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Intertidal Habitats and Ecology

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EXECUTIVE SUMMARY

Multiple climate-related drivers including temperature, sea-level rise, storminess and wave height are being documented to cause alterations in regional biodiversity, with increases in southern regional seas as warm water species extend their distributions faster than cold water species are retreating. As species are lost from areas, biodiversity is likely to decline, especially in habitats such as saltmarshes and seagrass beds where restricted connectivity between systems may limit propagule dispersal and inhibit the sustainability and recovery of impacted habitats. Although rocky intertidal habitats have a greater degree of connectivity, they are also at risk of regional biodiversity change as populations become established or disappear.

Changes in geographic distributions of intertidal species are continuing, with northern range limits of southern species in rocky intertidal habitats continuing to extend during the last 2 years. For example, the range limits of some southern species have moved up to 12km further north (e.g. *Osilinus* species) between surveys undertaken in July 2007 and July 2009. Population abundances of the topshell *Gibbula umbilicalis* have increased throughout the UK and in warmer southern areas they have switched to having two periods of gonad maturation per year (uni- to bivoltine). This was observed for the first time in 2008/2009. Such a strategy is more characteristic of populations inhabiting warm waters and lower latitudes. If the current responsive trend continues, shifts in dominant species of different taxa are likely to occur within the next decade. Established populations of cold water fauna are showing declines in abundance in the western Channel, but are still undergoing annual recruitment. The available evidence suggests that climate is acting on the adult reproductive stages and the post-recruitment stages of juveniles. Such information is not available at present for many soft sediment intertidal species.

Saltmarsh habitat is declining in the UK due to coastal squeeze, resulting from erosion at the seaward end of saltmarsh beds by increased storminess and relative wave height, and prevention of landward retreat by coastal defences built to combat increased flooding from storm surges. This is being offset in some areas by managed realignment and habitat recreation. Mudflats are at risk of rising sea levels and erosion, but more information is required to quantify the impacts on benthic composition and biomass. The dynamics of seagrass beds are different depending on their regional location, with increases and decreases in spatial extent occurring since 2007. Insufficient evidence is currently available to make a direct link between climate change and alterations in spatial coverage of Zostera habitats. There is little direct evidence for current impacts of climate change on soft sediment communities, but model predictions indicate a future loss of biomass and biodiversity.

Alien species already present on natural intertidal habitats are increasing in abundance. The few impacts on native species studied to date are predominantly negative but caution is urged in assuming that all introductions have adverse impacts for native communities, as some species can actually increase biodiversity.

Rocky intertidal systems are unlikely to be negatively impacted to a large degree by sea-level rise as most UK rocky habitats have sufficient habitat above existing high water springs levels to accommodate vertical shifts in biota in response to rising sea levels. Community

compositions are changing due to loss or gain of species and changes in relative dominance, which have the potential to alter structure and functioning of rocky ecosystems. Artificial 'stepping stones' created by increasing numbers of coastal defences along areas of soft sediment coastline have led to species artificially extending their range and subsequent colonisation of natural shores beyond these regions, and more species are expected to use this route across areas of unsuitable habitat in the future.

FULL REVIEW

1. What is already happening?

Intertidal habitats exist at the margin of the terrestrial and marine realms, and species occupying these ecosystems are subject to environmental challenges posed by both regimes. Diurnal tidal cycles and seasonal fluctuations in sea and air temperature mean that intertidal organisms are subject to thermal extremes: fluctuations in environmental temperature in the order of 25 °C can be experienced over a single tidal cycle. Additional stressors such as desiccation, current and wave forces, rapid fluctuations in salinity, oxygen availability and nutrient levels mean that organisms are often living close to their physiological tolerance limits, and are thus sensitive to further changes in aerial and aquatic regimes driven by global warming.

Climate has a pervasive influence at all levels of organisation in biotic systems because of temperature-dependent processes from enzyme reactions through to ecosystems (Cain, 1944; Atkinson *et al*., 1987; Stenseth *et al.*, 2002). Marine ectotherms respond faster than terrestrial species to environmental change as the typically short lifespans and sessile or sedentary nature of the adult and juvenile stages prevent escape from changing environmental regimes (Carr *et al.*, 2003, Helmuth & Denny, 2003). Intertidal invertebrates and macroalgae occupy low trophic levels and are responding quicker to alterations in climate than species at higher trophic levels (e.g. Smith, 1985; Barlow *et al.*, 1998; Jenouvrier *et al.,* 2003). They often show the first response in a cascade of effects up the food chain and are therefore sentinels of climate change impacts. In addition to temperature, several other climate-related drivers are also impacting, or are highly likely to have adverse effects upon coastal habitats including sea-level rise, increases in relative wave height and storminess, and the associated secondary effects arising from adaptation and mitigation activities in coastal regions. The evidence base for these is far less than for temperature–related impacts to date, but increasing research and monitoring efforts are demonstrating that intertidal habitats are already being impacted by several aspects of global environmental change.

Saltmarshes

Coastal saltmarshes (also known as 'merse' in Scotland) are concentrated at the landward edge of intertidal regions within estuaries and sheltered water bodies around the UK. They account for approximately 24% of the English coastline, 11% of the Welsh coastline and 3% of the Scottish coastline (www.ukbap.org.uk/PriorityHabitats.aspx). They are important breeding and refuge habitats for waterbirds and fishes and are a winter food source for passerines. Brackishwater regions of saltmarsh often have high plant diversity, are important habitats for invertebrate function and fish nursery grounds. Several terrestrial BAP species are also resident within saltmarsh systems. Saltmarshes in the UK are therefore designated as a BAP Priority Habitat, feature in Annex I of the Habitats Directive and are proposed for the OSPAR Threatened and/or declining Species and Habitats List (European Commission, 1992; OSPAR, 2008a). Many are listed as SSSIs, ten areas have been proposed as Special Areas of Conservation (SACs) and 27 major saltmarsh sites have been included in Special Protection Areas (SPAs).

Saltmarsh habitats function as a natural climate change mitigation measure due to their capacity as carbon sinks. Vegetation sequesters carbon, which is subsequently buried as the sediment accretes, at rates ranging from 30-343g carbon per square metre per year (Chmura *et al*., 2003). These habitats act as natural buffers against wave energy and flooding of low-lying land by seawater incursions. Awareness of the important role they play in flood defence has led to recognition within UK flood risk management and saltmarsh redevelopment plans (DEFRA, 2007; Nottage & Robertson, 2005).

The most recent Biodiversity Action Plan review in 2008 documents the status and extent of saltmarsh habitats as still in decline (www.ukbap.org.uk/PriorityHabitats.aspx). Isostatic rebound, sea-level rise and increased storminess are all thought to be factors that have increased the rate of erosion during the 2000s. There is evidence to suggest that a single storm event can trigger saltmarsh erosion, which then continues through internal processes which operate independently of external drivers (Van der Koppell *et al.,* 2005). Many saltmarshes are subject to erosion at the seaward edges due to high wave energy environments and coastal squeeze due to fixed flood defences and other artificial structures on the landward side. Few other links have been made between saltmarsh erosion and climate change, but lack of sediment supply due to coastal defence construction may be a secondary impact of adaptational measures being taken to combat climate change. The BRANCH assessment of the vulnerability of saltmarsh and mudflats in NW Europe showed how population increases and coastal defences have led to significant increases in the areas of both habitats in the high and very high vulnerability classes (Zhang *et al.*, 2007). The most recent estimate of the rate of saltmarsh loss to erosion is 100 hectares per year in the UK (www.ukbap.org.uk/PriorityHabitats.aspx).

Regional differences

Net erosion has occurred in recent decades in the western Channel, Celtic Sea and South-West Approaches and the Eastern English Channel regional seas areas (Allen, 1990). In the Western Channel, some saltmarsh habitat has disappeared completely (www.ukbap.org.uk/PriorityHabitats.aspx). Multiple causes have been identified, including temperature increases, sea-level rise, coastal squeeze (most pronounced in the southern North Sea and eastern English Channel regional seas) and isostatic rebound (lowering of land levels in the eastern English Channel region and raising around the Scottish Continental Shelf regional sea). The Severn Estuary system reached its maximum carrying capacity for saltmarsh in the 1980s and has been retreating inland since then as part of the continuing rise in sea levels (Allen, 1990). An increase in storm surge and relative wave height, and associated increases in wave energy have already been demonstrated to be contributing to saltmarsh retreat in the Irish Sea (Wheeler *et al.*, 1999; Devoy, 2008). Not all saltmarshes are retreating. Many that are located in north-west England and the Dyfi estuary in west Wales are showing a recent trend of accretion over the past two decades with the prevalence of coarse sediment and isostatic uplift cited as factors counterbalancing sea-level rise (Shi, 1993; www.ukbap.org.uk/PriorityHabitats.aspx). Saltmarsh in the Bury Inlet is accreting in most places, but the north-eastern boundary is eroding downstream of the Loughor Bridge (CCW, unpublished data). Sediment supply has been demonstrated to maintain elevation of marsh in south-east Essex with sea-level rise rates of 3 m per year in the 1980s, indicating that saltmarsh systems may be able to withstand future sea-level rise as long as sedimentation rates are sufficiently high (Reed, 1988).

Intertidal mudflats

Intertidal mudflats are predominantly located in the middle reaches of estuaries but also occur in other sheltered coastal areas. They are closely interlinked with saltmarsh habitats, which often form the border at the upper limit, and dissipate wave energy, reducing erosion risk for these sensitive habitats. Intertidal mudflats are listed as an Annex 1 habitat in the Habitats Directive, are a UKBAP Priority Habitat and are nominated for the OSPAR List of Threatened and Endangered Species and Habitats (European Commission, 1992; OSPAR, 2008a; www.ukbap.org.uk/PriorityHabitats.aspx). They are created by deposition of fine particulate sediments such as silts and clays (less than 0.063mm diameter) up to sandy gravels in low energy environments.

Climatic drivers are changing the geomorphology of open coast sediment systems via increasing wave height and storm surge and increased water depth. The morphodynamic state of beaches is being altered by increases in storm frequency and greater wave energy, which cause steepening of the beach slope and a change in the sediment particle composition towards coarser particles. In Charting Progress 2, the Southern North Sea, Eastern Channel, Western Channel and Celtic Sea and Irish Sea are all currently assessed as having large areas of intertidal sediments negatively affected by anthropogenic impacts (Benjamins *et al*., 2010). Climate is a contributing factor, but existing data from monitoring surveys cannot accurately separate the various effects of climate, such as temperature and sea level rise with from other factors such as damage from scallop fishing gear and bait digging.

Seagrass Beds

Seagrass beds occur in soft sediments within sheltered intertidal and shallow subtidal areas where there is protection from wave action. Seagrass plants stabilise sediment and provide a three-dimensional habitat for epifauna and flora and a sheltered environment for juvenile fish and cephalopods. They provide a major food resource for waterbirds, and the dead plant matter is a large organic source for intertidal benthic systems (Barnes & Hughes, 1982). Breeding populations of spiny and short-nosed seahorses, two species of pipefish, *Entelurus aequoraeus* and *Sygnathus typhle* are almost totally restricted to seagrass beds (OSPAR, 2008b). The productivity rate of seagrass beds (i.e. growth of new plant material) can reach 2g of carbon per square metre per day during the temperate growing season (OSPAR, 2008b) with high biomass of up to 5kg per m^2 (Barnes & Hughes, 1982). *Zostera* beds are therefore important habitats that support a biodiverse community, including nationally rare species, and are an important source of carbon to intertidal and subtidal coastal systems.

Seagrass beds are under threat from multiple anthropogenic pressures. The species *Zostera marina, Zostera marina var. angustifolia* and *Zostera noltii* occur in UK regional seas. They are individually classified as scarce (http://www.ukbap.org.uk/PriorityHabitats.aspx), and the overall status of seagrass beds in the UK is considered to be degraded (Wilding *et al.*, 2009). Loss and fragmentation of seagrass beds via nutrient loading and turbidity are the main drivers of decline globally (Short & Neckles, 1999). Pressure from activities such as land reclaim, coastal development and localised activities including recreational boat mooring and bait digging are a major cause of physical disturbance and damage to seagrass beds in UK regional seas (Wilding *et al.,* 2009). Physical disturbance and removal of seagrass habitat decreases the diversity and biomass of associated epifauna (Reed & Hovel, 2006). *Zostera* beds of even small spatial extent support such a higher biodiversity than the surrounding sediment that they are still classified as being of conservation importance (Hirst & Atrill, 2008). There is no published evidence for climate-driven changes to UK seagrass habitats, however, it is not unreasonable to expect that seagrass distribution and condition will be influenced by climate related changes in sea temperature, increased storm activity, changes in prevailing wind conditions and also by ocean acidification, shown to be important in other countries (Björk *et al.,* 2008).

Regional differences

The current status of individual seagrass beds shows mixed trends around the UK. Within the Solent Estuary, seagrass extent does not seem to have greatly changed over the last two decades (Lefebvre *et al*., 2009). The number of *Zostera* beds has increased in Wales during the 2000s. Post 2007 estimates are of an increase of approximately 27.8 ha, roughly 4.9% (Boyes *et al.*, 2009; Howsen, in prep.; Mercer, 2009; CCW, unpublished data). The exact dynamics of change cannot be pinpointed as there are gaps of several years between surveys at some sites. *Zostera noltii* prefers a low salinity environment for germination and increased freshwater runoff in recent years may have been the driver for the increased success of this species in Welsh seagrass beds.

Estuarine rocky habitats

Rocky habitat is relatively uncommon in UK estuaries, being mainly found in the Celtic and Irish Seas and north-west Scotland. It is located in low wave energy environments with reduced salinity, increased turbidity and siltation compared to open coast reefs. The coldwater macroalga *Pelvetia canaliculata* is found on the highshore of both estuarine and coastal rocky habitats. Abundances of *P. canaliculata* have shown a slight decline in the past few years on shores close to the mouths of estuaries in the western English Channel (Mieszkowska, unpublished data).

Estuarine soft sediment habitats

Soft sediment intertidal habitats are dynamic in nature, and are structured by a combination of factors including wave action, local hydrodynamics, wind direction and sediment transportation. Within an estuary, soft sediment benthic intertidal systems can be very variable in species composition, biomass, density and productivity, which can be driven by salinity, tidal range and sediment type (e.g. McLusky, 1989; Ysebaert *et al.,* 2003). Regions of soft sediment habitat currently situated around the outer regions of estuaries such as the Humber, in the northern North Sea support high macrobenthic biomass (Fujii, 2007) but low diversity and few rare species. Few direct impacts of climate change have been observed for estuarine soft sediment benthic habitats within the UK, but 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. Insufficient data is available to provide a comprehensive review of regional differences in climate impacts at the present time.

Rocky Intertidal

The majority of recent climate-driven impacts on rocky habitats have been observed at the individual species level. Northern range limits of warm water species are still extending northwards. The topshell *Osilinus lineatus* continues to colonise shores along the rocky coastline of north Wales beyond recent range limits where it has not previously been recorded. Numbers of individuals have risen significantly within populations around the Welsh and English coastlines during the 2000s, and the rate of increase in abundance is accelerating. The trend of decreases in the relative abundances of cold, and increases in warm water species of barnacles and limpets has continued during 2008 and 2009 (Mieszkowska, 2009b, 2010; Mieszkowska & Hawkins, unpublished data*). Populations of the cold water sugar kelp, *Saccharina latissima* (*Laminaria saccharina*) have remained stable around the UK coastline (Mieszkowska unpublished data) in contrast to many regions of Europe (Christie, pers. comm).

Phenological shifts are beginning to be detected in warm water gastropods in populations close to northern distributional limits in the UK. Reproductive cycles of the topshells *Osilinus lineatus* and *Gibbula umbilicalis* have shifted earlier in the year by 3 months during the last 2 decades (Mieszkowska *et al*., 2006). Gonad development in the southern limpet *Patella depressa* now commences 19 days earlier in the 2000s than during the 1940s (Moore & Hawkins, in review). Annual studies throughout the 2000s demonstrated that *G. umbilicalis* also switched reproductive strategies from a single to a double (bivoltine) gonad maturation per year in northern populations for the first time in 2008 (de Francisco Mora *et al.,* in review). This is characteristic of southern populations inhabiting warm waters at lower latitudes and may be indicative of the plastic nature of reproductive processes in response to changes in temperature. In contrast, the limpet *Patella vulgata*, which has a biogeographic range centred in cooler, more northerly latitudes has shown an increase in annual frequency of reproductive failure and a delay in onset of gonad production between the 1940s and 2000s (Moore & Hawkins, in review). Recruitment of rocky intertidal species has also been linked to the North Atlantic Oscillation. Larval dispersal corresponds to the oceanic component of the signal, whereas post-settlement success is related to the atmospheric commonent (Broitman *et al*., 2008).

Non-native species are dealt with in detail in *MCCIP ARC Science Review 2010-11 Non-natives* (Maggs *et al.*, 2010), but an important recent change is the increased colonisation of soft and hard intertidal habitats by the non-native Pacific oyster, *Crassostrea gigas*. This species was permitted to be farmed in estuaries around the UK as waters were considered to be too cold for it to reproduce, however, in recent years *C. gigas* has settled on natural habitat outside the farms. The worst affected area is in the eastern English Channel, where dense beds of *C. gigas* now occur. Natural estuarine and open coast colonisation is has increased over the last 2 years in the vicinity of oyster farms in the western English Channel and Scotland.

Regional differences

Most of the recent changes have been recorded in the Western and Eastern Channel and Celtic Sea, and Minches and western Scotland regional seas. During the last 2-3 years the southern red turf alga *Chondrocanthus acicularis* has noticeably increased its coverage in the mid and lowshore regions of shores where it has previously been recorded, and has colonised shores where it has not been previously seen in the Western and Eastern Channel regions (Herbert, Hawkins & Mieszkowska, unpublished data). The warm water kelp *Sacchoriza polyschides* has massively increased in abundance in kelp beds around the south-west of the UK within the last few years, whereas the cold water brown macroalga *Alaria esculenta* continues to decline in abundance. The native cold-water breadcrumb sponge, *Halichondria panacea* showed a sudden increase in abundance on rocky shores in the western English Channel during the spring of 2009 (Sugden, Mieszkowska & Hawkins, unpublished data). This species has not been seen in such high densities during the 2000s and is thought to have responded favourably to low temperatures during the winter of 2008/09, which was the coldest recorded during the 2000s (http://hadobs.metoffice.com/hadsst2/). Interestingly, this cold winter did not appear to exert any lethal impacts on warm water species.

The recent construction of coastal defences in the Eastern Channel has provided hard substratum in regions of soft sediment. Some rocky intertidal species have been using these structures as artificial habitat and have been able to successively colonise defences which are located within the dispersal distance of larvae along the coastline. Populations of the topshell *Gibbula umbilicalis* have now become established on natural rocky shore beyond these structures in the Eastern Channel and the southern North Sea (Mieszkowska and Hawkins, unpublished).

Intertidal chalk habitats

Intertidal chalk is classified as a UKBAP Priority Habitat, and is listed in Annex I of the EC Habitats Directive and the OSPAR list of Threatened and/or Declining Species and Habitats (OSPAR, 2008a). Intertidal chalk systems include sea caves and littoral fringe cliffs and platforms, which host micro-habitats of biological importance. Such exposures are rare in Europe, with those occurring in the southern North Sea and eastern English Channel regional seas accounting for 57% of all chalk habitats in Europe (ICES, 2003).

On the south coast of the Isle of Wight, severe storm events have caused infilling of rock pools and abrasion of intertidal chalk habitat by large amounts of flint that has been cast ashore from offshore shingle banks. This is particularly noticeable at Freshwater Bay on the south coast of the island, where positive impacts include promotion of recruitment of *G. umbilicalis* by increasing suitable microhabitat, but negative effects include disturbance of algae, limpets and the common anemone, *Actinia equina* which also colonise these rockpools (Herbert, unpublished data). Climate change also exerts secondary impacts on chalk habitats. Coastal defence work during the 1990s resulted in the modification of 74% of chalk habitat on the Isle of Thanet, 56% in the wider region of Kent and 33% in Sussex (Fowler & Titley, 1993). No update is available for the 2000s. Climate change is likely to be enhancing colonisation of intertidal chalk by non-native macroalgal species such as *Sargassum muticum* and *Undaria pinnatifida* in the eastern English Channel, which are a threat to native species restricted to this limited habitat. The non-native Pacific oyster *Crassostrea 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 few years (Herbert, unpublished data).

Biogenic reefs

The honeycomb worm *Sabellaria alveolata* is listed a UKBAP Priority Habitat, and reefs are only found on shores with moderate or strong wave action in south and west England, Wales, Scotland and Northern Ireland. Established reefs stabilise sediment and increase biodiversity within intertidal systems. The northern range limit of *S. alveolata* has been increasing in Scotland during the 2000s, and abundances have increased or remained stable at established sites in the western English Channel and Irish Sea regional seas (Frost *et al.,* 2004; Burrows *et al.,* unpublished data). These changes are thought to be a direct response to less severe winter temperatures in recent years. The *S.alveolata* reef in Pen Llyn a'r Sarnau SAC in north Wales has shown deterioration in the last few years, reversing the trend seen in the early 2000s. Long- and onshore sediment transport related to storminess is likely to be affecting the status of the reef, but is highly variable both within and between locations (Boyes & Allen, 2008; Brazier, unpublished data).

The blue mussel, *Mytilus edulis* occurs on many sediment types from muds and sands to rock platforms in sheltered to exposed locations in the inter- and subtidal, and its beds are another UKBAP Priority Habitat. *M. edulis* beds affect the transfer of organic matter between the pelagic and benthic zones and significantly contribute to carbon production within intertidal systems. They can form high density beds which function as a biogenic habitat for a wide range of epifaunal and infaunal species, and are an important food source for seabirds. The OSPAR Assessment process has identified beds in the western and eastern English Channel areas as 'under threat' from eutrophication and associated phytoplankton blooms, which are changing in frequency and intensity due to climate warming (OSPAR, 2008b).

Populations of the native oyster *Ostrea edulis* have declined throughout Britain over the last few decades due to a range of factors including overfishing, substratum loss, smothering, introduction of pathogens and synthetic compound contamination (Jackson *et al.*, 2008). *O. edulis* is listed in the OSPAR List of Threatened and/or endangered species and habitats (OSPAR, 2008a) No direct impacts of climate change have been attributed to the decline of *O. edulis* in UK waters. The introduction of the warm water non-native species of oyster, *Crassostrea gigas,* the slipper limpet *Crepidula fornicata* and the American oyster drill *Urosalpinx cinerea,* are thought to exert negative impacts on natural *O. edulis* beds via competition, pathogenic transfer, reduction in sediment bed quality and predation (Hancock, 1954; Minchin *et al.*, 1993; Miossec *et al.,* 2009). Their future impacts may increase, aided by a warming climate more suitable to their physiological niches. Limited research has been carried out to date to test these theories, but within estuaries in the Western Channel area, *C. gigas* appears to be breeding and settling on natural rock, outcompeting *O. edulis* within the intertidal zone (Mieszkowska, 2009a). Extensive reefs of Pacific oysters could also facilitate aquatic macrophytes (Reise & van Beusekom, 2008).

Note: References to unpublished data represent new data collected during 2007-2009. These data have been QA'd and analysed but are not yet published in peer-review manuscripts or grey literature reports. All such data referenced here has been collected by the author or collaborating researchers and we are confident that they accurately reflect current and ongoing changes within coastal habitats.

2. What could happen in the future?

Continuing climate change is likely to cause nutrient supply from land to decline, freshwater runoff from precipitation to increase, species introductions, temperature and sea-level to rise and pH to decrease in the UK. The effects are interrelated and will have an increasing likelihood of driving abrupt changes within intertidal ecosystems. Some systems may experience regime shifts, whereby the ecosystem rapidly changes from one stable state to another in which the ecosystem still functions, albeit differently to before, with different dominant species. Other switches may cause irreversible damage and alter or disrupt ecosystem structure and functioning. Increasing air and sea temperatures will continue to alter intertidal communities in UK regional coastal seas. The first and fastest changes are likely to continue to occur in the region of the biogeographic breakpoint that straddles the Celtic Sea and Western Channel, although climate impacts will occur throughout UK intertidal habitats. Warm water species will continue to increase in abundance within established populations and extend their northern distributional limits. Populations of cold water species will continue to decline in abundance, and further contractions of ranges are expected for those which reach their southern limits in the UK (*sensu* Mieszkowska *et al*., 2006, 2007). This is likely to be driven by warmer winter and spring temperatures acting to: 1. prompt earlier and potentially more frequent reproductive activity in mature adults, and 2. increased survival of new recruits due to greater temporal periods of exposure to food resources in spring and summer, coupled with exposure to less severe winter conditions. In contrast, low temperature thresholds necessary to trigger gametogenesis in cold water species may not be reached every year if mean winter coastal sea temperatures warm in the region of 2-

3°C predicted for the period 2070-2098 (UKCIP, 2009). Increased precipitation will result in increased freshwater runoff during winter, and greater upstream penetration of seawater during dry summers. Both processes are likely to affect intertidal species of marine origin, especially on the western coasts of the Celtic and Irish Seas and north-western Scotland (Benjamins *et al*., 2010), although there are no field data that demonstrate such impacts to date.

Saltmarsh extent may decrease further with predicted increases in sea-level rise, storm surge and relative wave height. Increased wave energy generated by rising wind strengths is predicted to contribute to increased erosion of the seaward edges of saltmarshes. UKCIP predictions for 2090-2099 of 29.8-45.6 cm sea-level rise (Medium emissions scenario), and over 1m exceedences of present-day high tides further support the predicted increase of erosion rates for UK saltmarshes. Model predictions for UK saltmarshes at risk forecast an increase from 30% currently vulnerable to 43% under the 2080s high sea-level rise scenario because of the lack of opportunity for autonomous adaptation through inland migration (Zhang *et al*., 2007). Greatest impacts are likely to occur in the Eastern Channel regional sea where changes in sea-level rise and storm surge are forecast to be most severe (UKCIP, 2009). The predicted increase in precipitation (50% central estimate prediction) in west Scotland and the Minches, Scottish Continental Shelf, northern and southern North Sea may enhance sediment supply to saltmarshes via freshwater runoff, but increased water flow could increase erosion rates relative to sedimentation processes. In contrast, a reduction in precipitation is forecast for the East and West Channel, Celtic Sea and Irish Sea areas which may result in a reduced supply of sediment to saltmarsh systems (http://ukclimateprojections.defra.gov.uk/content/view/1779/544/). Linkage of regional climate scenarios with local geomorphological and hydrological data is therefore imperative to allow accurate predictions at the regional seas scale.

Secondary impacts of climate change via coastal squeeze will continue due to the requirement for increased coastal defence construction and subsequent alterations in sediment supply. Managed realignment of agricultural land to assist the retreat of saltmarsh systems is already underway in the Wash, Chichester Harbour, Tees Estuary, North Norfolk Coast, Suffolk, Essex and Cromarty Firth. Restoration and creation of saltmarsh habitat is also under consideration which may slow the rate of loss, but these techniques do not yet have a sufficient spatio-temporal extent to demonstrate their effectiveness as a mitigation tool (http://www.uea.ac.uk/~e130/Saltmarsh.htm). Increased atmospheric CO_2 concentrations have been suggested as a potential offsetting mechanism via enhanced photosynthetic productivity, and subsequent peat production within the marshes.

Extensive areas of estuarine soft sediment intertidal habitats could be lost due to coastal squeeze as sea-level rise and constructed sea defences act in tandem to constrain the intertidal area available for colonisation, and subsequently drive impoverishment of benthic fauna. Sea-level rise may additionally cause a shift in both geographical location and vertical extent of intertidal zones, which would affect the total biomass and productivity of the benthos. The Humber estuary has experienced an average rise in sea level of 2-2.5mm per year over the last century (Winn *et al*., 2003) but recommended values for planning have been set at an average of 6mm per year, indicating an expected acceleration in the rate of habitat loss with continuing climate change. The exact nature of such changes are not known but model predictions indicate that soft sediment benthic habitats situated around the lower reaches of estuaries will be most at risk from future sea-level rise (Fujii, 2007).

Increases in sea level of 0.3m are predicted to result in a loss of spatial extent of the intertidal in the order of 6.7%, with a resultant loss of 6.9% of the total macrobenthic biomass (Fujii & Raffaelli, 2008). Studies conducted in artificially raised thermal environments close to powerstation outfalls raise caution as to the predictability of changes to benthic species (Schiel *et. al.,* 2004) and there are still large uncertainties in the projected impacts within soft sediment systems.

Increasing sea levels are likely to cause deepening and widening of estuaries, resulting in an increase in the tidal prism and tidal range (Kennish, 2002). Resultant increases in saline intrusion further inland (Scavia *et al*., 2002) may partially compensate for the decline in benthic biomass (Fujii & Raffaelli, 2008) but additional changes may exacerbate the decline in spatial coverage and species composition. Factors include the continuing trend of steepening coastal profiles due to foreshore erosion and the construction of sea walls (Hulme *et al*., 2002) and alteration of sediment regime by enhanced wave and tidal action. Such physical changes have potentially severe implications for intertidal benthic communities, with model forecasts of up to 22.8% declines in benthic biomass under the most severe climate scenarios (Fujii & Raffaelli, 2008). The decrease in intertidal mudflat area, combined with a reduction in biological integrity is also predicted to decrease the abundances of predatory bird and animals. (OSPAR, 2008b). Habitat creation and managed realignment are the current favoured methods of partially compensating for loss of intertidal mudflats caused by sea-level rise in the UK (Elliot *et al*., 2007).

Climate effects recently documented in European seagrass beds could provide an insight into future impacts that may occur in UK habitats. Reductions in biomass have occured in response to acute reductions in salinity during drought events in Portugal (Cardoso *et al*., 2008). Wasting disease, resulting from infestations of *Labyrinthula zosterae* is more prevalent in warmer environments and high turbidity water bodies. This species decimated UK populations of *Zostera* spp. during the 1930s (Rasmussen, 1977) and outbreaks are predicted to become more frequent in warmer intertidal systems (Wilding *et al.*, 2009). Future climate change may have both positive and negative impacts on intertidal seagrass beds in the UK. Geographical distribution will shift in response to increased temperature stress and changes in patterns of sexual reproduction, effects of salinity change on seed germination, propagule formation, photosynthesis, growth and biomass (Short & Neckles, 1999), changes in tidal range and flow and increased seawater intrusion into estuaries. Vertical alterations of seagrass beds will be driven primarily by sea-level rise due to increasing water depths, (Collins pers. comm.) but natural raising of the upper limit may be constrained in areas of coastal development and coastal defences. Indirect temperature affects may include plant community changes as a result of increased eutrohpication and changes in the frequency and intensity of extreme weather events. There is evidence of thermal plasticity in the closely related species *Zostera japonica*, with southern populations being adapted to warmer environments (Shafer et al., 2008). The species of *Zostera* found in the UK may also show adaptation to local thermal environments, and therefore some capacity to tolerate changes in thermal climate, but there is no direct evidence for this. Further increases in UV radiation are speculated to be detrimental to seagrasses, although preliminary studies suggest that the response is species-specific, with some species being able to produce more UV-B blocking pigments (Dawson & Dennison, 1996). Increased CO_2 concentrations may promote seagrass survival by increasing available carbon and reducing calcareous epiphytes (Short & Neckles, 1999; Hall-Spencer *et al*., 2008) but increase non-calcarious epiphytic algae which may negatively impact *Zostera* beds by increasing shading (Beer & Koch, 1996). Increased storminess, and associated increase in wave action extending into more sheltered areas may destabilise muddy sediments that *Zostera* spp. colonise. Loss of seagrass plants would destabilise the underlying substratum and reduce habitat availability for many associated species of epifauna, infauna and juvenile fish.

The species-specific nature and rate of recent and current changes suggest that community compositions will alter, with subsequent impacts on the structure and functioning of rocky intertidal ecosystems. If the warm water topshells *O. lineatus* and *G. umbilicalis* continue to accelerate their rate of population increase, they may outcompete other grazing species for available resources. Cold water species such as the barnacle *Semibalanus balanoides* may begin to suffer reproductive failure as autumn temperatures do not drop low enough to trigger gonad development, with predicted extinction in UK waters under future climate scenarios (Poloczanska *et al*., 2008). Laboratory experiments suggest a value of 10°C as the upper seawater temperature threshold for gametogenesis (Southward, 1958). Interestingly, the mean monthly winter sea surface temperature during 2006/07 and 2007/8 did not drop as low as 10°C, but cyprids and spat were recorded on shores in the western Channel during the following springs at densities typical of the last two decades (Hawkins & Mieszkowska, pers. obs.). Therefore this value may be a useful ballpark figure, but more research is required to fully understand thermal drivers and their impacts on key lifestages. Far less is known about thermal tolerances and impacts of warming sea waters on infaunal species in soft sediment habitats. Their ability to burrow may help them to avoid warming bottom waters to some extent, but there is insufficient field and laboratory data on the responses of epifauna and infauna to climate-driven changes in bottom temperature to make any firm predictions.

Sea-level rise will in time result in complete submergence of the lowshore sections of the intertidal zone, thereby reducing the available habitat for colonisation. This may be detrimental to soft sediment habitats, especially those with a shallow gradient. It is unlikely to have a significant impact on rocky shore habitats, however, as most rocky shores have sufficient vertical extent of rock above the current highwater mark to compensate for small increases in sea-level rise, and the entire rocky intertidal assemblage will probably just shift upwards to compensate. Sea-level rise and increased storminess are predicted to increase the rate of infilling of south and west facing estuaries in Wales, leading to increased sediment deposition and subsequent smothering of rocky outcrops (http://www.ukbap.org.uk/Default.aspx).

Increased storm surge events will alter species composition and dominance on rocky shores, with exposure-tolerant species of space occupiers including barnacles and grazers such as the limpets and kelps being favoured over more sheltered species. The midshore region is likely to see a gradual shift from algal dominance by fucoid species to grazing and space-occupying invertebrates (Hawkins *et al*., 2008; Burrows *et al*., 2009). Increased wave height will increase the vertical height available for colonisation due to raising of the splash zone. Conversely, soft sediment habitats will be negatively affected by increased storm surge and wave height, suffering from increased erosion, loss of intertidal spatial availability and alteration in sediment type. Coastal squeeze is likely to increase in spatial extent as accretion of the lower margins of soft sediment habitats acts synergistically with coastal construction preventing landward retreat, resulting in a constant thinning zone of intertidal habitat (Doody, 2001).

Significant mortality of *Mytilus edulis* has been demonstrated in response to increased air and water temperatures associated with heatwave events in field and laboratory experiments (Jones *et al*., 2009). The frequency and severity of such weather patterns is predicted to increase in the coming decades (IPCC, 2007), which is likely to result in more frequent losses of significant individuals within *M. edulis* populations in south-west England. Mussel beds trap sediment and can therefore keep up with sea-level rise to some extent (OSPAR, 2008b). They are also able to survive in an entirely subitidal environment so the geographical location of *Mytilus* beds is unlikely to alter significantly due to climate change in the next 10 -40 years.

Ocean Acidification

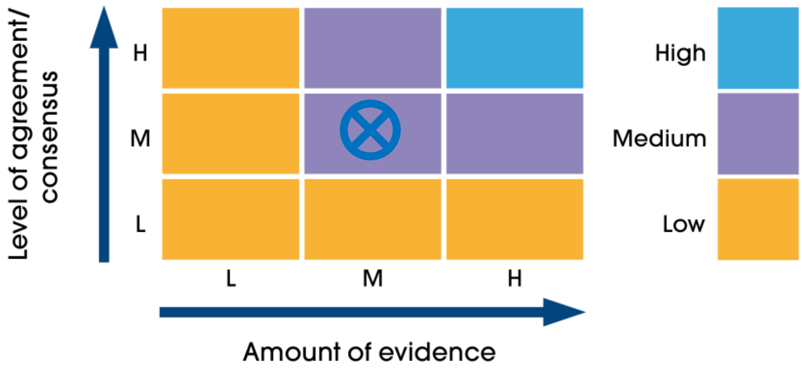
Research into the impacts of increased carbon dioxide concentrations in seawater on intertidal species and habitats is an emergent field, with little field evidence and varying results from laboratory studies conducted to date depending on the study species. The few studies at relevant pCO_2 levels impede our ability to predict future impacts on foodweb dynamics and the structure and functioning of intertidal ecosystems. It is, however, widely acknowledged that as the oceans continue to absorb CO_2 from the atmosphere, pH will decrease (i.e. water will become more acidic) and the carbonate chemistry will alter, causing a range of impacts on benthos. All relevant studies carried out on intertidal species to date, regardless of the habitat in which they occur, are dealt with together for the purposes of this report.

Few experimental data exist for intertidal soft sediment benthos, with the majority of laboratory studies focusing on subtidal species. A reduction in the ability to form calcarious shells is one of the primary predicted impacts of ocean acidification, as illustrated by research on the blue mussel, *Mytilus edulis*. Laboratory exposures to low pH levels of 6.5 caused the shells of *M. edulis* to begin to dissolve (Beesley *et al*. 2008). General health of exposed mussels was also impacted under less acidified conditions of pH 7.6 suggesting that a drop in the pH level of coastal waters of between 0.5 pH units is similar to that predicted over the 21st century (IPCC, 2007) is likely to reduce performance and survival of this species. Laboratory experiments have also shown that the reproductive ability and post-larval development of the rocky intertidal gastropod *Littorina obtusata* is impaired when exposed to seawater pH of 7.6 (Ellis *et al*., 2009), indicating that reproduction and recruitment may be negatively affected when environmental seawater pH decreases to this level. Chronic reduction of surface water pH to below 7.5 may also be severely detrimental to the acid–base balance of the purple sea urchin *Psammechinus miliaris* (Miles *et al*., 2006). These species do not occupy key functional or structural roles in rocky intertidal communities, but they do illustrate the breadth of taxa which are sensitive to decreases in oceanic pH. Acute exposure studies on the predatory crab *Necora puber* suggest that this species is more tolerant than other intertidal species studied to date, due to internal compensation by increasing concentrations of bicarbonate (Spicer *et al*., 2006). Such inter-specific variation may be manifested at the community level as interspecific interactions are altered by species-specific responses, similar to the responses to temperature increases outlined above.

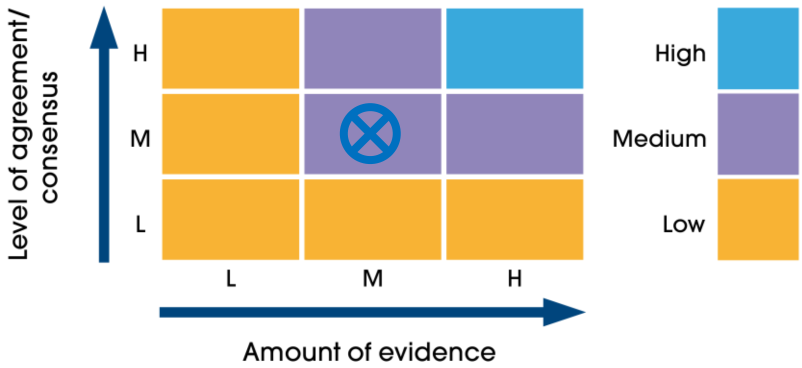
Importantly, acidification of the oceans will be superimposed on continually rising sea temperatures, and therefore we need to understand the potential synergistic impacts of these two climatic parameters. One example is the cold water barnacle, *Semibalanus balanoides*. Adult survival decreases by around 22% when exposed to pH levels of 7.7 (Findlay *et al*., 2009). This species is already declining in abundance at sites in the western English Channel region and Celtic Sea and is likely to be severely impacted when both temperature and pH of seawater are at stressful levels. There is the potential for widespread changes to intertidal systems via restructuring of communities due to variations in inter-species sensitivities, as indicated by new research on the impacts of ocean acidification on benthic assemblages (Widdicombe & Spicer, 2008).

3. Confidence in the science

What is already happening: **Medium**



What could happen: **Medium**



Overall

For this topic as a whole, we have a medium level of confidence both for ‘what is happening now’ and ‘what could happen in the future’. Whilst we are collating more information for all of the habitats, we don’t have sufficient biological (physiological / genetic) information yet to understand cause-and-effect, or the biological mechanisms by which most species are responding. There is a high level of confidence in conceptual and qualitative assessments, but quantitative information is still lacking for many intertidal habitats and this limits the predictive capacity for these systems.

Individual habitats

For all of the habitats discussed, confidence assessments are provided both for what is happening now and what could happen in the future.

In each case, the first score refers to the level of agreement in each case, the second to the amount of evidence. (e.g. for Intertidal mudflats below, medium refers to the vertical axis in the matrix (agreement) and low refers to the horizontal axis (evidence)).

Saltmarshes:

What is happening: Physical Impacts: high x high; biological impacts med x low

What could happen: Physical Impacts: med x med; biological impacts med x low

Intertidal mudflats:

What is happening: med x low, *What could happen*: med x low

Seagrass beds:

What is happening: med x med, *What could happen*: med x low

Rocky intertidal:

What is happening: high x high, *What could happen*: high x med

Estuaries:

What is happening: med x low, *What could happen*: med x low

Biogenic reefs:

What is happening: high x med, *What could happen*: med x low

Alien species:

What is happening: med x med, *What could happen*: med x low

4. Knowledge Gaps

The top priority knowledge gaps that need to be addressed in the short term to provide better advice to be given to policy makers are:

1. Lack of broad-scale, integrated monitoring designed to detect the impacts of climate change- temperature, storminess, sea-level rise on soft and rocky systems.
2. Information on the impacts of drivers, especially temperature, sea-level rise, storminess and precipitation, on the productivity and carrying capacity of systems.
3. Insufficient knowledge of biological mechanisms by which species are responding to climate change from the cellular to ecosystem levels.

5. Socio-economic impacts

Costing the economic and social value of goods and services provided by intertidal coastal habitats and the communities they support is still in its infancy for the UK and wider European Union. The ecosystem goods and services provided by intertidal and other coastal habitats are currently being evaluated as part of work being undertaken for the upcoming National Ecosystem Assessment. An initial draft of this report will be produced early in 2010 and will allow us to identify more precisely the cultural, economic and other types of benefit provided by these ecosystems. Seagrass beds and macroalgae were given an estimated global economic value of $US 19,000 per hectare per year a decade ago (Costanza *et al*., 1997) making them some of the most valuable plant resources on the planet. This estimate is likely to be revised in the current assessment being undertaken by the Deutsches Bank.

European directives such as the EC Habitats Directive and Water Framework Directive require habitats and areas to be monitored against benchmark reference conditions. These policy drivers do not take into account the fact that climate change can alter ecosystems irrespective of other anthropogenic impacts, and a site may not reach target conditions even if the state of the site or habitat has remained the same or improved with respect to direct human pressures. For example, sea-level rise will negatively impact upon intertidal habitats with a shallow vertical gradient and therefore present serious problems in meeting the requirements of the Habitats Directive for specific sites. This is something that must be borne in mind when setting the criteria for Good Environmental Status for the Marine Strategy Framework Directive, which is currently in the planning stages. In addition, the loss of saltmarsh habitat creates an obligation under the EU Habitats Directive to create a similar area of replacement saltmarsh. This has economic implications for member states in addition to statutory obligations arising from these directives. An overarching requirement is harmonisation of the Water Framework Directive, Marine Strategy Framework Directive and the Habitats Directive and the understanding that climate change will necessitate the development of moving baselines in order to achieve Good Environmental Status against the pervasive background of climate change.

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