Chapter 3: Natural environment and natural assets

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Lead contributor: Iain Brown (University of York/Stockholm Environment Institute)

Contributing authors: Richard Bardgett (University of Manchester), Pam Berry (University of Oxford), Ian Crute (Agriculture and Horticulture Development Board), James Morison (Forest Research), Mike Morecroft (Natural England), John Pinnegar (Centre for Environment, Fisheries and Aquaculture Science), Tim Reeder (Environment Agency), Kairsty Topp (Scotland's Rural College)

ASC contributor: David Thompson

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Implications of the vote to leave the European Union

This chapter was written before the results of the EU Referendum were known. Leaving the European Union is unlikely to change the overall scale of current and future risks from climate change, but in some areas it may affect policies and programmes important to address climate-related vulnerabilities.

If such policies and programmes are changed, it will be necessary for UK measures to achieve the same or improved outcomes to avoid an increase in risk. The Adaptation Sub-Committee will consider the impact of the EU Referendum and the Government's response in its next statutory progress report on the UK National Adaptation Programme, to be published in June 2017.

Contents

Key messages

Climate change presents a substantial risk to the UK's native wildlife and also to the vital goods and services provided by the natural environment to people. These goods and services include clean water, food and fibre, pollination, carbon storage, climate regulation, and natural protection from hazards such as flooding and erosion. Risks are typically exacerbated because the natural environment is already stressed by other non-climate pressures including pollution, habitat loss and fragmentation, and unsustainable use of soil, water and marine resources. These other pressures act to constrain the natural resilience of species and ecosystems to adjust and adapt to change, meaning that future climate change is likely to accelerate current rates of decline and loss of ecosystem function. Such combined stresses are particularly evident during extreme events when climate-related impacts are most apparent. Furthermore, evidence of longer-term shifts in species and ecosystems due to higher temperatures and sea-level rise is now discernible despite complex interactions. Shifts due to changes in rainfall patterns are currently more difficult to attribute with high confidence because of the large variability that occurs in the UK. The relative influence of climate change against other factors requires ongoing research and monitoring.

Climate change also provides potential opportunities for some species to expand their range (if habitat is available), extended growing seasons for crops and improved productivity in agriculture and forestry. Some evidence already exists for such positive outcomes but they are usually countered by other factors, both climatic and socio-economic. As also identified by other reports (for examplethe UK National Ecosystem Assessment and Foresight studies), realising opportunities and managing risks both require measures to reduce sectoral trade-offs over land, water and marine resources, while allowing space for adaptation by planning responses at landscape, catchment or ecosystem scale. On land, good management practice will be required to address an increased risk from soil erosion and the loss of organic material that releases carbon dioxide to the atmosphere. Changes in climatic suitability for species and habitats can be alleviated to an extent by further implementation of measures that maximise ecological resilience through the the quality, connectivity and diversity of habitat. Flooding of farmland is likely to become more frequent in some areas and new or more virulent pests, diseases and invasive species could pose additional risks to trees, crops, livestock and native wildlife. Increased water demand, including for agriculture, in combination with projections of reduced water availability in summer, is likely to challenge societal goals for healthy rivers, lakes and wetlands. The coast of the UK and its current distribution of species, habitats and land uses is particularly vulnerable to change because of existing pressures in combination with continued sea-level rise. In particular, sea level rise increases the risks associated with extreme water levels during storm events. Similarly, some change in the freshwater and marine environment is also inevitable and may potentially have major implications for fisheries and biodiversity. To sustain their multiple benefits, use of land and sea therefore require more flexible and coherent approaches than have occurred in the past. This will necessarily include recognition that maintaining all natural assets and services in their current geographical distributions will be increasingly unviable. The natural environment of the UK is also the result of human activity over millennia and it will be important that both its natural and cultural heritage are protected together.

The key risks and opportunities identified for the natural environment are summarised in Table 3.1. The assessment of the urgency is based on the expert judgement of the ASC, in consultation with the report authors and peer reviewers. See Chapter 2 for more detail on the method taken to assess urgency.

Table 3.1. Urgency scores for natural environment and natural assets

Risk/opportunity (relevant section(s) of chapter)	**More action needed**	**Research priority**	**Sustain current action**	**Watching brief**	**Rationale for scoring**
Ne1: Risks to species and habitats due to inability to respond to changing climatic conditions (3.2)	UK				More action needed to reduce existing pressures, improve size and condition of habitats, restore degraded ecosystems, and deliver coherent ecological networks. More action needed to factor climate change into conservation planning and site management
Ne2: Opportunities from new species colonisations (3.2)	UK				More action needed to deliver coherent ecological networks More action needed to factor climate change into conservation planning and site management.
Ne3: Risks and opportunities from changes in agricultural and forestry productivity and land suitability (3.3)		UK			More research needed into developing integrated land use planning based upon changing suitability More research needed on the nature and scale of changing land suitability and its impacts. More research needed into crop varieties, tree species and agricultural systems that are resilient to future climate change.
Ne4: Risks to soils from increased seasonal aridity and wetness (3.3)	UK				More action needec to reduce existing pressures on soils, increase uptake of soil conservation measures and restore degraded soils.

Ne5: Risks to natural carbon stores and carbon sequestration (3.3, 3.7)	UK				More action needed to restore degraded carbon stores, particularly peatlands. More research needed to account for climate change impacts on carbon stores in the UK GHG projections.
Ne6: Risks to agriculture and wildlife from water scarcity and flooding(3.4)	UK				More action needed to reduce pollution and over-abstraction and improve the ecological condition of water bodies Ensure decisions on use of water allow for necessary environmental flows and take account of climate change.
Ne7: Risks to freshwater species from higher water temperatures (3.4)		UK			More research needed on scale of risk and effectiveness of adaptation measures.
Ne8: Risks of land management practices exacerbating flood risk (3.3, 3.4)	UK				Deliver wider uptake of natural flood management in high-risk catchments especially where there are likely to be carbon storage, water quality and biodiversity benefits. Implement catchment-scale planning for flood risk management. Review potential for adverse flood risk outcomes from land management subsidies.
Ne9: Risks to agriculture, forestry, landscapes and wildlife from pests, pathogens and invasive species (3.7)			UK		Continue to implement surveillance and bio-security measures. Continue current research efforts into the impact of climate change on emerging and long-term risks. Develop cross-sectoral

					initiatives for risk assessment and contingency planning
Ne10: Risks to agriculture, forestry, wildlife and heritage from change in frequency and/or magnitude of extreme weather and wildfire events (3.3)			UK		Continue to build resilience of ecosystems to drought, flood and fire Continue current efforts to manage and respond to wildfires. Monitor heat stress impacts on livestock. Continue current efforts to manage impacts of high winds on forestry.
Ne11: Risks to aquifers, agricultural land and habitats from salt water intrusion (3.5)			England, Wales	Northern Ireland, Scotland	Continue actions to manage salinity risks to freshwater habitats. Monitor impacts on aquifers to assess whether risks are increasing.
Ne12: Risks to habitats and heritage in the coastal zone from sea-level rise; and loss of natural flood protection (3.5)	UK				More action needed to deliver managed realignment of coastlines and create compensatory habitat.
Ne13: Risks to, and opportunities for, marine species, fisheries and marine heritage from ocean acidification and higher water temperatures (3.6)		UK			More research needed to better understand magnitude of risk to marine ecosystems and heritage.
Ne14: Risks and opportunities from changes in landscape character (3.7)				UK	Monitor impacts and ensure climate change is accounted for in future landscape character assessments.

Box 3.1. Comparison with approach taken to the first UK Climate Change Risk Assessment

This assessment has taken a more holistic and cross-sectoral approach than the first CCRA in 2012. Although similar sectoral priorities have been identified, a greater emphasis has been placed on identifying the relationships and dependencies between risks. In particular, cross-cutting risks linked to the availability of land, water and marine resources have been highlighted using an ecosystem services framework. This allows an agenda for action to be developed consistent with the UK National Ecosystem Assessment and the work of the Natural Capital Committee.

The assessment of risks and opportunities facing terrestrial species, habitats and soils has benefited from a large expert-based analysis of climate impacts through the LWEC Report Card process. Specific emphasis has also been placed on assessing risks to pollinators. because of the additional societal benefits they provide through pollination . Assessment of crop production has been refined to provide a broader interpretation of the effects of climate change on crop yield changes. In addition, analysis of risks relating to grassland and livestock productivity, which are important for large areas of the UK, receive more comprehensive coverage. Freshwater ecosystems and water-related services are evaluated together to highlight synergies and the role of ecosystem-based adaptation. Similarly, risks to the key role played by coastal ecosystems in buffering flooding and erosion hazards is evaluated. A cross-cutting approach to assessing risks to and opportunities for the marine environment has been taken. . New ensemble model outputs of climate change in the marine environment have been used, including an assessment of uncertainty where this was not previously available. Finally, a cross-sectoral approach has been taken to evaluate risks associated with pests and diseases, wildfire and natural carbon stores in order to bring together existing evidence.

3.1 Context

This chapter considers the risks and opportunities from climate change that affect the natural environment. The implications of climate change for the multiple societal benefits provided by natural assets (or 'natural capital') are assessed, while also recognising the intrinsic value of the natural environment in its own right[1]. The chapter aims to highlight important adaptation-related links between biodiversity and ecosystem functioning with the many goods and services which people derive from the natural environment. This includes risks and opportunities for wildlife, agriculture, forestry and fisheries across terrestrial, freshwater, coastal and marine environments. The benefits people obtain from the natural environment, such as livelihoods, identity, security, health and well-being, provide important links with other chapters in the Evidence Report. These key linkages are further investigated in Chapter 8 (Cross-cutting issues).

3.1.1 Purpose of chapter

Climate is a key influence on the development and pattern of natural processes, therefore any changes in climate will have a strong interaction with the natural environment. Previous UK and global climate risk assessments have identified a high susceptibility of natural systems to present and future climate change because of this relationship. Risks may be both direct and indirect because of the many interconnections that exist between the environment and human use of natural resources. For this reason, previous assessments have highlighted the need to develop a cross-sectoral, integrated approach to adaptation. Both CCRA1 and the UK National Ecosystem Assessment identified the key role that ecosystems can play in helping to buffer and alleviate the risks from climate change. However, considerable evidence shows that the natural environment in the UK is already experiencing many other pressures in addition to climate change, including adverse management practices, habitat loss and fragmentation, pollution and overexploitation of natural resources. To place climate change in context, this chapter aims to highlight areas of interaction between climate change and socio-economic factors, including the relative contribution of climate change versus other drivers of risk, both now and in the future.

3.1.2 Approach to chapter

While recognising the intrinsic value of the natural environment, the structure of this chapter is based on assessing climate risks and opportunities against the key societal benefits provided by natural capital. Natural capital is the stock of natural assets from which humans derive a wide range of services that support and enrich life, including economic growth and people's health and well-being. A natural capital framework recognises that ecosystems are self-organising, interconnected and dynamic, existing at multiple scales. By adopting a systems perspective to analyse climate change risks, the chapter recognises that natural capital and ecosystems as a 'whole' are greater than the 'sum of their parts'.

The chapter assesses climate change risks to and opportunities for well-recognised goods and services, such as the provision of food, water and energy. Risks and opportunities of other less visible benefits are also assessed including: buffering against natural hazards (e.g. flooding and erosion), pollination, nutrient cycling and natural carbon storage. Less tangible but equally

[1] Recognition of intrinsic value is also consistent with recent international agreements developed under the aegis of the Intergovernmental Platform on Biodiversity and Ecosystem Services.

important benefits accrue through cultural associations with the natural environment, including their recreational, aesthetic and inspirational values. These benefits have high social and cultural significance, showing important interactions between this chapter and Chapter 5 (People and the Built Environment).

Ecosystem service relationships are typically complex and multi-layered (UK NEA, 2011, 2014). Intermediate ecosystem services are closer to natural ecosystem processes, while final services are more closely associated with human goods and benefits[2]. This chapter has generally aimed to follow this structure, proceeding from biodiversity[3] and ecosystem functions through to intermediate and final services and their benefits (Figure 3.1). However, for ease of explanation we have diverted from this structure occasionally to present topical issues together to help maintain policy relevance. In particular, a distinction is made between terrestrial, freshwater, coastal and marine environments.

From an economic perspective, some argue that valuation of ecosystem services should only be applied to those things directly consumed or used by a beneficiary (i.e. the benefits provided by final ecosystem services) because the value of constituent ecosystem processes and intermediate services is embedded within that estimate (UK NEA, 2011). However, it is acknowledged that such a perspective will only provide a partial rather than full societal value of natural capital and therefore the chapter aims to place equal emphasis across the ecosystem hierarchy.

The structure adopted for synthesising evidence is strongly aligned with the notion of ecosystem-based adaptation (EbA**)**, which is consistent with advocacy of the increased use of the ecosystem approach in policy development, and particularly in national adaptation policy (UK and devolved). With regard to climate change, the aim of EbA is to avoid piecemeal interventions for managing biodiversity and ecosystem services by instead designing more holistic joined-up strategies that aim to work in synergy with the natural environment and the maintenance of healthy functioning ecosystems. This is also consistent with the rationale for a thematic cross-sectoral risk assessment. In some cases, the framework also serves to identify strategic gaps in evidence that limit current confidence levels.

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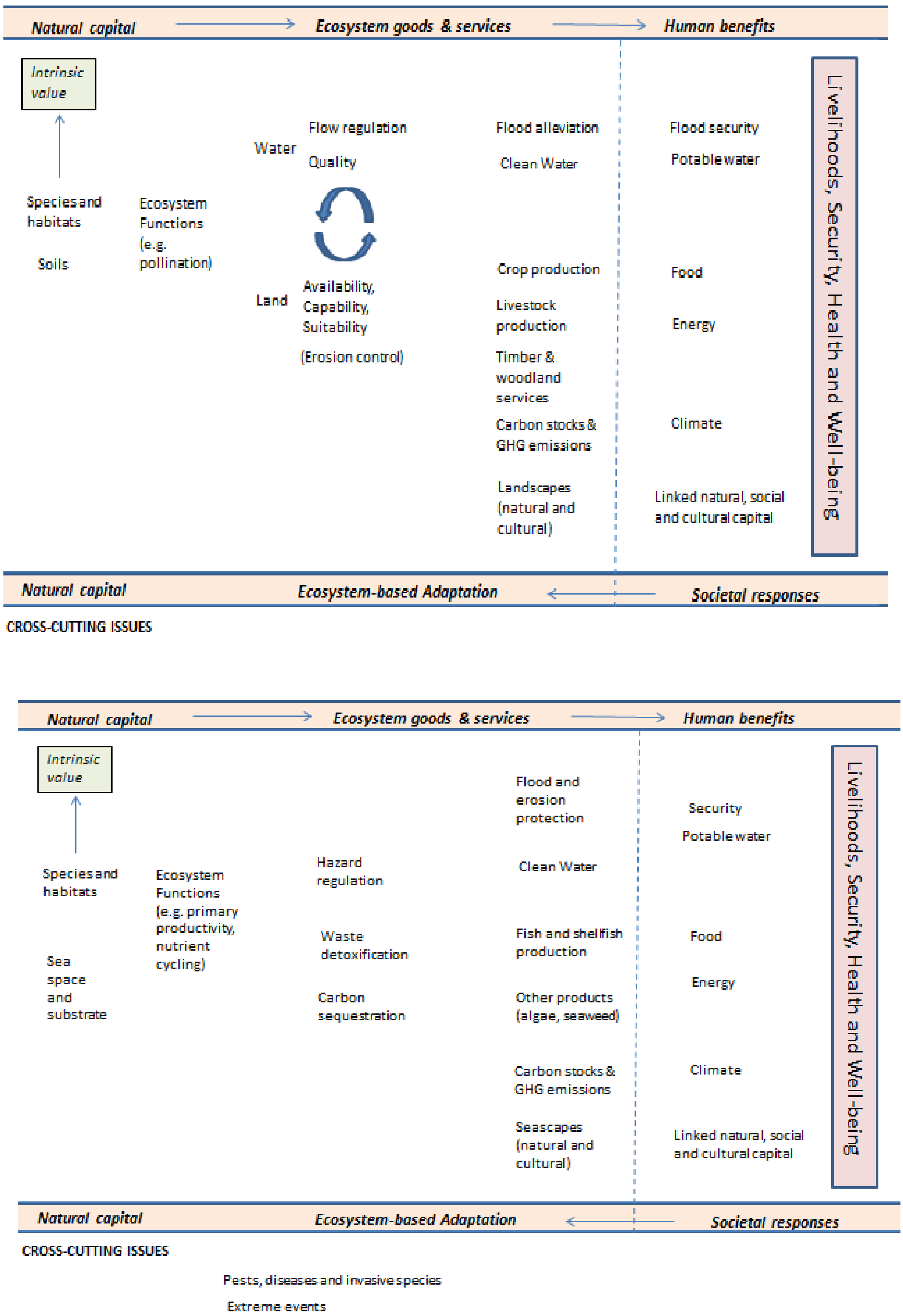
2 Goods and benefits are products or experiences derived from ecosystems but not connected to them, hence they may be traded or otherwise exchanged or shared. To realise these benefits may require input from other forms of capital (e.g. manufactured capital). Services retain a connection to ecosystems.

3 'Biodiversity' can (depending on its definition) actually be defined as an asset at multiple levels, because species (or genetic) functional diversity is a key indicator of healthy ecosystem function and regulation but also the presence of a diversity of wild species provides important cultural benefits to people (Mace et al., 2012; Harrison et al., 2014). Biodiversity therefore has an intrinsic 'existence' value in addition to its utilitarian value in underpinning ecosystem services to society. In the context of adaptation strategies, diversity (at multiple scales) may therefore be considered theoretically to provide an insurance strategy by acting to buffer ecosystems against change and therefore providing enhanced reliability for ecosystem processes and services, including those that help maintain food, water and energy security.

Figure 3.1. Relationship between biodiversity, soils and ecosystem services for (a) the terrestrial and freshwater environment and (b) marine and coastal environment



Source: CCRA Evidence Report authors.
Notes: As used to guide the structure of the chapter.

3.1.3 Comparision with approach to CCRA1

The first CCRA in 2012 took a sectoral approach to the assessment of climate risks. The CCRA1 sectors relevant to this chapter were Biodiversity & Ecosystem Services, Agriculture, Forestry, Water, Flooding, and Marine. Using a prioritisation approach based upon social, economic and environmental consequences, a long list of risks was reduced to a shorter list of priority risks. These priority risks were evaluated using climate metrics that aimed to quantify changes in risk based upon an assumed relationship between a climate variable and an indicator metric that was considered to represent the consequences. However, the climate-metric approach could not be realistically implemented for some sectors (notably Biodiversity & Ecosystem Services) because of the complexity of multiple interacting factors and its susceptibility to infer spurious correlations (e.g. for crop yields) due to other dominating socio-economic influences. Nevertheless, a broad consensus of views from stakeholders and scientists suggested that most of the priority risks had been successfully identified and that a more systematic rather than reductionist approach would be beneficial to focus more on these priority risks. It was also highlighted that further assessments should aim to address cross-sectoral issues that were under-reported in CCRA1 due to the sectoral structure, with an emphasis on requirements for adaptation actions.

3.2 Terrestrial ecosystems

3.2.1 Terrestrial species and habitats

Synthesis

The impacts of climate change can already be seen across a wide range of UK species and habitats. The current degraded state of many habitats and ecosystems exacerbates their vulnerability to climate change risks and extreme events. It also acts to constrain potential opportunities that may arise from the movement of continental species, particularly those which are rare and threatened in their current locations.

Future climate change is expected to lead to major changes in species distributions and interactions. Species which are restricted to the coldest parts of the UK are at long-term risk from warming temperatures, notably those in montane habitats. An important element of this risk is from increased competition from other species better adapted to warmer conditions. There are long-term, low-regret adaptation actions which can start immediately on nature reserves and other locations that can implement intensive conservation action:

- Protect populations of the rarest species by direct management (e.g. reducing competition).
- Identify 'refugia' locations where the chances of survival are best because of locally cooler temperatures and where conditions for existing species can be improved as far as possible.

A further category of species is those which are at risk in many of their present locations but could potentially colonise new areas. The resilience of these populations can still be enhanced by increasing the area and condition of existing sites and this should be a first priority as it is low regret and can start now. In addition, a number of longer-term responses need to start now because of their long lead time:

- Increase size and improve condition of sites to build population resilience.
- Develop ecological networks to enhance potential for natural colonisation of new sites.
- Develop options for managed translocation to suitable new locations.

Synthesis

- Ensure flexibility of site management objectives and condition assessment to recognise changing distributions of species.

The impacts of extreme events on vulnerable habitats are an immediate threat which is likely to increase with climate change. Wetlands are particularly vulnerable to reduced water availability. Urgent action is required to:

- Reduce factors which increase vulnerability, particularly to drought, for example by encouraging a diversity of species in woodlands, or by reversing drainage of wetland habitats.
- Ensure options for management are available (e.g. water control structures to maintain water in wetland habitats, access for fire service and developing early warning systems for wildfires).

The large-scale degradation of habitats is a long-term risk which requires immediate action because of the long time periods for successful restoration. Targeted actions include:

- Restore degraded wetlands, including blanket bog.
- Work at the catchment scale to restore ecosystem function (linking terrestrial and freshwater ecosystems)
- Ensure semi-natural habitats are of sufficient size for ecological processes to operate and build resilience.

Context and policy

Legal protection of wildlife stems from the National Parks and Access to the Countryside Act 1949, which created powers to designate nature reserves and what are now termed Sites of Special Scientific Interest (SSSI) (Areas of Special Scientific Interest in Northern Ireland). Of subsequent legislation, the Wildlife and Countryside Act 1981 is of particular note. There are currently four statutory country conservation bodies, Natural England, Natural Resources Wales, Scottish Natural Heritage and the Northern Ireland Environment Agency. The Joint Nature Conservation Committee plays a UK-wide advisory role. A total of 2.6 million hectares (10.6%) of the UK land area is designated as A/SSSI.[4]

A new tier of protection was introduced by EU Directives. Special Protection Areas (SPAs) are designated under the Birds Directive (1979) and Special Areas of Conservation (SACs) under the Habitats Directive (1992), which required the establishment of 'a coherent European ecological network' covering terrestrial, freshwater and marine environments. The majority of SPAs and SACs are underpinned by SSSI designation In addition, there are sites notified under the Ramsar Convention on Wetlands of International Importance, which are also SSSI. As well as nationally and internationally protected sites, local councils can recognise locally important wildlife sites for protection in the planning system.

Conservation beyond designated sites is also required by both Birds and Habitats Directives. The Nature Conservation (Scotland) Act 2004 requires public bodies to take appropriate action 'to further the conservation of the overall diversity, richness and extent of the natural world'. Scottish ministers were required to designate a Scottish Biodiversity Strategy and publish a list of species and habitats considered to be of 'principal importance' for the conservation of

[4] 2015 figures; http://jncc.defra.gov.uk/page-4241

biodiversity. The Natural Environment and Rural Communities Act 2006 introduced similar requirements for England and Wales, as did the Wildlife and Environment Act 2011 in Northern Ireland.

At the international level, the UK is signed up to the UN Convention on Biological Diversity (CBD) and made a number of commitments at the Aichi Conference of the Parties in 2010. The EU Biodiversity Strategy in 2011 set out the EU response to the Aichi Targets. As biodiversity policy is a devolved matter, each of the four UK nations has its own strategy.

In England, the 2011 White Paper *The Natural Choice: securing the value of nature* set out the Government's vision for the natural environment. This recognised climate change as a serious threat to biodiversity and ecosystems, and emphasised the need to take an integrated landscape-scale approach to conservation in response to the recommendations of the *Making Space for Nature* review (Lawton et al., 2010). Alongside the White Paper, *Biodiversity 2020: A Strategy for England's Wildlife and Ecosystem Services* was also published in 2011. This strategy sets high-level outcomes that demonstrate how action being taken in England is contributing to implementing the UK's international and EU commitments. *Biodiversity 2020* is clear that climate change represents a challenge to the way biodiversity is conserved. The strategy includes an undertaking to 'keep under active review our response to, and our increasing understanding of, the implications of climate change' in the implementation of the strategy. There are also specific climate-related commitments, particularly a target to restore 15% of degraded ecosystems as a contribution to climate change mitigation and adaptation and consideration of the impact of climate change on Natural England's SSSI Notification Strategy.

The Welsh Government published the *Nature Recovery Plan for Wales* in December 2015.[5] This sets out the commitments to biodiversity in Wales, and identifies the issues to be addressed to reverse the loss of biodiversity. It includes objectives to tackle the key pressures including climate change, particularly by increasing the resilience of the natural environment, to improve the ability of species and habitats to adapt. The Nature Recovery Plan also sets out how Wales will deliver the commitments of the UN Convention on Biological Diversity and the EU Biodiversity Strategy.

The *2020 Challenge for Scotland's Biodiversity* was published by the Scottish Government in 2013 and is Scotland's response to the Aichi Targets and the EU Biodiversity Strategy. It supplements and updates *Scotland's Biodiversity: It's in Your Hands* (2004). *Scotland's Biodiversity: a Route Map to 2020* was published in 2015 to guide the collaborative work to meet the aims of the 2020 Challenge and the Aichi targets ober the next five years. The Route Map includes 12Priority Projects including actions for restoration of peatlands, woodlands, and freshwaters, as well as other actions for species, land management, etc. The three documents together comprise the Scottish Biodiversity Strategy.

Valuing Nature, A *Biodiversity Strategyfor Northern Ireland to 2020*–was published in 2015. The Strategy sets out how Northern Ireland plans to meet its international obligations and local targets to protect biodiversity. It builds upon the first Biodiversity Strategy published in 2002, but follows an ecosystem services approach. Reducing the impact of climate change is one of seven high-level challenges identified by the 2015 strategy.

5 http://gov.wales/topics/environmentcountryside/consmanagement/conservationbiodiversity/?lang=en

Current risks and opportunities

Species and interactions

All species of plants, animals and micro-organisms are sensitive to climate. This is manifested in the climatic limits within which each species can survive: different species are adapted to different conditions, which determines where they occur globally. The mechanisms through which these limits operate vary. In some cases physiological constraints prevent organisms living beyond their climatic ranges, however interactions between species can also play an important role. One species may not be able to persist without another; for example, pollinator, host or food source. Alternatively, a species may be outcompeted by other species in sub-optimal climatic conditions. Even within species climatic tolerances, climate and weather can influence the timing of seasonal events, the success of breeding and interactions within species, with consequences for populations, biological communities and ecosystems.

There is a strong consensus in the scientific community that climate change has caused changes in the UK's biodiversity and ecosystems in recent years (Morecroft and Speakman, 2015). There is now very clear evidence that many species are moving to higher latitudes and altitudes, both within the UK and internationally, responding to changes in climate by colonising new areas as the climate becomes suitable [High confidence]. The UK has arguably the best information in the world on species distributions and how they have changed. A recent study of 1,573 animal species with northern limits in the UK showed that most had moved northwards over the past four decades (Mason et al., 2015). The average (mean) northwards movement was 23 km per decade in the first half of the period and 18 km per decade in the second half.

Many factors can influence the distribution of an individual species, including geology and soils, management history and interactions with other species, as well as climate. However, rising temperatures are the only environmental change that can realistically explain this general trend across the full range of species. The evidence also suggests that although species are shifting their range northwards, for many species groups it is at a lesser rate than would be expected based upon the change in temperature due to natural or land-use constraints on their dispersal ability (Hickling et al., 2006; Devictor et al., 2012) [High confidence]. There are also examples of species which have colonised the UK from continental Europe and spread northwards across a large part of the country, for instance the Small red-eyed damselfly. The evidence is less clear for plant species, many of which disperse relatively slowly (Morecroft and Speakman, 2015).

While an expansion northwards in species ranges has been clearly demonstrated in the UK, there is less evidence of a retraction of species ranges in their southern parts (Thomas et al., 2006). This would be expected for two main reasons. Firstly, relatively few species are at their southern range margins in the UK, whereas many more are at their northern range margin. Secondly, there is considerable inertia, for example, in long-lived species of plants a population may survive for decades even if it could not successfully reproduce. Even among short-lived species, populations may decline steadily over a period of years or decades before they actually disappear. In contrast, at northern range margins, a small number of individuals may be able to colonise a new site and be recognised in a short period of time. For species at the northern edge of their range in the UK there is the potential for new areas to be colonised and populations to increase, and there is evidence of species, such as the Dartford warbler, benefiting in this way. However, not all species are able to disperse quickly and not all species that could potentially colonise new areas are doing so – the rates of colonisation differ markedly between species. It is also the case that suitable habitat conditions need to be present for a species to spread, so a chalk grassland specialist will not be able to colonise if there is no chalk grassland, which in turn depends on both geology and management history. Species at their southern range margin in the UK are at significant risk of being lost from current parts of their range. A study has shown that this has happened with colonies of montane butterfly species, such as the Mountain ringlet, being lost from low altitudes while they are surviving at higher elevations (Franco et al., 2006). It is also the case that species expanding at their northern range margin in the UK, may well be declining at their southern range margin in other countries.

While changes in range are taking place, there are also natural mechanisms that increase the chances of some species surviving in their present locations, at least for more moderate levels of climate change. Some species may be able to survive as a result of genetic adaptation (Neaves et al., 2015) and species which have not shifted their distribution are more likely to have changed phenology, which may also improve their chances of persistence (Amano et al., 2014). Species are more likely to persist in some areas than others as the climate warms, these areas, termed 'refugia', are characterised by locally cool microclimates (such as on northern slopes and at higher altitudes), less intensive land use and a lower rate of temperature increase (for example close to the coast) than elsewhere (Suggitt et al., 2014).

Both range expansions and contractions are influenced by land management decisions and resulting habitat condition, as well as climate. [High confidence] In the past five years a number of publications have shown the value of SSSIs in facilitating colonisation at the northern margin (Thomas et al., 2012) and enabling persistence at the southern margin (Gillingham et al., 2015).

Climate risks and opportunities are apparent within current species distributions as well as at the margin. The capacity to identify population trends at a national level is constrained by the relatively small number of groups of species for which good regular monitoring data is available (for example butterflies, moths and birds). However, recently published analyses of population trends in 501 terrestrial species of mammals, birds, aphids, moths and butterflies, suggested that long-term trends in weather variables had a significant impact on 64% of species. There were significant variations in species responses to temperature and precipitation which make it difficult to generalise for taxonomic groups. However, there were significant long-term impacts of climate change on overall abundance trends in aphid polualtions, which increased by 0.7% per year between 1970 and 2010 and moth populations whichdeclined by 1.4% per year between 1975 and 2010. Particularly large increases (>30% per decade) were found in populations of five species across different groups, and similarly large large declines were found in nine moth species (Pearce Higgins et al., 2015). The analysis identified generally positive effects of temperature upon terrestrial invertebrates during spring and summer, but negative effects of warm, wet winters on moths and butterflies and of precipitation at other times, and of hot summers upon upland freshwater macroinvertebrates. Within groups, species-specific variation in population trends indicated that climate change also had impacts on bird communities.

Large variations in rainfall have been recorded in recent decades and there are some examples of the ecological impacts of extreme events. These cannot generally be individually attributed to climate change but they are evidence of climate sensitivity and general vulnerability to the frequency of extreme events, particularly drought episodes. Morecroft et al. (2002) reported contrasting impacts of drought in the mid-1990s, with some invertebrate species benefiting and others being adversely affected depending on the habitats they are adapted to and ability to colonise new areas. During a drought there are risks of sensitive tree species being lost, such as beech, with far-reaching consequences for woodland habitats (Cavin et al., 2013). Wetlands and water courses are vulnerable to drying up, resulting in declines or loss of species they support, although there is a degree of resilience in some ecosytems (Moss, 2015).

Oliver et al. (2015) recently showed that the impacts of droughts on butterfly populations are reduced in larger areas of semi-natural habitat compared with small areas. This suggests that restoration or creation of large areas of habitat could be an effective component of climate change adaptation. Furthermore, landscape diversity has been shown to enhance population stability in some species, apparently by increasing the variety of local ecological and climatic niches by which these species can adapt to changing annual and longer-term conditions (Oliver et al., 2010).

Changes in temperature are affecting phenology of species [High confidence]: in particular there has been an advance in the timing of spring life cycle events, with Thackeray et al. (2010) reporting an average advance of 11.7 days from 1976 to 2005 across a wide range of taxa. This may in turn have an impact on the interactions between species, including trophic relationships, pollination and competition (including between canopy and ground flora in woodlands).

Climatic effects on the almost 250 migratory species that are found in the UK are complex and include changes in other parts of the world, such as sub-Saharan Africa. However, some have been well documented: most avian species are migrating less (e.g. wading birds are increasingly overwintering on the east rather than west coast) and summer migrants are arriving earlier (Morrison and Robinson, 2015).

Habitats and ecosystems

Habitats will be affected by the direct impacts of climate change on the species of which they are composed, and the interactions between those species. In addition, climate change interacts with a wide range of other pressures affecting habitats and may exacerbate ongoing risks. These include land use change, pollution, habitat fragmentation, adverse management practices and recreational pressures. To date, drought is likely to have been the most significant climate-related driver of vegetation change, although it is suggested that the impacts have been limited by the natural 'inertia' of plant communities (Carey, 2015). [Low confidence]

Montane: Alterations related to climate change in some montane habitats have been observed. In particular, the composition of montane vegetation in the north-western Scottish Highlands is changing as some arctic-alpine species are declining (Ross et al., 2012). An analysis of mean snow cover duration from October to May in the central Scottish Highlands showed a decrease between 1979 and 2003, during which time mean October to May temperatures and mean precipitation increased (Trivedi et al., 2007). Repeat surveys of snow-bed vegetation in Scotland have shown a shift towards a community of more open character, with an increase in species such as Highland Rush and a decrease in specialist snow-bed liverworts (Morecroft and Speakman, 2015). Over the past 60 years, on Ben Lawers (central Highlands, Scotland), montane habitats and acid grasslands have become more homogeneous, primarily due to climate change, while upland fens, marshes and swamps have been less affected and have retained more of their original character (Ross, 2015). In addition, soil acidification, resulting from atmospheric nitrogen and sulphur deposition, which is often higher in montane regions due to enhanced average annual rainfall, is possibly leading to a decline in species richness and composition in the associated acid grasslands and heaths, although overgrazing may also be contributing to this trend (Ross et al., 2012). A broader-scale study of factors affecting *Racomitrium* heaths across Europe, including sites in Snowdonia, Cumbria, Southern Uplands and Scottish Highlands, found that nitrogen deposition was the main pressure affecting habitat condition and loss, with climate (primarily temperature) and grazing impacts as secondary factors (Armitage et al., 2012).These slightly different findings on the drivers of change may be due to the different scale of study and also the focus of the latter on the *Racomitrium* heaths, which have a higher percentage of mosses and lichens that are particularly sensitive to pollution.

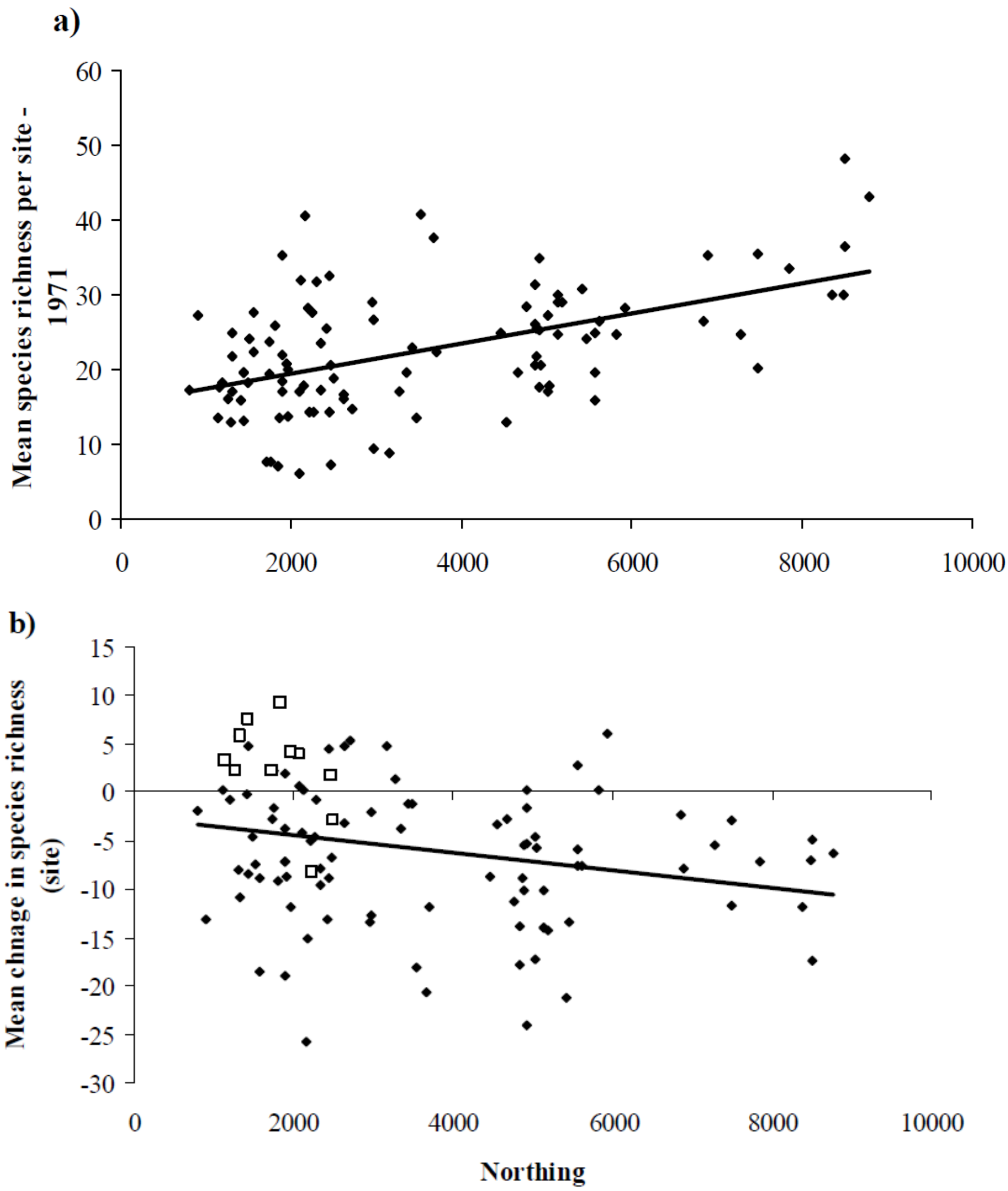
Wetlands: These ecosystems contain communities of species that are adapted to saturated conditions, either seasonally or all year round, such as bogs, mires and fens. Currently, many blanket bogs have been degraded as a result of drainage, overgrazing, burning and nutrient runoff pressures, which lead to a greater vulnerability to drying and erosion (ASC, 2015; Morecroft and Speakman, 2015).

Grasslands: Some habitats are sensitive to reductions in precipitation and to drought. There is evidence that in southern England drier summers have affected species composition of semi-natural lowland grasslands (Morecroft et al., 2002). There is also experimental evidence from 13 years of climate change manipulations of limestone grassland at Buxton, northern England, of the habitat's overall resilience to climate change, with constancy in the relative abundance and dominance of many growth forms, small shifts in abundance of some species and only few losses in response to drought and winter heating compared to inter-annual variations due to climate (Grime et al., 2008). Some changes in species abundance and community composition (Fridley et al., 2011), as well as genetic responses by individual species (Ravenscroft et al., 2015), have been identified.

Heathlands: Currently, climate change is not the most significant pressure although some specialist heathers of lowland heath are disappearing due to reduced rainfall in south-eastern England (Carey, 2015). Also, a re-survey of vegetation plots in the Scottish Highlands after 50 years showed changes in species composition and a decrease in species richness in dwarf-shrub heaths (Ross et al., 2012) and in oceanic-montane liverwort-rich heaths (Flagmeier et al., 2014). These changes were thought to be associated with grazing, runoff, increasing temperatures and changes in precipitation patterns. Other non-climatic pressures include habitat loss, air pollution, recreation and lack of (appropriate) management. A 13-year experiment assessing the impacts of nitrogen deposition on the structure and functioning of the lowland heath habitats in Surrey, southern England, found that drought sensitivity increases under elevated deposition (Southon et al., 2012).

Woodlands: A repeat survey of 103 British woodlands (1970 – 2001) detected a long-term decline in the species richness of ground flora (excluding bryophytes and lichens (Figure 3.2). This is thought to be mainly due to the deterioration of woodland quality and reduced management, although changes in the timing of seasonal events (such as earlier tree-leafing) may also have had adverse implications for ground flora and other woodland species due to phenological mismatches (Kirby et al., 2005). The data also shows that there have been gains in species richness in some woodlands in southern England as a result of the 1987 storm, illustrating how extreme events can be beneficial for biodiversity in the longer term. The study found little overall change in dominant tree species. However, there is evidence from other studies of shallower-rooted trees (such as beech) being replaced by deeper-rooted species that are less sensitive to drought (Morecroft and Speakman, 2015). Long-term changes in the ground flora of Wytham Woods, Oxfordshire, have been attributed to multiple drivers, including changing nitrogen deposition and deer numbers (Corney et al., 2008).

Figure 3.2. Change in mean species richness across 103 British woodland sites between 1971 and 2001



Source: Kirby et al. (2005).
Notes: Northings are the distance north (in km) from an origin located south-west of the Isles of Scilly as used by the British national grid. Graph (a) depicts mean in species richness in 1971 against northing (i.e. how far north the site is located) and thus is showing increased species richness further north.Graph (b) depicts the mean change in species richness between 1971 and 2001. This shows that the decrease in richness was greater the further north the site was located, although this is partly offset by the gain in species richness at sites affected by the 1987 storm (open squares).

Future risks and opportunities

Species and interactions

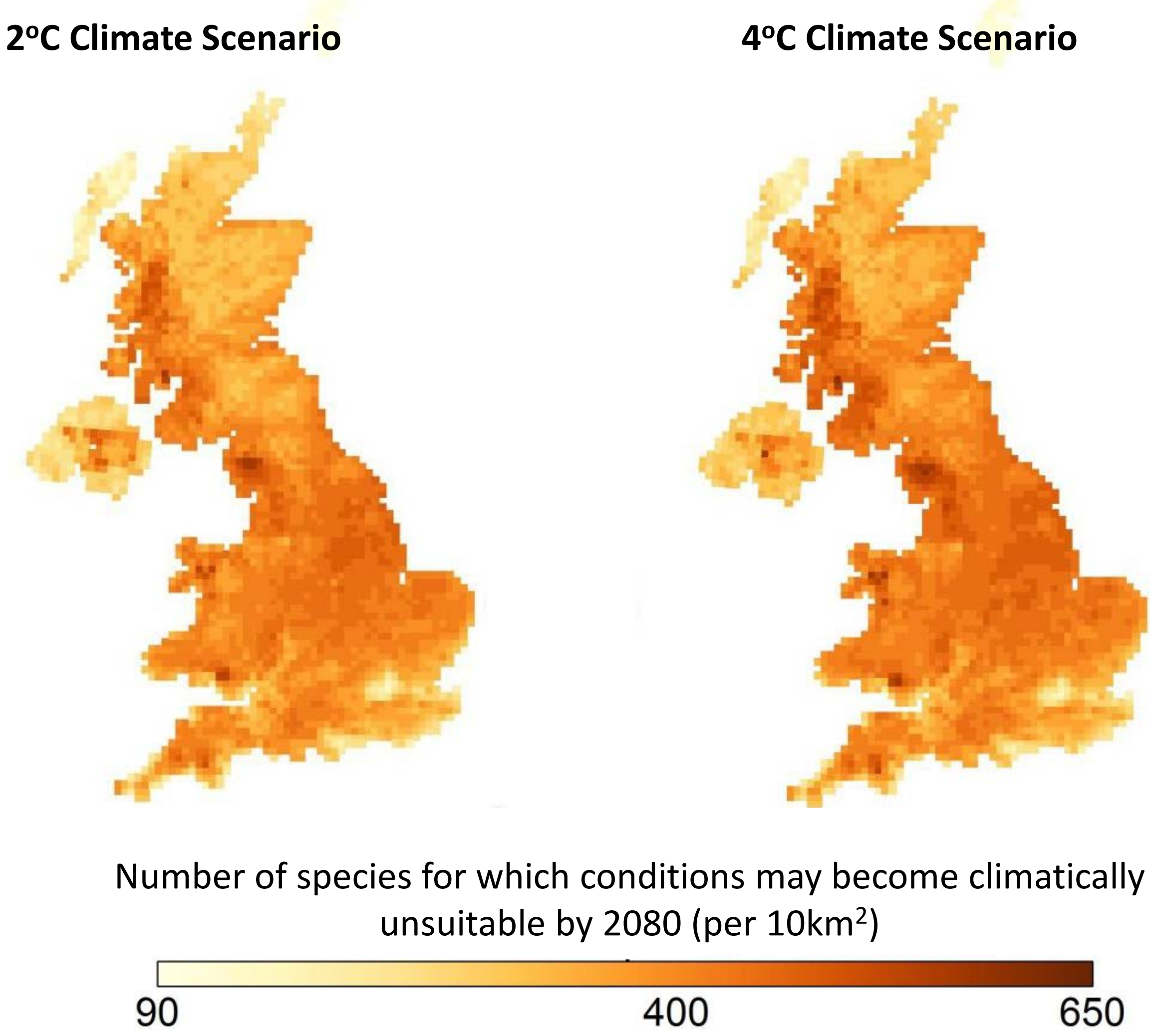
Projected changes in temperature and water availability, both annually and seasonally, are likely to have a significant impact on many aspects of species biology, and the interactions between species. This in turn will have consequences for population size, the composition of biological communities, the survival of species in their present locations and their potential to colonise new locations (Morecroft and Speakman, 2015).

The trend towards species colonising more northerly and higher altitude locations can be expected to continue. We would also expect to see more evidence of species declining and becoming locally extinct at the southern and low-altitude margins of their range.

The association between species distributions and climate can be modelled to provide an indication of species potential climatic range. This is referred to as bioclimatic envelope modelling. It should be noted that new climatically suitable areas may not actually be colonised, either because of a lack of suitable habitat or inability of species to move across the landscape. Bioclimate modelling is not likely to be a reliable guide to future potential in cases where actual current species distribution does not reflect the current modelled distribution. This may be due to the influence of a range of other factors on species distribution, including soils, geology, land management or intrinsic species rarity. Bioclimate modelling is nevertheless a useful technique for exploring potential changes across a wide range of species. Various studies of this sort have been carried out for the UK in recent years (e.g. Walmsley et al., 2007; Berry et al., 2007; Pearce Higgins et al., 2015).

Using new developments in bioclimate modelling, an assessment of changes in species potential ranges was carried out for CCRA 2017. This assessed change in potential range for over 4,700 native British species across 17 taxonomic groups against three climate scenarios (Beale et al., 2015). The results of the modelling show that there is projected to be a significant amount of variability between and within taxonomic groups, with species potentially gaining and losing climate space and some experiencing little change. Although 16 out of 17 broad taxonomic groups showed potential range expansions, localised species extinctions were also identified (Figure 3.3). Because more species are at their northern limit than at their southern limit in the UK, there is a larger number of species for which the area of suitable climate is projected to expand. However, the opportunity for species to occupy these potential new locations is constrained by the ability of species to move across the landscape and the availability of suitable habitat conditions in these places. As has been seen with recent range expansions, many species are unlikely to be able to fully exploit the potential opportunity that a warming climate presents.

Figure 3.3. Projected numbers of native UK species for which conditions may become climatically unsuitable by 2080



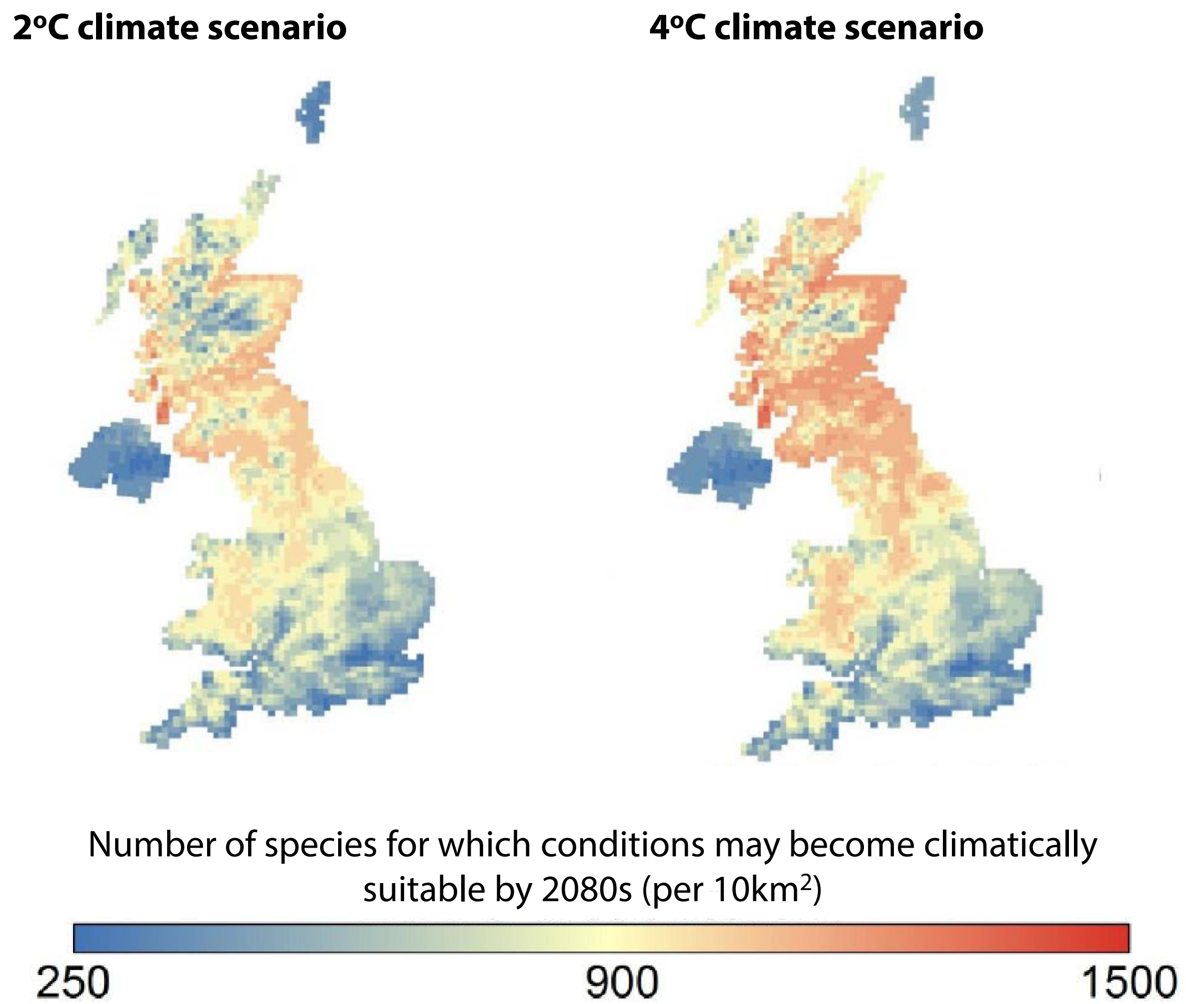
Source: Suggitt et al (2015) for the ASC.
Note: Projected numbers of species for which suitable climatic conditions may be lost from10 km grid cells for 2°C and 4°C climate scenarios in the 2080s across all taxonomic groups.

The same modelling work projected that climate in substantial areas of northern England and Scotland might become suitable for many new species, although this is less true of upland compared to lowland areas of northern Britain (Figure 3.4). It is important to note that these results do not account for potential colonists from further south in continental Europe thatcould be significant. It is also the case that species with the potential to expand their range in the UK may retract in other places.

The modelling shows that there is a higher probability of change (in terms of both local extinctions and colonisations) within protected areas (SACs, SPAs and SSSIs) than in non-protected areas in the wider countryside. The greater change expected within protected areas is likely to reflect the general tendency for protected areas to be sited in regions of atypical environmental conditions. Rare and specialist species in such areas are more sensitive to changes in climatic conditions than generalist species.

All bioclimatic envelope studies show that species associated with cold montane habitats are likely to see a contraction of their potential range to the most northern and high-altitude locations, with extinction risk greatest in England and Wales. [High confidence] Even the Scottish Highlands may fail to provide suitable conditions if climate change continues on its present trajectory into the second half of this century. Additionally, modelling shows that as changes in climate develop, species of more intermediate latitudes may start to lose suitable climatic conditions in the south of the UK. This loss of suitable conditions is one of the most serious risks to biodiversity in the UK.

Figure 3.4. Projected numbers of native UK species for which conditions may become climatically suitable by 2080



Source: Suggitt et al for the ASC (2015).
Note: Projected numbers of species for which climatic conditions may become suitable by 2080 per 10 km grid cell for 2°C and 4°C climate scenarios across all taxonomic groups.

As the climate warms further, the possibility of changes in the interactions between species will increase. For example, distributional changes may result in new combinations of species occupying particular locations, giving rise to so-called non-analogous communities (Keith et al., 2009). Current approaches to conservation include planning based on the presence of recognisable plant communities characterised as broad habitats or priority habitats. This approach is likely to require regular revision. It also increases the prospect of a breakdown in phenological synchrony between species, leading, for example, to an absence of food species or pollinators at the right time for some organisms.

Some species will only be able to be maintained in some locations if adjustments to catchment management and drainage are made to enable wetlands to persist, for example by holding water back higher in catchments and reducing abstraction.

Habitats and ecosystems

If mean January temperatures move above 7°C and mean July temperatures above 22°C[6] then climatic conditions may become more suitable for Mediterranean or sub-tropical vegetation on suitable substrates(Carey, 2015).[Low confidence] This would have potentially profound implications for the UK's current suite of habitat types, with significant change particularly likely for lowland heath, dry acid grassland, lowland calcareous grassland and maritime cliffs and heaths (Carey, 2015).

Montane: in Scotland, snow cover and its duration are projected to decrease due to increases in temperature [High confidence] , with the possibility of no snow cover below 900 m under a UKCIP02 2080s High scenario (Trivedi et al., 2007). This could lead to the loss of snow-bed communities through the invasion of more competitive species from lower altitudes (Callaway et al., 2002), such that the community becomes dominated by low shrubs such as *Calluna vulgaris* and *Vaccinium myrtillus* (Trivedi et al., 2007). There is little opportunity for adaptation in these habitats.

Wetlands: these communities are at risk from decreased water availability. Wetter winters and drier summers would probably see alterations in the plant communities and could lead to Purple moor grass and rush pastures changing to acid grassland (Mitchell et al., 2007). One of the most studied wetland habitats in relation to climate change is blanket bog. Blanket bog is generally found in cool, wet climates with waterlogged peat soils in the north and west of the UK. Recent studies have investigated the risk of suitable conditions not persisting in parts of the UK. Using an ensemble of bioclimatic envelope models, Clark et al. (2010) inferred that climate change is likely to pose increasing pressure on blanket bog, with large areas of existing habitat (31 – 89%) being modelled as less climatically suitable by 2050, particularly under high climate change scenarios.This pressure was greatest where blanket bog was on the warm and dry margins of its distribution in the south-west England and in the eastern half of the UK (e.g. North York Moors). Similar results were obtained using a separate, globally calibrated bioclimatic envelope model, PeatStash, under various UKCIP02 scenarios (Gallego-Sala et al., 2010). By 2071 – 2100 under a high-emissions scenario, the blanket peatland bioclimatic space was estimated to be ~84% smaller than baseline (1961 – 1990), with only parts of the west of Scotland remaining bioclimatically suitable. Increasing summer temperatures and resultant soil moisture deficits with a lowering of the water table were inferred to be the main drivers of the projected changes in habitat extent. Similarly, Carey et al. (2015) also found that, by the 2080s, 59% of modelled blanket bog areas would change NVC type, and Li et al. (2016) found that a warmer climate increased the risk of surface desiccation and resultant habitat loss through peat erosion (see Section 3.3.1). As with all bioclimate envelope modelling, the results need to be interpreted

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[6] The warmest parts of the UK currently have mean temperatures of 5 to 6°C in January and c.17°C in July.

with care. Factors such as relative humidity levels, which reflect important local upland conditions, may not be adequately represented in larger-scale projections of climate change (Lindsay, 2010). Blanket bog is a very diverse habitat and has also proved to be resilient to change over long periods of time, as species composition can change in response to a changing climate (Dise, 2009; Lindsay, 2010). When blanket bogs are in good condition they can modify their natural hydroclimatological conditions by impeding drainage and producing almost permanently saturated conditions, particularly for the larger, deeper bogs. Even if conditions are sub-optimal for bog formation and active growth they may nevertheless persist because of the high water tables. The threat to blanket bog is, however, significantly exacerbated if they are in poor condition, including where they have been drained or lost their vegetation cover, meaning they are not actively functioning. A high proportion of the UK's blanket bog is believed to be in degraded condition as a consequence of drainage, burning, grazing and atmospheric pollution (ASC, 2013); restoration measures such as blocking drainage channels (grips), preventing burning or reinstating vegetation can therefore significantly increase resilience.

Heathlands: lowland heath habitats are projected to be significantly affected by changes in vegetation composition under the range of UKCP09 projections (Low, Medium and High projections for the 2020s, 2050s and 2080s), with habitats in south-east England being particularly affected (Carey et al., 2015). [Medium confidence]The modelling suggests that lowland heath habitats could be replaced by species-poor acid grassland after the 2050s (Britton et al., 2001). Reduced precipitation could see significant reductions in wet heath, and lowland wet meadows could also be affected (Carey, 2015). Lowland heaths are also likely to become more susceptible to wildfires, especially during drought periods (see Section 3.2.10). The model outputs suggest that the composition of upland heath and native pine woodlands will be particularly affected, but that such changes will also be affected by non-climatic factors and management responses. Upland heaths, especially in south-west England, could become more like lowland heaths and higher temperatures could lead to more extensive bracken invasion (Carey, 2015).

Adaptation

The National Adaptation Programme (2013) identifies four focal areas for adaptation within the natural environment:

- Building ecological resilience to the impacts of climate change.
- Preparing for and accommodating inevitable change.
- Valuing the wider adaptation benefits the natural environment can deliver.
- Improving the evidence base.

The Scottish Climate Change Adaptation Programme (2014) includes an objective to support a healthy and diverse natural environment with capacity to adapt. Scottish Natural Heritage promotes a set of adaptation principles to help nature adapt, ranging from reducing other pressures, and making space for natural processes to planning for change. A set of case studies illustrates how these principles are being applied in practice on National Nature Reserves.[7] The

[7] http://www.snh.gov.uk/climate-change/taking-action/adapting-to-change/helping-nature-adapt/turning-principles-into-practice/

principles are also reflected in the 2020 Challenge for Scotland's Biodiversity as ways to improve ecosystem health

Similar principles for adaptation have been recognised in UK conservation policy and evidence documents over the past decade (Hopkins et al., 2007; Mitchell et al., 2007; Smithers et al., 2008) and climate change adaptation is a well-recognised issue in many statutory and NGO conservation organisations.

The *Making Space for Nature Review* (Lawton et al., 2010) advocated increasing the number, size, quality and connectivity of wildlife sites in England as important elements of increasing the coherence and resilience of the network of conservation sites. This review included climate change alongside a range of other considerations. Recent publications since the review have provided further evidence of the benefits of this landscape-scale approach, particularly on the effects of reducing habitat fragmentation in increasing resilience to extreme events (e.g. Oliver et al., 2015; Newson et al., 2014). A direct consequence of the review was the establishment of Nature Improvement Areas, and there is evidence that they have considered climate change adaptation (Van Dijk et al., 2013). There are also a number of other landscape-scale conservation initiatives (Macgregor et al., 2012) including the Wildlife Trusts' Living Landscapes and RSPB's Futurescapes. Landscape-scale thinking is also being applied more widely, for example in various National Parks and Areas of Outstanding Natural Beauty and through the network of Local Nature Partnerships established by the 2011 Natural Environment White Paper. A new Facillitation Fund has been established to support farmers in England to work together to achieve Countryside Stewardship prioirities across the landscape.

Conservation approaches to adaptation are also increasingly focusing on the potential of refugia, where localised conditions may allow species to persist (Suggitt et al., 2014). Identification and protection of these places is part of emerging approaches to climate change adaptation (e.g. Natural England, 2015). It is also the case that species distributions are not necessarily directly limited by physiological limits but by interactions with other species which are controlled by climate. In particular, plants adapted to cold conditions may not be able to survive in competition with faster-growing species adapted to warmer temperatures (Ross et al., 2012). This may offer the potential for management intervention to maintain species, at least on a localised basis.

A further element of resilience which is particularly important in the context of climate change is the restoration and maintenance of ecosystem processes. A key element in this is ensuring that the hydrology of wetland habitats is maintained in the face of increasing uncertainty and extremes of precipitation. Restoration of mires by blocking drainage ditches to allow peat formation to restart is a key element which is being promoted through a variety of measures, including LIFE, Countrysidel Stewardship and the Scottish Government's Peatland Action programme. In England, restoration schemes have been initiated on around one-third of the total area of upland peat (ASC, 2013) and there has been widespread lowland heath restoration delivered through Heritage Lottery Fund programmes and the Aggregates Levy. The Forestry Commission has also been protecting, improving and expanding woodlands on the Public Forest Estate and with private landowners. There is, therefore, evidence of some progress being made in developing ecological resilience.

Preparing for and accommodating inevitable change is a challenge broadly recognised by the conservation community, although so far practical action is relatively limited. Species modelling research suggests that there are likely to be implications for the current approach to the management of protected sites, which is orientated towards meeting set conservation objectives. To continue to be effective, site objectives may need to be adjusted to take account of both extinctions and colonisations. Changes in species distributions are increasingly being recognised in management plans for conservation sites. For example, provision is being made for more frequent 'short stopping' by migratory bird species on the east coast of the UK, in contrast to 'traditional' overwintering in west coast locations (Morrison and Robinson, 2015). There are also examples of SSSI boundaries and management objectives being amended, for example at Porlock in Somerset and on the North Norfolk Coast. The European Commission published Guidelines on Climate Change and Natura 2000 in 2013.[8] A review of the Natura Directives is also soon to be published, which may highlight the need to respond to climate change. The potential for increasing connectivity of habitat patches to facilitate species colonising newly suitable areas as the climate changes has been widely recognised (Heller and Zaveleta, 2008) and will be promoted by landscape scale conservation projects. There are however some species with intrinsically slow dispersal rates for which this will not be suitable and risks from invasive species need to be managed (Knight et al., 2014).

3.2.2 Pollination

Synthesis

Observed decreases in range, and probably declines in abundance are attributed primarily to changes in land use and agricultural policy and practice. Some pollinator species may have high susceptibility to changes in climate space and seasonality, with the possibility of future mismatches with flowering dates. Current evidence suggests complete disruption in pollination services is unlikely because of the presence of multiple pollinator groups. However, significant impacts are possible because of the multiple pressures and the pivotal role of some vulnerable species.

There is a need for more research to better understand the potential for mismatches due to changes in climate space and seasonality and the extent to which pollination disruption may occur, as well as how climate and non-climate pressures (including use of neonicotinoids) may interact.

Context and policy

While some plants are self-pollinated or wind-pollinated, most flowering plants, almost all fruit crops, and some grain crops require pollinators in order to produce fruit and seed. Pollination, therefore, is an important intermediate regulating service for food production.

There is no UK-wide or country-specific legislation relevant to the protection and management of pollinators. In England, the National Pollinator Strategy (Defra, 2014) seeks to improve the status of pollinators through: supporting their survival on farmland, across the wider countryside, and in towns and cities; addressing pest and disease risks; raising awareness of their needs; and increasing the evidence base on their status and the services they supply. Some of this can be achieved through measures under the revised Common Agricultural Policy (CAP), for example Ecological Focus Areas and the Countryside Stewardship scheme.

Wales was the first country to introduce an Action Plan for Pollinators in 2013 and a Pollinators Taskforce has been established. The plan identifies climate change as one of a number of threats to pollinators, alongside land use intensification, habitat loss and fragmentation, disease and agrochemicals. Actions include reviewing how existing delivery mechanisms, including agri-

[8] http://ec.europa.eu/environment/nature/climatechange/pdf/Guidance%20document.pdf

environment schemes, can be better targeted to improve habitat for pollinators. The plan will feed into the Welsh Government's Natural Resource Management programme and other policies and strategies currently in development as part of the Environment (Wales) Act 2016. .

In Scotland, a ten-year Honey Bee Health Strategy was published in 2010 which has four outcomes around education and training, communication, surveillance and biosecurity, and research and development. A consultation on a draft Scottish Pollinator Strategy was published in December 2015.[9]

An All-Ireland Pollinator Plan 2015 – 2020 has been published.[10] The plan provides a framework to bring together pollinator initiatives across the whole island, and is the start of a process to protect Irish pollinators and the service they provide into the future. It is supported by 68 organisations, many of whom have accepted responsibility for specific actions.

Current risks and opportunities

Pollinators, like many other species, face multiple pressures, from habitat loss, pests and diseases, extreme weather, competition from invasive species and the use of some pesticides (Vanbergen, 2014; Goulson et al., 2015). It is thought that the loss of flower-rich habitats is the most likely cause of loss of species richness in wild bees and other pollinating insects, while managed honey bees are more threatened by pests and diseases. Climate change is interacting with these pressures, but the challenges of distinguishing between multiple stressors means the evidence for its impact remains limited.

There is some evidence of changes in pollinator species distribution (Sparks and Collinson, 2007), especially losses at southern range margins (Kerr et al., 2015). For example, in Wales the range of some bumblebees and solitary bees has contracted (Vanbergen, 2013). Attributing these pollinator changes to climate change is not straightforward because of the complexity of plant–pollinator interactions. Research into the rate of extinction of bee and flower-visiting wasp species in Britain from the mid-19th century to the present day has attributed losses primarily to changes in land use and agricultural policy and practice (Ollerton et al., 2014; Senapathi et al., 2015). In Wales, agriculturally related land use intensification, habitat loss and fragmentation is seen as the main cause of pollinator loss (Welsh Government, 2013). A study of wild bee decline in the 20th century in the Netherlands suggested that climate sensitivity was not an important factor (Scheper et al., 2014). An analysis of the effects of land-cover change on the species richness and community composition of bee and wasp pollinator communities in Britain found that sites surrounded by arable expansion had a greater decline in species richness, but it was not possible to investigate to what extent fertiliser and pesticide usage were implicated (Senapathi et al., 2015). Bee species richness was positively affected by increases in surrounding woodland or urban area within a 1 km radius.

A comparison of species loss between 1930 and 1990 with losses recorded since 1990 suggests that some wild bees in Great Britain are starting to recover, possibly as a result of agri-environment measures (Carvalheiro et al., 2013). A meta-analysis found that agri-environment schemes have significant positive effects on pollinator species richness and abundance (Scheper et al., 2013).

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9 http://www.snh.gov.uk/about-scotlands-nature/species/invertebrates/land-invertebrates/pollinator-strategy-consultation

10 http://www.biodiversityireland.ie/projects/irish-pollinator-initiative/all-ireland-pollinator-plan/

There has been debate about the impacts of insecticides on pollinators, especially neonicotinoids, because of their widespread use and persistence. There is no consensus, partly due to differences in field and lab methods and dosages applied, but a review of the evidence is provided by Godfray et al. (2014). This suggests that even at experimantal doses that match those to which insects are exposed in the field, neonicotinoids can have a variety of sub-lethal effects on honey bee and bumblebee colonies, which in combination with other agrochemicals and disease could lead to colony collapse (Van der Sluijs et al., 2013). A partial restriction on the use of neonicotinoids was introduced at the EU level in 2013.

Spatial and temporal mismatches in the phenology of host plants and pollinators are recognised as a potential risk, but there is no current evidence that can be attributed to climate change (Hegland et al., 2009). Potential future mismatches will depend on the species, with specialists more likely to experience a mismatch. Multi-species plant–pollinator groups may be less susceptible to disruption by climate change (Rafferty and Ives, 2012). There are other more physiologically-based consequences of climate change, including issues of thermoregulation and foraging (see review by Willmer and Stone, 2004; Scaven and Rafferty, 2013). Many bees need considerable time for thermal regulation, which includes periods of basking in the sun to warm up or remaining in shade or inside the nest to cool down. This time comes at the cost of foraging, with negative consequences for pollination. For example, plants with flowers that open later in the day could receive fewer visits if pollinators have reduced activity by that point in the day, leading to reduced fruit and seed set (Wilcock and Neiland, 2002). Also, pollinators may have to restrict the length and time of their foraging trips to avoid overheating, which could alter patterns of pollen flow. Therefore physiological responses to warming may result in reduced pollination even without changes in species composition or phenological overlap.

Future risks and opportunities

Current susceptibility can give a guide to future vulnerability to climate change. For example, *Subterraneo bombus* (a species of bumble bee) appears to have narrow climatic ranges, with small colonies that start late in the year. They are also dependent on flowers with long floral tubes, such as red clover, which have become less common due to changes in agricultural practices (Williams and Osborne, 2009). A number of studies have modelled the impact of climate change on the potential suitable climate space of species, including pollinators. For example, Rasmont et al. (2013) have modelled 56 species of bumblebees in Europe at a 50 km scale up to 2050, using IPCC 2001 climate scenarios and a set of socio-economic storylines. All the results are scenario dependent, but five species might expand their ranges, four to 17 species (depending also on the dispersal assumptions) might more or less maintain their status quo, whilst suitability is projected to decrease for 34 to 52 species. Thirteen rare and localised species were not modelled due to lack of adequate data and they were thought likely to decrease under climate change.

Changes in species distribution and the timing of phenological events may lead to spatial or temporal mismatches between pollinators and their host plants (Petanidou et al., 2014), but many of the studies on this phenomenon are currently from areas outside Europe. One study modelled the current and future (2050) distributions of orchards in Great Britain and their pollinators using the SRES A1B emissions scenario, finding that while currently there is a large overlap, this does not continue into the future (Polce et al., 2014). The pollinators, however, are likely to persist in current orchard areas. A meta-analysis of UK phenological data showed that flowering dates have advanced more rapidly than invertebrate phenology, with the gap widening more recently, thus increasing the possibility of future mismatches, with a consequent reduction of pollination rates and seed production (Thackeray et al., 2010). [Low confidence]

Complete disruption in pollination services is thought to be unlikely (Smith et al., 2011) because pollination networks involve multiple species interactions which are relatively resilient (Hegland et al., 2009).

Climate change may also exacerbate the spread of non-native plants and pollinators but their effect on pollination is largely unknown. A field manipulation experiment has shown that invasive non-native species can have mixed effects on the pollination of native plants (Lopezaraiza-Mikel et al., 2007). It is also possible that trade in *Bombus terrestris dalmatinus* (for pollination of crops in greenhouses and poly-tunnels) could lead to the spread of disease into native bumblebee populations (Goulson, 2010). Although there is no consensus as to whether non-native bees have had a negative effect on native populations, they can contribute to pollinating native plants (Goulson, 2003).

Adaptation

Recent studies of non-syrphid flies that act as pollinators found that they may be more important for pollination than the traditionally studied syrphid *Diptera* (Orford et al., 2015). These species could be important in providing future pollination services.

In England, the Wild Pollinator and Farm Wildlife Package options under the new Countryside Stewardship scheme should be of particular benefit to wild pollinators, farmland birds and other species. Government is also working to implement the National Pollinator Strategy, including innovative work by Natural England to partner NGOs with landowners through the Landscape for Wild Pollinators Initiative.

Improving green space in urban areas (notably residential gardens) can provide food resources and suitable habitat for pollinators, with bee richness shown to be positively affected (Senapathi et al., 2015).

3.3 Soils and land use

3.3.1 Soils

Synthesis

Evidence shows a range of climate change-related risks are threatening the many services that soils provide, notably those that relate to soil biota, soil organic matter, and soil erosion and compaction. Soils are, however, exposed to multiple pressures and often highly heterogeneous, and as a consequence there are competing explanations for current impacts. The consensus view is that land use has been a more significant influence than recent climate change in determining impacts. Future climate projections suggest that climate, in combination with the other pressures, will lead to a major increase in risk due to the effects from higher temperatures, more intense rainfall and increased aridity, depending on location and soil type. Problems related to wetness are more likely to continue to occur in wetter areas in the north and west of the UK, whereas aridity is likely to become an increasing risk in drier areas of the south and east. These issues will be further modified based upon soil structure and the presence of organic matter and biological activity in different soils, in combination with land management on cultivated land or semi-natural habitats.

Unless management measures to maintain vegetation cover and good soil structure are further implemented, soil will become more susceptible to water erosion and potentially wind erosion. Due to higher temperatures and reduced soil moisture, soil is also likely to experience decreases in organic matter, with adverse consequences for crop production, soil biodiversity and carbon storage. In some locations, potential opportunities may arise for increased organic matter and biodiversity due to warmer temperatures and higher primary productivity if appropriate land management is implemented. In all circumstances, and particularly in areas of high risk, improved awareness and implementation of soil conservation measures will be required to address changes in risk.

The challenges of competing explanations for understanding changes in risk, particularly for soil organic matter and carbon stocks, identify a priority to further develop research and monitoring to improve knowledge of the interactions of climate change with different land-use systems and ecosystems on soil functions.

Context and policy

Soils are the foundation on which all terrestrial ecosystems function. They influence hydrology, vegetation composition and biogeochemical cycling and in so doing provide multiple functions that underpin a wide range of ecosystem services, such as food and fibre production, carbon sequestration and water regulation. Soils vary tremendously in their physical, chemical and biological properties; at any given location the type of soil that forms depends on a particular combination of soil-forming factors, including parent material, climate, topography and vegetation. The average rate of soil formation is about 0.1 mm a year, or one metre every ten thousand years (Montgomery, 2007). This slow rate of formation effectively means that soils are a non-renewable resource.

UK soils are relatively young in a global context, having mostly formed since the last glacial retreat some 10,000 to 15,000 years ago. Despite this, there is enormous diversity of soil types across the UK. This reflects the wide variation in soil-forming factors, but also the UK's long history of agriculture, forestry, mining and industry that has completely transformed the properties of many soils (UK NEA, 2011). Due to the maritime climate of the UK, with cool temperatures and relatively wet conditions, soils generally contain more organic matter and experience greater leaching (i.e. loss of nutrients and other soil constituents) than soils in eastern and southern Europe. There is a marked contrast between mineral soil types, which primarily form in the lowlands, and upland soils, which have surface horizons rich in organic matter and are generally more nutrient-poor and acidic, occurring in a wetter climate (UK NEA, 2011).

The fertility of soil depends on many factors, including its mineralogy and texture, the supply of organic matter and water and the diversity of organisms that live in the soil, which are responsible for the breakdown of organic matter and mineralisation of nutrients on which plant growth depends. Much of this soil biodiversity is microbial, with thousands of species of bacteria and fungi being found in a handful of soil; but soils also contain many different types of animals, including nematodes, microarthropods and enchytraeids, and larger organisms such as earthworms, ants and moles. The activities of these organisms, and their interactions with each other and plants, affect a range of ecosystem processes and services, including soil formation and structure, nutrient cycling, the production of food and fibre, climate regulation, and disease and pest control (Bardgett and van der Putten, 2014). The importance of soil biodiversity is increasingly recognised as providing benefits to human health through its role in the suppression of disease-causing soil organisms and provision of clean air, water and food (Wall et al., 2015).

The costs of soil degradation for England and Wales are estimated to be between £0.9 bn and £1.4 bn per year, with a central estimate of £1.2 bn (Graves et al, 2015). About 45 % of total quantified annual soil degradation costs are associated with loss of organic content of soils, 39% with compaction and 13% with erosion. It is estimated that 20% of the annual costs of soil degradation are associated with loss of provisioning services linked with agricultural production, both reduced output and increased costs. The remaining 80% of degradation costs are associated with loss of regulating services, the bulk of this (49% of all costs) linked to GHG emissions. Flood related costs (flood damage and flood risk management) account for about 19% of total costs and water quality related costs (both drinking water and freshwater water) account for about 11% of quantified costs (Graves et al, 2015). In addition to their implications for the natural environment, soil-related problems such as compaction and erosion, can also cause damage to cultural heritage features (e.g. archaeology).

UK soils are not protected by stand-alone legislation. Measures to promote or require the protection of UK soils occur indirectly through other legislation, although there are some measures to promote or require the protection of soils embedded in other legislation such as that for the control of pollution and management of contaminated land. The primary instrument for the protection of agricultural soils is the EU CAP, which requires that farmers meet the cross-compliance standards of Good Agricultural and Environmental Condition (GAEC) and also promotes soil conservation through agri-environment schemes. Non-regulatory mechanisms also promote sustainable soil management, through quality assurance schemes (e.g. Red Tractor or the Soil Association organic certification).

In England, a soils strategy, *Safeguarding our Soils – A Strategy for England*, was published in 2009. This set a policy aim that by 2030 all England's soils will be managed sustainably and degradation threats tackled successfully. The 2011 Natural Environment White Paper, *The Natural Choice*, confirmed the continuation of this policy aim and established a four-year research programme to explore how degradation can affect the soil's ability to support vital ecosystem services. The White Paper also committed to end peat use for horticulture by 2030. In 2012 the National Planning Policy Framework made clear that local authorities should not grant planning permission for peat extraction from new or extended sites.

The 2006 *Environment Strategy for Wales* includes a high-level outcome that 'soil is managed to safeguard its ability to support plants and animals, store carbon and provide other important ecosystem services'. The strategy proposes that changes in soil carbon will be monitored to assess progress in meeting this outcome. The main mechanisms for improving soil management referred to in the strategy are CAP and land-use planning, particularly for non-agricultural soils. The strategy is currently being reviewed in light of changes in policy since 2006, including the Environment Act (Wales) 2016 and the Welsh Government's Natural Resource Management programme. National land-use planning policy in Wales (*Planning Policy Wales*) states that local authority development plans should encourage land uses and land management practices that help to secure carbon sinks.

In Scotland, a *Soils Framework* was established in 2009 to co-ordinate actions to protect soils. It describes key pressures, particularly climate change, and relevant policies to combat those threats, and identifies the future focus for soil protection. The framework's vision is that soils are recognised as a vital part of the Scottish economy, environment and heritage to be safeguarded for existing and future generations. *Scottish Planning Policy* (2014) states that the land-use planning system should in principle seek to protect soils from damage such as erosion or compaction. Where peat and other carbon-rich soils are present, local authorities are required to ensure that the likely effects of development on CO_2 emissions are minimised as much as possible.

There is no stand-alone soil strategy or framework in Northern Ireland. However, the 2015 Strategic Planning Policy Statement for Northern Ireland stresses the importance of peatland to Northern Ireland for its biodiversity, water and carbon storage qualities.

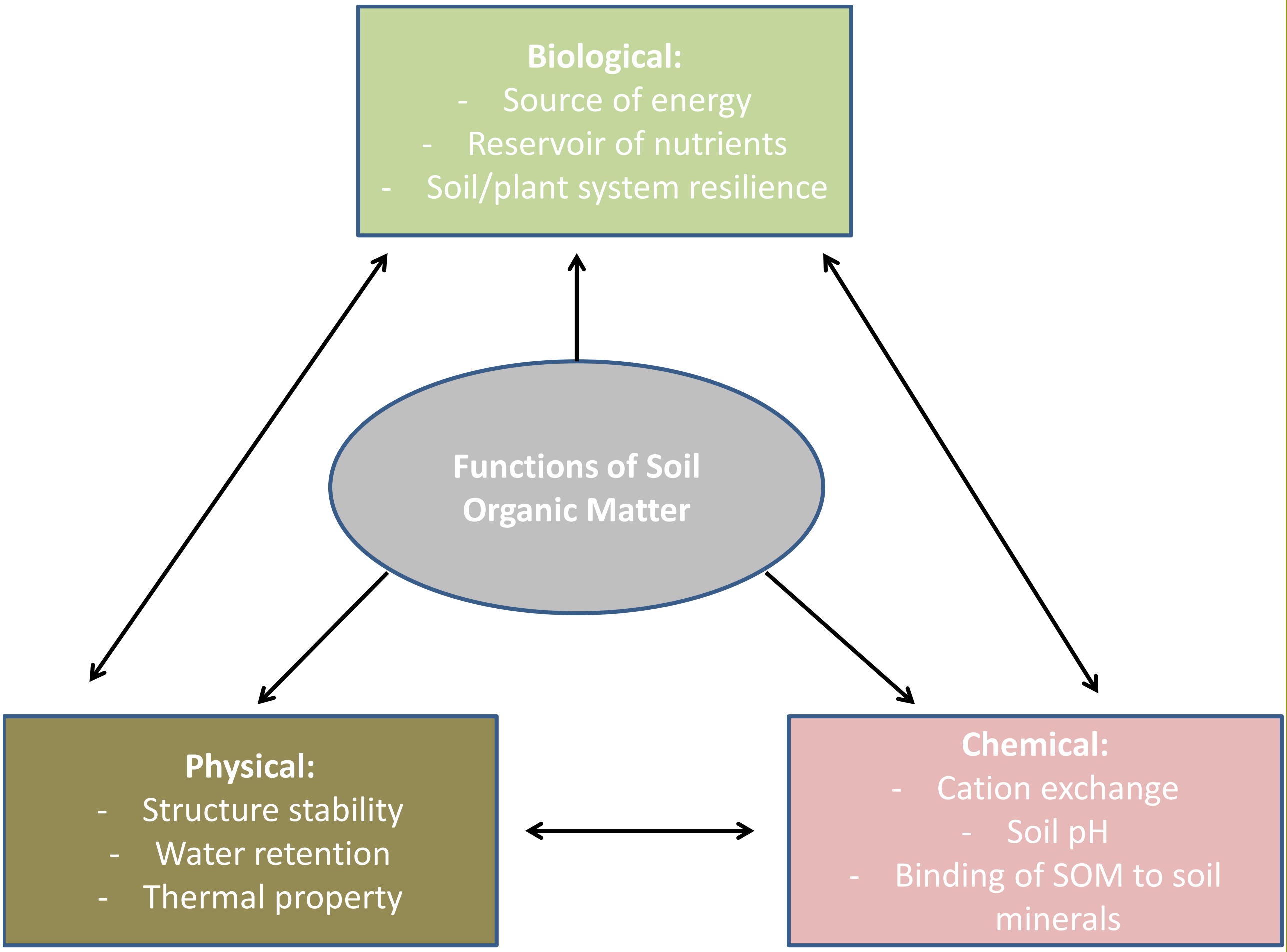
Current risks and opportunities

Soil chemical and biological processes are controlled by a complex set of factors, but most importantly by the balance between soil temperature and soil moisture. Temperature is a key factor that can control many terrestrial biogeochemical processes. Soils processes, properties and functions are therefore all sensitive to changes in climatic conditions.

Soil organic matter

A range of biological, physical and chemical processes convert dead plant material into soil organic matter (SOM) (Lehmann and Kleber, 2015), which plays a fundamental role in the functioning of soil (Figure 3.5). SOM retains both nutrients and water, thereby improving plant growth, and contributes to the formation of soil structure. Further, because SOM contains more carbon than vegetation and the atmosphere combined, it plays a crucial role in the global carbon cycle. Given this, changes in SOM can have important long-term implications for soil fertility and climate mitigation (Section 3.2.9). In general, decomposition of organic matter is retarded in cool, wet areas or where drainage is impeded; hence, the organic matter content of soils tends to be greater and deeper in such areas, reaching a maximum depth in peat soils.

Figure 3.5. Mulitiple functions provided by soil organic matter



Source: CCRA Evidence Report authors.

The impact to date of climate change on soil organic carbon (SOC) in UK soils is still a matter of debate due to differences in analytical methods and their interpretation. [Low confidence] Changes in SOC take place over a long period of time and are governed by many factors such as climate (temperature and precipitation) and edaphic factors (such as soil parent material and clay content). Chapman et al. (2013) reported no change in overall total soil carbon stock at 100 cm depth for a 25-year period across Scotland, although there were increases in SOC for soils under woodland and significant decreases in soils under improved grassland and arable cultivation. Another study by Reynolds et al. (2013) covering a 30-year period (1978 – 2007) reported no change in SOC across Great Britain for the depth of 15 cm. Further, Hopkins et al. (2009) found no consistently significant changes in SOC stocks due to climate change over a long-term (>100 years) experiment on grassland soils.

Data from the National Soil Inventory (NSI) revealed historical (1978 – 2003) losses of carbon from soils across England and Wales at a rate of 0.6% per year, which occurred irrespective of land use, and this was taken to suggest a link to climate change (Bellamy et al., 2005). However, Smith et al. (2007) inferred that land use change was the dominant influence in these data and climate change was only responsible for 10 – 20% of the SOC loss reported by Bellamy et al. (2005).

There is good evidence for the relationship between land use and SOC. Both the Countryside Survey 2007 and the NSI suggest that a loss of SOC has occurred on carbon-rich soils used for intensive arable cultivation (notably fenland peats), although this may be attributable to a change in equilibrium conditions related to tillage rather than climate change (Kirk and Bellamy, 2010). Similarly, a national study of English grasslands revealed that SOC concentrations were lowest in intensively managed (improved) grassland (Ward et al., 2016). Using isotope experiments, Evans et al. (2007) also reported negative impacts of agricultural intensification in the UK on SOC.

More recently, Barraclough et al. (2015) re-analysed the NSI data for England and Wales and detected the same decline in SOC as reported by Bellamy et al. (2005). Using space-for-time substitution methods, this re-analysis suggested that very little (0 – 5%) of the decline could be attributed to climate parameters (temperature and precipitation) for mineral and organo-mineral soils. However, 9 – 22% of SOC loss on semi-natural organic soils, such as bog and heathland, was tentatively attributed to increasing temperatures indirectly stimulating SOC loss through changes in vegetation and litter quality as grass species increase at the expense of shrubs and mosses. From this, Barraclough et al. (2015) postulate that bog and heathland habitats become transitional to grassland wherever mean annual temperatures exceed a threshold value of 7°C.

In summary, the current consensus is that UK SOM and SOC contents are at present more sensitive to land use than climate change in mineral soils, but that for organic soils climate change plays a greater role. Differences in detection of SOC trends between NSI data and CS2007 data in England and Wales are believed to be due to the land-use and habitat classes used and associated statistical power of detection. Future analysis would therefore benefit from standardised sampling protocols, including sampling of SOC to depth, and greater consideration of the role of temperature on organic soils of semi-natural habitats and the loss of topsoil SOC from carbon-rich horizons under intensive arable cultivation. Detection of climate change impacts is likely to be masked by other factors, most notably recovery from acidification (Monteith et al.,2012) as noted by the CCRA1 and evident in water quality data through dissolved organic carbon (DOC) levels (section 3.4).

Land management can potentially exacerbate the sensitivity of soils to loss of organic material from a changing climate, increasing the vulnerability of soils of some locations to drought and loss of key functions such as productivity, particularly in eastern England. Furthermore, shifting patterns of land use resulting from changes in climate could also affect SOC (see Section 3.2.4).

Soil nitrogen mineralisation, by which organic matter is converted to inorganic N forms, is temperature and rainfall dependent (Rustad et al., 2001). In a study covering 665 locations across Britain, Rowe et al. (2012) found that mineralisation increased with increasing mean temperatures. This has implications for N pollution, in the form of nitrates and nitrous oxide (N_2O) emissions. In saturated soils, lack of oxygen limits mineralisation. Organic-rich soils (histosols) that have been drained for agricultural production have higher mineralisation rates than mineral soils, and consequently higher N_2O emissions.

Soil ecosystems and biodiversity

Climate change can significantly affect the abundance, diversity and activity of soil biota, although effects are highly variable and depend strongly on environmental context (De Vries and Bardgett, 2015). In general, however, both elevated atmospheric CO_2 and warming are likely to increase the abundance of bacteria and fungi in soil, and of most faunal groups, thereby causing changes in the structure of the soil food web. [Medium confidence] A review by

Blankinship et al. (2011) reported that microbial biomass is significantly increased by elevated CO_2, but bacterial abundance is negatively affected by warming. Drought conditions typically decrease the biomass and abundance of most microbial and soil faunal groups and change microbial community structure, increasing the abundance of fungi relative to bacteria. By contrast, little is known about the effects of increased rainfall and flooding on soil biota, and reported effects are highly variable (De Vries and Bardgett, 2015). The effects of atmospheric nitrogen deposition also have an influence: measurements of mineralisable soil nitrogen at 665 locations in the UK suggest that increasing temperatures are likely to amplify effects of nitrogen pollution on semi-natural ecosystems (Rowe et al., 2012), potentially accelerating reductions in soil biodiversity due to declines in species that persist at low N levels (Section 3.2.1).

Changes in land use and management intensity can have negative impacts on soil biota and so increase their vulnerability to climate change. Many studies report soil biodiversity declines as a result of the conversion of land to agriculture and from agricultural intensification (Bardgett, 2005). A recent study of multiple sites across Europe, including the UK, revealed that intensive agriculture reduces soil biodiversity, making soil food webs less diverse with fewer functional groups (Tsiafouli et al., 2015). In all regions of Europe surveyed, species richness of earthworms, collembolans and oribatid mites was negatively affected by increases in land-use intensity.

Soil erosion and compaction

Erosion is a natural process, and occurs both by water and wind. Estimates for England and Wales suggest a loss of around 2.2 million tonnes of topsoil every year (Harrod, 1999) and 17% of arable land shows signs of water-driven erosion (Environment Agency, 2004). Rates of erosion can increase as a result of intensive cultivation, deforestation and overgrazing; in cultivated fields erosion rates can sometimes reach 10 mm/yr (Montgomery, 2007). Steeper slopes are more vulnerable to erosion by water, while fine sandy soils and organic soils can be vulnerable to wind erosion. Information on rates of soil loss by erosion in the UK is limited, but estimates range from 0.02 to 1.27 t/ha/yr for mineral soils, to as much as 10 t/ha/yr in cultivated arable fields (Verheijen et al., 2009). Higher rates of water erosion occur when bare soils are exposed to intense periods of rainfall and gullying due to overland flow, on either arable soils during the sowing season or upland soils that have lost their vegetative cover (notably peatlands). In Scotland, it has been estimated that around 35% of peatlands show signs of erosion, which has been attributed to both land use and climate change (notably increased rainfall in west Scotland) (Lilly et al., 2010), and in England the majority of upland peat has been described as being in a degraded condition (ASC, 2015). It is estimated that around one-third of cropland soils in England are at moderate to very high risk of erosion (Evans, 1990; Knox et al., 2015).

Although there is evidence that more rainfall is falling in heavy events (Watts and Anderson, 2015), evidence that erosion rates are increasing remains inconclusive. [Low confidence] Different types of land use and maangement can cause localised effects, and the climatic influence is most apparent during extreme events, which by their nature are sporadic and difficult to attribute to a long-term trend. In addition to the effects of local soil losses on plant productivity, the primary impact of water erosion is the transfer of soluble and insoluble material, including artificial fertilisers, to streams, which can cause severe problems for water quality, particularly where the buffering role of vegetation and soil has been reduced in catchments (see Section 3.2.5).

Compaction occurs due to the use of machinery or presence of livestock on vulnerable soils, particularly when waterlogged, resulting in damage to soil structure, reduced aeration and penetration of plant roots, and the potential for increased erosion due to reduced water infiltration and increased runoff from overland flow. While a number of small-scale studies have found locally occurring increases in soil compaction, there has been no systematic study of the national extent, or severity of, this issue; as a result, it is not currently possible to provide a quantitative assessment of the current state or trend across the UK and the role of climate change is not fully established. [Low confidence] The constraints of soil wetness for land use as arable or improved grassland have apparently increased in some western districts of the UK due to increased rainfall in recent years (see Section 3.2.4), but compaction may also be associated with use of larger machinery when soils are wet and vulnerable to damage. Good practice modifications, such as low pressure tyres, may be used to address trafficability problems. In England, across a range of soil types it has been reported that 30% of soils are in a good condition in relation to soil compaction, with 50% moderate and 20% poor (Defra project AC0114: Anthony et al., 2012). Similarly, a survey of 300 grassland fields in England and Wales found 10-16% of fields had problems with compaction and were in poor condition, including extensive grazing areas in addition to improved grasslands (Newell-Price et al., 2012).

Future risks and opportunities

Future changes in temperature and precipitation could potentially have considerable impacts on soils and their biodiversity. Rising atmospheric concentrations of CO_2, are also likely to influence soils indirectly, via changes in plant growth. There is a high degree of uncertainty about how climate change will affect soils in the UK due to limitations on the current evidence and the difficulties of distinguishing the role of climate from other factors. Nevertheless, the majority of climate projections imply a trend towards reductions in soil moisture, most notably in the eastern districts of the UK, due to an increased frequency of warmer, drier summers. The consequent changes in soil water regimes will be highly dependent on soil type and, in combination with elevated temperatures and CO_2 levels, will have an impact on rates of soil physical, biological and chemical processes, and hence on soil function and ecosystem services.

Soil organic matter

Projected future changes in SOM and SOC remain highly uncertain. The loss of SOM due to climate change would affect the stability of soil structure, topsoil water-holding capacity, nutrient availability, soil erosion and land use. Increases in atmospheric CO_2 concentrations and temperatures would be expected to enhance primary productivity (so increasing SOC), but may also enhance soil microbial activity and decomposition rates (reducing SOC stocks). Interactions with land use and nitrogen deposition will also be key influences.

Model-based simulations by Cooper et al. (2010) using the ECOSSE soil carbon model suggest that climate change would only cause relatively small changes in SOC levels for England and Wales to 2080 (medium emissions), but this analysis did not account for land-use change. Experimental evidence has shown highly variable responses of SOC to climate drivers. For example, in some studies warming enhances carbon loss from peatlands due to accelerated decomposition (Dorrepaal et al., 2009). However, warming has been shown to have little effect on soil carbon pools in grasslands despite large changes in vegetation and microbial community composition (Zhou et al., 2012). Furthermore, while warming can result in SOC loss in the short term, this effect can diminish with time as soil microbes adapt to warmer temperatures and SOC loss is compensated for by increased plant growth (Melillo et al., 2011). A meta-analysis of field studies across a broad range of ecosystems revealed that elevated CO_2 typically increases root growth and stimulates the discharge of easily degradable sugars, organic acids and amino acids from roots (Nie et al., 2013). This stimulates microbial activity and SOM mineralisation and hence carbon loss from soil (van Groenigen et al., 2014). A UK-based study showed that DOC release is more sensitive to atmospheric CO_2 concentrations than to warming or hydrological changes associated with drought (Freeman et al., 2004).

Higher mineralisation and denitrification due to increased temperatures are expected to result in higher nitrous oxide (N_2O) emissions from soils (Abdalla et al., 2010). Higher quantity and frequency of rainfall in late autumn and winter would also increase denitrification and thereby N_2O emissions (Dobbie et al., 1999). Changes in rainfall amount and intensity, along with increased temperatures, are also likely to influence leaching and mineral weathering rates. This in turn could have implications for soil pH. [Low confidence]

Soil ecosystems and biodiversity

While much uncertainty surrounds the impacts of climate change on soil biota and ecosystems (De Vries and Bardgett, 2015) , any changes in the abundance of soil organisms and the structure of the soil food web will significantly influence soil functioning (Bardgett and van der Putten, 2014) and human health (Wall et al., 2015). Changes in climate are expected to affect the abundance and activity of soil microflora (e.g. bacteria, fungi and protozoans), with implications for decomposition of organic matter and hence carbon storage, nutrient cycling and soil fertility (Turbé et al., 2010). A UK-based study showed that reductions in soil food web complexity and diversity due to intensive land use impaired the ability of soil biota to recover from drought, leading to increased losses of soil carbon and nitrogen (De Vries et al., 2012). Similarly, in a European-wide study, changes in soil food web structure were strongly related to C and N cycling (De Vries et al., 2013). Also, Handa et al. (2014) found across a broad range of ecosystems that the loss of key components of the decomposer community consistently reduced litter decomposition, and Wagg et al. (2014) found that experimental reductions in soil biodiversity reduced ecosystem multifunctionality. These studies all suggest that changes in the diversity and composition of soil food webs due to climate change could have implications for soil functioning, although the exact nature of these responses remains uncertain. [Low confidence]

Changes in the abundance of individual groups of soil organisms are also likely to affect specific soil processes. For instance, increases in the abundance of enchytraeid worms due to climate warming have been shown to increase rates of organic matter mineralisation in peatland (Cole et al., 2002; Briones et al., 2007). Elevated atmospheric CO_2 increases rates of SOM decomposition due to increased activity and abundance of many trophic groups in the soil food web (van Groenigen et al., 2014). Increases in the biomass and activity of soil microbes due to warming can result in greater respiration and rates of N mineralisation (Dorrepaal et al., 2009; Zhou et al., 2012), although effects on SOC are uncertain.

Soil erosion and compaction

Climate change has the potential to affect soil erosion via a variety of routes. Projected increases in the frequency and intensity of erosive rainfall events will have direct impacts on erosion rates (Pruski and Nearing, 2002). Simulation models predict that a 10% increase in winter rainfall in the UK could increase soil erosion by 150% in wet years, although the long-term average is not expected to increase significantly as erosion would be more similar to now for other years (Favis-Mortlake and Boardman, 1995). For England and Wales, Cooper et al. (2010) projected water erosion rates to increase by an average of 0.1 t/ha/yr by the 2080s, while McHugh (2007) inferred significantly higher increases (by an average of 0.6 t/ha/yr) over the same period. For many areas, climate models suggest more intense seasonal drying in summer could be followed by increased rainfall rates in autumn/winter, providing conditions that could lead to a large increase in rates of erosion by water. [Low confidence]Temperature and CO_2-driven changes in plant biomass could also increase erosion rates, due to faster decomposition from increased soil microbial activity (i.e. reducing organic matter content and binding of particles) (Nearing et al., 2005). [Low confidence] Increases are predicted to be greatest overall in upland areas, although with hotspots for specific pollutants, notably for phosphorous loads in intensive agricultural areas where manure applications are high (typically associated with extensive pig and poultry farming) (Cooper et al., 2010). As future climate projections for wind remain very uncertain then no significant changes in wind erosion rates can be inferred with any confidence but it is very likely to remain an important issue in its current problem areas.

It is also likely that shifts in land use as a result of climate change, including the adoption and diffusion of new crops, could alter future rates of erosion (Nearing et al., 2005). The complexities of land-use decisions under climate change mean there is a high degree of uncertainty about future rates of soil erosion. Modelling at six sites in Northern Ireland using different combinations of climate and land-use scenarios showed the largest increase in soil erosion occurs with a shift towards arable cropping, especially for maize (Mullan, 2013). This indicates that the indirect impacts of climate change on land use are likely to be more influential than climate change alone.

Modelling of upland peat soils by Li et al. (2016) using a range of UKCP09 projections shows an increased risk from desiccation and erosion due to a shift to a warmer, drier climate. The risk is highest for locations where blanket peat is present in the south west and eastern side of the UK, which already experience a warmer, drier climate. As noted in Section 3.2, peatland in poor condition (affected by erosion, drainage, burning etc.) is more vulnerable to climate change and more likely to degrade further and more rapidly, leading in particular to increased losses of soil organic carbon and reduced potential for adaptation/mitigation through restoration.

With regard to soil compaction, climate change may reduce the period of time when soils are at high risk due to being in a waterlogged condition. [Low confidence] Modelling by Cooper et al. (2010) suggests that a reduction in the number of days when soils are at field capacity in England and Wales means that soils will be slightly less vulnerable (valid across multiple UKCP09 projections). Similar work in Scotland likewise suggests a general trend for a reduced number of days in eastern areas when soils are waterlogged and vulnerable (Brown et al., 2008). However, in some locations, such as south-westand north-west England, Wales and most of west Scotland and Northern Ireland, localised risk of soil compaction will remain into the future because of the wetter climate and presence of poor-draining soils, which will continue to limit access to land, and constrain stocking levels during the winter and early spring (Cooper et al., 2010).

Adaptation

No-till and reduced tillage in arable systems can have benefits for maintaining SOM and SOC and the retention of nutrients and water. However, while no-till usually increases SOC in surface soils, it can cause reductions in SOC at depth and, in some situations, increase N_2O emissions (Six et al., 2004). Around 40% of cultivated land under discs and tines use reduced tillage systems, and 4% use a no-till or direct-drill system (Defra Farm Practices Survey and GHG indicators[11]). Reduced tillage and no-till systems are best suited to medium and heavy soils (Soane et al., 2012), although in recent years reduced tillage has been increasingly adopted on lighter soils. Approximately 60% of farms apply organic manures to at least one field on

[11] https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/448954/ghgindicator-2mitigation-29jul15.pdf

the farm (British Survey of Fertiliser Practice, 2013) which can also maintain levels of SOM when applied regularly.

Installation and renewal of existing underdrainage systems peaked in the 1970s and 1980s and declined rapidly until recently. Wet weather in 2011 and 2012 exacerbated soil drainage issues on many UK farms and gave rise to an increasing interest in the maintenance of field drains. Defra Farm Practices Survey data indicates that the proportion of farms 'improving drainage' to reduce soil compaction increased from 48% in 2008 to 62% in 2012.

An estimated one-third of upland peat in England is under some form of restoration (ASC, 2013), while in Scotland restoration work has begun on 5,580 hectares across 100 sites, with plans for a further 3,000 hectares to follow.

The risk of losses of SOM and soil biodiversity, and increases in soil erosion, highlight the key role of good soil management practices to enhance long-term soil resilience. Reactive measures will tend to be too late to avert long-term damage. The soils of the UK are highly diverse and considerable variations in soil properties can occur even at local levels, meaning that measures need to be customised to local conditions. National agencies provide advice on good practice (e.g. the Environment Agency's *Think Soils* guidance), which could be further developed to incorporate resilience to climate change. Increasing the availability of low-cost soil testing for farmers and land managers would be beneficial. There is also a need to enhance strategic monitoring of sensitive soils (e.g. carbon-rich organic or organo-mineral soils). As previously identified by the ASC, there remain significant knowledge gaps in the effectiveness and uptake of different soil conservation measures (ASC, 2013). This could be addressed through a systematic evaluation of conservation measures that account for a range of the local conditions.

3.3.2 Land availability and capability

Synthesis

There is good evidence that the biophysical capability of the land to support its many functions has changed over recent decades as the climate has changed. This is exemplified by changes in land classification, which has conventionally been used to indicate land quality for agricultural production. Future projections indicate much larger changes can be expected based upon shifts in temperature and precipitation patterns. Changes in the growing season in combination with wetness or drought-related factors imply that options and flexibility in land-use will be strongly affected, as happens at present mainly following extreme years (good or bad). Potential opportunities are particularly recognised in the north of the UK due to warmer, drier summers, but increasing constraints are expected in the south and east due to limits on water availability. Wetter winters could increase problems with waterlogged soils, especially if occurring in conjunction with wetter spring or autumn conditions.

More land is expected to be at risk from flooding in the future, which is likely to also challenge some existing land uses. Over 1 million hectares of high-grade agricultural land is projected to be at high risk (1 in 75 years or greater) by the 2050s under a 4ºC climate scenario. Agricultural land in the coastal zone is also susceptible to a shifting coastline.

Current land-use planning is mainly based on a static interpretation of land availability and capability rather than taking a strategic longer-term approach. As a consequence, land-use decisions may not be matched well with expected future conditions. Land-use planning needs to be forward-looking to plan for and maximise best use of available land resources, acknowledging that both supply of and demand for land-based services will change in space and time. A research priority is to assess the implications of existing strategies for drainage and irrigation on the availability of land for different uses, and alternative management actions for existing land uses (e.g. to improve water efficiency). This would support further development of good management practice and the regulatory framework. Land-use planning has a strategic role in co-ordinating actions across multiple sectors to manage risks, but incorporation of adaptation objectives has to date been limited, partly due to lack of accessible data and tools. In some locations, planned change of land use to reduce risks over a larger area is very likely to be beneficial, but this requires considerable lead time to facilitate a smooth transition from existing uses.

Context and policy

The capacity and quality of the land resource influences the services it can provide and the ability to sustain the demand for those services through time. As land is a finite resource there is a need to ensure that its current use does not degrade its future potential.

The land area of the UK is close to 25 million hectares. The dominant land use is agriculture, which occupies about three-quarters of the UK land area. According to the UK National Ecosystem Assessment (2011) enclosed farmland makes up 42% of land cover, comprosing both arable and intensive grasslands. Semi-natural grasslands make up 16% and mountain, moors and heaths a further 19% of land. Woodland accounts for 12% and freshwater and coastal habitats together make up 3%. Finally, urban and associated developed areas cover 7% of land. The area of developed land has been steadily increasing (by 0.6% in total between 2000 and 2010), as has the area of broadleaved woodland.

Rural land-use planning in the UK is primarily based on land capability classification. The Agricultural Land Classification (ALC) system is used in England, Wales and Northern Ireland , whilst the Land Capability for Agriculture (LCA) is used in Scotland.[12] Classification is based on bioclimatic metrics, soil properties (texture, structure, stoniness etc.) and soil-climate interactions (wetness, drought risk, erosion risk). For ALC, these factors are used to classify the land into five main grades, Grade 1 being excellent quality and Grade 5 being very poor quality, whilst in Scotland seven main classes are recognised with subdivisions to cover the wide variations in land quality. A large proportion of the Grade 1 land in England is on peat soils that have been drained with consequent high carbon emissions (section 3.7.1).

In all four UK nations, planning policy aims to steer urban development away from those areas of land that have the greatest agricultural potential.[13] Land resources in some areas of the UK are under high pressure for development as good-quality land often occurs around large urban centres that continue to expand, notably in south-east England. An estimated 6,600 hectares of Grades 1 and 2 agricultural land, accounting for a little under 1% of the total area, was developed between 2001 and 2011 in England (ASC, 2013).

There is increasing recognition that land classification needs to incorporate its multifunctional capability. As the best-quality land is prioritised for intensive agriculture, other demands for land are implicitly forced to compete for space on land of lesser quality. As well as land for development, these other demands include land for energy (particularly renewable energy and biomass), woodland expansion, flood alleviation, water quality, nature conservation and landscape amenity. In practice, as most upland areas are considered important for nature conservation, tourism and cultural benefits, and are contrained by topography from major developments, then other demands for land are often effectively competing for a 'squeezed middle' between the uplands and the highest-grade agricultural land (Slee et al., 2014).

Scotland is the only UK nation with a statutory Land Use Strategy (Scottish Government, 2016). This has established a series of principles and proposals for sustainable land use that can account for the changing suitability of land for agricultural and forestry production. In Northern Ireland, an initial proposal for the development of a land-use strategy has recently been prepared by Northern Ireland Environment Link.

There are a wide range of sectoral policies that influence land-use decisions. These include the CAP (Pillars I and II), national biodiversity strategies (particularly plans for ecological networks), climate change mitigation policies, the EU Water Framework Directive, the EU Floods Directive and the European Landscape Convention. Land-use issues are particularly relevant to aspirations for 'sustainable intensification', which aim to deliver productivity increases while enhancing long-term environmental, social and economic benefits. A major objective of the current round of CAP Reform (2014 – 2020) is to balance food security with environmental protection.

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[12] Although ALC and LCA are different classifications, they have many similarities due to their common origin. Bioclimatic metrics provide measures of the length or intensity of the growing season and general wetness constraints; these metrics are combined with intrinsic soil and topographic properties and soil-climate metrics that define land-use constraints due to soil wetness factors and droughtiness. These metrics are defined using a long-term average, usually over a 20- or 30-year period; in addition, the LCA system includes seasonal influences via soil moisture deficits, whereas ALC does not, except for identifying the period when land is at field capacity. The final grade or class of the land is provided by the lowest constraint. Similar use of bioclimatic metrics is employed by the Forest Research Ecological Site Classification (ESC) to identify geographic limits on tree species suitability in the forestry sector.

[13] Defined by grades 1, 2, and 3a to give 'best and most versatile land' (BMV) in the ALC system and Classes 1, 2, 3.1 to define 'prime agricultural land' (PAL) in the LCA system.

Current risks and opportunities

As climate is a key component of land capability, climate change presents both risks and opportunities, depending on location. A basic premise of the ALC and the LCA is that a warmer and drier climate equates to a better grade or class of land. However, other climate factors, such as seasonal soil wetness and droughtiness, may constrain land-use options. Changes in land-use may also have secondary impacts on biodiversity, soils, water and ecosystem functions and therefore affect the balance of services that the land provides.

As identified in Section 3.3.1, soils with high water retention properties, impeded drainage or proximity to a groundwater table impose restrictions on land-use due to waterlogging. Ignoring these constraints may cause problems with soil compaction and erosion, although the problems may be partially alleviated by artificial drainage systems. Conversely, soils with coarse texture and low available water capacity are particularly vulnerable to drought risk in locations where the climate is drier. Drought-risk may at least be partially alleviated by irrigation.

Depending on location, the average length of the annual growing season has increased by about 15-35 days and its strength in the most recent decade, measured in annual growing degree days for grass, is 16 % higher than 1961-1990 and 6% higher than 1981-2010 (Kendon et al., 2015). [High Confidence] In some locations a longer growing season and milder winters have provided opportunities for a shift to autumn-sown crops. There is evidence of a shift to a drier bioclimatic regime in some locations since the 1960s (as indicated by soil moisture deficits), notably in eastern districts of the UK (Keay et al., 2014a; Brown and Castellazzi, 2015). This trend is complicated by patterns of interdecadal variability (which means detection of a trend is dependent on the chosen baseline). Wetness constraints continue to limit land use in western parts of the UK. The overall area of land severely constrained by wetness issues may have slightly reduced since the 1960s, but with an increased risk in some locations such as south-west Scotland due to wetter winters. [Medium Confidence]

In England and Wales, small overall changes in the areas of different ALC grades in recent decades have been identified; a general improvement in land quality due to warmer conditions has been countered, at least to some extent, by an increase in drought risk (Keay et al., 2014b). The same analysis suggests that the area of BMV (best and most versatile) land may have actually declined since the cooler period of the 1950s to 1980s as more land has become constrained by drought risk than has benefited from the warmer temperatures. In Scotland, an expansion in the area defined as prime agricultural land has occurred in the east since the 1960s due to a shift in average conditions towards warmer, drier summers, with only a very small area presently constrained due to droughtiness (Brown et al., 2008, 2011). In locations where risks from soil wetness have been alleviated by underdrainage, a major uncertainty is the current performance of the drainage systems. As grants for drainage installation were phased out in the late 1980s, some systems may now be exceeding their original design lifespan (Anthony et al., 2012).

There are indications that annual variability in land quality may have increased for some locations over recent decades, particularly in western districts (e.g. south-west Scotland) [Low confidence]; this is consistent with anecdotal reports from land managers who have noted increasing difficulties in anticipating weather conditions from year to year (Brown and Castellazzi, 2015).

It is currently difficult to attribute actual land-use changes to climate-related changes in land capability because of the multiple factors involved in land-use decisions. In addition, monitoring land-use change through time has been complicated by changes in UK land-use and land-cover classification systems. It has been suggested that the trend towards increased agricultural intensification and specialisation may in some locations be increasing risk by encouraging land uses that exceed biophysical land capability, therefore reducing climate resilience (Brown, 2013). The consequences of this may become apparent during years with poor weather; for example, the wet conditions in 2012 resulted in a 14% reduction in UK wheat yields. Seasonal weather variations from year to year are a key feature of the UK climate, and a degree of autonomous adaptation to this climate variability is a part of the annual production cycle. For example, the wet summer and autumn of 2012 necessitated a shift towards spring-sown crops, fallow or grassland, rather than autumn-sown crops, in many locations (before 2012 the trend was for spring-sown to autumn-sown crops). The recent expansion in maize cropping (see Section 3.2.7) is partly attributable to milder spring conditions in addition to economic drivers, but maize can be a risky crop as it is usually harvested late in the year and thus at risk from wetter autumn conditions. Maize cropping can also be associated with soil compaction and erosion, which can lead to water pollution and loss of soil nutrients, especially when planted on steeper slopes.

Future risks and opportunities

The projected future trend towards a warmer climate with drier summers (on average) for most of the UK has important implications for land capability. By the 2050s, climate projections indicate that the intensity of the reference growing season[14] is projected to increase by 100 – 400 degree-days, with the largest changes in the south (Brown et al., 2008; Keay et al., 2014a). Many areas in the north and west of the UK that are currently marginal for cultivation would therefore experience an improved land capability, assuming that the main limitation is currently climatic rather than soil or topographic constraints (Brown et al., 2008).[15][Medium Confidence]

However, areas that currently have droughtiness constraints or are close to becoming droughty, notably in the south and east, are likely to become more limited by water availability in the future due to increased soil moisture deficits. The location of the most productive areas of land is therefore likely to shift as the climate changes. [Medium Confidence] Water availability constraints will either require additional irrigation and water storage, or adoption of new management practices (e.g. cultivation of drought-resistant crop varieties) or a change in land use.

In England and Wales, the most striking feature of future ALC projections is the downgrading of land quality due to droughtiness. The proportion of sample sites that occur on BMV land decreases from 38% against the 1961 – 1990 baseline to 9% under a high-emissions scenario for 2050. For the same period and scenario, the proportion of sample sites categorised as Grade 4 (poor quality) agricultural land increases from 14% to 48%. By the 2080s for this scenario the proportion of sites that would be Grade 4 increases to 70% (Keay et al., 2014a; Figure 3.6). The use of a number of alternate drought indices to that employed for ALC also suggest that there would be a significant increase in drought risk in the South and East of England, although risk will vary in relation to the specific indicator crop (Keay et al, 2014a).

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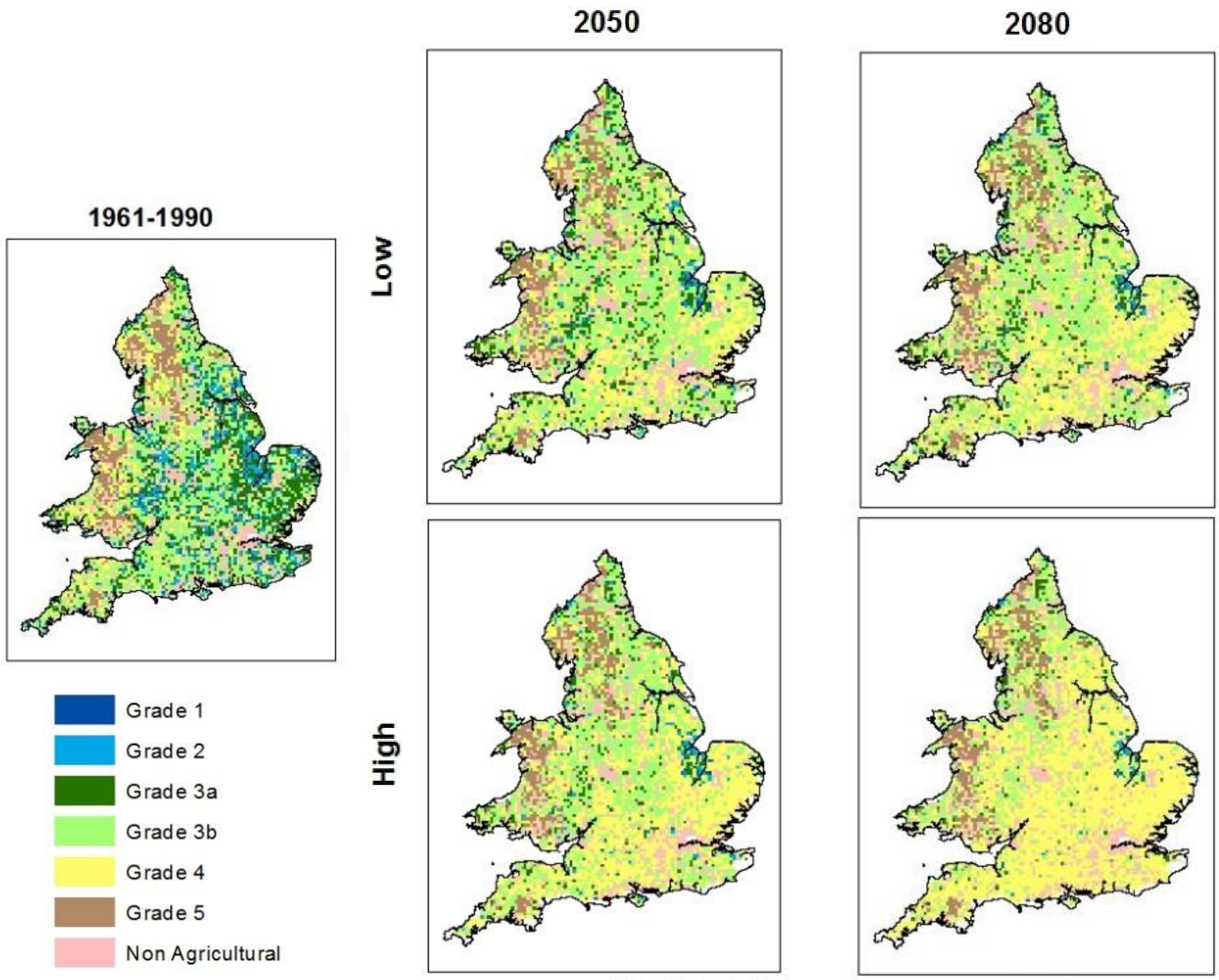
[14] The accumulated total of daily mean temperatures above a threshold value (using 5.5°C for grass) over the whole year.

[15] An important caveat is that this assessment assumes no change in intrinsic soil properties, such as may occur due to changing organic matter content (see Section 3.3.1).

Figure 3.6. Projected changes in Agricultural Land Classification for England and Wales based on low and high UKCP09 emissions scenarios for 2050s and 2080s.

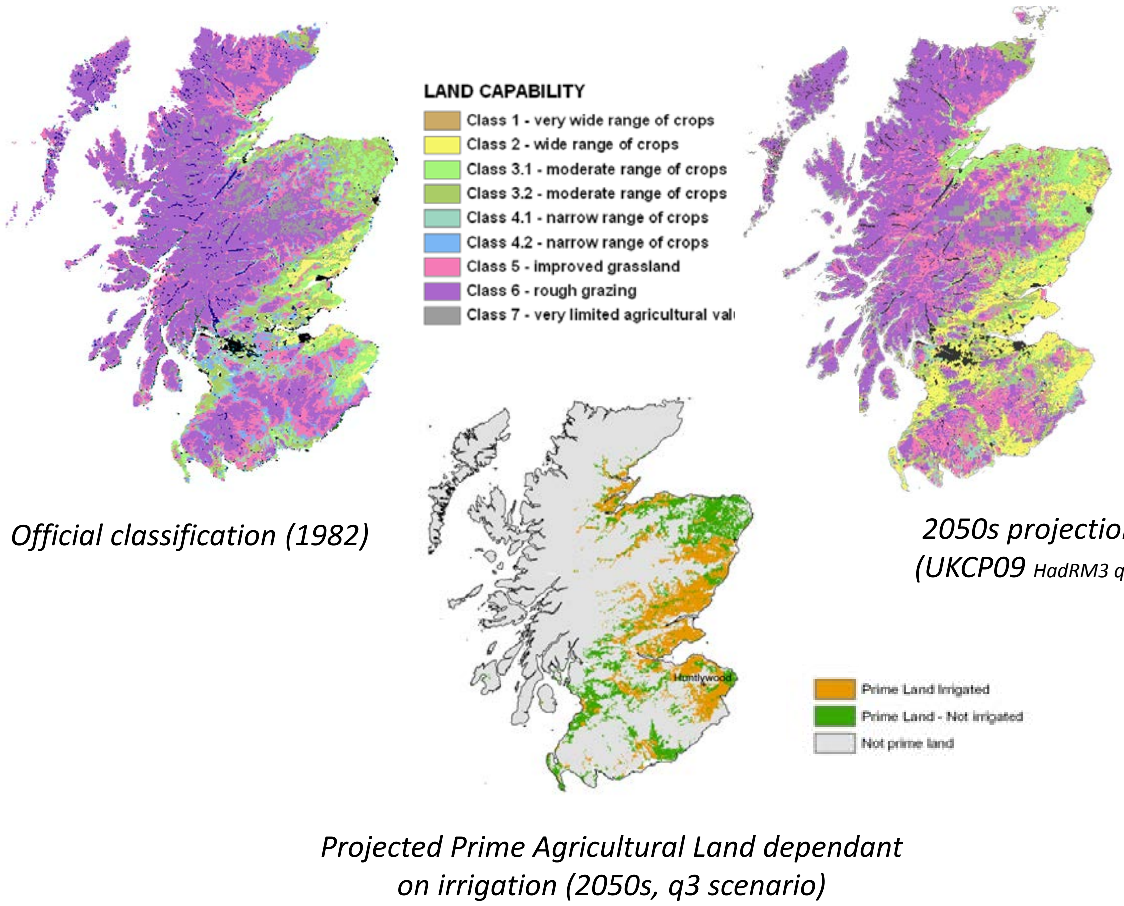


Source: From Keay et al. (2014a).
Notes: Projected ALC Grade (most limited of all 10 criteria) of the NSI sites with droughtiness using new MORECS regression and adjusted potato classification under different climatic scenarios.

Extrapolation of current trends in water demand shows that in a dry year in the 2020s the supply–demand gap could be 45 – 115 billion litres a year in England, with the upper-end estimate as large as the current annual demand. Although there is some evidence that demand has been declining in recent years (ASC, 2013), the implications for the cost of food production in these areas could be substantial.

In Scotland there is the potential for prime agricultural land to expand by up to 40% by the 2050s (from a 1981- 2000 baseline). However, 40 – 60% of this prime land may require irrigation to retain 'prime' status (based upon HadRM3 model: q3 and q16 runs) (Brown et al., 2011; Figure 3.7), with an additional irrigation demand of up to 50 Ml/km^2/yr estimated for this land. [Medium confidence]

Figure 3.7. Projected changes to Prime Agricultural Land for Scotland for 2050s and projected changes to area of PAL dependent on irrigation in 2050s.

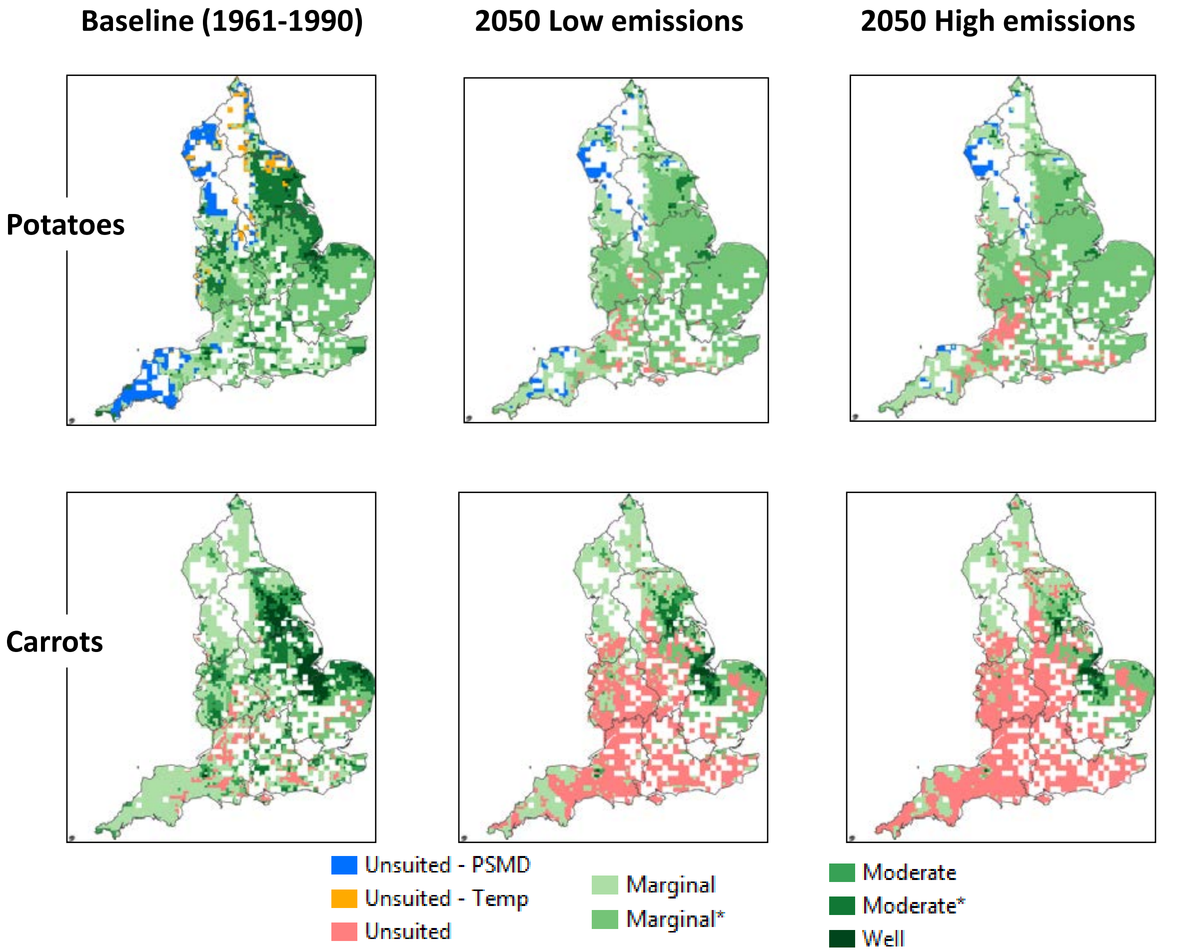


Source: Brown et al. (2011).
Notes: Figure shows recent and projected future changes in prime land based upon LCA classes (HadRM3 q3 medium emissions). Future changes are dependent on assumptions made regarding drought risk for indicator crops (potatoes, wheat) and whether additional irrigation would be available to counter increased soil moisture deficits.

Future climate changes imply changing suitability for specific land uses, notably for crops such as potatoes and cereals, based upon their current distribution (Figure 3.8). Current land uses that are vulnerable to the changing climate include potato cropping because of the high level of water use. It has been suggested that by the 2050s potato cropping will require irrigation-fed rather than rain-fed management across most of England and Wales (UKCP09 50% estimate, high and low emissions) (Daccache et al., 2012) [Medium confidence]. A more recent study (Keay et al., 2014b) estimates that the volume of water for irrigation would need to increase seven-fold by the 2050s (UKCP09 high emissions scenario) for present-day levels of potato production in England and Wales to continue. The same study found that the area of potato crops not requiring irrigation is projected to decline from 45,000 hecrates (1961- 90 baseline) to 1,400 hectares by the 2050s (high emissions scenarios).

Increased irrigation demand for potatoes has also been identified for Scotland (Brown et al., 2011) and Ireland (Holden et al., 2003). Ultimately, a geographical shift in potato-growing areas to less water-stressed areas in the north and west, in order to reduce irrigation needs, may become the most viable option for the potato industry, although this would require good management practice to counter the increased risk of soil erosion (section 3.3.1). Some types of horticulture with high water demands may also become less viable in their current locations unless more efficient irrigation systems are deployed.

Figure 3.8. Projected agricultural land suitability for potatoes and carrots in England under a low and high UKCP09 emission scenatios (2050s).



Source: ECI et al. (2013) for the ASC.
Notes: Agricultural land suitability for potatoes and carrots under three UKCP09 scenarios (Baseline, 2050 Low and 2050 High) for areas where crop was present in 2010 based on June Agricultural Census data.

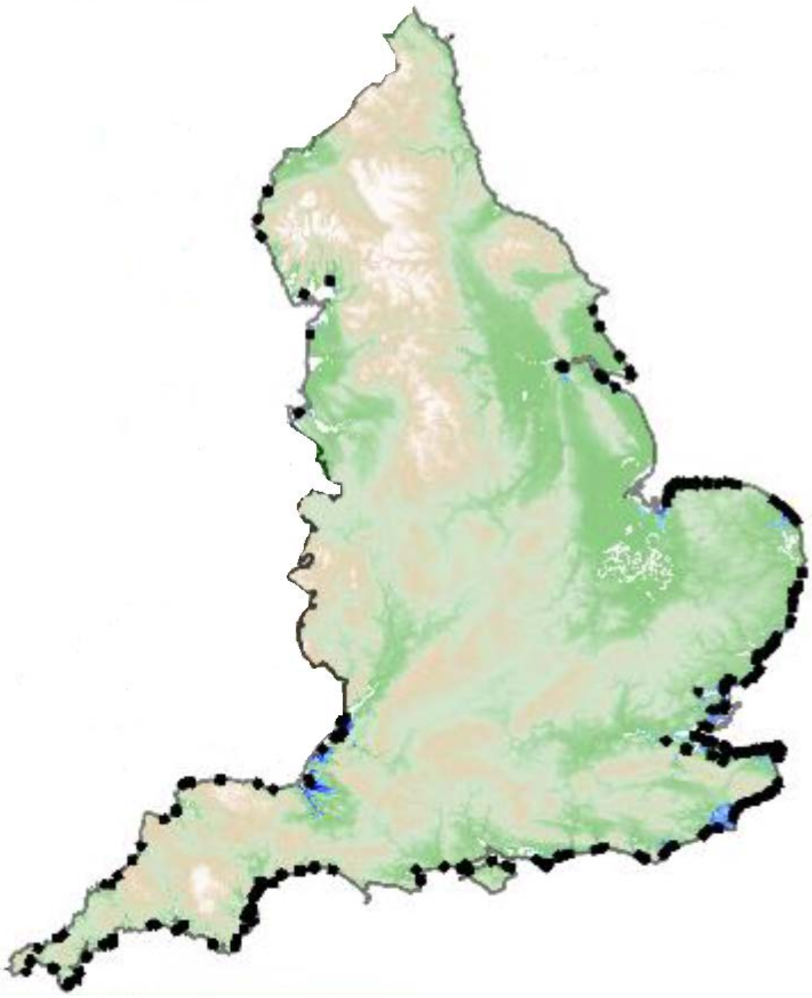
The implications for land-use of a projected trend to wetter winters shown by most climate models remain uncertain. Winter is normally a less active period due to problems caused by waterlogged soils on land accessed by machinery and livestock, but the length of this limited access period can be critical for many farming systems. For livestock systems, the increased length of the growing season into the winter months may be outweighed by the negative impacts of sward damage due to poaching from increased wetness. Current climate model projections tentatively suggest the most likely outcome is that constraints on seasonal access to land in spring remain broadly as at present; but that on average autumn access may be extended because of higher soil moisture deficits resulting from drier summers [Low confidence]. However, such conclusions are dependent on spring and autumn precipitation patterns, which have lower confidence in climate models.

Indicative results in England and Wales suggest a potential average increase of around 30 days when soils are workable by 2050 compared with 2000 (HadRM3model: q0 run) (Cooper et al., 2010), although soil wetness factors and associated risks from soil compaction will remain an issue in wetter areas of the UK. Managing wetness risk at acceptable levels is likely to be dependent on investment in drainage infrastructure, particularly to take advantage of an earlier start to the growing season. Further investment in maintenance of field drainage schemes for agricultural productivity would need to ensure that other ecosystem services (e.g. water quality and wetland biodiversity) are not compromised.

Flood risk may further reduce land availability and capability in vulnerable locations (notably in river and coastal floodplains). Land that is regularly flooded is only capable of supporting lower-value crops, pasture or woodland. The area of BMV or PAL in the UK at arisk of flooding is projected to increase. [Medium confidence]. Using an indicative 1in 75 year average risk level, flooding from fluvial, coastal and pluvial sources is projected to increase from 570,000 hectares (present day) to 750,000 hectares in the context of a 2°C rise in global mean temperatures by the 2080s; and to 940,000 hectares in the context of a 4°C rise (Sayers et al for the ASC, 2015). [Medium confidence].

Coastal flooding can impose additional constraints on agricultural land because saline contamination can necessitate high land remediation costs. An estimated 271,000 hectares of land in the coastal floodplain in England is protected by coastal habitats or artificial defences, of which over one-third (34%) is Grade 1 or 2 (ECI for the ASC, 2013). In the absence of additional adaptation, coastal defences will become increasingly vulnerable as sea levels rise to the 'toe' height of the defence foundations, leading to stronger and more continuous wave action on the weakest point of defence structures. With 1 m of sea-level rise, the length of highly vulnerable coastal flood defences in England would double from 110 km at present to around 220 km. If these highly vulnreable defences were all to fail during a 1-in-200-year tidal surge, up to 210,000 hectares could be inundated by seawater. The majority of highly vulnerable defences are located along the east and south coasts (Figure 3.9). In addition, a very small proportion of BMV land (<0.1%) was identified in CCRA 2012 to be at risk from coastal erosion.

Figure 3.9. Location of highly vulnerable coastal defences in England with 1 m mean sea-level rise



Source: Sayers et al (2015) for the ASC.
Notes: The location of coastal defences (black lines) assessed as highly vulnerable under 1m sea level rise. The map also shows the areas that would be temporarily inundated in a 1-in-200 year coastal surge event, assuming this causes the highly vulnerable coastal defences to fail. The exact scale of the impact would depend on where in the country the tidal surge occurs.

Climate-driven changes in land availability and capability will interact with changes in socio-economic demands for land. Preliminary high-level socio-economic analysis (CISL, 2014) has suggested that under a high-demand scenario, an additional 7.2 million hectares will be required by 2030. This is estimated from the upper end of extrapolated land demands from policy objectives for housing, woodland, bioenergy and nature conservation. The study also estimates that 5 million hectares could become available under a high-supply scenario, based on a combination of sustainable intensification (i.e. increased yields), waste reduction and changes in dietary preferences (principally reduced meat consumption). Under a low-supply scenario, only 1.2 million hectares becomes available, implying a potential supply–demand gap for land of up to 6 million hectares [Low confidence]. However, these estimates do not factor in the implications of climate-related changes in land capability discussed above.

Adaptation

All the aridity indices evaluated to date show the same trend, that levels of droughtiness are projected to increase significantly in future, particularly in the eastern side of the UK, thus reducing land capability. The magnitude of these changes will be influenced by the level of adaptation. The challenge for the agricultural sector is whether such adaptationoccurs as incremental changes, mainly through reactive adaptation as a response to poor farm performance, or whether there is a co-ordinated move to invest in new areas in anticipation of new opportunities.

Land-use planning currently shows only very limited use of adaptation planning. In agriculture, which is the main land use in the UK, there is little evidence for planned adaptation to enhance climate resilience and the dominant influence remains market forces and policy influences which currently pay little attention to planning for climate resilience (particularly CAP). The forestry sector, which represents the other main land use in the UK, operates over much longer planning cycles but interaction between the forestry and agriculture sectors, in terms of co-ordinated land use responses, is rather limited. Strategic land-use planning for national, regional and local authority purposes can operate over longer-term horizons but as yet shows limited incorporation of climate change adaptation in long-term decision-making, although mitigation measures are increasingly included. The role of planned land use changes to address flood risk, such as wetlands or woodland expansion, for integrated catchment or coastal zone management rather than localised interventions, also remains limited (see sections 3.4 and 3.5).

A research priority is to assess the implications of existing strategies for drainage and irrigation on the availability of land for different uses, and alternative management actions for existing land uses (e.g. to improve water efficiency). This should be linked to further development of good management practice and integrated with the regulatory framework.

The development of future scenarios of both supply and demand for land will need to account for a dynamic spatial component as climate change influences land quality in different regions of the UK and elsewhere. This identifies an increased need for the use of integrated land-use planning to also anticipate and maximise new opportunities in addition to risks.

3.3.3 Crop production

Synthesis

Crops are sensitive to weather and therefore a change in climate means that yields and overall national productivity are also very likely to change. Depending on the crop, and the rate and magnitude of climate change, there will be both risks and opportunities. The influence of climate at present is most evident during extreme years. The vulnerability of UK production at these times can lead to increased imports to meet demand. Future yield projections have considerable uncertainty due to the influence of multiple factors, the high degree of geographic variation in these factors, and sensitivity to the rate and magnitude of climate change (including CO_2 levels). Major crops such as wheat and sugar beet could potentially increase in yield, but not by as much as implied by CCRA 2012, and only if there is appropriate adaptation to changing climatic conditions. Production of some crops, such as potatoes, may be adversely affected by reduced water availability in their current growing areas.

Production is strongly influenced by market forces and the demand for different crops, and both are also projected to change in future. At present, climate adaptation tends to be mainly reactive, especially following adverse weather, although some planned initiatives at farm or business level are occurring to take advantage of new opportunities.

Improved resilience of crop production requires two types of adaptation measure to address both risks and opportunities. Firstly, strategic initiatives to maximise use of limited land resources (e.g. spatial planning, protection of better-quality land) to address current inefficiencies in the use of land for crops. Secondly, initiatives to develop and implement innovation in enhancing yield gains, including the uptake of new technology (e.g. genetics, crop breeding, precision farming) and enhanced management practices (e.g. crop rotations). In both cases, long lead times are involved in designing, developing and evaluating such initiatives as they translate from policy and research through to uptake by farming businesses. Measures also need to incorporate the influence of climate-related pressures on soil and water resources and biodiversity (notably pollinators), which emphasises the need to further develop good practice based upon quality, efficiency and sustainability of resources.

Context and policy

Based on the most recent set of published UK government statistics (Defra et al., 2015), 76% of land in the UK is classified as agricultural (18.5 million hectares), of which 93% is considered usable for the purposes of agricultural production (17.2 million hectares). Of this Usable Agricultural Area, 64% is grassland and 36% is considered ‘croppable’. In any given year, approximately 75% of the land that is suitable for cultivation of crops is used for this purpose (4.7 million hectares), with the remaining proportion being wooded, sown to grass or left uncropped.

- **Cereals:** In a typical year up to 70% of the cropped area grows cereals (c. 3.2 million hectares) of which wheat accounts for 61%, barley for 34% and oats for 4%, with rye and triticale making up the remainder. Total annual production of cereals has, in recent years, been about 21 million tonnes. Wheat production frequently exceeds domestic annual demand (c. 14 million tonnes) by as much as 20% but, in years when there is a poor harvest (such as 2012 and 2013 due to wetter weather), wheat imports may exceed exports by up to 15%. After seed production, other minor uses and waste are taken into account (c. 6% of annual production), UK wheat is used for milling and animal feed in approximately equal measure (each c. 6.6 million tonnes). Barley production routinely exceeds domestic demand (c. 5.3 million tonnes) with 10 – 20% of the crop being exported dependent on scale of production. Use of the crop for animal feed and malting is approximately in the ratio

1.8 to 1. Oat production is modest by comparison (c. 740,000 tonnes), with supply and domestic demand more or less in balance. The use ratio for milling and animal feed is approximately 2 to 1.

- **Break crops (oilseeds, beet, pulses, potatoes):** Cereals in general, and wheat in particular, cannot be grown continuously without adverse impacts on yields (primarily due to build-up of soil-borne fungal pathogens). Hence, rotation (on a two- to four- year cycle) with oilseed rape as the primary break crop has become the UK norm. Sugar beet, pulses, potatoes and, more recently, maize (for bioenergy production) also feature in rotations dependent on region and soil types. As the major break crop in UK cereal rotations, the area sown to oilseed crops (primarily oilseed rape) is largely a function of the cereal area and is consistently about 700,000 hectares. Yields fluctuate in the range 3 – 4 t/ha and annual production (c 2.4 million tonnes) routinely exceeds domestic demand by about 25% on average. Small areas of minor oilseed crops such as linseed and borage are also grown. Overall, the UK's annual import of processed vegetable oils (soya, sunflower, rapeseed and palm) amounts to around 270,000 tonnes, representing about 25% in excess of current UK production (based on rapeseed yielding 45% oil). Sugar beet and fodder beet provide good break crops in cereal rotations but the demand for fodder beet is limited and determined by the regional distribution of livestock. EU quotas (until 2017) and, particularly, the scale and regional location of processing capability currently limit UK sugar beet production, hence UK demand for sugar beet exceeds local supply by about 63%. The UK imports around 800,000 tonnes of soyabeans annually in addition to around 1.7 million tonnes of processed soyabean meal for animal feed. This dwarfs the UK annual harvest of dry peas and faba beans (c. 400,000 tonnes); less than 140,000 hectares of these crops are grown (just over 2% of the UK croppable area). Pulse crops are primarily grown for protein (in addition to their positive impact on soil fertility). In terms of protein produced per hectare, soyabeans are significantly more productive than peas and faba beans due to around 15% more seed protein (although seed yields of these crops are not dissimilar). With the exception of years when summer rainfall is exceptionally low or high (e.g. 2012) and productivity is reduced, the annual area of potatoes grown (c.140,000 hectares) and average yields achieved (c. 40 t/ha) consistently satisfy approximately 80% of UK demand for fresh and processed potatoes (c. 4.7 million tonnes.
- **Horticultural crops:** Vegetables and fruits are grown in the UK on an area of around 132,000 hectares plus a small area (2585ha in 2014) in glasshouses (primarily salad crops); however, these areas overstate actual land used for this purpose because two or more crops may be produced from the same area each season. Total UK demand is around 4.6 million tonnes, of which about 58% is provided from domestic production. These figures probably conceal significantly greater levels of wasted production that fails to reach the market due to fluctuations in price and demand and quality criteria. Fruit crops are grown on a small area (c. 36,000 hectares) and domestic production represents only 11% of UK demand for around 4 million tonnes. A significant proportion of imported fruit is seasonal or tropical and could not be produced in the UK, although domestic supply of apples (c. 32%) and strawberries (c. 70%) is substantial. Production of other fruit crops in the UK meets significantly lower proportions of domestic demand.

Defra is currently working on the development of a long-term Food and Farming Plan for England , to 'grow more, buy more, and sell more' British food. This is a response to the challenges and opportunities presented by projected increases in global demand for food and threats to food security, including from climate change. Meanwhile, current policy statements (updated in May 2015) make clear an ambition to make the farming, food and drink sector more productive and competitive by increasing resource-use efficiency and reducing waste. The Green Food Project (Defra, 2012) describes how the food system needs to change to increase food production while also improving the environment. The Government directs funds towards providing advice and encouraging actions that will both increase production and provide environmental protection. In 2013 the Government published the UK Strategy for Agricultural Technologies with the objective that the UK would become 'a world leader in agricultural technology, innovation and sustainability … and thereby contribute to global food security and international development'. Actions and innovation to mitigate and adapt to climate change represented a significant component of this strategy. The Government's 2013 National Adaptation Programme include sactions required to address flooding of agricultural land and water demand for irrigated crops, and also refers to the Agri-Tech Strategy and Sustainable Intensification Programme.

Current risks and opportunities

Crop production is sensitive to local weather conditions and this will often be reflected in national-level annual production figures if widespread favourable or poor weather is experienced in a particular year. Sensitivity to prevailing weather conditions varies with time of year and between different crops, and also to a lesser extent between crop varieties. Crops respond differentially to favourable or adverse weather conditions at particular stages of the crop production cycle. Hence, weather at sowing and planting, at crucial stages of crop growth and development (particularly flowering), and at harvest can have major impacts on crop yield and quality.

The diverse climate of the UK has meant that crop production has tended to be concentrated in the drier, warmer and sunnier regions in the east and south (see Section 3.3.2). These regions also tend to be the locations that have less inter-annual climatic variability. This allows routinely successful crop production by making possible the co-ordinated annual schedule of management practices that would otherwise be vulnerable to disruption.

The factors that influence crop growth and development, and thereby yield and quality, are, with the important exception of day-length (and rate of change of day-length), all affected to a greater or lesser extent by climatic factors and consequently by climate change. Climate can directly impact crop growth and development (beneficially or detrimentally) through the following factors:

- Temperature (mean and variance).
- Water shortage and excess (precipitation frequency, intensity and volume).
- Intensity of incident radiation (cloud cover).

In addition, indirect influences of climate occur through the following factors:

- Soil condition and availability of nutrients (see Section 3.3.1).
- Atmospheric CO_2 concentration.
- Incidence, distribution and identity of invertebrate herbivores and disease vectors.
- Incidence, distribution and identity of pathogens (bacteria, fungi, viruses and viroids).
- Identity, density and distribution of weed species.

There are complex differential interactions between all these factors in terms of their impact on crop growth and development, together with how they are managed. For example, flowering time is mediated by a complex interaction between temperature and day-length, while nutrient uptake by roots is influenced by soil temperature and rainfall. Indirect interactions are mediated through the plant's response to environmental factors. For example, elevated CO_2 concentration, through its effect on photosynthesis and thereby plant metabolism, has been shown to result in increased severity of losses due to certain herbivorous insects (e.g. Whittaker, 1999; De Lucia et al., 2012). Some genes involved in plant defence against pathogens fail to operate at high temperatures resulting in unanticipated disease susceptibility (Huang et al., 2006; Keen, 1990).

These interactions make it difficult to attribute observed trends in crop yield to recent climate change. CCRA1 was erroneous in implying that yield improvements in key crops such as winter wheat could be directly linked with temperature increases over recent decades. This is because a dominant factor over the same time period has been the adoption of new varieties as well as uptake of new technology and modified management practices (Semenov et al., 2012). In recent years, the yields of many (but by no means all) crops have reached a plateau. Where increases in yield have slowed or ceased, climate change has been suggested as a cause, but this depends on the crop. Analysis by Moore and Lobell (2015) found that climatic factors explained less than 5% of the change in UK barley and wheat yields since 1990. Market conditions and tailoring input costs (such as fertiliser use) to maximise margins are likely to have had more influence on wheat yields than climate variables over recent decades. However, in contrast, Jaggard et al. (2007) demonstrated that more than 60% of the yield increase in sugar beet since 1976 can be attributed to earlier sowing of the crop (an indirect effect of temperature elevation).

The relationship between crop yield and climate variables has been demonstrated by correlations between inter-annual values; the role of spring/summer radiation and summer/autumn soil moisture has been identified as particularly important (England: Atkinson et al., 2008; Scotland: Brown, 2013). [Medium confidence] This was exemplified by a reduction in yields during the poor weather of 2012 (UK wheat yields were reduced by c. 15% and potato yields by c. 26%). This relationship also seems to be consistent with phases of the winter North Atlantic Oscillation (i.e. positive phase correlated with higher yield[16]), at least in terms of the winter wheat quality as distinct from yield itself (Atkinson et al., 2005). [Low confidence]

The research referred to above suggests that the impact of climate on crop yields is currently most visible during years of extreme weather (good or bad) and in the absence of extremes it is socio-economic factors, including impacts of technology, that best explain any observed trends. [High confidence] Large increases (almost 100%) in the yield of UK-grown strawberries over the past two decades illustrate the point. Increases can be attributed to widespread adoption of innovative growing techniques and improved varieties, as well as seasonal extension. Interpreting national-level productivity data is further complicated because of potential changes in regional crop distribution. Forage maize is, for example, being grown much further north than was the case a decade ago and it is possible that the geographical range of other crops is also changing, such that suitable growing conditions may not be experienced consistently even if they do occur more frequently.

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[16] A positive phase of the NAO in winter (represented by warmer, wetter conditions) has a tendency to be followed by a negative summer NAO phase (sunny drier conditions) in recent decades, which although a weaker phenomena in summer, means that grain-growing areas of the eastern UK have an association with higher yields in these years.

Adaptive responses in crop production are currently a reaction to market opportunities that arise. These may derive from improved economic returns even when set against the risks of failure due to poor weather, and there are carry-over effects from year to year. For example, the adverse weather of 2012 had a knock-on effect in 2013 with the observed shift from autumn-sown to spring-sown cereal crops. There is therefore a concern that yields of some cereal crops could be particularly vulnerable to a run of poor years, as happened in some locations during the 1980s. The widespread use of varieties selected to perform well under favourable conditions may represent a risk if they are less resilient under unfavourable conditions than other varieties (Brown, 2013) [Low Confidence]. Moore and Lobell (2015) used the difference between short-term change and long-term change in yields to look for evidence of this type of large-scale adaptation but they found no evidence.

Future risks and opportunities

Future climate change will bring both risks and opportunities for crop production depending on the crop (and variety) and the rate and magnitude of climatic change. UK food security (in terms of food availability, price, safety and nutrition for UK citizens) and the import/export balance will be substantially dependent on the magnitude of changes in other key food-producing areas (see Chapter 7, International dimensions). Changes in global food and commodity markets will affect the economic performance of UK agricultural businesses, and the relative competitiveness of UK agriculture compared with the rest of the world. Analysis at European level suggests a relative gain for production of four key crops in northern Europe by comparison with southern Europe (Moore and Lobell, 2014, 2015). Socio-economic projections suggest increased UK and global demand for agricultural commodities that will also interact with the changing climate. Within the UK, climate change is likely to influence the relative advantages of different regions for growing crops, and this may cause a shift in production depending on the level of agronomic adaptation required in anticipation of or in reaction to the changing conditions. [High confidence] For example, warmer temperatures are projected to advance crop growth, with current wheat cultivars flowering around two weeks earlier and being harvested three weeks earlier.

The consensus view is that the increases in wheat yield projected for the UK in CCRA1 were overestimated because of their reliance on a temperature-based metric with exclusion of other factors including key thresholds. For example, national-scale studies have highlighted the effects of heat stress (above specific thresholds) during flowering on wheat yields in the UK (Semenov and Shewry, 2011). However, quantification of yield changes using multivariate crop models is complicated because of the difficulties of generalising from the site-specific conditions (for which the model is developed) to national-scale estimates. As also identified above, annual crop areas will also change. This will be partly as an autonomous response to climate as land capability changes (Section 3.3.2) but also due to socio-economic factors; both will have an impact on national-level production figures. At the global level, the AGMIP international multi-model project has predicted that wheat production will fall by 6% for each degree centigrade of global warming, together with increased variability of yield across regions and seasons (Challinor et al., 2014).

Potatoes are identified as a crop that is particularly vulnerable to climate change [Medium confidence]: this is due to projections of reduced water availability from a trend to drier summers and a reliance on irrigation to address excessive moisture deficits (as discussed in Section 3.3.2). In eastern England, where a significant proportion of the UK crop is grown, a substantial amount of water is used not only to grow the crop but particularly to enable a good-quality crop to be lifted from otherwise hard-baked ground. In England and Wales, Daccache et al. (2012) found that by the 2050s (under a central climate projection for the low emissions scenario) the area of land well suited and moderately suited for rain-fed potato production would decline by 88% and 74% respectively. The increase in the volume of water required for a switch from rain-fed to irrigated potato production is likely to be far greater than the incremental increase in current irrigation demand anticipated in already water-stressed areas of eastern and southern England. Similarly, in Scotland (Brown et al., 2011) and Northern Ireland (Holden et al., 2003) the area of land suitable for potato production has been projected to decline (dependent on climate scenario) unless additional irrigation is available to maintain yields and enable harvesting. These projections are based on the continued deployment of current varieties and an alternative adaptation strategy might be the development and adoption of more drought-tolerant varieties.

The benefits of warmer temperature for maize production are likely to encourage further expansion northwards and an increase in production. [Medium confidence] However, if this occurs at more sensitive sites (e.g. steeper slopes) then some negative impacts through soil erosion from bare ground are likely (see Section 3.3.1).

A major unknown in all future projections for crop productivity is the magnitude and impact of CO_2 fertilisation. Yield increases resulting from higher atmospheric concentrations of CO_2 will only be achieved if there are no other factors, such as water and nutrient availability, which act to limit yield. Under such circumstances, models suggest that increases in wheat yield of around 10 – 17.5% may be realised despite the earlier harvest date (AHDB project 539).

Productivity is markedly influenced by other biotic factors, and notably the incidence and severity of pests and diseases. Climate change will affect crop disease both directly (e.g. by affecting spore release, the frequency of suitable infection conditions, host resistance, speed of disease development and disease survival between seasons) and indirectly (e.g. through the production of susceptible crops in new locations). Much research has been published on the interactions between climate change and crop disease incidence and severity (e.g. in 2011 a collection of 12 papers on the subject was assembled in a volume of *Plant Pathology*). Generalisations are difficult because the environmental conditions conducive to major pest or disease outbreaks can be specific to the organisms involved. However, the time of occurrence of certain disease outbreaks is likely to change [High confidence], with milder winters and springs enabling pathogens such as rusts and mildews to become more severe in these seasons but less severe during hot, dry summer weather. Increased likelihood of warmer autumn conditions is likely to elevate the risk of root and stem rot pathogens that infect crops later in the year. Conversely, the relative severity of spring-infecting pathogens may not change due to earlier crop growth.

The integration of site-specific weather simulations (based on climate change scenarios) with epidemiological models of population dynamics for specific pests and pathogens can enable instructive predictions to be made. For example, Butterworth et al. (2009) made predictions about the regional impact on yield due to the incidence and severity of phoma stem canker disease on oilseed rape under different climate change scenarios. They predicted that the greatest yields (for fungicide-treated crops) would be in eastern Scotland and north-east England, with greater increases for the high-emissions scenarios and for the 2050s than for the 2020s. The yield losses from the disease were predicted to be greatest in south-east England (c.17% of total production). In addition to direct adverse impacts on crops from insect attack, increases in the distribution and populations of disease vectors are likely to bring about greater incidence and severity of diseases caused by viruses and phytoplasma.

Adaptation

Adaptation to climate change risks and opportunities may occur through changes in land use or through innovations in technology or crop management that improves yields. In each case, there are likely to be significant lead times to implement new measures, either through development of zoning in strategic land-use planning, or for research and development of improved crop varieties or introduction of novel crops that can provide yield gains. Some of these responses are also likely to involve important issues for other aspects of the natural environment, including biodiversity, soil and water resources.

There are significant uncertainties in future consumer demand for food, particularly regarding the balance between different food types. Business as usual projections in consumer demand based on historical trends are unlikely to hold true in the future. Issues such as dietary change, increasing consumer empowerment and personalised nutrition planning will affect patterns of demand and therefore supply (Foresight, 2011). Current discussions about the health impacts of sugar consumption and the need for obesity strategies suggest demand for sugar in developed countries may not continue to rise as it has in recent decades. Similarly, the Paris Agreement on climate change implies some restructuring of global diets. Changes in regulatory frameworks (e.g. water abstraction reform) may require crop production to adapt to new price signals and water availability constraints. Shifts in global patterns of demand will also affect trade and market-led decisions on land-use and crop choice in the UK. In short, patterns of demand for food and commodities may well change significantly due to socio-economic factors as well as the need to limit greenhouse gas emissions from agriculture. Understanding both demand- and supply-side risks and opportunities for UK agriculture, particularly in the context of global changes, is of critical importance in considering adaptation responses.

There is, however, relative certainty that the UK population will be higher in the 2050s than it is today. Central ONS projections suggest a 20% increase, to nearly 80 million people. Some studies have predicted that by 2050 per capita demand for cereals in the UK could increase by 14% from the current 350 kg a year (de Ruiter et al., 2016). Annual UK demand for cereals is therefore likely to rise, potentially as high as an additional 31 million tonnes from current levels.

Wheat is the highest-yielding cereal crop with the greatest diversity of uses, and it is possible that the above increase in demand could be met by higher UK wheat production. However, as highlighted in Section 3.3.2, land in the UK is a finite resource under competing pressures and its capability is also changing, which has implications for the maximum cropping area. If the area under wheat remains constant, an average yield of over 15 t/ha would be required to meet potential increases in cereal demand. This is a lower yield than the current UK (and world) record of 16.5 t/ha but still represents an improbable 1.6% year-on-year increase in average annual UK yields. Sustainable intensification to improve yields would also require additional measures to maintain natural capital, including soil quality (e.g. nutrients, organic matter, see Section 3.3.1), and improvements in water efficiency.

OECD-FAO projections suggest that global per capita demand for sugar could increase by 6% over the next 10 years (OECD/FAO, 2015). As noted above, it is doubtful that this global per capita increase is likely to translate to the UK and other developed countries, but if it were to do so then annual UK domestic demand for refined sugar could potentially increase from 2.3 million tonnes at present to nearer 2.9 million tonnes. Sugar beet is one crop where there is good evidence for recent climate change-related yield increases but, even if processing capacity was increased, meeting such demand implies doubling UK current levels of production. However, as noted above, the growing understanding of the role of sugar in adverse health outcomes may mean that demand projections in industrialised countries like the UK will be downwards rather than upwards.

It may become possible to consider arable rotations that include break crops such as grain maize, sunflower and other pulse crops like soya and navy beans. Heavy dependence on oilseed rape as the break crop in cereal rotations is not considered sustainable in the long term due to the build-up of soil-borne diseases. Hence, there is a need to increase crop diversity in rotations.

For pulses, increasing domestic yields is technically achievable, and projected rises in average temperatures during the growing season may make soyabean production an option in parts of the UK. However, substitution of current UK imports of seed protein would require an unrealistic ten-fold increase in the area allocated to cultivation of pulse crops.

An increasing UK population may not necessarily result in a corresponding increase in consumer demand for potatoes. AHDB Market Intelligence shows that demand in the UK has been on a downwards trend in recent decades as per capita consumption of alternative carbohydrates (i.e. rice and pasta) has increased. Nevertheless, if domestic demand were to increase in line with population growth then potentially higher yields as a result of longer growing seasons could go a long way to meet any potential shortfall. However, as discussed in Section 3.3.2, potato production has high water requirements and this is likely to be an increasing constraint on production in drier areas where water for irrigation is limited. Other non-climate factors are also already limiting production, for example cyst nematode infestation.

Access to adequate irrigation is also necessary for vegetable production and reduced water availability may counteract the expected benefits from an extended growing season, particularly in eastern areas. If suitable land were available for a significant expansion of vegetables (to ca. 190,000 hectares), there is scope to displace significant proportions of UK imports, but changes in land availability and quality (Section 3.3.2) means that this is assumption that requires further evaluation. Similar opportunities may exist for scaling up the production of tree fruit crops (apples, pears, plums, cherries), including outside traditional geographical areas, as well as introducing novel crops (apricots, peaches and table grapes). Production of soft fruit (berries) has grown substantially in recent years. In addition to meeting the potential increase in domestic demand, there is a realistic opportunity to also significantly increase exports.

Replacing vegetable oil imports with domestic rapeseed oil production would require average annual yields to increase to 4.5 t/ha, which is technically achievable. Production would need to increase by a further 20% by the 2050s to meet any increase in domestic demand as a result of population growth. The required yield increases over this period should, however, be readily achievable given that crops yielding over 6 t/ha are already recorded, although current problems with soil-borne diseases (as noted above for break crops) would need to be overcome.

The current ratio of cereals to other crops by land area in the UK is approximately 3 to 1. Projected increases in population imply increases in domestic demand, although there is significant uncertainty as to how this may play out. Unless significant yield gains can be achieved, it is possible that more potential croppable land will be used for cultivation of crops in the future. This would be likely to include expansion of the area used for cereals with more diverse, resilient rotations than are currently being practised. An increase in the croppable land area used for fruit and vegetable production is also possible with improving growing conditions (especially for fruit), and therefore a likely requirement for less reliance on imports. An increase in the area (and geographical distribution) of horticultural crops with commensurate increases in yield would represent a significant opportunity for the agricultural industry. Any expansion in total croppable land would need to be in association with sustained investment in elevation of yields, as well as geographical diversification of crop production and crop diversification within rotations.

Approaches to adaptation through three distinctive types of action could include

- **Genetics and adaptive crop breeding:** The development of new cultivars is already an active area of research but climate change may require an emphasis on a different range of traits than has been conventionally considered. Genetic improvement of crops through conventional breeding (hybridisation and selection) or the use of biotechnologies is a relatively slow process but is happening continuously in most crops as improved varieties are introduced which replace those with inferior attributes. This occurs more rapidly and regularly for seed-propagated annual crops (such as cereals and vegetables) than it does for crops that are propagated vegetatively (such as potatoes) and particularly perennial crops (such as tree fruit). Some guidance for plant breeders on the most appropriate traits for climate resilience may be provided through the use of ideotypes based upon modelling. For example, Semenov et al. (2014) suggested primary factors contributing to wheat yield increase were improvement in light conversion efficiency, extended duration of grain filling (resulting in a higher harvest index) and optimal phenology). Achieving greater yields will require resilience to adverse climatic conditions and simultaneous selection for increased tolerance to heat and drought stress. If the rates of environmental change and crop genetic adaptation proceed at the same pace then it could be argued that simply continuing an established process will be an adequate response. However, this would be complacent since there is no evidence that the pace of required genetic adaptation in all crops is sufficient to keep pace with environmental change. More importantly, however, the environmental conditions likely to be experienced at any particular latitude do not currently exist and have never previously been experienced (e.g. combinations of day-length, temperature and CO_2 concentration). Hence, the notion that it will be possible to adapt agriculture simply by moving southerly-adapted crop genotypes (and 'novel' crops) northwards may be a flawed assumption, although this has not been tested with field experiments. Day-length (and rate of change of day-length) is a key determinant of the timing of key developmental processes (tuberisation, bolting and flowering) influencing the yield and quality of harvested products (roots, fruits, seeds and leafy shoots). The continuing maintenance of germplasm collections. of relevance to UK crop genetic improvement is an important adaptation. The viability and suitability of potential novel crops that might be more fully integrated into UK agricultural systems have not been systematically analysed. Possibilities include grain maize, field-grown tomatoes, sunflowers, *Phaseolus* beans, soyabeans, table grapes and apricots.
- **Optimised land use planning and crop location:** Climate change will almost certainly require relocation of crops from one region of the UK to another (e.g. potato production from the semi-arid east to rain-fed production in the west), with all the necessary capital investments in farming businesses and regional infrastructure. Using the developing concept of 'precision farming', there are opportunities for innovative approaches to optimised regional land (and water) use. Well-founded decisions about suitability of land use for particular enterprises will be enabled through: integration of data from remote sensing or imaging techniques (satellite or UAV); existing and new databases (soil type, topography, historical land use); ground-based geophysical and biological appraisals of soil quality and health; and economic assessments (e.g. yield potential estimations). The underlying rationale is that deploying data integration at this scale will increase production efficiency through strategic business-to-business land-use planning at a regional level, hence removing the land-based constraints otherwise being experienced. This would represent a major step beyond the ad hoc land-trading and sub-contracting that currently takes place. If validated,

the commercial opportunities and efficiency gains at national and international level could be substantial.

- **Improved soil, water and crop management:** Changes in the integrated practices of soil, water and crop management at farm level will be fundamental to successful adaptation. Examples of the types of issues to be confronted include:
  - *Drainage:* Waterlogged farmland is unproductive and crop losses due to flooding have been significant recently. Investment in farmland drainage has been neglected for the past 50 years and recent flooding episodes have further highlighted problems. Data is lacking on the current state of farmland drainage systems, the scale of current investment (including inducement to invest) and priority areas in which to address waterlogging issues.
  - *Irrigation and associated water capture and storage:* The scale of recent and ongoing investment in water capture and storage (reservoirs) by farming businesses is thought to have been significant, although data on uptake is currently not systematically captured and it is not known whether the current levels of uptake are adequate to meet projected future demand.
  - *Soil management:* The physical properties of soil that enable both free drainage and water-holding capacity are substantially influenced by appropriate soil management practices, and particularly amendments with a range of organic materials (including animal manures, paper waste, straw etc.). However, optimum amendment practice is not well defined, regional demand for materials may exceed supply and requirements may be soil-type or land-use specific.
  - *Fertiliser timing:* Changes to the optimum timing and use of fertilisers on a range of crops are likely in response to anticipated changes in rainfall (intensity, frequency and volume) and crop growth rates (relating to temperature and CO_2 concentration).
  - *Protected/indoor cropping:* Demand for seasonal extension and the protection of crop quality is driving the increased use of various forms of physical protection (glass, polythene, fleece and gauze). High-value crops are increasingly being grown indoors, with supplementary lighting providing different wavelengths (that affect crop development and quality) provided by LEDs. Some crops are being grown exclusively in the absence of natural daylight. These trends are set to continue, but to date there has been no assessment of the extent of the various approaches and technologies,including implications for water resources and land use planning which would also be of importance with regard to climate change adaptation.
  - *Surveillance, forecasting, and management of pests, diseases and weeds:* Marked changes in the incidence, distribution, severity and identity of the diverse pests, pathogens and weeds that affect UK crop yield and quality can be anticipated with changing climate (see Section 3.7.2). Some limited work has been undertaken to explore specific crop and pest or disease combinations under different climate change scenarios and these provide the basis for a response (e.g. higher priorities for resistance breeding). Coincident with the threat of climate change is the development of resistance to available crop protection products, which has to be considered in the context of regulation and authorisation of usage.

3.3.4 Livestock production

Synthesis

Grassland productivity and other sources of fodder are linked to livestock productivity and are therefore included together in this section.

Adverse weather can affect livestock through stress and mortality from cold and heat, availability of water and availability of food including grass and fodder crops such as maize. In addition, weather can influence the prevalence of livestock diseases, especially during wetter periods. Risks are most evident during extreme events, especially as livestock production often occurs in more marginal agricultural areas compared with crop production. Although future climate change is highly likely to involve a reduction in cold-related stresses and potential benefits for grassland production from a longer growing season, risks from heat stress, water availability and wetter winter conditions are expected to increase. Grassland productivity may be reduced if conditions become too hot and dry or the soils become compacted or waterlogged. Opportunities may also become more evident, notably from longer growing seasons and potential increased production of forages, including improved prospects for alternative forage crops such as legumes and forbs.

Although there is clear evidence of climate-related sensitivity, evidence for risks and opportunities that can be unequivocally attributed to climate change is not well established. This may be partly due to the strong influence of year-to-year variability and the role of other factors in influencing production levels. Based upon the importance of livestock production for many rural areas of the UK, it will be important to keep monitoring the impacts and to further research the role of climate in any changes in production. This should then be regularly reviewed to assess whether further action may be required in the future to improve overall resilience levels.

Context and policy

Livestock production accounts for the largest proportion of land use in the UK and provides a major contribution to the rural economy through food and drink production. Climate is an important factor in livestock production, in terms of animal health and welfare issues and also through the availability of grassland and fodder crops. The quality and quantity of grass in turn affects the quality of the final products and the viability of livestock businesses. Dairy cows need lots of young nutritious grass and high-quality silage to support milk production, whereas beef cattle do best on grass that is more mature and of lower palatability. By contrast, sheep prefer swards with shorter grass. Effective management and use of grassland therefore often involves different types of livestock grazing the same grassland in a complementary manner. The livestock sector is also a relatively large contributor to greenhouse gas emissions (primarily methane) and therefore the need for adaptation measures to improve climate resilience is coincident with a major drive to reduce emissions in the sector.

The CAP is the primary policy mechanism affecting the UK livestock industry. The 2014 – 2020 CAP framework aims to support viable food production, sustainable management of the natural resources and climate action along with balanced territorial development. Livestock production also has important links with wider policy objectives. Scotland's Food and Drink policy identifies high-quality livestock produce as a significant area for expansion through the development of export markets as well as local markets. Northern Ireland has set targets for the expansion of dairy production by 2020 and increasing the turnover from the beef and sheep sectors.[17]

Current risks and opportunities

Grass and fodder crops

Meteorological conditions influence grass growth over the course of the growing season. Grass growth begins above a minimum temperature (5.5°C) and is stimulated by warmer weather, provided that there is sufficient soil moisture. Peak grass growth occurs in late spring and early summer. Growth in the late summer and autumn is restricted as temperature and solar radiation decrease. As a result there are variations in grass growth within and between years, making grass budgeting at farm level challenging. It is normal practice for farmers to aim to have sufficient silage to cope with very wet autumns or cold springs, but achieving this surplus can be difficult in marginal areas. The unpredictability of the weather can also present challenges in harvesting sufficient silage. Warmer springs may increase the prevalence of grassland weeds, which in some cases will affect the usability of the forage (Tiley, 2010). The drive to reduce costs in the dairy industry means there is interest in extending the outdoor grazing season (as is occurring in New Zealand), but the management challenge to achieve this is exacerbated in wetter areas. Furthermore, over half (55%) of cultivated grasslands in south-west England have soils which are structurally damaged (Palmer and Smith, 2013); this has implications for sward productivity.

Although grass yields improve with warmer conditions, they are also vulnerable to reduced soil moisture availability during drought. [High confidence]This is demonstrated by long-term data from the Park Grass experiment that has been run by Rothamsted Research data which shows a 1°C increase in maximum July/ August temperatures is associated with a 0.33 t/ha loss of yield. Further analysis of these data has shown that the significant relationship is actually between herbage growth rate and soil moisture (meaning a reduced yield with higher moisture deficit) and also that the pattern of inter-annual variability has a good relation with the North Atlantic Oscillation of the preceding winter (Kettlewell et al., 2006). By contrast, the results from a long-term study of Irish grass yields in a wetter climate indicate that production is influenced by temperature, solar radiation and evapotranspiration rates, but rainfall variations had no significant effect (Hurtado-Uria et al., 2013).

In recent years, the area of land that has been planted with forage maize has increased. Maize is more tolerant of drier conditions than grass, and may provide additional nutritional value for livestock as well as being used in anaerobic digesters for biogas, but requires a minimum temperature higher than grass to grow effectively. In England, there has been an increase from 118,000 hectares in 2004 to 183,000 hectares in 2014. Over the same period in Scotland, the area has approximately trebled (564 hectares to 1,318 hectares). The increase has in part been due to improvements in cultivation techniques (particularly the technology of sowing under plastic) and plant breeding. It may also reflect an assumption among growers that climatic conditions are becoming more favourable. However, due to a run of poor summers in recent years, the productivity of maize has been variable. When the average temperature for the growing season (April-September) fell acutely in 2012 the yields of both open-sown and plastic-sown maize in Northern Ireland collapsed to 33-35% of the long-term average (AFBI 2014).

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[17] http://www.agrifoodstrategyboard.org.uk/uploads/Going%20for%20Growth%20-%20Web%20Version.PDF

Heat stress

One of the main direct effects of climate on dairy production is heat stress, which can adversely affect milk yield and fat and protein content as well as cause animal welfare issues. In Scotland, Hill and Wall (2014) reported that heat stress thresholds (known as the thermal heat index) for dairy cattle have been typically exceeded for one day a year on average over the past 40 years. During the heat waves of 2003 and 2006, the threshold was exceeded on typically five days in the Midlands and south and east England (Dunn et al., 2014). Overall milk yields are thought to have been largely unaffected by these heat waves, although milk production declined by 30% in a single herd in south-west England during the 2006 event (Dunn et al., 2014). [Medium confidence]

Cold weather

The behaviour of cattle can be negatively affected by cold and wet conditions, with reductions in both lying and eating (Webster et al., 2008) and reduced feed intake (Graunke et al., 2011), and hence lower productivity. An extreme snow event in spring 2013 caused higher rates of calf and cow mortality than in the previous three years and also resulted in a lower number of lambs per ewe (BCMS, 2014).

Pests and diseases

The seasonal risk of the endemic disease liver fluke (Fasciola hepatica) is increased by a combination of higher rainfall with minimum average temperature above 10° C (Fox et al., 2012). Recent temperature increases and periods of heavier rainfall, particularly in areas of the western UK used for livestock grazing, are associated with increased prevalence of the disease causing the death of young cattle and sheep (Mason et al., 2012; Millar et al., 2012). [Medium confidence] There has also been increased incidence of the incursion of exotic animal diseases. One example is bluetongue, where the vector is an insect, and this disease has increased likelihood of establishing itself in the UK under a warmer climate. This is evidenced from Germany where the hot summer and warm winter of 2006/2007 resulted in an increase in the disease (Mehlhorn et al., 2008).

Future risks and opportunities

Grass and fodder crops

The sensitivity of production to small changes in seasonal variability make it difficult to predict future changes in yield and the associated impact on livestock productivity. Modelled predictions for Ireland, based on climate projections for a medium-emissions scenario (IPCC SRES A1B), suggest that by 2050 forage maize yields could increase by between 43% and 97% and that grass yields could increase by between 49% and 56%, compared to 1961-1990 baseline yields (Shrestha et al., 2014). Warmer temperatures will undoubtedly be of benefit for grassland productivity, particularly in marginal areas that currently experience difficulties during colder conditions. In winter, the extended growing season may provide opportunities for longer outdoor grazing, but this could be counteracted if increased precipitation increases the risk of damage to swards by poaching (Defra, 2014). A trend to warmer, drier summers could also have negative impacts on grassland productivity in drier areas (Zwicke et al., 2013). [Medium confidence]

Increased CO_2 concentrations are predicted to enhance legume and forb growth (Newton et al., 2014). However, grazing by sheep preferentially removes legumes and forbs, which suggests that the long-term sward composition is unlikely to be significantly affected by increased CO_2 concentrations. [Low confidence]

The nutritive value of feed and forage will be affected by increases in temperature and higher incidences of drought (Lee et al., 2013). Higher temperatures are likely to bring forward reproductive development and senescence. As the nutritive value declines, there is a risk that voluntary intake of the grass will decline (Clark and Wilson, 1993). This will have implications for the feed composition of diets and it may affect productivity (Lee et al., 2013).[Low confidence]

Warmer summers bring opportunities for growing more forage maize and other alternative forage crops. Although feeding forage maize to cattle reduces enteric methane production compared with grass, if the forage maize is sown as a replacement for long-term grass then forage maize will have negative consequences for carbon sequestration and therefore for overall greenhouse gas emissions (van Middelaar et al., 2013).

Heat stress

In southern Britain, heat stress in dairy cows is expected to increase from an average of one day a year under current climatic conditions to around 20 days a year by the end of the century (Dunn et al., 2014). Milk yields are expected to reduce by 0.89 kg with every percentage point that heat thresholds are exceeded (West et al., 2003). This may have implications for the profitability of the dairy industry due to the reduction in milk yield, as well as increased costs of running cooling systems to maintain animal welfare (Dunn et al., 2014). [Low confidence]

Increases in temperature are likely to result in an elevated animal welfare risk during transportation over long distances. [Low confidence] Vulnerability to heat stress is related to the age and size of the livestock.Pigs and poultry are typically housed but they have limited ability to dissipate heat. Maintaining internal temperature bounds for animal welfare and productivity is therefore a fundamental requirement (Valiňo et al., 2010). Under the current climate, natural and forced ventilation appears to be adequate. However, in the longer term the installation of cooling pads may be required.

Cold weather

The timing of extreme events will also continue to be important. For example, poor weather in the spring, as occurred during 2013, can have detrimental effects on sheep and cattle mortality and generally because this is a key time in the annual production cycle.

Pests and diseases

Wetter winters in the future may have implications for the prevalence of pests and pathogens. [Low confidence] Fox et al. (2012) predict increases in liver fluke risk in winter, particularly in wetter western locations of the UK, with Wales being most at risk. There is also likely to be an increased risk of livestock production being affected by exposure to endemic livestock diseases and greater incursion of exotic diseases, for example bluetongue and Schmallenberg virus (Skuce et al., 2013). There is also a risk that the prevalence of non-native weeds will increase (Bourdôt et al., 2010).

Adaptation

Declining nutritive value of grasses can be overcome through purchasing additional concentrates or modifying the management of the grassland to ensure that there is a sufficient supply of high-quality grass. The drought response of perennial ryegrass is partially controlled by genetic factors and thus there is the potential to breed varieties more suited to a future drier climate (Hatier et al., 2014).

The effect of heat stress can be reduced by providing access to drinking water and either artificial or natural shelter. Approximately 3.5 – 6.5 m^2 of shade is required per animal (Laer et al., 2014). The impact of cold weather conditions can also be alleviated by providing shelter and dry surfaces (Webster et al., 2008; Graunke et al., 2011; Laer et al., 2014).The adaptation measures that are appropriate to housed livestock are also appropriate for vehicles used to transport animals. Other adaptations include improved design of vehicles, reduced stocking density (Moran et al., 2009) and overnight transport. More radical adaptations could include a return to local slaughter facilities, resulting in minimal transport distance to slaughter.

In dairy herds, there has been a move towards cows being housed for longer periods. The change in climate may increase this trend as it makes the management of both the pastures and the cows more controllable. Any increase in housing may entail costs, with increasing manure storage requirements and additional fuel associated with the harvesting of the forage and the spreading of the manure. In grazed systems, possible adaptations include reducing overall stocking rates, changing pasture species, increasing the diversity of grassland sward and breeding species that are more stress tolerant (Lee et al., 2013).

3.3.5 Trees, wood production and forestry services

Synthesis

Trees and woodland are important sources of timber and fibre. They also provide many other ecosystem services, such as biodiversity, carbon sequestration and climate regulation, runoff control and catchment water supply, and recreation and amenity, meaning there are important interactions with other topics in this risk assessment. The current main risks to forestry production are damage and loss caused by pests and diseases and wind damage during exceptional storm events, although it is not possible to directly attribute these to the changing climate with confidence. The recent increases in temperature have lengthened the growing season substantially. This, together with the increased atmospheric CO_2 concentration, may be part of the reason that tree growth rates and productivity have increased recently in Europe. However, other recent environmental changes may be contributing, but there is little evidence from the UK.

Tree growth and forest productivity will be substantially altered by climate change, affecting all these ecosystem services. In the near and medium term (2030s) it is likely that the cooler upland and wetter areas will have improved growth of many tree species, and the drier, warmer lowland areas will have only small reductions due to increasing drought. In the longer term (2050s and 2080s), trees in present cooler and wetter upland areas are likely to still show increased growth compared with current conditions. However, growth is likely to be reduced in many other locations due to increasingly severe soil water reductions, particularly on lighter soils and in the south and east of Britain. The long lead time for forestry implies that there is high urgency for action to reduce risks. For softwood conifer plantations that reach maturity in shorter rotations (40 to 60 years) there is the possibility of change to more suitable species in time (for the 2080s at least). However, as rotation lengths for broadleaved species are longer than 100 years, the opportunity to change species is much more restricted, which will be a considerable problem for future hardwood production.

The suitability of existing tree species for the future climate will change, and in many cases decline, especially in the longer term. This is particularly the case for the main species currently used in softwood production in the UK, Sitka spruce, which is a species best suited to cool and moist conditions. Species suitability will also be affected by changing pest and disease risks.

Synthesis

Damage to trees and woodlands is likely to increase from pests and pathogens and from wind storms, droughts and wildfires. Pest and pathogen damage is likely to increase because of more suitable conditions for their spread, including more environmental stresses that will make trees more susceptible, and because of new introductions. Effective monitoring, control and risk-reduction measures are essential. In part these threats are exacerbated by the limited diversity of tree species planted during the expansion of UK forests in the past 60 to 80 years. Measures to diversify the species grown should reduce these risks and increase woodland functional resilience. However, the rate at which this can be achieved is very low, and is particularly problematic in broadleaved woodlands. Risk of wildfire damage will increase in southern and eastern Britain, particularly in south-east England, because of higher temperatures, drier conditions and the mosaic of fire-prone land covers, high population density and critical infrastructure. Measures to reduce wildfire risk and damage through open habitat management and contingency planning will be of key importance.

Context and policy

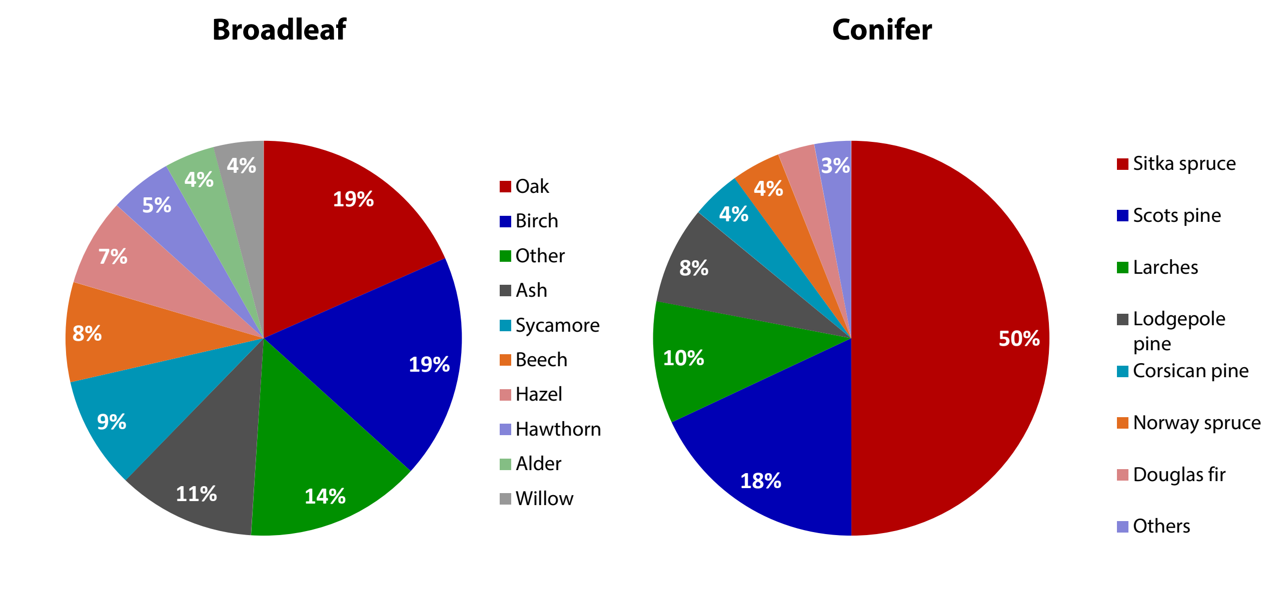
Across the UK there are 3.15 million hectares of forested land (Table 3.3) representing around 13% of total land cover (compared with an EU total of 37%). The area of woodland increased by 0.31% between 2000 and 2010, less than half the rate of the previous decade. Woodland areas have multiple functions: providing a habitat for wildlife and places for recreation, as well as supplying timber and other harvested wood products, including biomass for energy supply. Woodlands therefore provide a range of 'market goods' (traded goods such as timber) and 'non-market goods' (such as amenity benefits). They also provide key regulating services, such as carbon sequestration, climate stability, soil erosion protection, rainfall runoff reduction, and can help alleviate flood flows. While large conifer plantations in predominately upland areas form a large part of the forest area, woodlands on farms make up 28% of the total (2013 data). Most of the broadleaved woodland is in England and conifers dominate in Scotland (Table 3.2). The mix of conifer and broadleaf species is shown in Figure 3.10, which emphasises that half the conifer area is made up of one species, Sitka spruce. Oak, birch and ash make up almost half (47%) of the broadleaved area.

Table 3.2. Areas of forest in UK and countries under conifer and broadleaf species (2015) with proportion in state control or management

Country	Area (thousands of hectares)			
	Conifer	Broadleaf	Total	% public estate
England	339	965	1304	17
Wales	150	156	306	38
Scotland	1057	375	1432	33
Northern Ireland	67	45	112	55
Total UK	1614	1540	3154	28

Source: http://www.forestry.gov.uk/forestry/

Figure 3.10. Species composition of forested area in Great Britain (broadleaf and conifer)



Source: Data from Forestry Commission National Forest Inventory (NFI) in 2011 and 2013.
Note: Total areas are 1,277,000 ha for broadleaves and 1,357,000 ha for conifers.

New tree planting in 2014 covered 12,900 ha (with conifer species making up 17%) and the restocking (replanting after felling) area was 15,800 ha (73% conifers). This represents a potential change rate of 0.5% of the existing forest area per year.

UK forestry in 2014 provided exports of wood, pulp and paper to a value of £1.67 billion. This is approximately 23% of the value of imports of these same products (£7.18 billion). In 2013, 40,000 people were directly employed in forestry and primary wood processing. The sector contributed £1.9 billion in gross value added to the UK economy (Forestry Commission, 2016). British woodlands and forests also provide a wide range of non-market benefits, with the UKNEA (2011) estimating a total value of £1.26 billion per year (2010 prices). The key values were from recreation (£484 million per year) and biodiversity (£476 million per year), with large values also from landscape (£185 million per year) and carbon sequestration (£115 million per year).

Forests provide a major recreation resource: in 2015, 56% of adult survey respondents said they had visited woodland in the past few years.[18] There were an estimated 90 million visits to woodland in Scotland in 2013, which made up 23% of all outdoor recreation visits (SNH, 2014). The UKNEA estimate 250 – 300 million day visits to woodlands in the UK per year.

Only just over a quarter of the UK forest area is owned or managed by the Forestry Commission (FC), Natural Resources Wales (NRW) or the Forest Service in Northern Ireland. The proportion of woodland in the Public Forest Estate (PFE) varies from 17% in England to 55% in Northern Ireland (Table 3.3). Approximately half of UK woodlands and forest areas are owned by individuals or families, and about 13% by private institutions (Schmithüsen and Hirsch, 2010).

While most of the larger upland conifer plantations are managed with the prime purpose of softwood production, a substantial part of the lowland, typically broadleaf, forest area is in small woodlands which are not actively managed. There are estimated to be approximately 100,000 private owners of woodlands in the UK. The National Forest Inventory (NFI) indicates that 40% of GB broadleaved woodland owned by the private sector is unmanaged (NFI 2012, 2014).

Forestry is a devolved power and each administration has separate forestry policies and departments. All UK forestry policies refer to climate change risks and the role of forestry in mitigating climate change. Central to these policies is the UK Forestry Standard (UKFS, 2011) and its supporting guidelines, which include those on climate change. The UKFS provides the basis for independent certification of sustainable forest management through the UK Woodland Assurance Standard (UKWAS). The UKFS is also central to the management of the PFE, which is all certified to UKWAS, and in the requirements for planning, design and management of the private forestry sector in order to receive grant aid.

In England the *Forestry & Woodlands Policy Statement* (2013) sets three key objectives:

- Protecting the nation's trees, woodlands and forests from increasing threats such as pests, diseases and climate change,
- Improving their resilience to these threats and their contribution to economic growth, people's lives and nature,
- Expanding them to increase further their economic, social and environmental value.

Forestry Commission England has produced a *Climate Change Action Plan for the Public Forest Estate* (2011) and published a report in 2012 under the Adaptation Reporting Power. This highlighted several priority climate risks, including tree species suitability, lack of diversity, pest and disease outbreaks and increasing wildfires. It also noted several opportunities, such as use of new species and more demand for timber and wood fuel. The 2012 Plan has been reviewed and updated actions will be published later in 2016.

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[18] Public Opinion of Forestry Survey 2015 – UK & England, Forestry Commission www.forestry.gov.uk/pdf/pof2015ukeng.pdf/$FILE/pof2015ukeng.pdf,

The *Scottish Forestry Strategy* (2006) sets the policy framework for taking forestry forward through the first half of the century and beyond. Climate change is one of the '7 key themes'. Key challenges are around tree health, woodland expansion, managing invasive non-native species and biodiversity. A revised Forestry Commission Scotland *Climate Change Plan* was published in 2013.

The Welsh Government's *Woodlands for Wales* (2009) report is framed around four strategic themes, one of which is responding to climate change. The Welsh Government also produced a *Tree Health Strategy for Wales* (2013), which notes that climate change has removed some tree species from the potential portfolio.

The Northern Ireland *Forestry Strategy* (2006) is built around the sustainable management of existing woods and forests and a steady expansion of tree cover. One of the key themes of the Northern Ireland *Forestry Act (2010) Delivery Plan* for the Forestry Service is climate change, explicitly covering impacts of climate change and planting of better-adapted tree species.

Current risks and opportunities

Productivity changes

It is difficult to detect any trends in productivity in the UK that might be related to climate change. This is partly due to there being no systematic monitoring of tree growth trends in the UK. Furthermore, production data (e.g. timber harvest volumes) is determined by a range of factors including the length of rotations, total area of forest, mix of species being harvested and the state of the market. The NFI should provide more accurate information on growth changes when it completes its second five-year cycle of assessments in 2020.[19] Tree-ring studies have shown increases in radial growth across Europe, and there is limited data suggesting similar increases in England (Broadmeadow et al., 2009a; Barsoum et al., 2015). However, it is difficult to confidently ascribe this to climate change due to the effects of management practices (which are usually undocumented) and inevitable biases in sampling (i.e. only the rings of remaining trees are measured, not those that have died or been removed). The increasing forest productivity observed across Europe has been attributed to a combination of temperature increases, atmospheric nitrogen deposition and increasing atmospheric CO_2 concentrations (Broadmeadow et al., 2009a; Kahle et al., 2008). However, it is difficult to disentangle the influence of any of these particular drivers because of management changes and recovery from acid rain impacts on forest ecosystems (Kahle et al., 2008; Barsoum et al., 2015). One recent study used long-term (>100 years) experimental plot records in central Europe to show that recent Norway spruce and beech stands have grown faster than in the 1960s, particularly on more fertile sites (Pretzsch et al., 2014a). The authors attributed this to increased temperature and longer growing seasons. Unmanaged oak stands in the study area have also shown a growth rate increase, although there were fewer plots in the data set. There is no similar evidence for the UK.

Remote sensing on regional and global scales shows extension in forest canopy cover duration in deciduous vegetation in the temperate and boreal regions, both from earlier spring leafing and later autumn senescence [High confidence]. This is confirmed by other direct phenological observations that show earlier leafing and flowering in the UK [High confidence] (Sparks and

[19] For NFI reports see: http://www.forestry.gov.uk/forestry/infd-8tel28

Crick, 2015). The mean advance in spring leafing and flowering has been nearly two weeks. This may have an impact on the synchrony between the tree and the rest of the ecosystem (van Asch and Visser, 2007), including ground flora and herbivores (e.g. Oak processionary moth, Meurisse et al., 2012; Winter moth, van Asch et al., 2007; Roe deer, Plard et al., 2014). A recent review of pollinators and plant phenology responses suggests that most species have maintained their synchrony to date (Forrest, 2015), although there is a paucity of data. This may seem of limited relevance as all conifers and many broadleaved species of importance to UK forestry (e.g. oak, beech, ash) are wind-pollinated, but some such as cherry are insect-pollinated (see Section 3.2.2). The disadvantage with earlier bud burst is that it can result in increased risk of damage from late frosts, as occurred in May 2010 in England (Clark, 2013). However, there is no agreement on whether spring frost damage has increased in temperate zone forests (Augspurger, 2013; Vitasse et al., 2014).

There are reports of increased seed production related to a changing climate in some temperate tree species in continental Europe, including increased occurrence of mast years in beech in the UK (Nussbaumer et al., 2016). These increases might have significant effects on the woodland ecosystem as a food resource and for nutrient cycling, as well as possible benefits for tree seed collection.

Increased growth in warmer or longer growing seasons is dependent on sufficient moisture supplies. Tree-ring evidence clearly shows that growth can be substantially reduced in dry years (e.g. Cavin et al., 2013; Weemstra et al., 2013; Wilson et al., 2008). Drought can also reduce timber quality through cracking of stems and cause tree mortality, as was documented in 1976 and 2003 (Broadmeadow et al. 2009b; Green and Ray, 2009). Across Europe, major droughts have caused tree mortality in recent years (Allen et al., 2010) and there is increased evidence for drought-induced growth decreases, particularly in beech trees (Lindner et al., 2014; Cavin et al., 2014).

Pests and pathogens

There have been several recent examples of new tree pest and pathogen problems in the UK, many of which cause tree mortality, either rapidly or over a few years. For example, *Chalara fraxinea* (ash dieback) is a new disease in the UK that was first reported in 2012, and is now present in all English counties. By May 2016 it had been recorded in 39%, 16% and 13% of the 10 km squares in England, Scotland and Wales, respectively. A similarly new and rapidly spreading problem is *Phytophthora ramorum* development on larch. This was first observed to be infecting large areas in south-west England in 2009, and in the past five years approximately 2,000 hectares of infected larch stands have been felled in the region. It is likely that in the next decade all larch in Wales will be infected (*Tree Health Strategy for Wales*, 2013). A total of 575 sites were served with a Statutory Plant Health Notice in 2013 – 2014 because of *P. ramorum* infection, requiring a total of 4,800 hectares of woodland to be felled.[20]

Other new or growing pest and pathogen problems include *Phytophthora austrocedrae* infection of juniper, *Dothistroma septosporum* needle blight on pines (particularly Corsican pine, but also outbreaks on Caledonian pinewood sites), Oak processionary moth (OPM, *Thaumetopoea processionea*) in the London area, and Sweet chestnut blight (*Cryphonectria parasitica*) (Webber et al., 2016).

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[20] An up-to-date map of the *P. ramorum* outbreak is at: http://www.forestry.gov.uk/forestry/infd-86ajqa

It is not clear how much of this increase in forest pest and pathogen problems is climate change related, or whether it arises from the globalisation of the plant and timber trade (Webber et al., 2016). For example, OPM and chestnut blight problems were certainly caused by the import of infested or infected material, and there is no clear link to climate change. Oak decline is a growing problem in the UK (Webber et al., 2016), and a recent review has reported that 'recent forest declines have been attributed to an increase in both temperature and water stress, especially in southern Europe' (Sallé et al., 2014).

Increasing damage from grey squirrels, deer and wild boar is being reported, which is also seriously affecting woodland regeneration in some areas. While it is not clear how much of this increase is due to climate change, it is likely that warmer winters and earlier springs in particular have reduced mortality and allowed larger populations to build up [Medium confidence] (Broadmeadow et al., 2009a; Newman and Macdonald, 2015). Breeding phenology has advanced in the Red deer population on the Scottish island of Rum, which is associated with a warming climate (Moyes et al., 2011).

Invasive plant species

Invasive species such as rhododendron and *Gaultheria* are presently substantial problems for forestry, affecting ground and understorey flora, competing for water and nutrient resources and inhibiting natural tree regeneration. For example, the cost of clearing rhododendron from Loch Lomond and the Trossachs National Park has been estimated at £25 million (Invasive Species Scotland, 2015). However, there is no firm evidence that milder winters have increased the problem of these evergreen invasive species [Low confidence].

Storms

Wind damage to forests is a major problem to forestry in the UK and Europe [High confidence], where wind and snow storms cause approximately half of all damage to forests (EU, 2010; Nicoll, 2016). The Met Office has suggested that the intensity of strong winter cyclones has increased over the past century (Met Office and CEH, 2014). Table 3.3 summarises the consequences for forestry of some recent storms.

Table 3.3. Damage to forests from major storms in Europe since 1987

Date	Location and storm name	Damage (million m^3)
Oct 1987	Southern England and Northern France	12 (4 in England)
Jan to Mar 1990	England, Northern and Central Europe	110 (1 in England)
Dec 1999	Central Europe and Scandinavia (Lothar/Martin)	193
Jan 2005	Northern Europe (Gudren)	87
Jan 2007	Central Europe and Scandinavia (Kyrill)	54
Jan 2009	South west France (Klaus)	45
October 2013	Southern England (St Jude's Day)	2

Source: More information on these and other damaging storms is provided by Gardiner et al. (2013); data on 2013 storm in England from Forestry Commission NFI report (Forestry Commission, 2014).

Storms cause immediate damage (loss of timber stock, costs of clear-up) and can also disrupt markets and timber processing, as well as increase subsequent damage from insects, pests and fires. The risk of wind damage is well understood in UK forestry, particularly for conifer plantations in the uplands. The planning of rotation lengths, harvesting areas and thinning regimes usually includes measures to reduce the risk, including use of tools (e.g. ForestGALES, Hale et al., 2015) to support operational decisions.

Future risks and opportunities

Productivity changes

There are likely to be increases in tree growth rate in areas that are presently cooler and wetter due to lengthened and warmer growing seasons [Medium confidence]. Studies have estimated that, for example, Sitka spruce stand growth rates increase between 2.4 and 2.8 m^3/ha/yr for each 1°C warming. The effect of temperature increase on growth will vary depending on location, drought index and species sensitivity. Previous assessments, including that for CCRA2, highlight largely negative impacts on yield in the south and east of Britain (including much of lowland England), and generally positive changes to yield potential in the western and northern areas (at least in the short to medium term). There is, however, some uncertainty as the projections may not capture all the effects of changing climatic and soil conditions.

Warmer conditions will bring earlier bud burst in most native and non-native tree species currently important for forestry. Earlier bud burst is likely to lead to increases in productivity and growth rates. However, there are differences in the responsiveness between species with, for example, bud burst in oak and ash being more responsive to temperature changes than in beech (Vitasse et al., 2011; Vitasse and Basler, 2013; Roberts et al. 2015). Day-length is another key factor influencing bud burst, but this will not be affected by climate change (Vitasse et al., 2014). It is therefore possible that the rates of change in bud burst observed in recent decades may not continue into the future. Longer autumn periods before leaf-fall or dormancy are also likely, but as solar radiation input is less in the late autumn, the stimulatory effect on tree growth is likely to be smaller than that of earlier springs.

Growth will also be stimulated by the warmer temperatures within the season, particularly in the cooler regions. In the key evergreen conifer tree crops such as Sitka spruce, it is already the case that CO_2 uptake can occur even in late autumn and early spring if the weather is suitable, particularly if there are periods without low temperatures (Jarvis et al., 2009). This might increase the productivity advantage of some evergreen species over deciduous species in future climates.

Faster growth rates (either per tree or at stand scale) are not necessarily beneficial [Low confidence] as this may reduce timber quality unless different species (or different genotypes) are used. However, manufacture of particle board and engineered timber and use of harvested wood for bioenergy and biorefinery will be less affected by changes in wood properties because of more rapid growth. Faster growth may also cause nutrient imbalances [Low confidence].

Increased atmospheric CO_2 concentrations in the future may also increase growth rates [Medium confidence]. However, although experiments usually show a substantial increase in photosynthesis in the short term, the effect on plant growth is non-linear and depends on other environmental factors, particularly light, temperature, water and nutrient availability (Sigurdsson et al., 2013). Any effect is also highly species-specific.[21] Based on numerous experiments, there is no doubt that young trees grow faster with increased CO_2 (Jarvis et al., 2009; see also Norby and Zak, 2011) [High confidence], largely because of positive feedback with an expanding canopy.

There is very limited experimental data on the effect of increased CO_2 concentration on the growth of mature trees in forest environments. In general, the effects are usually smaller than those for young trees [Medium confidence]. Growth stimulation reduces when nutrients are limited (Norby et al., 2010), as is the situation in many semi-natural forest ecosystems. Across GB, about 24% of forested soil area can be classified as 'nutrient-poor', with a high proportion in Scotland (36%) due to the large areas of peat soils (Yamulki et al., 2012). In addition, tree nutrient status is declining across Europe, particularly for phosphorus, with a risk of deficiencies occurring in places (Jonard et al., 2015). This suggests that the size of the CO_2 growth stimulation might be restricted across a substantial area.

However, the Duke-FACE (Free Air CO_2 Enrichment) experiment with *Pinus taeda* showed growth stimulation throughout the ten years (Norby and Zak, 2011). In contrast, the Swiss tree canopy CO_2 enrichment study (150 ppm increase in ambient CO_2 concentration) in a mixed-species deciduous forest concluded after eight years that there was no change in growth (Bader et al., 2013)[22]. Similarly, the deciduous *Liquidambar* stand in the Tennessee FACE experiment showed a growth response only in the first six years, which declined over the next five, and most of the growth increase was in fine root mass, not in above-ground wood production (Norby et al.,

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21 For example, in the Swiss treeline experiment *Larix decidua*, with its indeterminate growth habit, showed a 33% above-ground growth response to increased ambient CO_2 while *Pinus uncinata* did not (Dawes et al., 2013).

22 Note that tree growth (increase in dry weight or volume) is not the same metric as net primary production (NPP), an ecological measure of productivity which measures net carbon uptake per area. NPP can increase, contributing to, for example, more litter or seed production, without necessarily an increase in tree growth. In four long-term temperate forest FACE (CO_2 enrichment) experiments, NPP increased by 23% (Norby et al., 2005).

2010). In experiments in northern Sweden on a nutrient-poor soil, growth of Norway spruce was not increased by higher CO_2 nor by air temperature (+4°C), unless nutrient supply was increased (Sigurdsson et al., 2013). These and other experimental results imply that the gradual increase in atmospheric CO_2 may not stimulate forest growth in general, but that this will depend on changes in the availability of nitrogen and other nutrients. [Medium confidence].

These experimental observations have renewed the discussion over whether tree growth should be considered to be (and modelled as) 'CO_2 supply limited' (i.e. driven by supply of carbohydrates from photosynthesis), or growth 'sink limited' (i.e. limited by the rate at which meristems in stems, buds and roots divide up to form new cells to grow with the carbohydrate available). This is a key question for the prediction of forest growth in the future and the modelling of the carbon cycle. Models that include or omit the atmospheric CO_2 increase effect on forest productivity produce very different predictions (Reyer et al., 2014; Lindner et al., 2014). In the UK situation, the paradigm of 'sink limitation' or a mixture of sink and source limitation for growth reinforces the view that future increases in temperature will have a large stimulatory effect on forest growth in upland and northern forests, and that water limitation will become particularly important in reducing growth in more southern and lowland areas [Medium confidence].

Increased CO_2 also reduces tree water loss (e.g. Bader et al., 2013). This is likely to reduce growth inhibition caused by drought (Lindner et al., 2014), although it is difficult to quantify [Medium confidence]). The reductions in water loss, even if they are not accompanied by substantial growth increases because of other limitations, may change the water balance of forested catchments, leading to higher water yield ([Low confidence] (Nisbet et al., 2011; also Gedney et al., 2006).

Tree species suitability changes

Climate change is predicted to drive tree species change across Europe [High confidence], with potentially severe economic impacts on forestry (Hanewinkel et al., 2013). Projected tree species responses in Europe have generally shown a northerly shift of range or a shift to higher altitudes (Lindner et al., 2014). It is unlikely that relatively immobile tree species will be able to keep pace with climate change naturally, apart from some species of early colonisers. However, forest managers have long planted species outside their natural ranges.

In the UK the choice of key species used in commercial forestry was based on the climate conditions prevailing during the major forestry expansions of the 20th century, and made great use of introduced conifer species (see Figure 3.11). Similarly, the continuing development and management of semi-natural and planted broadleaved woodlands largely relied upon the palette of native species that had arrived in Britain after the last ice-age. Many of these species will be less suitable for forestry (i.e. will have lower yield potential, although they may still survive) in the future warmer and drier conditions that are likely, particularly in southern and eastern areas [High confidence].

One exception is sweet chestnut, a native of southern Europe introduced in the Roman period, which is predicted to increase in suitability. However, this species may be under threat from a fungal blight disease (see above). Previous analyses have shown how species suitability for commercial timber production is projected to change with future climate (Broadmeadow et al., 2009b; Ray, 2008; ASC, 2013).

Warmer conditions may allow for the introduction of tree species that are more productive but not sufficiently hardy under present UK conditions, (Medium confidence) although the variability of recent winters probably deters planting of such species at the moment. While a warmer climate suggests there may be less winter damage from cold or from snowfall, there may be increased frost risk in the spring if leaf development is earlier (Low confidence) (Clark, 2013; Atkinson et al., 2013; Sunley et al., 2006).

Warmer autumns may also reduce hardening of buds and plant material (particularly a problem in tree nurseries and for providing planting material). The amount of seed chilling to break dormancy may also be reduced, which may affect natural regeneration (see Gosling et al., 2009), but may also lead to genetic adaptation in variable populations.

There is considerable evidence that genotypes from different latitudes (and possibly altitudes) have different germination responses to temperature, including chilling requirements (Medium confidence), which has implications for adaptation measures such as assisted migration (e.g. Midmore et al., 2015).

Drought

Any increase in the frequency and duration of prolonged drought periods will reduce tree growth (for examples, see previous section on current risks). Woodlands in the south and east of the UK and those on lighter and shallower soils will be at highest risk (High confidence). In some areas and on some soil types, summer drought stress may be exacerbated by wetter winter conditions causing waterlogging, reducing the rooting depth and thus the available soil water in summer (Low confidence). Increasing drought may cause particular problems for the establishment of young trees, either from natural regeneration or from planting.

Severe drought stress may lead to mortality and forest dieback, as was evidenced in several parts of Europe during the 2003 drought (Allen et al., 2010; Lindner et al., 2010). In addition, there are likely to be effects on street trees due to higher urban temperatures and reduced effective water availability in many urban soil substrates.

Petr et al. (2014) used probabilistic climate projections from UKCP09 with the Ecological Site Classification tree species suitability model to assess future yield class (potential productivity). They predicted substantial reductions in yield class in the 2050s and 2080s for three major tree species (Sitka spruce, Scots pine, and pedunculate oak) due to drought impacts, with reductions larger in lowland than upland sites. Some increases in potential yield were predicted in upland sites, particularly for oak. Potential production (estimated by combining yield class with present planted areas) in the state-managed forests for all three species in the 2080s was estimated to decrease due to drought by 42% in the lowlands and 32% in the uplands.

Waterlogging

Much of the more productive conifer production in the UK is in the uplands on poorly drained soils (often peat), with high water tables. In addition, many lowland woodlands are on heavy clay soils. Projected increases in winter rainfall will cause more waterlogging, which may restrict growth and reduce tree stability (Low confidence). Wet ground conditions will also restrict access for forestry machinery, reducing harvesting and thinning opportunities, and may cause damage to forest infrastructure such as roads, drains, culverts and bridges (Low confidence).

Pests and pathogens

Climate change is likely to favour the spread of some forestry pests and diseases through changing organism ranges (e.g. warmer climate may allow species distribution shifts to higher altitudes or latitudes), reducing winter mortality, increasing the number of generations in a growing season and increasing population sizes and therefore impacts (Medium confidence; Webber et al. 2016). Climate change is likely to affect the pest and pathogen organisms themselves, the host trees (and their distribution and growth) and the interaction between them (Sturrock et al., 2011); including particularly the stress status of the trees through drought or higher temperatures (e.g. Jactel et al., 2012; Telford et al., 2014), or by reducing root growth through waterlogging. Increased waterlogging and fluctuations in water tables may also cause more infection by soil-borne pathogens [Low confidence].

A meta-analysis of interactions between drought stress and pests and diseases (Jactel et al., 2012) found that water-stressed trees are less affected by damaging agents living in wood but more prone to pests and pathogens living on foliage. The authors conclude that while mild water-stress can increase secondary metabolite production and thus contribute to resistance mechanisms, severe water-stress may reduce it.

Particular insect pests as likely to increase with warmer conditions, although predicting the influence of climate change on individual species depends on understanding pest ecology and natural enemy responses (Webber et al., 2016). The green spruce aphid could increase with warmer conditions as their populations presently appear limited by cold winter temperatures (Straw et al., 2005). The pine processionary moth has been spreading northwards across Europe as temperatures increase (Battisti et al., 2005) and there are concerns that the damaging oak processionary moth (OPM) may do the same if present control measures are not successful. The spread of OPM is also likely to be influenced by the synchrony of oak leaf appearance and egg hatch. A recent outbreak of the very damaging wood-boring Asian longhorn beetle that arrived wood packaging from China was recorded in Kent. The beetle is just at the northern limit of its range in the Kent climate and consequently takes 3 years to go through its life cycle (Straw et al., 2015). If further unintentional introductions of the beetle occur, a warmer climate will reduce life-cycle duration to 2 years, which implies a faster rate of spread (Straw et al., 2015). However, not all insect pest problems are likely to increase; model calculations for the pine weevil (*Hylobius abietus*) suggest that in presently cooler areas, a rise in temperature will reduce the life-cycle from 3 to 2 years. This could reduce the cost of control through reduced insecticide use and through shorter fallow periods after felling which are used to reduce risk of damage to young transplants in areas being replanted (Wainhouse et al., 2014). Insect pests and pathogens are often associated with each other; for example, beetles and pathogens are both involved in oak declines.[23] Salle et al. (2014) predict increase in oak declines across Europe due to climate change modifying the range of ecological interactions between oaks and beetles. Predisposing factors for oak declines will also be affected by drought or late frosts, or insect attacks such as defoliations. Pests and pathogens are also more able to adapt to changing climatic conditions compared to their long-lived host (Sturrock et al., 2011).

The impact on trees from deer, squirrels and wild boar are likely to increase due to warmer winters reducing mortality, increased recruitment aided by longer growing seasons, and higher food supply resulting in larger populations (Broadmeadow et al., 2009b). However, the details of how different species will respond in different areas are not known. For example, if soil moisture deficits reduce forage production for roe deer during key reproductive periods, this may reduce

[23] Tree 'declines' are conditions characterised by a slow progressive deterioration in tree health or vigour, reduced tree growth accompanied by branch dieback.

populations (Irvine et al., 2007; Plard et al., 2014). Additional unknown factors are the response of diseases and pests within these populations, and their impacts.

Invasive species

It is suggested that climate change could increase problems with some existing invasive species [Low confidence], (e.g. DOENI Strategy for invasive alien species, 2013), as well as result in the introduction of new species. Existing invasive species such as rhododendron that compete with trees or that prevent tree regeneration might increase their elevation range in upland western areas with warmer conditions, increasing costs of control, and perhaps also increasing disease spread as rhododendron is a host for *Phytophthora ramorum*. However, there is no direct evidence of this. Changing climate conditions could make existing introduced tree species invasive, for example, by changing seed production or regeneration ability [Low confidence]. In addition, introduction of new tree species to benefit from climate change or to diversify species to increase resilience will have to guard against introduction of new species that might become invasive.

Storms

Projections of increased storm damage to trees and woodlands are highly uncertain (Lindner & Rummukainen, 2013). As discussed in Chapter 1, future changes in storm tracks and in extreme wind speeds are uncertain. The extent of damage also depends on the timing of a storm and the state of the ground, as wetter conditions will cause more tree overturning. For deciduous species, the increased wind load if trees are in leaf can contribute to damage (e.g. St Jude's Day storm in October 2013 in England). Warmer autumns, with consequent later leaf loss are likely to increase the risk of damage. Wind damage also depends on the soil wetness regime, as waterlogging reduces rooting depth and consequently tree stability (Nicoll, 2016).

Adaptation

Productivity changes

Selecting the right planting material for a particular site is key to ensuring subsequent tree establishment and productivity. Using tree planting material from different locations (termed 'provenances') is being promoted in order to better suit changing climatic conditions. The difficulty that climate change presents is choosing material that will establish in present conditions, and grow well in future changing conditions. Several research trials have shown that planting material from more southerly locations usually shows better early growth than locally adapted material (e.g. Hubert & Cundall 2006). In part this may be because of earlier bud-burst (Vitasse et al. 2009) and consequent longer growing seasons. However, there is little information on provenance growth comparisons over the longer-term and little systematic evidence across the main UK tree species on which to base provenance selection recommendations for future growing conditions. Some new multi-country trials of provenances and species have been set-up to assess provenance and species responses to different growth conditions,[24] but present results are obviously limited to establishment and early growth.

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[24] REINFFORCE project, http://reinfforce.iefc.net/

Changing productivity will require changes to forest management practices and timing. For example, where growth rates are increased, earlier thinning, harvesting and replanting regimes may be required. Where productivity decreases, longer rotations will be necessary, to supply the timber products required.Stand structure manipulation may need to change if competing ground vegetation growth increases.

Tree species suitability changes

Where new woodland is being planted, or harvested areas being restocked, there is the potential to change tree species, and information resources and tools have been developed to indicate the different species that could be used in UK conditions. Diversification is also essential to reduce pest and disease risk.

The ASC (2013) reported a recent increase in the number of conifer species being ordered from nurseries for the public forest estate in England. The 2015 Public Opinion of Forestry survey showed that 67% of respondents agreed that "different types of trees should be planted that will be more suited to future climates" (Forestry Commission, 2015). In the private sector, the take-up of diversification and other adaptation measures is limited, due in part to uncertainty. Most of the present conifer forest resource in the UK is already made up of introduced species, so further introductions should be possible, although there are severe time constraints in assessing likely performance in order to make appropriate choices for species with no history of widespread planting in the UK. Re-examination of remaining stands from past trial plantings in Britain when a range of non-native species were being assessed for performance can provide some essential information on the performance of several species that may have potential, although they were not selected for use originally.

The 2015 British Woodlands Survey on resilience (Hemery et al., 2015) identified that while most forest managers, owners and forestry professionals have seen signs of climate and other environmental change impacts, only 52% of respondents believed that "climate change is changing to such an extent that it will affect UK forests in the future", with 34% uncertain. Only 15% of the owners responding to the survey reported implementing any of the key climate change adaptation measures recommended in the UK Forestry Standard, and there was evident conservatism in owners about choosing planting material and in selecting non-native species (Hemery et al., 2015).

For semi-natural woodlands, effecting species change is even more challenging. However, it is worth noting that in North America assisted migration of tree species is being promoted for both the conservation of threatened species ("species rescue assisted migration") and at the larger scale, to maintain forest productivity, health and ecosystem services ("forestry-related assisted migration") (Pedlar et al., 2012).

The largest constraints on any change to tree species is the very slow rate of turnover in existing managed forests (with conifer rotations of 40 – 60 years, and broadleaf species over 100 years) and the slow rate of new planting. If current new planting and restock rates continue without other measures for diversification, the potential for species change is only approximately 1% of the current forest area per year.

Drought

Adaptation measures for drought risk could, in the longer-term, be changing tree species or genotypes to those with better performance and yields under drought. While work on provenance selection last century when several exotic timber species were being introduced did assess relative drought sensitivity, this was for the climate conditions at that time, and usually only examined early growth. More work needs to be done on selecting provenances for drought tolerance for future climate conditions (e.g. Eilmann et al. 2013; 2014), and to better predict species differences in drought tolerance.

A measure that can be adopted on existing stands is a reduction in tree density, either deliberately implemented by woodland managers, or occurring through increased competition, suppression and increased mortality (Fitzgerald and Lindner, 2013). Experimental work in the USA, Germany and Switzerland shows that increased thinning can reduce growth decline in dry years (van der Maaten 2013) and enhance recovery of growth from drought (D'Amato et al., 2013; Giuggiola et al., 2013; Sohn et al., 2013; Elkin et al., 2015). While there are no examples for the UK, reductions in tree density are reported to have been implemented in some forest stands in eastern France in response to water shortage.

Ground and understorey vegetation management can also improve availability of water to the trees (particularly for young trees), and is probably a necessary accompaniment to thinning measures. Planting methods may need to be modified to improve establishment success rates.

Waterlogging

Forest infrastructure specifications will need to be changed to reduce the risk from increased waterlogging.

Storms

Tools are available to help reduce wind damage from moderate storms by informing decisions on the timing of harvesting and thinning for conifer species. No such guidance is available for broadleaved species. There is little that can be done to reduce vulnerability from extreme storms, such as the 1987 storm in southern England, although contingency planning is critical to reduce immediate impacts. However, the 2015 British Woodlands Survey (Hemery et al., 2015) has shown that contingency planning among private woodland owners and managers to deal with major events such as storms and wildfires is uncommon (only 24% and 9% of owners confirmed they had contingency plans for wind and fire), even though it is recommended in the UK Forestry Standard. This lack of preparedness in the private sector is of considerable concern.

3.3.6 Wildfire

Synthesis

The risk of wildfire is identified as a cross-cutting issue for land use and management. Fire risk increases during hot, dry years and during unusually dry seasons (notably spring when dead vegetation increases fuel loads). Large-scale fires can cause severe damage to woodlands, heathland and grassland habitats, and to the species they support.

A trend towards drier summers suggests an increase in risk, with climate modelling suggesting a small increase in direct risks to the 2080s, with the greatest risk in the south of the UK. However, this modelling does not yet include indirect factors such as fuel loads and human behaviour.

Further investigation of localised risk factors linked to development of measures to improve awareness would benefit emergency planning and enhance preparations for a likely increase in future events.

Measures to reduce wildfire risk and damage through open habitat management and contingency planning will be of key importance. Although wildfire risk is recognised as an important issue within specific sectors (particularly forestry which often leads on proactive strategies), more emphasis should to be placed on developing a risk-based approach across sectors to ensure co-ordination of actions and resources. This should also include cultural heritage as wildfire can cause severe damage to designed landscapes, historic buildings, and monuments (Chapter 5).

Context and policy

Although not occurring with the same frequency or extent as in some other countries, wildfire represents a sporadic but serious risk to the natural environment in the UK. Wildfire can affect multiple sectors (notably forestry, agriculture and biodiversity) and multiple habitats (grassland, heathland, woodland and peatland).

While wildfire can damage the natural environment and forestry, with loss of timber, habitat and ecosystem services, it can also cause substantial short-term disruption to local populations and damage property and infrastructure. There are also potential health impacts from air and water pollution and even loss of life (Finlay et al., 2012).

Fire risk warnings can result in the closure of public access land, with potential economic impacts on tourist businesses. The most common locations for wildfires are adjacent to population centres. However, large fires in remote rural areas can have severe implications for the emergency services because they are difficult to access and can overstretch available resources. Wildfire is included in the UK National Risk Register and National Risk Assessment.

Wildfire risk is strongly weather related because drier conditions are more conducive to fire outbreaks. For example, on average there were 374 wildfires a day in the dry April of 2011 compared to 30 a day in January 2012 (Fire Statistics Great Britain, 2011-2012). Nevertheless, the relationship is complex because wildfire in the UK is usually caused by human actions and the severity is related to other risk factors that influence the fuel load and spread of fire, notably vegetation (type, age, dry matter, deadwood, etc.), land management and landscape structure.

Current risks and opportunities

Evidence for this risk has not substantially changed since CCRA1. Analysis of data on wildfires (Table 3.4) shows that in the four years from 2009/2010 to 20012/2013 there were over 200,000 wildfires in Great Britain, of which 26,000 (13%) were in woodland. An area of around 70,000 hectares was affected over that period. There is substantial year-to-year variation and the length of this record is not sufficient to show any trends.

Table 3.4. Total number of and area (hectares) burnt by wildfires in Great Britain

Year	Total number of fires		Total area burnt (hectares)	
	By all habitats in Great Britain	By forest and woodland in Great Britain	By habitats in Great Britain	By forest and woodland in Great Britain
2009/2010	61,184	6,621	13,095	256
2010/2011	65,300	7,994	17,565	1,333
2011/2012	61,544	9,185	37,380	8,883
2012/2013	23,550	2,503	3,490	440

Source: DCLG Incident Reporting System of wildfire attended by the England, Scotland and Wales Fire and Rescue Services analysed by Forestry Commission England.

Improvements in approaches to firefighting may have contributed to a reduction in large outbreaks in recent years, although drier summers have also been less common (Gazzard et al., 2016). Conversely, there is some suggestion that conversion to continuous cover management systems, with an increase in deadwood, forest floor litter, and so-called "ladder fuels" that facilitate fire spread into the canopy, may have increased forest fire risk (Stokes and Kerr, 2009).

For forest fires the most damaging events are in spring, when dry brash and litter from the previous growing season can provide fuel for fires. In some parts of the country, such as south Wales, fires often spread from adjoining areas supporting tussocks of dead Molinia grass.

Extended summer droughts, such as those experienced in 1976 and 1995, result in a secondary fuel source in late summer, as the ground vegetation dies and dries off. This is clearly demonstrated in fire statistics, which show peaks in years with extended summer droughts [High Confidence].

Forest fires in spring are generally surface fires, whereas in summer they can extend below the surface. When organic soils, particularly peat, are affected by fire, the damage can become extensive in depth and extent, because of the large fuel supply and difficulties of suppression, with implications also for increased carbon emissions and reduced water quality.

The conditions for a severe wildfire incident exist under present-day climatic conditions. For example, an outbreak in April/May 2011 at Crowthorne Wood and Swinley Forest (near Bracknell, south-east England) on mixed woodland and heathland used for recreation damaged 110 hectares of habitat. The incident was the most resource-intensive fire in Royal Berkshire Fire and Rescue Service's history (Oxborough and Gazzard, 2011). However, the consequences could have been much more severe if there had been a change in wind direction, as the area affected was close to high-density housing, infrastructure, commercial property and military installations.

Future risks and opportunities

Projections of drier summers with increased soil moisture deficits would suggest an increase in the number of fires and the area affected [Medium confidence]. This may be further exacerbated by possible changes in the frequency and intensity of droughts, although this is currently highly uncertain. It is also likely that weather conditions that promote wildfires will increase if there is an increased frequency of warmer and drier springs [Low confidence].

The McArthur Forest Fire Danger Index (FFDI) is projected to increase from a score of 6 in an average year currently to a score of 9 in England and Wales by 2100 (medium emissions scenario) (CCRA1). This would, however, only be classed as “moderate” risk and is considerably less than occurs with inter-annual variability. CCRA1 summarised these results for UK national parks (Table 3.5) showing the greater potential changes in the south. However, such results need to be interpreted with caution as the modelling has difficulties incorporating the fuel and weather interactions that cause a spike in wildfire incidences.

It is also important to note that such projections do not include socio-economic factors, land-use change and demographics. For example, lifestyle changes and increased uptake of outdoor leisure and recreation activities may exacerbate the risk through an increase in both accidental and malicious fires.

Table 3.5. Potential average changes in fire risk for National Parks in Great Britain

Country	National Park	% Difference in Fire Index (areal mean) (1980s-2080s)
England	Lake District	30%
	North York Moors	30%
	Northumberland	30%
	The Broads	30%
	Yorkshire Dales	30%
	Exmoor	40%
	Peak District	40%
	Dartmoor	40 – 50%
	New Forest	50%
	South Downs	50%
Scotland	Loch Lomond and the Trossochs	30%
	Cairngorms	30 – 40%
Wales	Pembrokeshire Coast	30 – 40%
	Brecon Beacons	30 – 40%
	Snowdonia	40– 50%

Notes: Values based on the McArthur Forest Fire Danger Index.

Wildfire risk is particularly likely to increase in the south and east of the UK, where there are contiguous areas of heathland and associated conifer tree stands and plantations together with high population densities and critical infrastructure. However, risk can also be expected to increase in northern parts of the British Isles, particularly on moorlands.

Warmer, drier conditions in spring and early summer (peak fire periods) are projected to increase which will also elevate wildfire risk [Medium confidence]. In the longer term (2050s onwards), if there are more frequent prolonged drought periods then this is likely to result in more extensive fire damage through a late summer fire period, with consequently hotter fires, as occurred in the 1976 drought [Low confidence].

Risk of forest fires may be elevated by increased tree mortality due to drought as well as pests and diseases (Section 3.3.5). Some tree diseases can also directly increase wildfire risk. For example, Dothistroma Needle Blight increases fuel loadings from shed needles, which may cause fire spread to the tree crown.

Adaptation

Most fires start in open habitats (grassland, heath and moorland) before spreading to woodland, so an integrated land-use–risk-management approach is essential. In addition, assessing future fire risk needs to consider the changing risks to different parts of the landscape mosaic over time (e.g. drier grasslands adjacent to young conifer stands).

Improved land management strategies for the prevention of wildfires and increased resilience to counter their spread may include:

- Controlled burning, grazing or mowing to remove fuel.
- Maintain high water tables and reduced soil moisture deficits (e.g. by blocking drainage ditches in suitable areas).
- Systematic provision of fire ponds.
- Planning for wildfire response within landscapes, including safety zones, water sources, orientation points and pre-planned control lines.
- Use of closure orders to restrict access.

Wildfire risk reduction through forest design plan changes has recently been promoted through publication of a Practice Guide for forest managers (Forestry Commission, 2014) and it is also integral to the Heather and Grass Burning Regulations and Code of Practice.

More systematic collection and analysis of data from wildfires throughout the UK is required to improve modelling and risk assessment tools.

3.4 Freshwater ecosystems and water services

Synthesis

At present, a clear climate-related trend in risk at a national scale cannot be distinguished for freshwater ecosystems and their services. This is due to the dominating role of large year-to-year climate variability and the influence of other factors (notably land use). A significant proportion of UK water bodies currently do not meet the standard of "good ecological status" according to the EU Water Framework Directive due to these multiple stresses.

Future projections for an increased incidence of warmer, drier summers are very likely to increase the risk of low flows and reduced water levels. In combination with higher water temperatures, this increases the risk of ecosystem disruption from reduced oxygen supply, thermal stress to species, reduced dilution of harmful pollutants and increased incidence of algal blooms in water bodies. Climate change would therefore provide further stress for water bodies that do not have good ecological status and may introduce new risks for water bodies that do have good status, depending on the magnitude of change.

Impacts would be exacerbated during periods of drought, although currently evidence for increased incidence of drought remains limited. The increased likelihood of more frequent periods of heavy rainfall could cause further raw water quality problems due to increased runoff/discharge of pollutants, effluents and sediments into water bodies, including elevated levels of dissolved organic carbon. In addition to environmental impacts, these problems would incur greater treatment costs for drinking water.

Risks may be further exacerbated in some catchments due to shifts towards more intensive land use, contributing greater pollution loads from diffuse sources. Depending on the rate of sea-level rise, existing freshwater aquifers may be at an increased risk of saline intrusion, with implications for drinking water supplies.

Ecosystems, particularly wetlands and woodlands, regulate and filter the flow of water through vegetation and soils (interception, evapotranspiration, infiltration, drainage, conductivity). Climate-related and human-related changes to ecosystems will therefore modify their role in buffering against extreme high flows (flood risk) and low flows, in addition to their role in water circulation and purification. Increasing evidence is available for these relationships but remains incomplete.

Action is being taken to improve water quality, but this will need to be sustained and increased in the future to manage the impacts of a changing climate. Additional action is also needed to ensure that the abstraction regime accounts for climate-driven changes in water availability and to develop viable strategies to maintain environmental flows in water-stressed areas.

An integrated, catchment-based approach to address risks and opportunities requires that both policy and science focus on key interactions between land use, terrestrial ecosystems and freshwater systems.

Further incentives may be necessary to bring vulnerable catchments back into the range of probable compliance with water quality targets. Further research and pilot studies are required to understand the benefits that may be provided from natural flood management schemes in alleviating flood risk. Most of these actions have long lead times because they require catchment-scale implementation and co-ordination to be effective. Anticipatory actions to reduce risk will be particularly important to reduce the risk of crossing irreversible thresholds in ecosystem function in water-stressed areas.

Context and policy

Climate has a direct influence on the functioning of freshwater ecosystems through the effects of temperature on water (thermal regime, including circulation and stratification of water in lakes) and through its interaction with the hydrological cycle to affect the water balance (i.e. between precipitation and evapotranspiration) resulting in changes to flow and water levels. In addition, freshwater ecosystems are sensitive to impacts on water quality which can interact with direct climatic influences to produce multiple stresses, particularly during extreme events. Rivers naturally provide hydrological habitat connectivity but this has often been reduced by man-made structures (e.g. flood defences) that disrupt the links between rivers and their floodplains. However, connectivity can also act to distribute poor water quality where this is a problem or serve as a conduit for invasive species.

As well as assessing climate risks to freshwater ecosystems, this section considers the role of the natural environment in the provision of clean water, in maintaining raw water quality (i.e. natural purification processes) and in moderating both high and low flows. Mediation of flows can be particularly important during extreme events when storage of water in soils and vegetation can help reduce flood peaks or sustain low flows during drought conditions.

Some habitats, notably wetlands (including floodplains) and woodlands, provide important ecosystem services through their natural water storage function, by modifying hydraulic surface roughness and soil infiltration properties, or by buffering and maintaining water quality. Land use, therefore, has a critical role in soil–vegetation–water interactions, as some uses can reduce the natural capacity of ecosystems to purify or store water or slow down runoff rates. Reduction of raw water quality can incur the need for increased treatment costs to meet minimum regulatory standards for drinking water and bathing water.

In combination with other influences (notably pollution risks), water flow and thermal regime can also have implications for water quality, hence quantity and quality are evaluated together as related risks. The natural role of ecosystem processes in regulating both water quality and quantity has important societal benefits but is also vital for maintaining the healthy functioning of the ecosystems themselves, therefore also supporting the other ecosystem services that these habitats provide (e.g. carbon storage).

Societal benefits of reduced risks from low or high flows and the maintenance of raw water quality are manifested in the benefits for land resources, particularly crop and livestock agriculture (Sections 3.3.3 and 3.3.4), in reduced costs for infrastructure for flood protection, water supply, or water treatment (Chapter 4) and in improved security, health and well-being benefits for people (Chapter 5).

As a consequence, complex feedback loops exist through which climate not only has direct effects on water provision as a final ecosystem service, but also has indirect influences through its relationship with ecosystem processes that both purify water and buffer extreme low or high flows.

Water also provides important cultural benefits, for example, through recreation, bathing and sense of place (Section 3.7.3). These interconnections between people and the natural environment are now increasingly recognised by the policy framework, although they are yet to be fully integrated into statutory regulation.

From a natural environment perspective, the key policy context is provided by the EU Water Framework Directive (WFD). The WFD, as transposed into national law, provides an integrated approach to sustainable water resource management and requires that all defined water bodies meet “good status” conditions (ecological and chemical) by 2015. Where this is not possible, and subject to the criteria set out in the Directive, water bodies should achieve good status by 2021or 2027. A water body can be designated as 'heavily modified' if there is a use that prevents Good Ecological Status from being achieved. In these circumstances additional mitigation measures are put in place to get as good an outcome as possible and an alternative objective of Good Ecological Potential is assigned.

The most up to date overall position for WFD compliance in the UK is summarised in Figure 3.11. The proportion of water bodies meeting good ecological status is 24% in England, 36% in Wales, 65% in Scotland and 22% in Northern Ireland.

As part of the six-year cycle of River Basin Management Planning (RBMP) procedures incorporated into WFD, water bodies are characterised in terms of existing pressures and economic uses, although climate change is not explicitly included (Figure 3.12). The main pressures identified are related to water quality, alterations in morphology, abstraction and, particularly in Scotland, changes in water levels and flows.

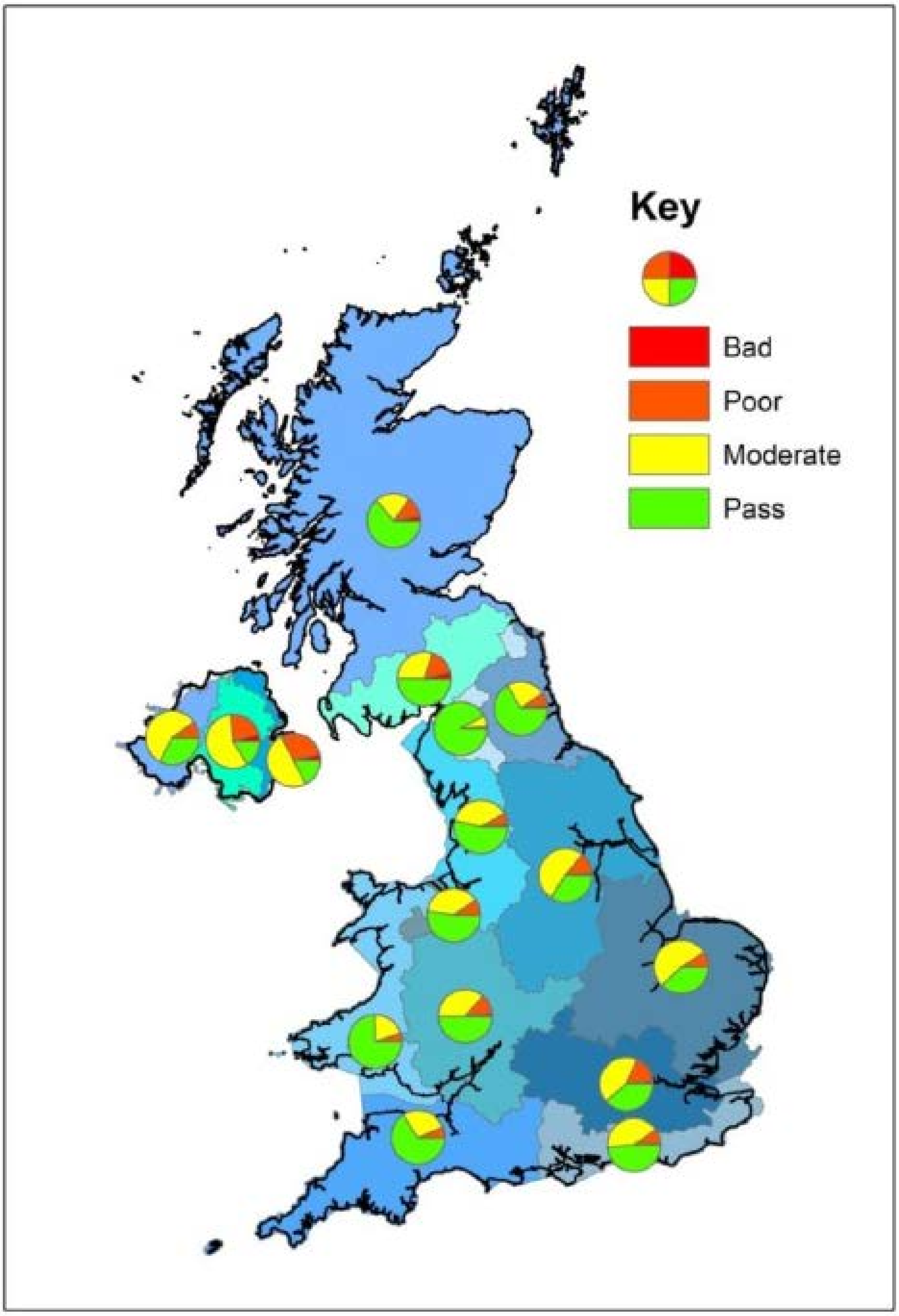
Climate change may not have been considered a major driver when the WFD was being drafted, but it now represents an important policy driver for reducing adverse effects from the headline pressures identified, as these pressures could be exacerbated by changes in thermal and hydrological regimes.

The objectives of the WFD also help deliver other important policy objectives linked to human well-being (Chapter 5) including statutory requirements defined by the Drinking Water Directive, Bathing Water Directive, Wastewater Directive and Nitrates Directive.

In addition, the EU Flood Directive (FD), as transposed into national law, provides a legislative framework for statutory assessment and mapping of flood risk, including the production of flood risk management plans by 2015 that include climate change and requirements for sustainable land-use practices. The six-year cycle of the FD is intended to be synchronised with that of the WFD through the RBMP process. The FD also explicitly refers to the benefits that may be gained from using Natural Flood Management (NFM) measures as alternatives or complements to "hard" engineering such as flood defence embankments.

Freshwater sites of high conservation importance, including rivers, lakes and wetlands, are designated under the EU Habitats Directive with an obligation to maintain "favourable conservation status". The WFD places a deadline of December 2015 to ensure the water-dependent features of SACs and SPAs have the conditions established to enable them to achieve Favourable Conservation Status. Many of these sites are also designated through the international Ramsar Convention for the protection of wetlands. Key species, such as Atlantic salmon, that are important in the notification of many European rivers under the Habitats Directive, also have a significant economic value in terms of commercial and recreational fishing.

Figure 3.11. Summary of current overall ecological status of UK water bodies



Source: Suggitt et al for ASC (2015).
Notes: Pies indicate the proportion of surface water bodies in each River Basin District (shaded) classed as 'bad', 'poor', 'moderate' of 'pass' (i.e. either 'good' or 'high'). Data were derived from: Environment Agency data for England and Wales (WFD cycle 2: predicted classification 2015); SEPA WFD cycle 2 online consultation tool (2013 data); and NIEA (2014) data.

Figure 3.12. Sources of WFD failures in (a) England and (b) Scotland

Source: Suggitt et al for the ASC (2015).
Notes: See notes for Figure 3.11 for data sources.

Current risks and opportunities

The climate sensitivity of freshwater ecosystems is particularly evident through short-term variability at inter-annual to interdecadal levels. This is typically associated with similar variability in precipitation patterns for the UK, and with a well-established link to different phases of the North Atlantic Oscillation (NAO). This inherent variability acts to confound detection of a clear long-term trend associated with anthropogenic climate change. Hence, both summer and winter flows have typically fluctuated by at least 300 – 350% between the driest and wettest years over the last three to four decades (Ormerod and Durance, as reported in Watts et al., 2012).

Some regions of the UK are generally more exposed to the amplified hydrological variability associated with the NAO: during positive phases, it can particularly exacerbate high flows in upland westerly locations and increase drought risk in lowland locations in eastern England (Strong and Maberly, 2011; Burt and Howden, 2013).

In an ecological context, localised correlations with the NAO have been identified for stream invertebrate fluctuations (Bradley and Ormerod, 2001), drought effects on salmonids (Weatherley et al., 1990) and acidification events (Ormerod and Durance, 2009). Similar NAO correlations have also been identified with surface water chemistry which can confound long-term trends (Evans et al., 2001).

In the following sub-sections we aim to distinguish relationships between the different climate-related risks acting on the freshwater environment and the resulting ecological and societal impacts. As previously noted, this is a particularly difficult task because of the combined impact of multiple stresses and the confounding effects of shorter-term "natural" variability[25], together with the differing local contexts for each system.

An additional stress on freshwater ecosystems is provided by invasive non-native species (INNS), which at present is largely a consequence of inadvertent or deliberate introductions, exacerbated by the conductivity provided by rivers and streams.

Thermal regime

Thermal regime is an important ecological influence because at higher temperatures most biological and chemical processes operate at faster rates. Many freshwater species have a specific thermal tolerance range, making them particularly sensitive to changes in water temperature.

Water temperatures have generally increased in rivers and lakes at similar rates to regional air temperatures since the 1970s or 1980s (Foley et al., 2012; Hannah and Garner, 2015; Langan et al., 2001) [Medium confidence]. In England and Wales, an average warming of 0.03°C/yr has been reported between 1990 and 2006 with 86% of monitored sites show a detectable temperature increase (Orr et al, 2014). The Scotland River Temperature Monitoring Network is a recent initiative to improve systematic monitoring of water temperatures, but it is presently too early to derive trends from these data.

[25] The influence of climate change on patterns of climate variability is also a topic of recent research (e.g. IPCC reports) although there remains considerable uncertainty.

This temperature change has modified the circulation of some lakes, particularly the process of stratification in which the thermal profile becomes more evident as a series of distinct layers, reducing circulation of water, oxygen and nutrients. Analysis at Blelham Tarn in Cumbria from 1968-2008 has shown a progressively earlier onset of stratification and later overturn (i.e. end of stratification) as the lake warmed. The duration of stratification has increased by 38 days and the presence of hypolimnetic anoxia (low dissolved oxygen conditions) has increased significantly (Foley et al., 2012).

There is some evidence of ecological responses to changes in water temperature [Medium confidence]. For example, in some catchments, this has been linked to reductions in fish species (Ormerod and Durance, 2013). In chalk streams in England, winter temperatures during the last ten years have begun to border the upper developmental limit for the eggs of both brown trout (*Salmo trutta)* and Atlantic salmon (*S. salar)* (Elliott and Elliott, 2010).

A confounding factor is that increasing stream discharge (volume of flow) appears to reduce sensitivity to warming among chalk-stream invertebrates (Durance and Ormerod, 2009), therefore the effects of warming may be masked in wetter years when discharge rates are higher. At Llyn Brianne (central Wales), spring invertebrate abundances have declined with stream warming by around 20% for every 1°C rise because species typical of cooler water conditions have been lost (Durance and Ormerod, 2007, 2010).

Changes in algal phenology in the English Lake District have been mediated both by temperature gain (Meis et al., 2009) and advances in stratification (Thackeray et al., 2008), and linked to changes in zooplankton phenology and nutrient concentrations locally in Windermere (Thackeray et al., 2013). Changes in prey use and fish food webs in Windermere have been linked to declining salmonids and increasing cyprinids (Winfield et al., 2012), with both temperature change and increased nutrients implicated.

The type of catchment can also have an influence on temperature-related sensitivity and its implications for freshwater ecology. In general, surface water-fed rivers on non-calcareous lithologies are generally more sensitive than spring-fed chalk rivers that are supplied by base flows from groundwater. However, there is presently only very limited information on temperature changes in groundwater systems.

River flows and water quantity

Long-term trends in river flows are difficult to distinguish from inter-annual variability. Rising trends between the 1960s and 1990s are evident from western and northern regions (e.g. ten-day average flows, Q5 flows[26]), but these are less significant in the context of longer hydrometric records, suggesting that they are most easily explained by the coincident and persistent positive phase in the NAO over that period [Medium confidence]. However, there is some indication that post-1960 increases in high flows were steeper than those found previously in the long-term record (Hannaford, 2015). In terms of flood risk this pattern of irregular climatic variability means that the UK has historically had flood-rich periods with higher flows (and conversely also low-flow periods). This variability needs to be factored into risk assessment together with longer-term trends due to climate change when planning for extreme events.

Seasonal flow trends have a high degree of spatial variability (Hannaford and Buys, 2012). Since the 1960s, the winter half of the year (September to March) shows increasing high flows across

[26] Q5 is a reference flow that occurs 5% of the time (ie. 5th percentile)

the whole of the UK [Medium confidence] but no obvious trend in winter low flows, except for a suggestion of decreased low flows in the west. Autumn flows have increased across much of the UK (especially central and south-west England and Wales, and eastern Scotland) but in spring there is only a weak trend towards decreasing flows (particularly lowland England).[27] No clear pattern is evident in summer. Flow trends post-1960 are consistent with changes in winter and autumn rainfall patterns but not for spring and summer: these differences are most likely due to other factors (e.g. changes in abstraction regimes) that require further research.

Furthermore, catchments of a similar type (e.g. geology, soils) that have had similar influences in terms of the variability and trend of precipitation patterns over recent decades also apparently have differing patterns of river discharge. This may be a consequence of different changes in land use (e.g. afforestation; urbanisation) although further research is required to provide a large-scale systematic analysis of different drivers for both high and low flows in the UK.

Many freshwater species are sensitive to extreme high and low flows, which can result in the simplification of food webs (Ledger et al., 2012; Ledger and Milner, 2015). Low flows usually present the greatest risk, as the quantity of water together with its temperature determines the level of dissolved oxygen available for microorganisms who derive food from organic compounds at the base of the food web; in extreme cases, streams may become ephemeral features with major implications for their ecology. Extreme high flows and associated sediment loads can damage fish spawning beds. Negative impacts from low flows have occurred in sensitive catchments during drier years but it is likely that this is also related to the level of abstraction that occurs in those catchments.

Development of a species vulnerability index for river birds has shown that vulnerability is associated with specialist traits that restrict species to in-channel riverine habitats (Royan et al., 2014). Vulnerable species include the common sandpiper, goosander and white-throated dipper that are commonly associated with fast-flowing, upland rivers and streams.

A recent estimate (Environment Agency and Ofwat, 2013) suggests that 13% of river water bodies in England and 4% in Wales are failing to support good ecological status due to abstraction. The same assessment suggests that 42% of groundwater bodies in England and 6% in Wales are also failing to meet good ecological status for this reason.

The WFD has driven the development of a nationally consistent screening approach for environmental flow requirements in the UK. The Environmental Flow Indicator (EFI) represents a precautionary estimate of the flow regime necessary to support good ecological status. EFIs are set with reference to natural flows for every river, lake or estuary water body. The difference between natural flows and the EFI equates to the amount of net abstraction considered acceptable at different levels of flow, usually between 10% and 20% below natural Q95 flow (i.e. the flow exceeded 95% of the time).

In some catchments, present-day abstractions would exceed the available resource at low flows if they were left unrestricted. This would mean that the EFI would not be met and freshwater ecology may be damaged. This is particularly the case in the south and east of England, but also in a small number of catchments in north-west England, Northern Ireland and Scotland (Figure 3.13). In many rivers today, abstraction licence constraints actively ensure that the EFIs are met and that the freshwater ecology is protected.

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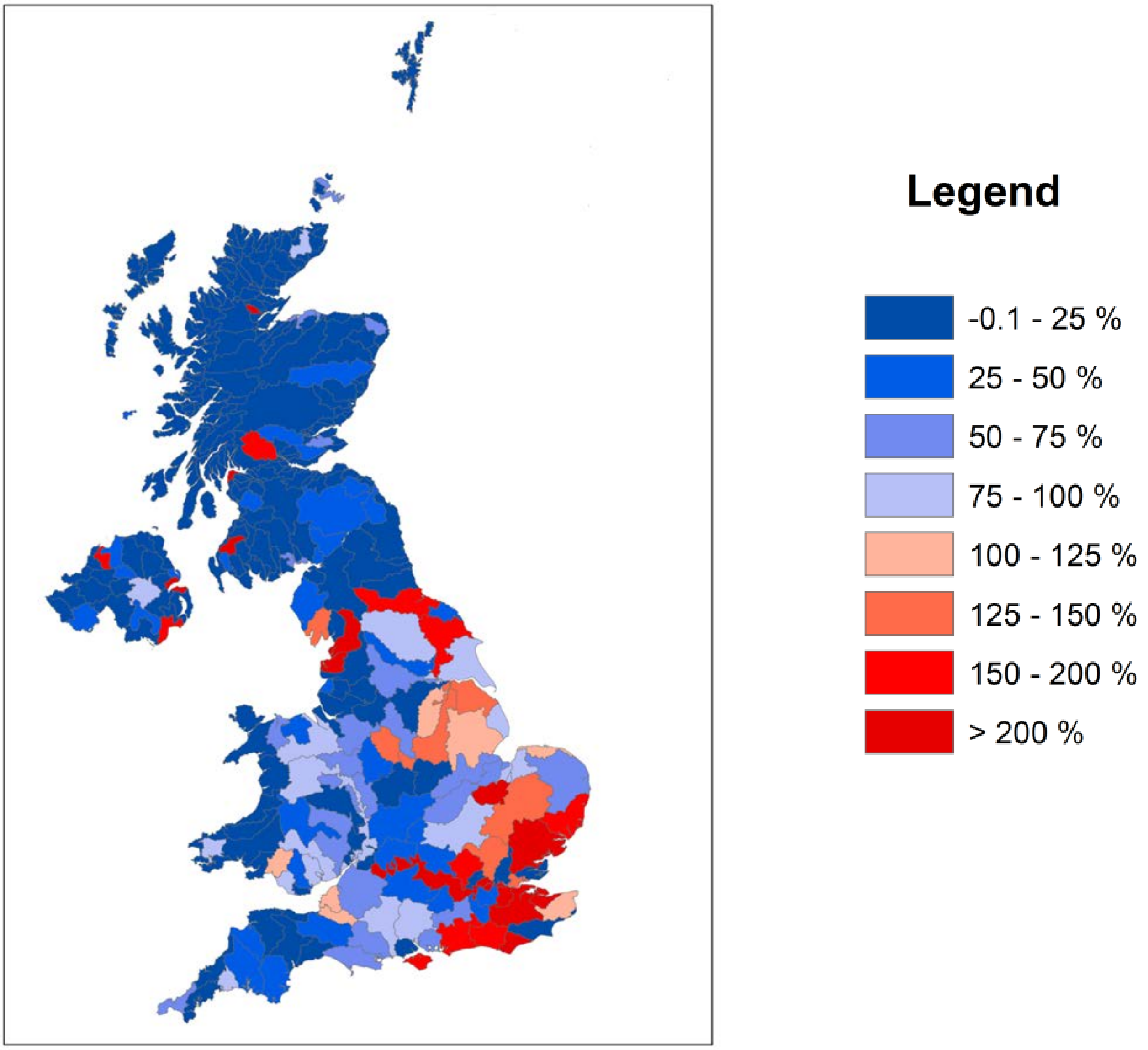
27 Over a longer timescale, many records actually show increases in spring flows from the 1930s/1940s to present.

Figure 3.13. Present-day abstraction demand as a percentage of the available resource during low flow conditions



Source: H.R. Wallingford for the ASC (2015).
Notes: Present day (2015) abstraction dema nd as a percentage of the available resource at the average of Q95 and Q70 low flow conditions.

In the coastal zone there are risks associated with saline intrusion from marine incursion into aquifers or other water bodies, which can impact on water resource availability in coastal locations. A total of 13 WFD failures attributed to saline intrusion have occurred in England and Wales, with 12 failures in Scotland; in both cases this total is relatively small compared to other sources of failure (AECOM et al, 2015, for the ASC).

Water quality

In addition to changes in thermal and hydrological regime, water quality is influenced by nutrient status, mobilisation of toxic substances and acidification potential. Water quality can be impaired by pollution from both point sources (e.g. industrial discharges, sewer outflows, treatment plants) and diffuse sources associated with land-use practices (e.g. artificial nutrients from fertilizer application, notably nitrates and phosphates). These interact with climate, particularly through the effects of heavy rainfall which can cause an increased risk of soil erosion, especially on steep slopes of bare cultivated soil (see Section 3.3.1) and runoff of pollutants into water bodies from both diffuse and point sources (notably sewer outfalls).

Resultant impacts on water quality are actually greatest during low flows due to the reduced dilution effect and greater concentration levels of pollutants; problems can also be exacerbated by flux of pollutants from heavy rain events following low flows. Water quality issues in lakes are further complicated locally by mixing processes, stratification in deeper lakes, ice cover and internal changes in dissolved oxygen concentrations.

The dominant influence on water quality in recent decades has been attributed to changes in diffuse and point pollution sources, especially in the lowlands (Howden et al., 2010). In rural areas, the shift to more intensive land use in recent decades has increased diffuse pollution from fertilisers, as well as reducing the buffering of water quality and quantity delivered by semi-natural habitats. Land management therefore has a crucial role in facilitating the interception, infiltration and storage of water rather than allowing pollutants to enter water courses through surface runoff.

There is no systematic, direct monitoring of the ecosystem processes that act to regulate water quality and quantity. For rivers, information on water quality trends from the Harmonised Monitoring Scheme (HMS), comprising 230 large river sites, shows that organic pollution indicators (oxygen demand and ammoniacal nitrogen) have universally declined since the 1980s, phosphate concentrations have reduced since the mid-1990s and heavy metal concentrations have substantially fallen. However, nitrate levels have shown little change over that time.

Nutrient runoff from land-use sources (notably agricultural fertilisers) can lead to eutrophication effects from microbial population explosions that deplete dissolved oxygen faster than it can be replaced and cause species loss at higher trophic levels as well as human health issues from toxins. A large-scale increase in cyanobacteria in the UK (and other countries) that has acted to increase the risk of algal toxins in lakes has been recorded since 1945, but mainly been attributed to elevated nutrient levels rather than temperature change (Taranu et al., 2015).

Diffuse pollution (especially nitrates) from agricultural intensification has been the main impact on UK groundwater quality (Shand et al., 2007). It has also been inferred that for coastal waters the majority of nitrate loadings and almost half of phosphate loadings are attributable to agriculture within the UK (OECD, 2012). Although fertiliser use has declined due to higher prices, with applications on grassland declining by 50% since the 1990s, it is still widely applied to arable land (AIC, 2014) which can cause problems when a large proportion of a catchment is under intensive cultivation.

Other impacts on water quality occur due to dissolved organic carbon (DOC) from the release of soil organic carbon (SOC) (see Section 3.3.1) into water courses and mobilisation of specific chemical pollutants, particularly from the legacy of past industry and mining. These pollutants all require potentially expensive additional treatment. The loss of DOC is controlled by rainfall but is also associated with solar radiation and temperature (Harrison et al. 2005. Freeman et al. (2001) reported an increase of 65% in DOC concentrations in freshwater draining from 20 sites across the UK and concluded that rising temperatures increased DOC loss from soils. Evans et al. (2005) reported a 91% increase in DOC in UK waters over a 15-year period. In a long-term study

Worrall et al. (2003) found that increases in DOC concentrations in rivers coincided with increases in mean summer temperatures. Worrall and Burt (2004) also confirmed that climate, especially severe drought, is a main driver for DOC loss. Concentrations of DOC have shown linear increases at all 22 sites covered by the UK Upland Water Monitoring Network established in 1988.

Monitoring data is also indirectly available through the distinction between "clear" and "humic" catchments in determining the relevant WFD standards for pH levels. Rates of increase are greatest in those catchments with the greatest proportions of SOC, which tend to be upland peatlands. Some of this observed increase in DOC is likely to be due to soils responding to reductions in sulphur deposition since the 1990s (Monteith et al., 2012) although it has also been suggested that burning and overgrazing may have disrupted the natural buffering capacity of the soil (Clutterbuck and Yallop, 2010).

Climate change may be implicated through changes in seasonal drying and rewetting cycles as well as fluctuations in the water table (Worrall et al., 2006) but this is often masked by other factors [Low confidence]. Furthermore, changes in atmospheric CO_2 concentrations may also be exerting an influence on plant–soil–water interactions, but this also remains uncertain.

These existing pressures make the impacts of climate change on water quality difficult to detect at the UK level [Low confidence]. Nevertheless, climate sensitivity is evident with episodes of poor water quality associated with specific weather events through both high flows (stormwater runoff) and low flows (reduced dilution of effluent discharges). Climate change effects on freshwater ecology are apparently more detectable at sites where other factors, such as poor water quality, are either absent or well understood (e.g. Durance and Ormerod, 2007, 2009; Viney et al., 2007).

Impacts of poor water quality on taxa other than fish, invertebrates and lake plankton are poorly known due to a lack of large-scale studies. Similarly climate-related impacts on ecological processes and the consequences for water-related ecosystem services remain poorly understood. Some evidence suggests that there may be elevated impacts on species at higher trophic levels (Ledger et al., 2012), and indirect effects on interspecific interactions and food availability for organisms may be more widespread than direct physiological effects through thermal tolerance (Cahill et al. 2013; Moss, 2015) [Low confidence].

Future risks and opportunities

Thermal regime

Increases in water temperatures in the future can be expected to broadly follow projected increases in air temperatures, but with local variations due to the relative sensitivity of different water bodies (e.g. buffering of groundwater-fed base flows). Reductions in flow are also likely to lead to greater increases in river temperature in summer [Medium confidence].

Similarly, change in the thermal profile of lakes is very likely to further modify seasonal circulation and stratification patterns with resultant ecological implications (including nutrient availability and dissolved oxygen levels). Smaller, shallower lakes are likely to be most at risk, with reduced circulation and increased residence times increasing the risk from cyanobacterial blooms and deoxygenation. Larger, deeper lakes are likely to be more sensitive to longer periods of stratification, reaching greater depths causing deoxygenation and loss of fish assemblages (Muir et al., 2012).

A continued decline in species adapted to cold conditions such as the Arctic char (*Salvelinus alpinus*) and those with complex life cycles such as the Atlantic salmon (*Salmo salar*) may be expected. This may be accompanied by increased prevalence of invasive fish species including the common carp (*Cyprinus carpio*), European catfish (*Silurus glanis*) and roach (*Rutilus rutilus*) (Durance and Ormerod, 2009; Britton et al., 2010; Winfield et al., 2008) [Low confidence].

These general patterns of thermal change are also likely to be substantially modified by other local factors in the catchment (e.g. land-use patterns, hydrogeology, hydropower). Therefore measures that influence these local factors by reducing existing stresses to provide more favourable conditions for natural adaptation will be particularly beneficial. For example, this can include further targeted planting of riparian woodland to enhance shading in "hot spot" locations).

There is currently rather limited information on expected future changes to ecosystem function, but deoxygenation in sensitive locations could destabilise freshwater ecosystems due to temperature rises in conjunction with reduced water flows and lake levels (Cox and Whitehead, 2009).

River flows

Future projections of river flows imply changes across the seasons as follows (all changes referenced against a 1961-1990 baseline):

- **Increases in average winter flows** [Medium confidence] (Christierson et al., 2012;[28] Prudhomme et al., 2012[29]). For the 2050s, modelled changes of -20% to +40% are indicated but with a smaller range of -20% to +20% for most of Scotland, except in the east where changes up to +40% were projected[30] (Prudhomme et al., 2012).
- **Reduced summer flows** [Medium confidence]. For the 2020s, Christierson et al. (2012) found a maximum flow reduction in August of 30% (UKCP09 median projection) with most sites having a reduction in flow even at the 75th percentile. For the 2050s, Prudhomme et al. (2012) found reductions of up to 80% by the 2050s, especially in the north and west of the UK.
- **Reduced spring flows** [Low confidence] (Christierson et al., 2012; Prudhomme et al., 2012). For the 2050s, hydrological modelling derived from the HadRM3 climate model ensemble (11 members) showed that most resulted in reduced flows (maximum -40%), but three members show increases in flow (up to +60% for central England).
- **No clear pattern in autumn flows** [Low confidence]. Prudhomme et al. (2012) found a wide range of results varying from -80% to +60%, with decreases most common in southern England.
- **Increases in the magnitude of flood events** has been modelled through the 21st century [Medium confidence for increase but Low confidence in magnitude] (Fowler and Kilsby, 2007; Fowler and Wilby, 2010; Kay et al., 2014a, b). With regard to the 1-in-20-year flood event in England and Wales, Kay et al. (2014a) indicate central estimate increases (assuming medium emissions) of 5 – 15% (2020s), 10 – 25% (2050s) and 15 – 45% (2080s), with the

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[28] Christierson et al. (2012) applied the UKCP09 probabilistic ensemble at each site.
[29] Prudhomme et al. (2012) used a rainfall-runoff mode with the HadRM3 11-member PPE (medium emissions).
[30] Christierson et al. (2012) applied the UKCP09 probabilistic ensemble at each site.

largest changes in south-east England. For Scotland, Kay et al. (2014b) indicate increases (assuming medium emissions) of 5 – 20% (2020s), 10 – 35% (2050s) and 15 – 45% (2080s), with the biggest increases in west Scotland (Argyll and west Highlands).

These projected changes imply that both high and low flows are likely to be significantly modified throughout the UK, with a greater prevalence of both winter high flows and summer low flows. Changes in flow are projected to be much greater in the higher climate change scenarios (i.e. 4°C world) compared to lower scenarios (i.e. 2°C world). Modelling of selected catchments in Scotland has shown the influence of local charactersistics on hydrological response, notably the effects in some locations of a diminished snowpack on enhanced winter flows (Capell et al., 2013, 2014).

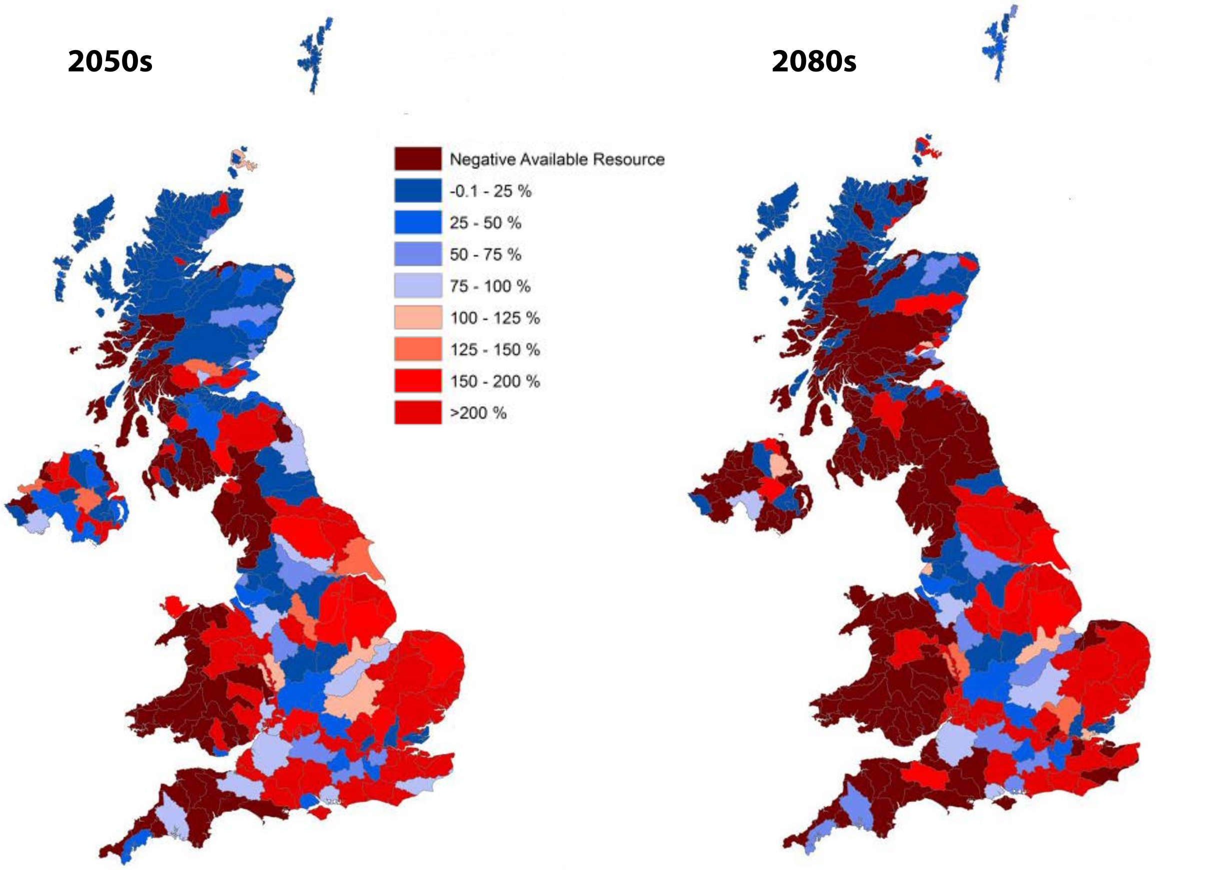
The interaction of these average trends in flow patterns with the high levels of short-term variability occurring in many UK rivers mean that predicting ecological impacts is subject to high uncertainty. In addition, such future flow projections are based upon direct climate change only and do not include indirect effects through land-use change or increased demands for water abstraction.

Although summer flows are expected to decline with climate change, a key determinant of the expected ecological consequences will be the policy taken to maintain environmental flows. If EFI thresholds are fixed at current levels, the absolute volume of water required to meet environmental requirements will remain the same despite a decline in water availability. Alternatively, environmental flows could be reduced but remain at a similar proportion relative to the reduced overall flow level. This would require less volume of water to meet EFI requirements but would be likely to have significant ecological impacts.

Decisions regarding environmental flows will have a major influence on sensitive species and ultimately the functioning of aquatic ecosystems. The potential ecological consequences of a long-term reduction in environmental flow values have not yet been investigated at a national scale.

The majority of catchments in the UK are projected to be unable to meet abstraction demand by the 2050s and 2080s under high emissions and high population growth scenarios (H.R. Wallingford for the ASC, 2015). This is assuming that EFI thresholds are fixed and there is no further adaptation beyond that already planned in water resource management plans (Figure 3.14). Under these scenarios, there would not be enough natural resources available to meet the fixed EFI requirements and therefore no additional resources available for human uses. This would especially be the case in the western half of the UK.

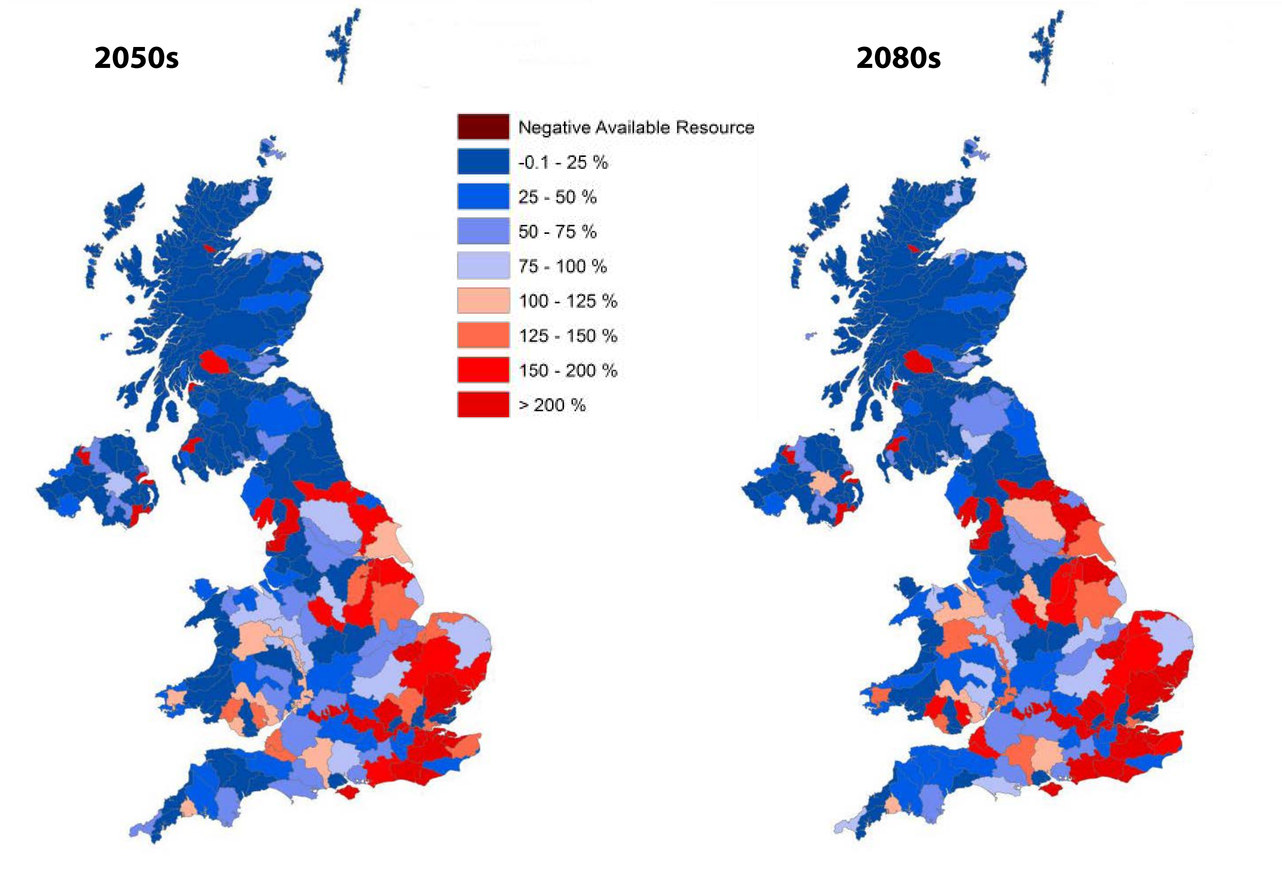
Figure 3.14. Abstraction demand in 2050s and 2080s as a percentage of the available resource during low flow conditions under a high climate scenario; high population growth; no additional action adaptation; and fixed environmental flows.



Source: H.R. Wallingford for the ASC (2015).
Notes: Negative Absolute Resources means that there is not enough natural resources to service the environmental flow requirement and therefore no resources available for human uses.

Reducing the EFI requirement so it is proportional to overall flows would mean more catchments would have water available (Figure 3.15). However, severe reductions in either environmental flows or water for abstraction (or potentially both depending on scenario) are still projected for south and east England and south Scotland (H.R. Wallingford for the ASC, 2015). The most resilient catchments are identified in north Scotland and in the west and central Midlands of England.

Figure 3.15. Abstraction demand in 2050s and 2080s as a percentage of the available resource during low flow conditions under a high climate scenario; high population growth; no additional adaptation; and proportionate'environmental flows.



Source: H.R. Wallingford for the ASC (2015).
Notes: Negative Available Resources means that there is not enough natural resources to service the environmental flow requirement and therefore no resources available for human uses.

The projections of future water availability also suggest that upland catchments are particularly at risk of facing negative resource availability (H.R. Wallingford for the ASC, 2015). This is likely to be explained by impermeable geology and lack of groundwater storage, meaning these catchments are more susceptible to short periods of below average rainfall. By contrast, lowland catchments that are supported by groundwater baseflow are more susceptible to longer periods of below average rainfall, particularly if it occurs over successive winters when groundwater recharge usually occurs. Changes in the seasonality of rainfall could disproportionately affect upland catchments.

Water quality

Projected future changes in raw water quality remain highly uncertain. The complex interaction of climate change with other factors (notably land-use change) will vary temporally and spatially due to catchment sensitivity. Analysis of six UK rivers shows that higher temperatures and reduced summer flows could increase the risk of algal blooms and increase suspended solids (Whitehead et al., 2009) [Low confidence]. Broad-scale modelling of nitrogen pollution in Scotland has shown that either increases or decreases in concentrations can be expected depending on location, season and climate scenario (Dunn et al., 2012). The same modelling study found the indirect effects of climate change, through influence on land-use change scenarios and nitrogen loads, to be greater than the direct effects of precipitation and temperature changes.

In addition, upland streams and lakes could experience altered acidification status (due to increased acid runoff), as well as increased dissolved organic carbon and turbidity [Low confidence]. Risks may be greater in western areas where increased salt deposition associated with increased storminess can delay recovey from acidification (Evans et al., 2001). Very limited information is available on changes in groundwater quality (Jackson et al., 2015).

The influence of precipitation on catchment hydrology means that shorter-term climatic variability is very likely to dominate over the next 20 years or so in combination with the influence of catchment heterogeneity. Water quality risks are most likely to be particularly exacerbated during extreme events, notably droughts, due to combined stresses in terms of temperature, low flows and water quality. Hence, the expectation is that the effects of climate change become more pronounced at vulnerable locations (particularly in association with other stresses) and in particular years which may become more predictable with seasonal forecasting (e.g. related to the influence of NAO phases).

Combined effects of land use and climate change may further exacerbate current water quality issues, including an expansion or intensification in the use of arable land in vulnerable catchments (assuming greater prevalence of low flows in summer) (Dunn et al., 2012). Although modelling work shows a reduced loss of phosphorous from soil due to lower projected runoff rates in the 2030s and 2050s, concentrations in water would not be reduced due to reduced dilution in low flows (Cooper et al., 2010) [Low confidence].

Adaptation

Planned adaptation measures may be distinguished in terms of the spatial scale at which they operate (local to regional) and their timescale to implementation. Much current adaptation planning is shorter-term and at a local scale, primarily developed on an opportunistic ad hoc basis rather than being longer-term or strategic (Muir et al., 2012).

Actions to reduce existing stresses provide an important stepping stone to enhance resilience to long-term climatic trends and extreme events: a study on the river Kennet (south England) suggested that reducing agricultural fertiliser use by 50% would provide the biggest improvement in nitrogen loads compared to lowering atmospheric sources or constructing river water meadows (Whitehead et al., 2006).

Low-regret adaptation options that have been identified include restoring natural river flows, increasing riparian shade, enhancing thermal refugia and removing thermal barriers or hot spots (Orr et al., 2015). However, as emphasised above, the feasibility of each of these options will depend on local catchment and river reach characteristics; an ongoing research and knowledge exchange programme linking monitoring data to specific interventions is required to better understand these contextual factors (Wilby et al., 2010).

Significant knowledge gaps remain in our understanding of the relative effectiveness of such interventions, indicating that further trials and monitoring are required to inform guidance for best practice.

Thermal regime

There is increasing interest in the use of riparian woodland planting to provide shade and reduce temperature stress in vulnerable and important locations (e.g. fish spawning sites). Initiatives include the Environment Agency's "Keeping Rivers Cool" project. Summer mean and maximum water temperatures are on average 2 – 3°C lower in shaded rivers versus open rivers which can create adaptation space for vulnerable species. One study on a river in the New Forest found that only 20 – 40% shading was required to keep river temperatures below the lethal limit for brown trout (*Salmo trutta*) (Broadmeadow et al., 2011). However, these initiatives are currently focused on specific priority catchments, and the full extent of new riparian planting in the UK in the context of planned adaptation strategies remains uncertain.

If adaptation in situ is unviable, as is the case for isolated lakes, then species translocation presents an alternative option. The use of translocation of vulnerable species to more resilient "refugia" environments has been trialled for coldwater fish in Scotland and the English Lake District (Adams et al., 2014).

Problems with reduced oxygen concentrations may potentially be addressed through targeted adaptation interventions such as aeration pumps if no other alternative is available (e.g. in urban areas) although this is a costly option and unlikely to be feasible as a widespread intervention.

River flows

Adaptation measures to increase ecological resilience include the restoration of natural river morphology and flow regime, including the use of riparian vegetation, to create more variable conditions and a wider range of micro-hydrological ecological niches for species as flow conditions change. When viewed as climate change responses, such actions remain in the early stages of development but there are many local examples of such modifications that have been developed primarily for other reasons (e.g. sport fishing) that could be used to improve the limited evidence base for adaptive management.

In the context of a regulatory approach, the Environment Agency and Natural Resources Wales has a Restoring Sustainable Abstraction (RSA) programme with the aim of addressing legacy abstraction issues and risks at priority sites. The Environment Agency also has water resources programme under WFD which aims to deliver environmental flows more widely in England.

There is increasing interest in the adoption of NFM schemes which maximise the use of natural fluvial and landscape features to reduce flood peaks. As most of these schemes are still in the early stages, the benefits remain to be fully established. Results from experiments in the Pontbren catchment (central Wales) suggested land-use changes (reduced grazing pressure or afforestation) could reduce runoff rates by 50% or more (Marshall et al., 2014), although caution is needed when extrapolating from any one example and these results are at a local level rather than for the catchment as a whole.

One barrier to uptake of NFM is that they require ongoing maintenance which is typically not included in capital costs to local authorities. A review of 53 different types of woodland NFM schemes ranging from 20,000 ha (SCAMP project) to small woodland blocks found that the lack of standardised monitoring meant that at present many of the reported benefits are anecdotal (JBA Consulting, 2015).

Evidence to quantify the benefits of NFM measures as a climate change adaptation measure is currently rather limited and usually based upon specific modelling studies. The evidence suggests that risk is significantly reduced but to a lesser extent in the case of extreme events (e.g. 1% annual or 1-in-100-year events) [Low confidence]. For example, analysis in the Tarland Burn catchment (north-east Scotland) shows that high levels of afforestation (> 50% woodland cover) with coniferous woodland could reduce flood peaks by up to 36% (maximum reduction) by the 2080s (depending on UKCP09 projection) for a 1-in-10-year (10% annual) rainfall event compared to the current land use, although this may also reduce summer flows (by up to 65%) which could exacerbate water quality issues (Iacob, 2015). As hydrological models involve different assumptions regarding vegetation, soil and climate parameters, further work is required to validate these assumptios against empirical data.

A challenge that occurs with NFM measures is that it isnot possible to guarantee a specific standard of flood protection, as is provided by conventional engineering design (e.g.1-in-100-year limits). The most likely adaptation pathway at present is an increased used of NFM schemes in combination with conventional engineering approaches, as either intervention by itself is unlikely to be effective in managing flood risk, even to present risk levels.

Water retention also has the potential benefit of providing increased water availability during the summer, which can be an important advantage during drier conditions and droughts. Farm storage ponds may have benefits both for water retention in winter and local water availability in summer. A recent NFU survey (2015) found that 41% of abstraction licensees in East Anglia are planning to invest in increasing on-farm reservoir capacity, or have already done so.

Current regulatory arrangements to maintain environmental flows and water quality standards will require regular review and possible revision to meet the changing reference conditions for hydrology and ecology. Actions to combat risks, including diffuse pollution and over-abstraction in the most vulnerable areas, may need to move towards more collective rather than individual arrangements to maximise resource efficiency (e.g. farm-based water-sharing schemes; catchment-scale measures to reduce runoff into watercourses).

Water quality

Negative impacts on water quality are currently addressed through targeted measures in RBMPs. Adaptation responses that more explicitly address interactions with climate change would therefore probably be best integrated with these existing catchment-based initiatives. Recent progress in addressing WFD obligations indicatesthat diffuse pollution is not as readily addressed through targeted measures as point-source pollution; and that diffuse pollution will continue to be an ongoing problem in catchments at risk.

The use of buffer strips on agricultural land to prevent pollutants reaching watercourses is becoming more widespread due to the availability of incentives primarily through the CAP). Because the effectiveness of buffer strips depends on catchment characteristics and pollution levels, the degree of buffering provided is variable (Stutter et al., 2012). Woodland (Nisbet et al., 2011) or heavy grasses (Lee et al., 2003) are often cited as providing the largest beneficial impact. Riparian buffers may also have multiple benefits by enhancing hydrological and ecological connectivity, providing water shading and reducing sediment loss and alleviatingflood risk (Stutter et al., 2012).

Several restoration projects have sought to improve the condition of upland wetlands by drain blocking (Parry et al., 2015). Monitoring to date suggests that blocking ditches leads to reduced DOC concentrations in the ditches, with an average reduction of 28% identified, albeit with considerable local variation (Armstrong et al., 2010). Data from the SCAMP project (United Utilities, 2014) suggests a slight reduction in water colour through time over a five-year period.

3.5 Coastal ecosystems and buffering of hazards

Synthesis

Coastal habitats are exposed to any sea-level rise and changes in coastal processes. Vulnerability is increased in many locations due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone ("coastal squeeze") as a natural response to sea-level rise. In addition, natural adaptive capacity is limited by reduced sediment supply due to hard coastal defences. Coastal habitats provide an important "soft" protection role by buffering wave and tidal energy, therefore loss of habitat can put additional stress on hard defences (including maintenance costs). A further issue is that if rollback does occur then adjacent terrestrial and freshwater habitats in the coastal zone are at risk unless compensatory habitat is available elsewhere.

Although there are potential opportunities for habitat enhancement and biodiversity in the coastal zone, existing stresses mean that risks presently dominate. There is evidence that the rate of sea-level rise is increasing, which is exacerbating risk; further increases in rate and magnitude are therefore considered increasingly likely. Climate change also means there is now a long-term commitment to sea-level rise which extends post-2100. For these reasons, ecosystems in the coastal zone, together with the functions and services they provide, are inferred to be at high risk.

Current policy has increasingly recognised the need to adapt to coastal change, notably through shoreline management plans. However, there are major challenges in implementing this at local level which means "hold the line" continues to be the dominant response in practice and managed realignment schemes cover less than 1% of the coast. Localised rather than strategic interventions disregard the complexity of interrelated risks in the coastal zone and can lead to inadvertent secondary consequences in other coastal locations, such as loss of habitat or increased exposure to flooding and erosion for settlements.

To address these risks, the evidence suggests more action is needed to proactively plan for and respond to a rapidly changing coastline, particularly in low-lying areas. This may require more dynamic policies to strategically protect communities together with natural, cultural and economic assets.

Coastal zone management is a cross-cutting issue that requires integration of objectives for the natural environment with those for critical infrastructure (Chapter 4) and people and the built environment (Chapter 5). In particular, co-ordinated action is required to:

- Review the strategic UK approach to coastal vulnerability and planning for the long term.
- Carry out more pilot studies into adaptive pathways for key estuaries and vulnerable stretches of coast and communities (following the Thames Estuary 2100 example).
- Support initiatives to heighten awareness of a changing risk profile on the coast and innovative responses to accommodate change.
- Conduct further research on the impacts of changing storm and wave regimes on coastal erosion and flood defence conditions, linked to a sustainable role for coastal ecosystems in moderating erosion and flood risk.
- Continue to monitor research on storm surge and sea-level rise for updated guidance.

Context and policy

Coastal ecosystems occur at the transition between terrestrial or freshwater processes and marine processes, usually comprising mosaics of habitats that are functionally interdependent. In addition to direct climate influences, these interrelationships are influenced by sediment availability and size, and therefore geomorphological processes related to wave and tidal energy, in combination with ecological processes related to salinity tolerance. For example, on barrier coasts, saltmarsh or sand dune systems may form behind a shingle ridge or sand bar that is orientated relative to the dominant wave direction.

Interrelationships in coastal systems are also very dynamic and likely to change through time as adjustments occur in response to shifts in driving forces such as wave regime, tidal currents and sediment supply. These naturally variable conditions act against human interventions that usually aim to maintain a static coastline. This is particularly the case during extreme events, when multiple factors can combine to increase water levels and produce much higher hydrodynamic forces.

Sea-level change acts as a key influence on coastal processes by modifying the position of the tidal energy frame relative to the land. Changes in sea level relative to the land are a consequence of global and regional eustatic changes in the oceans, together with local isostatic land movements. For the UK, as a legacy of ice loading from the last glaciation, the general isostatic pattern is land uplift in the north (~1 – 1.5 mm/yr maximum) and subsidence in the south (~-0.5 – 1 mm/yr maximum) (Shennan et al., 2009; Bradley et al., 2011). Eustatic changes in sea level are influenced by a warming climate due to melting land ice and thermal expansion.

The consequence of this is that sea level is rising but with local variations in the UK due to the pattern of isostasy. In addition, climate has an influence on coastal processes through the prevailing direction and location of the North Atlantic storm track and related atmospheric phenomena (including the NAO). Climate-driven changes in wave energy and the occurrence of extreme storm events can be important in influencing the balance between sediment erosion and accretion on the coastline. The occurrence of high water levels (e.g. storm surge events) can result in coastal flooding of the adjacent land.

These relationships can be further influenced by the interaction of vegetation and sediments in coastal habitats, for example:

- Seagrass beds that occur in soft sediments where there is protection from wave action (e.g. in estuaries), capturing and stabilising sediments and their nutrients for a range of species, but which are vulnerable to changes in turbidity and nutrient loading (Jones and Unsworth, 2016).
- Coastal saltmarshes that occur in the upper intertidal zone above mudflats or sandflats, and which feature vegetation communities that can trap fine sediment and allow the habitat to potentially keep pace with sea-level rise, provided enough sediment is available.
- Cliff vegetation, which to a varying extent can help resist cliff erosion by waves and intense rainfall, depending on the erosivity of rock or sediment.
- Sand dune systems, where pioneer vegetation communities act to stabilise sand movement against the effects of the wind.

As a consequence of their dynamic environment, coastal habitats and species have natural resilience if they are not constrained by space and other imposed factors, although this adaptability can depend on local site conditions and may produce unexpected changes as salinity regimes change (Hoggart et al., 2014).

In addition, the coastal zone features other habitats that may occur beyond the normal reach of marine influences, but can be affected during extreme events and are sensitive to a saline influence. These habitats are protected from tidal inundation by sea walls and regular tidal inundation does not occur, but they may be flooded during extreme events when defences are overtopped. Coastal grazing marsh is seasonally or periodically inundated pasture, usually with drainage ditches to maintain water levels that contain brackish or fresh water. Coastal lagoons occur as permanent or temporarily inundated water bodies protected by dune systems or shingle, and with a wide range of salinity regimes and specialist species.

On the more developed coastlines the habitat mosaic and distinction between marine and freshwater habitats is a consequence of the location of the coastal defences rather than a natural transition between habitats, which presents a major challenge for adaptation. Impacts of coastal defences can also occur at other locations along the coast due to modification of the pattern of longshore drift. Several notable examples exist where defences have resulted in a loss of sediment in the downdrift direction, causing increases in erosion there (and in some cases increased flood risk, if natural protection on a barrier coastline is reduced).

Species in the transitional coastal zone include those that use terrestrial, freshwater or marine environments together with specialists that use the dynamic conditions of coastal habitats to their advantage. Changes in general species distribution for terrestrial and marine species, and their attribution to climatic warming, are described in other sections (3.2.1 and 3.6.1 respectively), whilst this section specifically addresses the additional risks and opportunities that occur from coastal drivers, particularly sea-level rise.

Although coastal ecosystems provide many societal benefits (see Jones et al., 2011),[31] one of their most important ecosystem services is provided through the interaction of vegetation with tidal and wave regimes and sediments, therefore acting as a natural buffer from flooding and erosion hazards. Habitats and associated landforms (or subtidal features) contribute to coastal protection either directly by dissipating or attenuating wave energy, or indirectly through regulating sediment movements. In terms of direct protection, barrier features such as sand dunes and shingle bars can protect low-lying hinterlands from flooding if they are high or wide enough. With regard to indirect protection, saltmarsh and beaches act to attenuate wave energy and reduce its erosive force on backing features including cliffs and man-made defences which in turn can also protect low-lying hinterlands from flooding.

Dynamic coastal systems have the potential to be self-regulating in the face of rising sea levels; this can only occur if there is both an adequate supply of sediment and there is landward space for migration and adjustment of the different components relative to the tidal and wave energy frame.

For England and Wales, the ecological and geomorphological functions of the coast are incorporated into Shoreline Management Plans (SMPs) which aim to develop a long-term strategy for the sustainable management of the coast including the effects of climate change. Second generation SMPs aim to develop a strategy for the management of the coast up to 2100, each based upon distinctive littoral cells (zones of internal sediment circulation). Each SMP sets out the management policy for each stretch of coast (e.g. hold the line, managed realignment, no active intervention). SMPs are developed by Coastal Protection Authorities and decisions about the appropriate strategy for an area of the coast are made by the local authority responsible. The Environment Agency has a strategic overview role for the coast and works with Coastal Groups and Regional Flood and Coastal Committees to ensure that SMPs fairly reflect coastal processes and risks that cross local authority boundaries. SMPs are intended to guide

[31] Issues related to carbon storage and cycling are addressed with that from other environments in Section 3.2.9.

integrated planning and development on the coast, but they can meet with local opposition when proposing change to an existing 'hold the line' policy (O'Riordan et al., 2008).

SMPs have also been developed for some sections of the coast in Scotland. A National Coastal Change Assessment for Scotland is underway and due to be completed in late 2016, but data was not available for review at the time of this risk assessment.

As with terrestrial and freshwater species and habitats, conservation objectives for biodiversity are delivered through an international framework that includes the Habitats Directive (EU), Birds Directive (EU) and is underpinned by national biodiversity strategies as a contribution to the UN Convention on Biological Diversity (described in Section 3.2.1). The UK has special responsibility for seven habitats of European importance under the Habitats Directive, including coastal lagoons, grey dunes and dune slacks, estuaries, vegetated shingle banks, machair (a calcareous grassland found on shell-rich coastal substrates in north-west Scotland and a small area of Northern Ireland) and vegetated sea cliffs.

Coastal habitats have high biodiversity value, supporting a high number of species relative to their extent, for example, sand dunes, coastal shingle and maritime cliffs in England support 148 UK priority species (Webb et al., 2010). The coastal zone is also an important location for cultural heritage, particularly archaeological sites, and coastal processes can both reveal and destroy historic artefacts and features (Chapter 5). Some locations, such as Hengistbury Head (Dorset) are important both for nature conservation and for archaeology, emphasising the need for an integrated management strategy.

Current risks and opportunities

Coastal flooding events can result from extreme storm surges and wave events or in combination. The shallow North Sea is susceptible to storm surges, notably when cyclonic conditions amplify surge conditions that funnel into the southern North Sea and the Thames Estuary. Storm surges can also occur in the Irish Sea, although to a lesser extent than the North Sea but still with potentially severe consequences. The waves generated from the Atlantic Ocean result in the roughest conditions on the western continental shelf of the UK throughout the year, while more sheltered regions, for example, around south-east England, are relatively calm.

The wave climate around the British Isles has been studied in detail (see for example: Wolf and Woolf, 2006; Tsimplis et al., 2005) and has been shown to be strongly seasonal. January is the roughest winter month at most locations in an average year, although this can vary between November and March. Seasonality varies slightly from region to region. For example, spring tends to be slightly rougher than autumn in the north-west of the British Isles, but the reverse is generally true in the North Sea and the south-west approaches.

An estimated 28% of the combined English and Welsh coastline is currently experiencing erosion rates greater than 10 cm/year (Evans et al., 2004). Almost two-thirds of intertidal profiles in England and Wales have steepened over the past hundred years as a result of erosion due to the combined effect of sea-level rise and sediment depletion (Taylor et al., 2004); this is most likely caused by the presence of hard coastal defences and less dissipation of wave energy in the intertidal zone.

Long sections of the UK coastline (including 55% of the English coast – see ASC, 2013) are protected by hard engineering structures which prevent natural adjustment of coastal systems to a rising sea level, including the migration of habitats inland to remain in a similar position within the tidal frame. This "coastal squeeze" effect is most pronounced on the heavily defended coasts in the south and east of the UK. The UK NEA (Jones et al., 2011) estimated that coastal margin habitats (Sand Dunes, Machair, Saltmarsh, Shingle, Sea Cliffs and Coastal Lagoons) have declined in extent by 16% since 1945 due to development, land-use change and coastal squeeze[Low confidence].

Recent evidence suggests there has been an acceleration of sea-level rise in recent years (e.g. Rennie and Hansom, 2011), although with some uncertainty. This has particularly important implications for the natural resilience of coastal systems to accommodate change. At higher rates of sea-level rise, an increasing amount of sediment is required for the coastline to remain in a dynamic equilibrium and maintain the same horizontal profile (Orford and Pethick, 2006). However, much of the coast is depleted of sediment due to the use of coastal defences to resist erosion; fluvial sediment sources for the coast may also have been reduced in some locations due to defence structures. As a consequence the coastline is not in equilibrium with its driving forces. Where it cannot adjust by moving landward the profile is likely to further erode and steepen, which can also in turn increase nearshore wave energy.

The southern and eastern coasts of the UK have generally experienced the greatest impact from sea-level rise to date. This is partly due to the southern half of the UK having the highest rate of isostatic land subsidence, but also reflects higher rates of coastal squeeze and a relatively high proportion of soft sediments and low-lying coastline.

Northern and western coastlines, notably in Scotland, are generally less vulnerable to sea-level rise than southern and eastern coastlines because they have higher proportions of rocky coastline, although there are also some areas with soft sediments and low-lying land of high vulnerability. In addition, most land in northern UK is still experiencing postglacial isostatic uplift, which moderates the effect of rising sea levels. However, the impacts of relative sea-level rise and marine incursion of terrestrial habitats is beginning to be recorded at sites in northern Britain where there was no evidence previously (Rennie and Hansom, 2011; Teasdale et al., 2011) [Low confidence].

As a consequence of these changes, there have been significant losses of intertidal habitat area in recent decades in vulnerable locations. Some areas on the south coast of England lost 50% or more saltmarsh area between 1971 and 2001 (Baily and Pearson, 2007). Similarly, coastal grazing marsh is increasingly threatened by higher levels of salinity due to increased percolation and flood frequency. For the UK as a whole, habitat losses due to sea-level rise were estimated by the UK NEA over the preceding 20 years to be fairly small: approximately 2% for sand dunes and 4.5% for saltmarsh (Jones et al., 2011). Furthermore, some locations, notably on the west coast (e.g. Dee and Ribble estuaries, Solway Firth, Morecambe Bay) have experienced small gains in saltmarsh habitat, usually from expansion of the lower marsh onto adjacent sand or mud flats, and often aided by colonisation from common cord-grass (*Spartina anglica*), an invasive non-native species.

In Scotland, the distinctive machair habitat has been identified as particularly vulnerable to sea-level rise. The main machair areas are separated from the foreshore by systems of coastal dune ridges that provide protection from the sea, but in places the dunes have been removed by erosion. Much of the machair is not only low-lying, but in some cases below the high water mark, meaning even small changes in sea level could have a large influence on the habitat [Medium confidence].

Coastal habitats have also experienced the direct effects of climate change through changing temperature profiles, in a similar way to terrestrial and freshwater ecosystems. This has been most evident with rocky intertidal and subtidal species, where warmer-loving “southern” species are shifting northwards and colder-loving “northern” species declining (Hawkins et al., 2009). In addition, there is some evidence to suggest that high water temperatures and low dissolved oxygen have resulted in a delay in the upstream migration and poor estuary survival of adult Atlantic salmon (*Salmo salar*) (e.g. Solomon and Sambrook, 2004) [Low confidence]. Warmer coastal waters may also be increasing the risk of harmful algal blooms and hypoxia from eutrophication due to nutrient runoff from land sources.

The loss of coastal habitat has implications for coastal flooding and erosion risk to people and properties. Modelling suggests that up to 50% of wave energy is attenuated in the first 10 – 20 metres of vegetated saltmarsh (Möller, 2006), reducing the scale of artificial defences needed on the landward side. Up to 60% of observed wave reduction has been attributed to the role of vegetation on the saltmarsh and the same vegetation allows the marsh substrate to remain stable and resistant to surface erosion under a wide range of hydrodynamic conditions (Möller et al., 2014). An 80-metre width of saltmarsh has been estimated to reduce the height of seawall defence required from 12 metres to only 3 metres, resulting in capital cost savings of £2,600 – £4,600 per metre of seawall (based upon 1994 costs, see King and Lester, 1995). Using available data, the UK NEA tentatively estimated that for England this buffering role provides £3.1 – £33.2 billion worth of capital savings in sea-defence costs (Jones et al., 2011: 3) but this requires further substantiation. A recent assesment in Scotland has identified that £1.2bn of residential properties benefit from the protective function of sediment accretion (8,387 properties) (Fitton, 2015). Benefits are likely to be particularly important during extreme conditions, as experienced in many coastal areas of the UK during the winter of 2013/2014, for example, which caused severe erosion at many vulnerable locations. Kelp beds can also provide significant wave attenuation and protection of soft coasts (Angus and Rennie, 2014).

Future risks and opportunities

Although there is high certainty that sea levels will continue to rise around the UK coastline, the scale of increase is less certain. The Thames Estuary 2100 assessment, for example, planned for 75 cm (90 cm with land movement). Sea-level rise has been shown to significantly increase the frequency of extreme wave conditions in the nearshore zone (Chini et al., 2010). Recent evidence from the 2013/2014 winter storms (Wadey et al., 2015) suggests that projected changes in wave frequency and size may have been underestimated in UKCP09. Alternatively 2013/2014 may have seen anomalous extreme wave conditions from a sustained and vigorous westerly atmospheric circulation due to natural variability, as periods of extreme storminess have also occurred in the past. Similarly there is low confidence regarding future storm surge projections, but with the potential for increased storm intensity or frequency (Istorm, 2014). See Chapter 1 for further details.

The current loss of coastal habitats implies that even under relatively low levels of sea-level rise, continued and increased loss of habitat can be expected unless adaptation measures are implemented consistent with local and regional coastal dynamics. Higher rates of sea-level rise are very likely to cause much greater losses as natural resilience is exceeded. These losses are most likely to affect the intertidal zone and be exacerbated by local coastal squeeze issues. Up to 80% of intertidal habitats in England are estimated to be at risk of coastal squeeze as they are located seaward of fixed sea defences (ECI et al for the ASC, 2013). The Environment Agency estimates that 1,200 hectares of internationally protected (i.e. SAC/SPA) intertidal habitat in England and a further 500 hectares of freshwater habitat will be lost due to coastal squeeze by the mid-2020s. Natural Resources Wales have estimated that around 2,300 hectares of Natura 2000 coastal habitat in Wales will be lost by the end of the century due to coastal squeeze, of which some 260 hectares will be lost by 2025.

For coastal species, the complex topography and heterogeneous geology of the UK coastline often means that habitat patches are separated by long stretches of unsuitable habitat. This is likely to limit opportunities for species to colonise new locations as previous habitats are lost.

Difficulties in quantifying the extent of historic habitat loss at the UK level means that future projections remain tentative. The UK NEA (Jones et al., 2011) projected coastal margin habitat losses to reach 8% by 2060 [Low confidence]. However, for higher sea-level rise scenarios the potential losses may be significantly greater; the risk is then amplified due to the increased likelihood of threshold effects due to the decreased buffering role of sediment supply. Therefore, for higher sea-level rise scenarios there is a much greater risk of a catastrophic breach on a barrier coastline with consequent inundation and flooding of the hinterland (Hanson et al., 2010).

Model simulations have shown that a sea-level rise of 30 cm would result in the loss of approximately 7% of the intertidal area in the Humber estuary; this in turn would lead to 7% loss in the total biomass of macrobenthic invertebrates. The modelling also found that beach slope steepening (with resulting coarser sediments)could potentially lead to a further 23% loss of macrobenthic biomass (Fujii and Raffaelli, 2008). Modelling on a larger scale suggests that marshes with higher tidal ranges are less vulnerable (Simas et al., 2001). Even small losses of habitat could have major negative consequences for biodiversity, particularly as coastal habitats in the UK are of international importance to wintering populations of waders.

Further steepening of coastal profiles is likely to result in narrower, shallower beaches and may increase the retreat of soft cliffs due to the reduced buffering effect of the beach zone in dissipating wave energy (Dickson et al., 2007). [Medium confidence] Although there is considerable uncertainty regarding future wave regimes, relatively small changes in the direction and intensity of wave action may have major implications, as coastal landforms are highly sensitive to such variations, particularly due to the interaction of prevailing patterns of longshore drift with local tidal currents. This will vary across sediments of different sizes. Past reconstructions of coastal change have shown that even current areas of sediment accretion (e.g. nesses) which may be considered less vulnerable can be destabilised as wave regimes change, becoming areas of erosion and sediment transport rather than accretion.

At the terrestrial margin, coastal grazing marsh is also particularly vulnerable to sea-level rise and increased saline incursion, with consequences for modification of vegetation communities and implications for the large proportion of overwintering and migrating birds that use this habitat. Regular flooding may eventually cause some of this habitat to become saltmarsh, which may partly compensate for losses at the seaward margin but in turn would require identification of replacement areas for new grazing marsh.

Other high risk habitats include coastal lagoons and machair due to the increased risk of marine incursion and the high sensitivity of these habitats to changes in prevailing salinity conditions and water tables; lagoons often occupy rock basins and therefore have limited scope in adjusting to change (Angus, 2014). The dispersal ability of rare species typical of isolated coastal lagoons (e.g. sea anemones) and machair habitats is likely to be severely constrained meaning they have limited adaptation options.

Changes in hydrology due to shifts in seasonal rainfall and temperatures are very likely to result in significant changes in dune slack vegetation, and the more vulnerable locations may experience much larger changes in local hydrology (Curreli et al., 2013). For example, at Ainsdale in north-west England, modelling of falls in water tables of 1 metre by 2100 suggests there would be substantial modification of dune evolution and habitat mosaics (Clarke and Ayutthaya, 2010). There is already some evidence of drying out of dune wetlands based upon hydrology and vegetation surveys at a range of sites (Stratford et al., 2014).

As current levels of sediment depletion already constrain the natural adjustment of coastal systems to sea-level rise, a business-as-usual scenario with limited adaptation and a predominance of "hold the line" policies is very likely to reduce the buffering role that ecosystems provide in protecting against coastal flooding and erosion. When combined with higher rates of sea-level rise, there is a significantly increased risk of crossing a threshold beyond which habitat loss and coastal steepening lead to a large increase in flood risk on vulnerable coasts. This would require a major upgrade of artificial coastal defences to compensate for the loss of natural protection unless more emphasis is placed on 'softer' schemes that maintain sediment supply and allow a dynamic coastal profile. Based upon current evidence, the magnitude of risk and the identification of thresholds beyond which natural resilience is severely compromised remain poorly quantified. On barrier-type coasts, the risk is particularly pronounced due to the possibility of a barrier breach (e.g. erosion of shingle bars or sand dune ridge) and the flooding of low-lying hinterland. Current management practices for beach nourishment or mechanical reprofiling of barrier ridge elevations (e.g. Cley, North Norfolk) are likely to become increasingly unsustainable as sea-level rise increases.

Adaptation

The second generation of SMPs in England and Wales identified likely changes in coastal processes and implications for coastal management based on future projections up to 2100 (and similarly in Scotland for those areas with SMPs). The SMPs in England have aspirations to significantly increase the length of coastline being realigned, from around 1% in 2000 to 9% by the 2030s, rising to 16% by the 2080s (ASC, 2013). The SMPs therefore provide a sound basis for developing local and regional adaptation planning complemented by continued monitoring of coastal change to refine existing strategies (Nicholls et al., 2013). SMP policies have generally been agreed in the absence of detailed cost/benefit appraisal, and affordability considerations for each specific location. This means that continuing to hold current defence lines may prove to be unaffordable in practice in some locations. In other locations local funding contributions may be available to sustain coastal defences for longer period of time than the SMP policy suggests. This means that despite best intentions, SMPs may underplay the true risk of coastal flooding and erosion.

Local authorities have the flexibility to review their SMPs whenever they wish to do so, making them responsive to evolving situations on their coastlines. SMPs are regarded as living documents that can respond to changing risk scenarios. However, implementation of the policies identified by the SMPs, however, has proven to be difficult, particularly where they identify that a continued "hold the line" policy is unsustainable and that the coastline should be allowed to retreat inland (either through managed or unmanaged intervention). The current rate of implementation of managed realignment in England would need to increase five-fold from current levels for the SMP aspirations to be delivered (ASC, 2013).

Although the long-term strategy may be acknowledged by local communities and nature conservation interests, difficulties often occur in shifting from the current position of "hold the line" because, in the short-term, no alternatives are seen to satisfy the need for developmentspace or compensatory habitat (O'Riordan et al., 2008) .

In recent decades, soft engineering schemes have been increasingly used as a more flexible approach to complement the hard engineering of coastal defences, with the aim of enhancing sediment supply. Beach nourishment schemes can be used to increase beach volume, width and height and, therefore, the capacity to dissipate wave energy. However, the source of nourishment material is critical since dredging and sand and gravel extraction remove substantial amounts of sediment from coastal systems which can result in negative sediment budgets and significant habitat erosion. The sustainability of sediment transfer schemes such as beach nourishment in the face of long-term sustained sea-level rise has not yet been fully addressed, especially as increasing volumes of material will be required for transfers unless complemented by other initiatives.

Managed interventions through coastal realignment and associated schemes (e.g. regulated tidal exchange) have been implemented at some locations but only cover a small proportion of the UK coastline, often on formerly reclaimed land (currently there are over 100 schemes recorded in the OMREG database). This means that the dominant response remains either "hold the line" on defended coastlines, particularly for the south and east coasts, or of non-active intervention on less developed coastlines, typically in the north and west (excluding major estuaries).

Saltmarsh is capable of trapping available sediment for future habitat provision, particularly in estuaries with high sediment loads (sedmentation rates of up to 9mm/yr have been recorded: e.g. Teasdale et al., 2011). However, coastal realignment increases the tidal prism of the estuary and can modify tidal flow both locally to the site and in the estuary as a whole (French, 2006). Increased tidal flows result in the widening and deepening of creek systems and can lead to the erosion of existing marsh. Large schemes can cause major changes in the tidal volume of the estuary and increase tidal scour (French, 2006). Where realignment or similar schemes have occurred, the sediments of restored saltmarshes have been found to be less oxygenated than those at corresponding elevations of natural marshes (Davy et al., 2011). Poorly oxygenated sediments may inhibit plant colonization (French, 2006), and may result in greater nitrous oxide emissions than natural marshes and higher methane production but restored sites can also sequester carbon and reduce estuarine nutrient loads (Ding et al., 2010).

There remains uncertainty regarding the functioning of restored systems and the ability of sites to provide ecosystem services, meaning that it cannot be assumed that managed realignments will provide a like-for-like replacement for natural marshes (Mossman et al., 2012).

Coastal realignment to date has been focused on fine sediment habitats with the creation of saltmarsh and mudflat. However, coastal management aimed at reconnecting sand dunes and shingle structures to their source areas would also be likely to have considerable local benefits in vulnerable locations (Jones et al. 2011), as for example when sediment has been supplied in the past by areas of erosion elsewhere in the same littoral cell. Responses to 'coastal squeeze' in terms of providing compensatory habitat need to consider the type and location of habitat lost and gained. Losses of mudflat and lower saltmarsh are not adequately compensated by a realignment that forms upper saltmarsh and results in very different plant and animal communities.

3.6 Marine ecosystems

3.6.1 Marine species and habitats

Synthesis

Climate-related impacts are already evident in the marine environment through changes in water temperature and chemistry. The severity of impacts are expected to increase in the future and place further pressure on already stressed marine ecosystems.

Although there is high confidence that climate change is causing physical changes to the marine environment, the ecological impacts are much harder to assess because of the complexity of ecosystems and the potential for threshold effects when key species are lost. Higher water temperatures and increasing acidity could result in significant changes to the base of the marine food chain, possibly with major implications for fisheries and biodiversity. Given both the degree of uncertainty and the potential scale of the impacts, high priority should be given to further research.

Risks may be further exacerbated by marine pathogens and potentially by the introduction of invasive non-native species, whose spread is linked to increased marine transport globally, in addition to climate change.

Action is being taken to reduce adverse pressures, such as pollution and damaging fishing practices, and to increase the extent of protected marine sites. These steps should enhance the resilience of marine ecosystems to climate change, but it will also be necessary to continuously monitor and assess whether new, or additional, actions will be needed in the future. Continued surveillance and international co-operation to address the expected increased risks from marine pathogens and invasive non-native species will be required.

Context and policy

Virtually every issue that has been raised in this chapter for terrestrial, freshwater and coastal ecosystems has a direct parallel in the marine environment.[32] In the past, marine issues have received relatively little attention in national and international climate change risk assessments, although this situation is now changing and two chapters of the latest IPCC AR5 working group II report focus explicitly on climate change in the oceans, with a dedicated special report on the oceans and cryosphere scheduled to be published by the IPCC in 2018.

Marine issues do not only relate to the natural environment. There are also infrastructure, transport and energy issues (Chapter 4), human health issues (Chapter 5) and business issues (Chapter 6).

The knowledge base with regard to marine climate change impacts and adaptation has been significantly advanced in recent years through the UK Marine Climate Change Impacts Partnership (MCCIP) and its periodic report cards that collate information from the scientific community and make it available to policy-makers and industry practitioners.

The Marine and Coastal Access Act (2009), Marine (Scotland) Act (2010) and the Marine Act (Northern Ireland) 2013 set out how marine activities will be managed across the UK. The

[32] Issues related to marine carbon storage and cycling are addressed with that from other environments in Section 3.7.1.

legislation requires the development of a Marine Policy Statement to inform statutory marine plans. The first Marine Policy Statement was published in 2011, and requires that "marine plans and decisions consider how activities in the marine environment can adapt to the impacts of climate change". Statutory marine plans will be reviewed at least every three years for progress reporting, enabling new evidence on climate change to be included. Marine plans for all areas around England should be in place by 2021. In Scotland a National Marine Plan (NMP) was laid before Parliament in 2015, which explicitly considers actions needed for climate change adaptation.

The EU Marine Strategy Framework Directive (MSFD) of 2008 establishes a framework within which member states are required to "take the necessary measures to achieve or maintain good environmental status" (GES) in the marine environment by the year 2020 at the latest. Good environmental status is determined on the basis of a set of "descriptors" and the first of these (Descriptor 1- biological diversity) requires that the distribution and abundance of species are in line with prevailing "physiographic, geographic and climatic conditions". In December 2012, the UK government published an 'initial assessment' of UK seas including characteristics, targets and indicators of GES. In August 2014, the government published proposals for a co-ordinated monitoring programme for the ongoing assessment of GES, largely making use of existing statutory and voluntary monitoring schemes. The UK Government has recently consulted on a programme of management measures to meet the objectives of the MSFD. Over the next five years the MSFD indicators will be used to show whether good environmental status is being achieved in UK waters. The MSFD has been written with the explicit knowledge that marine systems are dynamic and it includes adaptation and exception sections that require climate and environmental variability be taken into account.

As of August 2015, approximately 16% of UK waters (the Exclusive Economic Zone, encompassing 773,676 km^2) are occupied by Marine Protected Areas (MPAs). The first statutory Marine Nature Reserve in the UK was established in 1986; and, since this time, legislation relating to MPAs has proliferated with a number of different designation mechanisms available (e.g. under the EC Birds and Habitats Directives, the UK Marine and Coastal Access Act, the Marine (Scotland) Act). Together, these designations contribute to the aim of establishing an 'ecologically coherent network of marine protected areas', and hence to delivering resilience against climate change. These MPAs together protect a representative range of habitats and species in the seas. Further designations are expected in the coming years, with Defra commited to designating at least 25% of English waters as MPAs by 2020. Marine Conservation Zones (MCZs) are being identified in Welsh inshore waters, with the first MCZ designated in 2014. Strangford Lough has been designated as Northern Ireland's first MCZ while in Scotland the Nature Conservation MPA network consists of 30 MPAs. In many cases, designation documents for individual MPAs include a brief assessment of future threats associated with climate change, but these are usually fairly high-level and generic.

The relevant national climate change adaptation strategies and plans in the UK nations include a number of commitments of relevance to marine ecosystems. For example, the National Adaptation Programme (NAP) (2013) encompassed 15 maritime actions, including: developing a programme of measures to achieve Good Environmental Status in waters, where the Programme of Measures will take account of 'prevailing conditions' including climatic changes; reviewing marine environmental monitoring and surveillance programmes to assess whether they are well suited to the detection of changes in state that can be associated with climate change; funding a 5 year research programme on ocean acidification, with the aim to increase the understanding and reduce uncertainties; and to publish a new report card summarising the latest evidence on impacts of climate change on the UK's marine environment. Similarly, the 2014 Scottish Climate Change Adaptation Programme included a number of marine focussed actions, including: to continue support for the Marine Climate Change Impacts Partnership (MCCIP); for the National Marine Plan (NMP) to set out objectives and policies in a way best calculated to mitigate, and adapt to, climate change; and to develop Regional Marine Plans (RMPs) from 2014 that shape regional objectives and policies for coastal and marine management and include policies relating to climate change adaptation.

Current risks and opportunities

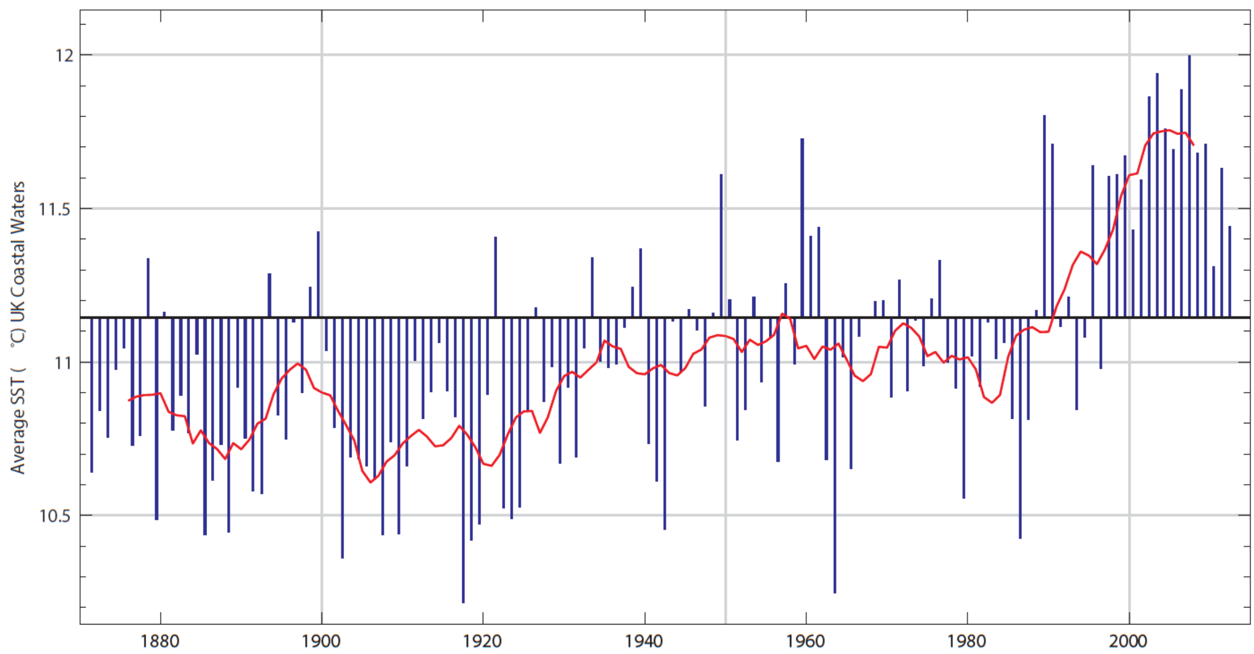
Climate drivers

The marine environment is influenced by a combination of atmospheric and oceanic drivers. Climate variability and long-term climate change impacts therefore manifest themselves differently than they do on land.

- **Temperature:** Sea-surface temperatures (SST) in UK coastal waters and in the North-east Atlantic have risen by between 0.1 and 0.5 ˚C/decade since the 1980s (Figure 3.16; Figure 3.17). Superimposed on the underlying upward trend through the 20th and early 21st Century are decadal variations with relative maxima around 1960 and in the 2000s and a relative minimum in the 1980s. Between 2008 and 2013, sea surface temperatures in most areas around the UK levelled off or were slightly lower than observed in 2003-2007 but temperatures in 2014 were again relatively high. The observed temperature changes have been due to a combination of global climate forcing and natural variability in the ocean–atmosphere system. The Atlantic Multidecadal Oscillation (AMO) is thought to be a representative pattern of the internal variability; decadal scale variations observed in UK waters share similarities with features of the progression of the AMO.
- **Salinity:** The salinity of the upper ocean (0 – 800 metres) to the west and north of the UK has generally been increasing over the last 3 decades following a particularly fresh period (with relatively low salinity) in the 1970s. A relative minimum occurred in the mid-1990s and present-day conditions have tended to be saline with evidence of a freshening signal coming from the North Atlantic since 2013. In the North Sea salinity is influenced by relatively salty North Atlantic water flowing in from both the north and south balanced against fresh water from river runoff. The variations over periods of 2 -10 years are large relative to any apparent trend since the 1970s. The western English Channel is influenced by North Atlantic water, tidal currents and local weather conditions. There is no discernible long-term trend in over a century of observations, but in recent years salinity has been higher than average. Since the mid-1960s the salinity of the Irish Sea has exhibited no significant long-term trend. Maxima occurred in the late 1970s and late 1990s; present conditions are close to the long-term mean.
- **Storms and wave height:** Waves and storm-force winds are a significant feature of UK waters, particularly in autumn and winter. In the UK, wave data have only become routinely available since the 1950s. All wind and wave time series show a great deal of variability including inter-annual and inter-decadal fluctuations. However in some notable cases a distinct trend has been observed within the wider variability. Bacon and Carter (1991) inferred an increase in mean wave height "over the whole of the North Atlantic since 1950" from observational data. Similarly, Vikebø et al. (2003) using mainly Norwegian sources, reported a positive trend in mean significant wave heights (~4cm/yr) starting in the 1960s, mainly in northern parts of the North Sea. Wang et al. (2012) produced a reconstruction of

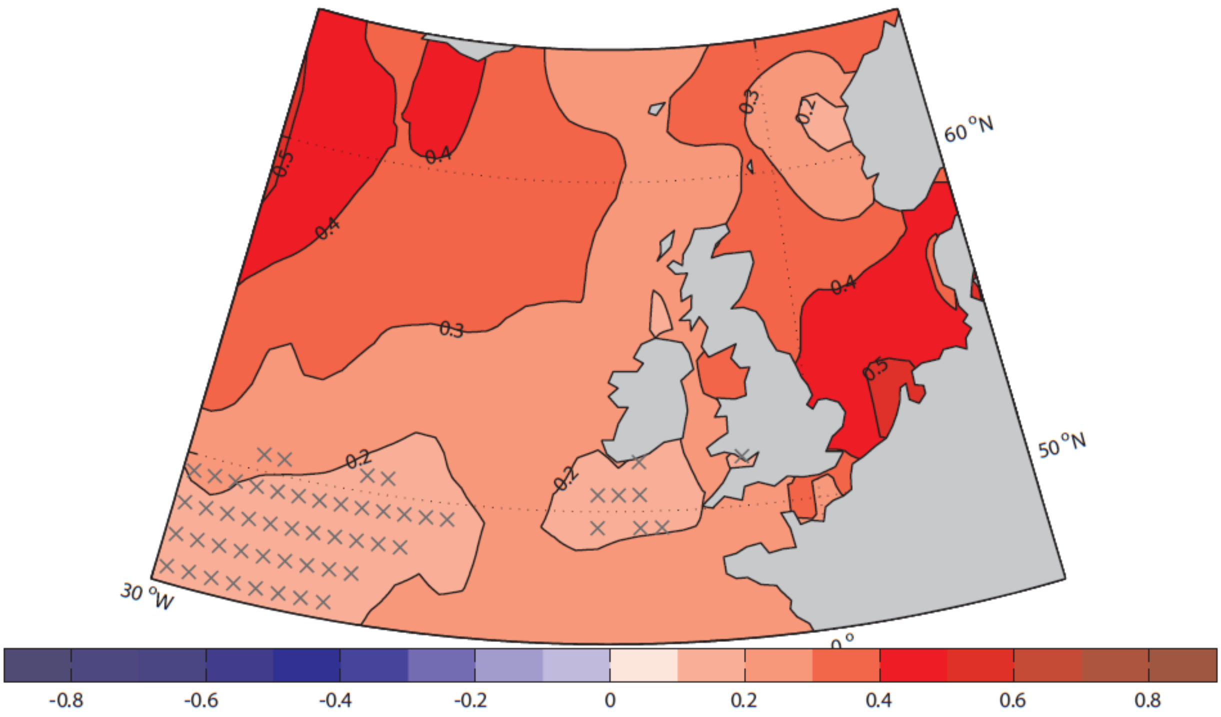
North Atlantic wave heights from 1871-2010, based on a 56-member hindcast ensemble of SLP (sea level pressure) measurements/model outputs. This study, suggested a small positive long-term trend in wave heights. Although no individual storm can be regarded as exceptional, the clustering and persistence of the storms experienced during the winter of 2013/2014 was highly unusual (Matthews et al., 2014). A report published by the UK Met Office and the Centre for Ecology and Hydrology provided an exhaustive summary of these storms, their effects and the likely causes (Slingo et al., 2014). For the UK, the single largest indicator of winter atmospheric circulation, including strength and track position of storms, is the NAO, defined as the difference in atmospheric pressure between the Azores and Iceland. Studies show that over the last century there has been a significant increase in the intensity of strong winter storms for the high latitude North Atlantic region reaching to the north of the UK (Woolf and Wolf, 2013).

Figure 3.16. Time series of average sea surface temperature (SST) in UK coastal waters



Source: Figure 4 in Dye et al., 2013a.
Notes: Blue bars show the annual values relative to the 1971-2000 average and the smoothed red line shows the ten-year running mean. Data are from the HadISST1.1 data set.

Figure 3.17. 30 year tend in annual average sea surface temperature (°C/decade) from 1983 to 2012



Source: Dye et al., 2013a.
Notes: Data are from the HadISST1.1 data set . Hatched areas have a slope which is not significant at the 95% confidence level.

- **Acidification:** Ocean acidification is closely linked with climate change as they share the same driver: increasing atmospheric CO_2. Ocean uptake of CO_2 has increased surface ocean hydrogen ion concentration by ~30% to date, and decreased surface carbonate ion concentration by ~16% globally. These effects are expected to greatly intensify in the next 100 years unless strong and urgent mitigation measures are taken at the global scale (Williamson et al., 2013). Ocean acidification is usually measured as a long-term decrease in the pH of surface ocean seawaters. pH is a logarithmic scale that reflects the concentration of hydrogen ions, and ranges from highly alkaline (base) pH 14, to highly acidic pH 1. At present, seawater around the UK tends to range from pH 7.9 to pH 8.2 (see Ostle et al., 2016). It is highly likely that UK coastal waters, ecosystems and habitats will be significantly impacted this century if global CO_2 emissions continue to rise. Measurements at ocean time series stations together with transect-based observations confirm long-term changes in the ocean carbonate system, indicating an anthropogenically induced surface pH decrease of ~0.002 units per year since 1990 (Doney et al., 2009; Orr, 2011; Bates et al., 2012). In the North Atlantic, there have been marked spatial differences in the rate of pH decrease over that period, with ocean acidification apparently occurring more rapidly in the European region (on-shelf and off-shelf) than either in the Caribbean or central Atlantic (Schuster et al., 2009). Superimposed on the relatively consistent large-scale and long-term trends, many marine systems – particularly those in economically important coastal and shelf seas – show high temporal and spatial variability in carbonate chemistry (Hofmann et al., 2011). In UK and European shelf seas, both observations and modelling show that CO_2 levels in near-surface seawater can currently vary between 200 – 450 ppm, contributing to a pH variability of as much as 1.0 (typically 0.3 – 0.4) over an annual cycle (Blackford and Gilbert, 2007; Artioli et al., 2012). Overall however, collated pH measurements from waters around the UK seem to suggest a long-term decline over the past 30 years. North Sea pH has decreased at a rate of around 0.0035 pH units per year (Ostle et al., 2016).

- **Oxygen concentrations:** Reduced oxygen concentrations in marine waters have been cited as a major cause for concern globally (Diaz and Rosenburg, 2008), and there is evidence (Queste et al., 2013) that areas of low oxygen saturation have started to proliferate in the North Sea. Extensive research has been carried out on the effects of acute hypoxia (low oxygen), but far less on long-term chronic effects of low oxygen zones, such as those in the North Sea, especially with regard to consequences for commercially important fishes and shellfishes. Whether or not the observed changes in oxygen saturation of waters around the UK are a result of long-term climate change remains unclear, however reduced oxygen concentrations are predicted to occur more extensively, more frequently and for longer periods of time in the future, and it is known that there are certain thresholds below which oxygen levels significantly affect the aerobic performance of marine organisms (see Townhill et al. in press). .

Changes in the distribution and productivity of marine species

Many studies have now been published (and reviewed by MCCIP) that describe changes in the distribution and productivity of marine organisms around the UK.

There have been extensive changes in planktonic ecosystems in terms of overall plankton production, biodiversity and species distribution. In the North Sea the population of the previously dominant and important zooplankton species, the coldwater species *Calanus finmarchicus*, has declined in biomass by 70% since the 1960s (Beaugrand et al., 2003). Species with warmer-water affinities (e.g. *Calanus helgolandicus*) are moving northward to replace the declining coldwater species, but are not as numerically abundant or as nutritionally (i.e. less lipid-rich) important.

Over the last 50 years there has been a progressive increase in the presence of warmwater species in the more temperate areas of the North-East Atlantic and a decline of colder-water species. The seasonal timing of some plankton production has also altered in response to recent climate change. This has consequences for plankton predator species, including fish, whose life cycles are timed in order to make use of seasonal production of particular prey species (Edwards et al., 2013). A particular concern is the possible impact on lesser sandeel *Ammodytes marinus* as a low trophic level fish species underpinning the foodweb for numerous dependent predators – including seabirds, sea mammals and piscivorous fish. Annual reproductive output ('recruitment') of sandeels is known to be heavily influenced by seawater temperature and the availability of zooplankton food. A shift from large energy rich *Calanus* copepods to smaller copepod species will impact the specific growth potential of sandeels and hence many other components of the marine ecosystem (Azour et al., 2015).

Recent warming has caused some coldwater demersal (bottom-dwelling) fish species to move northwards and into deeper water (e.g. cod, plaice), while some warmwater demersal species have become more common in or "invaded" new areas (e.g. John Dory, red mullet). Centres of distribution have generally shifted by distances ranging from 48 to 403 km (Perry et al., 2005) and the North Sea demersal fish assemblage deepened by ~3.6 m per decade between 1980 and 2004 (Dulvy et al., 2008). Further detail on this topic is included in section 3.6.2, although shifts in fish distribution can be difficult to discern because intensive fishing pressure and climate change interact such that stocks may appear to have a more northerly distribution than previously but this may reflect loss of southerly populations due to over-exploitation, rather than movements.

Pelagic (blue-water) fish species are showing particularly marked distributional shifts, with mackerel now extending into Icelandic and Faroe Island waters, whilst sardines and anchovies are moving into Irish and North Sea environments. In the North Sea eight times as many fish species have increased distribution ranges (mainly small-sized warm-water 'Lusitanian' species) compared with those whose range decreased (primarily large cold-water 'Boreal' species) (Hiddink and ter Hofstede, 2008). Teasing apart the relative influences of climatic cycles such as NAO or AMO and general climatic warming as drivers for these changes is an important challenge (Simpson et al., 2011).

There is strong evidence that climate warming has influenced the relative timing (phenology) of fish annual migrations and spawning events in European waters, with potentially significant effects on population sizes and juvenile recruitment. Where spawning is too early or too late to capitalise on available food resources, annual recruitment can be strongly affected. For example, the spawning times of cod in the North Sea have changed significantly since the late 1960s, leading to mismatches with the timing of blooms of essential zooplankton prey and, therefore, reductions in annual recruitment success (Beaugrand et al., 2003). Four out of seven sole stocks were shown to have exhibited a significant long-term trend towards earlier spawning in a study of market sampling data in England and the Netherlands (Fincham et al., 2013).

Many fish species migrate between rivers and the sea as part of their life cycle (i.e. they are 'diadromous'), and climatic variability can impact them in both environments (Lassalle et al., 2009). Observed declines in salmon are strongly correlated with rising temperatures in oceanic foraging areas, with temperature affecting growth, survival and maturation at sea. Freshwater temperatures have also increased significantly in the last 40 years (Section 3.4), with implications for the survival of juvenile diadromous fish, including both anadromous species (river-spawning species such as salmon, trout, shad) and catadromous species (sea-spawning species such as eels and flounder). For eels, climatic changes in the spawning areas of the Sargasso Sea are probably impacting reproduction and larval survival globally.

The impact of climate change on marine mammals remains poorly understood, due largely to the difficulty of obtaining substantive evidence. Range shifts have been observed in a number of cetacean (dolphin and whale) species, and these have been linked to increasing sea temperatures. However, the mechanisms causing those changes remain uncertain, and for some species, it is difficult to differentiate between short-term responses to regional resource variability and longer-term ones driven by climate change (Evans and Bjørge, 2013).

Warmer-water dolphins, such as the short-beaked common dolphin and striped dolphin, have apparently extended their shelf sea range further north in recent years off western Britain and around into the northern North Sea (Evans et al., 2003; MacLeod et al., 2005). Common dolphins are now seen quite regularly in the North Sea, even in winter, and this may reflect the expanding range of typically warmer-water fish species such as anchovy and sardine.

There is no clear evidence that climate change has directly affected either of the two UK breeding seal species (grey and harbour seal), although elsewhere in the world scientists have attempted to link changes in seal population dynamics and life history parameters to climate change. Increases in most grey seal populations and declines in some harbour seal populations could be linked in some way to climate-mediated changes in food supply, although other factors (depletion of food resources from fishing, recovery from epizootics, interspecific competition, density dependent effects) are probably more important (Evans and Bjørge, 2013).

There is increasing evidence that the overwintering distributions of many coastal wading birds have shifted in recent decades in response to warming. In the last decade, this has resulted in declines in usage of east coast sites in favour of the Netherlands, although during recent cold winters this trend has been partially reversed (Pearce-Higgins and Holt, 2013). These changes have probably resulted from a redistribution of individuals rather than changes in survival, either in response to an altered tendency towards cold-weather movements or changing juvenile settlement patterns.

Increasing evidence also exists for similar distribution changes occurring in sea ducks, which may be taking advantage of ice-free conditions in the Baltic, and in coastal wintering waterfowl.. Many UK wintering waders and waterbirds breed in Arctic and subarctic regions that have experienced some of the greatest warming trends globally in recent years. They are therefore potentially vulnerable to additional impacts of climate change outside of the UK (Pearce-Higgins and Holt, 2013).

Seabird (e.g. gulls, gannets and auk) breeding populations in the UK increased in size over much of the last century, but since 1999 these populations have declined by an average of 7.5% (Daunt and Mitchell, 2013). Climate change is considered to be one of the main drivers of these declines. Warmer winter sea temperatures have resulted in major changes in abundance and species composition of plankton that have in turn contributed to the reduction in abundance and quality of seabird prey species such as sandeels, with knock-on effects for seabirds (MacDonald et al., 2015). In addition, increasing incidence of extreme storm events has resulted in frequent seabird 'wrecks' i.e. dead seabirds washed up on beaches. In winter 2013-2014, persistent gales caused prolonged poor foraging conditions at sea, resulting in tens of thousands of seabirds, mainly auks (half of which were puffins) being washed up emaciated on the UK's Atlantic shores (Schmitt, 2014).

Changing distribution of available prey resources may mean that certain breeding sites become untenable in the future and consequently that conservation designations of certain protected UK sites (for example, those associated with nesting little terns) might need to be revisited, also European Union 'Special Protection Area' (SPA) desigantions for overwintering grounds. Furthermore, there is growing evidence that breeding phenology is changing, with seabirds becoming increasingly desynchronised from their prey. However, regional variations in the impacts of climate change are apparent, with weaker effects on seabird demography in the Irish Sea, Celtic Sea and English Channel compared to the North Sea (Daunt and Mitchell, 2013).

Impacts on 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 (as discussed in Section 3.5). Within an estuary, soft sediment benthic intertidal systems can be very variable. Recent research has yielded valuable information about the interactive effects of climate-related and other pressures. In studies of intertidal soft sediments the combined effects of increased temperature, inorganic nutrients and organic matter were additive, rather than antagonistic or synergistic (Fitch and Crowe, 2011; O'Gorman et al., 2012). This highlights the need for future studies to consider cumulative rather than individual effects of climate and other stressors (Jones et al., 2013).

Climate-driven impacts on rocky habitats have been observed at the individual species level, with northern range limits of some warmwater species extending northwards while the range limits of coldwater species are retreating. For example, the topshell *Phorcus (osilinus) lineatus* continues to colonise shores along the rocky coastline of North Wales beyond recent range limits (Mieszkowska, 2012). Similar patterns have been observed in another warmwater topshell, *Gibbula umbilicalis*, along the far eastern English Channel and around into the southern North Sea (Spencer et al, 2012).

During the past five years, long-term survey sites have reported a continuation in the trend for decreases in the relative abundances of coldwater species of barnacles and limpets and an increase in warmwater species. Multidecadal cycles in relative abundances of the coldwater barnacle *Semibalanus balanoides* and warmwater *Chthamalus* spp. are strongly correlated with both local sea surface temperatures and show strong links to the basin-scale AMO, which reflects long-term fluctuations in the marine climate (Mieszkowska et al., 2012).

Phenological shifts are occurring in warmwater gastropods in populations close to northern distributional limits in the UK. Reproductive cycles of *P. lineatus* and *G. umbilicalis* have shifted earlier in the year by three months during the last two decades. Gonad development in the southern limpet *Patella depressa* now commences 19 days earlier in the 2000s than during the 1940s (Moore et al., 2010; Mieszkowska et al., 2013).

Seagrass beds occur in soft sediments within sheltered intertidal and shallow subtidal areas where there is protection from wave action. There is no published evidence for climate-driven changes to UK or Irish seagrass habitats, however given existing knowledge of lethal thermal limits and physiological performance under raised marine temperatures, it is not unreasonable to expect that seagrass distribution and condition will be influenced by climate-related changes, increased storm activity, changes in prevailing wind conditions and also by ocean acidification, shown to be important in other countries (Borum et al., 2004; Bjork et al., 2008). Positive responses with regard to this particular habitat may occur, as sea temperatures increase and more CO2 is available. The changes are not certain but studies suggest that we may witness an increase in range of seagrass in the UK. Authors have suggested that areas where beds may appear and thrive should scoped-out and looked at for increased protection in the future (Natural England 2013).

There is evidence that climatic processes influence species abundance and community composition in inshore and offshore subtidal habitats, as well as those in the intertidal zone. Kröncke (2011) conducted an integrated analysis on benthic communities of the Dogger Bank (Central North Sea) over 1920-2010. The results from this study indicated that both the direct human impact resulting from fishing activity as well as climate change affected the Dogger Bank macrofauna in the 20th century.

There is no obvious signal of warming effects in southern and south-westerly sediments, although changes to the species dominating crustacean assemblages in the Bristol Channel and the occurrence of previously undocumented species in the western Channel (e.g. the brittle star *Amphiura incana* and the shrimp *Athanas nitescens*) suggest some degree of climate influence (Birchenough et al.,2013).

Non-native marine species

Increasing evidence is now available to show that climate change has led to the northwards range expansion of a number of invasive non-native species (INNS) in the UK and Ireland, such as the Asian club tunicate *Styela clava* and the Pacific oyster *Crassostrea gigas* (Cook et al., 2013).

Providing definitive evidence of the direct linkage between climate change and the spread of the majority of INNS is extremely challenging due to the influence of other confounding factors, such as arrivals or secondary transmission via shipping or aquaculture. Localised patterns of water movement and food supply may also be complicating the overall pattern of northwards range expansion by preventing the expansion of some INNS, such as the slipper limpet *Crepidula fornicata* and the Chilean oyster *Ostrea chilensis*, from a particular region (Cook et al., 2013).

More than 90 INNS have been identified in British and Irish (including Republic of Ireland and Northern Ireland) marine and brackish environments, of which over 60 are now established (Roy et al., 2012; Minchin et al., 2013). Their arrival has been principally due to shipping, including ballast waters and sediments, fouling of hulls and other associated hard structures, and imported consignments of cultured species (Minchin et al., 2013). The majority of marine INNS in Britain originate from the North Pacific, followed by the North-west Atlantic, in regions with broadly equivalent climatic conditions (Minchin et al., 2013). Many are initially reported at sites in proximity to ports, marinas and aquaculture facilities, particularly in the English Channel, with a number subsequently spreading northwards to the North or Celtic Seas (Minchin et al., 2013).

The ctenophore (comb jelly) *Mnemiopsis leidyi* was first reported in the North Sea and Baltic Sea in 2006 (Oliveira, 2007) and has recently been detected in UK waters for the first time. This species is a major cause for concern given previous invasions of the Black and Caspian Seas, and the devastating impact on recipient ecosystems and fisheries. Recent modelling studies focused on *Mnemiopsis* in the North Sea found that temperature is the crucial factor controlling winter distribution and future proliferation (David et al., 2015; Van der Molen et al., 2015).

Marine pests, pathogens and diseases

Reports of increased abundance of jellyfish over recent decades have raised concerns over the possible role of climate change in influencing outbreaks (e.g. Atrill et al., 2007). In the UK, fish farms have suffered severe damage because of the stinging jellyfish *Pelagia noctiluca* (at Uist, northern Scotland in November 2014, and Glenarm Bay, Northern Ireland in November 2007) and the Torness nuclear power station in Scotland had to be temporarily closed down in June 2011, as a result of large quantities of jellyfish blocking its cooling water intake screens (see Chapter 4).

The Continuous Plankton Recorder (CPR) survey has indicated an increasing occurrence of jellyfish in the central North Sea since 1958, which may be positively related to the NAO and Atlantic inflow (Lynam et al., 2004; Atrill et al., 2007). High jellyfish numbers are potentially detrimental to fisheries, both as competitors with and predators of larval fish (Purcell and Arai, 2001). In particular, negative impacts on herring larvae have been noted (Lynam et al., 2005). In recent years there has been considerable debate about whether or not gelatinous plankton have increased globally as a result of climate change (e.g. Purcell, 2012; Brotz et al., 2012) and hence whether jellyfish are likely to proliferate further in the future.

A global literature review has suggested that marine pathogens are increasing in occurrence, and that this increase is linked to rising seawater temperatures (Harvell et al., 1999). There is particular concern regarding the poleward spread of bacterial water-borne infectious diseases, mediated by anthropogenic warming of the marine environment.

One such pathogen group, non-cholera *Vibrios*, are a globally significant yet poorly understood cause of morbidity and mortality in humans (see Chapter 5). Approximately a dozen *Vibrio* species are known to cause disease in humans (Austin, 2005), and infection is usually initiated from exposure to seawater or consumption of raw or undercooked seafood (Dechet et al., 2008; Baker-Austin et al., 2013).

Several key data sets have emerged since CCRA1 that provide new information on this risk. In the UK, there have been very few domestically acquired cases reported in the last 20 years, however, the fact that vibriosis is a largely underreported clinical condition probably masks the true extent of disease in the UK. Recent data have shown a clear increase in vibriosis in Northern Europe, with outbreaks typically linked to warm weather episodes (Baker-Austin et al., 2012). In the US, it is estimated that infections associated with *Vibrios* increased by 47% between 1996 and 2005, whereas almost all other agents of food-borne disease (bacterial, viral and parasitic) declined over the same period. The annual estimated cost to the US economy associated with marine *Vibrios* ranges from $6.5 to $34.9 billion, more than any other aquatic pathogen (Ralston et al., 2011).

Recent work by Cefas has identified pandemic (O3:K6) *V. parahaemolyticus* in UK shellfish and water samples (Powell et al, 2013), although no clinical cases linked to these strains have yet been reported in the UK. *Vibrio* species proliferate rapidly at temperatures above 18°C. By contrast norovirus, another major cause of shellfish-acquired gastroenteritis, occurs most frequently in the UK during cooler winter conditions and following periods of high precipitation with the resulting increased runoff from sewers (Campos and Lees, 2014). Historically, the principal sources of microbiological contamination of UK coastal waters have been continuous sewage discharges (CSOs). The Marine Conservation Society (MCS) estimates that there are about 31,000 CSOs impacting UK coastal waters (MCS, 2011). Several pathogens of human health relevance are found in these discharges, namely norovirus, hepatitis A, *E. coli* and *Salmonella* species (Baker-Austin et al., 2013). Very little is known about the effects of climate change on non pathogenic marine bacteria and viruses, although it is recognised that this part of the marine community strongly influences energy flow through lower parts of the marine food chain.

A number of algal species found naturally in UK waters produce toxins that are dangerous to humans. Filter-feeding bivalve shellfish are capable of concentrating toxin-producing algae from the water column, leading to a direct route of human exposure. A perceived increase of harmful algal blooms (HABs) has been globally registered with important ecological and economic consequences due to their effects on coastal marine resources (Miraglia et al., 2009). Overall, the number of reported cases of algal biotoxin ingestion is low in the UK. The surveillance system in place in the UK plays an important role in reducing the human health impact of these toxins, although outbreaks of shellfish poisoning still occur regularly, and closures of aquaculture sites and shellfisheries occur each year. Regional differences continue to be seen in the distribution of HABs around the British Isles, with impacts mainly observed in the south and west coast of Ireland and in regions with a strong Atlantic influence (Bresnan et al. 2013).

HABs occupy a variety of ecological niches in the ocean and climate change can have an impact in diverse ways. Examination of the HAB data collected by the CPR shows that the distribution of some HAB species in the North Sea has altered the since the 1960s with a general decrease in abundance observed along the east coast of the UK. A recent analysis of six phytoplankton taxa from the CPR showed that an increase in wind speed and SST were associated with a decrease in some dinoflagellates genera, while these climatic conditions favoured some diatom taxa, including the toxin producing Pseudo-nitzschia 'seriata' type diatom (Hinder et al., 2012). Blooms of *Karenia mikimotoi*, continue to impact the UK coastline. During the summer of 2012, a widespread bloom of *K. mikimotoi* was observed along the western coast of Ireland where it persisted from May until September. The 2012 bloom resulted in considerable mortalities of fish.

K. mikimotoi blooms do not occur every year, although an increase in the incidence of blooms has been suggested as a potential impact of historical climate change by Bresnan et al. (2010).

An increase in the duration of stratification of the water column may influence the future abundance of HABs in UK waters. This is particularly relevant in shelf areas where offshore high biomass *K. mikimotoi* blooms have been hypothesized to initiate and subsequently impact coastal areas along the west of Ireland and northeastern Scotland. Conversely, an increase in wind speed and duration may reduce the duration of stratification in the water column. This may result in a decrease of some HAB dinoflagellate species. Little is known about the impacts of ocean acidification or changes in offshore circulation on the incidence of HABs. The role of offshore blooms in seeding coastal blooms remains poorly characterised and the lack of monitoring on the shelf edge compounds this knowledge gap.

A recent survey in Scottish waters has revealed the presence of domoic acid in the urine and faeces of harbour seals (*Phoca vitulina*). The impacts of these toxins on the health of marine mammals are unknown and a more detailed study is currently being undertaken (Jensen et al., 2015).

EC Regulation 854/2004 (the EU Shellfish Hygiene Directive) prescribes the legal controls that are placed on the production and marketing of live bivalve shellfish to ensure that contaminated shellfish are not placed on the market. In the UK the Food Standards Agency (FSA) and Food Standards Scotland (FSS) are required, to undertake extensive programmes of monitoring at shellfish production sites and of marine phytoplankton (algae) from harvesting waters. The results of these programmes are used to determine whether an area should be open or closed to harvesting depending on the levels of microbiological and chemical contaminants detected. In addition, bivalve mollusc (shellfish) harvesting areas are classified by monitoring the levels of the sewage-associated bacterium *E.coli* in shellfish flesh, and through 'sanitary surveys' that identify possible sources of microbiological pollution.

Marine pathogens and risks to human health are also covered in Chapter 5 of CCRA 2017, in a section titled 'Risks to health from poor water quality'.

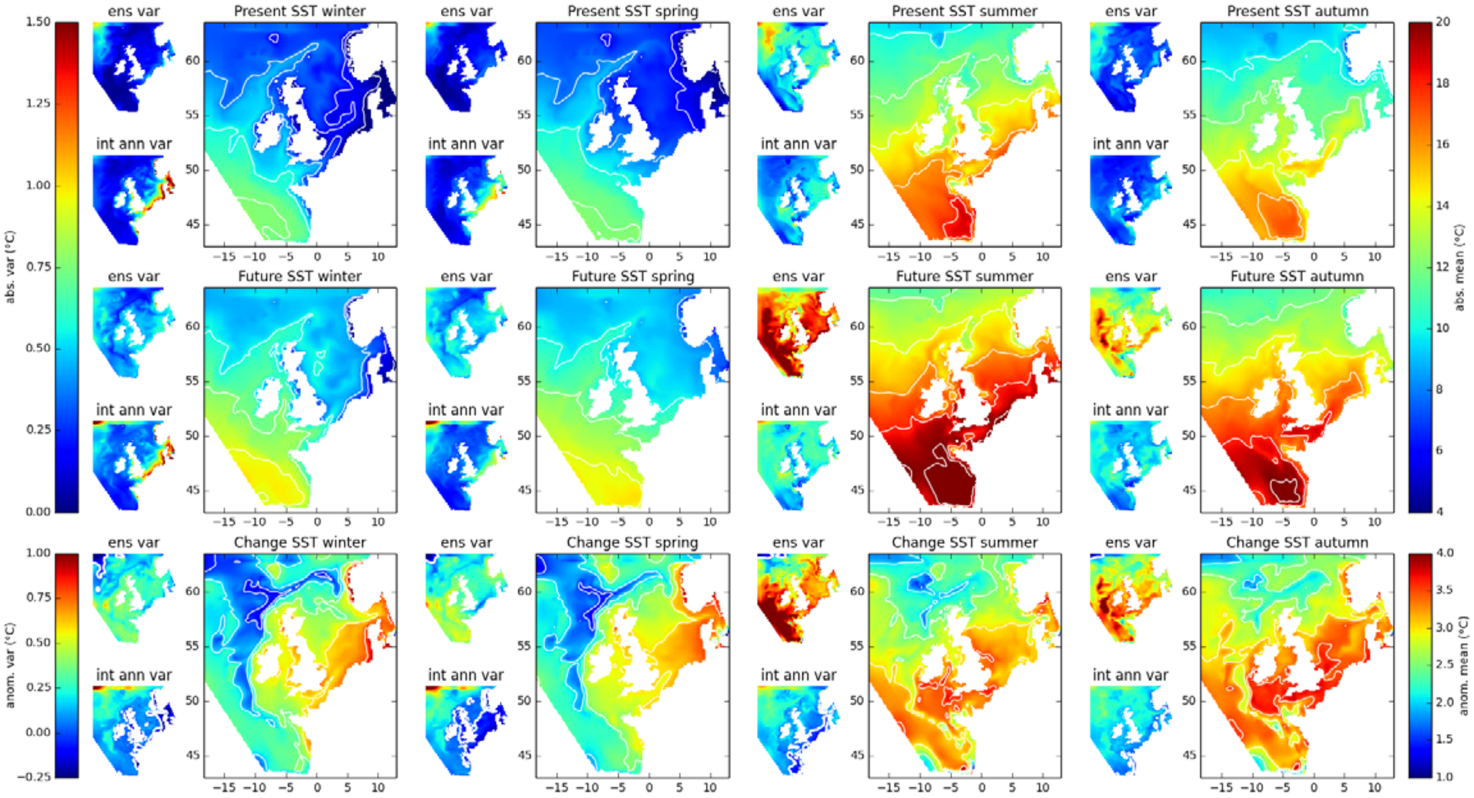
Future risks and opportunities

Climate drivers

- **Temperature:** Over the 21st century, continued warming is anticipated in the shelf seas around the UK and Ireland, although perhaps at a lesser average rate to that observed in the last 30 years. Natural variability, driven by atmospheric and oceanic processes, introduces a level of uncertainty that makes it difficult to predict the exact direction of temperature change over the next decade. UKCP09 included for the first time a projection of possible change in the hydrographic conditions of the seas around the UK and Ireland, providing a basis for many subsequent impact studies. However, these initial projections were only based on one model run, one climate scenario, and outputs were available for one future time horizon (2080s). New maritime climate model outputs have now been made available by the UK Met Office for CCRA 2017, and as a result of the Defra-funded MINERVA project (see Tinker et al., 2015, 2016). The authors used an ensemble approach (comprising 11 ensemble model runs) to give a measure of uncertainty in the future state of the entire north-west European shelf seas. The simulations were run as transient experiments from 1952-2098 under a medium emissions scenario (SRES A1B). Ensemble mean outputs suggested a shelf and annual mean Sea Surface Temperature (SST) rise of 2.90°C ± 0.82°C over the next 80 years (Figure 3.18), and a freshening of -0.41 psu ± 0.47 psu. There is a spatial pattern to the warming, with greatest winter/spring warming anticipated in the south-west North Sea, and summer/autumn warming in the Celtic Sea and North Sea. In the winter and autumn, the Near Bottom Temperature (NBT) across the shelf warms at a similar rate as SST. In the summer, the SST in stratified regions tends to increase more than the NBT (Tinker et al., 2016). Researchers in Ireland recently published a projection for the Irish Sea specifically (Olbert et al., 2012) under the same medium emissions scenario but run over the period 1980 – 2099. They suggest that the Irish Sea SST will rise by around 1.9°C over the next 70 years and that the timing of both the maxima and minima in the annual temperature cycle may shift later by about two weeks. Warming is strongest in the shallow waters along the coastline and in the eastern Irish Sea and weaker in the deep waters of the western Irish Sea.

- **Salinity:** There is considerable uncertainty regarding future salinity. Current projections suggest that the shelf seas and adjacent ocean may be slightly fresher (less saline) in the future than at present. However, models find it difficult to quantify large-scale, long-term changes in salinity as these are the net result of climate-driven changes in precipitation, evaporation, ocean circulation and ice-melt (Dye et al., 2013b).
- **Storms and wave height:** It is has proven exceedingly difficult to provide accurate projections for the North Atlantic Oscillation (NAO) (IPCC, 2013). However, wintertime NAO is very likely to exhibit large natural variations in the future of similar magnitude to those observed in the past, but different climate models suggest different long-term outcomes in terms of whether or not the NAO will tend towards more positive or negative values in the future. Some multi-model studies (e.g. Karpechko, 2010; Gillett and Fyfe, 2013) suggest overall that the NAO is likely to become slightly more positive (on average) in the future due to increases in greenhouse gas (GHG) emissions (IPCC, 2013). Zacharioudaki et al. (2011) used a system of nested global and regional climate models to project the future (2061-2100) wave climate of much of the west-European shelf seas under a range of different global warming scenarios and projected wind fields. This study suggested that mean and extreme wave heights will increase around the south-west of the UK and in the Irish Sea by as much as 10% during winter. Grabemann and Weisse (2008) suggested increases (up to 18% of present values) in significant wave height (SWH) in the North Sea by the end of the 21st century, with an increase in the frequency of extreme wave events anticipated over large areas (under SRES A2 and B2 scenarios).

Figure 3.18. Projections of sea surface temperature: present day (1960– 1989; upper row), future (2070 – 2098, middle row) and difference (bottom row) for each season



Source: Tinker et al., (Accepted pending revisions).
Notes: Each main plot, representing the ensemble mean, is accompanied (to the left) by the two components of ensemble variance: the inter-annual variance (labelled int ann var, small upper panel); and the ensemble variance, the variance of the 30-year mean of the 11 ensemble members (labelled ens var, small lower panel).

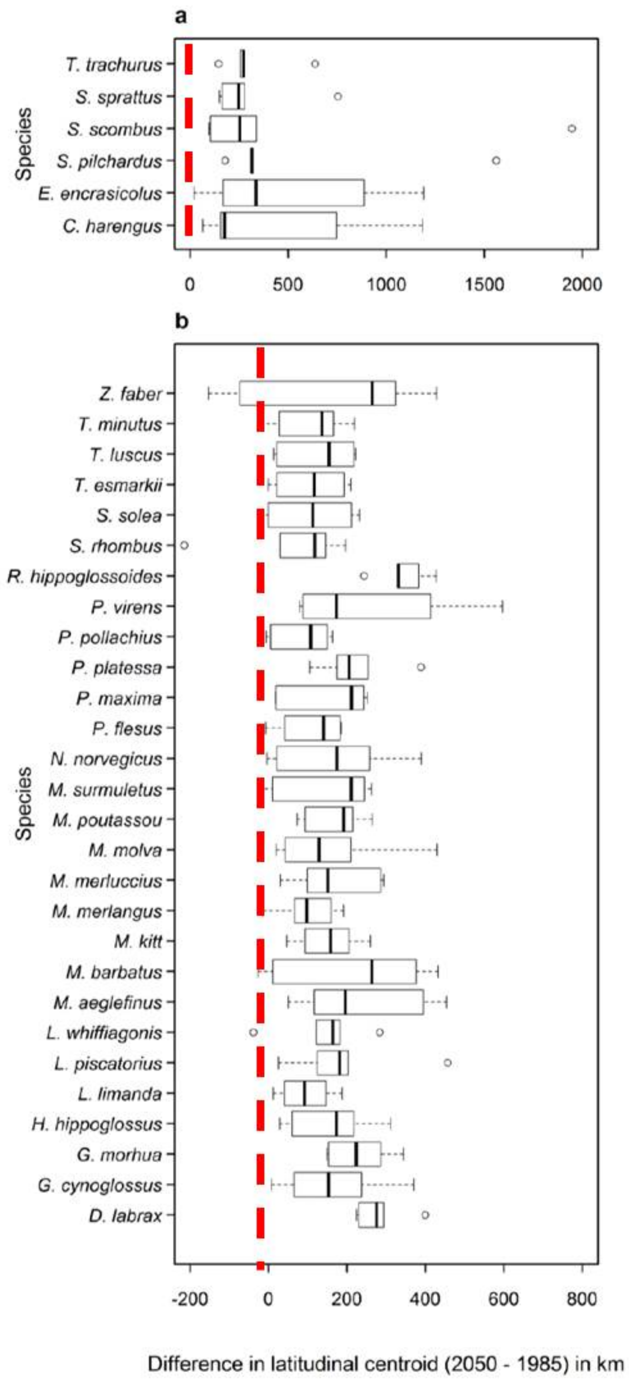
- **Acidification:** Modelled estimates of future seawater pH around UK coastal and shelf waters are generally consistent with global projections. However, variability and uncertainties are greater due to riverine inputs, biotically driven processes (and their own variability) and geochemical interactions between the water column and underlying sediments (Blackford and Gilbert, 2007; Artioli et al., 2012). Whilst there are many global-scale models of ocean acidification in the upper ocean, full-depth modelling for regional seas, such as around the UK, is not so well-developed. In the Regional Ocean Acidification Modelling (ROAM) project of the recently completed UK Ocean Acidification Research Programme (UKOA), a coupled physical-ecosystem model was used to project future values for pH and aragonite saturation state for the North Western European Shelf (see Ostle et al., 2016). The model was forced with data representative of the IPCC AR5 RCP 8.5, as simulated by the UKMO HADGEM model. The model suggests a clear decrease in both pH and saturation state, with areas around the south coast of Norway showing the strongest decrease. Surface waters will start to become undersaturated gradually from around 2030 and more rapidly from 2080. By the end of the century the model estimates that an area of surface water of approximately 300,000 km2 could become undersaturated. The pH trends estimated from this model for OSPAR regions II (Greater North Sea) and III (Celtic Seas) were -0.0036 and -0.0033 pH units per year, respectively (see Ostle et al., 2016).

- **Oxygen concentrations:** Projections of future oxygen concentrations in the North Sea are few and far between, although modelling for three locations suggests marked declines in the concentration of O_2 by the 2080s as a result of simulated changes in the balance of phytoplankton production and consumption, changes in vertical mixing (stratification) and changes in oxygen solubility with temperature (medium emissions) (van der Molen et al., 2013). A parallel study by Meire et al (2013) for the “Oyster Grounds” site specifically suggested that bottom water oxygen concentrations in late summer will decrease by 11.5% by the 2080s.

Changes in the distribution and productivity of marine species

As with organisms on land, a great deal of modelling has been carried out to anticipate future changes in the distribution and productivity of marine species. For example, analysis has been carried out (Jones et al., 2012; Defra, 2013) looking at the relative merits of three different “bioclimatic envelope modelling” approaches to predict distributions of 44 marine fishes and invertebrates around the British Isles. The ensemble projections suggested northward shifts for most fish species, at an average rate of 27 km per decade (the current rate is around 20 km per decade for common fish in the North Sea) (Dulvy et al., 2008). There is, however, considerable variation within and among species projections. Overall, median projected rates of shift were significantly greater for pelagic than demersal fish species, at 277 km and 168 km respectively, over the 65 years. The species predicted to move the furthest and fastest were anchovy and sardine (Jones et al., 2012; Defra, 2013) (see Figure 3.19).

Figure 3.19. Projected change (in km) in latitude from 1985 to 2050 for (a) pelagic fish species and (b) demeral fish species



Source: Jones, M. (2013) PhD Thesis. ueaeprints.uea.ac.uk/47819/1/2013JonesMCPhD.pdf
Notes: Change is shown in latitudinal centroids based on species distribution models and climatic data sets.

Rutterford et al. (2015) used the same fish survey data sets for the North Sea, together with generalized additive models (GAMs), to predict trends in the future distribution of species, but came to the conclusion that fish species over the next 50 years will be strongly constrained by availability of suitable habitat, especially in terms of preferred depths. This study made use of the new ensemble model outputs made available by the Met Office for CCRA 2017 (Tinker et al., 2015, 2016; see figure 3.20).

Horse mussel beds (*Modiolus modiolus*) currently appear as a designated feature in ten MPAs. Based on future climate change projections, there is a risk that this feature will no longer be represented in the UK MPA network by 2100 due to rising sea temperatures (Gormley et al., 2013). Changes in the marine environment and in species distributions could impact upon the preservation of some marine archaeological sites in the near future. Over the last 20 years, increased frequency of the shipworm *Teredo navalis* has been reported along the southern Baltic Sea coasts of Denmark, Germany, and Sweden as well as North Sea coasts of the Netherlands, indicating possible range extensions into previously unoccupied areas. Climate envelope modelling (Appelqvist et al., 2015; Paalvast and van der Velde, 2011) has suggested that near-future climate change is not likely to change the overall distribution of *T. navalis* in the region (including around the British Isles), but might prolong the breeding season and increase the risk of shipworm establishment at the margins of the current range.

Most seabird species in the UK are at the southern limit of their distribution range. As a result, changes in species' ranges due to future climate change are likely, with associated reductions in overall population size. By the end of the 21st century, great skua and Arctic skua may no longer breed in the UK and the range of black guillemot, common gull and Arctic tern may shrink to such an extent that only Shetland and the most northerly tips of mainland Scotland will hold breeding colonies. Many other species could shift their distribution north, no longer breeding in south-eastern England (Huntley et al., 2007; Daunt and Mitchell, 2013).

Fourteen of the 15 wader species modelled using data from France, Ireland, the Netherlands and the UK showed positive relationships between winter temperature and density, but negative correlations between summer temperature and density (Pearce-Higgins et al., 2011). In a separate study, 22 out of 47 wintering waterbird species modelled projected increases in population density by the 2050s (medium emissions), with 10 species showing a decline (Pearce-Higgins and Holt, 2013).

Impacts on habitats

The impacts of sea-level rise and coastal change on intertidal habitats are discussed in Section 3.5.

While shallow southern and south-eastern UK waters are predicted to show the greatest increases in sea surface temperature and near-bottom temperatures under the UKCP09 projections, deep-water habitats to the north and north-east of the UK may also be impacted. The scleractinian coldwater coral *Lophelia pertusa* forms biodiverse and functionally important deep-water reef habitats (Davies et al., 2009). The majority of the world's *Lophelia* reefs are concentrated in the North Atlantic (Guinotte et al., 2006) but they are represented in UK waters in northern and western Scottish waters, as well as canyons on the edge of the Celtic Sea. Experimental studies on the coral's physiology suggest that they are highly vulnerable to ocean acidification and modelled predictions showed that much of the Atlantic Ocean may become corrosive to aragonitic shells and skeletons within the next century. Consequently coldwater corals may be amongst the most vulnerable organisms to ocean acidification and warming (Birchenough et al., 2013).

By 2060, over 85% of known deep-sea coldwater coral reefs in UK waters (mostly to the west of Scotland) could be exposed to waters that are corrosive to them as a result of undersaturation of aragonite. Seven MPAs are designated for the protection of coldwater corals (MCCIP 2015;

Jackson et al., 2014). The East Mingulay Special Area of Conservation may be one of the few places where coldwater corals are still in non-corrosive waters by 2099. Calcification appears to be the process most sensitive to future changes in carbonate chemistry. Sensitivity differences between taxa may reflect species-specific responses to different carbonate chemistry variables, as well as differences in the ability of species and groups to regulate internal pH (Williamson et al., 2013). For all organisms, prolonged exposure to pH values lower than the conditions under which they evolved will require more energy for internal pH regulation, reducing the energy available for growth, maintenance or reproduction. Organisms with an active high-metabolic lifestyle, such as teleost fish and cephalopods, may be better able to cope with future ocean acidification than those with low-metabolic lifestyles, such as bivalves and echinoderms (Melzner et al., 2009). A wide range of laboratory-based studies (mostly short-term, on single species) have shown that a diverse range of marine organisms is potentially affected by levels of surface ocean pH projected for 2100 under business-as-usual scenarios (Williamson et al., 2013). Most impacts are deleterious, however, some species (particularly marine plants) could benefit whilst others seem unlikely to be affected. Meta-analyses have been carried out by Hendricks et al. (2010) and Kroeker et al. (2013) for a wide range of organisms, and by Liu et al. (2010) for microbes. In general, echinoderms, molluscs, calcareous algae and corals appear to be more sensitive than crustaceans, fishes and non-calcareous algae (phytoplankton and seaweeds) (Williamson et al., 2013).

Horse mussel beds (*Modiolus modiolus*) currently appear as a designated feature in ten marine protected areas. Based on future climate change projections, there is a risk that this feature will no longer be represented in the UK marine protected area network by 2100 due to rising sea temperatures and ocean acidification (Gormley et al. 2013).

The UK's territorial deep sea covers a huge geographic area and supports a wide range of environments and biological communities. The deep sea is remote, difficult and expensive to study and as result our understanding of its ecology is still very limited, with detailed studies restricted to a small number of localities. At depths below 200 metres, ecosystems are not directly affected by climate-driven factors such as sea surface temperature, storm surge or significant wave height. However, deep-sea organisms are reliant on plankton and detritus sinking from surface waters for food. Climate-driven changes in surface ocean productivity will therefore have a major impact on deep-sea ecosystems (Hughes and Narayanaswamy, 2013).

Marine non-native species

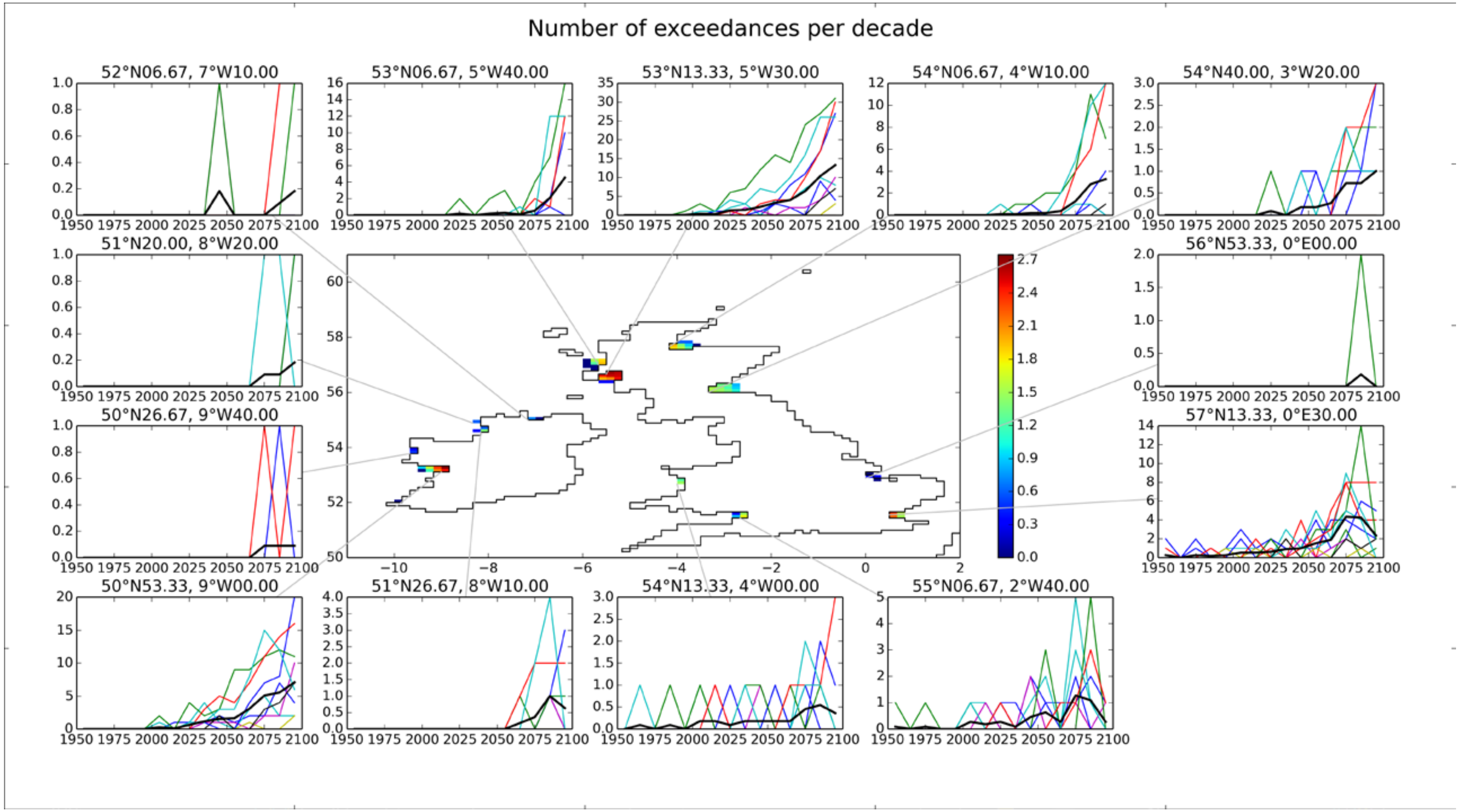
Bioclimate envelope models have been used to "predict" suitable habitat for non-native marine species. A notable example includes Herborg et al. (2007), who examined the potential habitat of Chinese Mitten Crab *Eriocheir sinensis*, a species that reproduces in estuaries but spends most of its juvenile life upstream in rivers, where it causes severe river bank erosion. Similarly Jones et al. used bioclimate envelope models to assess the distribution of suitable habitat for the Pacific oyster *Crassostrea gigas* in north-west European waters both now and in the future under conditions of climate change (Jones et al., 2013).

Further analyses of this type are underway for a shortlist of 19 marine non-native species using the new Met Office model outputs (Tinker et al., 2015, 2016) (see Figure 3.22) as a result of the Defra-funded MINERVA project (Townhill et al., in press).The Maximum Entropy (Maxent) species distribution model was used to predict future habitat suitability, assuming a single SRES emission scenario, A1B. This study suggests that suitable habitat ranges for all species will move poleward at a global scale by up to 843km, and generally northward within the UK shelf seas by up to 115km by the end of the century. Within the shelf seas area, the species with the greatest anticipated northward latitudinal centroid change by the 2080s are cord grass *Spartina townsendii* var. *anglica* (115km), wireweed *Sargassum muticum* (110km), Asian club tunicate *Styela clava* (90km), Pacific oyster *C. gigas* (86km), Asian shore crab *Hemigrapsus sanguineus* and kuruma prawn *Penaeus japonicus*, both (81km).

Marine pests, pathogens and diseases

Most *Vibrio* species that are of concern for human health grow preferentially in warm, low salinity seawater. The projected significant warming of UK waters (Figure 3.21), particularly during summer and autumn months, and anticipated lower salinity as a result of changes in precipitation are likely to significantly increase the prevalence of these pathogens and consequently increase the number of reported clinical cases in hospitals and health centres. Sea surface temperatures around large portions of the UK coastline are anticipated to regularly exceed the 18°C threshold implicated in "at risk" periods elsewhere in Northern Europe, particularly for *Vibrio* wound infections (Baker-Austin et al., 2012). Several UK estuarine areas (notably the Forth, Thames and Severn) are predicted to frequently experience temperatures >18°C and salinities <30psu in the future; that is, the number of 'exceedances' will likely increase, particularly after 2050 (Figure 3.20). It is predicted that this will result in a significantly increased risk of *Vibrio* wound infections or *Vibrio*-associated shellfish poisonings.

Figure 3.20. Number of actual and projected exceedances of environmental conditions that increase risk of *Vibrio* infections per decade (1950 to 2100)



Source: Cefas and Met Office analysis based on the ensemble model outputs provided by Tinker et al. (Accepted pending revision) funded by Defra SEPF project.
Notes: Incidents where sea surface temperatures exceed 18°C and salinity is lower than 30psu. Thick black line indicates the ensemble mean; coloured lines indicate the trajectories of the 11 individual ensemble members.

Adaptation

Impacts on marine species and habitats

Many features for which MPAs have been designated are potentially vulnerable to climate change, meaning the ongoing utility of MPAs as a conservation tool could be affected.

Where an MPA has been identified for its physiographic features (e.g. large shallow inlet and bay) or seabed features (e.g. mudflats or rocky reef), climate change could result in changes to the constituent flora and fauna, rather than the distribution or extent of the feature itself. Such changes are unlikely to compromise the achievement of conservation objectives. Protecting the healthy condition and functionality of these physiographic and seabed habitats might, in such circumstances, be more important than retaining specific species assemblages.

In contrast, where the main feature of an MPA is a named biogenic habitat (e.g. seagrass beds) or a species, the consequences of climate change could compromise the achievement of conservation objectives. In a worst-case scenario, species or habitats could be lost entirely from the MPA (MCCIP 2015). A key objective must be to reduce other anthropogenic pressures in order to build resilience.

Acidification

Ocean acidification is an issue that has received considerable scientific attention since CCRA 2012. Scientific work on ocean acidification and its impacts is being carried out by a wide range of UK research centres, university groups, government bodies and other organisations. In 2010 most of that effort was brought together by the UK Ocean Acidification (UKOA) research programme, a £12.4 million, five-year initiative funded by Defra, DECC and NERC that reached its conclusion in June 2015. Subsequent work funded by Defra (the PLACID initiative) has aimed to sustain monitoring programmes and to investigate consequences for commercial shellfish specifically. Efforts are also underway (funded by NERC) to collate all available pH and carbonate chemistry measurements for UK waters as an input to the next OSPAR Quality Status Report Intermediate Assessment, scheduled for 2017 (see Ostle et al., 2016).

There are similar initiatives throughout Europe as well as elsewhere in the world (e.g. the US, Japan and Australia). Further research is, however, urgently needed in order to “scale up” from laboratory experiments, where species are exposed to elevated CO_2 or low pH, to better understand what sensitivities will mean in terms of populations in the wild and hence for the functioning of natural ecosystems (Le Quesne and Pinnegar, 2012). Possible short-term adaptation actions to mitigate the impacts of ocean acidification are unclear. The main route to avoiding this problem will be through a reduction in anthropogenic carbon emissions, and therefore a long-term reduction in the amount of carbon dioxide being absorbed into the oceans. At a smaller scale various geoengineering solutions have been proposed in order to reduce the pH of seawater at certain locations, for example adding crushed limestone or enhancing local phytoplankton growth. However, such solutions are not viable at the large scale without significant manipulation of global ocean-atmosphere system (Mathews et al., 2009; Williamson and Turley, 2012).

Marine pests, pathogens and diseases

EU regulations prescribe the legal controls that are placed on the production and marketing of live bivalve shellfish to ensure that contaminated shellfish are not placed on the market. In the UK, the Food Standards Agency (FSA) and Food Standards Scotland (FSS) are required to undertake extensive programmes of monitoring at shellfish production sites and monitoring of marine phytoplankton (algae) from harvesting waters. The results of these programmes are used to determine whether an area should be open or closed to harvesting depending on the levels of microbiological and chemical contaminants detected.

Bivalve mollusc (shellfish) harvesting areas are classified by monitoring the levels of the sewage-associated bacterium *E. coli* in shellfish flesh, and through "sanitary surveys" that identify possible sources of microbiological pollution. In addition, local authorities are responsible for collecting water and shellfish samples at the required frequency from the designated sites and for sending these to relevant testing laboratories to look for algal biotoxins. Water and tissue samples are routinely analysed for the presence of harmful algal species which may be responsible for the production of Amnesic Shellfish Poisoning (ASP) toxins, lipophilic toxins (including Diarrhetic Shellfish Poisons (DSP)) and Paralytic Shellfish Poisoning (PSP) toxins. Such well-established programmes should be sufficient to detect any significant changes in the microbial or algal community that might occur as a result of future climate change.

Norovirus and *Vibrio* species are, however, not currently notifiable pathogens in the UK. There is no requirement for statutory surveillance and monitoring. In 2012 the FSA concluded that there remains a serious lack of quantitative information on the prevalence, distribution and levels of norovirus in oyster harvesting areas in the UK. The systematic surveillance data generated by Cefas and other agencies will be used to develop the FSA's policy on virus contamination of bivalve shellfish and contribute to a European review of data on norovirus in oysters.

With regard to *Vibrio* species, scientists, including those based in the UK (Baker-Austin et al., 2012) and at the European Centre for Disease Prevention and Control (ECDC) have used data on sea surface temperature and salinity to create a real-time model that shows coastal areas that are environmentally suitable for *Vibrio* growth. The instantaneous Vibrio Risk Map is available through the ECDC's E3 Geoportal website, and it is hoped that this tool will be used extensively by health departments as an early warning system to prevent infections and save lives.

3.6.2 Marine fisheries and aquaculture

Synthesis

Changing stock levels and shifts in the distribution of species could have both positive and negative implications for the commercial fisheries sector. Observed and projected changes in planktonic ecosystems could also have impacts on marine fisheries, although the nature and extent remains uncertain.

More research is needed to better understand these changes and their potential consequences for fleets, markets and dependent communities. Policy interventions will aim to build resilience in fish stocks by reducing overfishing and other adverse pressures.

The consequences of climate change for UK marine aquaculture industry are poorly understood. Climate change may modify the range of species that can be successfully farmed in the UK. Increased storm frequency and severity, along with ocean acidification, could in the future have significant adverse impacts.

Context and policy

Commercial fishing is an important economic activity in coastal regions of the UK. In 2014, a total of 451,000 tonnes of fish and shellfish were landed by UK vessels, with a first-sale value of £615 million. The UK fishing fleet (including those in the Isle of Man and Channel Islands) consists of over 6,000 vessels, directly employing nearly 12,000 people, while the processing sector employs an additional 18,000 people (MMO, 2015). The UK is a net importer of fish, with imports amounting to around 720,000 tonnes per year and exports amounting to 500,000 tonnes.In general, many of the fish products that consumers choose to eat in the UK, such as cod, haddock, tuna and prawns are deived from imports, whereas many of the most important fish and shellfish that UK fisheremen actually catch, e.g. mackerel and Nephrops (langoustine) are exported. The UK imports 91% of its cod (mostly from cold countries further north), 53% of its haddock and 99% of its shrimps and prawns (MMO, 2015).

Recreational sea angling has economic significance in the UK. Surveys in 2012 suggest that there are 884,999 sea anglers in England, spending £1.2 billion each year on this activity, and supporting over 10,400 full-time equivalent jobs (Armstrong et al., 2013). Changes in species distribution due to climate change, could have serious consequences for this sector, both negative and positive – but these are poorly understood.

Annual UK aquaculture production (2012) amounts to around 177,000 tonnes of finfish, of which 94% is in Scotland, and 27,300 tonnes of shellfish (14% in England, 31% Wales, 27% Scotland and 28% Northern Ireland) and the industry employs over 3000 people (Ellis et al., 2015). The dominant species produced are Atlantic salmon, mussels and oysters. Global production of seafood products from aquaculture has grown substantially in the past decade, reaching 74 million tonnes in 2014, compared with 32 million tonnes in 2000. Aquaculture continues to be the fastest-growing animal food producing sector and currently accounts for nearly half (44%) of the world's food seafish production.

The EU Common Fisheries Policy (CFP), first introduced in the 1970s, has been revised and reformed a number of times over the last four decades, most recently in 2013. It is used by the EU to manage the conservation of marine biological resources. Various measures that support the implementation of the CFP cover aquaculture and the processing and marketing of fisheries and aquaculture products. Key reforms to the CFP in 2013 included a phased ban on discarding of target fish, i.e. a 'Landing Obligation' effective as of 1 January 2015; a legally binding commitment to fishing at sustainable levels; and increasingly decentralised decision making, taking place at regional rather than the pan-European level.

The CFP's objective is to be "environmentally, economically and socially sustainable" and to contribute to the availability of food supplies. There is no explicit mention of climate change in the original text or subsequent amendments but a number of the key functions within the CFP are known to be sensitive to climate change. Limits on the number of fish allowed to be caught, total allowable catches (TACs), are set by the Council of Fisheries Ministers, following proposals by the European Commission based upon scientific assessments of managed fish stocks. One component of the scientific advice is an estimate of the number of juvenile fish entering a population (termed "recruitment"). Environmental conditions, including climate change, are amongst the factors that contribute to the variability of recruitment from year to year.

TACs are shared between EU nations through "relative stability" arrangements. These arrangements are based on member state catches during a historical reference period (Morin, 2000) and give each member state a fixed percentage share of each stock. Climate change has the potential to shift the geographical distribution of commercial species so that they no longer match the existing patterns of exploitation. Some flexibility is provided by the CFP as quotas can be swapped each year between member states which could be used if distributions of managed stocks shift into new areas, or retreat from traditional ones. However this is currently a very complicated process and swaps are not always straightforward, especially in the new era of the Landing Obligation. Efforts to actively engineer climate change adaptation, could only be achieved through wholesale renegotiation of relative stability arrangements on a regular basis, and this is something EU member states have resolutely refused to do. If previously unregulated but increasingly important stocks need to be managed using TACs then member states will however, negotiate with each other to determine their respective relative stability shares of these 'new' resources.

The new 'landing obligation' will be introduced gradually between 2015 and 2019 for all commercial fisheries (species under TACs, or under minimum landing sizes). The obligation stipulates that once the least plentiful quota species in a mixed fishery is exhausted, the whole fishery must cease operation. Baudron and Fernandez (2015) have argued that many commercial fish stocks are beginning to recover under more favorable climatic conditions. For example northern hake (*Merluccius merluccius*) a warm-water species, has recolonized the northern North Sea from which they had largely been absent for over 50 years. These changes have implications for the management of other stocks. Notably, if discards are banned as part of management revisions, the relatively low quota for hake in the region could be a limiting factor that may potentially result in the closure of the entire demersal mixed fishery, jeopardizing livlihoods of commercial fishermen, especially in Scotland (Baudron and Fernandez, 2015).

Some sharing arrangements ("fisheries agreements") are in place between EU and non-EU countries (such as Norway, Iceland, Greenland and the Faroe Islands) that allow TACs to be set for stocks that are shared.

The Straddling Fish Stocks and Highly Migratory Fish Stocks Agreement (1982) is part of the implementation of the United Nations Convention on the Law of the Sea (UNCLOS) and aims to "ensure the long-term conservation and sustainable use" of straddling fish stocks (e.g. cod and halibut) and highly migratory fish (e.g. tuna and oceanic sharks). Environmental conditions are one of the factors that influence variability of recruitment to a stock as well as distribution and thus levels of sustainable yield that can be taken from that stock.The need to manage stocks in line with prevailing marine environmental conditions is clearly recognised within the agreement, including the use of best available science to maintain or restore stocks at levels capable of producing maximum sustainable yield (Article 5). Article 2.2. of the new CFP also aims to ensure that 'exploitation... restores and maintains populations of harvested species above levels which can produce the maximum sustainable yield'.

Exploitation of shellfish stocks in UK waters is generally regulated through national legislation (the Sea Fisheries (Shellfish) Act 1967) or local bylaws rather than at the EU level. The government is able to mandate the improved management of private or natural shellfisheries. Orders restrict fishing rights in a specific area of the sea or tidal waters. They cover one or more named species of shellfish and are granted for a set period. Under conditions of future climate change it is possible that commercial shellfish species might not be able to persist in certain areas, and hence that long-established management measures such as 'Several/Regulatory orders' might need to be revisited in the future.

Current risks and opportunities

Fisheries

Long-term changes in temperature and other marine variables often coincide with observed changes in fish distribution and hence the locations where commercial fishery operations achieve high catch rates. Analysis of Scottish and English commercial catch data spanning the period 1913-2007 has revealed that the peak catches of target species such as cod, haddock, plaice and sole have all shifted distribution latitudinally, albeit not in a consistent way (Engelhard et al., 2011). Over the past century, for example, cod catches have shifted steadily north-eastward and towards deeper water in the North Sea (Engelhard et al., 2014). Sole catches have retreated away from the Dutch coast southwards towards the eastern Channel, whereas plaice catches have moved steadily north-westwards towards the central North Sea (van Keeken et al., 2007; Engelhard et al., 2011). Peak haddock catches have moved very little in terms of centre of distribution, but their southern boundary has shifted northwards by approximately 130 km over the past 80 – 90 years.

The commercial fishery sector has witnessed and responded to a number of new opportunities in recent years, as warmwater species have moved into UK and Irish waters and their exploitation has become commercially viable for the first time. Notable examples include new or expanding fisheries for sea bass, red mullet, John Dory, anchovy, hake and squid. Theoretically, in the northern hemisphere warming results in a distributional shift of species northwards, and cooling draws species southwards (Burrows et al., 2007). Both northerly and southerly shifts have been observed for the whole North-east Atlantic region since the 1970s for individual species (Heath, 2007). More species shifted south than north between the 1970s and 1980s (a relatively cool period) and vice versa between the 1980s and 1990s (a relatively warm period). This seems to parallel observed interdecadal changes in air and seawater temperatures.

Distribution shifts will have “knock-on” impacts for commercial fisheries because changes in migration or spawning location affect the “catchability” of individual species. Populations may move away from or towards areas where fishing fleets operate or where spatial restrictions on fishing are in place. In addition, species distributions may migrate across national boundaries. A notable example of the consequences of such changes is events arising from rigid quota allocations between Norway and the EU, and between Iceland, the Faroe Islands and the EU. In 2009 it became apparent that North Sea mackerel had moved away from the Norwegian Sector, possibly as a result of excessively cold conditions near the Norwegian coast, resulting in disagreements over permissible catches by Norwegian boats operating in EU waters. Shortly after, Iceland and the Faroe Islands unilaterally claimed a quota for mackerel since the species had attained high abundance in their territorial waters, having previously been largely absent. The debate over equitable quota allocation has raged for more than five years, with EU countries accusing Iceland and the Faroe Islands of threatening stock sustainability. It is unclear whether the apparent changes in mackerel distribution were a result of long-term climate change (Astthorsson et al., 2012; Hughes et al., 2015), but fisheries scientists anticipate many more territorial and quota dissagreements of this type in the future as a result of climate change.

A similar incident is now occurring in the English Channel and southern North Sea region with regard to access to European anchovy. Anchovy stocks are currently depleted in the Bay of Biscay where Spanish and French vessels have exclusive access rights, but are increasing further north along southern coasts of the UK (Petitgas et al., 2012).

South of Ireland, new fisheries have recently opened up for boarfish *Capros aper*, a small, previously unimportant species that is converted to fish meal for aquaculture. Landings have grown rapidly from less than 120 tonnes in 2001 to more than 139,000 tonnes in 2010 (ICES, 2011). Boarfish became increasingly prevalent in French and UK survey catches after 1990 (Pinnegar et al., 2002). In the past boarfish outbreaks had been linked to storms and variability in offshore climate. Their appearance after 1990 across the whole North Atlantic basin may be linked to a series of strong positive anomalies in the NAO and quota restrictions are now being put in place to ensure sustainable exploitation of the newly emerging, but very large-scale fishery..

The fishing industry and scientists have known for over 100 years that the status of fish stocks can be greatly influenced by prevailing climatic conditions. Recruitment variability is a key measure of stock productivity, and is defined as the number of juvenile fish surviving from the annual egg production phase to be exploited by a fishery. Recruitment is critically dependent on the match or mismatch between the emergence of the larvae and availability of their zooplankton food as well as other processes that affect early life-history stages (see Petitgas et al., 2013). Empirical data on exploited populations often show strong relationships (positive and negative) between recruitment success, fisheries catches and climatic variables.

Some species around the UK benefit from warmer conditions, whereas others suffer. In the case of cod, there is a particularly well-established relationship between recruitment and sea temperature (Drinkwater, 2005). In the North Sea, close to the southern limits of the species' range, warmer conditions lead to weaker than average year classes.

Extensive fishing can cause fish populations to become more vulnerable to short-term natural climate variability (e.g. Ottersen et al., 2006) by making populations less able to "buffer" against the effects of the occasional poor year. In the case of cod, climate change has been estimated to have been eroding the maximum sustainable yield at a rate of 32,000 tonnes per decade since 1980 (Cook and Heath 2005).

In terms of value, the most important fishery for the UK is Norway lobster (scampi) *Nephrops norvegicus* (£98 million, 30,300 tonnes in 2014). *Nephrops* are particularly important in Northern Ireland, where they represent 60% of the total value and 38% of the catch. Climate impacts were not found by Zuur et al. (2003), who analysed landings-per-unit-effort (LPUE) series for 13 *Nephrops* populations. However, González Herraiz et al. (2009) found that LPUE from Spanish trawlers operating on the Porcupine bank (West of Ireland) was negatively related to the NAO index. Experiments on Irish Sea *Nephrops* demonstrated that, at higher temperatures, larval stages are typically of shorter duration (Dickey-Collas et al., 2000), hinting that climate warming could speed up growth rates and potentially influence recruitment patterns.

Scallops *Pecten maximus* are the most important shellfish fishery in the UK in terms of tonnage (£58 million, 38,500 tonnes in 2014). Numbers of young scallops recruiting each year have been positively related to seawater temperature in the spring, and warmer conditions seem to coincide with higher egg production (Shephard et al., 2010). Weiss et al. (2009) suggest that edible crab *Cancer pagurus* exhibits a particularly narrow larval temperature tolerance range and so recruitment in this species may be highly vulnerable to climate change. By contrast, spider crab *Maja brachydactyla* are thought to benefit from warmer seawater temperatures. Early hatching of spider crabs on the French Coast adjacent to the Channel Islands has been related to higher winter/spring sea temperature (Martin and Planque, 2006).

The winter of 2013/2014 has been identified as one of the stormiest (in terms of wind speeds, wave heights, and so on) of the past 66 years (Matthews et al., 2014). The UK fishing industry was severely disrupted, with many vessels tied up in port for at least five months, along with severe damage to fishing boats and harbour facilities (Andrew, 2014). The lack of fish reaching markets resulted in higher prices. Many of the key climate change risks affecting maritime industries and ecosystems relate to future patterns of storm frequency and severity, yet this is one of the most uncertain areas of climate prediction, with very little confidence in model outputs.

Aquaculture

To date, there has been little published research on the effects of climate change on UK mariculture (although see Callaway et al., 2012; Gubbins et al,. 2013). However, some changes in the sector have been attributed to a changing climate, for example, increases in the incidence of norovirus associated with bivalve shellfish have been associated with increases in precipitation, runoff and sewerage overflow and the establishment of feral non-native populations of Pacific oysters could have been related to elevated seawater temperatures (Gubbins et al., 2013).

In August 1984, the jellyfish *Phialella quadrata* was reported to have caused the death of 1,500 Atlantic salmon smolts at a location in Shetland (Bruno and Ellis, 1985; Seaton, 1989). This appears to be the first documented case of salmonid mortality related to jellyfish in Scottish waters. In November 2007, the small jellyfish *Pelagia noctiluca* swamped cages of a salmon farm in Northern Ireland, killing an estimated 250,000 fish (Doyle et al., 2008). Whether or not these events reflect a proliferation of jellyfish in recent years and hence are linked to long-term climate change remains unclear (see section 3.6.1), but further research is clearly warranted (Callaway et al., 2012) and is currently underway.

Storm damage is a major concern to the aquaculture industry. Storms are a limiting factor in the distribution of mussel beds in intertidal areas (Callaway et al., 2012). In addition a study of fish escapes in Scotland found that of the 2.18 million fish that escaped during the seven years covered by the research, 38% escaped during a single storm event in 2005 (Taylor and Kelly, 2010).

Future risks and opportunities

Fisheries

Considerable progress has been made in recent years with climate projections for fish and fisheries. Some of these advances have followed the application of complex ecosystem or fish distribution models (e.g. Blanchard et al., 2012; Cheung et al., 2011; Lindegren et al., 2010). Other advances have involved detailed economic modelling to determine the potential impact of climate change on fishing fleets and on regional economies (e.g. Defra, 2013).

Modelling suggests that distributions of exploited species will continue to shift in the next five decades, both globally and in the North-east Atlantic (Jones et al., 2014; Cheung et al., 2011). The total maximum catch potential is projected to decrease slightly within the UK Exclusive Economic Zone (EEZ) by the 2050s (Jones et al., 2015), resulting in a 10% decrease in net present value, assuming a 'high' emission SRES A2 scenario In contrast, a separate study using a very different model methodology predicted a 24% increase in catch potential by the 2050s (Blanchard et al., 2012) assuming an SRES A1B 'business as usual' emissions scenario. The differences in the projected changes from these two studies is likely to be due to assumed changes in net primary production that drove the two underlying biogeochemical models. The model used by Jones et al. (2015) anticipated a decrease in net primary production available for fish, while Blanchard et al. (2012) assumed an increase. Clearly a detailed examination of what might be expected to happen to plankton productivity in waters surrounding the UK will be crucial for accurately determining the likely consequences for UK fisheries, although most recent studies have predicted a decline (see section 3.6.1 which highlights an observed reduction in plankton productivity in recent years).

Many efforts to predict climate change impacts on future fisheries productivity (i.e. recruitment, stock biomass and maximum sustainable yield) have tended to focus on cod. One study predicts that cod stocks in the Celtic and Irish Sea will disappear altogether by 2100, while those in the southern North Sea will decline (Drinkwater, 2005). Elsewhere, stock productivity will increase (e.g. around Iceland). Cod will likely spread northwards along the coasts of Greenland and Labrador, occupying larger areas of the Barents Sea, and may even extend into some parts of the Arctic Ocean. At present most of the cod eaten by consumers in the UK come from the Barents Sea (Norway) or Iceland.

Climate change may make stocks more vulnerable to overfishing by reducing the overall carrying capacity of the stock (Jennings and Blanchard, 2004). Cook and Heath (2005) examined the relationship between sea surface temperature and recruitment in a number of North Sea fish species (cod, haddock, whiting, saithe, plaice, sole). These authors concluded that if the recent warming period were to continue, as suggested by climate models, stocks which express a negative relationship with temperature (including cod) might be expected to support much smaller fisheries in the future.

Ocean acidification may have direct and indirect impacts on the recruitment, growth and survival of commercial fisheries. The impacts are suggested to be particularly significant for animals with calcium carbonate shells and skeletons such as molluscs, some crustaceans and echinoderms. Four of the ten most valuable marine fishery species in the UK are calcifying shellfish and analysis suggests that losses in the mollusc fishery (scallops, mussels, cockles, whelks, etc.) could amount to £55 – £379 million per year by 2080 depending on the CO_2 emission scenario chosen (Pinnegar et al., 2012).

Changes in species distribution due to climate change could have serious consequences for the recreational fisheries sector, both negative and positive, but these are poorly understood. Climate change is having a profound impact on fish populations popular with recreational sea fishing (Simpson et al., 2011) e.g. seabass. Sought-after species such as cod and pollock may decline, but there might be opportunities to increase participation with new warmwater species moving into UK waters (e.g. sea bass, sea bream). Fisheries scientists have a reasonable understanding of the recreational sea fishing sector in terms of what species are currently favoured, and where and when they are targeted (Armstrong et al., 2013), but limited insight into how this sector might develop in the future. For example, possible impacts of climate change on the recreational sea angling sector could stem from any changes in storminess, since anglers typically operate from small boats or from exposed shorelines.

Aquaculture

Commercial shellfish often have very specific habitat requirements (e.g. sediment type, depth range, food availability) and their ability to shift distribution and track climate change may be compromised compared to more mobile species such as finfish. To date, comparatively few studies have attempted to model future responses of commercially important shellfish species around the UK, although bioclimate envelope modelling studies have been conducted for Pacific oyster (Jones et al., 2013) and both coldwater and warmwater mussel species (*Mytilus edulis* and *M. galloprovincialis* respectively) (Fly et al., 2015). These studies have suggested northward shifts in habitat suitability for commercial shellfish in the future, but have also highlighted major uncertainties with regard to changes in coastal primary production and how filter-feeding shellfish might respond to such changes.

Opportunities for farming warmer-water fin-fish species such as sea bass (*Dicentrarchus labrax*) and turbot (*Scophthalmus maximus*) may emerge from rising seawater temperatures, as has been envisaged for Norwegian waters (Bergh et al., 2007). However, an increase of 2°C may well adversely affect some species currently being farmed in Scotland as the thermal optima for the animal's physiology may be exceeded for long periods of time during the summer months. Aquaculture of species such as Atlantic cod and Atlantic halibut may not be possible in the south of the country or may be limited to areas of deep-water upwelling where the water is cooler than elsewhere.

An increase in sea surface temperature is also likely to affect the growth of farmed shellfish. A predictive modelling study of farmed shellfish growth in Northern Irish waters (Strangford Lough) suggested that production of mussels would decrease by 10% and 50% under +1ºC and +4ºC scenarios respectively. Oysters, due to their differing physiology, were predicted to experience much less growth limitation under the same scenarios (Ferreira et al., 2008).

Some of the most damaging but least predictable effects of climate change relate to the emergence, translocation and virulence of diseases, parasites and pathogens. Not all effects on disease will be detrimental. Some viral infections occur between narrow temperature ranges, often 10 – 12ºC (Bricknell et al., 2006), usually during spring and autumn. Under warmer conditions this temperature window may decrease in the spring (and occur earlier in the year) as more rapid warming of water occurs in spring. Conversely, if cooling of the environment is delayed during the autumn, this temperature window may become extended and occur later in the year. Additionally, warmer water conditions may allow the establishment of exotic diseases, which are currently excluded as the climate is too cool to permit transmission (Gubbins et al., 2013).

The potential level of threat from climate change depends on the degree of exposure that individual shellfish or fin-fish farm operations have to prevailing environmental conditions, or the need to obtain broodstock from the wild. Bivalve farming, which depends on wild spat for stock, plankton for food and water quality for health, is highly susceptible to various effects of climate change. In comparison salmon farming is more independent of the natural environment since feed supply and offspring are sourced from hatcheries or elsewhere in the world, where climate change might or might not be a problem (Callaway et al., 2012).

Ocean acidification may result in financial losses of £60 – £125 million per year from the shellfish aquaculture sector (oysters, mussels, etc.) if CO_2 concentrations were to reach levels of around 740 ppm (Pinnegar et al., 2012). Acidification may have detrimental effects on mollusc spatfall, making natural seeding of mussel or oyster farms less efficient or untenable in the future.

In the north-west United States researchers have linked the collapse of oyster seed production to an influx of low pH water. For the past several years, the Pacific Northwest oyster industry has struggled with significant losses due to ocean acidification as oyster larvae encountered mortality rates sufficient to make production non-economically feasible. Carbon dioxide rich, low pH water was being brought up from the ocean depths onto the Pacific coast via increased upwelling and this had serious consequences for oyster hatcheries in Oregon and Washington State. Significant investment ($500,000) in pH monitoring, at hatcheries and offshore is now allowing shellfish growers to switch off seawater pumps to hatcheries when the most corrosive conditions are detected/anticipated and hence shellfish production is once more viable, and the industry is recovering (Barton et al., 2012). Such adaptation actions might eventually be needed in the UK and Ireland. Recent model outputs for this particular region suggest that surface waters will start to become under-saturated ('corrosive') with regard to aragonite, from around 2030 and especially so from 2080.

There is therefore a strong economic reason to improve understanding of physiological and behavioural responses to ocean acidification and techniques for modelling the upscale implications (Le Quesne and Pinnegar, 2012).

Adaptation

Well-managed fish stocks are much less susceptible to climate impacts than those that are already heavily degraded as a result of overfishing. In some cases, fish stocks will not be able to sustain the same level of fishing pressure in the future as they have experienced in the past as a consequence of climate change. Fishing vessel operators are used to dealing with constantly changing weather, stock sizes and market prices, hence there is an expectation that they will be able to autonomously adapt to climate change in the future. However, the adaptive capacity of some segments (e.g. small vessel operators) is likely to be more constrained than others.

In 2015, Seafish (the trade body for the commercial fisheries sector) published a report 'Understanding and responding to climate change in the UK wild capture seafood industry' . This study drew heavily upon earlier outputs from the UK Marine Climate Change Impacts Partnership (MCCIP), as well as the sea fisheries component of the 2012 'Economics of Climate Resilience' project (Defra, 2013). The exercise relied on research evidence and industry experience, engaging around 40 fishery stakeholders as well as scientists. Adaptation responses will differ between domestic and international parts of the UK wild capture industry:

- In a domestic context, suggested adaptation responses included: improving scientific advice and data collection through partnership working, reviewing fisheries governance for regulated and non-regulated species, enhancing vessel operational safety, investing in port resilience, and assessing freight ferry vulnerability.
- In an international context, suggested adaptation responses include: reviewing of key sources of existing supply and available options, assessing the impact of changes in specific regional supplies, ensuring international management regimes provide early resolution on 'rights to fish', incorporating climate change into vessel and gear design, and improving resilience and capacity of overseas facilities.

The key adaptation actions highlighted in the 'Economics of Climate Resilience' report (Defra, 2013) for the UK fishing industry included:

1. Travelling further to fish for current species, if stocks move away from UK ports.
2. Diversifying the livelihoods of port communities, this may include recreational fishing where popular angling species become locally more abundant (e.g. sea bass).
3. Enhancing vessel capacity if stocks of currently fished species increase.
4. Changing gear to fish for different species, if new or more profitable opportunities to fish different species are available.

5. Developing routes to export markets to match the changes in catch supplied. These routes may be to locations (such as southern Europe) which currently eat the fish stocks which may move into the UK EEZ.
6. Stimulating domestic demand for a broader range of species, through joined up retailer and media campaigns.

In 2011 Sainsbury's ran a campaign "switch the fish", that challenged the supermarket's customers to try an "alternative" finfish species at reduced cost, including some of those that are more reflective of current climatic conditions in waters around the UK (e.g. mackerel, seabass, pollack, hake etc.). Following a year of high profile awareness campaigns, Sainsbury's reported an annual increase of seabass sales of 57%, and pollack 15% (Defra, 2013). Such campaigns may need to be expanded further in the coming years to ensure that UK consumers are able to eat sustainable seafood given global patterns of climate change.

In 2016 two EU-funded Horizon 2020 research projects (CERES and CLIMEFISH) were commissioned, with the aim to "provide the knowledge, tools and technologies needed to successfully adapt European fisheries and aquaculture sectors in marine and inland waters to anticipated climate change". This will include economic assessments of climate change impacts on these two sectors, but also the development of practical adaptation measures that fishermen, farm managers and aquatic resource managers can adopt to successfully adapt to forthcoming climate change threats or to capitalise on emerging opportunities.

3.7 Cross-cutting issues

3.7.1 Carbon storage and GHG emissions

Synthesis

In terrestrial ecosystems, carbon is stored in soils and vegetation. Warmer temperatures and CO_2 fertilisation benefits for primary productivity may potentially increase carbon stocks but this may be countered by losses due to enhanced soil respiration during higher temperatures. Current evidence for changes in carbon stocks suggests land use and management is the dominant influence and the role of climate is less clear (Section 3.3.1 reviews evidence for soil carbon). This also means that risks and opportunities from future climate change remain uncertain.

Despite this uncertainty, the poor condition of some peatland areas and of other organic soils indicates that higher temperatures and the likelihood of drier summers are likely to substantially increase the loss of carbon stocks, notably for the drier eastern side of the UK. As climate is an important influence on land-use patterns, changes that affect land allocation for different types of agriculture and forestry will also have an impact on carbon stocks in soils and vegetation (as discussed in Section 3.3.2). Losses may be exacerbated by land-management practices, such as the continued use of lowland peat soils for intensive arable production. In addition, GHG emissions from methane and nitrous oxide will also be modified by changes in both land use and climate, but these emissions may be partially mitigated by further support for and implementation of good management practices.

There is a need to better account for the impacts of climate change on carbon stocks inprojections of future GHG emissions and removals from land use and forestry. Further action to restore degraded natural carbon stores, such as peatlands, is also needed as it is likely that climate change will further exacerbate carbon losses unless their resilience is improved..

Carbon is also stored in coastal and marine ecosystems (“blue carbon”). Althoughknowledge is improving, we have limited information on how climate change may affect these stocks. Nevertheless, potential loss of coastal intertidal areas (Section 3.5) suggests an increased risk of loss of carbon stocks and sequestration potential.

Context and policy

Carbon is naturally stored in soils and vegetation. Vegetation growth also acts to sequester CO_2 from the atmosphere into plant tissues which can then be transferred to soil carbon through litter, humus and root exudates. Soils can also be a source of carbon emissions through decomposition and respiration, which may be accompanied by losses of methane (mainly from wetlands) and nitrous oxide (mainly from artificial fertilisers), both powerful GHGs. The largest carbon stocks occur in soils, particularly organic (carbon-rich) soils, as exemplified by peatlands, which are deep organic soils that are usually waterlogged and actively sequester carbon due to retarded decomposition rates and colonisation by specially adapted species, notably *Sphagnum*. The largest vegetation stocks and overall sequestration rates occur in woodlands, although these carbon stores can be lost with felling or degradation of woodlands.

Estimates of the amount of carbon stored in UK soils vary, depending on depth and survey method. For the purposes of UNFCCC inventory reporting, a standardised inventory was produced which estimated the total soil carbon stock to be 9838 ± 2463 million tonnes. Since then, improved estimates have been derived for Scotland (ECOSSE, 2007), England (Natural England, 2010) and Wales (Jones and Emmett, 2013) and of soil carbon stocks in the UK's forests (BioSoil: Vanguelova et al., 2013).[33]

Forests are an important component of the UK carbon balance.[34] Estimates of the carbon content of woodland have tended to increase as previous studies were found to have underestimated the carbon held in root stock. The most recent "bottom-up" estimate is 213 million tonnes of carbon (Forestry Commission, 2014a). This is based on data from the first five-year cycle of the National Forest Inventory to calculate the amount of carbon in living trees (as at 2011), including roots, stems, branches and leaves but excluding biomass contained in other vegetation associated with the stand (e.g. shrubs and herbs). Broadleaf trees were assumed to be in leaf.

Carbon stored in woodlands is approximately equally split between coniferous and broadleaved species; Sitka spruce plantations contain 55 MtC, compared to 31 MtC for oak woodlands. According to the UK's GHG inventory, 17 $MtCO_2$ was sequestered by UK forests in 2013 although this is forecast to decline from the 2030s as the national forest age structure changes and plantations from the 1960 – 1980s reach harvest date (Buys et al., 2014).

"Blue carbon" is defined as the carbon stored in coastal and marine habitats and sediments. Typically, blue carbon is thought of as only being coastal, but recent studies (Alonso et al., 2012; Birchenough et al., in press) suggest that offshore habitats around the UK are also important carbon stores. It is possible to characterise different marine habitats on the basis of their potential as carbon sinks, that is, the rate at which they sequesteratmospheric CO_2 , and the duration of carbon storage. Rates of carbon sequestration are particularly high in saltmarsh and sand dunes (UK NEA, 2011; Beaumont et al., 2012). Data concerning the role of offshore habitats as sinks of blue carbon are comparatively scarce, but a dedicated study has been commissioned as part of the NERC/Defra-funded Shelf Sea Biogeochemistry programme.

Targets for GHG emissions are an integral component of climate change mitigation policy, including a statutory target of an 80% reduction by 2050 compared to a 1990 baseline. Progress on these targets is measured through the UK's national Greenhouse Gas Inventory, which records emissions from energy, industry, transport, buildings and waste. GHG emissions from and sequestration by vegetation and soils (although not coastal or marine habitats) are accounted for within the Land Use, Land Use Change and Forestry (LULUCF) sector and the Agriculture sector of the inventory.

Current risks and opportunities

Climate interacts directly with the carbon cycle on land through its influence on the rate of plant and soil processes. In addition climate indirectly impacts on the carbon cycle through its influence on land-use and management choices and therefore the type of plants and soils available to store or sequester carbon. Climate change can directly affect sequestration rates through a longer and more intensive growing season and increased CO_2 concentrations.

As discussed in Section 3.3.6, there is some evidence of enhanced tree biomass growth in recent decades across Europe, but this may also be attributable to non-climate factors (i.e. enhanced

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[33] See Suggitt et al (2015) for the ASC for a review and summary of the different estimates.

[34] See NFI reports: http://www.forestry.gov.uk/forestry/infd-8tel28 and http://www.forestry.gov.uk/pdf/FCRP018.pdf/$FILE/FCRP018.pdf

nitrogen deposition; recovery from sulphur deposition; forest management changes). Evidence for the influence of climate on soil carbon (Section 3.3.1) remains divergent but the consensus is that land-use change is currently the dominant factor [Low confidence]. Nevertheless, in some upland locations temperature change effects on soil respiration may be contributing to losses of carbon to the atmosphere (as CO_2) and water bodies (as DOC), especially in areas of unvegetated peat.

Relationships between climate and GHG emissions (or uptake) are indirectly mediated through interactions between climate change and land-use change (Section 3.3.2). A shift in land use from intensive agriculture (particularly arable) to forestry or rough grazing can allow carbon sequestration whilst change in the opposite direction will result in net carbon emissions. In addition, agricultural land uses are also significant emitters of methane (from livestock) and nitrous oxide (N_2O) from fertiliser application and livestock, both important GHGs.

However, at present the main factors influencing land-use change are policy (especially CAP) and crop or livestock prices. Hence, as has been shown for Scotland using NFI data (Brown et al., 2014), most recent new tree planting has occurred in the uplands on less agriculturally productive land. This planting has often been on wetter carbon-rich soils (although planting on deep peat/blanket bog is now proscribed by the Forestry Standard) which incurs initial carbon emissions due to disturbance, particularly where drainage is required.

Carbon stocks in coastal sediments accumulate over the course of hundreds to thousands of years, but upon disturbance they may be converted into atmospheric CO_2 within a very short period of time (Pendleton, 2012). While offshore shelf sea sediments store little carbon per hectare, because of their spatial scale they hold considerable amounts of carbon overall. The large spatial scale occupied by some of the blue carbon sinks presents innate difficulties when considering habitat management and restoration. In some cases, the extent of areas already lost to disturbance or conversion are larger than the area currently covered by these habitats. Most studies on the effect of climate change on coastal habitats have focused on the loss of carbon sequestration capacity, while the conversion of pools of already stored carbon has been largely overlooked (Pendleton, 2012).

Future risks and opportunities

Future projections of LULUCF GHG emissions and removals do not include a climate component because of fundamental uncertainties, even with regard to present-day conditions. Enhanced storage of carbon due to a longer growing season and CO_2 fertilisation is likely to be countered by a loss of carbon from enhanced soil respiration due to higher temperatures. At present it is difficult to evaluate which will be the dominant process (Section 3.3.1) and it will also depend on changes in soil water regimes. Nevertheless, in currently vulnerable areas (e.g. unvegetated or degraded peat), higher temperatures and the likelihood of drier summers, particularly in the eastern side of the UK, would be likely to substantially increase the loss of carbon stocks. Hence, the role of land management in enhancing soil resilience, by maintaining peat-forming vegetatation cover (particularly key species such as *Sphagnum* on peatland) for example, or limiting tillage during cultivation, will be important for both climate adaptation and mitigation objectives.

Expected future changes in tree growth (Section 3.3.5) also have the potential to affect carbon stocks (both positively and negatively) but are subject to considerable uncertainty and are dependent on individual species' responses and their resilience to other climate-related risks (e.g. drought; pests and pathogens; wildfire). The type of woodland planted in the future (i.e. confier/broadleaved) will have a key bearing on future carbon sequestration and storage. In addition, tree stocking densities and hence carbon storage may be reduced through drought-related mortality and impacts from pests and pathogens, as well as through managed reductions in order to enhance resilience.

Coastal carbon stores such as inter-tidal habitats will be increasingly affected by sea level rise (as discussed in Section 3.5.) Carbon could be lost where inter-tidal habitats are unable to naturally migrate inland in response to rising sea levels due to the presence of fixed coastal defences or other land uses ('coastal squeeze'). However, where inter-tidal habitats are able to migrate inland, there could potentially be carbon gains through accumulation of sediment.

Climate-related changes in land use are highly likely to influence future carbon stocks in both soils and vegetation. Projecting future land use is highly uncertain, as there are a wide range of global, demographic and localised factors involved. The UK NEA (2011) developed a series of socio-economic scenarios to assess possible land-use changes up to the 2060s. Under one scenario (*Green and Pleasant Land*) a preservationist attitude arises with low-input agricultural systems adopted to conserve a range of ecosystem services. In the *World Markets* scenario, the focus is on achieving high economic growth and agricultural production by removing barriers to trade. The NEA estimated changes in UK land cover (from a 2007 baseline) by the 2060s under these different scenarios (Table 3.6). A high and low climate scenario were also considered for each socio-economic scenario.

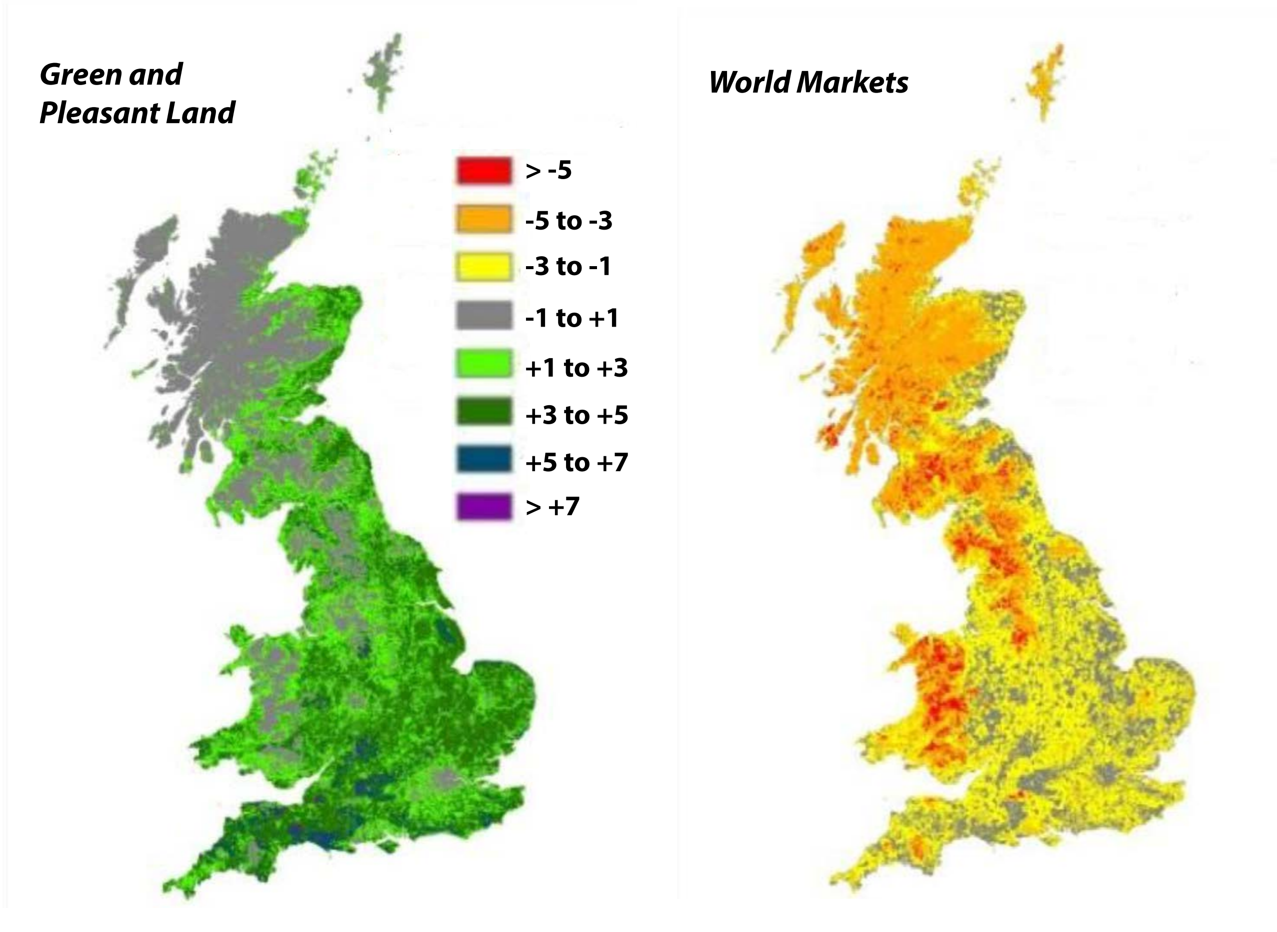
Table 3.6. Percentage change in land cover by NEA scenario and climate scenario.

Land cover type	Socio-economic and climate scenarios			
	Green and Pleasant Land		***World Markets***	
	High	**Low**	**High**	**Low**
Arable	–8%	–6%	+2%	+4%
Improved grassland	–7%	–6%	–7%	–6%
Broadleaved woodland	+5%	+6%	–1%	–0.6%
Conifer	–1.5%	–2%	+1%	–0.3%
Urban	No change	No change	+8%	+8%
Semi-natural grassland	+10%	+6%	–2%	–3%
Upland	+0.8%	+1.0%	–2.5%	–3%

Source: Suggitt et al for the ASC (2015) using data from the UK NEA (2011).
Notes: Both NEA scenarios assume a global decline in resource availability, but differ in terms of their assumptions about land management priorities, public environmental awareness, well-being, ecological footprint and the levels of governance and adaptation capacity. Estimates of the carbon stock of different land cover types were used to calculate the amount of carbon under each scenario combination, thereby representing final "equilibrium" changes in carbon content following the change in land cover, and as such can take anything from 50 – 75 years to complete. Hence, the influence of climate on soils and vegetation was not modelled mechanistically (e.g. vegetative succession or growth) and further work is required to provide more robust estimates of the combined effects of climate and socio-economic change on future land use.

For the *Green and Pleasant Land* scenario, soil carbon stocks in Great Britain are projected to increase by 7% under a low climate scenario, rising to 9% under a high climate scenario (Figure 3.21). Under *World Markets,* intensification of farming results in a 9% loss of soil carbon stocks (low climate scenario) across Britain by the 2060s. Wales has the highest rates of soil carbon loss overall at 14%.

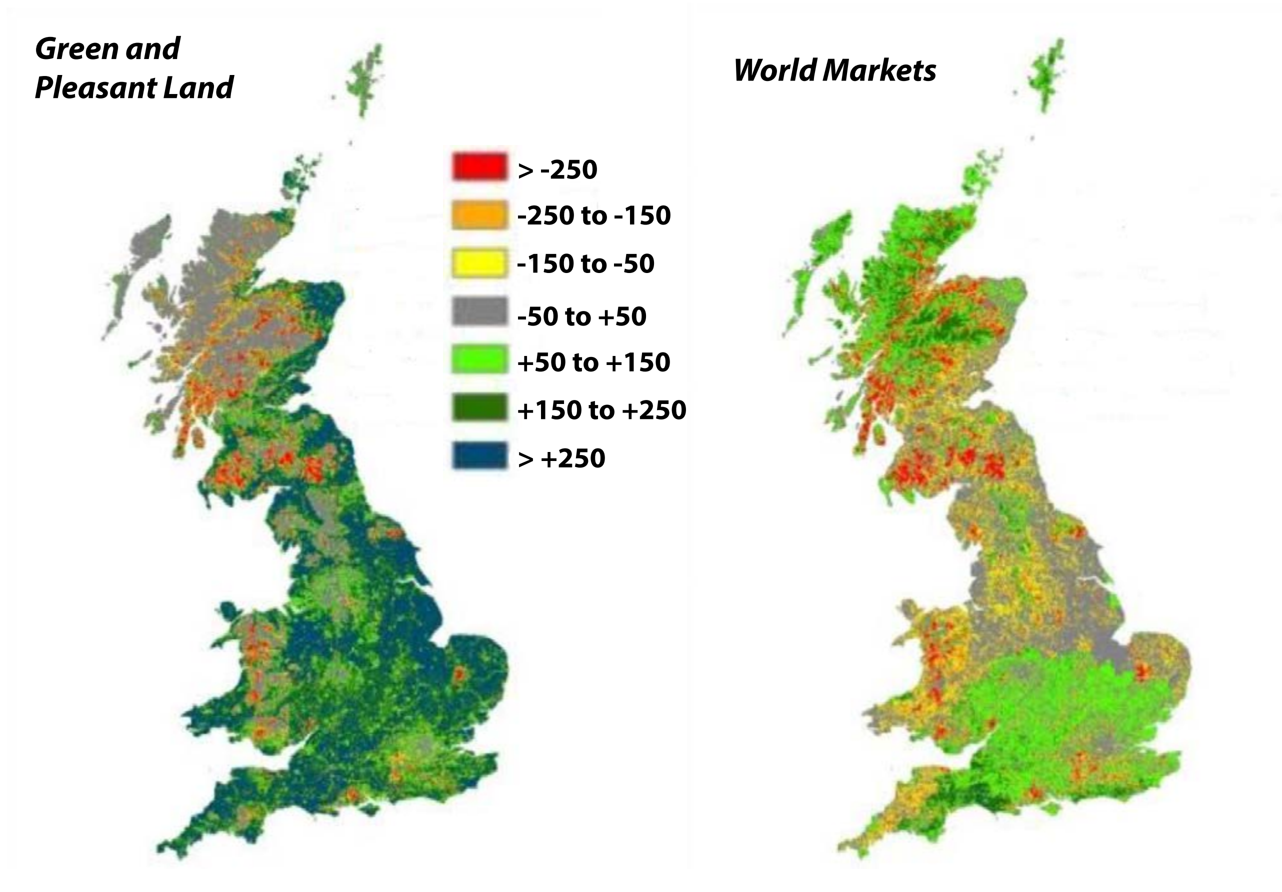
Figure 3.21. Projected changes in soil carbon stocks by the 2060s under the *Green and Pleasant Land* and *World Markets* socio-economic scenarios and a high climate change scenario



Source: Suggitt et al (2015) for the ASC.
Notes: See above (Table 3.7). Units are tonnes of Carbon per 1 km x 1 km grid cell.

Projected increases in forest carbon stocks are significant higher under the *Green and Pleasant Land* scenario, increasing by 28% under a low climate scenario and 23% under a high climate scenario (Figure 3.22). This reflects an assumption that broad-scale afforestation programmes are largely successful at replacing farmland that is no longer economically viable due to reduced water availability. However, under *World Markets*, forest carbon is projected to reduce by 8% across Britain, although there are gains in lowland areas under the high climate scenario which reflects the assumption that water availability and soil aridity limits intensive agriculture in these areas.

Figure 3.22. Projected changes in vegetation carbon stocks by the 2060s under the *Green and Pleasant Land* and *World Markets* socio-economic scenarios and a high climate change scenario.



Source: Suggitt et al (2015) for the ASC.
Notes: See above (Table 3.7). Units are tonnes of Carbon per 1 km x 1 km grid cell.

Adaptation

The constituent countries of the UK have a number of strategic targets to increase forest cover over the coming decades (Table 3.7). If delivered, these planned expansions represent an important opportunity for increasing carbon sequestration and carbon stocks.

Table 3.7. Forest cover estimates of UK countries and woodland creation targets

Country	Cover estimate	Target
England	10%	12% by 2060
Scotland	18%	25% by 2050
Wales	15%	Increase of 4.8% by 2030 (aspirational target)
Northern Ireland	6%	12% by 2055

The restoration of degraded peatlands will increase the resilience of these habitats to climate change and potentially reduce carbon losses. See Sections 3.2.1 and 3.3.1 for further discussion.

3.7.2 Pests, pathogens and invasive species

Synthesis

This risk is identified as cross-cutting because although individual threats may be recognised by specific sectors, some can spread from one sector's interests to another's. This is particularly the case for some emerging threats.

Low winter temperatures currently act as a climatic control on many pests and diseases that pose a risk to trees, crops, livestock and native wildlife. Milder, wetter winters could increase the risk from damaging pests and pathogens and may also facilitate the spread of invasive species. Any increase in pests, pathogens and invasive species has the potential to further threaten native wildlife already stressed by other pressures and could have serious implications for agricultural production (crops and livestock), forestry, and ecosystem services such as landscape amenity value.

Continued action is needed, including preventative measures and surveillance, with increased co-ordination across sectors. More risk-based research is also required on the potential consequences of changing climatic conditions for the spread of pests, pathogens and invasive non-native species. It will be important to monitor whether incidents and impacts are becoming more common and the extent to which climate change is a factor.

Context and policy

There is generally a high level of scientific consensus that the risks from pests, pathogens and invasive non-native species (INNS) could significantly increase due to climate change, but the risks from individual species and pathogens are highly variable. A shift to warmer winters implies a greater survival rate for INNS and pathogens that have previously been constrained by low temperatures [High confidence]. Much of the risk evaluation to date has been based upon possible impacts on human health or productivity in forestry and agriculture, rather than on natural ecosystems processes, and therefore may be underestimated. Risks have the potential to be interlinked as pests can act as hosts for the transmission of disease.

The risks from pests, pathogens and invasive species can affect multiple sectors. Risk management measures from one sector can affect another, and can often only begenuinely effective if implementedacross sectors. Diseases occur due to pathogenic micro-organisms (e.g. bacteria) and viruses. To assess risk from diseases, the pathway of transmission is used to distinguish vector-borne, water-borne and air-borne diseases. Some of these diseases are zoonotic in that they can be passed from animals to humans which has implications for human health (Chapter 5).

Particular emphasis is placed on recognising emerging infectious diseases (EIDs) which are caused by pathogens that: (i) have increased in incidence, geographical or host range; (ii) have changed pathogenesis; (iii) have newly evolved; or (iv) have been recently discovered or newly recognised. More is known about domesticated livestock EIDs and crop plant EIDs than EIDs in the wild. Most outbreaks occur when a disease moves to a new host but the possibility of "wildcard" events due to genetic recombination to form novel pathogens (as with the H1N1 virus) cannot be discounted, notably for plant diseases.

Disease emergence can occur through gradual changes in climate (e.g. change in vector distribution; increasing water or temperature stresses on plants). Alternatively, emergence can be related to greater frequency of specific weather events (e.g. dry weather tends to favour insect vectors and viruses, whereas wet weather favours fungal and bacterial pathogens). Thus, climate change can lead to the increased emergence of pre-existing pathogens as major disease agents or can provide the climatic conditions required for introduced pathogens to emerge.

Vector-borne diseases are those in which an organism (notably insects, ticks, or mites) carries a pathogen from one host to another, generally with increased virulence of the pathogen in the vector. Some vectors are able to move considerable distances and pathogens can be introduced to new geographic areas by: (i) travel of humans and international trade; (ii) animal movement, for instance, of livestock; (iii) migratory birds; (iv) changing agricultural practices; (v) wind.

Water-borne diseases are those that are predominantly transmitted through infected water containing pathogens, either through direct contact (e.g. bathing or washing) or consumption (in food or from drinking).

Air-borne diseases are caused by pathogens transmitted through the air and depend on several physical variables endemic to the infectious organism. The most notable environmental factors for transmission of air-borne diseases are temperature and relative humidity, which influences the spread of the droplets containing the infectious particles, together with winds and human or animal behaviour. Ultraviolet light is harmful to both viruses and bacteria, meaning that there can be a seasonal component for incidence. Plant fungal diseases are also affected by rain "quality", that is, the rate of fall rather than the total amount.

Pests represent either native or non-native (alien) organisms that cause damage to native species, ecosystems or productivity of the rural sector. Of particularly high concern is the introduction of INNS that can modify relationships in native ecosystems to the detriment both of biodiversity and the wider benefits (ecosystem services) that humans receive.

Specific biosecurity measures in the agricultural and forestry sectors have been developed to address diseases and pests (see Sections 3.3.3, 3.3.4, 3.3.5).

Under the definitions of the Convention on Biological Diversity (CBD), INNS are identified as those introduced by human agency. The CBD commits parties to prevent the introduction of, or enable the control or eradication of, such species. The Bern Convention on the Conservation of European Wildlife and Natural Habitats obligates "strict control" on INNS and similar provisions are also made in the EU Habitats and Birds Directives and the EU Invasive Alien Species Regulation (2015).

The Great Britain Invasive Non-native Species Strategy recognises that the most cost effective and least environmentally damaging approach to INNS is through preventing the introduction of such species into the wild, rapid response and early intervention. The risk assessments and actions plans prepared by the GB Non-native Species Secretariat include reference to climate change. Long-term horizon-scanning exercises to identify future threats are also being developed.

Pressures from invasive non-native species are also a key component in assessing good ecological status for the WFD and in the MSFD. Insects can constitute a nuisance in law (civil law or statutory environmental protection), particularly where they interfere with the use or enjoyment of land.

Current risks and opportunities

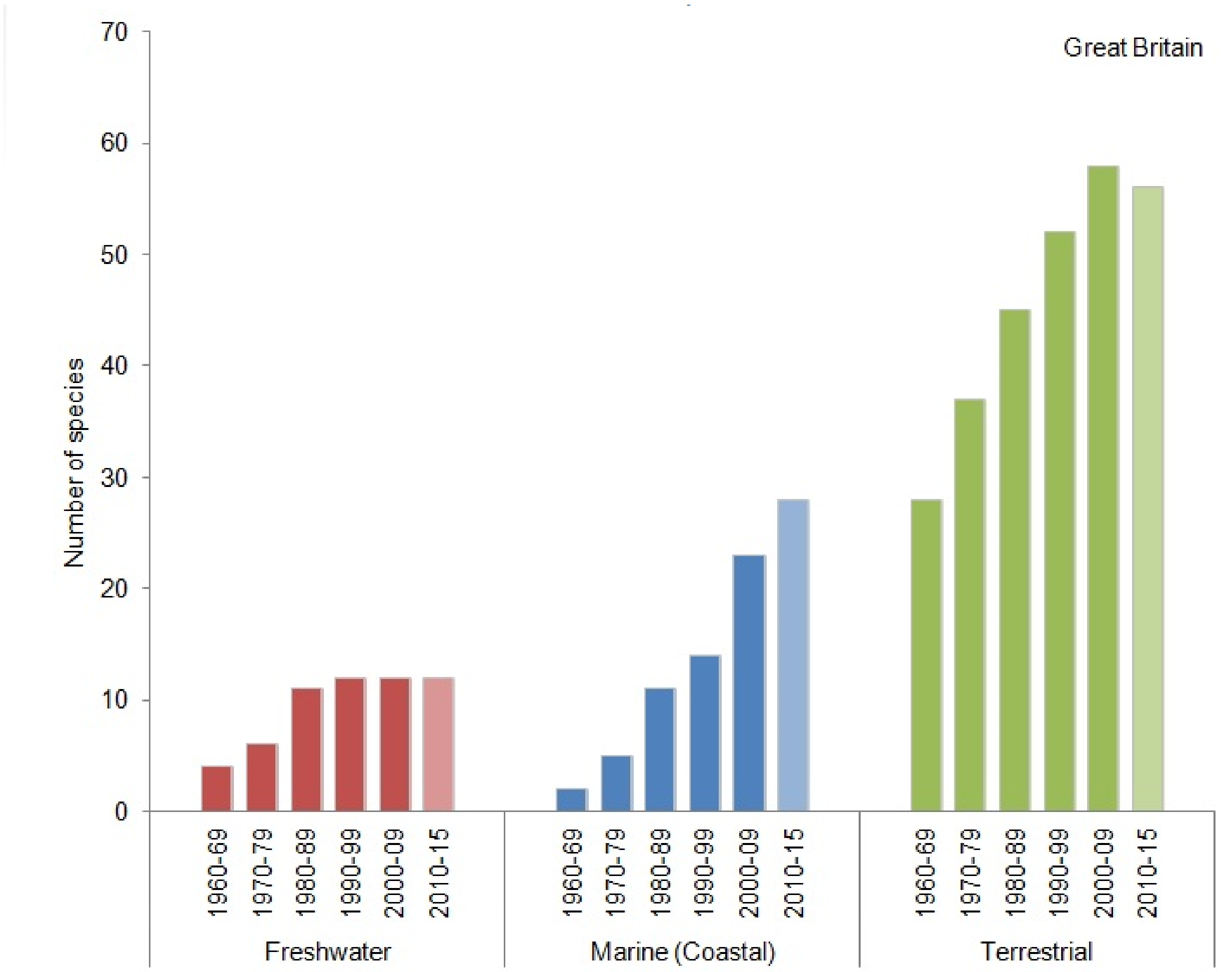
For pests and diseases that are already well known in terms of their risks for the UK, established mechanisms are in place to improve awareness, surveillance and reporting. Previous outbreaks mean that links with weather and climate conditions are usually known, which can be used to further enhance awareness of high-risk conditions.

However, complex interrelationships of multiple factors mean that cause–effect relationships are often not fully established. For example, parasitic nematodes represent one of the most pervasive and significant challenges to grazing livestock, and their intensity and distribution are known to be strongly influenced by climate (van Djjk et al., 2010; Heffernan et al., 2012). Nevertheless, the exact causes of the current pattern of increasingly widespread low-level infection with sporadic high-intensity outbreaks are poorly understood due to limited longer-term data availability, although changing climate has been suggested as a possible driver.

For EIDs, the primary risk is considered to be anthropogenic introduction into the UK with the risk level potentially further elevated by expansion of global trade patterns. Containment measures therefore require international co-operation and knowledge exchange programmes to improve awareness of the changing distribution of risk, the factors contributing to risk and the relative success of different measures to manage risk. Each of these different components will vary with the specific nature of different pathogens.

With regard to non-native species, there has been a significant rise in recorded species across all environments (Figure 3.23). Of the 3,050 non-native species recognised and recorded in Great Britain, 1,919 are considered to be established and of those 179 are considered to be invasive non-native species (i.e. exerting a negative impact on nativebiodiversity).

Figure 3.23. Trends in recorded Invasive Non-Native Species in Great Britain



Source: Botantical Society of Britain and Ireland, British Trust for Ornothology, Centre for Ecology and Hydrology, Marine Biological Association, National Biodiversity Network Gateway.
Notes: As established on or along more than 10% of land or coastline. The last time period covers a shorter period than the other bars (2010-2015).

Specific issues related to the risks from marine non-native species, pests and pathogens are described in Section 3.6.1.

Future risks and opportunities

Four key factors will influence future risks.

- First, climate change will act directly on pests, pathogens, hosts or vectors. In particular, milder winters, higher nocturnal temperatures and higher overall annual temperatures will enable increased winter survival of plant pests, pathogens, accelerated vectors and pathogen life cycles, as well as increased sporulation and infectiousness of foliar fungi. It is also possible that so-called 'sleeper species' that are not currently a problem become more invasive in future.
- Second, climate change will affect the habitats present in a region, the community of hosts that can live in them and their life cycles.
- Third, climate change is likely to modify dispersal patterns. For example, reduction in Arctic sea ice may open up a new sea route from east Asia to Europe which may increase threat of invasion by Pacific marine species as this route will be faster and cooler than the current route through the tropics.
- Fourth, climate change will act indirectly on other drivers of disease such as through land-use change, trade patterns, migration and so on.

A summary of anticipated changes in disease is provided in Table 3.8. Vector-borne diseases have been identified as the most likely to change in risk level due to climate change (Brownlie et al., 2006) [Medium confidence]. Climate sensitivity of vector-borne diseases and their risk of transmission are influenced by:

- Changes in temperature and precipitation, directly by affecting pathogen–host interaction and indirectly through ecosystem changes and species composition.
- Climate warming influencing the spread of vectors into new areas that were previously too cold.
- A change in the incubation period of pathogens in invertebrate vectors.
- The movement of human population due to climatic factors, which can expand the distribution of pathogens.
- A decline in biodiversity altering predator–prey relationships, particularly a decline in the predators of vectors that can result in increased vector populations.

Ruminant livestock are at considerable risk of bluetongue, a viral disease that has already spread into southern Europe. Recent climate warming has influenced a spread in vector distribution of the disease, as well as the appearance of novel vectors (Guis et al., 2012). For parasitic nematodes in livestock, Fox et al. (2015) demonstrate that small changes in climatic conditions around critical thresholds may result in dramatic changes in parasite burdens; identifying these thresholds may be critical for risk management.

Table 3.8. Possible implications of climate change for major animal and plant diseases	
Animal diseases	**Plant diseases**
Increased risk from expansion of vectors for bluetongue, West Nile virus, canine leishmania. Reduced risk of air-borne spread of foot-and-mouth. Possible reduction in fascioliasis, although wetter winters may enhance risk in some areas. Unpredictable change for anthrax, haemochosis, summer mastitis. Little or no change for avian influenza, brucellosis, BSE, classical swine fever, locally spread foot-and-mouth, mastitis, Newcastle disease, rabies, salmonelloses.	Uncertain but possible increase in risk of root and stem base fungal crop pathogens (e.g. take-all, eye spot). Decrease in some stem and leaf fungal pathogens (leaf blotch, net blotch, yellow rust, late blight, light leaf spot, phoma stem canker), although may be countered by earlier disease onset. Increased risk of powdery mildew. Unpredictable change in brown rust. Milder winters may favour vector-borne viral diseases (e.g. barley yellow dwarf virus).
Source: Brownlie et al., 2006.	

Most studies have concluded that insect pests will generally become more abundant with climatic warming due to several interrelated processes, including range expansions, phenological changes, increased overwinter survival and population changes..

A continued rise in CO_2 will affect pest species directly and indirectly, depending on the characteristics of pest and host species. Aphids (which feed on the phloem of their host-plants using hypodermic-like mouth-parts) can transmit plant diseases caused by bacteria and viruses; mild winters and warm springs favour higher aphid survival and development rates, and hence may increase the risk of plants being infected . Wetter winters may increase the risk from liver fluke (fascioliasis), which is vectored by water-sensitive lymneid snails (Fox et al., 2011).

The most vulnerable ecosystems are likely to be those where a “keystone” species is threatened, such as species that mediate competition between herbivores or plants, or nitrogen fixation in soils. In this case ecosystems are at risk of a domino effect, where changing species interactions can result in the disruption of ecosystem function and key services.

The complexity and resilience of ecosystems normally acts as a buffer against such extreme impacts, but human-related stresses (e.g. simplification of habitats to facilitate intensive production systems) can erode this resilience. It has been postulated that species with low rates of reproduction, and that have invested heavily in disease immunity, are likely to be lost first when ecosystems are degraded by human actions. This means that those remaining have higher reproductive rates or have invested less in immunity, thereby providing an increased number of disease hosts (Keesing et al., 2010).

Adaptation

In addition to measures to restore or maintain natural ecosystem resilience against these risks, the key requirements for adaptation are biosecurity surveillance and early warning systems; these may also require further international co-operation to better anticipate new or emerging threats. The development of risk-ranking approaches to categorise the high number of potential and actual threats at different stages of emergence and invasiveness is likely to be needed. A further precaution to counter the spread of pathogens or INNS will be epidemiological modelling, based upon present and future landscapes, to identify hot spots and barriers for particular species. In some cases, this may include the design of landscape-scale measures to reduce dispersion of INNS or pathogens.

3.7.3 Landscapes and sense of place

Synthesis

Landscapes are the product of both natural and human (cultural) processes that help shape people's sense of place and local identity. [35] Changes in the character of the landscape are an important, but often under-recognised, influence on individual and shared attitudes to change.

Landscape character assessments aim to identify and understand the key features that combine to make up each unique landscape. National-scale mapping projects show how shifts in land cover and land use have changed landscape character in many locations across the UK over recent decades. Climate change is likely to have been a contributing factor, both directly through biophysical changes in land cover and the composition of species, and indirectly by influencing land use decisions. However, we know much less about how people have perceived and interpreted these changes.

As highlighted in the preceding sections of this chapter, natural changes in the landscape will continue to occur and these will interact with human responses (climate-related or otherwise) in very diverse ways reflecting the large variety of UK landscapes. Although research on this subject is increasing, stimulated in part by the UK National Ecosystem Assessment, understanding the influence of climate change on landscape character is at an early stage. However, climate change is increasingly likely to challenge attitudes that do not recognise the dynamic nature of landscapes.

Landscape character assessment and other tools (such as landscape visualisation techniques) can play an important role in capacity-building initiatives that aim to enable and encourage a wider awareness of the role of landscapes in inclusive climate change decision-making.

Context and policy

The natural environment provides important, but often intangible, inspirational, aesthetic, educational, cognitive, spiritual, recreational and amenity benefits to people. In the context of an ecosystem services framework, these benefits are usually referred to as cultural ecosystem services, although unlike other ecosystem services they are the consequence of an enourmous number of diverse interactions between people and the environment that have produced the distinctive landscapes (and seascapes) of the UK. The awareness and appreciation of wildlife in its natural setting is, for many people, a core component of the cultural benefits provided by nature. Changes in species and habitats therefore have a potential impact on the interactions between people and the natural environment.

Landscapes represent the unique mosaic of natural and human elements that distinguish a particular area. The numerous benefits, tangible and intangible, are often described in the context of "sense of place" (UK NEA, 2014) and have important links with human identity, health and well-being (Chapter 5). Changes in the natural environment have important implications for

[35] This section also includes the amenity value of the marine environment ("seascapes"). Built heritage and the historic environment is covered in Chapter 5.

such relationships, although it is important to recognise that landscapes are dynamic features that have evolved over previous millennia.

The importance of local landscapes is signified by the European Landscape Convention (ELC), which the UK has ratified. The ELC highlights that landscapes are a product of natural and cultural factors, with all landscapes having their own distinctive value. Areas of particularly distinctive landscape value that are deemed to be of national importance are designated as either National Parks or Areas of Outstanding Natural Beauty (AONBs), as well as National Scenic Areas in Scotland. A common tool that has been used throughout the UK to synthesise the key features of different geographic areas is Landscape Character Assessment (LCA).

Current risks and opportunities

National-scale mapping projects have shown how the land has changed in terms of both land cover (biophysical component) and land use (socio-economic component). These planned and unplanned changes include the expansion of woodland cover, the specialisation and intensification of agricultural land, and dynamic successional changes in semi-natural habitats (e.g. heathland to grassland). These changes have acted to modify both the individual features that make up the landscape and the landscape mosaic itself. As a consequence the landscape character of some areas has substantially changed whilst others continue to look, to a greater degree, as they have in the past.

Climate change is likely to have been a contributing factor in these changes, either directly through its influence on land cover, or indirectly by favouring some land uses over others in specific locations. In addition, responses to climate change, both through adaptation and mitigation, are now also becoming a significant influence on landscape change. Thus far, the effects have been most strongly evident through mitigation measures, such as the development of wind farms and other renewable energy initiatives. However, adaptation is also an important driver behind the planned expansion of woodland in the UK (Section 3.3.5). Coastal landscapes are being modified through both planned responses (e.g. managed realignment) and unplanned responses (e.g. no active intervention) to sea-level rise (Section 3.5).

Landscape perception is a personal and subjective experience, therefore detailed survey data are required to derive overall conclusions. Although there has been research on general perceptions of environmental change (Upham et al., 2009), to date there has been little research on how people have perceived climate change in the landscape. Some relevant information has been collated through the monitoring of wildlife, particularly through citizen science projects (e.g. Nature's Calendar survey by the Woodland Trust). In Scotland, climate change effects on the landscape have been used as a way of engaging local communities in thinking about both landscape change and adapting to climate change.[36]

As yet, therefore, we have rather limited evidence on how people perceive change and whether they see climate change as a risk or on opportunity to their local landscape or to iconic landscapes of national/international importance.

Future risks and opportunities

Changes in land cover and land use will undoubtedly continue to occur with the magnitude and pace of climate change being a key factor influencing this change (Freque-Baxter and Armitage,

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[36] www.snh.gov.uk/publicationsdata-and-research/publications/search-the-catalogue/publicationdetail/?id=1869

2012; Twigger-Ross, 2013). For example, any increases in incidences of drought are likely to affect landscapes dominated by semi-natural habitats as well as urban settings due to the impacts on greenspace and street trees. Such changes are likely to increasingly challenge attitudes that aim to maintain current landscape character as the status quo.

Adaptation

Partly because of their intangibility, changes in landscape character are often systematically undervalued in land use change assessments and strategic planning documents. The importance of landscape character and 'sense of place', particularly to local people and communities, can therefore be neglected in land use decisions. This can have implications for adaptation, as cultural perceptions of landscape can be both an enabler or barriers for the implementation of adaptation actions (e.g. Adger et al., 2011).

To date, tools such as LCAs have not been widely used to integrate future projections of land use and land cover change. However, LCAs have the potential to help communities explore how change may directly and indirectly impact on their local landscape (e.g. Dockerty et al., 2006; Wang et al., 2015). In many locations, climate change can be perceived as both a risk and an opportunity for local landscapes depending on future aspirations and perceptions of change.

3.8 Conclusions

3.8.1 Discussion

The evidence reviewed in this chapter demonstrates the complexity of climate change risks to the UK's natural environment. Ecosystems are sensitive to multiple climate variables and these variables interact with a range of other factors to define current and future risk.

The evidence strongly suggests that climate change is already having an influence on the natural environment and the goods and services it provides. We can say with good confidence, based upon current climate sensitivity and future climate projections, that in general risks (along with some opportunities) are very likely to increase.

There is strong evidence of changes to species distributions in terrestrial, freshwater and marine ecosystems. This trend is related to increases in both air and water temperatures for which the climate change signal is now strong enough to be detected against natural variability. By contrast, a trend is more difficult to detect for risks associated with changing precipitation or wind patterns. Although there is evidence of wetter winter conditions in the north and west of the UK in recent decades, this cannot be conclusively attributed to climate change. Similarly, a shift to a more westerly airflow, with implications for exposed locations, especially on coasts and mountains, has not been established as a long-term trend.For the coastal zone, there is high confidence of ongoing, and possibly accelerating, sea-level rise.

These trends suggests there is a clear rationale for ensuring action is taken now to build the resilience of the natural environment so it is more able to accommodate change in the future. Anticipatory adaptation is necessary because of the complexity of the systems involved and the long lead times from policy development to implementation on the ground. It is also usually much easier to take action now than to try and restore biodiversity and ecosystem functions and services once they have been irreversibly degraded.

Climate change risks are exacerbated because they occur in combination with existing pressures, particularly for biodiversity, soils and water. A key finding from this chapter is that many risks are interrelated, therefore adaptation responses need to be co-ordinated so that risks are not tackled in isolation. There are trade-offs between different sectors in the way risks to soils, land and water are being managed. In the absence of further planned adaptation it seems likely that reactive responses will dominate, which will potentially exacerbate inefficiencies and resource depletion. For freshwater, the requirement to achieve good ecological status of water bodies under the EU WFD is an important driver for adaptation. But maintaining good status in the future will require further integration with land use and holistic catchment-scale approaches. With the exception of the Land Use Strategy in Scotland, there are no similar approaches to the WFD for delivering integrated adaptation in respect of soil and land resources.

Across many risks, notably in relation to agricultural and forest productivity, differences between the north/west compared to the south/east of the UK have been highlighted, with climate change seeming, as a broad generalisation, to provide new opportunities in the former and increased risks for the latter.

Although the evidence for this chapter strongly suggests that risks will outweigh opportunities, it is important for adaptation that any potential benefits from a changing climate are realised. This is particularly the case for changes in the current distribution of species, crops and land uses. It is also possible that some ‘risks’ could become opportunities, or at least reduced risks, if proactive adaptation measures are implemented now, as for example with crop productivity and yields. This will not occur with simply continuing ‘business as usual’ strategies, as more co-ordinated investment is required to maximise potential opportunities.

In the marine environment, although the EU MSFD recognises the need for adaptability due to changing environmental conditions, in practice this is often difficult to achieve. The interpretation of ‘fair’ national quotas for fishery stocks, given the observed and projected changes in species distributions, is a particularly challenging issue.

This chapter has aimed to develop a systematic approach to risk assessment, with a view to aiding the identification of policy priorities for forthcoming national adaptation programmes. In this regard, further work is still required as prevailing attitudes in both science and policy tend to favour taking a sectoral approach. Nevertheless, a series of cross-cutting issues have been identified that should act as a stimulus for further cross-sectoral initiatives.

Particularly notable is the importance of co-ordinated efforts to monitor and address the risks from pests, diseases and INNS. Climate change is very likely to lead to an increase in these risks, but there is significant uncertainty as to the nature of new threats. The degraded condition of many ecosystems is also likely to increase virulence when pests, diseases and INNS become established. In some cases, however, non-native species may provide opportunities, such as by restoring ecosystem function to degraded sites where native species can no longer be re-established. Furthermore, the processes of change are likely to challenge traditional attitudes regarding distinctions between ‘native’ and ‘non-native’.

Finally, there is a growing recognition that taking an ecosystem-based approach has the potential to deliver more synergies between climate change mitigation and adaptation. Actions to protect and restore natural carbon stores, such as woodlands, peatlands and saltmarsh, can not only contribute to the UK’s efforts to reduce GHG emissions, but also directly increase the resilience of the natural environment and communities to climate hazards such as flooding and erosion. There is considerable scope to further develop measures that link climate adaptation and mitigation responses in more synergistic strategies.

3.8.2 Priorities for action in the next five years

The summaries at the start of each section of the chapter highlight specific measures that are important for addressing risks and opportunities. Many measures can be considered low-regret strategies, because they enhance resilience to climate-related risks (particularly extremes) to some extent, regardless of the degree of future climate change.

There are several examples, notably for biodiversity, in the coastal zone and for water services, where policy initiatives have not yet been fully implemented on the ground to a stage where it is evident that risks are being reduced. In these cases, the main barriers to achieving enhanced resilience appear to be a lack of incentives to support implementation at a local level.

A further necessary step is to be able to accommodate change, as much of the current interpretation of 'resilience' actually in practice purports to maintaining the status quo. However, the evidence suggests that policy measures to accommodate change are much less developed at present. One of our key findings is that attempts to maintain the status quo are very likely to exacerbate some aspects of risk, because some degree of change is inevitable .

The natural environment is particularly vulnerable to crossing thresholds that can potentially result in non-linear ecosystem responses, with the risk of major implications for human well-being. As resilience is reduced or lost, the buffering the natural environment provides against hazards such as flooding, erosion or pests reaches a point where it is sharply reduced. ,It should therefore be a priority to sustain and monitor the capacity of the natural environment to continue to provide vital buffering against climate hazards, and to understand more about how this capacity could be further enhanced . These issues are particularly evident in the coastal zone where the science is very confident that sea level rise will continue for many decades (and even centuries) to come. The key uncertainty is the magnitude (or rate) of future sea-level rise and the prospect of passing a threshold beyond which risks rapidly increase. Hence a more transformational strategy to sustain coastal assets in the UK is likely to be necessary to be better prepared for future change and avoid potential threshold effects.

Similar strategic priorities can be highlighted for land-use planning which could deliver wider benefits than occur through piecemeal or sectoral strategies that tend to favour the status quo. This will be particularly necessary to accommodate adaptation initiatives such as'space for nature' (to facilitate species movements), 'space for water' (to alleviate downstream flooding), or coastal realignment, none of which can be implemented in isolation due to the need to integrate multiple land uses. Such initiatives also have long lead times and therefore responses need to be implemented now in order to be in place to address climate risks, and to be integrated with strategies for communities and built infrastructure. Policy development for the Land Use Strategy in Scotland as an outcome of the Climate Change Act 2009 (Scotland) can be seen as a positive example in this context, albeit in its early stages in incorporating adaptation in addition to climate mitigation requirements.

A major challenge that emerges from this chapter is reconciling production systems, particularly in agriculture and fisheries, with the other benefits that the land or sea provide. To facilitate this, we have followed an ecosystem approach to identify multiple ecosystem services that are in many cases supplied by the natural environment, often to people living some distance away in towns and cities. With climate change likely to influence changing demands in towns and cities, this provides an even stronger rationale to incorporate the integrated role of land and sea into planning approaches so that changing patterns of supply and demand for ecosystem services can be accommodated.

Knowledge of many ecosystem services, although considerably improved by initiatives such as the UK NEA, remains limited and it is important that further research is carried out on their spatial and temporal variability and trends. Nevertheless, the crucial role of these services is now increasingly recognised (e.g. alleviation of flood and erosion risk) and uncertainty in knowledge of service flows should not act against a recognition that UK climate resilience is strongly related to having adequate stocks of natural capital and the protection of natural assets.

Important gaps in evidence exist for our knowledge of existing change and, as a result, our understanding of future pathways and the efficacy of different types of adaptation intervention. Therefore, at a time of likely unprecedented change, it is particularly important that existing monitoring initiatives (e.g. Environmental Change Network, National River Flow Archive, Nature's Calendar, etc) are continued and complemented by additional monitoring in under-represented areas (notably the uplands). We also strongly suggest that more effort should be made to ensure timely release of this monitoring data, including national-scale climate data, so that research can keep pace with the demand for evidence to support decisions in policy and practice.

3.8.3 Knowledge gaps

This evidence review has identified a number of important evidence gaps for each topic area. Addressing these specific gaps should occur in tandem with policy responses, whilst recognising the level of confidence in existing knowledge and the need to take a precautionary approach. However, as concluded above, there is also a strong case for more integrated approaches that investigate evidence across traditional sectors or academic disciplines, including the potential benefits from ecosystem-based approaches to climate adaptation and mitigation.

An increased number of studies are taking a cross-cutting approach (e.g. Holman et al., 2008; Ray et al., 2014; Brown and Castellazzi, 2014; Brown et al.,2015; Harrison et al., 2015). However, research still predominantly favours traditional scientific approaches rather than interdisciplinary or transdisciplinary research. In addition, those studies that have addressed this broader agenda have tended to evaluate trade-offs or synergies between risks that are more easily quantifiable based upon existing data, and there has been more emphasis on risks rather than opportunities. These issues are particularly pertinent in evaluating extreme events which typically expose systemic interdependencies across sectors, but for which the basic climate-related data and the evaluation of risks and responses remains seriously deficient.As highlighted above, more emphasis therefore needs to be placed on data quality and availability in addressing evidence gaps, including adequate description of key assumptions, metrics and indicators.

Terrestrial species and habitats

There is a need to further develop monitoring and analysis to separate the effects of climate and other environmental changes (including air pollution, succession and changing land use and land management) on populations and on communities acrossthe full diversity of UK environments. Better knowledge of climate sensitivity is critical in the design of adaptation strategies for species of high conservation value. Long-term field experiments therefore need to be supported and maintainedalongside other surveillance and monitoring.

There are some particular areas of weakness in current monitoring schemes, including for mammals and many invertebrate groups. There is also further scope to develop time-series population data to understand recent change and develop improved projections. Evidence in the UK provides many more examples of species advancing at their leading range edge than retreating at their trailing edge which requires further investigation as potentially indicative of an 'extinction debt'. Greater monitoring of the population dynamics processes that should eventually lead to range shift at the trailing edge would be desirable to put this pattern in context. New phenological data and information from museums also needs to be incorporated into existing data sets. In particular, there is little work on the phenology of mammals (e.g. hedgehogs, bat species).

More research is required to understand the effectiveness of adaptation strategies, including how ecological networks function in a changing climate, the role of refugia, and the optimum balance between larger, more resilient sites and a larger number of small sites which may facilitate dispersal. There is poor understanding of the dispersal ability and gene flow of many individual species and hence the extent to which they may be able to track changes in suitable climatic conditions. More research is needed to better understand ways in which landscape structure and microclimate interact with species' life history strategies to cause genetic isolation and affect the ability of species to colonise new sites. A key uncerertainity is the extent to which evolutionary change will be able to keep pace with rapid climate change.

Climate-related interactions on key ecosystem processes and functions (e.g. nutrient cycling, productivity, etc) remain a key knowledge gap. The potential implications of climate-driven threshold effects on the provision of key ecosystem services is poorly understood.

Further evidence is required on the relative importance of extreme weather events (drought, heatwave, flooding) in driving abundance and range shifts.The additional role played by changes in CO_2 concentations also needs to be better understood. .

Research is required to improve the design of restored and created habitats to ensure they will be resilient to a range of future climate scenarios.

There is a need to better understand the resource requirements of migratory species throughout their life cycle, especially on stop-over/fuelling sites, and whether such species are vulnerable to climate change (particularly those migrating from Africa).

The area requirements and habitat preferences of species which might colonise the UK are unknown. Further research is also required to understand the potential risk of facilitating the spread of INNS through the expansion of ecological networks

Pollination

The main requirement is to better understand potential asynchronies that may occur and the extent to which they might disrupt pollination, together with the scope for improved pollination through species colonisation. Further research is needed into how climate and non-climate pressures interact and their combined effect in causing loss of species and pollination services.

Soils

There is a need for enhanced monitoring of soil biodiversity, soil properties and soil processes, linked to national inventories, to better interpret long-term patterns of change.

Further research is needed on the impact of climate change on soil formation and nutrient cycling. Knowledge of changes in soil biota and associated risks for soil and human health remains at an early stage and requires further monitoring, experimental and modelling work.

The evidence for changes in SOC requires further research by using standardised methodologies and sampling strategies to evaluate the role of different influencing factors, particularly across different land-use types on both intensively managed and semi-natural land.

Current evidence for changes in both soil erosion and compaction is limited and mainly based upon local examples. Further work is required to develop and validate suitable larger-scale approaches using modelling techniques complemented by survey data. This should aim to identify high-risk areas and the role and uptake of different land-use practices in reducing or exacerbating erosion/compaction risk at different times of the year.

Land availability and capability

The empirical basis for land capability classification needs reappraisal against observed changes in land-use practices and climatic conditions. The use of indicator crops in land capability classification shouldbe reviewed in the light of new crops and varieties that may provide acceptable alternatives. This particularly applies to the use of indicator crops to define drought risk thresholds.

There is limited knowledge of the extent of field drainage systems in the UK and their effectiveness in alleviating wetness constraints in a changing climateThe role of drainage (or in some cases drain-blocking) as an adaptation measure also needs to be more fully evaluated against the full range of risks and opportunities it provides (including flood risk). The influence of changes in land capability on land-use change needs detailed investigation using fine-scale data (e.g. IACS land-use data,[37] soil and topographic data).

Assessments of changes in land capability have assumed that soil properties remain unchanged. Further work should be undertakento identify critical soil properties that need to be incorporated into adaptation actions as measures of good land management practice (e.g. maintenance of soil organic matter).

The system of land capability classification shouldbe extended to include forestry, multifunctional land use and ecosystem services, in order to provide a firmer foundation for integrated land-use planning.

Crop production

Four examples where additional research is needed to facililtate improved adaptation are as follows:

- Predicting the rate of ingress and the incidence and severity of crop pests and diseases under different climate change scenarios through the integration of spatially explicit climate models with models of pest population dynamics and disease epidemiology.
- Models of crop growth, development and yield are available for several important UK crops (wheat, sugar beet, potatoes) and models that allow the predicted scheduling for supply chains of fruit and vegetables also exist. An audit of the available models and their robustness in practice is needed alongside new research to integrate these models with spatially explicit climate projections.
- Adaptive crop improvement in response to future climate scenarios will require theoretical crop genetic/phenotypic ideotypes to be constructed (e.g. flowering time, vernalisation requirements, day-length responses, tolerance of high temperature events) and tested in silico. Genes and gene combinations identified as contributing to the traits required will need to be assembled and hypotheses about proposed phenotypic performance tested under controlled growth-room conditions. This is long-term and time-consuming work that

[37] Integrated Agricultural Control System (IACS) data used for EU CAP payments to farmers.

needs to be started now if applicable results are to be available in time to respond to likely climatic changes.

- The UK (via Defra) has been actively assembling a substantial crop pest and pathogen risk register. Work is now required on two fronts to add value to this exercise: (i) sampling systems and optimum approaches to surveillance and early detection/reporting of exotic (and established) pest and disease occurrences; (ii) analysis of relative probabilities of the introduction and establishment of exotic pests and pathogens coupled with estimations of likely impacts on yield and quality.

Livestock production

Within the UK, the impacts of climate change on livestock systems, and the most appropriate adaptations, are system specific. Underlying soil–climate–fodder crop interactions are also likely to change. There is a lack of knowledge as to how changing climatic conditions will interact and combine, as well as a lack of knowledge regarding the viability of specific local adaptations for livestocksystems. More work is required on extending the current applicability of models of livestock systems to higher temperatures and being able to describe the effect of variability in climate.

In general, there is a lack of research into the effect of a changing climate on beef and sheep system dynamics, although some information available for dairy herds is also applicable to these livestock systems. In addition, the focus to date has been placed on the effects of climate change on highly productive grassland, and there is a lack of knowledge of how climate change will impact on permanent pastures and more marginal grasslands, and of the knock-on consequencesfor the livestock industry.

There will be opportunities for alternative forage crops and diversification of existing grassland swards. However, there is inadequateknowledge on how to optimise the sward composition and the forage grown, and their interactions with animal productivity.

Trees, wood production and forestry services

There is no systematic evidence or monitoring of past and current tree growth rate trends in UK forests, which is a major knowledge gap. In addition, the following priorities are identified:

- Better and more quantitative information on the drought sensitivity of the main UK tree species.
- More information on the suitability of new species and provenances for introduction.
- Assessment of tree pest and disease risks with increasing temperatures and their interactions with drought stress.
- Knowledge of wind risk and its management for broadleaved species.

Freshwater ecosystems and water services

We currently have very limited knowledge of the impacts of climate change on aquatic ecosystems, including the effects of changes in thermal and hydrological regime and water quality, particularly on key species (e.g. invertebrates, fish) and across trophic levels. More information is needed on the interaction between climate variability and change, particularly how this relates to extreme flow events and the vulnerability of sensitive locations. In addition, most of the existing research in this area has looked at the influence of either climate change or land-use change on water quality, but not how they may act together on catchment water quality and aquatic ecosystems.

More evidence is needed to better understand the efficacy of different adaptation options (e.g. riparian planting for water shading) in different catchments to develop a stronger evidence base for targeted interventions. In particular, more research and monitoring of water-related ecosystem services at catchment scale is urgently required because of the paucity of current data linked to climate sensitivity and climate change. Existing data areoften derived from a few localised examples. Research isneeded that covers the variety of different catchment types occurring across the UK and assesses multiple benefits, including flood risk reduction, maintaining flows and water availability during droughts, and the combined benefits for water quality.

A particularly important knowledge gap exists regarding the benefits of different types of natural flood management scheme at catchment scale (including co-benefits for other ecosystem services), and their potential role in reducing climate change risks, both for high flows and low flows.

Coastal ecosystems

There is some uncertainty regarding the current rate of sea-level rise and whether there has been a recent acceleration in the rate of rise as suggested by some of the satellite data and geomorphological and ecological evidence. This has important implications for the most suitable future projections on which to base both research and future planning.

As identified by CCRA1, there are still major limitations in the evidence base at national and UK scale for coastal habitat monitoring and evaluation of change, as most of the evidence is from specific areas that may not be indicative of the diversity of UK coastlines. The interaction of geomorphological and ecological processes and related changes in hydrodynamic forcing and salinity constraints at regional scale (e.g. for littoral cells) remains a major uncertainty, although there is good evidence for some locations (notably estuaries such as the Humber and Blackwater) that provide pilot studies for further knowledge exchange initiatives.

Knowledge of coastal systems and processes (e.g. from modelling studies) suggests that there are thresholds for sea-level rise and extreme events beyond which the resilience of the natural environment is considerably reduced. This increases the potential for crossing dangerous thresholds for flood and erosion risk (e.g. due to a barrier breach). Further work is required to provide better information on these thresholds in order that they can be incorporated into management plans.

Marine ecosystems

In 2014 the UK MCCIP published a list of priority research needsfor UK marine systems:

- Improved understanding of the combined effects of climate warming and ocean acidification on ecologically, and economically, important marine organisms and particulalry of the implications of changes in plankton communities for fisheries,seabirds and marine mammals.
- Better assessments of climate change impacts on marine mammals at the population level in order to discriminate between regional population responses and those occurring on a wider geographical scale. Long-term monitoring at a sufficient spatial scale is a component of this requirement.
- A better understanding of how changing observation techniques can affect the consistency of sea temperature records; for example, satellite measurements of sea surface temperature versus measurements physically taken in the ocean itself.

- A sustained, quality-controlled and global-scale observational database to provide information on baseline pH or pCO_2 concentrations, variability and trends in the carbonate chemistry system in coastal waters, shelf seas and the open ocean.
- Baseline data on the distribution and structure of deep-sea biological communities in UK waters and how they vary in time.
- Knowledge of large-scale benthic (bottom living) species distribution within UK waters in order to detect changes over large areas of the seabed and patterns of benthic response to climate change.
- Better monitoring and joined-up epidemiological reporting systems to aid understanding of human health impacts from pathogens occurring in, or released into, the marine environment.This would assist in linkingclinical cases back to climatic events.
- Regional model outputs that provide insight into dissolved oxygen levels around the UK and likely consequences for marine biota.

Marine fisheries and aquaculture

Priority research needs with regard to marine fisheries and aquaculture include:

- Improved predictions of climate impacts on primary productivity. Some regional climate models (linked to models of phytoplankton) anticipate an increase in system productivity around the UK and Ireland, whereas other models (and recent observations) suggest a decrease. Understanding the direction of change will be critical to improvements in future fish and fisheries models.
- Assessment of the social and economic implications of climate change on fishing fleets in the UK. Attention should be paid to identifying the winners and losers that might result from future climate change, including the particular fleets or ports that would be affected.
- Improved insight into the possible threat posed by ocean acidification, particularly with regard to the UK shellfish industry.
- Better wind and storm projections as well as improved understanding of how any changes in these variables might impact maritime safety, and/or access to fishery resources. Fishing operations can be greatly influenced by prevailing weather conditions. If storm severity or frequency changes, then fishing livelihoods could be heavily impacted.
- Research into indirect consequences of climate change on marine fisheries, for example, as mediated by changes in prey availability or exposure to predators. In addition, whether or not changes in marine habitats as a result of climate change and ocean acidification might have long-term consequences for fisheries resources.
- Improved monitoring and surveillance of non-native species, as well as jellyfish to be able to determine the indirect impact of climate change on aquaculture risk.
- Updated epidemiological risk assessments to include climate change impacts on aquaculture pathogens.
- Research into the relationship between fish farm escapes and changing frequency of storm events (return periods etc.).

Carbon storage and GHG emissions

Evidence for both the direct and indirect influence of climate on carbon stocks and GHG emissions is currently limited. Business-as-normal projections for changes in the LULUCF sector of the UK GHG Inventory have been produced but they have yet to be modified based upon climate change projections and different socio-economic scenario assumptions.

Similarly, the future land-use projections presented in the chapter, although indicative of the key role that land-use decisions can play in emissions pathways, assume a rather unrealistic approach to equilibrium conditions. Hence, there is a strong requirement to improve the evidence base by developing transient projections of changes in both vegetation and soil carbon stocks based upon the interaction of climate and socio-economic factors with land-use change and land management through time (e.g. including forestry rotations, livestock numbers, arable management, fertiliser application, changes in soil properties).

Pests, pathogens and INNS

There is a need for improved baseline distribution data, biological information and long-term, regional scale data on non-native species for terrestrial, freshwater and marine environments. For disease, the main requirements are: improved evidence of the factors that initiate EIDs, improved modelling of both scenarios for disease outbreaks and of INNS dispersal, and further assessment of alternative risk management strategies.

Landscape and sense of place

There is a significant knowledge gap regarding perceptions and appreciation of the multiple benefits from landscapes and seascapes Pilot initiatives that incorporate assessment of the direct and indirect implications of climate change on landscape character have been developed by Natural England, Scottish Natural Heritage, the Scottish Government, Natural Resources Wales and the Wildlife Trusts ("Living Landscapes" initiative). There is therefore considerable scope to link these approacheswith further use of Landscape Character Assessment in understanding and responding to climate change.

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Committee on Climate Change
7 Holbein Place
London
SW1W 8NR

www.theccc.org.uk

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