

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

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EXECUTIVE SUMMARY

- The lack of dramatic observed responses of intertidal species from 2002 to 2018 is consistent with the lack of increase in sea temperatures since 2000. Despite the continued global upward trend in temperature, UK regional sea temperatures have remained stable or declined over the same period, with only those in western Scotland increasing. This hiatus followed a period of rapid warming from 1980 to 2000 when many range shifts occurred.
- Short-term fluctuations in abundance of climate-sensitive species have continued through the mid-2010s, and most of these changes in abundance are consistent with changes in temperature at timescales of less than a decade, offering further evidence for the continued sensitivity of intertidal species to climate.
- Leading range edges of Lusitanian topshells are continuing to move northwards in North Wales and south-east England. The Community Temperature Index (CTI) shows considerable promise for use as a measure of the combined response of multiple species to climate change. Preliminary analysis of spatial patterns in UK rocky shore communities shows that the CTI follows temperature closely, and that changes in sea-surface temperature over time are matched by changes in CTI.
- The lack of an upward trend in UK sea temperatures is unlikely to continue, since longer-term trends appear to be less regionalised than short-term ones, and further changes are likely to be more dramatic in the next decade as local temperatures catch up with global trends.
- A heatwave event in summer 2018 caused heat damage to the high and midshore fucoids in UK regional seas.

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1. WHAT IS ALREADY HAPPENING?

Rocky intertidal habitats

Whilst many species are undergoing oscillations in population abundances across the UK sections of their biogeographic ranges, few species are showing changes in the leading or trailing ranges of their distributions.

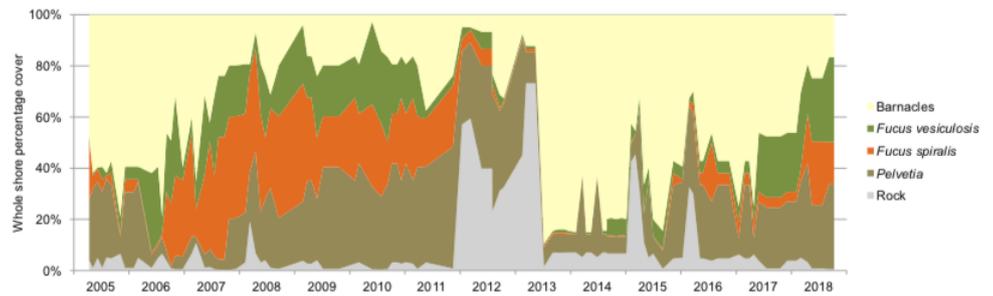
Repeated surveys in 2014 and 2015 of rocky shore sites first visited in the early 2000s (2002–2006) around Scotland revealed a general increase in large brown canopy-forming algae, shown by significantly more-frequent increases than decreases in abundance (Burrows *et al.*, 2017b). *Fucus vesiculosus*, for example, increased at 74% of the 154 sites surveyed, by an average of two abundance categories. *Fucus spiralis* increased at 55% of sites and declined at 24% of sites, while *Fucus serratus* increased at 66% of sites and declined at 18% of sites, both species by an average of around one abundance category. Mid-shore *Ascophyllum nodosum* and high-shore *Pelvetia canaliculata* increases were also significantly ($p < 0.05$) more frequent than decreases, albeit with a smaller average magnitude of change (less than half of an abundance category). While these changes are consistent with responses of cold-water species during a period of slight cooling (Burrows *et al.*, 2017a), care must be taken to view these as part of longer-term trends. Intertidal fucoids are very variable in abundance, even cyclic, on short time scales (Hartnoll and Hawkins, 1985; Little *et al.*, 2017), with fluctuations tied to positive and negative feedbacks in their population dynamics and those of their main limpet grazers (Southward and Southward, 1978). Photographs of an intertidal outcrop near Oban, Scotland, repeated every 1–2 months since 2005, show that large changes in intertidal seaweeds can happen over periods of less than a decade (Figure 1).



(a)

(b)

Figure. 1a, b: Changes in intertidal seaweeds at a single site near Oban since 2005. (a) Less macroalgae on 7/6/2005, (b) increased cover on the upper shore 27/6/2018.



(c)

Figure 1 – contd. (c) Cover of main space-occupying taxa estimated from photos taken at 1–2 monthly intervals of the above reef showing two periods of increased algae (2008–2012 and 2018 onwards) interspersed with periods of dominance by barnacles.

Kelp abundances around the UK have shown a shift towards increasing abundance of Lusitanian species including *Sacchoriza polyschides* and *Laminaria ochroleuca* in recent decades in response to warming of the marine climate (Brodie *et al.*, 2014; Mieszkowska *et al.*, 2006b; Smale *et al.*, 2015). Boreal kelps have not shown a significant decline in abundance to date; however, they are predicted to decline in abundance and undergo range retractions with continued climate change (Brodie *et al.*, 2014). A meta-analysis of intertidal brown macroalgae around the UK coastline found regional differences in abundance trends, with declines in the southern region, but no change or increases in central and northern regions of the UK (Yesson *et al.*, 2015).

The Lusitanian toothed topshell *Phorcus lineatus* has undergone a further extension of the northern, leading range edge in North Wales since 2014, with isolated individuals recorded at Rhos-on-Sea and Llanddulas for the first time in 2015; a small, multi-age population was recorded at both sites in 2016, 2017 and 2018 and individuals were found at Penmaenmawr for the first time in 2019. One adult individual was also found at Little Orme for the first time since the only previous records in 2009 and 2010. Isolated individuals of the Lusitanian species of limpet *Patella depressa*, and barnacle *Chthamalus stellatus* have been recorded for the first time on MarClim sites in the eastern English Channel in 2017 and 2018 (Mieszkowska 2018b; MarClim, unpublished data).

Multidecadal cycles in relative abundances of the boreal barnacle *Semibalanus balanoides* and warm-water barnacle *Chthamalus* spp. are strongly correlated with both local sea-surface temperatures, and a ‘Warm Index’ of barnacle abundance shows strong links to the basin-scale Atlantic Multidecadal Oscillation. In contrast, there are weak or no observed relationships with the North Atlantic Oscillation for either species. The shorter lifecycle of *S. balanoides* compared to the chthamalids and the increase in spring and summer temperatures to which newly settled *S. balanoides* recruits have been exposed during the last decade are likely

mechanisms by which barnacle densities are responding to low-frequency temperature variability expressed in the Atlantic Multidecadal Oscillation (Mieszkowska *et al.*, 2014a).

Biogenic reefs

Biogenic reefs are important habitat providers and contribute to coastal protection. The honeycomb worm *Sabellaria alveolata* is a Lusitanian warmwater sedentary tube-dwelling polychaete that creates tubes from shell fragments and suspended sediment (Wilson, 1971) forming extensive reefs (Figure 2) that provide important habitat for many intertidal species. *S. alveolata* is covered by Annex 1 of the EC Habitats Directive. In the UK, the presence of this species is limited to the western peninsulas. Surveys in the 1980s (Cunningham *et al.* 1984), 2000s (Frost *et al.* 2004) and long-term monitoring as part of the MarClim project shows how *S. alveolata* disappeared from the North Wales and Wirral coastlines after the cold winter of 1962/3, with population declines also occurring after the cold winters of 2009/10 and 2010/11 (Firth *et al.* 2015). In the 2000s and 2010s *S. alveolata* has recolonised locations from where it had previously disappeared, and increased in abundance at many shores, possibly in response to recent warming.

S. alveolata has been recorded on many artificial coastal defence structures (Firth *et al.*, 2015, MarClim data), demonstrating another impact of climate change on this species via proliferation of coastal defences to help combat increased storminess, rising sea levels and habitat loss.



Figure 2: *S. alveolata* reef at Bude, south-west England.

Invasive non-native species

A brief overview of the main changes related to invasive non-native species (INS) and intertidal habitats presented here. Artificial man-made habitats can often support higher densities of INS than natural rocky shores due to reduced competition from established native species, more-vacant habitat and year-round settlement allowing opportunistic colonisation of vacant space (Mineur *et al.*, 2012).

The red alga *Gracilaria vermiculophylla* has invaded soft-sediment intertidal estuarine systems at Dundrum Bay and Carlingford Lough in north-eastern Ireland since 2008. New populations were recorded in Christchurch Harbour and Brownsea Island in Dorset, and Kingsbridge Estuary in Devon in 2015, and the River Glaslyn in North Wales in 2017 with DNA barcoding carried out on all populations to confirm species identity (Krueger-Hadfield *et al.*, 2017).

The red alga *Caulacanthus okamurae* has been recorded in recent years on rocky intertidal habitats along the English Channel coastline and the Celtic sea (Mieszkowska, 2018a; MarClim, unpublished data). Introduced into southern Biscay from the north-west Pacific in the 1980s, *C. ustulatus* has spread to many other parts of the Atlantic coast and was recorded in the Normano-Breton Gulf in 2005 (Department of the Environment, 2017).



Figure 3: *Caulacanthus okamurae* at Tinside, south-west England.

The invasive kelp *Undaria pinnatifida* was recorded at Neyland in Milford Haven in 2017 (Mieszkowska, 2017). A new MarClim long-term monitoring site was set up at this location and *U. pinnatifida* was recorded there again in the 2018 annual survey. No other new records of *U. pinnatifida* have been recorded around UK regional seas in the past few years; however, existing populations still maintain their population densities (Araújo *et al.*, 2016; MarClim, unpublished data).

The Pacific oyster *Magellana gigas* has been farmed in estuaries around the UK since the 1960s. As recently as 2002, waters were still considered to be too cold for it to reproduce and therefore present a substantial risk of invasion to natural systems (Herbert, 2012). In recent years, *M. gigas* has settled on natural habitat outside of farms and spread around the UK coastline as warmer temperatures have facilitated survival and reproduction in wild populations (Campbell, 2012). The worst affected area is in the eastern English Channel, where dense beds of *M. gigas* now occur. An intensive study of a wild population in south Devon confirms that the population is reproductively active and self-sustaining (Campbell, 2012; MarClim, unpublished data). Natural estuarine and open-coast colonisation has increased over the last two years in the vicinity of oyster farms in the western English Channel and Scotland. Evidence suggests that the risk to biodiversity from wild settlement of Pacific oysters relates not so much to local changes in species diversity *per se* but to the extent of habitat transformation (Herbert *et al.*, 2012). *M. gigas* has colonised chalk platforms along the eastern section of the Eastern Channel and Southern North Sea, including Ramsgate, Margate, Forelands and the Thanet coast in the last decade (Herbert *et al.*, 2012; MarClim, unpublished data).

The sea squirt *Corella eumyota* has been recorded in North Wales in previous years since 2012, however, it had disappeared from the MarClim site at Menai Bridge in 2016 and was not present in 2017 or 2018 (Mieszkowska, 2018c; MarClim, unpublished data).

Extreme weather events

The ‘Beast from the East’ and Storm Emma that occurred in the winter months at the start of 2018 did not have any discernible effects on rocky intertidal species. A heatwave event occurred in the summer of 2018. The fucoids *Pelvetia canaliculata*, *Fucus spiralis* and *Fucus vesiculosus* that occur in the high- and mid-shore showed physical evidence of heat damage, with tissues turning brown (Figures 3 and 4). No large mortality events were observed at any of the MarClim long-term time-series sites around the UK coastline, and effects were site-specific, with not all species being affected at all sites. Heat-damaged algae were recorded in the north-east and south-west of England, North Wales and the Isle of Man during 2018 MarClim time-series surveys.



Figure 4: Heat damaged *Fucus spiralis* at Souter, north-east England.



Figure 5: Heat damaged *Fucus vesiculosus* at Mountbatten, south-west England.

Coastal defences

The threat of flooding, sea-level rise, storm surges and coastal erosion has led to an increased amount of artificial structures being installed to defend our coastline. Artificial structures cover 29.8% of the English coastline and 7.6% of the coastline of Ireland (Masselink and Russell, 2013). Little research has

been published since the last MCCIP report in 2013 about the impacts of artificial structures on intertidal biodiversity around the UK coastline (Mieszkowska *et al.* 2013). In the south-east of England, artificial coastal defences have acted as stepping stones for rocky intertidal species across areas of soft sediment habitat, with species including *Patella depressa*, *Steromphala umbilicalis* and *Melaraphe neritoides* using structures to breach habitat barriers and colonise natural rocky habitat where they could not previously reach via natural dispersal (MarClim data).

Community Temperature Index (CTI)

Linking changing abundance of species to their thermal affinity allows the assessment of the effects of warming for species assemblages at defined locations. This pattern can be summarised into a single measurement using the Community Temperature Index (CTI), the weighted average of the thermal affinities of the species that make up the community, where w_i is the weight for species i , and N is the number of species in the community (in single surveys or aggregated across areas and years), STI is Species Temperature Index.

$$CTI = \frac{\sum_{i=1}^{i=N} STI_i w_i}{\sum w_i}$$

Weighting by abundance or frequency of occurrence allows the effects to be assessed of changing relative abundance without necessarily a change in species composition. CTI for rocky shore communities tracks changes in coastal sea temperature over time, especially over longer periods, during both warming and cooling. Thermal *affinities* of rocky shore species were obtained by matching their global distributions with average coastal sea-surface temperatures to give the Species Temperature Index (STI), here the median sea temperature experienced, and Species Thermal Range (STR), a measure of the spread of temperatures occupied by each species (Burrows *et al.*, 2017a).

Spatial patterns in Community Temperature Index on UK and European rocky shores presented here are based on data recorded by the MarClim programme. Trends in CTI over time used data collected by repeated visits to sites in the MarClim programme since 2001 (Mieszkowska *et al.*, 2006; Mieszkowska *et al.*, 2014b) and collected by the SOTEAG (Shetland Oil Terminal Environmental Advisory Group) environmental monitoring programme around Sullom Voe since 1976 (Burrows *et al.*, 2002; Hiscock, 1981).

CTI for animal species showed a much higher spatial turnover and a much greater sensitivity to sea surface temperature than CTI for algae. Community Thermal Diversity (abundance-weighted standard deviation of species

thermal midpoints, STIs) was higher for communities of animal species than for plants and algae. Average temperature range (Community Thermal Range) was similar for plant and animal communities. Values of Community Thermal Bias (CTI minus local temperature) showed that animal communities were generally composed of species whose thermal affinities were centred in waters 1–2°C warmer than the sites surveyed, while communities of plants and algae tended to comprise species with STIs 1–3°C colder than local temperatures.

CTI declined by 0.5°C in south-west England and Wales over 17 years from 2002 to 2018, with most of the decrease in the first 10 years. CTI increased in Shetland by a similar amount, steadily rising from 1980 onwards after an initial decline post 1976. Changes in CTI reflected species abundance changes in the two regions. The pattern of changes in species abundance in south-west Britain relative to their thermal affinities (STIs) generated CTI changes that were consistent with a shift towards a community dominated by cold-water species.

Negative thermal bias, where the community is dominated by cold-water species and potentially vulnerable to the effects of warming (Stuart-Smith *et al.*, 2015), was seen for communities of macroalgae across most of the UK, increasing from eastern Scotland and north-east England towards south-west Britain and Ireland. This suggests greater negative effects of climate warming on intertidal macroalgae in the south-west of the UK. Observed changes in abundance of large brown algae in the UK over recent times match this expectation, with declines in abundance seen only in the English Channel and Celtic Sea and no changes in central and northern regions (Yesson *et al.*, 2015). The trends in abundance of macroalgae in this study follow this pattern in south-west Britain, with macroalgae increasing during a period of stasis or slight cooling since 2002. Animal communities in wave-sheltered conditions tend to be composed of species with cooler thermal affinities than in wave-exposed conditions, not seen in algal communities.

Communities of intertidal animals, by contrast, have mostly positive thermal bias across the study region suggesting that intertidal animals may benefit from the effects of warming since, on average, they have distributions centred in warmer areas. Warm-water animal species declined with cooling in the south-west, but the main contribution to changing CTI in Shetland was the decline in cold-water animals with warming.

Despite the weak response to annual changes in average annual sea surface temperature, compositional changes shown by CTI were broadly in line with the longer-term trend in temperature. The spatial pattern of CTI for animals is better aligned to the east–west gradient of winter temperatures in the UK, while CTI for algae better matches the north–south gradient of summer

temperatures, suggesting the limiting processes may occur in different seasons for the two groups.

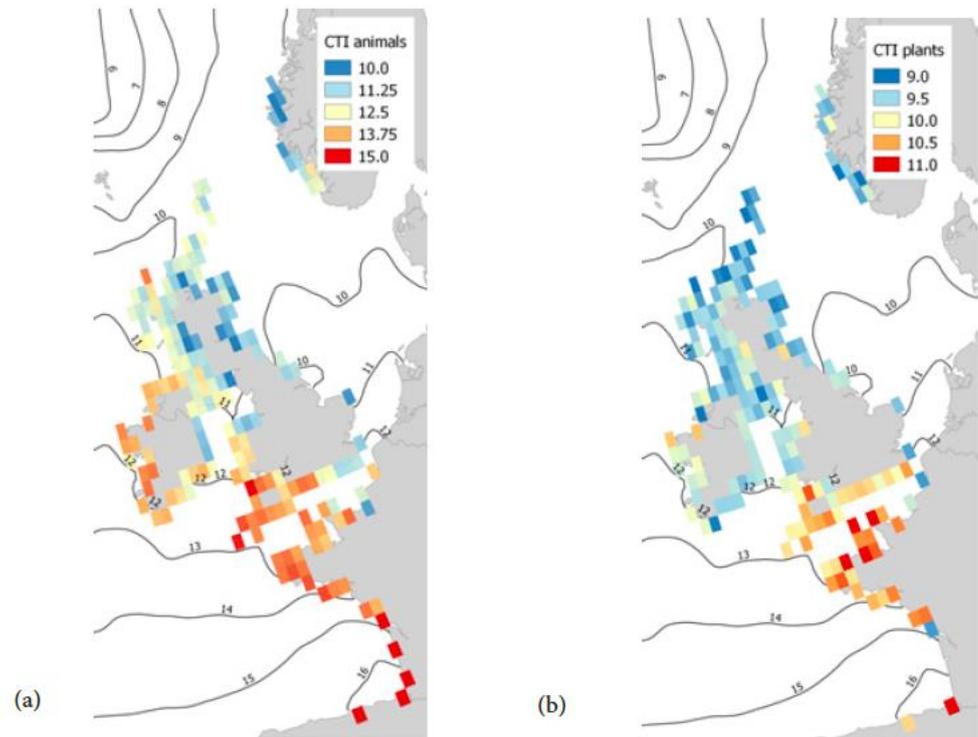


Figure 6: Geographical patterns in CTI for (a) animals and (b) plants and algae. Community Temperature Index values are shown as averages in 0.5°-degree grid cells. Isotherms are annual sea-surface temperature at 1°C intervals from the NOAA OISSTv2 HR dataset. (Burrows *et al.*, 2017a.)

Saltmarsh habitats

Over the past 50 years, re-alignment schemes have been used to restore degraded saltmarsh habitats and create new areas for saltmarsh plants to colonise. Some of these schemes are still ongoing, for example in Northumberland. This restoration process is slow, with varying levels of success with biodiversity difficult to restore and carbon storage equivalent to natural sites occurring over approximately 100 years or more (Adams *et al.*, 2012, Mossman *et al.*, 2012, Burden *et al.*, 2013). Saltmarsh habitats show nuanced responses to climate change, including sea-level rise and average temperature increases. In addition, changes in geographic tidal amplitude and sediment texture have local impacts on the characteristics of saltmarsh habitats (Skov *et al.*, 2018, Goudie, 2013). There have been estimates of loss in cover of marshes across the British Isles (Mieszkowska *et al.*, 2013) but despite this, marshes appear to be keeping pace with sea level rise (Cahoon *et al.*, 2000, van der Wal and Pye, 2004). In keeping with national trends there

is slightly more floral diversity in south-east marshes compared to north-west marshes and long-term time-series suggest little change in these assemblages (Haynes, 2016). There has been a noticeable decrease in nesting birds in marsh areas over the past decade. One possible driver could be the increase in abundance of the non-indigenous species *Spartina anglica* (Malpas *et al.*, 2013, Sharps *et al.*, 2015, 2016, 2017), although other factors including terrestrial land use change may also be responsible. More work on the fauna associated with saltmarsh habitats is needed to establish the current status of the faunal community living in and associated with these habitats (MCCIP, 2018; Skov *et al.*, 2018).

Soft-sediment intertidal habitats

There have been few clear indications of the effects of climate change on soft-sediment communities in the UK to date, but this is due in part to the inherent difficulty in making observations in these habitats as opposed to, for example, the intertidal rocky shore. Many species remain hidden in the sediment and quantifying changes in distribution and abundance is relatively difficult and it is not at present possible to provide a comprehensive review of regional differences in climate impacts. Models of future climate impacts are being developed based on historical alterations in estuarine structure and subsequent erosion and increasing coarseness of grain size due to local geomorphological changes and mesocosm approaches are yielding predictions on impacts of climate change variables (temperature; atmospheric CO₂) on biodiversity-ecosystem functioning (Bulling *et al.*, 2010). These indicate that increasing temperature and atmospheric CO₂ reduce nutrient levels and have a negative effect on marine invertebrate biodiversity, although the interactions between various drivers are complex. There are some obvious recent changes though, in relation to the spread and extent of the Pacific Oyster *Magellana gigas*. This species was first introduced to the UK in 1890, however, it was the reintroduction in 1962 under licence from MAFF for aquaculture that resulted in the successful invasion of natural rocky and soft sediment intertidal habitat in the UK (Walne, 1971; Herbert *et al.*, 2012). Whilst this species attaches to hard substrata on the rocky shore, it is now found in dense beds associated with soft sediment in estuaries, including the River Thames estuary as is the invasive clam *Venerupis philippinarum* (Worsfold, pers. comm.). Settlement in *Magellana* species is facilitated by a conspecific chemical cue (Pawlik, 1994) and it is likely that the establishment of a few pioneers leads rapidly to further settlement and colonisation. Climate projections are thought likely to result in *M. gigas* successfully recruiting annually in south-west England, Wales and Northern Ireland by 2040 in response to continually warming marine environmental temperatures (Maggs *et al.*, 2010). The risk to human health from this commercially farmed species is likely to be greater than the risk to native marine biodiversity. Much climate-related change is occurring beside shifting species' range margins, with likely effects on the functioning of ecosystems, including the structure of marine food webs.

Ocean acidification

Ocean acidification poses a growing threat to marine life, with generally negative effects on shallow-water organisms, particularly shell-building organisms, and sometimes positive as for some species of non-calcifying macroalgae (Connell and Russell, 2010; Kroeker *et al.*, 2013). Year-long mesocosm studies on effects of ocean acidification on rocky intertidal species, including *Nucella lapillus*, have proved useful in showing how acidification may affect the biology of the species over ecologically relevant timescales (Queirós *et al.*, 2014). Complex effects including increased feeding rates but greater distances covered while feeding have been included in models that scale up to population abundance by the end of the 21st century, predicting widespread declines with knock-on impacts on rocky intertidal communities. Similar studies involving both short- and long-term experiments on other intertidal organisms continue to produce positive and negative effects on traits related to ecological performance.

Mesocosm experiments investigating species interactions have shown that grazing activity of *Littorina littorea* altered under warmer, more acidic seawater conditions, but this was driven by a change in species composition of the biofilm on which they were grazing (Russell *et al.*, 2013), highlighting the importance of indirect effects of anthropogenic drivers on marine species.

Socio-economic impacts

The National Ecosystem Assessment for the UK carried out in 2011 identified the range of ecosystem goods and services that marine systems offer to UK society, including food, blue carbon, blue biotechnology, coastal protection, tourism, cultural heritage, physical and mental health benefits and learning experience (UKNEA, 2011). The National Ecosystem Assessment for the UK Follow-On quantitatively values additional ecosystem services including the identification of the role of intertidal ecosystems as natural capital, examining their features and the role that they play in providing natural capital (UKNEAFO, 2014).

2. WHAT COULD HAPPEN IN THE FUTURE?

There is still a degree of uncertainty related to how coastal systems will be affected by ocean acidification due to the lack of data, the predominantly short-term durations of experimental research, and the related problems with scaling up observations to predict future impacts (Connell and Russell, 2010; Porzio *et al.*, 2011; Dupont *et al.*, 2013). Interactions with changes in sea temperature are likely to exacerbate the decline in some boreal cold-water species, however, the species-specific nature of observed changes in intertidal species in response to both climate change and ocean acidification

demonstrates that forecasting future changes will not be a simple process that can be applied across all species.

Rocky intertidal habitats

Further extensions of the leading range edges may occur on both natural and artificial habitat if sea and air temperatures begin to increase again around the UK regional seas. Population abundances across UK coastlines are likely to undergo natural oscillations due to lifecycle events, and Lusitanian species may continue to expand their leading range edges if the marine climate begins to warm again.

Saltmarsh habitats

The saltmarsh pioneer, *Sporobolus (Spartina) anglica*, may extend its geographical range northwards as temperature rises. *S. anglica* colonises lower onto intertidal flats than other marsh pioneers, and marsh areas in the north could expand as a consequence (Loebl *et al.*, 2006). The rapid colonisation of *S. anglica* over tidal flats in sites with large wintering populations of waders and wildfowl is of concern because of the birds' loss of habitat for feeding and roosting (Davidson *et al.* 1991). It usually produces a monoculture which has much less intrinsic value to wildlife than the naturally species-diverse marsh (Davidson *et al.*, 1991). *S. anglica* is not the major dominant species in most UK marshes, however, once it has become established at a location then accretion occurs, causing successional community change.

Long-term monitoring of marshes is crucial for successful adaptive management and needs integration across sites, regions and connected systems. Long-term monitoring has the added benefit of preventing misinterpretation of natural marsh changes that may be part of natural long-term cycles (e.g. Pringle, 1995).

Intertidal soft sediment habitats

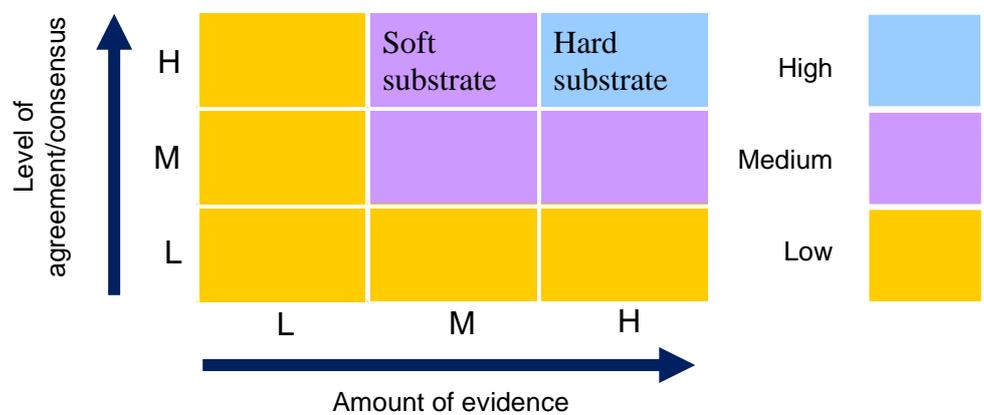
There is insufficient information to be able to currently determine what impacts climate drivers, such as increased temperature, decreasing pH, increased wave fetch, altered precipitation and salinity, will have on these ecosystems.

The effects of discrete extreme temperature events were investigated as a driver of change by simulating heat waves in a large outdoor mesocosm system designed to preserve natural sediment temperature profiles, solar and tidal cycles, and faunal densities. Community composition effects were not identified overall nor was the number or abundance of shallow dwelling organisms that may be more vulnerable to extreme temperatures at the sediment surface. For the polychaete *Alitta virens* and the bivalve

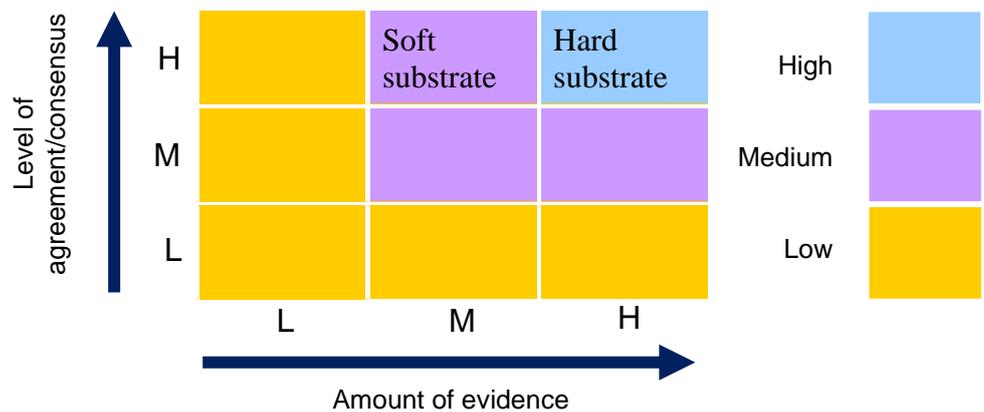
Cerastoderma edule, which exhibit different burrowing abilities, neither species exhibited higher mortality as a result of the heat wave simulations performed. Changes in energy reserves, however, suggested sublethal effects for both, which has implications for their vulnerability to the increased frequency, intensity, and duration of these events predicted for the future (White, 2018).

3. CONFIDENCE ASSESSMENT

What is already happening



What could happen in the future?



The confidence assessments have been done separately for hard and soft substrate as they differ in terms of both consensus and amount of evidence. More information and agreement is available for hard substrate systems.

4. KEY CHANGES AND EMERGING ISSUES

1. *Continued funding of time-series.* Funding for sustained monitoring and surveys is not provided by UK Research and Innovation, and Statutory Nature Conservation Body (SNCB) funding is declining each year. Without continued funding, the time-series that are tracking climate-driven changes in intertidal biodiversity are under imminent threat of being stopped.
2. *Increase in soft sediment benthic biodiversity monitoring.* A few isolated monitoring schemes exist around UK regional seas but these do not cover a wide geographic area, making changes difficult to detect.
3. Using the best scientific information available to support decision-making is fundamental to the implementation of national and international policies on conservation of biodiversity and sustainable use of resources. Reliable information and adequate scientific data to support the knowledge needs of different groups of stakeholders and decision-makers is not, however, always available.

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