can trigger dune erosion, but the response is unpredictable and other factors, such as the timing and duration of storm events and the seasonal variability in dune morphology, and the ability of the system to restore dune formation, are important (Esteves *et al.* 2012).

6. Managing coasts to enable adaptation to change

Dynamic coastal systems have the potential to be self-regulating in the face of rising sea levels but only if there is an adequate supply of sediment to allow accretion *and* if there is landward space for the coast to roll-back into. There are moves to manage coasts at scales defined by boundaries related to the movement of sediments (de la Vega-Leinert & Nicholls 2008) and management plans are increasingly incorporating schemes that retreat sections of coast landward. Such managed coastal realignment usually occurs on low-lying areas of coast that are vulnerable to flooding, where sea defences are no longer economically viable to maintain. The coastal defence is realigned landward and the existing seaward wall breached to allow tidal inundation. Current coastal realignments have focussed on fine sediment habitats with the creation of saltmarsh and mudflat. However, coastal management aimed at reconnecting sand dunes and shingle structures to their source areas would also have considerable long-term benefits (Orford & Pethick 2006).

Managed realignment (MR) is carried out on sites of historic land claim and the surface elevation relative to current sea level will depend on the period since reclamation for agricultural use, during which there will have been shrinkage and consolidation of the sediments (Crooks *et al.* 2002). Sites low in the tidal frame at the time of reactivation may be subject to rapid accretion of sediment once tidal inundation commences, for example rates of up to 41 mm yr^{-1} were found at Tollesbury MR (Garbutt *et al.* 2006). Managed realignment of low lying land, particularly in estuaries with high sediment loads, is therefore capable of trapping available sediment for future habitat provision (Orford & Pethick 2006). However, coastal realignment increases the tidal prism of the estuary and can modify tidal flow both locally to the site and in the estuary as a whole (French 2006). Increased tidal flows result in the widening and deepening of creek systems and can lead to the erosion of existing marsh. Large schemes can cause major changes in the tidal volume of the estuary and increase tidal scour (French 2006). However, some schemes, such as at Alkborough on the Humber, have been specifically designed to provide flood storage for extreme events (ABPmer 2004).

Elevation in relation to the tidal frame is the major determinant of saltmarsh plant colonization and, at low-lying sites, a period of accretion is usually required. Once site elevation is suitable for colonization, halophytic species can quickly establish and the percentage of the species pool found on an MR site can be similar to those found on natural reference marshes after just one year (Mossman, Davy & Grant 2012). However, the plant communities of MR sites are dominated by

pioneer and low marsh species. This may be because most MR sites were restored less than ten years ago and the communities indicate the early-successional state (Mossman, Davy & Grant 2012), for example Hughes, Fletcher & Hardy (2009) found that pioneer *Salicornia* communities were replaced by later successional species within five years. Areas of saltmarsh that have developed after sea defences were accidentally breached during storm surges and remained unrepaired can indicate the development of restored saltmarsh over longer time periods; most of these accidentally restored sites are more than 50 years old. Plant communities of these older restored sites are also different from those on natural marshes (Garbutt & Wolters 2008; Mossman, Davy & Grant 2012). Characteristic species of mid marsh levels, such as *Limonium vulgare*, *Armeria maritima*, *Triglochin maritima* and *Plantago maritima* are scarce on restored sites of all ages (Mossman *et al.* 2012; Mossman, Davy & Grant 2012); the lower abundance of these species is likely to lead to absence of specialist invertebrate species that are exclusively dependent on them (Agassiz 2000).

The sediments of restored saltmarshes have been found to be less oxygenated than those at corresponding elevations of natural marshes (Davy *et al.* 2011; Mossman *et al.* 2012; Mossman, Davy & Grant 2012). Poorly oxygenated sediments may inhibit plant colonization (French 2006), and may result in greater nitrous oxide emissions than natural marshes (Adams, Andrews & Jickells 2012) and higher methane production (Ding, Zhang & Cai 2010). However, MR sites can sequester carbon and reduce estuarine nutrient loads (Adams, Andrews & Jickells 2012). Restored marshes are able to provide a range of other ecosystem services. Sites that are at appropriate elevations in the tidal frame and are sufficiently vegetated may attenuate wave action (Möller 2006). Managed realignment sites are utilised as nursery habitat and feeding habitat by fish, including commercially important species (Colclough *et al.* 2005; Fonseca, Colclough & Hughes 2011). Benthic infaunal species can quickly colonize the sediments of some restored sites, although after 5 yr differences from natural marshes remain in abundance, diversity and community structure (Mazik *et al.* 2010). Studies from the USA have indicated that infaunal community structure can take decades to develop fully (Craft 1999). Bird assemblages utilising managed realignments can quickly become similar to adjacent existing habitat (Atkinson *et al.* 2004; Mander *et al.* 2007); however, wading bird community composition is dependent on the infaunal assemblages; for example Eurasian Oystercatcher was absent from several managed realignment sites because the density of large bivalves was very low (Atkinson *et al.* 2004).

There is significant uncertainty regarding the functioning of restored marshes and the ability of sites to provide the complement of ecosystem services; this is exacerbated by a lack of knowledge regarding the functioning of natural marshes. Vegetation community development is probably the most studied aspect of managed realignments and this work has highlighted differences between the restored and natural sites. It cannot be assumed that managed realignments will provide a like-for-like replacement for natural marshes (Mossman, Davy & Grant 2012), but managed realignment does create valuable habitat and there is significant need to create secondary habitat, particularly if there is a risk of the primary habitat being lost completely in some areas. Saltmarsh habitat created by managed realignment is probably cost-effective and can provide significant benefits in terms of habitat creation, biodiversity, sediment trapping and some ecosystem services.

Rising sea levels and coastal squeeze will result in considerable conflict between the need to maintain intertidal and other coastal habitats, such as dune systems, by allowing the natural movement of coastlines and through managed realignment, and the need to protect valuable inland coastal habitats, such as grazing marsh and saline lagoons until they have been relocated or recreated elsewhere. The vulnerability of coastal wetland habitats, and the options and ease of their replacement, have been recently assessed in order to inform adaptation strategies (Defra 2011).

Some lagoons have established behind sea-walls or other artificial barriers (Bamber 2010). However, saline lagoons are ephemeral by nature and are highly vulnerable to changes in coastal morphology. Many lagoons are protected from tidal inundation by sand or shingle barriers. Policies that allow natural coastal rollback will cause losses of area unless the lagoons are able to migrate

landward and, in many cases, this is unlikely (Spencer & Brooks 2012). Sea level rises may also result in increases in the breaching and overtopping of barriers. This will lead to changes in water quality and salinity, with resulting changes in species composition. There are significant challenges in re-creating new saline lagoon habitats to replace existing ones. Lagoonal species vary in their tolerance of salinity and whilst some species can tolerate a wide salinity range, others are much more restricted (Bamber 2010). Suitable locations with appropriate balances of freshwater and saline inputs may be rare. Furthermore, some specialist species are poor dispersers (Barnes 1988).

Coastal grazing marsh is threatened by increases in salinity due to increased percolation, salinization of groundwater and flood frequency; regular flood frequencies of >1 per year will cause reversion of grazing marsh to saltmarsh (Richards *et al.* 2008). Currently, 4046 ha of coastal and floodplain grazing marsh grassland and ditches are at risk from coastal flooding under current climatic conditions; a further 473 ha are at risk by 2030 under a scenario of medium emissions and degraded coastal defences (Defra 2011). There will be intense competition between saltmarsh and grazing marsh for space (Richards *et al.* 2008). Abandonment of agricultural land due to increased flooding may result in gains in grazing marsh area, but these areas will remain highly vulnerable to further saline flooding and reversion to saltmarsh (Defra 2011). New grazing marsh habitat should be created prior to further losses occurring, particularly since major flood events are unpredictable. Large areas are currently being established through schemes such as the Environment Agency Regional Habitat Creation Programmes, particularly in The Fens (e.g. Hilgay, Norfolk; Frampton, Lincolnshire)). However, further losses will occur around the country and additional suitable sites should urgently be sought.

7. Confidence in the science

There is moderate confidence (medium evidence and agreement) that increases in sea and air temperatures are resulting in range shifts, changes in phenology and community composition. However, the majority of evidence comes from rocky intertidal habitats and there is very limited evidence regarding specialist species of other coastal habitats. There is limited evidence of the biological mechanisms behind observed and predicted changes.

Changes in precipitation are more likely than not to impact on the vegetation of coastal habitats. However, the projected increase in winter rainfall and decrease in summer rainfall will have opposing effects on most coastal habitats; the net result of these opposing effects is not known. Whilst there is moderate confidence in the potential effects of changes in rainfall on some habitats (e.g. dune slacks), there is limited evidence for others. Overall, confidence in the direction and magnitude of the effects of changes in rainfall on coastal habitats is limited.

There is robust evidence that most coastal habitats are nutrient-limited. Continued nutrient enrichment is likely to interact with, and exacerbate, some of the effects of climate change. There is moderate confidence of the interaction between nutrient enrichment and climate change in some habitats, for example increased winter rainfall causing higher nutrient laden run-off into water courses and estuaries. However, evidence in other habitats is limited.

There is moderate evidence and agreement that rising sea levels have resulted in the loss of salt marsh, mudflat and sandy beaches. There is robust confidence in the prediction that future losses of coastal habitats will occur. However, coastal systems are dynamic and it is difficult to separate the impacts of rising sea levels from those of coastal management, including sea defences. Future coastal zone management and adaptation will have significant influence on the quantity, quality and location of future coastal habitats.

8. Research gaps

In all habitats, there is a lack of long-term, integrated monitoring designed to detect the impacts of climate change.

There is insufficient information and understanding of the productivity, nutrient cycling and other ecosystem processes and services in coastal habitats; how these will respond to changing climates is therefore very poorly understood.

There is insufficient information on the interaction between changing climate and other factors that influence successional processes, such as nutrient deposition and habitat management.

The inter-relationships between complex and dynamic coastal habitats, particularly regarding sediment transport, are poorly understood.

The effectiveness of habitat restoration or creation to establish biodiversity-rich and ecologically functioning habitats is poorly understood. The ecosystem processes and functions of restored habitat have received limited attention. Research should also be targeted at making improvements to the design of restored/created habitat where limitations to the benefit have been found.

*Table 1. Correspondence between the habitats covered in this review and those of the UK Biodiversity Action Plan (BAP) and Annex I of the Habitats Directive. * denotes Priority habitat under the Habitat Directive; [†] denotes habitat for which the UK has special responsibility (high proportion of European habitat area (McLeod et al. 2005).*

Habitat	BAP	Habitat Directive
Coastal Grazing Marsh	Coastal and Floodplain Grazing Marsh	
Saline lagoons	Saline lagoons	Coastal lagoons*[†]
Maritime cliffs and slopes	Maritime cliffs and slopes	Vegetated sea cliffs of the Atlantic and Baltic coasts[†]
Shingle structures and beaches	Coastal Vegetated Shingle	Annual vegetation of drift lines Perennial vegetation of stony banks[†]
Machair	Machair	Machairs[†]
Coastal Sand Dunes	Coastal Sand Dunes	Embryonic shifting dunes Shifting dunes along the shoreline with *Ammophila arenaria* Fixed dunes with herbaceous vegetation*[†] Decalcified fixed dunes with *Empetrum nigrum** Atlantic decalcified fixed dunes (*Calluno-Ulicetea*)* Dunes with *Salix repens* ssp. *argentea* (*Salicion arenariae*) Dunes with *Hippophae rhamnoides* Humid dune slacks[†] Coastal dunes with *Juniperus* spp*
Intertidal rock	Intertidal chalk Intertidal Underboulder Communities Estuarine Rocky Habitats *Sabellaria alveolata* reefs	
Intertidal sediment	Intertidal Mudflats Sheltered Muddy Gravels Peat and Clay Exposures	Mudflats and sandflats not covered by seawater at low tide Estuaries[†]

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	with Piddocks	
Seagrass Beds	Seagrass Beds	
Coastal saltmarsh	Coastal saltmarsh	*Salicornia* and other annuals colonizing mud and sand *Spartina* swards (*Spartinion maritimae*) Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*) Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*)

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