The Impacts of Climate Change on Coastal and Intertidal habitats in the UK

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KEY FACTS

What is already happening?

- Coastal, intertidal and shallow nearshore ecosystems tend to be highly diverse and variable at local, regional and global scales. The impacts on these systems of anthropogenic activities and climate change are highly complex, cumulative and interlinked. Therefore, quantifying habitat-specific responses to changes in climate is extremely challenging.

- Increasing storm frequency and severity are altering tidal flows, exposing saltmarshes to stronger wave action and accelerating erosion. As sea levels rise, mudflats shift, low-marsh areas flood more often, and habitats migrate landward. This drives changes in plant communities and the loss of flood-sensitive species. Together, these pressures highlight how vulnerable saltmarsh habitats are to climate change.

- Climate change may be facilitating the spread of established invasive non-native rocky shore species, with recent range extensions observed in the English Channel, southern North Sea, and the Celtic Sea.

- As machair – a rare coastal grassland found only on the exposed western coasts of Scotland and Ireland – is limited to low-lying coastal areas with high winter water levels, it can be particularly vulnerable to changing weather patterns, relative sea-level rise (RSLR), and increased frequency of storm events.

- Inter- annual climate variability fundamentally influences the ecological functioning of dune slacks. Reduced rainfall limiting water-table recharge, erosive pressure from storm surges and RSLR may be capable of forcing more-significant shifts in dune slack hydrology and ecology than seasonal variation.

What could happen in the future?

- Rising sea levels and more frequent, intense storms will continue to alter coastal sediment transport, and – together with freshwater inputs and human activities – are reducing the extent of UK saltmarshes, with an estimated 11% loss projected by 2060 without putting effort into restoration. Shifts in temperature, rainfall patterns and the increasing occurrence of heatwaves also risk driving major ecological regime changes within saltmarsh habitats, altering species composition and overall ecosystem function.

- Anthropogenic and Marine Heat Wave (MHW) impacts are likely to be more intense in shallow and intertidal waters where the temperature stress is the greatest and exposed plants are likely to experience desiccation, suggesting that the intertidal seagrass *Zostera noltii* may be particularly vulnerable.

- Cumulative impacts of climate change and direct anthropogenic stressors including high suspended sediments and eutrophication are likely to increase the vulnerability of seagrass beds to climate change.

- Increased frequency and intensity of heatwave events may start to negatively affect the abundance of cold-affinity boreal species in rocky intertidal systems. Some cold-water boreal species, such as the barnacle species *Semibalanus balanoides* and *Balanus crentaus* may disappear from some rocky intertidal habitats in south-west England.

- The coincidence of storms with large tides and sediment availability may be a key factor in long-term dune loss. Rates of dune erosion are expected to continue to increase due to climate change.

- For maritime cliff and slope habitats, increased rainfall in the future may lead to increased slope failure, particularly affecting the movement of groundwater in softer lithologies. The role of RSLR in accelerating soft cliff retreat is shown in a modelling study that estimates future erosion rates three to seven times higher in North Devon and Yorkshire by 2100 based on the current predictions of RSLR. This is an increase much greater than previously predicted.

- Although the focus of restoration programmes is often on the active habitat creation, the importance of conserving existing habitats through removal of pressures is likely to bring the largest benefits for climate change mitigation and adaptation. Restoration practices should also plan beyond single species and single habitats to a multi-habitat seascape.

OVERVIEW

In the United Kingdom, coastal, intertidal, and shallow nearshore ecosystems are under increasingly severe pressure from a multitude of climate change and human-induced drivers, including:

- Relative sea-level rise (RSLR)
- Changes in the frequency and intensity of storms
- Storm surge events
- Changes in the frequency, amount and seasonal distribution of precipitation
- Changes in wave action
- Warming seawater and air temperature
- Increased frequency and duration of heatwaves and marine heat waves (MHWs)
- Ocean acidification
- Coastal erosion
- Saline intrusion
- Coastal development and changes in land-use practises
- Coastal and marine pollution, nutrient enrichment.

Ecosystem processes and functions fundamental for healthy coastal, intertidal and shallow nearshore habitats are dependent on biotic and abiotic factors such as tides, waves, winds, flora, fauna, food web interactions and sediment dynamics. These factors are susceptible to climatic change while being vulnerable to, and often negatively affected by, human activities. These pressures can lead to changes in ecosystem processes and functions as well as habitat extent and condition; they can significantly alter species distribution patterns. Important ecosystem services provided by coastal, intertidal and shallow nearshore habitats such as flood alleviation, carbon storage, water filtration, sediment stabilisation, and provision of food and nursery grounds to associated species, can also be fundamentally affected by the combined impacts of climate change and human pressures.

Coastal, intertidal and shallow nearshore ecosystems tend to be highly diverse and variable at local, regional and global scales and the impacts of anthropogenic activities and climate change are highly complex, cumulative and interlinked. Therefore, quantifying habitat-specific responses to changes in climate can be extremely challenging (Zimmerman, 2021).

Climate change impacts vary regionally across the UK coasts. Risks from extreme water levels are increasing across the UK, driven by rising sea levels, with the net rate of sea-level rise is slightly higher in the south of England and lower in some parts of Scotland (Haigh *et al*., 2022). Coastal erosion is expected to increase, in part due to sea-level rise, with low-lying and soft-sediment coasts in the east of England most vulnerable (Masselink *et al*., 2020), whereas the sediment supply and transport characteristics may continue to make some coastal habitats in the north more resilient to RSLR (Ladd *et al*., 2019).

Seasonal rainfall patterns across the UK are expected to change significantly, with increased winter rainfall particularly in the north-west, and reduced summer rainfall, albeit with the possibility of an increase in episodic summer storms[1]. Air temperatures are rising, particularly in the south, and warming sea temperatures are most pronounced in the North Sea. While Marine Heatwaves (MHW) have so far been short-lived in UK seas they are likely to intensify, and seabed ecosystems within southern North Sea and the English Channel have been predicted to become most impacted (Jacobs *et al*., 2024). Measurements of anthropogenic CO_2 in the North Atlantic are higher than for any other ocean basin (Stoker *et al*., 2020).

This MCCIP evidence update provides a review of climate change impacts on coastal, intertidal, and shallow nearshore habitats from across the UK listed below, drawing out new local and regional examples and evidence that has emerged since the separate MCCIP evidence reviews on coastal[2] and intertidal[3] habitats were published in 2020. The habitats covered in this review include:

- Saltmarsh
- Sand dunes
- Machair
- Maritime cliff and slopes
- Rocky intertidal habitats
- Intertidal soft sediments
- Intertidal and subtidal seagrass beds (*Zostera noltii* and *Zostera marina*).

Intertidal and subtidal seagrass beds were not covered in any of the 2020 MCCIP reviews and as such, a more detailed section is included here. Vegetated shingle and coastal saline lagoon habitats have not been included in this review as no new evidence or studies on climate change impacts were found for these habitats since the last MCCIP reviews in 2020. Shallow subtidal biogenic reefs are not covered in this review, as they will be addressed in the ‘Shallow, shelf and deep-sea habitats’ MCCIP review (Hill *et al.,* 2026).

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[1] Dataset Collection Record: UKCP18 Probabilistic Climate Projections (ceda.ac.uk), [Accessed on 19 January 2026]

[2] 11_coastal_habitats_2020.pdf (mccip.org.uk) [Accessed on 19 January 2026]

[3] 12_intertidal_habitats_2020.pdf (mccip.org.uk) [Accessed on 19 January 2026]

WHAT IS ALREADY HAPPENING?

Saltmarsh

Saltmarshes generally occur between mean high-water neap tides and high astronomical tides at temperate latitudes. They are widely distributed around the UK, although one third of the total extent is found at five sites (The Wash, Inner Solway, Morecambe Bay, Burry estuary, Dee estuary) (Burd, 1989). Saltmarsh extent has reduced by around 85% since the mid-1800s, principally due to historical reclamation for agriculture and more recent activities such as port development (Green *et al*., 2021).

The character and dynamics of saltmarshes are governed by the interplay between four physical factors: sediment supply, tidal regime, wind-wave climate, and sea-level change (Fagharazzi *et al*., 2012). Over the past 150 years in the UK, the vertical and lateral expansion of saltmarshes in the north and erosion in the south can be explained by variations in sediment supply and its wave-driven transport (Ladd *et al*., 2019).

RSLR and changes in temperature, precipitation and storminess can all affect saltmarshes, primarily 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. As long as there is an adequate sediment supply to maintain vertical accretion, saltmarshes can keep pace with RSLR (Ladd *et al*., 2019; Fagherazzi *et al*., 2020). UK average accretion rate has been estimated at 4.5 mm/yr, with considerable variability between sites (0.68–7.88 mm/yr), and it positively correlates with tidal range; therefore, marshes with both higher tidal ranges and suspended sediment loads are most resilient (Masselink, 2024). Increased complexity in saltmarsh creek network morphology also increases resilience to rising sea levels through improved flow concentration, sediment transport and deposition by increasing sediment accumulation rates (Cornacchia *et al*., 2024). Storm generated sediment makes an important contribution to saltmarsh accretion and expansion (Pannozzo *et al*., 2023).

However, lack of accommodation space, whether due to natural elevation in topography, reduced creek complexity, or presence of hard defences and infrastructure, can affect saltmarsh accretion rates and their ability to keep pace with RSLR (Bost, 2024). The lateral extent of a saltmarsh can be reduced as deeper water and larger waves cause erosion to the seaward edge, which could also be exacerbated by an increase in storminess. Erosion predominantly affects lower marsh communities which are more vulnerable to wave action, although mid- and high saltmarshes are susceptible to internal erosion through creek expansion. Landward migration of saltmarsh could compensate for these losses, but only in places without hard sea defences (Schuerch *et al*., 2018; Horton *et al*., 2018).

Although there is currently little evidence regarding the sensitivity of saltmarsh species to climate change (Paul *et al*., 2022), changes in temperature, precipitation and the occurrence of heatwaves have the potential to cause major ecological regime shifts within UK saltmarsh habitats and species communities. Observations and experiments in other temperate regions show a range of climate impacts, for example drought inhibiting seedling establishment (Ostertag *et al*., 2023). Where warmer temperatures and increased CO_2 have favoured invasive marsh species such as *Spartina anglica* in the Wadden Sea (Loebl *et al*., 2006; Koop-Jakobsen and Dolch 2023), reduced plant diversity and geomorphological change is negatively affecting the resistance of these saltmarsh habitats to climate change (Granse *et al*., 2021). As plant diversity is linked to soil stability (Ford *et al*., 2016), such changes may lead to increased erosion and loss of saltmarsh. Both drought and flooding have been linked to sudden dieback events in the USA, and their frequent occurrence may convert a marsh to a mudflat which may be less resilient to RSLR due to reduced capacity for wave attenuation and sediment capture (Rolando *et al*., 2023). Climate change can also affect keystone species important for ecosystem function such as cockles, which act as important bioturbators within tidal flats (Zhou *et al*., 2022). Climate-change pressures can also be exacerbated in saltmarshes with reduced resilience due to other pressures. For example, excess nutrients can cause smothering by opportunistic macroalgal blooms, biodiversity loss and community shifts towards resilient but soil-destabilising species such as *Elymus athericus* (Geoghegan *et al*., 2018; Ní Longphuirt *et al*., 2016; Valéry *et al*., 2017).

Restoration of saltmarsh, to mitigate historical and ongoing habitat losses, 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. To date, the most ambitious saltmarsh managed re-alignment and restoration scheme in the UK is the RSPB Wallasea Island Wild Coast project on the Essex coast, which aimed to transform nearly 800 ha of farmland back to wetland habitat, approximately 400 years after reclamation by the end of 2018 (ABPmer, 2018). 115 ha of this is now established saltmarsh habitat. Overall in England, a total area of c. 2950 ha of re-aligned saltmarsh had been created by 2020[4] and there are long-term plans to re-align 10% of the coastline by 2030, rising to 15% by 2060 (Adaptation Sub-Committee, 2013).

Saltmarsh habitat restoration as a 'Nature-Based Solution' is growing in popularity due to the multiple natural capital benefits associated with these habitats. These include biodiversity enhancement (Vanderklift *et al*., 2022) and coastal protection, by attenuating waves (Fairchild *et al*., 2021; Gilbertson *et al*., 2020; Möller *et al*., 2014; Hanley *et al*., 2020), reducing storm surges (Pannozzo *et al*. 2021) and minimising coastal erosion (Pétillon *et al*., 2023). Restored saltmarshes can also reduce levels of excess nutrients (Watson *et al*., 2020) and turbidity (Nardin *et al*., 2018). Saltmarshes accumulate and stabilise carbon rich sediment, depending on

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[4] OMReg Map - ABPmer [Accessed on 19 January 2026]

sedimentological conditions (Ladd *et al*., 2019; Schuerch *et al*., 2019) and the condition and management of the marsh (Dale *et al*., 2017; Mariotti and Carr, 2014; Spencer *et al*., 2008). A recent review of UK saltmarsh carbon accumulation found that restored saltmarshes sequester nearly double the amount, at an average rate of 8.2 metric ton of CO_2 equivalent per hour per year and 15 tCO2e h-1y-1 compared to natural and restored marshes, respectively (Mason *et al*., 2022). However, if rates of sea-level rise exceed the ability of the marsh to keep pace, saltmarshes will be submerged and may become sources rather than sinks of carbon (Gore *et al*., 2024). Although biodiversity and ecosystem functioning in re-aligned marsh are often incomparable to semi-natural marsh (Lawrence *et al*., 2018), knowledge of how the biogeophysical systems work is improving and enabling better site-specific design of restoration for ecosystem service benefits such as carbon storage, coastal protection and biodiversity (Dale *et al*., 2021). Restoration activities also help increase the resilience of saltmarsh habitats to the impacts of climate change, although the time scales required for a restored saltmarshes to unlock its natural capital potential are yet to be fully understood.

Sand Dunes

Coastal sand dunes are formed from sand 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 in the UK are dry dune grassland and dune slacks: a seasonal wetland, with dune heath on some acidic sites. A tiny number of sites have naturally colonised dune woodland.

Climate change can affect coastal sand dunes in several ways. These include direct loss of habitat due to coastal erosion, coupled with accelerated RSLR, 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. Indirect effects on dune hydrological regimes include coastal erosion which lowers the water table, and RSLR which raises the water table (Clarke and Sanitwong Na Ayutthaya, 2010). Specific processes sensitive to climate change include the rate and direction of sand movement, which is governed by the wind climate.

Sand dunes provide storm protection through the stabilisation of substrates, preventing coastal erosion and reducing the attenuation of wave energy and flood risk. Unlike hard defences, sand dunes are naturally dynamic and with appropriate management, capable of adapting to rising sea levels and shifts in the frequency of storm events (Hanley *et al*., 2014). However, in England and Wales, many sand dunes lack natural dynamism and approximately 35% of the total dune frontage has net erosion or is protected by hard defences, 35% have experienced net stability, and 30% have accreted seawards (Pye *et al*., 2007c). The extent of frontal dune erosion is expanding because of increased storminess and RSLR, and this may have negative impacts on the extent of some dune habitats and the effectiveness of dune systems as natural flood defences. The consequences of such changes will vary between locations, reflecting differences in natural processes and beach-dune sediment budgets (Pye *et al*., 2007c). At a regional level, studies of shoreline change have indicated that c. 50% of sand dunes in Cornwall are undergoing erosion while 21% are stable and 28% are prograding or accreting (Brodie *et al*. 2023). The coincidence of storms with large tides and reduced sediment availability was attributed as a key factor to long-term dune loss.

Dune slack habitats have been drying out in recent years in the UK, most notably in the south and west, where a 30% loss in extent was recorded between 1990 and 2012, and where the remaining habitat is showing shifts in extent and composition from wetter to drier plant communities (Stratford *et al*., 2014). Dune slacks are particularly vulnerable to changing weather patterns, as only small shifts in water table levels can result in species change (Davy *et al*., 2010; Curreli *et al*., 2013; Rhymes *et al*., 2018; Dwyer *et al*. 2021a; Dwyer *et al*., 2021b). Furthermore, even though seasonal patterns of rainfall and soil moisture content can affect both dry and wet dune slack vegetation, inter- annual climate variability has been found to have a fundamental influence on the optimal ecological functioning of dune slacks and may be capable of forcing more significant shifts in dune slack hydrology and ecology than seasonal variation (Martin *et al*., 2018; van Willegen *et al*. *in press*).

One of the key climate driven threats to sand dune biodiversity in England and Wales is the loss of structural diversity through ‘stabilisation’, which has been attributed to the combined effects of atmospheric deposition of nitrogen compounds, increases in temperature, and changes in precipitation patterns and wind speeds, promoting vigorous vegetation growth that favours species which thrive in nutrient rich conditions at the expense of slower growing dune specialists, leading to loss of diversity and stabilisation of successional communities (Pye *et al.,* 2014; Jones *et al*., 2008; 2010; Gao *et al*., 2020). The most recent assessment of the conservation status of the UK’s dunes indicated ‘bad, declining’ or ‘bad, no change’ condition, mainly due to overstabilisation and scrub cover (JNCC, 2019b,c).

Stabilisation can further reduce resilience to climate change (in already-constrained systems), since artificially tall and fixed frontal dune crests restrict the natural landward transfer of sand through which dunes adapt to rising sea levels. The inability of sand to move inland within dune complexes is likely to drive enhanced movement alongshore or offshore, gradually reducing the total volume of available sediment, and thus the capacity to adjust to changing conditions. Evidence from studies of stabilisation at Sandscale Hawes, north-west of England, and Newborough Warren in North Wales, suggest that a trend towards warmer, slightly wetter and less windy climate has been the main driver of stabilisation, although atmospheric nitrogen deposition, increasing atmospheric

CO_2 concentrations, reduced grazing intensity (including declines in rabbit populations) and reduced human disturbance, have played contributing roles. At Sandscale Hawes, a tendency for shoreline progradation on the west-facing coast may also be a factor (Jones *et al*. 2008; Jones *et al*., 2010; Pye *et al*., 2020).

The traditional approach to coastal defence has been to counter flood risk with hard engineering, but measures such as seawalls are expensive and inflexible, and often deliver unexpected environmental outcomes (Firth *et al*., 2014). By contrast, vegetated shorelines such as sand dunes, provide a natural defence that offer adaptability, flexibility and cost-effectiveness with the additional benefit of the other ecosystem services (Hanley *et al*., 2020). The approach to dune management has changed over recent decades as new research has improved our understanding of the natural dune functioning and the impacts of climate change. For dunes to be healthy, self-sustaining, biodiverse and resilient they must be dynamic, and dune management in England and Wales is now aimed at supporting natural processes, encouraging mobility of sand in the dunes, and reversing the effects of stabilisation (Arens *et al*., 2024). Lack of sediment supply and inappropriate coastal management are the key concerns. The main sources of sand in the past were marine reworking of glacial sediments on the seabed and in coastal cliffs, however sand supplies are now less significant. The construction of sea defences has also affected sediment supply: cliff defences halt erosion and groynes can interrupt longshore drift that transports sediment in a prevailing direction. Hard defences, such as sea walls, can increase sand loss from beaches by reflecting wave energy and thereby, increasing sediment mobilisation and resulting in scour. Sea walls and revetments (a facing, usually of stone or concrete, to sustain an embankment) can also lead to fossilisation of the landward dunes by disconnecting beach and dune systems. Furthermore, offshore dredging and associated port infrastructure can impact sediment supply and movement (Pye *et al*., 2007c). Wider-scale impacts on sediment supply have occurred through the alteration of our estuary systems, successive land claim, and drainage (Denning *et al*., 2024).

The current approach to sand dune conservation management in England and Wales aims to rejuvenate sand dune habitats, allowing them to maintain and enhance their successional stages. While there is an existing policy of working with natural processes around sand dune dynamics (Jones *et al*., 2021), the new management approach aims to kick-start this by initiating carefully planned interventions to remove vegetation and enhance wind speeds and sand supply on site, thus avoiding hysteresis effects where current vegetation cover prevents natural mobility (or which would only occur at much higher wind speeds). Re-introducing natural dynamism to sand dune systems improves habitat resilience and the provision of multiple ecosystem services (Creer *et al*., 2020; Feagin *et al*., 2019; Arens *et al*., 2024). Such management interventions also facilitate natural processes critical to dune evolution and adaptation, such as crest buildup and rollback and improve connectivity between beach and dune complexes, thus creating a more sustainable system that provides more flood protection and ecological and socioeconomic benefits (Bridges *et al*., 2021). Management interventions in dune systems in England and Wales have in some cases involved large-scale remobilisation, leading to some controversy in the academic press, although the consensus amongst most dune ecologists is that larger scale interventions are necessary to re-instate natural processes and reverse the decline of many rare plant and animal species (Delgado-Fernandez *et al.,* 2019; Pye and Blott 2020; Arens *et al.,* 2020; Creer *et al.,* 2020; Austin and Walker-Springett, 2021; Cooper and Jackson, 2021).

The Scottish Government's management strategy for coastal dune habitats has a strong focus on the encouragement of natural processes, as this is likely to optimise the representation of habitats and the species they support. Although there could be net losses of some coastal habitats to others, the result should be a retention of naturally functioning coastal habitat that is best able to adjust to climate change. The Scottish Government has analysed change and trends on the whole Scottish coast using a range of techniques, allowing the most vulnerable areas to be identified and targeted (Hansom 2017). For example, certain dune systems, such as Torrs Warren in south-west Scotland, have become stabilised due to scrub and tree invasion and may therefore need to be targeted for management action. Nitrogen deposition in Scotland is highest in the south-east, which is also where the strongly nitrifying Sea-buckthorn is most frequent and now controlled through a co-ordinated Sea-buckthorn management programme (Denning *et al*. 2024). The next phase of this work provides guidance and funding for all Scottish coastal Local Authorities to develop Adaptation Plans.

Finally, increased presence, abundance and distribution of invasive non-native species (INNS) can be indirectly linked with climate change. Invasive plant species enter dunes systems through multiple pathways and can cause substantial ecological damage to the ecosystems including a loss of biodiversity, changes in soil fertility, increased water use, with localised effects on the water table level and sometimes water chemistry. INNS can cause shifts in community composition associated with sand dunes, for example over 460 non-native taxa were recorded in the sand-dune vascular plant inventory of the Sefton Coast between 2005/06 and 2018, with twice as many non-native plant species observed during the 2018 survey (Smith, 2020). Although some plant species with ranges previously limited to southern England are expanding naturally due to climate warming (Walker *et al*., 2023), invasive plants frequently arrive to dune systems by accidental or deliberate release from adjacent gardens or parks, or via dumped garden waste near dune habitats (Houston, 2023; Denning *et al*., 2024).

The Climate Change Risk Assessment (CCRA3) (Berry and Brown, 2021) identified the need for enhanced monitoring, surveillance and early response measures to manage the risk to UK terrestrial species and habitats from pests and pathogens, especially in view of the need for improved international co-ordination following Brexit; regulations and resources need to keep pace with the increasing risks and management measures required and consider the changing portfolio of risks, for example, from INNS relative to native species. Further research is required to understand specific climate responses and thresholds of high-risk pest and pathogen species, and the potential change in risk associated with different adaptation options. A systematic risk screening on more than 1300 alien plant species identified the priority species that can have a serious ecological impact on the European protected habitats of Atlantic dunes and saltmarshes (Adriaens *et al.*, 2023). The report advises on the development of preventive actions, rapid removal of newly emerged exotic species and surveillance systems. Recommendations are included for better invasive non-native species monitoring and management for the Habitats Directive.

Machair

Machair is a form of calcareous dune grassland restricted globally to northern Scotland and the north-west of Ireland. It does not occur in Northern Ireland. The Annex I habitat is referred to as ‘machair grassland’ to distinguish it from the wider functional ‘machair system’ that includes the beach, dune, machair grassland, marshes, swamps and, where the sea gains entry to low-lying sections, saltmarsh and sandflats. The Scottish area of machair grassland has been calculated from a 1990s survey dataset as 11,678 ha (Habitat Map of Scotland). In the UK, machair occurs mainly on exposed, low-lying coasts that are increasingly vulnerable to inundation due to sea-level rise (Angus and Hansom, 2021). The habitat has been defined in detail by Angus (2006). Machair biodiversity is highest in Uist, where rotational cropping of fodder cereals supports a wide range of crop ‘weeds’ and the associated fallows are also diverse.

As machair has a restricted geographical extent which is limited to low-lying coastal areas with high winter water levels, it can be particularly vulnerable to climate change impacts such as changing weather patterns, RSLR and increased frequency of storm events. Existing records already show seasonal increases in precipitation around the times of ploughing and harvest, with decreases during the summer growing season. Key issues with the impacts of RSLR and increased storminess on machair relate to water management, including keeping seawater from overtopping the dune ridge or contaminating the machair water table, and ensuring that precipitation can be discharged to the sea.

The machair habitat found in the Scottish Outer Hebrides at Uist is possibly the finest in ecological terms and has been subject to a detailed study of exposure to climate change. This study revealed unusually extended linkages across not only the adjacent coastal habitats, but also with the nearshore marine environment, socio-economics and a historical legacy of inland water management (Angus, 2018). Many areas of machair are drained loch beds that lie below the level of mean high-water springs (MHWS), so that the dune ridge is of critical importance in protecting these areas from marine incursion.

The aforementioned linkages at Uist have increased vulnerability to climate change in two main respects. Firstly, the extensive beds of kelp *Laminaria hyperborea* west of Uist were found to dampen wave energy, however rising sea levels would progressively disengage the wave base from the seabed and its kelp, so that wave impact on frontal dunes will increase over time. Secondly, the drainage network can only discharge surplus inland water to the sea when the sea is at a lower level than the inland water. Rising sea levels not only reduce the 'head' of fresh water, but also reduce the time available for discharge during the tidal cycle. With winter precipitation likely to increase, significantly in some scenarios (Kay *et al.,* 2011), machair flooding by fresh water is likely to become more frequent (Angus, 2018). Risks could be amplified through the compound effects of these climate risks, and through the linkages that exist between these habitats and the sea at Uist. Changes in storm climate could increase the likelihood of both marine and freshwater flooding on machair habitats. Low pressure systems raise sea level by around 1cm for every hectopascal (millibar) below standard pressure of 1013.5 hPa. This means the wave base is raised off the seabed and the kelp beds during storms, so that waves retain more of their energy, increasing the probability that waves will overtop or breach the dune cordon. Such storms can also be associated with very high rainfall, yet the raised sea levels make it even more difficult to discharge this water to the sea. Saline water, even if diluted by rainwater, can be toxic to many plants and animals associated with this ecosystem.

Maritime Cliff and Slopes

Coastal cliffs composed of friable material such as clay and shale have little resistance to erosion and generally have a shallower gradient in contrast to the near vertical faces of hard-rock cliffs. In exposed situations soft cliffs can be highly dynamic, shaped by erosive action of wind and waves which support a wide diversity of vegetation types with variable maritime influence (JNCC, 2004). Cliff and slope erosion is important for constantly exposing new surfaces for recolonisation by plants and allows early successional vegetation types to be maintained (Denning, 2023).

Erosion is a critical natural function of both hard and soft cliffs. Free-functioning coastal processes are a significant source of sediment within the coastal and marine system, 'feeding' adjacent beaches and providing material for the development of sand dunes and shingle structures that offer natural flood management (Short, 2012; Masselink *et al*., 2020). Cliff erosion along the Holderness coast in Yorkshire has been estimated to supply as much as 3 million cubic metres 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).

Maritime cliffs are highly sensitive to climate change (Natural England and RSPB, 2019). Rising sea levels, increased storminess and erosive wave action accelerate basal undercutting, altering cliff profiles and causing episodic slumping or collapse. Rising sea levels have the effect of eroding the beach, steepening the foreshore and exposing soft-rock cliffs to greater erosion (Denning, 2023). The Future Coast database (Defra, 2002) describes sensitivity to climate change and future cliff recession potential. The extent to which sections of cliff have moved away from historical rates of change is an important means of assessing how climate and sea level may affect their stability (Doody, 2020). Soft cliffs are dynamic in nature and can erode rapidly, with the fastest rates of recession occurring on the south and east coasts of England. For example, Earlie *et al*. (2018) recorded erosion rates three to five times higher than the long-term average for two exposed stretches of coastline at Porthleven and Godrevy in Cornwall. Historical cliff-retreat rates have been shown to range considerably, for example between 2 and 25 cm per year within just a small stretch of coastline in Devon (Shadrick *et al*., 2022).

Changing winter rainfall patterns can also lead to increased groundwater pressure, reducing the stability of maritime cliffs and slopes leading to collapse (Natural England and RSPB, 2019; Burden *et al*., 2020). As cliffs undergo erosion or collapse, habitat patches become smaller and more isolated (Doody, 2020; Shadrick *et al*., 2022). Erosion can create new habitats, such as crevice and ledge-plant communities, many of which contain rare or endemic species (JNCC, 2004; Denning, 2023). However, the pace of climate-change driven erosion may exceed many species' ability to adapt if it happens too rapidly for ecosystems to recover or recolonise (Burden, 2020).

Habitat loss and fragmentation due to a combination of coastal retreat and lack of space for cliff-top habitats to roll back is a serious risk to both coastal slope vegetation and invertebrates reliant on cliff-top habitat. The quality of some of the vegetation types may also be affected by climate change. An increase in rainfall may affect specialist, drought tolerant, annual plants (therophytes) as the area of bare ground on hard rock cliffs rises because of erosion (Denning, 2023). Many therophytes are adapted to saline influence but only within tolerable ranges, and excess salinity may have impacts upon their physiology inhibiting key life stages such as germination and seedling establishment. Therophytes are adapted to low-competition, open**,** disturbed zones (Grime, 1977), and may succumb to faster growing perennials or invasive species under altered climatic and sediment regimes. Overall, increased erosion may be favourable for over stabilised habitats, but unfavourable for those already being eroded where there is a lack of space to move landwards (Denning, 2023). Sea-level rise and more frequent storm surges also increase exposure of these coastal ecosystems to flooding (Bricheno *et al*., 2023) and saltwater intrusion (Richardson *et al*., 2024).

Increased annual average temperatures with hotter and drier summers can alter vegetation composition in favour of invasive species, resulting in a loss of native biodiversity. Hottentot fig (*Carpobrotus edulis*) is a pioneer of disturbed sites and coastal areas such as cliffs and sand dune systems[5]. Physiological tolerance of this species to stress factors such as salinity, drought and excess light affects patterns of native species diversity contributing to its invasive success and confer advantage under climate change scenarios representing a serious threat for coastal habitats (Campoy *et al*., 2018).

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[5] *Carpobrotus edulis* (L.) N.E.Br. In BSBI Online Plant Atlas 2020 [Accessed 9 February 2026]

Coastal protection measures put in place to reduce or prevent climate change impacts may affect the natural zonation of maritime grassland and cliff-top heathland vegetation. Over-stabilisation results in the reduction of bare ground and the loss of early pioneer communities (West, 2020), leading to grass- dominated swards or the development of scrub and woodland. Erosion tends to be favourable for over-stabilised habitats, whereas already eroding natural maritime cliff and slope habitats can get squeezed due to a lack of space for landward recession, presenting a serious risk to both the natural vegetative communities and the invertebrates reliant on them (Denning, 2023).

Approaches to coastal protection in the UK are changing. Where the economic justification for protecting the coast is uncertain, even including where houses may be at risk, more sustainable, “nature-based” alternatives are being explored. For example, managed re-alignment has taken place in the village of Happisburgh in Norfolk, where existing groynes have been removed, allowing the cliff to recede landwards (Denning, 2023).

Rocky Intertidal Habitats

Rocky intertidal habitats refer to a range of temperate intertidal hard substrate ecosystems. In the UK, these intertidal rock habitats are dominated by different foundational organisms, including barnacles, molluscs, and fleshy and coralline macroalgae, which create biogenic habitat for species-rich assemblages of invertebrates, fish, macro- and microalgae, for example. Rocky shores provide ecosystem services such as wave attenuation, habitat, and food resource provision for both marine species and humans.

The MarClim project[6] is the most spatio-temporally extensive time-series for rocky intertidal species globally. Extending back to the 1950s, this time-series dataset consists of annual data on the distribution and abundance of 87 species of invertebrate and macroalgae for 100 sites around the UK coastline since 2002, providing evidence for climate-driven changes to species within rocky intertidal habitats. This time-series data forms the basis for much of the evidence and projections in this section.

The response of species within rocky intertidal habitats to climate change is highly species-specific. Most show oscillations in population abundances around UK Regional Seas due to the life cycles of species or natural stochasticity (where the ecosystem state cannot be precisely predicted, even with knowledge of all the factors affecting that process). All species have a thermal tolerance range where physiological performance can decline towards the thermal tolerance limits, or mortality occurs once these limits are exceeded. However, many rocky shore species in the UK have not yet reached their climate driven upper thermal limits. This can be seen in citizen science data collected across the UK for the Capturing Our Coast project, where survey data suggested that thermal range location may influence trends in abundance for individual invertebrate species on UK coasts.

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[6] Marine Biodiversity and Climate Change, Marine Biological Association [Accessed 24 November 2025]

Populations of algae are patchier at the site scale towards trailing range edges, with high, space-limited abundance in occupied areas and low elsewhere, giving a consistent upper limit to abundance across the thermal range. For invertebrates, smoother variations in population density have been observed between favourable within-range and unfavourable marginal locations (Vye et al., 2020).

Some invertebrate species continue to show changes in the leading or trailing edges of their biogeographical distributions in response to warming of the UK marine climate. One example is the leading range extension of the barnacle Perforatus perforatus, a Lusitanian species (i.e. it has evolutionary origins in warmer waters from lower latitudes), which was recorded on Welsh MarClim surveys in 2022, 2023, and 2024. The previous leading range edge was recorded in Pembrokeshire, whereas a new population has become established on the Llŷn Peninsula, Gwynedd, representing an extension of 270 km, or 1.04 degrees latitude in the leading range edge. This recent range limit extension is likely due to climate change, as suitable habitat has existed between these two locations for many decades.

These species-specific changes are driving community-level changes. Community Thermal Index analyses show that communities of animal species in Shetland showed a shift in dominance towards warm-affinity species (‘thermophilisation’) with local warming from 1980 to 2018 but the community of macroalgal species did not. From 2002 to 2018, communities in south-west Britain showed the reverse trend in CTI: declining average thermal affinities over a period of modest temperature decline (Burrows et al., 2019, Chust et al., 2024).

A recent review of macroalgal species using time-series from UK coastlines categorized 7% of species as threatened (1% Critically Endangered—CR, 3% Endangered—EN, 3% Vulnerable—VU), and 55% as Data Deficient (Brodie et al., 2023). Heatwaves in recent years have caused sublethal heat damage in macroalgae, with mortality-induced reductions in abundance of a few species of invertebrates after a heatwave event in 2020. Mid- and lowshore fucoid algae showed much less damage than highshore fucoids, although no fatal damage was recorded (Mieszkowska and Sugden, 2022; Mieszkowska et al., 2022; MarClim, unpublished data). Cold-affinity boreal species were more negatively impacted by heatwaves than warm-affinity lusitanian species that have warmer thermal tolerance ranges. Cold spells in winters had greater negative effects on warm-affinity species, although this was not supported for summer cold spells (Mieszkowska et al., 2021). Winter warm spells tend to have a more negative effect on cold-affinity species than on warm-affinity species.

Worldwide, the arrival, spread, and establishment of invasive non-native species in the marine environment has been linked to numerous pathways such as shipping and aquaculture, with the likely secondary effect of climate change further facilitating spread via secondary introductions (Katsanevakis *et al.*, 2013). For UK rocky shores, invasive non-native species that are already established continue to spread around the coastlines. In addition, a small population of the invasive crab *Hemigrapsus takanoi* has been recorded at Hampton Pier, in Kent in 2022 (MarClim, unpublished data). This is the second record on the north Kent Coastline and one of nine records in total along the southern North Sea and eastern English Channel coasts of the UK, including one individual in 2014, two in 2017, and five individuals in 2020, as shown in the NBN Atlas[7]. The invasive red alga *Caulacanthus okamurae* is spreading across sites along the coastline of the English Channel and Celtic Sea (Mieszkowska, 2018; MarClim, unpublished data; also see NBN Atlas[8] and World Register of Marine species[9]).

Intertidal Soft Sediments

Intertidal soft sediments, including UK shores with intertidal mud-and sand flats and beaches, are dynamic environments that act as natural buffers against coastal flooding and erosion (e.g. Brunetta *et al*., 2019; Reed *et al*., 2018). These habitats are highly productive and of significant ecological importance (Bouma *et al*., 2014). They occur extensively throughout the UK along open coasts and in lagoonal inlets and estuaries. Intertidal flats have been estimated to occupy an area of approximately 2700 km^2 in the UK (Murray *et al*., 2019). Recent analyses of satellite imagery have revealed a global trend of diminishing area of intertidal flats (Murray *et al*., 2019) but UK- specific analyses are hindered by measurement uncertainties for highly dynamic intertidal flats typical of UK settings (Mason *et al*., 2010).

Abiotic controls are critical to intertidal habitat characteristics (e.g. Paterson *et al*., 2019) but the impacts of gradual long-term climate-related changes in baseline (e.g. sea level, temperature, pH) are difficult to detect against a background of high short-term variability. Nevertheless, most beaches have steepened over the past century (e.g. Masselink *et al*., 2020) in combination with coarsening of sediment particles. For intertidal estuarine flats, most studies have focused on the response to changes at timescales shorter than those related to climate change, leading to conceptual models of the response of intertidal flats to changes in external forcing (e.g. Friedrichs, 2011). The general response to RSLR is a redistribution of sediments within estuaries that results in landward and upward migration with rising sea level (e.g. Masselink *et al*., 2020 and references therein). Sediment supply is critical to this response, and the presence of coastal defence structures may block migration and lead to a narrowing of the intertidal zone, also called ‘coastal squeeze’. Intertidal flats are subjected to a range of physical and biological drivers across spatial and temporal scales and understanding the combined effect that results from biological-physical interactions is fundamental to understanding the impacts of climate change on these systems (Solan *et al*., 2023).

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[7] https://species.nbnatlas.org/species/NHMSYS0001697250 [Accessed on 30 March 2026]

[8] https://species.nbnatlas.org/species/NHMSYS0021060121 [Accessed on 30 March 2026]

[9] https://www.marinespecies.org/aphia.php?p=taxdetails&id=496188#distributions [Accessed on 30 March 2026]

Even though intertidal ecosystem processes are impacted by large-scale climate-induced ocean variability (e.g. Cloern and Jassby, 2012), assessing the impacts of climate change on soft sediment communities is challenging due in part to the difficulty in conducting observations and interactions with anthropogenic pressures. Published UK-based long-term monitoring studies are lacking, but evidence from other temperate intertidal soft sediment systems in the Wadden Sea suggest that macrofauna communities respond to sea-level rise mediated abiotic sediment changes (Singer et al., 2023). There is also evidence of favourable long-term evolution of intertidal soft sediment benthic ecosystems (e.g., Beukema and Dekker, 2020; Brebant et al., 2025). However, the importance of local conditions (e.g. Brebant et al., 2025) make transferability to UK systems somewhat speculative. Studies often rely on mesocosms and experimental approaches instead extrapolations from which can be inconsistent with responses in the natural environment (Rimmer 2025). Individual drivers affect species behaviour leading to changes in nutrient turnover and primary production (e.g. Bulling *et al*., 2010; Hicks, *et al*., 2011). The response of soft sediment communities to long term climate-driven is also significantly complicated by short-term (seasonal) variability (Godbold and Solan, 2013), by the non-additive and non-linear effects of multiple drivers (Bulling *et al*., 2010; Hicks, *et al*., 2011), and by local community dynamics (Godbold *et al*., 2011). Altogether, climate-related change is significantly more complex than shifting species' range margins with likely effects on the functioning and vulnerability of benthic ecosystems and the ecosystem services they provide.

Intertidal and Subtidal Seagrass Beds

Seagrass beds are unique biogenic habitats formed by flowering plants (angiosperms) that have adapted to living in saline conditions. Seagrasses reproduce through both pollination and rhizome extension, which enables them to effectively colonise extensive intertidal and shallow subtidal areas of the UK waters when the environmental conditions are suitable (Unsworth and Cullen-Unsworth, 2014). Seagrass distribution spans UK coasts, estuaries, lagoons, lochs, and offshore islands with two dominant species: eelgrass (*Zostera marina*) and dwarf eelgrass (*Zostera noltii*[10]). *Z. marina* is found in shallow, fully marine conditions, mainly on muddy to relatively coarse sediments, whereas *Z. noltii* typically occurs on wave-sheltered intertidal muddy fine sands and sandy muds on the mid and upper shore (d'Avack *et al*. 2022a, b). They can also be found on pebble and rubble substrates (e.g. substantial beds in Strangford Lough in Northern Ireland), near other sedimentary biogenic habitats such as oyster reefs, maerl beds and mussel beds, and within mosaics of other coastal habitats such as saltmarsh and mudflats (Gamble, 2021). In England, seagrass distribution and extent are highest in the south and south-west coasts, although intertidal and subtidal seagrass beds are also found on the east coast and the north-east, including Lindisfarne (National Seagrass Layer, 2025[11]). Scotland

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[10] The spelling *Zostera noltii* (Hornemann, 1832) is used here, however spelling *Z. noltei* is also commonly used in scientific literature.

[11] National Seagrass Layer (England) - data.gov.uk [Accessed on 19 January 2026].

alone holds 20% of the seagrass beds in north-west Europe, with both *Zostera* spp. widely recorded on both west and east coasts (Kent *et al*., 2021). In Wales, the most substantial seagrass beds are located on the coasts of Anglesey and Pembrokeshire (Stewart and Williams, 2019). The current extent of seagrass beds is approximately 17km^2 in total in Northern Ireland, however the area of suitable habitat could be as high as 215km^2 (Strong *et al*., 2021).

Seagrasses form complex three-dimensional habitat structures that provide food and shelter for a diverse range of marine fish, invertebrates and plants. As bioengineers, they play a crucial role by establishing positive feedbacks that lead to the local environment becoming more suitable for both their own productivity and for their associated flora and fauna (Maxwell *et al*., 2017), for example by modifying the carbonate chemistry and pH of their local environment (Unsworth *et al*., 2012; Pacella *et al*., 2018). Other ecosystem services provided by seagrass beds include filtration and trapping of suspended sediments, reduction of nutrient, bacterial and viral loads, stabilisation of sediments and slowing of water movement, thus providing protection against erosion of coastal sediments and improving ecosystem resilience to climate change (Unsworth *et al*., 2021). The role of seagrass beds in climate change mitigation through carbon capture has gained increasing attention globally (Fourqurean *et al*., 2012; Duarte and Kraus-Jensen 2017) and within England (Lima *et al*., 2022; Swaile *et al*., 2022), Scotland (Shafiee, 2021), Wales (Stewart and Williams, 2019) and Northern Ireland (Strong *et al*., 2021). However, seagrasses also produce and release methane, which could significantly impact their role in climate change mitigation (Schorn *et al*., 2022, Williamson and Gattuso, 2022).

For seagrass ecosystems to thrive, they require appropriate physical and chemical conditions that promote plant growth and survival, such as adequate nutrients and light for photosynthesis as well as suitable water depth, seabed substrate and wave exposure. Seagrass growth and distribution are also directly affected by trophic interactions in the ecosystem, primarily through predation and competition for resources. The feedback between seagrasses, other organisms and abiotic conditions can be important for the stability and resilience of seagrass ecosystems, and if disrupted, their structure and functioning can be fundamentally changed, leading to reduced seagrass health and loss of habitat (Maxwell *et al*., 2017). It is the balance of these factors that ultimately determines seagrass survival, and climate change impacts alongside anthropogenic pressures can alter this balance.

In the mainland UK waters, extensive loss and degradation of seagrass habitats has occurred in the past century. Recent estimates suggest a 44% habitat decline since 1936, of which 39% has taken place since the 1980s. The most severe estimates extending further into the past, are as high as 92% (Green *et al*., 2021). The loss of seagrass habitat in the UK is not unique; the global declines have been widespread and substantial (Waycott *et al*., 2009, Dunic *et al*., 2021), although in some bioregions there is evidence of recovery where pressures have been successfully removed (de los Santos *et al.*, 2019). In the UK, the long-term declines in seagrass habitat distribution and extent have largely been attributed to disease; coastal development; changes in land-use practises, physical disturbance; reduced water quality from coastal inputs of nutrients and sediments; and the intensifying impacts of climate change (Jackson *et al.*, 2016; Jones and Unsworth, 2016; Unsworth *et al.*, 2021).

The key climate change stressors impacting seagrass habitats in the UK waters are warming sea temperatures, ocean acidification, RSLR, changes in rainfall patterns, and increasing frequency and severity of extreme weather events such as storms and marine heatwaves (MCCIP, 2018; d'Avack *et al.*, 2022a, b). The effects of climate change stressors differ between the UK regions. As coastal ecosystems tend to be highly diverse and variable at local, regional and global scales and the impacts of anthropogenic activities and climate change are also highly complex, cumulative and interlinked, quantifying habitat-specific responses to the inevitable changes in climate can be extremely challenging (Zimmerman, 2021). However, understanding the current health of seagrass beds in the UK waters and the key pressures affecting their survival and productivity enables us to evaluate the resistance and resilience of these ecosystems to the intensifying impacts of climate change.

Cumulative impacts of climate change and direct anthropogenic stressors including high suspended sediments and eutrophication are likely to decrease the resilience and increase the vulnerability of seagrass to climate change due to reduced light penetration that prevents seagrass from accelerating the rate of photosynthesis in response to increasing water temperature and/or altered water chemistry (Zimmerman, 2021). Recent studies in the UK shores have shown that many seagrass beds are currently in poor condition due to reduced water quality resulting from eutrophication, as indicated by excessive nutrient levels within seagrass plants and reduced light penetration in the water column (Jones and Unsworth, 2016; Unsworth, 2022). This tends to occur in eutrophic waters due to increased phytoplankton biomass and direct shading by competing epiphytic microalgae and filamentous macroalgae (Gustafsson and Boström, 2014). The sources of excess nutrients in seagrass beds in England, Wales, Ireland and Northern Ireland are predominantly from urban sewage and agricultural effluent origins, indicating that eutrophication in the UK seagrass beds is likely to be due to the widespread inefficiencies of current sewage treatment and farming practices (Jones *et al.*, 2018). The nutrient enrichment status in Scottish seagrass beds is yet to be reported on (Kent *et al.*, 2021).

In addition to eutrophication, anthropogenic activities such as dredging, beach nourishment, sediment discharge and coastal protection reduce water quality through increased turbidity and sediment mobility, resulting in a further reduction in the light available for seagrass photosynthesis (Munkes *et al.*, 2015). In general, eutrophic and turbid areas within English waters are most prevalent in sheltered coastal areas such as estuaries and shallow inlets and bays, which overlap significantly with the current and historic seagrass beds and coastal areas with seagrass habitat restoration potential (Swaile *et al*., 2022). It is therefore likely that many of the seagrass habitats in English waters are less resilient to climate change impacts due to cumulative impacts of pressures from anthropogenic sources, therefore addressing these pressures is important to enable and enhance seagrass habitat recovery. Sustained local and regional efforts to improve water quality based on sound science have been successfully applied to increase seagrass productivity and survival elsewhere, including Chesapeake Bay in the USA (Zimmerman, 2021). In Milford Haven, Wales, where historical pollution encroachment and oil spills had previously reduced seagrass extent and distribution, focused water quality management efforts have resulted in intertidal *Z. noltii* habitat recovery, although the subtidal *Z. marina* beds have not shown improved health and condition in this location (Bertelli *et al*., 2018).

If water quality issues and other anthropogenic pressures are addressed to make areas of seabed suitable for seagrass recovery, restoration or creation, focused effort on increasing seagrass habitat distribution and extent within UK waters is likely to provide a myriad of ecosystem benefits, from coastal protection to improved local conditions for associated biodiversity. History suggests seagrass beds can fluctuate and recover from dramatic losses. Recent attempts of restoration have shown that seagrass beds can regain abundance, which is encouraging and should help stimulate conservation initiatives globally (Valdez *et al*., 2020). In the UK, projects and partnerships to restore seagrass habitats at scale are gaining popularity[12]. However, to date, evidence of seagrass restoration success in the UK is still patchy and limited, with long-term success yet to be proven. Although reduction of removal of anthropogenic pressures is important for improving seagrass habitat recovery over time, restoration projects have commonly prioritised active restoration over addressing existing pressures. The EU Life Recreation ReMEDIES project was the first UK partnership to tackle the reduction of existing recreational pressures damaging the seabed including anchoring and mooring, alongside trialling methods of active seagrass restoration at scale[13]. The project also had a significant emphasis on public engagement, education and behaviour change to promote public awareness and change perceptions in the conservation and restoration of these sensitive habitats, which is fundamental for restoration success.

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[12] The UK's biggest seagrass restoration project - Swansea University [Accessed on 19 January 2026]

[13] Home - Save Our Seabed [Accessed on 19 January 2026]

WHAT COULD HAPPEN IN THE FUTURE?

Saltmarsh

RSLR, storm frequency and flooding are projected to increase with global warming (Palmer *et al*., 2018). Global Mean Sea Level (GMSL) to rise is predicted between 0.28 and 1.32m by the year 2100 (IPCC, 2019; Horton *et al*., 2020), which may lead to coastal saltmarsh habitats either declining or expanding in size, depending on their capacity to migrate landward (Schuerch *et al*., 2018). Rising sea levels and more frequent and intense storm events will continue to alter sediment transport around the coast, which combined with freshwater and anthropogenic inputs will likely impact the areal extent of saltmarshes (MCCIP, 2018), with a projected loss of 11% of UK saltmarsh between 2010–2060 (Hudson *et al*., 2021). A study by Horton *et al*. (2018) showed a greater than 80% probability of saltmarsh retreat for the whole of Great Britain by 2100. Effects will vary locally, depending on the factors mentioned above, combined with saltmarsh morphology, management strategies and local ecology. Erosion, subsidence, hypoxia and loss of water clarity may also affect associated habitats, which could in turn affect hydrological and ecological connections. Hence, there is a need for conservation efforts to consider locations where future habitats will develop (Lawler *et al*., 2020).

Saltmarshes can keep pace with RSLR where there is an adequate sediment supply to maintain vertical accretion and space for habitat transition (Moritsch *et al*., 2022). Saltmarshes with both higher tidal ranges and suspended sediment loads may therefore be more resilient (Masselink *et al*., 2024). Fluvial sediment supply provides further resilience to RSLR (Wu *et al*., 2020). However, there is a complicated relationship with saltmarshes and the adjacent mudflats which affect the equilibrium on infilling, erosion and stability (Mariotti and Carr, 2014), and subsidence and autocompaction may limit the ability of marshes to keep up with sea-level rise (Saintilan *et al*., 2022).

Rising sea levels alter the balance of coastal freshwater-saltwater interaction both at the marsh surface and in the subsurface hydrology causing changes in saltwater intrusion. Resulting elevations in soil water salinity trigger the mortality of salt-intolerant vegetation eventually altering the ecosystem function (Zhang *et al*., 2022). As sea levels rise and river flows reduce, saltwater will move further inland, resulting in increasing saline intrusion which will be most detrimental for upper estuarine, low salinity (oligohaline) and tidal freshwater zones (Little *et al*., 2022). Changes in land use for agriculture, construction and extraction, combined with changes in catchment management and precipitation patterns may further alter the transport of nutrients, sediment and pollutants by rivers and porewater (Olds *et al*., 2016; Newton *et al*., 2020). This in turn can have implications to plant zonation and nutrient and carbon cycling (Xin *et al*., 2022). The effects of the combined anthropogenic and climate change hazards over the next century are not yet well understood but are currently being investigated by

CHAMFER[14], by combining existing UK hydrological, meteorological and sedimentological models with field validated coastal habitat models.

Increased frequency of storm events could bring larger waves, which may laterally erode the marsh but can also deposit large quantities of sediment. This initially builds up the marsh which in some cases can improve resilience to RSLR (Pannozzo *et al*., 2021). However, the subsequent consequence of this may be lowering of the mudflat and steepening of the saltmarsh, exposing the edges to lateral erosion (Schuerch *et al*., 2019). RSLR will also cause deeper waters and bigger waves to 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). Although landward migration of saltmarsh could compensate for these losses, much of the extent of estuaries is 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 on the coast, such as dredging, also potentially increase the vulnerability of marshes to climate change by diminishing and disrupting natural sediment supply which will slow down saltmarsh growth, further reducing its resilience and natural capacity for recovery (Burden *et al*., 2020). As storms are projected to more frequently occur over autumn and winter in the UK, this may aid seed dispersal of some species of saltmarsh plants which benefit from extreme water levels depositing seed further up the shore, such as *Spartina* spp. and *Salicornia europaea*. These windows of opportunity can also be windows of risk as seeds may be swept off tidal flats, depleting the local seedbank (van Regteren *et al*., 2019), Zhu *et al*., 2022).

Seasonal changes in freshwater flow also impact saltmarsh habitats. Wet winters bring more rain and groundwater, which saturates the saltmarsh soils and lowers salinity. This creates favourable conditions for plant growth, especially in the upper and middle zones of the marsh (Boorman, 2019; Feng *et al*., 2025). During the dry summer months, the tides take over, increasing salinity and potentially shifting the balance of plant life. As climate change alters rainfall patterns and accelerates sea-level rise, understanding this interaction becomes even more important. If freshwater inputs decrease due to drought or other factors, saltmarshes may struggle to maintain their current structure and function. Freshwater inputs play a critical role in shaping these environments, and their influence may be key to predicting how saltmarshes will fare in the future (Montalvo *et al*., 2024). Data analysis conducted of the relationship between groundwater levels and marsh sediment redox potential suggests that terrestrial hydrology influences carbon sequestration and marsh resilience. Lower groundwater levels resulting from pressures such as excessive abstraction or prolonged drought for example have been linked to increased redox potential and rates of decomposition in marsh sediments resulting in slowed rates of accretion and reducing capacity for carbon sequestration (Guimond *et al*., 2025). This underscores the need for a deeper understanding of the role of terrestrial

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[14] UK Coastal Hazards, Multi-hazard Controls on Flooding and Erosion | CHAMFER (noc.ac.uk) [Accessed 21 January 2026]

ground and surface water flows in sustaining coastal ecosystems and their vulnerability to impacts of climate change.

Even where saltmarshes can keep pace with RSLR, a growing body of evidence suggests that climate change may disturb the specific environmental conditions and ecology linking the terrestrial, river and estuarine systems these unique zones support, disrupting the distinctive and productive communities that play an important role in the functioning of the coastal or estuarine ecosystems as a whole (Little *et al*., 2022). The loss or reduction of these zones may result in a net loss of function, with critical implications for the ability of these habitat mosaics to continue to provide key ecosystem services into the future (Ensign and Noe, 2018; Little *et al*., 2022).

Some ongoing loss of habitat will be mitigated by an increase in restoration of intertidal ecosystems as nature-based coastal defences (Bouma *et al*., 2014; Hanley *et al*., 2020), for carbon offsetting, or for biodiversity and protection of bird populations in particular. However, restoration of saltmarsh may not recreate functional habitat, or provide ecosystem services equivalent to those from natural systems. Restored sites in the UK take approximately a century to attain equivalent soil carbon pools (Burden *et al*., 2013), and many decades for plant communities to resemble those of natural marshes, if indeed at all (Mossman *et al*., 2012). Poor soil drainage of re-aligned sites can lead to a build-up in nutrients and contaminants, resulting in lower biodiversity and slower community maturation (Garbutt and Wolters, 2008; Spencer *et al*., 2008; Spencer *et al*. 2017). Furthermore, low plant diversity has been linked to soil stability (Ford *et al*., 2016), therefore low-diversity restored habitats are potentially less resilient to changing climatic conditions such as increased wave energy, RSLR and storm surges. These effects may also be exacerbated by high nutrient levels decreasing root biomass and further destabilising the soil (Penk *et al*., 2020). More-frequent high-flow, and low-flow, flood events may result in an increase in peak concentrations of diffuse-source and sewer pollutant loads, and wastewater-derived pollutants respectively (Whelan *et al*., 2022). This could in turn affect coastal nutrient loading and thus the resilience and function of saltmarsh communities.

Sand Dunes

In the UK, RSLR and changes in the coastal movement of sediment are projected to have already contributed to a 2% loss in area between 1999 and 2020, and future estimates predict coastal habitat losses of around 8% by 2060 although more data are required to validate these projections (Jones et al., 2011). The prevailing opinion is that sand dunes have potential to adapt to some climate change impacts through natural processes (Burden et al., 2020) and dunes will migrate landwards or 'roll over' in response to RSLR (e.g. Defra, 2012). However, past and present interventions have often reduced or constrained sediment processes while anthropogenic development has reduced the ability of dunes to migrate landwards, resulting in a reduced capacity of dunes to respond naturally and adapt. The mobility of sand dunes is likely to apply only to the seaward sectors as the landward sectors are too stable to mobilise, the dune systems are likely to lose extent and become narrower, with a loss of more stable vegetation types as they are covered by land blown sand.

Coastal dune systems with limited sediment inputs are expected to exhibit increased erosion rates due to climate change impacts, notably RSLR and potentially increased storminess. The extent of frontal dune erosion may expand in the next century with negative impacts on the extent of some dune habitats and the effectiveness of dune systems as flood defences. However, the consequences of such changes will vary between locations, reflecting differences in natural processes and beach-dune sediment budgets (Pye et al., 2007c). The amount and type of vegetation cover will also affect sand movement and sand capture by vegetation over time. Different plant species trap sand in different ways, resulting in a range of different types of dune formation (Zarnetske et al., 2018). Conversely, increased occurrence of flood- and storm events can also have significant physiological impacts on the survival, growth and reproduction of dune vegetation, especially on supralittoral plants inhabiting the dune edge which are typically tolerant of salt spray but not of seawater immersion (Edge et al., 2020; Hanley et al., 2020). Nevertheless, the coincidence of storms with large tides and sediment availability may be a key factor in long term dune loss. Rates of dune erosion are expected to continue to increase due to climate change.

Research using both in-situ observations and a coastal retreat modelling approach predicted most of the dune retreat to occur over a small number of energetic winters characterised by dune roll-over whereby sediment removed from the dune face is deposited on or behind the dune crest (Masselink *et al*., 2022). Observed sand-dune retreat rate was two to three times greater than that predicted, suggesting that RSLR alone may be insufficient to explain observed sand-dune retreat, and that increased winter storminess may also be implicated (Masselink *et al*., 2022). A combination of in-situ analysis and the application of dune retreat models can provide useful insights into future dune evolution crucial for coastal planners and managers to decide how to protect communities in the short term and into the future.

Climate change scenarios for the remainder of this century also anticipate warmer, drier summers. The net reduction in annual recharge to groundwater is likely to cause lowering of the water table (Clarke and Sanitwong Na Ayutthaya, 2010), resulting in the loss of dune-slack habitat to dune grassland (Curreli *et al*., 2013), as well as more subtle shifts in community composition (Rhymes *et al*. 2016; Rhymes *et al*. 2018; Dwyer *et al*., 2021b). Dune slacks located in the north-west of England have been predicted to experience lowered water tables by up to 100 cm by 2080, which will pose a serious threat to dune-slack specialist communities (Denning *et al*., 2024). In Scotland, few scarce species are likely to be impacted, whereas in England and Wales whole communities are endangered by this change since only 20 cm change in the average water table regime can lead to a shift from one plant community to another (Jones *et al*. 2021). Some plant species with a relatively broad hydrological niche, such as petalwort (*Petalophyllum ralfsii*), will require careful management to maintain open conditions (Callaghan *et al*. 2021), while others such as *Hydrocotyle vulgaris* have more constrained niches (Dwyer *et al*., 2021b) and may therefore be more vulnerable to climate change. Consequently, 'turf stripping' has been carried out on some sites to lower the vegetation surface to retain conditions suitable for dune-slack plants. Changing climatic conditions are also likely to increase the rate of reproduction and spread of INNS (Houston, 2023).

Machair

Despite the strong correlation between machair distribution and exposed coasts, the Dynamic Coast project does not anticipate widespread losses from erosion due to climate change impacts (Fitton *et al*., 2017). The most likely scenario is that the dune ridge will migrate landwards, so that croft land within the machair will be converted to dune. Breaching or overtopping of the dune would lead to flooding with sea water, which is toxic to most machair plants and the crops that are associated with high arable biodiversity. This could lead to a reduction in machair cropping, with a consequent reduction in the associated biodiversity. In the most extreme cases, a single breach in the dune ridge could lead to linear flooding of low-lying machair, and its conversion to intertidal sediment flat and/or saltmarsh, so that it is important to safeguard the integrity of the dune ridge (Angus, 2018; Angus and Hansom, 2021). Beaches fronting machair systems might be subject to more-frequent development of ephemeral 'beach crust' concretions as temperatures rise (Angus, 2023), with unknown consequences for biota.

For machair habitats, the immediate management priority is to enhance the protective function of the dune cordon. Overtopping by waves can be reduced by excluding livestock from the dune ridge using fencing, allowing the dunes to grow vertically. The problem is not so much the grazing, but the damage to the root systems of marram by hooves. Although there are obvious disadvantages of erecting fences in a dynamic environment, alternatives such as collars linked to GPS boundaries are often too expensive to be viable for small-scale machair restoration initiatives.

Convincing crofters that the full potential protection afforded by the dune ridge involves allowing it to move landwards with RSLR is challenging, as habitat roll-back is not matched by roll-back of land holdings, so that there is a net loss of croft land. Other aspects of the strategy such as re-engineering the drainage network and planning for asset transfer, require major funding, which is potentially available, but none of the organisations involved have yet been able to identify sufficient resources to make a funding bid.

Maritime Cliff and Slopes

It is very likely that stretches of soft rock cliff currently undergoing erosion will experience increased erosion rates in the future due to RSLR (Masselink and Russell, 2013). Increased rainfall in the future may also lead to increased slope failure, particularly affecting the movement of groundwater in softer lithologies (Burden *et al*., 2020). For example, in North Devon and Yorkshire, a study modelling future erosion rates estimated the rates of soft cliff retreat to increase by up to three to seven times the present rates by 2100, based on the current predictions of RSLR; an increase much greater than previously predicted (Shadrick *et al*., 2022).

Some evidence suggests that even historically stable rock coasts can be highly sensitive to RSLR and should therefore be included in the future planning for global climate change response. As cliff retreat is primarily driven by erosion and strongly linked to the rate of RSLR and increased storminess (Ashton *et al*., 2011; Trenhaile 2014), climate change will have a direct impact on risk associated with coastal hazards in the coming century and beyond, even on historically stable coastlines (Shadrick *et al*., 2022).

However, the response of cliff-front waves to future RSLR is complex and depends on shore platform geometries and RSLR scenarios, indicating that future cliff retreat rates may not accelerate homogeneously under RSLR (Matsumoto 2024). Caution is therefore needed in the interpretation and application of forecasts and modelled data. It is important to understand the local drivers of coastal cliff erosion to assess the future consequences of climate change. For example, a detailed study of the evolution and erosion history of cliffs in East Sussex, southeast of England, showed a persistent decrease in erosion rates to less than half the maximum rate observed in the late 19th to early 20th century despite RSLR (Dornbusch, 2022).

Rocky Intertidal Habitats

Leading range edges of Lusitanian species will continue to move polewards, with population abundances increasing across the biogeographic ranges on UK coastlines. In contrast, trailing range edges of boreal species will continue to retract polewards, with population abundances decreasing on UK coasts (Mieszkowska *et al*., 2021). These changes are likely to continue to be species-specific, with some species from groups such as the fucoids and red algae showing no changes in either leading or trailing range limits (Mieszkowska *et al*., 2002; Mieszkowska and Sugden, 2002). Data suggest that predictions of average species behaviour in response to climate-driven changes in the thermal environment can be reasonably accurate and for some guilds of species this could conform to an abundant centre distribution (Vye *et al*., 2020). Experimental research carried out on the invasive Pacific oyster, *Magellana gigas* has shown that both warming and ocean acidification predicted for the year 2100 did not affect species fitness (Pack *et al*., 2020). The consistency of responses to climate change across different communities and with general expectations based on species thermal characteristics suggests strong predictive accuracy of responses of community composition to anthropogenic warming using Community Thermal Index (CTI)) analyses (Burrows *et al*., 2019).

Predicted increased frequency and intensity of marine- and atmospheric heatwaves and storm events are also likely to have both sublethal and lethal impacts on intertidal species. However, as these species are adapted to extreme environments the impact of extreme events is likely to be short lived. Forecasting impacts of extreme events will be difficult due to the site-specific nature of recent extreme events (Mieszkowska *et al*., 2021) but recovery back to natural community assemblages typical of the region is likely within five to ten years of an extreme event occurring. Decreases in abundance and local extinctions, especially high on the shore may occur if the frequency of occurrence of heatwaves increases and return times shorten (Benedetti-Cecchi, 2001).

Vertical zonation of macroalgae may shift down the shore in response to physical factors such as wave force and heatwaves acting on short timescales, coupled with pervasive climate change and increased wave intensity occurring on decadal timescales. These climate drivers will have the greatest direct impact at the upper end of the vertical distributions of ecosystem engineers. If species interactions intensify in the lower shore due to climate related changes, species may retreat upper shore due to competitive exclusion and changes in dominance of component species. This combination of direct and indirect effects has already been recorded further south, in the Azores (Martins *et al*., 2019), and may start to occur in the UK if extreme heatwave events become more prevalent.

Intertidal Soft Sediments

Effects of climate drivers on intertidal soft sediment systems (e.g. changes to geomorphology, species, and communities) will result from the combined interdependencies of multiple climate-induced changes as well as local context and human activities (e.g. Nicholls *et al*., 2018; Paterson *et al*., 2019). This makes inferring place-based effects from conceptual understanding, generic models, or generalising results from other locations, particularly challenging. The geomorphology of intertidal soft sediment systems is likely to be most sensitive to changes due to RSLR, wave climate, and sediment input (Masselink *et al*., 2020). However, modelling based projections at local scales are highly uncertain due to a combination of ‘cascade of uncertainty’ (Wilby and Desai, 2010) and limitations in the models used to determine the geomorphology of intertidal soft sediments systems (e.g. Amoudry and Souza, 2011). Ecological effects are even more complex due to the multiple interdependencies involved (e.g. Nicholls *et al*., 2018). There is insufficient information to be able to currently determine the overall response to changing climate-induced drivers such as increased temperature, sea-level rise, decreasing pH, increased wave fetch, altered storm frequency, precipitation and salinity combined with any shifts due to human activities. In many situations, the specific dynamics and variability of intertidal soft-sediment systems imply that responses to changes in local context and human activities can dominate the more-gradual response to climate change.

Mesocosms can elucidate the changes due to a small subset of drivers, often single drivers, but these are subject to the challenges highlighted above. For example, a large outdoor mesocosm system was used to determine the effects of discrete extreme temperature events (White, 2018). For the polychaete *Alitta virens* and the bivalve *Cerastoderma edule*, which exhibit different burrowing abilities, neither species exhibited higher mortality because of the heat wave simulations performed. Changes in energy reserves, however, suggested sublethal effects for both, which has implications for their vulnerability to the increased frequency, intensity, and duration of these events predicted for the future.

Intertidal and Subtidal Seagrass Beds

Seagrasses provide ideal species for the development of predictive models on the organism- and population-level responses to climate change impacts due to their relatively simple morphology and life history, and because they are relatively easy to manipulate in laboratory and field experiments (Zimmerman, 2021). Globally, there is now a strong and consistent evidence base from laboratory and mesocosm studies, field observations and simulations that suggest increased CO_2 exposure has a positive impact on seagrass photosynthesis, leaf growth, reproduction, and survival (Palacios and Zimmerman, 2007; Jiang *et al.*, 2010; Zimmerman *et al.*, 2017; Liu *et al.*, 2020). Furthermore, numerous studies have demonstrated that increased CO_2 availability can counter the negative impacts of warming sea temperatures on seagrass growth and survival (Zimmerman *et al.*, 2017; Liu *et al.*, 2020; Zimmerman, 2021). This reinforces the increasingly popular hypothesis on seagrasses benefiting from ocean acidification and having the capability to offset the deleterious effects of gradually warming ocean temperatures where light availability is sufficient to maintain metabolic functions (Zimmerman *et al.*, 2017; Zimmerman, 2021). Warming sea-surface temperatures could also increase the seed germination rate and flowering density of seagrasses (Potouroglou *et al.*, 2014) or reduce senescence during winter months (MCCIP, 2018), although higher seawater temperatures have also been associated with shoot mortality and retarded growth of seagrasses (Tang and Hadibarata, 2022). Increased die-off could also occur, especially in shallow waters where localised temperature increases could reach the species' thermal tolerance thresholds.

Although *Z. marina* habitats around the coasts of the British Isles have been assigned medium sensitivity to global warming under medium, high and extreme global warming scenarios (d'Avack *et al.*, 2022a), the species has an extensive geographic distribution in the Northern hemisphere with a wide thermal range (Lee *et al.*, 2007). A recent study examining the thermal limits of seagrasses and their vulnerability of warming climate estimated an upper thermal limit of 24ºC for *Zostera marina* at higher latitude regions such as the UK (Marbà *et al.*, 2022). This implies the species may be able to tolerate and adapt to the gradually increasing temperatures in UK waters if light and nutrients are available for its altered metabolic needs. Intertidal *Z. noltii* beds have been assigned high resistance and resilience to the extreme scenarios of both global warming and ocean acidification, suggesting the species is not sensitive to these pressures (d'Avack *et al.*, 2022b). Laboratory experiments have shown that *Zostera* spp. can tolerate the combined impacts of warming sea temperatures and ocean acidification better than other seagrass species such as *Posidonia* spp. (Invers *et al.*, 2001). With *Zostera* spp. predominant, the role of seagrass beds in buffering the impacts of warming sea temperatures and ocean acidification by modifying their local carbonate chemistry and ameliorating ocean acidification of the surrounding water could therefore become increasingly important with the accelerating rate of climate change in the UK waters (Koweek *et al*., 2018). However, although nutrient loading has been shown to enhance vegetative growth and sexual reproduction of *Zostera* spp. in some oligotrophic coastal and estuarine ecosystems (Qin *et al*., 2021), seagrass beds in the UK are reportedly in poor health due to reduced water quality (Jones and Unsworth, 2016) and may therefore already be close to their threshold limits for light availability and other consequences of nutrient enrichment. Therefore, UK seagrass beds may be unable to respond to the warming sea temperatures and increasing CO_2 availability through accelerated rates of photosynthesis.

In addition to direct impacts on seagrass survival, other climate change effects such as RSLR, increasing severity and frequency of storm events (Hanley *et al*., 2019), and changing rainfall patterns are all likely to result in further reductions in water quality within coastal waters through increased turbidity, siltation and run-off of nutrients and sediments from water courses (Mossman *et al*., 2015), adding to the already complex mix of factors causing light attenuation within UK seagrass habitats. For example, rising sea levels will impact seagrasses in the deeper end of their subtidal range through changes in the light environment (caused by increased depth) and turbidity (due to altered tidal range) (Aoki *et al*., 2020). Seagrass plants experiencing light limitation have also been shown to allocate more resources to maintain leaf growth over rhizome growth (Gustafsson and Boström, 2014), which has implications for seagrass carbon storage but also the survival of seagrass beds in potentially stormier seas.

Light availability for seagrass beds within the intertidal and shallow zones is less likely to be limiting, however these habitats could be more affected by the direct, physical impacts of RSLR and increased frequency and duration of extreme weather events causing high winds and waves, storms and storm surges (Unsworth *et al*., 2015; Hanley *et al*., 2019). Both *Z. marina* and *Z. noltii* beds may be able to expand their ranges and migrate landwards to compensate for RSLR if not constrained by the lack of suitable habitat or steep topography, in which case recovery is expected to be limited at best (d'Avack *et al*., 2022a, b). The seagrass beds occurring in the shallower end of the *Zostera* spp. depth range that already experience longer exposures to sunlight, emersion, high temperatures and higher risk of desiccation (Unsworth and Cullen-Unsworth, 2014) are particularly at risk of the intensifying global warming and decreasing summer rainfall.

Perhaps one of the most serious climate change effects on the UK seagrasses will be the increasing frequency and duration of marine heatwaves (MHWs). Over the past century, MHWs have doubled in frequency, become more intense, lasted for longer, and extended over larger areas (IPCC, 2022a), and their impacts are predicted to intensify further (Frölicher *et al*., 2018; Oliver *et al*. 2018). Both UK *Zostera* species are highly sensitive to MHWs (d'Avack *et al*. 2022a, b). Evidence from laboratory studies has shown severely reduced shoot densities and increased rate of *Z. noltii* dieback after a relatively sudden rise in water temperatures (Repolho *et al*., 2017). Long-term field experiments in North America recorded severe declines in *Z. marina* shoot densities after a single MHW event (Aoki *et al*., 2020). This study also identified a threshold depth beyond which seagrasses were unable to recover after a sudden temperature stress due to the limited light availability for the changed metabolic needs of the seagrass plants, suggesting MHWs may also limit the depth range of subtidal seagrasses including *Z. marina* further towards shallower waters (Aoki *et al*., 2020). In the UK however, MHW impacts are likely to be more intense in shallow and intertidal waters where the temperature stress is the greatest and exposed plants are likely to experience desiccation, suggesting that the intertidal *Z. noltii* may be particularly vulnerable to the intensifying impacts of MHWs. In addition to the observed seagrass dieback, significant losses of sediment carbon associated with seagrass beds have also been recorded following MHWs (Aoki *et al*., 2021), implying that MHWs could accelerate climate change impacts further through increased release of sediment carbon to the atmosphere.

In addition to the direct impacts of climate change on the chemical and physiological functions of seagrasses, the trophic interactions within seagrass ecosystems are likely to be altered owing to the intensifying impacts of climate change. As seagrasses are relatively poor competitors for light (Zimmerman, 2021), opportunistic macroalgae and epiphyte species may already have a competitive advantage within many UK seagrass ecosystems that are suffering from poor water quality. Warming sea temperatures are likely to exacerbate this competitive advantage, causing further reduction in available sunlight for seagrass photosynthesis and respiration (Hughes *et al*., 2004; Mossman *et al*., 2015). Climate change stressors may also decrease the natural defences of *Zostera* spp. to predation; for example, Zayas-Santiago *et al*. (2020) observed significant changes in the chemical pathways of *Z. marina* following CO_2 enrichment, which may result in the seagrass leaves becoming more vulnerable to grazing and opportunistic pathogens such as the *Labyrithula* complex often held responsible for the wasting disease (Zimmerman 2021).

Overall, the sensitivity of seagrass ecosystems to the multiple climate change stressors described above is likely to vary between the coastal areas of the UK, depending on the multitude and severity of simultaneous pressures affecting their performance, productivity and survival. In the North Sea, where the coastal and estuarine regions are already subjected to high levels of turbidity and eutrophication, seagrass beds may be particularly vulnerable to RSLR, compounded by increases in storm swells and heavy rainfall events due to strong tidal regimes and the effects of storm surges in this region (IPCC, 2022a, b). In the south and east coasts of England, where sea levels are rising faster and the land is also sinking slightly (Morecroft and Speakman, 2015; Stoker *et al*., 2020), seagrass beds may be particularly vulnerable to climate change impacts due to the excess nutrient loads and high macroalgal cover that are affecting water quality and seagrass health (Unsworth, 2022). Warming is expected to be greatest in the

English Channel and the North Sea, with smaller increases in the outer UK shelf regions (Stoker *et al*., 2020), implying seagrasses in these areas may in the future be particularly limited by light to compensate for the increased need for respiration. Seasonal mean and extreme waves are generally expected to increase slightly to the south-west of the UK, where some of the healthiest and largest English subtidal seagrass beds are located, reduce to the north of the UK and experience little change in the North Sea (Jenkins *et al*., 2009), although there are large uncertainties around these predictions. More recent projections of significant wave height (SWH) suggested an increase in annual maximum SWH off the south-west of the UK and in parts of the Irish Sea but noted that these projections should be viewed as indicative of the potential changes with low confidence (Palmer *et al*., 2018).

More UK-specific evidence is required to determine the local and regional scale impacts of climate change stressors on seagrass habitats alone and in combination with the multitude of anthropogenic pressures discussed above. However, sufficient availability of light for photosynthesis stands out as a key limiting factor for seagrass health and productivity and is the common consequence of most if not all the anthropogenic and climate induced stressors UK seagrass beds are facing. Water quality, and specifically its impact on light availability, is therefore intrinsically linked with the resilience and sensitivity of seagrass ecosystems to climate change impacts. Ultimately, due to the poor health of UK seagrass ecosystems, removing all the stressors and restoring the integrity of trophic interactions is perhaps not realistic, and climate change is likely to lead to alterations in the extent and distribution of seagrass beds in the UK waters (Unsworth *et al*., 2015). Improving the resilience of these ecosystems through management actions to achieve and maintain adequate water quality may provide a means of reducing such loss or degradation while securing the ecosystem services that seagrass beds provide into the future.

Active habitat restoration may play an important role in seagrass ecosystem recovery in the UK, and partnerships to restore seagrass habitats at scale are already gaining popularity[15]. Habitat suitability modelling and mapping potential restoration areas can provide useful tools for direct restoration efforts (UK Seagrass layer[16]; Swaile *et al*., 2022), however climate projections have so far largely been excluded from habitat restoration suitability mapping and ongoing restoration initiatives even though climate change impacts are likely to influence the success of habitat restoration and/or creation actions. For example, measured sediment resuspension thresholds in combination with wave predictions based on future changes in sea level and wind patterns could be used to further assess to which extent existing and planned seagrass meadows are vulnerable to changing wind conditions (Smit *et al*., 2021). Refining the measure of light availability in the UK habitat suitability models could also be used to pinpoint where management effort for water quality improvements is key to seagrass habitat

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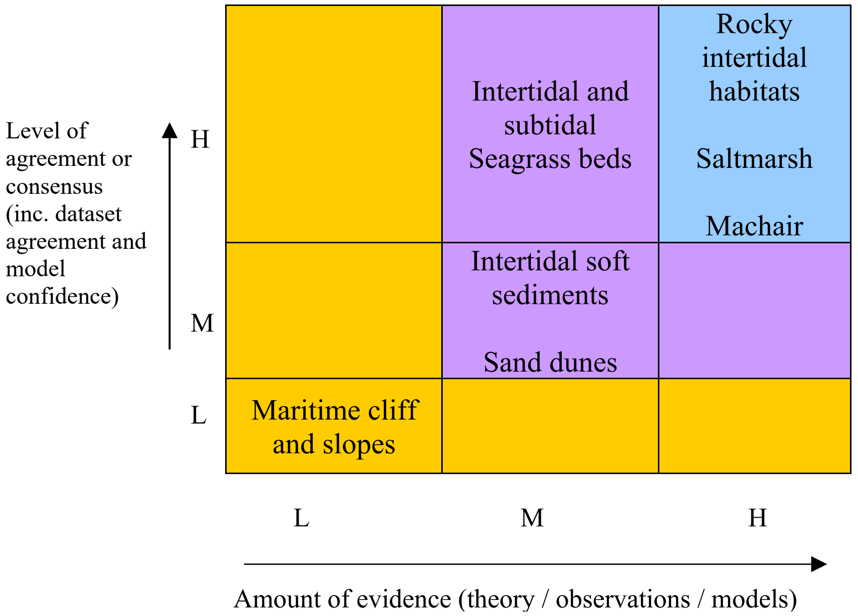
[15] The UK's biggest seagrass restoration project - Swansea University [Accessed on 8 March 2026]

[16] National Seagrass Layer (England) - data.gov.uk [Accessed on 8 March 2026]

restoration efforts now and in the future. Laboratory studies on carbon isotope fractionation in seagrass leaves and its sensitivity to light availability have enabled development of index of light availability in seagrass populations, which has been used to predict seagrass biomass in light-limited estuaries in USA (Ruesink *et al*., 2015). Further defining these parameters in habitat restoration potential models could help refining the most suitable locations for restoration efforts and could help to increase the currently low success rates of active seagrass restoration efforts.

CONFIDENCE ASSESSMENT

What is already happening?

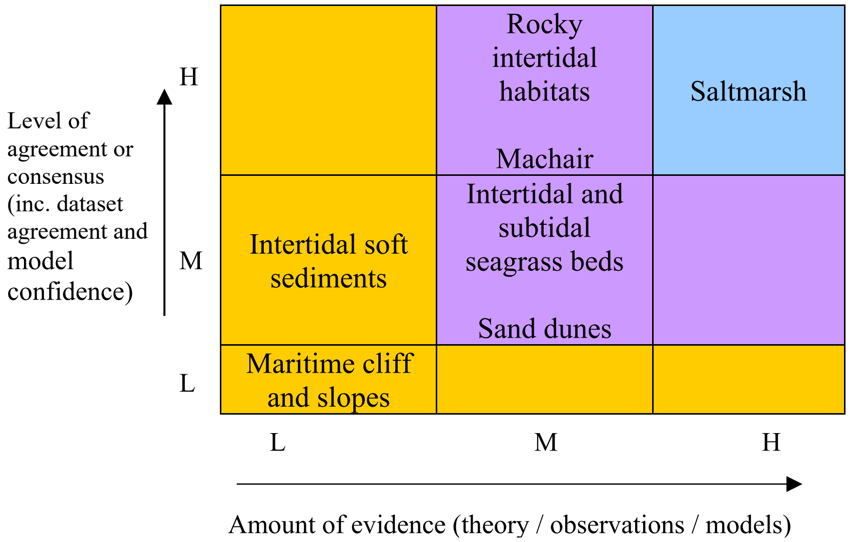


	L	M	H
H		Intertidal and subtidal Seagrass beds	Rocky intertidal habitats Saltmarsh Machair
M		Intertidal soft sediments Sand dunes	
L	Maritime cliff and slopes		

Level of agreement or consensus (inc. dataset agreement and model confidence)

Amount of evidence (theory / observations / models)

What could happen in the future?



	L	M	H
H		Rocky intertidal habitats Machair	Saltmarsh
M	Intertidal soft sediments	Intertidal and subtidal seagrass beds Sand dunes	
L	Maritime cliff and slopes		

Level of agreement or consensus (inc. dataset agreement and model confidence)

Amount of evidence (theory / observations / models)

Rationale for confidence assessments

The spatio-temporal stochasticity associated with forecasting storm events presents a major limitation to our ability to predict when and where coastal and shallow nearshore ecosystems will be affected by climate change. Significantly increased erosion and flooding frequency and duration over coming decades can be predicted with some certainty. The threats associated with extreme storms are likely to affect coastal habitats across the sublittoral, intertidal and supra-littoral range (Hanley, 2020).

As different coastal landforms have different properties and levels of sensitivity to climate impacts, they will respond differently to weather and climate events even along the same stretch of coast. This means there may be a low predictability of the net response of coastal habitats to climate impacts. Ongoing climate change is increasing the rate of coastal change. However, the response is not uniform and shows increasing variability over time (Knight, 2024).

Saltmarsh

There is high certainty of RSLR and coastal hazards affecting the geomorphology of saltmarshes, however uncertainties remain with respect to erosion, accretion and infilling relationships between saltmarshes and adjacent mudflats (Foster *et al*., 2013; Zhou *et al*., 2022). Differences in definitions of the habitat also hamper management strategies and reporting, especially when area comparisons are involved (Angus and Dargie, 2024). Nevertheless, the evidence base on climate change impacts and adaptation of saltmarsh is more comprehensive than for any other coastal habitat, and the current push to collect more data on saltmarsh to determine the potential for their inclusion in the UK Greenhouse Gas Inventory to contribute towards Net Zero greenhouse gas emission targets has resulted in a rapidly improving evidence base[17]. Evidence supporting seascape-scale restoration is also building. Furthermore, there is a high level of consensus and evidence for the effect of nutrients on saltmarsh ecology (Boorman and Hazelden, 2012; Penk *et al*., 2020), although how this may interact with climate change has a low level of evidence. Uncertainties also remain around the impact of coastal hazards on community ecology of saltmarshes on short to long-term timescales.

Sand Dunes

Sand dunes will be subject to increasing dynamism because of RSLR (high level of certainty) and higher or more frequent strong winds (low level of certainty). This dynamism is likely to involve landward translation of the dune ridge (high level of certainty, see, for example, Dynamiccoast.com) but landward sections of the dune system are likely to remain in place due to high levels of stability. Summer drought and altered rainfall patters are likely to lead to lowering of the summer water table, with a shift towards

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[17] UK Blue Carbon Evidence Partnership - Evidence Needs Statement (cefas.co.uk)
[Accessed on 8 March 2026]

drier slack communities and the possible full loss of some dune slack habitat to fixed dune grassland. This has a moderate level of uncertainty due to a lack of knowledge on the interplay between inter-annual and long-term variation in water tables and their effects on the plants involved. Due to the high level of recreational use of dunes, with facilities such as caravan sites and golf courses, there is a real possibility that attempts to protect these activities (possibly involving maladaptation) could compound impacts on dune ecosystems already being impacted by climate change, by preventing many of the natural dynamic processes which allow a degree of self-regulation.

Machair

Almost all machair lies below the 10 m contour, and in places in South Uist and Benbecula, significant areas of machair lie below the existing level of mean high-water springs (2.03 m). With increasing RSLR and increasing winter precipitation (high level of certainty), as well as the reduction in wave attenuation provided by kelp beds *Laminaria hyperborea*, even existing levels of storminess involve a higher risk of flooding by sea water or fresh water (or both). The possibility of more frequent or more serious storms (low level of certainty) increases the risk of flooding. Though the habitats within machair ecosystems already have a high level of dynamism and because of this, relatively high resilience, the main threat to biodiversity relates to adverse impacts on crofting, either directly or due to loss of motivation because of flooding. Existing records show seasonal increases in precipitation around the times of ploughing and harvest, with decreases during the summer growing season. Reductions in crop yields could lead to reductions in the low input rotational cropping that currently delivers high levels of biodiversity, leading in turn to replacement of fodder crops by imported feed, or a reduction in cattle rearing, both of which would lead to a decrease in machair biodiversity (medium level of uncertainty). Climate change trends and scenarios thus present less of a direct threat to machair biodiversity than the indirect impact posed by the adverse impacts on agricultural activity by local crofters (smallholders).

Maritime Cliff and Slopes

The varied geophysical and climatic characteristics of coastal zones make them susceptible to a range of extreme natural events, such as erosion, flooding and cliff instability which are common features of coastlines. Operating on different timescales, coastal erosion being a relatively constant, low magnitude process, whereas flooding and landsliding are higher magnitude, episodic events that are relatively more difficult to predict (Moore and McInnes, 2020).

Coastal response to sea-level rise is strongly determined by site-specific factors, rather than a global change in sea level or a regional change in wave climate. Any predictions of general coastal response due to climate change will therefore have a low confidence. In the absence of detailed studies and the limited availability of long-term coastal change data, then confidence in local or regional predictions of coastal response to climate change will be low. In the absence of a clear understanding of the coastal-change processes, and a reliable predictive tool, the default position is to assume that present-day coastal change will persist; however, it is very likely that currently stretches of coast undergoing erosion will experience increased erosion rates due to sea-level rise (Masselink and Russell, 2013).

Rocky Intertidal Habitats

Climate change impacts on the abundance and distribution of intertidal invertebrates and macroalgal species in the UK rocky shores have been monitored comprehensively through the MarClim programme, which is the most spatio-temporally extensive time-series dataset for intertidal ecosystems in the world. However, forecasting future impacts of extreme events, such as MHW and storm events, is difficult due to their stochastic, site-specific nature. When combined with interacting anthropogenic impacts, future predictions on climate change impacts on rocky shores become less certain, however, the use of CTI analyses is showing some promise in addressing this uncertainty (Burrows *et al*., 1019).

Intertidal Soft Sediments

Impacts of gradual long-term climate change are often significantly smaller than the short-term variability characteristic of these environments, which makes them difficult to detect. The coupled biophysical system has yet to be fully integrated and addressed holistically. This is especially important for future assessment where several reinforcing or competing effects combine. The importance of local context makes transferability and generalisation of case studies difficult.

Intertidal and Subtidal Seagrass Beds

For seagrasses, there is comprehensive empirical evidence on the physiological responses of different species to climate change impacts including warming and ocean acidification from laboratory, mesocosm and field studies from across the world (Palacios and Zimmerman, 2007; Jiang *et al*. 2010, Zimmerman *et al*., 2017; Liu *et al*. 2020). Predictive models of organism- and population-level responses of seagrasses to future climate change impacts (Zimmerman, 2021) also imply high confidence in climate change projections. However, studies from UK waters, on UK-specific intertidal and subtidal species (*Z. marina* and *Z. noltii*) are more limited, especially on the cumulative impact of multiple pressures (climate change and human induced) on existing seagrass beds and on areas suitable for seagrass habitats restoration. The adaptive response of seagrass ecosystems to climate change in UK waters are also less understood.

KEY CHALLENGES AND EMERGING ISSUES

Nature-Based Solutions

Nature-based solutions (NBS) describe actions to protect, sustainably manage and restore natural and modified ecosystems to address societal challenges effectively and adaptively for the benefit of human well-being and biodiversity (IUCN, 2016). NBS have become the focus of climate change adaptation and mitigation discussions in the UK and globally since UNFCCC COP27 where, for the first time, the decision text included encouraging Parties to consider NBS or ecosystem-based approaches.

Within coastal ecosystems, NBS tend to centre around utilising natural elements and processes to enhance holistic flood and coastal erosion risk management for the benefit of local communities (Environment Agency, 2017; Moraes *et al*., 2022; Van der Meulen, 2022). Despite the growing popularity of coastal NBS, traditional hard engineering approaches are still being adopted in many urbanised locations because NBS are often not socially, technically or economically feasible. In these cases, alternative approaches of 'greening the grey' are used to improve the multifunctionality and ecological value of hard (engineered) coastal and estuarine infrastructure (Apine *et al*., 2024; Naylor *et al*., 2017). With modified UK coastal areas being particularly vulnerable to the impacts of changing climate, hybrid solutions that combine NBS with hard coastal defences have been shown to provide the best outcomes for hazard risk reduction, cost-effectiveness and climate mitigation (Huynh *et al*., 2024). NBS should therefore be considered as part of a wider portfolio of coastal management strategies (Knight, 2024).

In the UK, the increasingly popular NBS for coastal, intertidal and shallow nearshore ecosystems include restoration and creation of coastal wetlands including saltmarsh habitats, dune management, beach nourishment, and restoration and creation of intertidal and shallow nearshore habitats such as seagrass beds (Defra, 2022). In addition to flood risk and coastal erosion alleviation, habitat restoration within the UK coasts in gaining momentum to unlock the multitude of additional natural capital benefits that these habitats provide, including biodiversity protection and enhancement, increased carbon sequestration and storage potential, provision of fish nursery habitat, and improving nutrient and turbidity levels. A review of studies worldwide suggests that saltmarsh planting (*Spartina*, *Juncus*, *Scirpus*, *Salicornia* and *Suaeda*) can enhance ecological functions related to climate change mitigation including shoreline protection, productivity and carbon storage and demonstrate potential to achieve equivalence to natural marshes after five to 25 years (Liu *et al*., 2024).

The Restoring Meadow, Marsh and Reef (ReMeMaRe) initiative[18] has enabled a growing community of practitioners, researchers and government agencies to come together in a common cause to restore at least 15% of our

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[18] Restoring Meadow, Marsh and Reef (ReMeMaRe) | Estuarine & Coastal Sciences Association (ecsa.international) [Accessed 19 January 2026]

coastal and estuarine habitats in England for nature and people by 2043. ReMeMaRe has produced valuable tools and guidelines to standardise the approaches for local restoration initiatives, from habitat restoration principles and handbooks to existing habitat extent and restoration potential maps[19]. Understanding how to 'future proof' restoration efforts to secure naturally functioning ecosystems that are resilient to changing climate is fundamental for unlocking the natural capital benefits associated with coastal and marine habitat restoration. Furthermore, although the focus of restoration programmes is often on the active habitat creation, the importance of conserving existing habitats through removal of pressures is likely to bring the largest benefits for climate change mitigation and adaptation (Smeaton, 2023). Further details on novel research and delivery projects on habitat specific restoration are given in the relevant habitat sections above.

Seascape Approach to Restoration

Restoration practice is currently dominated by single habitat approaches underpinned by single species monocultures, potentially limiting the range of benefits that restoration can provide. There is increasing evidence to suggest that for ecosystem restoration to meet its full potential in delivering resilience to climate driven environmental change, restoration practices should plan beyond single species and single habitats to a multi-habitat seascape (McAfee *et al*., 2022; Zabin *et al*., 2022). Where multiple habitats are co-restored, their positive interactions mutually benefit each other to stabilise and even accelerate ecosystem recovery of connected habitats such as interacting patches of seagrass meadow, oyster reefs (McAfee *et al*., 2021b) and saltmarsh (Derksen-Hooijberg *et al*., 2018), creating living shorelines and generating biological feedbacks which reduce hydrodynamic stress, promoting sediment accretion and propagule settlement (Smith *et al*., 2020).

The concept of seascape restoration has emerged to restore multiple degraded coastal and inshore habitats within an ecosystem in unison to tackle the biodiversity and climate crises (Garbutt, 2024). Healthy seascapes are characterised by many interacting species and interdependent habitats that co-create ecological functions to build stability and resilience at seascape scale. A new Seascape Restoration Statement[20], adopted by the ReMeMaRe initiative, aims to support bigger, better and more connected restoration at the seascape scale in English estuaries and coastlines. The Solent Seascape Project[21] is pioneering large-scale co-restoration of coastal habitats and investigating synergistic effects of this approach.

Fundamentally, a multi-pressure approach combined with the seascape approach is needed for better restoration outcomes at ecosystem and seascape levels. Addressing the key local manageable pressures common to

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[19] Tools and guidance | Estuarine & Coastal Sciences Association (ecsa.international) [Accessed 19 January 2026]

[20] ReMeMaRe 2023: Seascape Restoration (office.com) [Accessed on 19 January 2026]

[21] Home - Solent Seascape [Accessed on 19 January 2026]

multiple habitats, such as nutrient enrichment, coastal development, and recreational mooring, is critical for practical conversation, restoration and management actions and will improve ecosystem resilience to climate change (Tregarot *et al*., 2024). Restoration efforts should also consider locations where future habitats will exist (Lawler *et al*., 2020). Ongoing and unprecedented global environmental change may ultimately require new ways of doing restoration that are fit for the future (Bullock *et al*., 2022).

The Role of Genetics in Future Proofing Marine Conservation and Recovery

Better understanding of species traits and thermal tolerances are needed to make more robust predictions of future impacts of climate-driven changes. This includes sustaining time-series that are tracking climate-driven changes in intertidal biodiversity. In the UK, genomic sequencing of species, including those associated with vulnerable coastal and intertidal habitats is undertaken by the Darwin Tree of Life Project. Ecological genomics studies will allow a greater understanding of the genetic mechanisms underpinning of species responses to climate change.

The role of genetic diversity may be important in determining whether protected and/or restored habitats such as intertidal and subtidal seagrass beds will survive and thrive. A study on the genetic diversity of the UK seagrass beds found four genetic clusters for *Z. marina* which were connected by gene flow but some of which had low genetic diversity (Finger and Lilley, 2024). Although the populations are not genetically isolated and thus potential local adaption by mixing seed sourced from different populations is unlikely to be disrupted, the seed and transplant sources for restoration should match the genetic cluster groups to ensure that restoration sites will consist of plants best suited for their environment. For *Z. noltii*, more data is needed before a UK-wide comparisons can be made (Finger and Lilley, 2024).

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