Chapter 5: Changing Ocean, Marine Ecosystems, and Dependent Communities

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Table of Contents

Executive Summary .......................................................................................................................... 3

5.1 Introduction ............................................................................................................................. 7

5.2 Changing Oceans and Biodiversity ....................................................................................... 8

5.2.1 Open Ocean ....................................................................................................................... 8

5.2.2 Changes in Physical and Biogeochemical Properties ....................................................... 8

Box 5.1: Time of Emergence and Exposure to Climate Hazards ............................................. 32

5.2.3 Impacts on Pelagic Ecosystems ....................................................................................... 35

5.2.4 Impacts on Deep Sea Floor Systems .............................................................................. 45

5.2.5 Risk Assessment of Open Ocean Ecosystems ................................................................. 54

5.3 Changing Coastal Seas and Biodiversity ............................................................................ 55

5.3.1 Introduction to Changing Coastal Seas ........................................................................... 55

5.3.2 Oceanographic Drivers .................................................................................................... 56

5.3.3 Coastal Ecosystems .......................................................................................................... 57

5.3.4 Assessments for Coastal Ecosystems .............................................................................. 73

5.4 Changing Marine Ecosystem Services and Human Wellbeing ........................................ 75

5.4.1 Changes in Key Ecosystem Services ............................................................................... 76

Box 5.2: Responses of Coupled Human-Natural Eastern Boundary Upwelling Systems to Climate Change ........................................................................................................................ 79
5.4.2 Climate Risk, Vulnerability and Exposure of Human Communities and their Wellbeing .......... 93

Box 5.3: Harmful Algal Blooms and Climate Change ..................................................................... 96

5.5 Risk-reduction Responses and their Governance .................................................................. 115

5.5.1 Ocean-based Mitigation .......................................................................................................... 115

5.5.2 Ocean-based Adaptation .......................................................................................................... 123

Box 5.4: Coral Reef Restoration as Ocean-based Adaptation .......................................................... 128

5.5.3 Governance Across All Scales ............................................................................................... 144

Box 5.5: Policy Responses to Ocean Acidification: Is there a Governance Gap? ......................... 148

5.6 Synthesis .................................................................................................................................. 151

5.7 Key Uncertainties and Gaps ....................................................................................................... 151

FAQ5.1: How is life in the sea affected by climate change? .............................................................. 152

References .................................................................................................................................. 154

Appendix 5.A: Supplementary Material .......................................................................................... 220

5.A.1 Projection of Time of Emergence of Habitat Suitability Index of Marine ................................. 223
Executive Summary

Ocean and cryosphere are important components of the earth system. The ocean is essential for all aspects of human well-being and livelihood. It provides key services like climate regulation, through the energy budget, carbon cycle and nutrient cycle. The ocean is the home of biodiversity ranging from microbes to marine mammals that form a wide variety of ecosystems in open pelagic and coastal ocean.

The oceans have warmed unabated since 2004, continuing the multi-decadal ocean warming trends documented in the IPCC Fifth Assessment Report (AR5). Ocean warming dominates the increase in heat energy stored in the climate system and pervades the ocean from the surface to bottom. The upper ocean (0–700 m) and 700–2000 m layers have both warmed from 2004 to 2016 (virtually certain) and the abyssal ocean (below 4000 m) continues to warm in the Southern Hemisphere (high confidence). The observed upper ocean temperature or heat changes are very likely to have a substantial contribution from anthropogenic forcings since 1970. There is increased agreement between coupled model simulations of anthropogenic climate change and observations of changes in ocean heat content (high confidence). The overall warming of the ocean will continue to increase this century even if radiative forcing stabilizes (e.g., RCP2.6, high confidence). Under the RCP2.6 and 8.5 scenarios, by 2100 the ocean is likely to take up about 3 and 6 times, respectively, the roughly 500 × 1021 J that the oceans have already taken up since the start of the 20th century. {5.2.2.2}

The ocean is stratifying; observed warming and high latitude freshening are both surface intensified trends making the surface ocean lighter at a faster rate than deeper in the ocean (high confidence). It is very likely that stratification in the upper few hundred meters of the ocean will increase significantly in the 21st century. This trend reduces surface exchange with the deep ocean, affecting heat and carbon uptake, re-oxygenation of ocean and nutrient cycles. {5.2.2.2}

Multiple datasets and models show that the oceanic sink for atmospheric CO₂ has continued to strengthen in the recent two decades in response to the growing burden of CO₂ in the atmosphere (virtually certain), while an important degree of temporal variability has emerged. A consistent picture emerges from independent ocean surface and interior datasets (covering the last few decades and the period since the industrial revolution), as well as ocean models that ocean carbon uptake has continued to increase commensurate with the atmospheric carbon dioxide growth rate (high confidence). {5.2.2.3} It is very likely that the ocean has taken up about ~25±5% of total anthropogenic emissions in the recent two decades. Evidence is growing that the ocean carbon sink is dynamic on decadal timescales, especially in the Southern Ocean, which affects the total global ocean carbon sink (medium confidence). {5.2.2.3} Despite our improved understanding of the magnitude of ocean carbon uptake and its variability, there is still substantial uncertainty regarding the underlying mechanisms. {5.2.2.3}

The ocean is continuing to acidify in response to ongoing ocean carbon uptake. It is virtually certain that open ocean pH is declining by ~0.02 pH units per decade where time-series observations exist. The anthropogenic pH signal has already emerged over the entire surface open ocean (high confidence) and emission scenarios are the most important control of surface open ocean pH relative to internal variability for most of the 21st century at both global and local scale (virtually certain). {5.2.2.3} Changes to the deep ocean are more complex as they are controlled by parallel changes to ocean circulation. {5.2.2.3 and Box CC-Southern Ocean}

1 FOOTNOTE: In this Report, the following terms have been used to indicate the assessed likelihood of an outcome or a result: Virtually certain 99–100% probability, Very likely 90–100%, Likely 66–100%, About as likely as not 33–66%, Unlikely 0–33%, Very unlikely 0–10%, Exceptionally unlikely 0–1%. Additional terms (Extremely likely: 95–100%, More likely than not >50–100%, and Extremely unlikely 0–5%) may also be used when appropriate. Assessed likelihood is type set in italics, e.g., very likely (see Section 1.9.3 and Figure 1.4 for more details).

2 FOOTNOTE: In this Report, the following summary terms are used to describe the available evidence: limited, medium, or robust; and for the degree of agreement: low, medium, or high. A level of confidence is expressed using five qualifiers: very low, low, medium, high, and very high, and type set in italics, e.g., medium confidence. For a given evidence and agreement statement, different confidence levels can be assigned, but increasing levels of evidence and degrees of agreement are correlated with increasing confidence (see Section 1.9.3 and Figure 1.4 for more details).
The ocean is observed to be losing oxygen and oxygen minimum zones have expanded. Overall, the oxygen content of the global ocean has declined by about 2%, largely in responses to changes in ocean circulation and respiratory demand (high confidence). \(5.2.2.4\) The largest reductions in oxygen have been observed in the Southern Ocean, South Atlantic and North Pacific (medium confidence), but there is low confidence for changes in the tropical ocean due to natural variability and limited agreement across studies. Over the next century oxygen declines of 3.5% by 2100 are predicted by CMIP5 models globally (medium confidence), with low confidence at regional scales, especially in the tropics. The largest changes in the deep sea will occur after 2100. \(5.2.2.4\). Where oxygen is already low, even very small declines in oxygen availability can lead to decreases in biodiversity, nutrient cycling, and ecosystem productivity. \(5.2.2.5, 5.2.3\) and Box 5.1

Ocean primary productivity is projected to show regional variability in response to climate driven changes in nutrient supply with implications for pelagic and seafloor ecosystems. There is high confidence that future changes to ocean primary productivity will be driven by region specific changes in magnitude and ratio of nutrient supply. \(5.2.2.5\) In general, CMIP5 models project a small decrease in global organic matter production (medium confidence) with increases in high latitude (low confidence) and decreases in low latitude (medium confidence) in response to changes ocean nutrient supply. There is currently low confidence in appraising past open ocean productivity trends, including those determined by satellites, due to newly identified region-specific drivers and the lack of corroborating in situ time series datasets. \(5.2.2.5\)

Emergence of novel ocean conditions for organisms from plankton to mammals are driving changes in eco-physiology, biogeography and ecology and biodiversity (high confidence). Observed and projected population declines in the lower-latitude range boundary (medium confidence), expansion in the poleward boundary (high confidence), earlier timing of biological events (high confidence), and overall shift in biomass and species composition (very high confidence) are consistent with expected responses to climate change. \(5.2.2, 5.2.3, 5.3.2, 5.3.3\) Changes in biogeography and community structure are projected to continue in the 21st century (high confidence), with potential global animal biomass projected to decrease by 4.8% ± 3.6% s.d. and 17.2% ± 11.1% s.d. under RCP2.6 and 8.5, respectively, by 2090–2099 relative to 1990–1999 (likely). \(5.2.3.1\) Scope for adaptation for many organisms to cope with novel environmental conditions is limited (medium confidence), particularly those higher up in the ocean foodweb and for high carbon emission scenarios. \(5.2.3, 5.3.3\)

All coastal ecosystems are observed to be under stress from the combination of climate change impacts in the ocean and from land sources as well as other human activities (high confidence). Habitat erosion and retreat from rising sea level and more severe storms are leading to losses of vegetation and benthic fauna e.g., in sandy beaches and saltmarshes (high confidence). \(5.3.3.4, 5.3.3.5\) Extreme events such as heat waves and storms are exacerbating the rate of ecosystem changes, such as those observed in kelp forest and seagrass meadow (high confidence). \(5.3.3.4, 5.3.3.9\) Herbivory intensified by warming will result in physical and physiological stress on coastal vegetation and reduce their productivity (medium confidence). \(5.3.3\) Compounding effects of warming, deoxygenation, acidification and changes in nutrient supplies exacerbates the decrease in species richness and spatial heterogeneity in coastal ecosystems (medium confidence). \(5.3.3\)

Almost every major coral reef system (shallow and deep) are vulnerable to climate change with clear regional differences in their sensitivities (high confidence). Ocean warming, acidification, rising sea level and more severe storms impede reef resilience on a global level and augment reef destruction (high confidence). \(5.3.3.6, SROCC Chapter 6\) Shallow coral reefs that are not degraded by other impacts such as overfishing and nutrient enrichment could constitute an important refuge to reefs degraded by climate change. However, loss of deep-water coral reef habitat is virtually certain under projected ocean acidification (aragonite under-saturation) through dissolution and intensified bio-erosion of the non-living matrix. \(5.3.3.6\)

Benthic communities in deep-sea habitats will experience structural and functional changes that affect the carbon cycle in this century under all emission scenarios (medium confidence). This is suggested by the strong positive relationship between annual Particulate Organic Carbon (POC) flux and abyssal sediment community oxygen consumption combined with projected changes in biomass \(5.2.4.2\). Much of the
abyssal seafloor is expected to experience declines in food supply \{5.2.4.4\} that will diminish benthic biomass, change community structure and rates of carbon burial \textit{(medium confidence)}. The majority (82\%) of the mapped seamounts are predicted to experience reduced POC flux under RCP8.5 in 2100, resulting in declines in benthic biomass \textit{(medium confidence)} \{5.2.4\}.

Seafood provision from fisheries and aquaculture are impacted by climate change \textit{(high confidence)}, reducing the revenue from fisheries sectors \textit{(medium confidence)}, and impacting the livelihood of the dependent communities \textit{(medium confidence)} and food security of vulnerable people \textit{(medium confidence)}). Fisheries catches and their composition are already affected by the effects of warming, deoxygenation and changes in primary production on growth, reproduction and survival of fish stock \textit{(high confidence)} \{5.4.1\}. Shellfish aquaculture is also sensitive to ocean acidification \textit{(high confidence)} \{5.4.2.2.1\}. In the 21st century, potential global fisheries catches are projected by multiple models to decrease with increasing dominance of warmer water species with increasing CO₂ emission \textit{(medium confidence)} \{5.4.2\}, although the changes in realized catch will depend strongly on fishing intensity \{5.4.1.3.1\}. The decrease in catch in the tropical ocean is projected to be among the largest regional declines \textit{(very likely)}.

Farmed species will be exposed to increased risk of disease and harmful algal blooms \{5.4.1.3.1\}. Consequently, people who depend on fisheries and related-sectors will experience substantial decline in their income, livelihood and availability of animal-sourced nutrients \textit{(medium confidence)} \{5.4.2.1.2\}.

Ocean ecosystems’ role in climate regulation, supporting culture and recreation and their intrinsic values that are important for human wellbeing are being threatened by climate change because of the loss and degradation of biodiversity and ecosystems functions \textit{(high confidence)}. This is evidenced from reduction in nutrient cycling in the deep sea ecosystems through warming- and hypoxia-induced reduction in biodiversity \textit{(medium confidence)} \{5.4.1.2.2\}, reduction in the quality and quantity of tourist attractions as coral reefs are degraded from warming and increased severity of storm events \textit{(high confidence)} \{5.4.1.3.2\}, reduction in carbon stock and sequestration in salt marshes through reduced habitat availability for fauna requiring open vegetation structure as a result of sea level rise \textit{(high confidence)} \{5.4.1.3.2\}, and potential loss of opportunities for using ocean ecosystems for education and the relationship with Indigenous knowledge and culture \textit{(medium confidence)} \{5.4.1.1.2, 5.4.1.2.2, 5.4.1.3.2, 5.4.2.1\}.

Climate change impacts on the Ocean is expected to substantially increase the risks on human health and conflicts within and between political entities in the 21st century \textit{(medium confidence)}.

Elevated risks of water-borne disease, food poisoning and pollutant contamination on human health are directly related to climate change through increase prevalence of pathogens and harmful algal bloom, and increased contaminant bioaccumulation under warming and high CO₂ conditions \textit{(high confidence)} \{Box 5.3, 5.4.2.1.1\}. The projected decline in fish supply and key fish micronutrients threaten food security of coastal communities that are strongly dependent on seafood, such as coastal Indigenous people and many tropical countries \{5.4.2.1.2\}. Redistribution of fish stocks under climate change increases the risk of dispute between countries or regions \textit{(high confidence)} while reduced food security and livelihood of socially marginalized human populations exacerbates inequality and social unrest \textit{(medium confidence)} \{5.4.2.1.3\}.

Blue carbon ecosystems, such as mangroves, salt marshes and seagrasses, can help provide a nature-based solution with multiple co-benefits. Some 151 countries around the world contain at least one of these coastal blue carbon ecosystems and 71 countries contain all three. Carbon storage, per unit area, in vegetated marine habitats can be much greater than for terrestrial habitats \textit{(high confidence)} \{5.5.1.1\}.

Successful implementation of measures to maintain and promote carbon storage in coastal ecosystems should significantly assist several countries in reaching the net zero emissions target of the Paris Agreement \textit{(high confidence)} \{5.5.1.1\}. Conservation of these habitats would also sustain the wide range of ecosystem services they provide and assist with climate adaptation through improving critical habitats for biodiversity, enhancing local fisheries production, and protecting coastal communities from sea level rise and extreme weather events \textit{(high confidence)} \{5.5.1, 5.5.2\}. The climate mitigation effectiveness of other natural carbon removal processes in coastal waters, such as seaweed ecosystems, \{5.5.1.1.3\} and proposed non-biological marine CO₂ removal methods are smaller or currently have low feasibility currently \{5.5.1.1.4\}.

Blue carbon ecosystems cannot replace the need for the very rapid phase-out of fossil fuels, and the control of all other emissions, nor for further research and development of environmentally-friendly negative emission technologies. The maximum global mitigation benefits of cost-effective coastal wetland
restoration is unlikely to be more than 2% of current total emissions from all sources. However, the
protection and enhancement of coastal blue carbon can be considered as a ‘no regrets’ option, in addition to,
rather than replacing, other mitigation measures. The potential for climate mitigation by increasing
biological productivity in the open ocean is limited, \{5.5.1.2.1\} since nearly all the extra carbon removed
would be returned to the atmosphere on decadal timescales. Other human interventions to enhance marine
carbon uptake, e.g., ocean alkalisation (enhanced weathering), would also have contested governance, with
risk of undesirable non-climatic consequences. \{5.5.1.2\}

Given strong carbon emission mitigation, management of other non-climatic human stressors on
ocean ecosystems and improving the resilience and resistance of built infrastructure offers cost-
effective opportunities for climate risk reduction for the ocean \(\text{high confidence}\). Managing existing
non-climatic human drivers such as overfishing, habitat degradation, pollution, demographic changes and
poverty can substantially reduce climate risk on ocean ecosystems and dependent human communities with
large co-benefits in supporting sustainable development \(\text{high confidence}\). \{5.5.2.2\} Built-infrastructure
adaptation are optimally by integration with natural infrastructure and ecosystem-based approaches \(\text{high
confidence}\). \{5.5.2\} The effectiveness of adaptation approaches becomes low under high greenhouse gas
emission scenario \(\text{high confidence}\). \{5.5.2.5\} Integrated adaptation planning, with non-climate change
related co-impacts, remains an under-achieved ambition, especially in developing countries \(\text{high
confidence}\). \{5.5.2.1, 5.5.2.4\}

The effectiveness of coastal ecosystem-based adaptation is supported by a growing body of literature
and increased international funding for this approach. Adaptation through ecological engineering such
as restoration of natural habitats can support biodiversity and enhances ecosystem services that could
provide support livelihood, economic benefits and help build resilience in vulnerable dependent
communities to climate change \(\text{high confidence}\). \{5.5.2.1\} Ecosystem-based approaches are more effective
if supported with other adaptation approaches like community-based adaptation, adaptation planning within
frameworks and supported by good governance and effective implementation \(\text{high confidence}\). \{5.5.2.1,
5.5.2.2\}

Existing international and national legislation and related processes are not sufficient to provide an
effective adaptation response to reduce risk of climate change in the marine system \(\text{high confidence}\).
Adjustment of legal tools and techniques of international conventions such as the United Nations Convention
on the Law of the Sea (UNCLOS) could facilitate more effective responses to challenges created by climate
change for the Ocean. \{5.5.4\} However, such adjustment is heavily dependent on further development,
modification and implementation of detailed regulations e.g., fisheries management or marine spatial
planning by relevant international, regional and national institutions. Climate risks can also be further
reduced by shortening the lead time before implementation of these adjustments relative to the time of
emergence of climate stressors and their impacts in the marine system \(\text{medium confidence}\). \{Box 5.1,
5.5.4.1\}
5.1 Introduction

The ocean is a key component of the Earth System. It stores the heat trapped in the atmosphere resulting from rising greenhouse gases, it stores carbon dioxide in its ocean depths and it masks and slows surface warming of the Earth. It is the home to the largest continuous ecosystem, providing habitats for rich marine biodiversity, and is an essential source of food and livelihood that contributes to the world’s food, human health and security. The ocean also supports other essential services to humans through, for example, climate regulation, tourism, renewable energy, aesthetic appeal and sustaining local and traditional knowledge. Governance in the ocean also has a different set of approaches, challenges and opportunities from other systems that requires special treatment particularly under a changing climate.

The IPCC Fifth Assessment (AR5) reports from Working Group I (WG1) have shown from the available research that there are ongoing changes to the physical and chemical state of the ocean. AR5 WG1 report (IPCC, 2013) concluded that (1) ‘ocean warming dominates the increased energy stored in the climate system with more than 90% the energy accumulated since 1971’; (2) ‘the ocean has absorbed about 30% of the emitted anthropogenic carbon causing ocean acidification’ and (3) it is ‘extremely likely that human influence has been the dominant cause of warming since mid 20th century’. AR5 WG1 Chapter 1 summarizes observed changes in the oceans (Rhein et al., 2013), Chapter 10 addresses the detection and attribution of human influence (Bindoff et al., 2013) and Chapter 12 provides projections of the physical system (Collins et al., 2013).

The Working Group II of IPCC AR5 shows that marine ecosystems are changing with implications for human dependent society through changes in ecosystem services, with Chapter 5 covering coastal ecosystems (Wong et al., 2014), Chapter 6 covering ocean processes (Pörtner et al., 2014), Chapter 30 (Hoegh-Guldberg et al., 2014) covering the regional differences between ocean basins, and Chapters 14–20 covering general aspects related to adaptation and impacts, risks, vulnerabilities, and opportunities of different societal sectors that are also applicable to the ocean. These assessments concluded that ocean warming, acidification and deoxygenation are affecting marine life from molecular processes to organisms and ecosystems, with major impacts on the use of marine systems by human societies. Specifically, IPCC AR5 WGII highlights that ‘marine species have shifted their ranges, seasonal activities, migration patterns, abundance, and demonstrated altering of species interactions in response to ongoing climate change’ (high confidence, Chapter 6 and 30). AR5 WGII also concludes that ‘future climate change will cause species richness to increase at mid and high latitudes (high confidence) and to decrease at tropical latitudes (medium confidence), resulting in global redistribution of catch potential for fishes and invertebrates, and implications for food security (medium confidence)’. Also, ‘ocean acidification poses risks to ecosystems, especially sensitive ones such as coral reefs (medium to high confidence)’.

This report updates these earlier assessments with new evidence on the changing ocean and marine environment. Unlike previous assessment reports, this ocean focused chapter undertakes an integrated assessment of the coupled human-natural marine system, with the current research and knowledge of how changes in physical, chemical and biological properties of the ecosystems interact with risks and vulnerabilities of the dependent communities under climate change. We also specifically assess methods and approaches that have been taken to reduce risk and the governance options that might mitigate the risks of anthropogenic climate change. Radiation management techniques (also known as sunlight reflection methods) are excluded here. Such approaches are direct geo-engineering that are addressed in the IPCC Special Report on Global Warming of 1.5°C (SR1.5) and are outside the scope of the current report. However, two mitigation measures are considered, first by maintaining carbon stores in the marine environment and secondly the potential to enhance carbon uptake are included (Section 5.5.1) and compared with the efficacy of natural marine systems.

The chapter design is structured around a set of guiding questions. What are the key changes in the physical and biogeochemical properties of the ocean and major habitats and their taxonomic groups (Section 5.2 and 5.3)? How have these changes impacted key ecosystems, risks to ecosystems services and human wellbeing (Section 5.4)? And finally, how effective are coastal blue carbon and open/deep ocean initiatives for mitigating carbon emissions and providing pathways for adaptation for marine dependent communities (Section 5.5)? These guiding questions mean that the report covers both regional and global scales and
across aspects of human social systems, including governance and institutions, and adaptation pathways for dependent communities.

While the geographic scope of this chapter is global, Chapter 3 covers the Polar (i.e., Arctic and Antarctic) ecosystems and circulation changes in more detail, Chapter 4 covers the effects of sea level and its implications, and Chapter 6 covers extremes and abrupt events. While some of the sub-systems are addressed in other chapters, we draw on their assessments when necessary. This chapter uses IPCC calibrated language around scientific uncertainty described in Section 1.8.3.

5.2 Changing Oceans and Biodiversity

5.2.1 Open Ocean

The open ocean extends from a water depth of 200 metres into the deeper regions and is the subject of this section. The world’s coastal and shelf seas (extending from the coasts to depths of 200 metres) are discussed in Section 5.3. The assessment of the open ocean includes the changing physical and chemical properties (Section 5.2.2) and their impacts on the pelagic ecosystems (Section 5.2.3). The physical and chemical changes include the observed and projected changes from the available climate models for two scenarios (Representative Concentration Pathway or RCP2.6 and RCP8.5) to 2100. The drivers of changes in the pelagic ecosystems are the main focus of Section 5.2.2. The section includes changing temperature and circulation, oxygen, ocean acidification, nutrients and net primary productivity. The pelagic ecosystems are further sub-divided into epipelagic (water column that is exposed sunlight, Section 5.2.3.1), the deep pelagic (water column that is not exposed to sunlight, Section 5.2.3.2) and the deep sea floor ecosystems (5.2.4). The open ocean has perhaps the clearest signals of climate change in the observations of the physical environment and these have been attributed to rising greenhouse gases. This relative simplicity contrasts with the complexity of the coastal and shelf seas.

5.2.2 Changes in Physical and Biogeochemical Properties

5.2.2.1 Introduction to Changing Open Ocean

The ocean is getting progressively warmer and fresher; there are also changes in fundamental chemistry, including ocean acidification and oxygen loss (Rhein et al., 2013). Each of these global scale trends is readily detectable in oceanic observations, well understood scientifically, and is consistently projected by climate models. Each of these has been directly attributed to anthropogenic forcing from changing concentrations of greenhouse gases and aerosols (Bindoff et al., 2013). These trends in the global average ocean temperature will continue for centuries after the anthropogenic forcing is stabilized (Collins et al., 2013).

The impacts on ocean ecosystems and human societies are primarily driven by regional trends and by the local manifestation of the global-scale changes. At these smaller scales, the temperature, salinity, nutrient and oxygen concentrations in the ocean are also expected to exhibit well-understood basin and local-scale changes. However, the ocean also has significant natural variability at local and basin scales with timescales from days to decades and longer (Rhein et al., 2013). This variability can mask the underlying observed and projected trends (see Box 5.1). The impact of multiple stressors on marine ecosystems, which can compound risks, is also less well understood. There is new evidence and understanding about multiple stressors which has accumulated since the last assessment report (e.g., Gunderson et al., 2016). The most severe impacts of a changing climate will typically be experienced when conditions are driven outside of the range of previous experience at rates that are faster than human or ecological systems can adapt (Pörtner et al., 2014; Box 5.1).

This section summarizes our emerging understanding of the primary changes to the ocean, along with an assessment of several key areas of scientific uncertainty about the changes in the ocean. Because many of these long-term trends have already been extensively discussed in previous assessments (IPCC, 2013), much of this summary of the physical changes will be brief except where there are significant new findings. The processes driving ocean mixing and its rich geography and temporal variability are complex. Thus, our
projections potentially could not account for all the changes in circulation or mixing triggered by anthropogenic climate changes.

5.2.2.2 Changing Temperature, Salinity, Circulation

5.2.2.2.1 Global scale and projected regional ocean changes over the 21st century

The ocean is warming as a direct result of anthropogenic changes to the radiative properties of the atmosphere and the heat budget of the Earth (very likely) (Bindoff et al., 2013). Over each of the past few decades, our ocean observing system has documented an increase in ocean temperature (Figure 5.1) that corresponds with an uptake of about 93% of the excess heat accumulated in the Earth system over this time (Bindoff et al., 2013; Rhein et al., 2013). The ocean heat uptake of 274 ZJ\(^{3}\) during 1970-2010 period is equivalent to an energy imbalance of 0.42 W m\(^{-2}\) (with respect to the Earth surface area 5.10 x 10\(^{14}\) m\(^{2}\)) (Rhein et al., 2013). Thermal expansion has contributed about half of the observed global mean sea level rise during this period (Church et al., 2013; Table, 13.1).

To understand the recent and future climate, we use coupled ocean-atmosphere models with the full time history of atmospheric forcing (greenhouse gases, aerosols, solar radiation and volcanic eruptions) for the historical period and projections of the concentrations or emissions of these forcing agents to 2100. For these projections the RCPs of atmospheric emissions scenarios are used as specified by the Climate Model Intercomparison Project, Phase 5 (CMIP5) (see Section 1.8.2.3, and also IPCC AR5). This chapter focuses on the low and high emissions scenarios RCP2.6 and RCP8.5, respectively. When these scenarios are used to drive climate models, it is possible to simulate the recent and future patterns of changes in the ocean temperature, salinity and circulation (and other oceanic properties such as ocean oxygen concentration and ocean acidification, Section 5.2.2.3 and 5.2.2.4). The direct comparison of the simulated historical changes with observations is undertaken to detect climate change, to attribute the causes of climate change to the forcings in the system, and to establish the credibility of the model. Attribution studies also reject competing hypotheses to explain the global ocean changes such as natural forcing such as solar variability and volcanic eruptions (refer to Chapter 1) Bindoff et al. (2013). Finally, the projections of ocean changes also inform the detection, attribution and projection of risks and impacts of ecosystems (Sections 5.2.3 and 5.3.4) and human wellbeing (Section 5.4) under climate change.

\(^{3}\) FOOTNOTE: ZJ is Zettajoule and is equal to 10\(^{21}\) joules.
Figure 5.1: Ocean heat content changes: (a) Time series from 1950 to 2100 of globally integrated ocean heat content changes in ZJ relative to the 2004 to 2008 average as inferred from observations by 4 independent efforts (Green lines; Palmer et al., 2007; Lyman and Johnson, 2014; Cheng and Chen, 2017; Ishii et al., 2017) processed as in Johnson et al. (2018) and as simulated by an ensemble of 17 CMIP5 coupled climate modes for the historical period (Magenta) and projected into the future with RCP2.6 (Blue) and RCP8.5 (Red) forcing; the inset sub-panel in (a) is an expanded version of this figure for the 1990-2018 time period; (b) and (c) Observed heat uptake between the 2004-2008 pentad and the 2012-2016 pentad converted to heat fluxes averaged over (b) the 0-700 m depth range and (c) the 700-2000 m depth range converted into W m$^{-2}$, as determined by the SODA 3.4.2 reanalysis product (Carton et al., 2018); (d) and (e) the 17 model CMIP5 ensemble mean simulated heat uptake for the same time periods and depth intervals as (B) and (c), with stipling indicating regions where the ensemble mean change is not significantly different from 0 at the 95% confidence level based on the models' temporal variability; (f) Projected CMIP5 ensemble mean RCP8.5 heat content uptake for the top 700 m from (2081 to 2100) minus (1986 to 2005), with stipling indicating regions where the projected changes are not significant at the 95% confidence level; (g) like (f) but for the 700 m to 2000 m depth range.
The globally integrated ocean heat content changes from reanalysis of ocean observations and as projected with an ensemble of CMIP5 climate models are shown in Figure 5.1a. These two estimates are consistent in the historical period, once the limitations of the historical ocean observing network and the internally generated variability with a single realization of the real world are taken into account (see Section 5.2.2.2). The ensemble of climate models project that under the RCP2.6, the ocean will likely take up an additional 1500 ZJ of heat (Figure 5.1), or about 3 times the roughly 500 ZJ that the oceans have already taken up since 1900 (Roemmich et al., 2012). With the RCP8.5 scenario, the oceans will very likely take up about twice as much more heat as RCP2.6 (Figure 5.1). Even for a low carbon emissions pathway of RCP2.6 there is only a 66% chance of avoiding 2°C of surface air temperature warming relative to preindustrial times and the ocean will continue to warm for several centuries to come (Collins et al., 2013). It is virtually certain that the ocean will continue to take up heat throughout the 21st century, at a rate that depends upon which radiative forcing scenario we collectively choose to follow.

The regional patterns of the ocean temperature and salinity changes (Figures 5.1 and 5.2) associated with climate change reflect both increased net downward heat fluxes due to the evolving atmospheric composition and related changes in the net fresh-water forcing from the atmosphere, land and cryosphere, as carried by the ocean circulation. These patterns also reflect changes in the ocean circulation itself, driven by changes in wind, heat, and fresh water forcing of the ocean. The ocean circulation plays a major role in redistributing heat laterally and in regulating the exchange between the surface and ocean interior. While ocean circulation plays a critical role in shaping large-scale distributions of temperature and salinity, most of the changes in ocean circulation are relatively modest, and the most obvious signatures of climate change in the ocean are in changing temperature and salinity distributions.

The ensemble average of the CMIP5 climate models projects widespread ocean warming over the coming century, concentrated in the upper ocean (Figures 5.1f and 5.2) (Kuhlbrodt and Gregory, 2012). The anthropogenic heat will penetrate into the ocean following well-established circulation pathways (Jones et al., 2016). The greatest vertically integrated heat uptake occurs where there is already the formation of interior waters, such as Antarctic Intermediate Water along the Antarctic Circumpolar Current (Frölicher et al., 2015) or North Atlantic Deep Water in the Nordic Seas (Figure 5.1f), but all watermasses that are subducted over decades are expected to experience significant warming (see Figure 5.2). The warming in the subtropical gyres penetrates deeper into the ocean than other gyres (roughly 15 to 45°N and 15 to 45°S in Figure 5.2), following the wind-driven bowing down of the density surfaces (the solid lines in Figure 5.2) in these gyres (Terada and Minobe, 2018). The greater warming at 700-2000 m in the Atlantic than the Pacific or Indian Oceans (Figure 5.1g) reflects the strong southward transport of recently formed North Atlantic Deep Water at these depths by the Atlantic Meridional Overturning Circulation (AMOC). Two areas that commonly exhibit substantially reduced near-surface warming over the course of the 21st century are the northern North Atlantic, where a slowing AMOC (see chapter 6) reduces the northward heat transport and brings the surface temperatures closer to what is found in other ocean basins at these latitudes (Collins et al., 2013), and the southern side of the Southern Ocean, where water upwells that has been submerged for so long that it has not yet experienced significant anthropogenic climate change (Armour et al., 2016). Most of these projected warming patterns are broadly consistent across the current and previous generations of climate models (Mitchell et al., 1995; Collins et al., 2014) as well as observations and understood theoretically. These evidences give high confidence that the projections describe the changes the real world will experience (high agreement, robust evidence).

FOOTNOTE: The 17 CMIP5 coupled models used here were selected based on the availability of ocean data from the historical period, RCP2.6 and RCP8.5 projections, and corresponding control runs to correct for model drift. The models used are ACCESSI.0, BCC-CSM1-1, CCM4, GFDL-CM3, CMCC-CESM, CNRM-CM5, CSIRO-BOM, CanESM2, GFDL-ESM2G, GFDL-ESM2M, FGOALS-S2.0, GISS-E2-H, HadGEM2-CC, INM-CM4, IPSL-CM5A-LR, MPI-ESM-LR, and MRI-CGCM3. Up to 3 ensemble members were included per model, and all changes are relative to a control run with an identical initial condition but with preindustrial forcing. A table with a description and citations for each of these models, along with more detailed discussion of the use of climate model output, can be found in Flato et al. (2013).
Figure 5.2: Side-view basin-averaged zonal-mean trends [change per 50 years] in water-mass properties by basin as observed (average of 2012-2016 minus average of 2004 to 2008) and CMIP5 projected with RCP8.5 forcing (average of 2081 to 2100 minus average of 1981 to 2000) trends in water-mass changes with RCP8.5 forcing: top-to-bottom (Atlantic, Pacific, Indian, Global), with observed changes above the corresponding projected changes; left-to-right.
The near surface salinity of the ocean is both observed and projected to evolve in ways that reflect the increased intensity of the Earth’s hydrologic cycle (Durack, 2015). Generally speaking, the ocean surface in areas of the ocean that currently have net evaporation are expected to become saltier, while areas with net precipitation are expected to get fresher (Rhein et al., 2013), as the current patterns of precipitation and evaporation are generally expected to be amplified (Held and Soden, 2006). At longer timescales of decades to centuries, the larger scale changes in the ocean circulation and basin-integrated fresh water imbalances emerge in the near-surface salinity changes, as shown in Figure 5.2, with an increasingly salty tropical and subtropical Atlantic and Mediterranean contrasting with a freshening Pacific and polar Arctic emerging as robust signals across the suite of climate models (Collins et al., 2013). The freshening of the high latitudes in the North Atlantic and Arctic basin is consistent with the widely expected weakening of the Atlantic Meridional Overturning Circulation (discussed in Chapter 6) and a decline in the volume of sea ice (discussed in Chapter 3).

Deeper in the ocean, salinity changes reflect changes in the rates of formation of water-masses or their newly formed properties. Due to their various origins, ocean water-masses tend to be relatively fresh (like Antarctic Bottom Water, Antarctic Intermediate Water or North Atlantic Intermediate water) or salty (like Mediterranean Water, Red Sea Overflow Water, or North Atlantic Deep Water) (Figure 5.2). The dominant ocean heat uptake by the southern hemisphere in the early 21st century is expected to become more balanced between the hemispheres as the asymmetric cooling by aerosols is reduced (Shi et al., 2018). Over time, as these water-masses are transported away from their sources in marginal seas or at the ocean’s surface, these distinctive properties are diluted by mixing with ambient interior ocean waters. Changes in the interior ocean salinities can reflect either changes in the rates of formation of these water-masses or changes in the properties of their source (Purich et al., 2018). Thus, projected freshening of the Southern Ocean surface leads to a freshening of the Antarctic Intermediate Water that is formed and subducted here, flowing northward from the Southern Ocean as a relatively fresh water-mass at depths of 500-1500 m (Figure 5.2). Increased surface salinity in the Atlantic subtropical gyres are pumped into the interior by the winds, leading to an increased salinity of the interior subtropical gyres, along with contributions from increasingly salty Mediterranean water (Jordà et al., 2017). Conversely, fresh-water capping of the northwestern North Atlantic inhibits deep convection in the Labrador Sea and the consequent production of Labrador Sea Water in some models (Collins et al., 2014), and contributes to the increased salinity of the North Atlantic between 1000 m and 2000 m depths by the reduction of the signal from a relatively fresh water-mass (Figure 5.2).

The ocean circulation at the end of the 21st century is expected to be qualitatively similar to that today. The projections suggest that some major ocean current transports will exhibit a modest increase (such as the Kuroshio Extension) (Terada and Minobe, 2018) or a small decrease (such as for the Indonesian Througflow) (Sen Gupta et al., 2016); many current transports are expected to exhibit smaller than 20% changes by 2100 with RCP8.5 forcing. The heat transported by the AMOC, in particular, contributes to the relatively clement climate of northern Europe and the North Atlantic Basin as a whole, although the wind-driven ocean gyres also contribute to the meridional ocean heat transport (see the review by Buckley and Marshall (2015)). As a result, there is a concern that significant changes in ocean circulation could lead to localized climate changes that are much larger than the global mean. Projected and observed changes in the AMOC and the rates of formation of deep water-masses in the North Atlantic are discussed in Chapter 6, along with the possibility of abrupt or enduring changes as a result of forcing from Greenland. A significant reduction in the Atlantic Meridional Overturning circulation would, in turn, modestly weaken the Gulf Stream transport, which also has a substantial wind driven component (Frajka-Williams et al., 2016). The Antarctic Circumpolar Current is projected to be subject to strengthening westerly winds and substantially reduced rates of bottom water formation, as discussed in the Cross-Chapter Box 5 in Chapter 3. Climate-change induced changes of the circulation in other mid-latitude basins may be difficult to detect or reliably project because of significant natural variability at inter-annual (e.g., El Niño) to decadal (e.g., the Pacific Decadal Oscillation) timescales. Most aspects of the large-scale wind-driven ocean circulation are very likely to be qualitatively similar to the circulation today, with only modest changes in transports and current locations.
Mesoscale eddies (geostrophic rotating vortices with spatial scales of 10-100 km that penetrate deeply into the water column, and are often described as the ocean’s weather, analogous to the midlatitude atmosphere’s high and low pressure systems) play an important role in regulating the changes to the larger scale ocean circulation, especially in the Antarctic Circumpolar current, as is discussed more fully in the Box CC-Southern Ocean. In addition, sub-mesoscale eddies (surface intensified, rotationally influenced motions with smaller horizontal scales of 100s of m to about 10 km and intrinsic timescales of a few days that especially arise in association with fronts in the ocean’s surface properties) are known to be particularly important in the dynamics of the near-surface ocean boundary layer (see the review by Mahadevan (2016)). Sub-mesoscale instabilities are associated with restratifying overturning circulations that can limit the thickness of the well-mixed ocean surface boundary layer near fronts (Bachman et al., 2017). Moreover, submesoscale motions generate strong vertical velocities that drive fluxes of nutrients from the interior ocean into the euphotic zone or create pockets of reduced mixing with increased phytoplankton residency time within the euphotic zone (Lévy et al., 2012), so that the sub-mesoscale structures are often readily visible in images of ocean colour. Intense mesoscale eddies are known to create favourable conditions for submesoscale instabilities both observationally (Bachman et al., 2017) and in numerical studies (Brannigan et al., 2017). Intensifying Southern Ocean eddy fields will likely have a significant impact on biological productivity, ecosystem structure, and carbon uptake, both directly and via submesoscale processes.

The exact mechanisms driving the abyssal ocean variability (below 4000 m) are still poorly understood owing to limited spatial and temporal coverage of oceanic datasets and significant differences between various climate models, but climate models consistently predict a climate-change induced long-term abyssal warming trend originating in the Southern Ocean due to a reduction in the formation rates of cold Antarctic Bottom Water (Heuzé et al., 2015). Although the abyssal modes of natural variability are not as pronounced as closer to the surface, deep ocean heat content can vary on relatively short time scales through the communication of topographic and planetary waves driven by changes in the rate of deep water formation at high latitudes (Kawase, 1987; Masuda et al., 2010; Spence et al., 2017). Both North Atlantic Deep water (NADW) and Antarctic Bottom Water (AABW) have shown variability in production rates over the past half century. The observed southern intensified global deep warming may be driven by a slow down in AABW formation rates owing to freshening of shelf waters, changes in local winds driving cross shelf mixing, or larger scale dynamics controlling the spin up or down of Southern Ocean gyres influencing the density of outflowing waters over deep sills. It may also reflect changes in deep Southern Ocean convection. The Weddell Polynya is a large opening in the wintertime ice of the Weddell Sea that is kept ice-free despite intense cooling of the ocean by convective mixing bringing up warm and salty water from the deep ocean. (See Box 3.2 for a more extensive discussion of polynyas and the Weddell Polynya in particular.) The Weddell Polynya was present in the first three years of infrared satellite observations of wintertime sea-ice concentrations in the early 1970s, but it has been closed since 1974, only to reopen for the first time in decades in 2016. The prominent Weddell Polynya in the early 1970s greatly increased the volume of the coldest waters in the deep Weddell Sea. Weddell Polynyas are documented to drive abyssal cold and salty signals and can spread /thermal signals as waves further and faster than could be explained by slow advective signals (Martin et al., 2015; Zanowski and Hallberg, 2017); these waves do not directly heat individual water parcels, but instead warm the ocean where they cause the coldest deep layers to spread laterally and thin. However, recovery from the large Weddell polynya of the early 1970s can only explain about 20% of the observed abyssal warming trend (Zanowski et al., 2015).

The ocean’s properties are changing most rapidly in the near surface waters that are more immediately exposed to atmospheric forcing. As a result of the surface-intensified warming, the upper few hundred meters of the ocean are becoming more stably stratified (Helm et al., 2011; Talley et al., 2016b). The combination of surface intensified warming and near-surface freshening at high latitudes leading to a projection of more intense near-surface stratification (the downward-increasing vertical gradient of density after effects of compressibility have been corrected for) across all ocean basins (Figure 5.2) is a robust result with a high agreement across successive generations of coupled climate models (Capotondi et al., 2012; Bopp et al., 2013). By contrast, the bottom intensified warming in the abyss (see Figure 5.3) which is consistent with a slowing in the rate of bottom water formation, is also associated with a reduction in the abyssal stratification of the ocean (Johnson et al., 2014; Desbruyères et al., 2016b). Both of these changes have consequences for the evolving turbulence and ocean water-mass structure. Based on observational evidence, theoretical understanding and robust climate model projections, it is very likely that stratification in
the upper few hundred meters of the ocean below the mixed layer will increase significantly in the 21st century over most ocean basins as a result of climate change, and abyssal stratification will likely decrease.

The dynamical consequences of increased stratification are understood with very high confidence (see, for instance, Gill (1982) or Vallis (2017)). The projected amount of increase (or decrease) in stratification will vary by region, with the time-horizon in question, between forcing scenarios, and to some extent across models. An increase in stratification will increase the lateral propagation of internal Rossby waves (which set up the basin-scale ocean density structure) proportionately, and it increases the buoyancy frequency and the lateral propagation speed of internal gravity waves and boundary waves by about half the percentage change of the stratification itself. Increasing stratification increases both the length of the internal deformation radius (a typical length scale in baroclinic eddy dynamics) and the horizontal scales of internal tides (see Section 5.2.1.2.4) proportionately with the changes in the internal gravity wave speeds. For the same forcing, increasing in stratification reduces the geostrophically balanced slope of density surfaces and to a lesser degree the vertical excursions of internal gravity waves. For the same turbulent kinetic energy dissipation, locally increased stratification reduces the turbulent vertical diffusivity of heat, salinity, oxygen and nutrients (see Section 5.2.1.2.5). Increasing interior ocean stratification acts to restrict the depth of the ocean’s surface mixed layer. The flattening of density surfaces by increased stratification inhibits advective exchange between the surface and interior ocean (Wang et al., 2015), with consequences for the uptake of anthropogenic carbon (Section 5.2.1.3), the evolving oxygen distribution (Section 5.2.1.4) and the supply of nutrients to support primary production (Section 5.2.1.5).

### Observing and attributing anthropogenic climate changes in the ocean

Identifying the patterns of anthropogenic climate changes in oceanic observations is complicated by the presence of basin-scale natural variability with a range of timescales, from tidal to multi-decadal, and due to the difficulties associated with maintaining high-precision observing systems spanning the ocean basins. Historically, scientific research expeditions starting in the 19th century have provided occasional sections measuring deep ocean properties (Roemmich et al., 2012). Greater spatial and temporal coverage of temperatures down to about 700 m from expendable bathythermographs (XBTs) along commercial shipping tracks started in the 1970s (Abraham et al., 2013). Since the early 2000s, thousands of autonomous profiling floats (Argo floats) have provided high-quality temperature and salinity profiles of the upper 2000 m of ice-free regions of the ocean (Abraham et al., 2013; Riser et al., 2016). Further advances in autonomous floats have been developed that now allow these floats to operate in seasonally ice-covered oceans (Wong and Riser, 2011; Wong and Riser, 2013), and more recently to profile the entire depth of the water column down to 4000 or 6000 m (Johnson et al., 2015; Zilberman, 2017) and to include biogeochemical properties (Johnson et al., 2017). Autonomous floats have revolutionized our sampling and accuracy of the global ocean temperature and salinity records and increased certainty and confidence in global estimates of the earth heat (temperature) budget, particularly since 2004 (Von Schuckmann et al., 2014; Roemmich et al., 2015; Riser et al., 2016), as demonstrated by the convergence of various observational estimates of the
changes in the heat budget of the upper 200 m in Figure 5.1a. New findings using data collected from such observing platforms mark significant progress since AR5.

Observationally based estimates of global ocean heat content changes prior to the Argo era are complicated by systematic instrumental errors and limited data coverage. Since AR5, there have been improvements in our ability to understand and correct for XBT instrumental errors. New estimates also attempt to avoid a tendency to minimize temperature changes arising from traditional data-void filling strategies that effectively revert to the mean in the absence of observations. Supplementing *in-situ* observations with satellite or using model-based covariances of oceanic variables in time and space, have been found to provide more accurate approaches to enhance the information that can be obtained from sparse internal-ocean data. Five estimates of ocean heat uptake in the upper 2000 m in 1971-2010 cited by AR5 ranged from 3.2 ZJ yr\(^{-1}\) to 5.3 ZJ yr\(^{-1}\) during the 1971-2010 period (Roemmich et al., 2013) (Figure 5.1). However, the three more recent independent estimates that do a better job of accounting for instrumental biases and the sparsity of interior ocean observations, suggest larger and more consistent heat uptake rates of 5.8±1.0 ZJ/yr (Ishii et al., 2017), 6.3±0.8 ZJ yr\(^{-1}\) and 6.3±1.8 ZJ yr\(^{-1}\) during the same 1971-2010 period. These same re-analyses exhibit more rapid ocean heat uptake in recent decades, with estimates of between 8.9 ZJ yr\(^{-1}\) and 9.5 ZJ yr\(^{-1}\) ocean heat uptake in the upper 2000 m from 1991 to 2017.

These updated estimates of ocean heat uptake are consistent with CMIP5 ensemble of climate models (see the inset panel in Figure 5.1a). While the collection of the worlds’ climate models have been criticized for having an ensemble mean that does not exhibit the observed ‘hiatus’ or ‘slowdown’ of global mean surface temperature increase in the early 21st century (Meehl et al., 2011; Trenberth et al., 2015), it is increasingly clear that this is at least in part due to the redistribution of heat within the climate system from the surface into the interior ocean and between ocean basins. Individual realizations of climate models do show decades with slow increases in mean surface temperature comparable to what was observed, even though these cases exhibit continued interior ocean heat uptake, and every ensemble member exhibits surface warming closer to the ensemble-mean over multidecadal timescales (Meehl et al., 2011; England et al., 2015; Knutson et al., 2016). Recent Ocean Heat Content (OHC) estimates show warming quite similar to the average of CMIP5 models, both during the 1971-2010 period highlighted in AR5, and over the 2004-present Argo era (Figure 5.1a). OHC changes in CMIP5 models closely match observational estimates over the past 50 years; the ensemble mean of the models gave an average ocean warming trend of 0.40±0.09 W m\(^{-2}\) 6.4±1.4 ZJ yr\(^{-1}\) for the upper 2000m from 1971 to 2010 compared to recent observations ranging from 5.8 ZJ yr\(^{-1}\) to 6.3 ZJ yr\(^{-1}\) (Cheng and Chen, 2017; Ishii et al., 2017). This overall consistency between observationally-based estimates of ocean heat uptake and climate model simulations of the historical period gives greater confidence in the accuracy of the climate model projections for changes in the coming century.

Detailed regional patterns of trends in temperature and heat content at depths of 0 m–2000 m during the early 21st century are consistent in various analysis efforts, owing to the improved observing network (Roemmich et al., 2015; Desbruyères et al., 2016a) (Figure 5.1b and 5.1c). All ocean basins have experienced significant warming over the past two decades (Figure 5.1b and 5.1c), with the greatest warming in the Southern Ocean (Roemmich et al., 2015; Trenberth et al., 2016), the tropical / subtropical Pacific Ocean (Roemmich et al., 2015), and the tropical / subtropical Atlantic Ocean (Cheng and Chen, 2017).

Large scale patterns of natural variability at interannual to decadal time scales can mask the long-term warming trend in the upper 700 m (Figure 5.1b), particularly in the tropical Pacific and Indian Ocean (England et al., 2014; Liu et al., 2016). The most significant upper 700 m warming occurred in a large extratropical band of the Southern Hemisphere between 30° S and 60° S mostly centred around 40° S, in the tropical and northern Indian Ocean, and in the Northern Pacific. Warming of the extra-tropical Southern Ocean is mainly concentrated in the southern Indian and Pacific subtropical gyres. Marginal seas, such as the Mediterranean and Red seas have also exhibited notable warming. Conversely, over the past decade there have also been regions of cooling in the upper 700 m, notably in the North Atlantic around 40° N–60° N. In the North Atlantic, extremely cold and relatively fresh surface and subsurface conditions have persisted since the year 2014 (Josey et al., 2015) linked both to circulation changes (Robson et al., 2014; Smeed et al., 2014; Yeager et al., 2015) and anomalous atmospheric conditions (McCarthy et al., 2015; Josey et al., 2018), which together have caused a cooling pattern in regional ocean heat content trends (Figure 5.1b). This robust cooling of the North Atlantic is consistent with a the persistent reduced intensity and heat transport of the North Atlantic Current (Smeed et al., 2018) that very likely contributes to record low densities in the deep
Labrador Sea due to deep ocean warming since 1995 and the role played by long-term freshening (Robson et al., 2014), and may have contributed to a subsequent intensification of deep convection in the Labrador Sea since 2012 (Yashayaev and Loder, 2017). Cooling trends over the past decade in the upper 700 m also found in the south-western Indian Ocean (near Madagascar) and within most of the northern subtropics (10° N and 30° N) and eastern basins of the Pacific Ocean. Observations show that most of the global ocean heat uptake over the past decade has been stored in the southern hemisphere subtropical gyres. These southern subtropical gyres are driven, in part, by the surface winds, which have intensified in recent decades, facilitating the penetration of heat to deeper depths (Gao et al., 2018). The ensemble of freely running CMIP5 climate model simulations also shows that internal variability can be expected to mask the long-term warming trend at regional scales in the upper 700 m over the past decade. As indicated by the stippling in Figure 5.1b, the CMIP5 multi-model ensemble mean trends over the period of 2006-2016 are regionally significant at the 95% confidence level compared with natural variability in only a limited part of the ocean basins. Averaged over the whole 21st century, by contrast, the ensemble of CMIP5 models show statistically regional significant upper 700 m heat content trends almost everywhere (Figure 5.1f).

At depths of 700–2000 m, observations in all of the ocean basins show broadly warming trends in the well-observed Argo era (2006-present), with particularly significant warming patterns in the Southern Hemisphere extratropics around 40° S and the subpolar North Atlantic (Figure 5.1c). These observed changes support the notion that deep ocean heat content has been continuously increasing. As a result, regional climate change signatures emerge from confounding natural variability sooner in the 700 m-2000 m depth range than in upper 700 m of the ocean, where interannual modes of variability have a larger influence on the circulation (for a more complete discussion see Johnson et al. (2018)).

Despite the patches of cooling water in the upper 700m described above, every one of the world’s ocean basins volume averaged over depths of 0 m to 2000 m has experienced significant warming over the last decade (Figure 5.1 b and c, Figure 5.2, and also Desbruyères et al. (2016a)). The greatest warming has been observed in the Southern Ocean, the tropical and southern subtropical Pacific Ocean, and the tropical/subtropical Atlantic Ocean (Figure 5.2). Indeed, the Southern Hemisphere extratropical ocean accounted for 67%-98% of the total ocean heat increase for the period of 2006 to 2013 (Roemmich et al., 2015). The warming of the ocean in the instrumental record since 1970 has continued unabated (Figure 5.1a). For the recent period (2004 to 2014) analyses are all in agreement and show that the oceans have continued to take up heat with a rate of 0.4–0.6 W m\(^{-2}\) averaged over the surface of the Earth (very likely), and thus are in the same range as reported by the IPCC AR5 WGI (Rhein et al., 2013).

There are many papers that describe the observed change in ocean temperature and ocean salinity, and it is clear that the ocean state has changed (Abraham et al., 2013; Ishii et al., 2017). The main difficulty of the ocean temperature (and salinity) measurements in the global oceans is the presence of internal variability. Internal variability in short records and on regional scales can swamp the underlying climate change signal. As can be seen in Figure 5.1 (Panels d and c) the historical periods 2003 to 2016 are very smooth and have a relative different pattern of change compared with the corresponding observed changes (Figure 5.1 Panels b and c). Detection and Attribution studies take the internal variability into account and separate the underlying climate signals with the same spatio-temporal sampling as the observations, and apply a range of statistical tests to determine the coherence of the observations with the co-sampled observations (Bindoff et al., 2013; AR5 WG1 Box 10.1).

Since AR5, the use of different and updated oceanographic data sets and increase in the number of ensembles of the CMIP5 simulations has improved the overall detection and attribution of human influence. The newer CMIP5 simulations have better and more complete climate forcing (over earlier simulations) and there are a greater number of ensembles. Together these measures increase the coherence of the simulations and reduce noise. For example, an isotherm approach used to reduce the noise from the displacement of isotherms in the upper water column allowing detection in each of the mid-latitude ocean basins was achieved on 60 year time series (Weller et al., 2016). Using all the available ocean temperature and salinity profiles the Southern Ocean showed that the warming and freshening patterns were consistent primarily with increased human induced greenhouse gases and secondarily from ozone depletion in the stratosphere, but inconsistent with internal variability (Swart et al., 2018).
Using energy conservation for the earth, simplified climate models also show the earth’s energy balance the observed trends are extremely unlikely (<5%) to be caused by internal variability (Huber and Knutti, 2011). Together the evidence from the IPCC AR5, with the new evidence on regional scales across the global oceans, and though the use of energy conservation, we conclude that the observed upper ocean temperature or heat changes changes are very likely to have a substantial contribution from anthropogenic forcings.

The global ocean below 2000 m has warmed significantly between the 1980s and 2010s (Figure 5.3), contributing to ocean heat uptake and through thermal expansion to sea level rise (Purkey and Johnson, 2010; Desbruyères et al., 2016b). The observed deep warming rate varies regionally and by depth reflecting differences in the waters influencing particular regions. The deep and abyssal North Atlantic, fed by North Atlantic Deep Water (NADW), has reversed from warming to cooling over the past decade, possibly associated with the North Atlantic Oscillation (e.g., Yashayaev, 2007; Desbruyères et al., 2014) or longer-term weakening in North Atlantic overturning circulation (Caesar et al., 2018; Thornalley et al., 2018). The strongest warming is observed in regions of the deep ocean Antarctic bottom water (AABW) (Purkey Sarah et al., 2014). Regions of the ocean fed by AABW from the Weddell Sea have exhibited a possible slowdown in local AABW warming rates (Johnson et al., 2014), while the Pacific, fed by AABW from the shelves along the Ross and Adelie Coast, has continued to warm at an accelerating rate between 1990 and 2018 (Desbruyères et al., 2016b).

To date, assessment of deep ocean (below 2000 m) heat content has mostly been from ship-based data collected along decadal repeats of oceanographic transects (Figure 5.2b) (Talley et al., 2016b). While relatively sparse in space and time compared to the upper ocean, these transects were positioned to optimize sampling of most deep ocean basins and provide the highest quality of salinity, temperature and pressure data. Argo floats capable of sampling to 6000 m have just started to populate select deep ocean basins. At present this Deep Argo data is still in its infancy and has not yet been incorporated into the deep ocean warming estimates, but are near-certain to significantly advance our understanding and ability to monitor the deep ocean warming on smaller spatial and temporal scales in years to come, once decadal information is available from the full global array (Johnson et al., 2015). The development of the deep Argo program, complemented by indirect estimates from space (Llovel et al., 2014; Von Schuckmann et al., 2014), will strongly reduce the current large uncertainties of deep ocean heat content change estimates in the future.

Although climate models are able to capture the dynamics of the climate system, all numerical models have approximations and biases. The most commonly used type of ocean component in climate models is known to exhibit numerically induced diapycnal mixing that can be a significant fraction of the physical mixing (Ilıcak et al., 2012; Megann, 2018). Because so many ocean models exhibit the same sign of bias, there is a systematic warming of the lower-main thermocline is not cancelled out when taking the average over the ensemble of all the models in CMIP5. These biases are widely known within the ocean modelling community, and various groups are working to reduce these biases in future climate models with better ocean model numerics and parameterizations. To correct for any linear consequences of model biases, climate model projections are always taken as the difference from a control run without the anomalous forcing. However, some aspects of the ocean response to climate change are nonlinear, and model biases can introduce uncertainties into climate projections. In the case of heat uptake, this is of the order of 10% uncertainty, while for the rate of steric sea level rise (which depends on the nonlinear equation of state of seawater) the uncertainty in CMIP5 models is of the order of 20% (Ilıcak et al., 2012).

5.2.2.2.3 Tides in a changing climate

Both human structures and ecological systems in the coastal zone are directly impacted by tidal amplitudes, which contribute to high-water levels. The tides are the response of a forced-damped-resonance system (Arbic et al., 2009). The M2 tide is the dominant tidal constituent in most places, with a period of ½ lunar day, or 12 hours, 25 minutes; the M2 tides are created by the differential motion of the solid Earth and oceans in response to the gravitational attraction of the moon (Newton, 1687; Laplace, 1799). The astronomical forcing evolves only slowly, however the tidal damping and basin resonance at tidal frequencies can change in response to climate change. Tides are well documented to have changed dramatically over geological timescales, with much larger amplitude global-scale tides during the last glacial maximum (e.g., Arbic et al., 2004; Egbert et al., 2004; Arbic et al., 2008; Griffiths and Peltier, 2009) when shelf areas were greatly reduced by the lower sea level, and tidally driven mixing was also more intense, amplifying the overturning circulation (Schmittner et al., 2015). The tides have also been changing on more recent time scales. For
example, Colosi and Munk (2006) found an increase in the amplitude of the principal lunar semidiurnal tide $M_2$ in Honolulu of about 1 cm over the past 100 years, which they attributed primarily to changes in oceanic stratification that bring about changes in the internal $M_2$ tide phase relative to the external $M_2$ tide, increasing constructive interference. Several recent studies have analyzed historical coastal tide gauge data and found amplitude trends of order 1-4% per century (Flick et al., 2003; Ray, 2006; Jay, 2009; Ray, 2009; Woodworth, 2010; Müller et al., 2011). In some locations, the changes in the tides have been of comparable importance to changes in mean sea level for explaining changes in high water levels (Jay, 2009). For many individual tide gauges, the trends in tidal amplitude are strongly positively or negatively correlated with local time-mean sea level trends (Devlin et al., 2017). Müller et al. (2011) model the response of tides to changes in water column thickness arising from sea level rise and glacial isostatic adjustment; the water column thickness changes explain some of the tidal changes, but by no means all of it. Another source of secular tidal changes, changes in oceanic stratification, modifies the rate of energy conversion from the barotropic tides to the internal tides (Jayne and St. Laurent, 2001) the vertical profile of turbulent viscosity on shelves (Müller, 2012), and the propagation speed of the internal tides (Zhao, 2016). Both sea level and stratification are expected to exhibit robust secular positive trends in the coming century due to climate change, at rates that are significantly larger than historical trends. As a result, it is very likely that majority of coastal regions will experience statistically significant changes in tidal amplitudes over the course of the 21st century.

Because coastal tides are near resonance in many locations, small changes in sea level and bay shape can change the local tides significantly. For example, the insertion of tidal power plants can have a significant impact on the local tides (Ward et al., 2012). Various observational and modeling studies demonstrate that sea level rise has spatial heterogeneous impacts on the tides, with some locations experiencing decreased tidal amplitudes and others experiencing increased tidal amplitudes (Pickering et al., 2012; Devlin et al., 2017). Such tidal changes have implications for designing flood defenses, for tidal renewable energy, for tidal flushing timescales of estuaries and embayments, and for navigational dredging requirements (Pickering et al., 2012). Pelling et al. (2013) demonstrate that the rapid coastline changes in China’s Bohai Sea have altered the tides in that region and throughout the Yellow Sea (Hwang et al., 2014). Pelling and Green (2014) examine the impact of flood defenses as well as sea level rise on tides on the European Shelf. Local changes to tides are very likely to result in measureable changes in many places from climate drivers (listed above) as well as from direct human coastal adaptation measures.

5.2.2.2.4 Ocean turbulence in a changing climate

Ocean turbulent mixing is a key process regulating the ocean circulation and climate. Turbulent mixing is important for the uptake and redistribution of heat, carbon, nutrients, oxygen and other tracers (properties that are carried along with the flow of water) in the ocean. Due to the importance of turbulent mixing on the lateral and vertical distribution of ocean properties, it is anticipated that ocean mixing changes can have significant impacts on marine ecosystems and on the rates of heat and carbon uptake by the ocean (Schmittner et al., 2009; MacKinnon et al., 2017). It has become increasingly clear that turbulent mixing in the ocean is not constant in space or time. Global estimates of both the turbulent kinetic energy dissipation rate and the diapycnal diffusivity, two measures of ocean turbulence, vary over several orders of magnitude throughout the ocean (Figure 5.4) (Polzin et al., 1997; Waterman et al., 2012; Whalen et al., 2012; Alford et al., 2013; Hummels et al., 2013; Sheen et al., 2013; Waterhouse et al., 2014; Kunze, 2017). Turbulent mixing both dissipates energy and diffuses tracers across isopycnals (Fischer et al., 2013). For a given energy dissipation rate, the turbulent diffusivities of heat, salinity, nutrients and other tracers tend to be smaller with stronger stratification. This dependency on stratification helps explain why the observationally inferred diffusivity in the heavily stratified main thermocline (250 m-1000 m depth) is of similar magnitude to those deeper in the water column, while the turbulent energy density and dissipation rate are much stronger at the shallower depths. Consistent global patterns in the variability of turbulent mixing are found using a range of techniques, including microstructure measurements, tracer release experiments, and indirect inference techniques, reflecting global patterns of the underlying physical processes that produce turbulence (Ledwell et al., 2010; Waterhouse et al., 2014; Kunze, 2017). Turbulence also fluctuates in time, is modulated by tidal cycles (Klymak et al., 2008) and seasonal changes (Sloyan et al., 2010; Whalen et al., 2012). In the mixed layer and directly below, turbulence changes according to local conditions, such as the winds, heating rates and local stratification (Sloyan et al., 2010; Moum et al., 2013; D'Asaro, 2014; Tanaka et al., 2015) at diurnal to seasonal and longer timescales. These variations in near-surface turbulence must be taken into account for climate models to reproduce the observed seasonal cycle of surface properties and spatial structure of the depth of the thermally well-mixed near surface layer of the ocean. The dramatic
spatial and temporal patterns of ocean turbulence help shape ocean tracer distributions (heat, dissolved greenhouse gases, nutrients) and will regulate the distribution of temperature in a changing climate.

Figure 5.4: Estimate of the average diapycnal diffusivity between 250–1000 m calculated by applying fine structure techniques to Argo float data. Only bins with at least three estimates are plotted and regions with insufficient data are coloured grey. This figure was created using updated data through April, 2018 with the techniques from Whalen et al. (2012).

Ocean turbulent mixing requires energy sources. Surface wind and buoyancy forcing, the larger-scale ocean circulation itself, and the barotropic tides are all thought to be significant sources of the energy that drives mixing (Wunsch and Ferrari, 2004). Often this energy first passes through the ocean’s pervasive field of internal gravity waves (Wunsch and Ferrari, 2004). Such waves are analogous to waves on the ocean surface, but are the much larger, slower cousins that exist within a stratified fluid. They propagate and refract through the varying ocean circulation, often breaking into turbulent mixing far from their sources (Eden and Olbers, 2014; Melet et al., 2016; Meyer et al., 2016; Zhao et al., 2016).

Many of these energy sources that drive turbulent ocean mixing are expected to change with a changing climate. The energy contributing to the internal waves from the winds and the subsequent turbulence will be altered by changes in storm activity and sea-ice coverage. For example, the increasing extent of ice-free Arctic Ocean has already been observed to lead to increased wind-driven internal waves (Dosser and Rainville, 2016). The Southern Annular Mode is expected to intensify as a result of climate change (Young et al., 2011; Jones and Cheung, 2015), bringing with it stronger winds, and more wind-energy input over most of the Southern Ocean and a more intense mesoscale eddy field (Hogg et al., 2015). These intensified eddies will energize the internal wave field where they flow over rough bathymetry, in turn increasing topographically generated mixing in the Southern Ocean (Melet et al., 2015). Changes in the near-bottom stratification will alter the rate that the barotropic tides generate internal waves, thereby altering the strength and distribution of the tidally generated mixing. Given the sparsity of the ocean observing system and the large spatial and temporal variability of ocean mixing, climate-change related changes in the overall levels and patterns of ocean turbulence are only just now beginning to be documented observationally (Dosser and Rainville, 2016).

Despite the difficulties with observationally documenting ongoing changes in turbulent mixing, several of the robustly projected changes in the state of the large scale ocean will directly impact ocean turbulence over the coming century. The projected more intense Southern Ocean wind forcing and mesoscale eddies (Cross-Chapter Box 5 in Chapter 3) are very likely to lead to more intense turbulent vertical mixing (by all metrics) in the Southern Ocean. The increased stratification in the tropics and subtropical gyres that have been noted in Section 5.2.2.2 as a very likely consequence of surface-intensified climate change are likely to lead to a net reduction in the vertical diffusivities of nutrients and other gases within the main thermocline, reducing the
flux of nutrients into the euphotic zone and increasing the gradient in oxygen concentrations between the near surface ocean and the interior. Dramatic seasonal reductions in Arctic sea ice are virtually certain to increase wave-driven mixing in the upper Arctic Ocean and at rough topography surrounding the Arctic Ocean basin.

Interactions between ocean turbulent mixing and other parts of the climate system can play a significant role in amplifying climate changes and must be taken into account in projecting climate changes. For example, changes in internal waves in the Arctic Ocean can in turn drive mixing that warms the surface by drawing upon large subsurface reservoirs of warmer, saltier water, either locally or over rough topography at the basin margins (Carmack et al., 2015; Rippeth et al., 2015; Meyer et al., 2017); this positive feedback mechanism can contribute to rapid Arctic Ocean warming, loss of multiple year ice and increased periods of ice-free conditions. These changes impact societies around the Arctic basin (as discussed in Chapter 3 Sections 3.5 and 3.3, Box 3.3) and trigger further climate feedbacks due to changes in the ocean circulation and Earth’s albedo. Climate-induced changes to winds, tides and ocean stratification have the potential to change ocean turbulence leading to potentially significant impacts on the marine ecosystem and the oceans ability to mitigate/moderate climate change.

5.2.2.3 Changes in Ocean Carbon

The concentration of dissolved inorganic carbon (DIC) in the sea changes in response to (i) the exchange of CO₂ across the air-sea interface, (ii) consumption of seawater CO₂ during organic matter production via photosynthesis, (iii) CO₂ production during the respiration of ocean organic matter, (iv) the formation of mineral forms of calcium carbonate (CaCO₃) by marine biota, (v) the (mostly abiotic) dissolution of these carbonates, and (vi) ocean transport and mixing (Sarmiento and Gruber, 2002). These processes and their interactions lead to a complex spatio-temporal pattern of variability in DIC in the sea and since these processes are of ‘natural’ origin, one often refers to their impact as the ‘natural’ carbon cycle (Sarmiento and Gruber, 2002; McNeil and Matear, 2013).

The human-induced increase in atmospheric CO₂ perturbs the natural carbon cycle through three means: First, it adds additional, i.e., anthropogenic CO₂ to the ocean via enhanced air-sea gas exchange. This anthropogenic CO₂ accumulates first in the surface ocean and is then transported to depth by ocean currents and mixing. This is the primary process that governs the evolution of the net global uptake of CO₂ by the ocean (Sarmiento et al., 1992). Second, newly accumulated CO₂ alters the CO₂ chemistry in the ocean (Zeebe, 2012) causing ‘ocean acidification’ (see below). Third, the global climate change induced by the increase in the atmospheric concentration of CO₂ and other greenhouse gases alters main ocean carbon cycle processes in response to the changing ocean temperature, circulation and biological processes, with potential feedbacks back to atmospheric CO₂ (Joos et al., 1999; Ciais et al., 2013).

Since AR5, new global-scale data synthesis products, novel methods for their analyses, as well as progress in modeling have substantially increased our quantitative understanding of each of the above processes, improving also the confidence levels. The most important progress concerns the data-based quantification of the temporal variability of the ocean carbon sink. While AR5 assessed primarily the climatological mean processes governing the ocean carbon cycle, the most recent work now permits us to assess how these processes have changed in recent decades in response to climate variability and change. Here we focus specifically on the open ocean carbon cycle.

The analyses of the steadily growing number of surface ocean CO₂ observations (now more than 20 million observations, SOCATv6 (https://www.socat.info/index.php/2018/06/19/v6-release/) demonstrate that the net ocean uptake of CO₂ from the atmosphere has increased from around 1.2±0.5 Pg C yr⁻¹ in the early 1980s to 2.0±0.5 Pg C yr⁻¹ in the years 2010-2015 (Rödenbeck et al., 2014; Landschützer et al., 2016). Once the outgassing flux of 0.45 Pg C yr⁻¹ stemming from river derived carbon (Jacobson et al., 2007) is accounted for, these new observations imply that the global ocean uptake of anthropogenic CO₂ increased from 1.7±0.5 Pg C yr⁻¹ to 2.5±0.5 Pg C yr⁻¹ between the early 1980s and 2010-2015 (Rödenbeck et al., 2014; Landschützer et al., 2016; Le Quéré et al., 2018). This increase is supported by the current generation of ocean carbon cycle models (Le Quéré et al., 2018), and commensurate with the increase in atmospheric CO₂.
Thus, there is high confidence from surface ocean carbon data that the ocean sink has increased in the last two decades in response to the growth of atmospheric CO2. The multiple lines of evidence indicate that it is very likely that the ocean has taken up about 25±5% of the global emissions of CO2 since the mid 1980s from the burning of fossil fuels, cement production, and land use change.

Alongside the global scale picture, these newly available surface ocean observations also reveal a substantial degree of variability at interannual and decadal scales (Rödenbeck et al., 2015; Landschützer et al., 2016; Le Quéré et al., 2018). Most notable are the air-sea CO2 flux variations in the tropics linked to ENSO variations (Rödenbeck et al., 2015; Landschützer et al., 2016), as well as the strong decadal variations in the high latitudes, especially the Southern Ocean (Landschützer et al., 2015; Munro et al., 2015; Ritter et al., 2017). Although the data scarcity in the Southern Ocean remains a challenge for identifying such decadal variations and trends (Fay et al., 2014; McKinley et al., 2017), the analysis of multiple data sets using different methods support the existence of this variations (see also chapter 3). Fluctuations in the Southern Ocean CO2 flux are important as they impart a substantial imprint also on the global uptake fluxes with reduced Southern Ocean uptake in the 1990/2000 period being characterized by an exceptionally weak global net uptake of only about 0.8 ± 0.5 Pg C yr⁻¹. While the weakening ocean CO2 sink during the 1990s was primarily caused by a southward shift and intensification of the westerly winds over the Southern Ocean that enhanced the outgassing of natural CO2, the subsequent strengthening of the sink from 2000 onwards appears to arise from a combination of changes in wind, temperature, and circulation. The causes for the decadal changes in the high latitudes of the northern hemisphere are less clear (Landschützer et al., 2016).

The majority of ocean carbon cycle models do not capture this decadal variability (Le Quéré et al., 2018), for reasons presently not well understood. This uncertainty precludes a formal attribution study at this stage, but the majority of the analyses suggest that decadal scale fluctuations in ocean-atmosphere CO2 fluxes are more likely the result of natural (unforced) variations in the climate system and to lesser degree a result of human-induced climate change (McKinley et al., 2017). That said, the weakening trend of the Southern Ocean sink in the 1990s was consistent with the existence of the stratospheric ozone hole, which helped to drive the southward shift of the westerly wind belt over the Southern Ocean (Gillet and Thompson, 2003).

There is growing evidence that the ocean carbon sink exhibits decadal variability at regional scales that significantly alter the globally integrated sink (medium confidence), but there is limited understanding of the role of specific driving processes.

Substantial progress since AR5 was also made with regard to the determination of the oceanic uptake and storage of anthropogenic CO2 within the ocean’s interior. This is largely a result of the continuing efforts to re-measure DIC along many of the repeat hydrographic lines that were occupied during the 1980s and 1990 (Talley et al., 2016a), alongside the preparation of a global quality-controlled database of ocean interior observations (Olsen et al., 2016a). Several studies analyzed the changes in the amount of anthropogenic CO2 that have accumulated between different occupations in the different ocean basins (Wanninkhof et al., 2010; Pérez et al., 2013; Woosley et al., 2016; Carter et al., 2017), confirming that the anthropogenic CO2 taken up from the atmosphere is transported to depth, where most of it is stored. Using a newly developed reconstruction method, Gruber et al. (2018) extended these results to the globe. They find that between 1994 and 2007 the global ocean has accumulated an additional 32 ± 5 Pg C of anthropogenic CO2 taken up from the atmosphere, bringing the total inventory for the year 2007 to 150±20 Pg C. Extrapolating this estimate to the year 2010 gives an inventory of 158±18 Pg C, which is statistically indistinguishable from the ‘best’ estimate provided by Khatiwala et al. (2013) of 155±31 Pg C and more recently also found from a steady-state ocean model (DeVries, 2014). This estimate also confirms the model-based estimate of AR5, which suggested an ocean inventory of anthropogenic CO2 in the year 2011 of 155 Pg ±31 Pg C (AR5, chapter 6).

Thus, there is high confidence that over the period 1750 until 2011, about 29±5% of the global anthropogenic emissions have accumulated in the ocean. The consistency between surface ocean observations and the ocean interior data-based reconstructions provide robust evidence that this uptake fraction has not changed in a statistically significant manner in the last few decades and remains consistent with AR5.

Detailed analyses of the spatial structure of the change in storage of anthropogenic CO2 confirm the variable nature of the ocean carbon sink suggested by the surface observations. This is the case especially in the North Atlantic, where the sink was substantially lower between 1994 and 2007 relative to the prior decades.
(Pérez et al., 2013) and the Southern Ocean, most likely a consequence of changes in ocean circulation
(DeVries and Weber, 2017). The change in anthropogenic CO$_2$ between 1994 and 2007 occurs throughout
the upper 1000 m, but with very different penetration depths, reflecting largely differences in the efficiency,
with which the anthropogenic CO$_2$ is transported from the surface to depth (Figure 5.5). This spatial
distribution of how the amount of anthropogenic CO$_2$ has changed between 1994 and 2007 is remarkably
similar to the distribution of anthropogenic CO$_2$ reconstructed for 1994 (Sabine et al., 2004). This is a result
of primarily the long time-scales over which the anthropogenic CO$_2$ has been perturbing the global carbon
cycle, having led to the establishment of a transient steady state (Gammon et al., 1982).

![Figure 5.5: Vertical sections of the change in anthropogenic CO$_2$ between the JGOFS/WOCE era (~1994)
and the Repeat Hydrography/GOSHIP era (~2007). Shown are the zonal mean sections in each ocean basin
organized around the Southern Ocean in the center. The upper 500 m are expanded. Contour intervals are 2
µmol kg$^{-1}$. From Gruber et al. (2018).](image)

Regarding the future fate of the ocean carbon sink, there has been limited new insights since AR5. AR5
concluded based on their analyses of the results from the CMIP5 that while it is *virtually certain* that the
ocean will continue to uptake CO$_2$ from the atmosphere, there was much less clarity about the magnitude and
direction of the climate change induced changes in this uptake (Ciais et al., 2013). Most models agreed that
the warming and the increased stratification caused a reduction in the uptake, particularly in the high
latitudes, but there was no consensus with regard to the role of biology. The work since then focused on
either the limits of the linear feedback analysis framework (Schwinger et al., 2014; Schwinger and Tjiputra,
2018), or the long-term evolution, i.e., extending the simulations beyond 2100 (Randerson et al., 2015).
Although the evidence is still rather limited, the existing results suggest that global warming will lead to a
sustained reduction in the global ocean uptake of CO$_2$ from the atmosphere, and that this carbon-climate
feedback might be larger than that for the land.

The oceanic uptake of anthropogenic carbon not only increases the ocean’s pool of DIC, but alters also its
chemistry. The DIC pool is made up of three species of inorganic carbon: (i) bicarbonate (HCO$_3^-$), (ii)
carbonate (CO$_3^{2-}$) and (iii) aqueous carbon dioxide (CO$_2(aq)$), which are typically found at a ratio of 100:10:1.
These three inorganic carbon species are coupled via a set of reversible reactions known as the ‘buffer
system' and their relative abundance is strongly controlled by ocean pH. For example, if the ocean pH lowers (or acidity increases) by one pH unit from its typical value in seawater of 8.1 to 7.1, then CO$_2$ concentration becomes more abundant, while CO$_3^{2-}$ concentrations are reduced. A further implication of the coupled carbon speciation reactions is that as the ocean absorbs more CO$_2$ from the atmosphere, the pH drops, known as 'ocean acidification, which then drives a concomitant reduction in CO$_3^{2-}$ ions. The impacts of ocean acidification and carbonate ion reductions (i.e. pH and CO$_3^{2-}$ ion reductions) on habitats and specific taxa are considered in greater detail in Sections 5.2.3 and 5.3.3.

Analyses of direct measurements of ocean acidity trends from ocean time-series and derived pH changes from shipboard studies show consistent decreases in ocean pH have occurred over the past few decades. pH reductions range between 0.013 to 0.026 pH units decade$^{-1}$ over records that span up to 25 years in some places and pH declines are found everywhere. Trends calculated from direct measurements from repeat occupations of specific locations show a similar value of around -0.02 pH units decade$^{-1}$ for diverse oceanic regions (Appendix 5.A, Table 1). At larger spatial scales (e.g., for broad ocean basins and regions), pH trends are assessed using shipboard observations of the fugacity of CO$_2$ and estimates of ocean alkalinity (Takahashi et al., 2014; Lauvset et al., 2015). Between 1991–2011, mean ocean pH has declined by 0.018 ± 0.004 decade$^{-1}$ in 70% of ocean biomes, with the largest declines in the Indian Ocean (−0.027 decade$^{-1}$), eastern Equatorial Pacific (−0.026 decade$^{-1}$) and the South Pacific subtropical (−0.022 decade$^{-1}$) biomes, with slightly lower rates of change in the Atlantic and Southern Oceans (Lauvset et al., 2015). In general, trends in the stability of mineral forms of aragonite and calcite (known as the 'saturation state') follow those of pH, with high latitude regions most vulnerable to under-saturation as they naturally have lower mean values. In addition to long term trends that are derived by excluding the seasonal cycle, the amplitude of seasonal changes in pCO$_2$ and hence also pH, have increased by 2.2±0.4 µatm decade$^{-1}$ between 1982-2014, which would enhance the acidification exposure to organisms (Landschützer et al., 2018).

pH trends are primarily attributed to the rising DIC levels due to the increased flux of anthropogenic carbon into surface waters from the atmosphere (Lauvset et al., 2015). Observations suggest that these lowered pH waters are being transferred into the ocean interior by overturning circulation in the north and south Atlantic (Rios et al., 2015), as well as the North Pacific (Watanabe and Kawamiya, 2017). However, for both the surface and in the ocean interior, natural variability is an important local player alongside secular changes (Lauvset et al., 2015; Rios et al., 2015). For example, the seasonal cycle in carbon speciation variables (pH, calcite and aragonite saturation states) is equivalent to around 10 years of the long-term trend at some sites (Takahashi et al., 2014; Sutton et al., 2016). Equally, oscillations linked to natural climate variability associated with El Nino, Pacific Decadal Oscillation and North Pacific Gyre Oscillation and other climate modes can be important drivers of variability (Sutton et al., 2017). Changes in alkalinity and in particular the ratio of alkalinity to DIC may be particularly important in driving regional trends in calcite and aragonite saturation states (Lenton et al., 2012; Takahashi et al., 2014).

We assess that it is virtually certain (high confidence – high agreement) that ocean pH is declining by ~0.02 pH units per decade everywhere time-series observations exist. This trend can be attributed to rising atmospheric CO$_2$ levels, although datasets remain sparse at high latitudes. There is low confidence that the amplitude of the seasonal changes in pH that organisms are exposed to has increased and for attributing drivers of change at regional scales.

In parallel to the ocean uptake of anthropogenic carbon, open ocean pH is projected to decline by 0.07 and 0.31 pH units for RCP2.6 and RCP8.5 scenarios, respectively (Bopp et al., 2013; Gattuso et al., 2015), with good agreement across the range of CMIP5 models (Figure 5.6a). Projected changes in pH are quite uniform with a relatively weak spatial variations and are greatest in the Arctic Ocean and the high latitudes of the Atlantic and Pacific Oceans due to their lower buffering capacity and are lowest in contemporary upwelling systems (Figure 5.6b). Other ocean regions show changes similar to the global average. Similar declines in the concentration of CO$_3^{2-}$ ions is predicted by the CMIP5 models (Bopp et al., 2013; Gattuso et al., 2015), with high latitude and Arctic Ocean regions first to be undersaturated due to their systemic vulnerability (Orr et al., 2005). Models also predict that the seasonal cycle in pH will be amplified in the future. In the ocean interior, the progressive decline in pH due to ongoing anthropogenic carbon uptake will be modulated by changes to ocean overturning and water mass subduction (Respandly et al., 2013) and may be augmented by modifications to interior ocean respiration rates (Chen et al., 2017). Seafloor changes in pH over the next century are highly localized and are linked to transport of surface anomalies to depth, with over 20% of the
North Atlantic sea floor deeper than 500m experiencing pH reductions greater than 0.2 units by 2100 under the RCP8.5 scenario, including canyons and seamounts designated as marine protected areas (Gehlen et al., 2014a). Changes in pH in the abyssal ocean (>3,000 m) deep, are greatest in the Atlantic and Arctic Oceans, with lesser impact in the Southern and Pacific Oceans by 2100, mainly due to circulation timescales (Sweetman et al., 2017). Model skill is generally good for trends in carbon speciation variables, such as pH, thus increasing confidence in their projections over the coming century (Bopp et al., 2013; Frölicher et al., 2016) (Figure 5.6a), although persistent model biases in the simulation of seasonal cycles should be noted, especially in the Southern Ocean (Mongwe et al., 2018).

Overall, there is high confidence that the future ocean will experience pH drops of between 0.1 or 0.3 pH units by 2100, depending on whether we follow the RCP2.6 or RCP8.5 emissions scenario respectively, that are exacerbated in polar regions. There is medium confidence, due to their reliance potential for parallel changes in ocean circulation, that the Arctic and North Atlantic seafloors will experience the largest pH changes over the next century.

Figure 5.6: Panels a, d, g and j display observation-based estimates (black lines) and simulated global changes over the period of 1950 to 2100. For surface pH, upper 100 m nitrate concentrations, O₂ concentration averaged over 100 to 600 m depth, and NPP integrated over the top 100 m. Panels b, e, h and k show spatial patterns of simulated change in surface pH, upper 100 m nitrate concentrations, O₂ concentration averaged over 100 to 600 m depth, and NPP integrated over the top 100 m. Panels c, f, i and l display time series of the percentage of total uncertainty ascribed to internal variability uncertainty, model uncertainty, and scenario uncertainty in projections of global annual mean
Anthropogenic trends in carbon speciation variables (pH, saturation of calcite and aragonite) in the open ocean emerge from the background variability by 2010 or 2020 for the tropical ocean or regions of low and high latitude upwelling, respectively (Keller et al., 2014; Rodgers et al., 2015), with the detection of open ocean pH trends requiring around 15 years of data is needed for the resolving of a pH trend (Henson et al., 2016). For aragonite saturation, the anthropogenic signal has already emerged from the background over much of the ocean (Rodgers et al., 2015). By the end of century, the anthropogenic pH signal is projected to have emerged over the entire surface ocean (Frölicher et al., 2016). While internal variability and model uncertainty are most important for the next few decades, the largest source of uncertainty in the projection for end-of-century open ocean pH changes (Figure 5.6c) at both global and local scale is the emissions scenario (Frölicher et al., 2016), with large changes in the trajectory and magnitude of pH changes between the RCP8.5 and RCP2.6 scenarios (Figure 5.6a and c). Time of Emergence is longer in coastal regions with higher variability of the carbonate species (Sutton et al., 2018).

5.2.2.4 Changing Ocean Oxygen

Ocean oxygen (O2) levels at the surface are controlled by the balance between oxygen production during photosynthesis and temperature controlled equilibrium with the atmosphere through air-sea exchange. Deeper in the water column, consumption of oxygen during respiration and redistribution by ocean circulation and mixing are dominant. Due to the localisation of photosynthetic activity and relatively short timescales of atmospheric equilibrium, surface waters are close to atmospheric equilibrium and relatively high in oxygen. In contrast, little in situ production means that O2 consumption during respiration and the interior transport of oxygen rich surface water by ocean circulation dominate in the interior ocean. Oxygen minimum zones (OMZs) arise in the interior ocean where the circulation cannot supply enough oxygen enriched waters to meet the respiratory demand, especially in the shadow zones in the eastern sides of tropical oceans where the time-mean advective pathways of the circulation are blocked by topography rather than originating in the directly ventilated surface ocean. Changes to the extent of low oxygen regions in response to climate change can promote the ocean source of nitrous oxide (Martinez-Rey et al., 2015; Battaglia and Joos, 2018), which is a potent greenhouse gas, and have occurred in the geologic past in response to climate warming (Praetorius et al., 2015).

Changes in ocean oxygen has been appraised by analysing datasets going back to the 1960s. The most recent study estimates oxygen levels have dropped by 961 ± 429 Tmol dec−1, or by 2% in total since the 1960s (Schmidtko et al., 2017). By region, the largest trends are seen in the Equatorial Oceans that contain the lowest oxygen waters, North Pacific, Southern Ocean and South Atlantic (from 210 to 119 Tmol dec−1 over or 1-3 umol kg−1 decade−1 in the upper 1,200m (Figure 5.7), while the Arctic Ocean shows strong a decline (Schmidtko et al., 2017). While other studies agree in the bulk trends and at high latitudes, there are differences at the regional scales in the tropics related to data merging and analysis methods that affects confidence locally (Helm et al., 2011; Ito et al., 2017). Syntheses of datasets from local time series tend to document stronger trends, with oxygen declines of over 20% at sites in the Northeastern Pacific between 1956-2006 and the California Current between 1964-2006 (Levin, 2017). Despite holding the highest inventory of oxygen in the ocean, oxygen levels in Southern Ocean contributed 25% to the global decline between 1970–1992 (Helm et al., 2011) and have fallen by over 150 Tmol decade−1 from the 1960s to present (Schmidtko et al., 2017). Observations along ocean cruises as part of the CLIVAR programme have also documented broad thermocline oxygen declines in the northern hemisphere oceans, accompanied by oxygen increases in subtropical and southern hemispheres (Talley et al., 2016b).

Around 15% of the global ocean loss of oxygen since the 1960s, primarily in the upper ocean is associated with warming induced declines in saturation (Schmidtko et al., 2017), similar to previous estimates (Helm et al., 2011). The role of other processes, namely changes in ocean circulation or oxygen consumption during respiration, are more difficult to quantify directly due to the lack of clear proxies and their often overlapping nature. For example, while it is likely that increasing ocean stratification is driving an important part of the
decline in oxygen (Talley et al., 2016b), this is not as clearly quantifiable as temperature driven changes to stratification (Levin, 2017). In specific ocean regions summarised by Levin (2017), detailed analyses of oxygen trends have attributed changes to ocean physics in western Northern Pacific (Whitney et al., 2013; Sasano et al. (2015), the southern California Current region (Goericke et al., 2015), Santa Barbara Basin (Goericke et al., 2015) and St Lawrence Estuary (Goericke et al., 2015). In regions of high mesoscale activity, such as the tropical north Atlantic, low oxygen eddies can have a significant impact on oxygen dynamics (Karstensen et al., 2015; Grundle et al., 2017).

![Figure 5.7](image-url) Absolute change in dissolved oxygen (umol kg⁻¹ decade⁻¹) between water depths of (a) 0 and 1,200 m, and (b) 1,200 m and the sea floor over the period 1960-2010.

Oxygen is also declining in the deepest parts of the ocean, where a global oxygen decline of 70 Tmol yr⁻¹ below 1,200 m has been determined. By region, the strongest declines in deep ocean oxygen occur in the Arctic, North Pacific and Southern Oceans, while the North Atlantic has experienced a moderate oxygen increase below 1,200 m (Figure 5.7). Regional changes in the deep ocean are attributed to changes in large scale ocean circulation that affects the ventilation of the deep ocean by Pacific, Atlantic and Antarctic deep waters (Watanabe et al., 2003; Stendardo and Gruber, 2012). Indeed, globally, the observed oxygen decline is negatively correlated with ocean heat content changes (Ito et al., 2017). Changes to respiration rates, either due to temperature enhancement or in the amount/quality of organic material can also be important and the enhanced respiratory demand associated with an intensified Monsoon has been invoked as a driver of the expansion of the Arabian Sea OMZ (Lachkar et al., 2018).

Overall there is high confidence from high agreement and robust evidence that the the oxygen content of the global ocean has declined. There is medium confidence that the largest regional changes have occurred in the Southern Ocean, North Pacific and South Atlantic, but there is low confidence in tropical ocean oxygen declines due to limited agreement from limited evidence.

Oxygen ocean changes are affected by climate variability on interannual and decadal timescales, especially for the tropical ocean OMZs (Deutsch et al., 2011). ENSO variability affects the heat content and respiration rates in the tropical Pacific, which then regulates oxygen solubility and respiratory demand (Ito and Deutsch, 2013; Edebbar et al., 2017). These drivers may then be combined with modifications to overturning and ventilation of OMZs by lateral jets (Duteil et al., 2012). Centennial scale studies based on isotope proxies for low oxygen regions have demonstrated fluctuations in OMZ extent linked to decadal changes in tropical trade winds that affects interior ocean respiratory oxygen demand, suggesting that it is difficult to attribute recent changes in the Pacific OMZ to anthropogenic forcing alone (Deutsch et al., 2015). Parallel work based on oxygen observations (Czeschel et al., 2012), as well as modelling (Duteil et al., 2018) supports the importance of decadal scale variability in the eastern tropical Pacific OMZ. There is some evidence for the potential of a modulating impact on tropical Pacific oxygen from atmospheric deposition of nitrogen and iron (Ito et al., 2016; Yang and Gruber, 2016).
At the global scale, there is high agreement that only around 10-20% of the observed oxygen decline can be explained by a warmer ocean, leading to high confidence that other processes associated with ocean physics and biogeochemistry has driven the majority of the observed oxygen decline. For the tropical Pacific OMZ, there is medium confidence arising from medium agreement from medium evidence that low frequency decadal changes in ocean physics have controlled past fluctuations in OMZ extent.

Future changes in oxygen can be appraised from CMIP5 models that account for the combined effects of ocean physics and biogeochemistry. Globally, oxygen is predicted to decline by 3.45±0.44% across the range of CMIP5 models (Bopp et al., 2013) (Figure 5.6d). Increased tropical ocean stratification reduces interior ocean oxygen by diminishing pathways of ventilation in the subtropical gyres and by inhibiting turbulent mixing with the oxygen-rich surface ocean (see Section 5.2.2.2.4). This relatively robust global modelled trend (Figure 5.6d) however masks important uncertainties in the projection of regional trends (Figure 5.6e), particularly in the tropical ocean OMZs (Bopp et al., 2013; Cocco et al., 2013; Cabré et al., 2015). In these regions, an overall projection of oxygen increases is more uncertain as the inter-model standard deviation is over four times greater than the projected trend (Bopp et al., 2017). The uncertainty in the trends in tropical ocean OMZs arises due to the fact that oxygen depletion due to warming induced reductions in oxygen saturation are opposed by oxygen enrichment due to reduced oxygen consumption during respiration in response to predicted declines in marine export production, as well as biases due to model resolution in the tropics and the length of the model spin up (Bopp et al., 2017). At the seafloor, bathyal (200–3000 m depth) regions of the North Pacific, North Atlantic, Arctic and Southern Oceans may see oxygen declines by 0.3% to 3.7% by 2100 (relative to 2005), while abyssal ocean changes are lower and localised around regions in the North Atlantic and Southern Ocean affected by deep water formation (Sweetman et al., 2017). There is high confidence that the largest changes in deep sea systems will occur after 2100 (Battaglia and Joos, 2018).

Accordingly, there is medium confidence in the projected 3.5% loss of total ocean oxygen by 2100, relative to 2005, across CMIP5 models, due to the combination of warming and increased stratification and low confidence at regional scales (especially the tropical ocean) due to inter-model variability and reduced skill (Bopp et al., 2013). Simulations extended to 2300 suggest that by 2150 the trend of declining tropical ocean oxygen (both in terms of concentrations and volume of low oxygen waters) may reverse itself, mainly due to the effect of strong declines in primary production and organic matter fluxes to the ocean interior (Fu et al., 2018), but with low confidence due to limited evidence. At the global scale, 10,000 year intermediate complexity model simulations find that ocean oxygen loss shows near linear relationships to equilibrium temperature, itself linearly related to cumulative emissions, and any climate mitigation scenario will reduce peak oxygen loss by 4.4% per degree Celsius of avoided warming (Battaglia and Joos, 2018).

A median of just over 25 years of data is required for an interior ocean oxygen trend (average concentration over 200-600 m depth) to emerge from the natural background, but more than 40 years data is needed in specific dynamic regions (Henson et al., 2016). By 2100, the anthropogenic signal in interior ocean oxygen is projected to have emerged above uncertainties associated with internal and inter-model variability for around 25% of the ocean area in the 100-600m depth range (Frölicher et al., 2016) (Box 5.1). In particular, the oxygen signal in the northern subtropical Pacific is predicted to emerge by 2030 to 2040 (Frölicher et al., 2016; Long et al., 2016). As seen for pH, the scenario uncertainty is also an important component of the total uncertainty in projection of interior ocean oxygen by 2100 (Figure 5.6f), but the contributions of internal variability and model uncertainty remain important, especially at more regional scales, by 2100 (Frölicher et al., 2016). At the zonal scale, robust detection of anthropogenic trends in oxygen emerge from climate models much faster, due to reduced noise (Andrews et al., 2013). Overall, it is likely that time of emergence estimates are conservative given that models tend to underestimate the low frequency decadal scale variability in the tropical Pacific Ocean.

Taken as a whole, we assess that there is high confidence that ocean oxygen loss is occurring today and will increase in the future in response to climate change. While projections at regional and decadal scales may be uncertain due to local processes and natural variability, there is a robust link between global warming and ocean oxygen loss.

5.2.2.5 Changing Ocean Nutrients and Primary Productivity
Light and nutrients support the base of marine food webs and nutrient levels in the surface waters of different parts of the open ocean result from the interplay of external nutrient supply (e.g., from dust and ocean mixing/upwelling) and internal cycling (consumption by primary producers and replenishment during degradation of organic matter). Nutrients that are remineralized in the interior ocean need to be advected or mixed back into the surface ocean before they can again support phytoplankton growth. The depth of ocean mixing also controls the light environment that phytoplankton are exposed to, but outside of dark polar regions during winter, spatial variations in phytoplankton activity reflect changes in nutrient levels. Indeed, the overall production of particulate organic matter by phytoplankton primary production is regulated by available nutrient resources and is an important component of ocean biogeochemical cycling and atmospheric CO₂ uptake (see Section 5.2.2.3), as well as being a source of food for both pelagic and benthic ecosystems. The greatest rates of phytoplankton productivity occur in regions where nutrient supply is maximised by upwelling and upwards mixing of deep nutrient reserves (Falkowski, 1998) (Figure 5.8). Equally, the distribution of specific forms of plankton is closely linked to the match between the relative supply of different nutrients and their requirements (Hagstrom et al., 2017), which themselves can be strongly sensitive to warming (Yvon-Durocher et al., 2015; Yvon-Durocher et al., 2017). For example, the competitive success of nitrogen fixing plankton in the future ocean will be controlled by warming (Fu et al., 2016), the relative supply of iron, nitrogen and phosphorus (Ward et al., 2013) and ocean acidification (Hutchins et al., 2013; Hutchins et al., 2015).

Overall, there is high confidence from multiple lines of evidence that future changes to phytoplankton primary production and community structure will occur from changes in magnitude of nutrient supply and the ratio of these different nutrients.

**Figure 5.8:** Map of the dominant limiting resource (Moore et al., 2013), updated to include new experiments from the north Pacific, tropical Atlantic and south east Atlantic (Browning et al., 2017; Shilova et al., 2017). The background is depth integrated primary productivity using the Vertically Generalized Productin Model (VPGM) algorithm. Symbol colouring indicates the primary limiting nutrients inferred from chlorophyll and/or primary productivity increases following artificial amendment of: N (green), P (black), Fe (red), Co (yellow) and Zn (cyan). Divided circles indicate potentially co-limiting nutrients, e.g., a red-green divided circle indicates Fe-N co-limitation.
Since AR5, our understanding regarding the role of different nutrient resources in different regions has grown. Diverse studies (including shipboard experiments and use of protein biomarkers) have highlighted nitrogen and phosphorus limitation in the stratified tropical ocean regions accompanied by widespread iron limitation at high latitudes and in upwelling regions that typically have elevated levels of productivity (Figure 5.8) (Moore et al., 2013; Saito et al., 2014; Browning et al., 2017; Tagliabue et al., 2017). Moreover, more extensive experimental work has demonstrated nitrogen-iron co-limitation at the boundary of the south Atlantic gyre system (Figure 5.8) that is linked to the common occurrence of simultaneous drawdown of nitrogen (as nitrate) and iron levels that is typical of all gyre/upwelling boundaries, driving greater microbial diversity (Browning et al., 2017). These findings echo the substantial overlap in nutrient stress biomarkers seen in transition between upwelling and gyre systems in the Pacific (Saito et al., 2014).

There is high confidence that changes in ocean primary production will be controlled by changes to iron in the upwelling regions and the Southern, North Atlantic, and sub-Arctic Pacific Oceans, with nitrogen and phosphorus being important in the low productivity tropical ocean gyres (robust evidence and high agreement).

Nutrient changes in the surface ocean over the last few decades reflect modifications to physical supply by mixing and upwelling, as well as external input, typically from dust and internal cycling. In the open ocean, increasing stratification is occurring in the major tropical regions (Dave and Lozier, 2013; Talley et al., 2016b; Kwiatkowski et al., 2017; and see also Section 5.2.2.2). Increased stratification is expected to retard nutrient supply to the surface by suppressing upwelling and turbulent mixing (see also Section 5.2.2.2.4). Supply of iron at in the Southern Ocean is mostly regulated by winter mixing (Tagliabue et al., 2014), as well as changes linked to modifications to iron supply linked to sea ice, icebergs and glacial melt. Increasing inputs of anthropogenic nitrogen from the atmosphere are perturbing ocean nutrient levels (Jickells et al., 2017). In the western Pacific in particular, additional atmospheric nitrogen input has enriched the system in nitrogen, which raises the nitrogen to phosphorus ratio and induces a progressive shift towards phosphorus limitation in this region (Kim et al., 2014). Enhanced microbial turnover of dissolved organic nitrogen and phosphorus may act to counter balance reduced vertical supply due to enhanced stratification, but this requires microbial turnover to mainly occur in the upper ocean (Letscher et al., 2013; Letscher and Moore, 2015). Microbial turnover of dissolved organic nitrogen and phosphorus in the upper ocean depends critically on their lability (readiness to change) (Banse, 1968; Letscher et al., 2013) and the presence of the necessary metal cofactors for the hydrolysing enzymes (Mahaffey et al., 2014; Browning et al., 2017).

Overall, the marked specificity of the processes governing nutrient supply in distinct ocean regions, results in low confidence regarding the ability to project the impact of future changes, due to limited evidence of past changes that are robustly understood and reproduced by models.

Much effort has been made to extract information on the trends in primary production over the recent past using satellite datasets, which have the potential to provide a global picture or change. These datasets rely on mathematical algorithms to convert ocean colour into ocean chlorophyll (an index of phytoplankton biomass), phytoplankton carbon or primary productivity. Globally, a range of studies have reported insignificant changes in open ocean chlorophyll of $<\pm 1\%$ yr$^{-1}$ (Boyce et al., 2014; Gregg and Rousseaux, 2014; Boyce and Worm, 2015; Hammond et al., 2017). For specific open ocean regions, the overall low change estimated globally may mask much larger changes at regional scales (Colella et al., 2016; Hammond et al., 2017; Mélin et al., 2017), with recent datasets merged from different satellites reporting trends of $\pm 4\%$ in different regions with increases at high latitudes and moderate decreases at low latitudes (Mélin et al., 2017), but corroboration with in situ time series is lacking. Attempts to compare satellite trends with in situ data from open ocean time series sites have found substantial mismatches (Saba et al., 2010), but display reasonable correlation in higher biomass coastal regions (Kahru et al., 2009) and, at present, satellite records are not yet long enough to unambiguously isolate long term climate related trends from natural variability (Beaulieu et al., 2013). Indeed, satellite derived trends associated with natural climate variability, e.g., El Nino or the Indian Ocean Dipole can be substantial (Brewin et al., 2012; Currie et al., 2013; Racault et al., 2017). While some studies report good intercomparability across different satellite datasets (Mélin et al., 2017), others highlight mismatches between various satellite algorithms for primary productivity in absolute values and decadal trends (Gómez-Letona et al., 2017).
In conclusion, despite the medium agreement between satellite-based studies in the literature, the reliability of these trends is undermined by the lack of corroborating in situ measurements and other validation time series. Consequently, there is low confidence in satellite based trends in global ocean primary productivity, especially at regional scales where distinct sets of poorly understood processes dominate.

Globally, CMIP5 models project overall declines in surface ocean nitrate concentrations (Figure 5.6g) that are localised in tropical upwellings, subpolar gyres and other frontal systems (Figure 5.6h). Iron concentrations are projected to increase in the future, due to changes in ocean physical lateral transport and reduced consumption (Misumi et al., 2013). Currently, projections from CMIP5 model simulations do not account for the changing supply of nitrogen or iron from the atmosphere (e.g., accounting for anthropogenic sources), which lowers confidence in these projections - especially in regions experiencing high levels of atmospheric deposition.

Globally, net primary productivity trends in CMIP5 models mirror the decline in nutrients, with a decline in the low latitudes and an increase at higher latitudes that results in a low global overall decline (Figure 5.6j). The apparent virtual compensation in the effect of primary production is underpinned by substantial spatial and inter-model heterogeneity in the projected changes (Figure 5.6k). For example, in the tropical ocean a decline of 11 ± 24% in primary production is projected across the suite of CMIP5 models by 2100 under RCP8.5, with lesser changes under more moderate scenarios (Laufkötter et al., 2015). However, this large inter-model range can be restricted to ± 6% if emergent constraints from the historical record that link variability productivity to temperature anomalies are used, a four-fold decline in inter-model uncertainty (Kwiatkowski et al., 2017). Via this emergent constraint approach, there is a projected decline in global primary production of 6 ± 3% (Kwiatkowski et al., 2017). Projected declines in primary production in different biomes across CMIP5 models are generally attributed to increased stratification that reduces in surface nutrient concentrations in low latitude biomes (Cabré et al., 2014; Laufkötter et al., 2015; Fu et al., 2016) (Figure 5.8), but in some models the temperature enhancement of growth is the dominant driver (Laufkötter et al., 2015). Interacting controls by iron and light drive the response of primary productivity in the Southern Ocean (Laufkötter et al., 2015; Leung et al., 2015). On multi-century timescales, it is projected that enhanced physical-biogeochemical feedbacks act to suppress biological activity for a millennium (Moore et al., 2018). This arises due to the retention of essential nutrients in the ocean interior as changes to winds, sea ice and ongoing warming reduce the usual resupply to the surface ocean, which exacerbates the declining trend in CMIP5 model projections to 2100.

In the surface ocean, the progressive decline in upper ocean of major nutrients (nitrogen and phosphorus) with increasing stratification will affect the food quality available to upper trophic levels via its impact on the carbon to nitrogen to phosphorus ratio of organic matter. Projections with variable stoichiometry climate models suggest that in oligotrophic regions and the Arctic Ocean food quality will decline by over 20% due to changing phytoplankton nutrient ratios driving reductions in the nutritive content of organic matter (Kwiatkowski et al., 2017).

Changes in organic matter flux from surface waters (known as export production) in CMIP5 models are generally closely linked to changes in primary production in the surface layer. In general, CMIP5 models predict declines in export production of around 1–20% from 2000–2100 for the RCP8.5 scenario (Bopp et al., 2013; Fu et al., 2016; Laufkötter et al., 2016). Models with a more detailed representation of the marine food web tend to predict a stronger decline in export production than total organic matter production, highlighting the role of food web changes in driving changes to organic matter supply to the ocean interior (Fu et al., 2016). Parallel to these declines in export production, the flux of organic material to sea floor is also predicted to decline globally, with increases projected only in the polar regions (Sweetman et al., 2017). The projected changes in export production can be larger than global primary production because they are affected by both the magnitude of organic matter production in the surface ocean, and also how food web structure modulates the ‘transfer efficiency’ of particulate organic material, which then affects the sinking speed and lability of exported particles through the ocean interior to the sea floor (Bopp et al., 2013; Fu et al., 2016; Laufkötter et al., 2016).

Confidence in model projections can be appraised using the model skill in reproducing distributions of the limiting nutrient in specific ocean regions (Figure 5.8) as this will underpin projected changes in primary productivity. High model skill in reproducing surface distributions of nitrate and phosphate (Laufkötter et
al., 2015), raises confidence in projections in nitrogen and phosphorus limited systems. In contrast, poor model skill in reproducing iron dynamics (Tagliabue et al., 2016) lowers confidence in projections in iron limited regions (Figure 5.8). Despite apparently high skill in the reproduction of bulk nitrate and phosphate fields, skill is reduced when underlying biogeochemical and physical mechanisms (Duteil et al., 2012) or the ratio of resources is assessed. Indeed, properly representing the ratio of different resources correctly has been shown to have significant impacts on the response of primary productivity to environmental change (Moreno et al., 2017). Model skill in reproducing patterns of primary production and export from satellite derived estimates range from poor to average (correlation coefficients of 0.1-0.6 across different models (Laufkötter et al., 2016; Moreno et al., 2017)), but it should be noted that complete comprehensive observational datasets do not exist for these metrics with very few in situ observations. Improving model skill in ocean biogeochemical processes is important as trends in the future magnitude of changes in nutrient supply and associated impacts on organic matter production by primary production are related to the magnitude of model biases in contemporary ocean biogeochemistry (Fu et al., 2016).

Trends in surface nitrate, primary production and export production emerge from background variability after a median of around 30 years, but can take >50 years in some regions (Henson et al., 2016). Using CMIP5 models, it is suggested that detecting trends using ocean observing systems will be challenging because they often require a long time series (~35 years) and the results are only statistically representative of a small area (Henson et al., 2016). The anthropogenic trend in primary production does not emerge from background uncertainty at any geographic point due to internal and model dynamics by 2100 (Frölicher et al., 2016). Unlike pH and oxygen, scenario uncertainty is not a dominant component of the uncertainty in future projections of nutrient levels and organic matter production, which is instead dominated by model uncertainty, especially at regional scales (Figure 5.6 i and l) (Frölicher et al., 2016). That the between model uncertainty in the modelled trends in nitrate levels and productivity are larger in the uncertainty between the RCP2.6 and RCP8.5 scenarios (Figure 5.6 g and j) highlights the impact of the incomplete representation of these processes in climate models.

In summary, there is high confidence that future modifications to ocean temperatures, stratification and circulation will affect nutrient supply to the ocean surface, with medium confidence in the trends in nitrogen and phosphorus, but low confidence in projections of changes to iron supply. Incomplete representations of the major processes in climate models and in some cases, poor model skill, undermine confidence in projections. There is medium confidence (high agreement, limited evidence) from CMIP5 models that changes in nutrient supply will lead to modest decline in global productivity and a larger decline in export production by 2100. At regional scales, there is high confidence in strong spatial variability emerging, but low confidence in the direction of change. In general, uncertainty increases at each step along the cascade from changes to ocean physics, to nutrient supply, to primary production and finally to export production as our understanding worsens concomitantly.

[START BOX5.1 HERE]

**Box 5.1: Time of Emergence and Exposure to Climate Hazards**

The concept of time of emergence (ToE) is defined as the time at which the signal of climate change in a given variable emerges from a measure of the background variability or noise (AR5 WGI 11.3.2.1, SROCC Glossary). In associating a calendar date with the detection, attribution and projection of climate trends, the concept of a ToE has proved useful for policy and planning particularly through informing important climatic thresholds and the uncertainties associated with past and future climate change (Hawkins and Sutton, 2012). However, there is not a single agreed metric of ToE and the ToE for a given variable thus depends on choices regarding the space and time scale, the threshold at which emergence is defined and the reference period (AR5 WGI 11.3.2.1). Recently, the ToE concept has been expanded to include variables that describe ocean ecosystems, considering essential variables such as pH, carbonate ion concentrations, aragonite and calcite saturation states, nutrient levels and marine primary productivity (Box 5.1, Figure 1) (Ilyina et al., 2009; Friedrich et al., 2012; Keller et al., 2014; Lovenduski et al., 2015; Rodgers et al., 2015).

Here, we wish to emphasise how the ToE for essential variability may be linked to organisms and their exposure to hazard. In this regard, an important distinction to prior work reported in IPCC AR5 WGI
(Stocker et al., 2013) is that organisms respond to climate hazards that emerge locally, rather than to the global and basin-scale averages reported in IPCC AR5 WGI. As the noise from internal variability is greater at local scales than for bulk averaged quantities, emergence is much later. Moreover, variability is also much greater in the coastal ocean than for the open ocean considered here, which will be important for discussing hazard exposure for coastal species. For example, although signals of anthropogenic influences have already emerged from internal variability in the late 20th century for global and basin-scale averaged ocean surface and sub-surface temperature (very likely) (AR5 WGI SPM), their ToE and level of confidence vary greatly at local scales and in coastal seas (Frölicher et al., 2014). Pelagic organisms with small range size may thus be more (or less) at risk to warming with earlier (or later) ToE at the scale of the area that they inhabit.

ToE assessments typically quantify the internal variability using the standard deviation of the detrended data over a given time period (Keller et al., 2014; Rodgers et al., 2015; Henson et al., 2016; Henson et al., 2017), the scenario and model uncertainty associated with different climate scenarios and across available climate models (Frölicher et al., 2016), and in some cases the autocorrelation of noise (Weatherhead et al., 1998). As more components of ‘noise’ are accounted for, the ToE lengthens. Here we show the local ToE for five key ocean condition variables (Box 5.1, Figure 1). Calculated using the preindustrial period and presented relative to the period 1986-2005 as representative of the observation period, this accounts for internal and model variability (Frölicher et al., 2016). Interior ocean oxygen depletion emerges from local background variability only in the north Pacific subtropical gyre, Arabian Sea, southeastern Indian Ocean, north and central eastern Atlantic Ocean, whereas nitrate and net primary productivity will not generally emerge, except at some local regions, until the next century. From an observational standpoint, similar analyses that account for autocorrelation of noise suggest time series of around a decade are sufficient to detect a trend in pH or SST, whereas datasets spanning 30 years or longer are typically needed for detection of emergence at local scales for oxygen, nitrate and primary productivity (Henson et al., 2016).

**Box 5.1, Figure 1:** Time of emergence of key ocean condition variables: (A) sea surface temperature (SST), (B) surface acidity (pH), (C) interior oxygen (O₂), (D) surface nitrate (NO₃), and (E) depth integrated net primary production (NPP). The year of emergence represents the year when the mean change relative to the reference period of 1986-2005 is above the standard deviation of each variable over the historical period (Frölicher et al., 2016) (Section 5.2.2).

The rapidity of change, encompassed in the ToE, can be linked to concepts of exposure to hazard and vulnerability of biota. As organisms have evolved to be adaptable to natural variations in the environmental conditions of their habitats, changes to their habitat conditions larger than that typically experienced or specific biological thresholds such as upper temperature tolerance may become hazardous (Mora et al., 2013). Thus, the ToE of biological response is dependent on the species’ biology and biogeography as illustrated by the differences in ToE of habitat suitability between Skipjack tuna (larger home range with narrower temperature tolerance) and Atlantic cod (small range size with wider temperature tolerance) (Box 5.1, Figure 2). Generally, large proportion of the habitats of both species will experience ToE with the 21st
Species with fast generation times relative to the ToE of key habitat conditions (e.g., phytoplankton) may evolve more quickly to environmental change and be less vulnerable to climate change than longer-lived, slower generation time species (e.g., large sharks) (Jones and Cheung, 2018). However, evidence on evolutionary adaptation to expected climate change is limited, thus while shorter generation time may facilitate adaptation to environmental change, it does not necessarily result in successful adaptation of organisms’ habitats above which large biological impacts occur, e.g., warming threshold for coral bleaching (Pendleton et al., 2016) and temperature and oxygen threshold for fishes such as Atlantic cod and tunas (Deutsch et al., 2015; Box 5.1, Figure 2). These thresholds increase the biological sensitivity to ToE and shorten the time of organismal response relative to ToE of the respective ocean variables.

Box 5.1, Figure 2: Projected emergence of changes in environmental habitat suitability for two different fishes with widely different preferences and tolerance to ocean conditions and biogeography. (a, b) predicted present-day (average of 1950-2000) habitat suitability index of (a) skipjack tuna (*Katsuwonus pelamis*) and (b) Atlantic cod (*Gadus morhua*). (a, b) Time of emergence of changes in predicted habitat suitability index under RCP8.5 using the ocean variables in Box 5.1, Figure 1 and the method of Frölicher et al. (2016) for (c) skipjack tuna and (d) Atlantic cod (5.A.1).

Earlier ToE and their subsequent biological impacts on organisms and ecosystems increase the urgency of policy responses through both climate mitigation and adaptation (5.4.2, 5.4.3). However, the rapid emergence of stressors at the local scale in the near-term (already past or in this decade) such as warming and ocean acidification and the resulting impacts on some of the more sensitivity or less adaptive biodiversity and ecosystem services may post challenges for international and regional policies as their often require multiple decades to designate and implement (Box 5.5). In contrast, scope for adaptation for national and local ocean governance can be more responsive to rapid changes (5.5.2). This highlights the opportunities for multi-level adaptation that allows for reducing climate risks that are expected to emergence of stressors and impacts at different time frame (Mackenzie et al., 2014).

[END BOX 5.1 HERE]
5.2.3 Impacts on Pelagic Ecosystems

Marine pelagic ecosystems (from the surface ocean down to the sea floor) face increasing pressures due to the multiple environmental drivers from the changing climate (outlined in Section 5.2.2) and increasing intensity of human activities. Findings from AR5 highlight shared biological principles that drive responses of marine microbes, plants, invertebrates and fishes to single and multiple climate change drivers (AR5 WGII; Pörtner et al. (2017)). Particularly, temperature predictably affects the rate of biochemical reactions that determine organisms’ physiological performance, with an optimal range of temperature above and below which whole-organism biological functions performance decreases, impacting growth, reproduction and survivorship of the organism (Pörtner et al., 2014; Poloczanska et al., 2016).

Despite this emerging understanding, science-based conservation and restoration policies require a mechanistic understanding of ecosystem-level responses to multiple interacting climatic and human drivers. While such holistic approaches facilitate improved projections of the ecological impacts of climate change, they must deal with a number of emerging complexities, such as the differentiation between the long-term climate trends (e.g., progressive acidification and rising temperatures) and the short-term climatic variability (see Chapter 6) ranging from the seasonal to multi-annual scale (Henson et al., 2017).

As was discussed in AR5 WGII (Pörtner et al., 2014) and assessed in Section 5.2.2.5, short time series of two decades or less in length are too short for determining trends in the ocean (Henson et al., 2010; Chavez et al., 2011), and longer time-series are needed to establish the relative influences of climate change and variability and other human non-climatic influences (Edwards et al., 2013). To overcome this drawback, multiple lines of evidence – ranging from experiments to models and field observations – are combined for the detection and attribution of biological changes, future projections and risk assessments. This section will follow the subdivision of the pelagic system into the epipelagic ocean (<200 m, the uppermost part of the ocean that receives enough sunlight to allow photosynthesis), and the deep pelagic ocean, containing the ‘mesopelagic’ (200–1000 m depth) and ‘bathypelagic’ (>1000 m depth) zones.

5.2.3.1 The Epipelagic Ocean

Since the AR5, there have been a wide range of advances in understanding the effect of climate change on the surface, epipelagic ecosystem, across observations, experimental and modelling studies. The surface ocean is much more exposed to climatic and non-climatic influences from human than other open ocean ecosystems due to its close interaction with the lower atmosphere and the impact of human activities. The following assessments include the interactions between multiple climate and non-climate stressors, instead of the effects of individual drivers; the latter was solidly established in AR5. These updates from AR5 result in more comprehensive understanding of the future risks of the epipelagic organisms (spanning from microbes to marine mammals) and ecosystems, as well as their scope and limitation of adaptation, including shifts in distribution, phenology and biodiversity.

5.2.3.1.1 Detection and attribution of biological changes in the epipelagic ocean

Recent evidence from mechanistic understanding, physiological experiments and observations since AR5 continue to support the effects of changing temperature and oxygen level on pelagic marine ectotherms (organisms that are dependent on external sources of body heat). Changes in temperature beyond the thermal preferences of the organism affects body functions, impacting growth and reproduction. The dominant hypothesis explaining the physiological mechanisms remain the temperature-dependent oxygen capacity limitation (Pörtner et al., 2014; Pörtner et al., 2017; Pörtner et al., 2018). Although alternative mechanisms explaining temperature-sensitivity of some marine organisms may exist (Lefèvre, 2016), the conclusion that biological functions are impacted by warming beyond species’ thermal preferences and tolerances remain robust with high agreement of evidence. Anoxic conditions hinder marine life that obtain oxygen from the ambient waters, while hypoxic conditions reduce their metabolic scope for growth, and consequently limits the ability of animals to forage, avoid predation or fend off diseases (Kroeker et al., 2013). Since AR5, experiments continue to support that increasing CO₂ and decreasing pH in the ocean disrupts organism’s vital biological processes such as growth, reproduction and calcification through increased corrosion of calcium carbonate exoskeletons, elevated metabolic demand from disruption of the intra-cellular and inter-cellular acid-base balance, and neuro-sensory and behaviour such as foraging and predatory avoidance, with the precise effects varying between species and ecosystems (Sett et al., 2014; Cattano et al., 2018).
Organisms’ responses to multiple climate drivers -ocean warming, acidification, deoxygenation, and changes in nutrient or food supplies- have also been observed in experimental and field studies (Boyd and Bressac, 2016; Breitburg et al., 2018) (Figure 5.9). Sensitivity to ocean acidification varies across species and such variations may interact with the effects of warming and low oxygen directly and indirectly. Warming may exacerbate the effects of ocean acidification on the rate of photosynthesis in phytoplankton (Lefevre, 2016). However, the effects of the interactions between warming and ocean acidification on phytoplankton production varies between the phytoplankton’s climatic zones. In a recent meta-analysis of published experimental studies, primary production by temperate noncalcifying plankton increases with elevated temperature and CO₂, whereas tropical plankton decreases productivity because of acidification (Nagelkerken and Connell, 2015). Also, temperature increases consumption by and metabolic rates of herbivores but not on secondary production; the latter decreases with acidification in calcifying and noncalcifying species. This effect creates a mismatch with carnivores whose metabolic and foraging costs increase with temperature (Nagelkerken and Connell, 2015). Reported physiological responses to multiple climatic drivers include the decrease in aerobic scope for growth as demand for metabolic oxygen increases, neuroendocrine responses that affect the behaviour of the organisms, immune responses that affect the organisms’ defence against parasites and pathogens, disruption of ionic- and osmoregulation, and changes in reproductive biology (Whitney et al., 2013). reducing development of larvae of some sea urchins (echinoderms) (Ferrari Maud et al., 2014), or increasing predation rate of reef fishes (Nagelkerken and Munday, 2016) and behaviour in general (Nagelkerken and Munday, 2016), although interactions vary strongly between species and biological processes (Gobler and Baumann, 2016; Lefevre, 2016) (medium confidence). Hypoxia and acidification can also limit the thermal reaction norm of organisms and exacerbate their sensitivity to warming (Mackenzie et al., 2014; Pörtner et al., 2017). Limitation of nutrient and food availability and predation pressures can further increase the sensitivity of organimsal groups to climate change in specific ecosystems (Riebesell et al., 2017). Climate change also affect organisms indirectly through the impacts on competitiveness between organisms that favour those that are more adaptive to the changing environmental conditions (Alguero-Muniz et al., 2017) and trophic interactions (Seebacher et al., 2014).

Overall, since AR5, empirical observations from experiments have establish physiological mechanisms and detected significant responses of combined effects of warming, ocean acidification and low oxygen on phytoplanktons and pelagic marine ectotherms (high confidence). Sensitivity and responses vary between taxonomic groups and organism that are adapted to different climatic zones (high confidence). Species at the population level are shaped by organisms’ physiological sensitivity modified indirectly by trophic interactions (high confidence).

New evidence since AR5 continues to support that shifts in distribution range and phenology from phytoplankton, zooplankton to fishes are consistent with expectations from physiological understanding of the organisms’ temperature preferences and changes in environmental temperature and other oceanographic conditions (Pörtner et al., 2014; Britten et al., 2016; Poloczanska et al., 2016) (Figure 5.9). Specifically, spatial and temporal occurrences of marine biota follow changes in environmental conditions. Warming is related to expansion of leading edge of their geographic range and contraction of trailing edge in the past (Feary et al., 2014). Timing of biological events of marine biota are also shifting to earlier in the years under warming, which is consistent with the expectation from the close relationship between temperature and these biological events (Poloczanska et al., 2016).

Overall, the rate of historical geographic range shift in the last century, towards cooler higher latitude waters and deeper layers, is partly correlated with the rate of ocean warming, the slope of their habitat’s temperature gradient and species’ biological characteristics, but with group-specific differences (Pinsky et al., 2013; Jones and Cheung, 2015; Poloczanska et al., 2016). For example, analysis of the Continuous Plankton Recorder data-series from the North Atlantic show that the range of dinoflagellates and copepods tended to closely track the velocity of climate change (the rate of isotherm movement). In contrast, the distribution range of the diatoms shifted much more slowly (Chivers et al., 2017) and its distribution seems to be primarily influenced by multi-decadal variability rather than from the average temperature trend (Edwards et al., 2013; Harris et al., 2014).
In the North Atlantic, zooplankton are advancing northwards by 11.6 ± 2.9 days per decade faster than any other marine group (Richardson, 2008; Schlüter et al., 2010). Copepods, cladocerans, ctenophores, and larvae of decapods, echinoderms, molluscs and fish are all peaking earlier in the spring season (Mackas et al., 1998; Bertram et al., 2001; Edwards and Richardson, 2004; Greve et al., 2004; Costello et al., 2006; Burthe et al., 2012), although not all species are advancing (Philippart et al., 2003; Edwards and Richardson, 2004; Asch, 2015; Crespo et al., 2017), and phenology is probably less relevant in tropical systems. In the California Current, larvae of offshore, pelagic fish species are found earlier in the year in response to sea surface warming, whereas larvae of coastal, demersal species are found later (Behrenfeld et al., 2006).

**Figure 5.9:** Evidence of climate change responses of marine organisms to changes in ocean conditions under climate change. (a) Physiological evidence of responses to multiple drivers: [+] additive, [x] synergistic, [-] antagonistic; (b) different lines of evidence on changes in organism’s body size and (c, d) field observations on changes in latitudinal

<table>
<thead>
<tr>
<th>a. Physiological impacts</th>
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**Physiological level**

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<tr>
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<th>Field data</th>
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**Organism level**

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<thead>
<tr>
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<th>Fishes (N = 185)</th>
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<tbody>
<tr>
<td></td>
<td>Benthic invertebrates (N = 48)</td>
<td>Sea birds (N = 4)</td>
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**Population level**

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<tr>
<th>d. Change in phenology</th>
<th>Phytoplankton (N = 24)</th>
<th>Zooplankton (N = 115)</th>
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<tr>
<td></td>
<td>Fishes (N = 17)</td>
<td>Sea turtles, seabirds, mammals (N = 65)</td>
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**Figure 5.9:** Evidence of climate change responses of marine organisms to changes in ocean conditions under climate change. (a) Physiological evidence of responses to multiple drivers: [+] additive, [x] synergistic, [-] antagonistic; (b) different lines of evidence on changes in organism’s body size and (c, d) field observations on changes in latitudinal

**Figure 5.9:** Evidence of climate change responses of marine organisms to changes in ocean conditions under climate change. (a) Physiological evidence of responses to multiple drivers: [+] additive, [x] synergistic, [-] antagonistic; (b) different lines of evidence on changes in organism’s body size and (c, d) field observations on changes in latitudinal
range and phenology (based on Poloczanska et al., 2013). Positive values represent direction of range and phenology shifts that are consistent with expectation from climate change. T – temperature, OA – ocean acidification, O2 – ocean deoxygenation, and others – other variables mostly include salinity, nutrient and primary production.

Variations in the rate of shifts in distribution and phenology between different groups of organisms may relate to their biological and ecological characteristics. In general, marine ectotherms (those whose body temperature change with the temperature of the environment) are more sensitive to climate change than endothermal organisms (those that generate heat to maintain their body temperature). Species that evolved in fluctuating and variable environments (e.g., seasonally variable temperate environment) often become eurythermal (i.e., the ability to tolerate or adjust to a wide range of temperature) while those that evolved in more stable environments become stenothermal (i.e., living within a narrow temperature range) (Cheung and Pauly, 2016; Pörtner et al., 2017). Based on existing observations and ecological theories, larger body size, higher dispersal ability, availability of suitable habitats and food also contribute to the successfullness of range expansion (Sunday et al., 2011).

Overall, the biogeography and phenology of marine organisms in the epipelagic systems, with expansion in the poleward boundary (high confidence), earlier timing of biological events (high confidence), and overall shift in biomass and species composition (very high confidence). Such changes are consistent with expected responses to ocean warming based on theories and experimental evidence that scale from individual organisms’ physiological responses to population level effects under warming. Uncertainties exist in the detailed physiological mechanisms driving the observed macroecological changes. However, such uncertainties do not undermine the attribution of biological impacts to warming.

AR5 concluded that there had been limited field observations that robustly demonstrate biotic responses attributable to anthropogenic ocean acidification, partly because of the large natural variability (Pörtner et al. 2014). More recent evidence since AR5 continues to suggested mixed results on whether impacts of ocean acidification on calcifying zooplankton are already observable (Howes et al., 2015). Attempts were made to tease apart the relative roles of regional and multi-decadal variability in the effect of ocean acidification. (Rivero-Calle et al., 2015) reported, using Continuous Plankton Recorder (CPR) archives, that stocks of coccolithophores (a group of phytoplankton that forms calcium carbonate plates) over the last five decades have increased by 20%, and that this increase is linked (by statistical analysis) to higher CO2 concentrations. Their results agree with other recent observational study in the subtropical North Atlantic that attributes an increase from 1990 to 2014 in coccolithophore abundance to ocean acidification. However, the outcome of (Rivero-Calle et al., 2015) analysis depends on whether global (Mauna Loa archive) or regional (Takahashi archive) CO2 time-series are employed in the statistical analysis, with global CO2 and the Atlantic Multidecadal Oscillation (AMO), respectively, being the main drivers attributed to this floristic shift (medium confidence).

Overall, detectable changes in abundance of califying phytoplankton and zooplankton have observed in some regions of the epipelagic ocean (medium confidence), but increased ocean acidification may not have caused sufficient biological changes beyond natural variations (medium confidence).

The differences in rate of responses warming between biota and regions result in re-organization of pelagic communities, with increasing dominance of warmer water species (Stuart-Smith et al., 2015) and potential mismatch of biotic interactions (Thackeray et al., 2016; Pecl et al., 2017). Distribution shifts of copepods are accompanied by a decrease in their mean size, leading to a system with higher metabolism, more recycling and less export (Attrill et al., 2007; Poloczanska et al., 2013). The warming-stratification hypothesis suggests warming decreases of phytoplankton and zooplankton biomass in the tropical ocean because of nutrient limitation associated with enhanced stratification and a shallower mixed layer, and increases biomass in polar regions through greater water column stability and faster growth (Richardson and Schoeman, 2004; Boyce et al., 2010; Laufkötter et al., 2013). There is some support for this hypothesis. Although there were methodological challenges, a data synthesis from the tropical Atlantic showed a 10-fold drop in biomass from 1950 to 2000, related to reduced phosphate concentration and thinning of the thermocline. Similarly, a recent modelling study showed a 5% decline in zooplankton biomass in the past 50 years (Steinberg et al., 2012). However, the best available tropical data (from the Bermuda Atlantic Time Series) shows a 61% increase in zooplankton biomass associated with sea surface warming and increased stratification.
As endotherms, mammals and birds are more buffered to the effects of temperature changes than reptiles, which as ectotherms, rely on external heat sources to regulate their body temperature. For reptiles, including lizards, turtles and snakes, temperature directly effects important life history traits including hatching size, sex, viability and performance (high confidence) (Hays et al., 2003; Pike, 2014; Dudley et al., 2016; Santos et al., 2017). This is particularly important for marine turtles as changing temperatures will affect the hatching sex ratio because sex is determined by nest site temperature (high confidence) (Hatfield et al., 2012; Santidrián Tomillo et al., 2014; Patricio et al., 2017). Rising temperature will also lead to a loss of breeding substrate as the associated rise in sea level can reduce the available nesting or pupping habitat for land breeding marine turtles, lizards, seabirds and pinnipeds (very likely) (Fish et al., 2005; Fuentes et al., 2010; Funayama et al., 2013; Reece et al., 2013; Katselidis et al., 2014; Patino-Martinez et al., 2014; Pike et al., 2015; Reynolds et al., 2015; Marshall et al., 2017).

Changes in ocean temperature, and pH will also indirectly impact marine mammals, seabirds and reptiles by changing the abundance and distribution of their prey (high confidence) (Polovina, 2005; Polovina et al., 2011; Doney et al., 2012; Sydeman et al., 2015; Briscoe et al., 2017; Woodworth-Jefcoats et al., 2017). While indirect effects of changing climate are harder to document, examples are becoming apparent. The distributions of some of these large animals is determined by the occurrence and persistence of oceanic bridges and barriers that are related to climate driven processes (Ascani et al., 2016; McKeon et al., 2016). For example, the decline of Arctic sea-ice is effecting the range and migration patterns of some species and is allowing the exchange of species previously restricted to either the Pacific or Atlantic oceans (Virtually certain) (Alter et al., 2015; George et al., 2015; LaIдр et al., 2015; MacIntyre et al., 2015; McKeon et al., 2016; Breed et al., 2017; Hauser et al., 2017). It is hard to predict how these new invasions will change the existing food webs and species interactions. For example, as killer whales, Orcinus Orca, inhabit Arctic waters, narwhale’s, Monodon monoceros, are avoiding the use of key habitats that are already in decline to avoid predation (Medium confidence) (Bost et al., 2009; Sydeman et al., 2015) (see Chapter 3).

The habitat requirements of marine mammals, seabirds and sea turtles are associated with bathymetric and mesoscale features that facilitate the aggregation of their prey (Bost et al., 2015; Kavanaugh et al., 2015; Hindell et al., 2016; Hunt et al., 2016; Santora et al., 2017). The persistence and location of these features are linked to variations in climate (Crocker et al., 2006; Baez et al., 2011; Dugger et al., 2014; Abraham et al., 2017; Youngflesh et al., 2017) and to foraging success, juvenile recruitment, breeding phenology, growth rates and population stability (high confidence) (Costa et al., 2010; Ancona and Drummond, 2013; Ducklow et al., 2013; Chambers et al., 2014; Descamps et al., 2015; Abadi et al., 2017; Bjorndal et al., 2017; Fluhr et al., 2017; Youngflesh et al., 2017). These changes have been most pronounced in polar environments, where sea-ice is an important physical feature that is required for both breeding and hauling out by birds and mammals (Ducklow et al., 2006; Siniff et al., 2008; Jenouvrier et al., 2009; Costa et al., 2010; Murphy et al., 2013; Constable et al., 2014; Barbraud et al., 2015; George et al., 2015; MacIntyre et al., 2015; Amelineau et al., 2016; Renner et al., 2016; Ferguson et al., 2017; Fernandez-Arcaya et al., 2017; Hamilton et al., 2017; Hauser et al., 2017; Stempieniewicz et al., 2017) but also forms a critical link in the structure of polar marine habitats (Costa et al., 2010; Murphy et al., 2016) (see Chapter 3).

A consistent theme in both predicted as well as observed changes in megafauna attributed to climate change is that there will be winners and losers (Virtually certain) (Trivelpiece et al., 2011; Hazen et al., 2013; Clucas et al., 2014; Constable et al., 2014; George et al., 2015). For example, while some Adélie and Chinstrap Penguin populations have plummeted, Gentoo Penguin colonies have increased (Cimino et al., 2013; Ducklow et al., 2013; Korczak-Abshire et al., 2013; LaRue et al., 2013; Dunn et al., 2016; Youngflesh et al., 2017). Such changes are quite regional, Adélie Penguin populations at the southernmost limits of the range are increasing (Trivelpiece et al., 2011), while those at the northern limits are declining (Cimino et al., 2013; Ducklow et al., 2013; Trathan et al., 2015). Long term population changes associated with climate change have also been observed for temperate species of seabirds and shifts in distribution of marine mammals and seabirds (High confidence) (Henderson et al., 2014; Hiscock and Chilvers, 2014; Ramp et al., 2015).

Rockhopper, Eudyptes chrysolome, and Yellow Eyed Penguins, Megadyptes antipodes (Rey et al., 2014; Morrison et al., 2015; Mattern et al., 2017), have declined, while populations of Gentoo, Pygoscelis papua, and Magellanic Penguins, Spheniscus magellanicus, Imperial Cormorants, Phalacrocorax atriceps, Dolphin,
Larus scoresbii, and Kelp, L. dominicanus, gulls have increased, some significantly (Boersma and Rebstock, 2014). However, increases in Magellanic penguin populations occurred at higher latitudes colonies, whereas they declined at the lower latitude colonies (Weimerskirch et al., 2012). This pattern is consistent with a shifting habitat and species range associated with climate change (highly likely). Laysan, Phoebastria immutabilis, and Wandering, Diomedea exulans, albatross have responded positively to climate change as they have been able to take advantage of the increased intensity of winds, allowing them to forage farther and faster making their foraging trips shorter increasing their foraging efficiency and breeding success (Descamps et al., 2015; Thorne et al., 2016).

Climate change is also associated with an increase in the frequency and intensity of storms and other extreme weather events (high confidence) (Vargas et al., 2006; Boersma and Rebstock, 2014; Henderson et al., 2014; Shirasago-Germán et al., 2015; Descamps et al., 2017; Pardo et al., 2017) (Chapter 6), causing increased risk on population viability of marine mammals and seabirds (medium confidence). An increase in storms between 1983 and 2010 resulted in increased reproductive failure in Magellanic Penguins at Punta Tombo, Argentina (Cavole et al., 2016). An unusual ‘warm-water anomaly’ off the Northeast Pacific Ocean resulted in closures of commercially important fisheries and mass strandings of marine mammals and seabirds (Newell et al., 2015) and a reduction in Shag, P. aristotelis, and Kittiwake, Rissa tridactyla, breeding success in the North Sea (Nicoll et al., 2017). Depending on the age class of petrel and a regional effect, there could be both positive and negative effects of tropical cyclones on Round Island Petrels, Pterodroma arminjoniana (Riofrío-Lazo et al., 2017).

Some species like Galapagos penguins (Spheniscus mendiculus), sea lions (Zalophus wollebaeki), fur seals (Arctocephalus galapagoensis), and flightless cormorants (Phalacrocorax harrisi), have very small populations with restricted ranges and do not have alternative habitats to utilize (Vargas et al., 2005; Gallagher et al., 2015; MacLeod and Steinfartz, 2016). In general, specialist species are more vulnerable than generalists (see Figure 5.9). This can be in terms of diet or habitat requirements, or in terms of existing life history traits (Pörtner et al., 2014). The future for species that have limited ranges, limited habitat availability or small populations is quite precarious (medium confidence).

In conclusion there is a consistent theme with multiple lines of evidence that predicted and observed changes in megafauna will occur, and that there will be both winners and losers (high confidence, high agreement).

5.2.3.1.2 Future changes in the epipelagic ecosystem

The projected rate of change in the ocean (see Section 5.2.2) raises questions about how readily individuals, species, communities and foodweb can respond to multiple climate drivers (including CO2, oxygen, nutrients and temperature) and shifting patterns in the future ocean (Schaum et al., 2013; Boyd et al., 2016; O’Brien et al., 2016).

Planktonic microbes (bacteria, archaea, unicellular algae, and protozoans), with short turnover times (hours to days), would adapt faster than larger animals, due to their phenotypic plasticity (Schaum et al., 2013). Microbes are thus expected to adapt quickly to rapidly rising of CO2 or warming, partly compensating for the negative effects (Schaum and Collins, 2014). The response of microbes to multiple drivers is complex, however, since it may involve interactive effects (i.e., synergisms and antagonisms) between changing biologically influential properties (Breitberg et al., 2015; Hutchins and Boyd, 2016; O’Brien et al., 2016) (medium confidence). The majority of studies suggest that reduced pH will result in a shift in the microbial communities associated with invertebrate hosts, with direct impacts on host health and disease susceptibility, although the magnitude of effects will vary among species (Lee et al., 2015) (medium confidence). Another indirect CO2 effect on the bacterial and phytoplankton community could be through viral lysis, which is reported to respond positively to pCO2 (Brussard et al., 2013; Crawford et al., 2017) (low confidence).

A wide range of N2 fixation responses of diazotrophic cyanobacteria (ranging from no change to increases or decreases in N2 fixation rates) have been recently observed in laboratory experiments under temperature and CO2 conditions that are similar to RCP8.5 projected for the end of the 21st century. These variable responses are possibly due to differences in the mechanisms of N2 fixation between strains and species (Eichner et al., 2014; Gradoville et al., 2014) (medium confidence). Moreover, nitrogen fixation by Trichodesmium appears to be significantly influenced by interactions between elevated CO2 and iron and phosphorous co-limitation.
with global implications for ocean carbon and nitrogen cycling (Walworth et al., 2016; Schulz et al., 2017; Walworth et al., 2018) (medium confidence).

In situ experiments with natural plankton communities revealed some consistent patterns in phytoplankton responses to ocean acidification (summarised in Pörtner et al., 2014). Across a wide range of climate zones and ecosystem types, autotrophic standing stocks (chlorophyll a) were consistently higher at high pCO2, more pronounced in smaller-sized taxa, with variable responses depending on nutrient availability.

Haptophytes (coccolithophores and *Phaeocystis* sp.) are the only phytoplankton class known to respond negatively to ocean acidification (Hoogstraten et al., 2012; Bermúdez et al., 2016; Riebesell et al., 2017). Coccolithophores are vulnerable to projected end of century pH (Sett et al., 2014) (medium confidence). However, new evidence, based on long-term experiments of acclimation or adaptation to increasing temperatures in combination with elevated CO2, show that growth, calcification and carbon fixation rates of coccolithophores at high CO2 are modulated by temperature, are species-specific, and could increase calcification (Khanna et al., 2013). Calcification of planktonic foraminifera will be negatively affected by acidification (Roy et al., 2015) (medium to high confidence), and their populations are predicted to experience the greatest decrease in diversity and abundance in sub-polar and tropical areas, under RCP8.5 (Brussaard et al., 2013).

At the level of mesozooplankton, experimental studies simulating future scenarios of warming and acidification are contradictory. Some studies suggest that acidification and warming act synergistically to increase the impact on zooplankton (Garzke et al., 2016), whereas other studies suggest that acidification effects partially counteract some temperature impacts on abundance and body size (Mayor et al., 2015). Other evidence suggests that the impact of ocean acidification on non-calcifying zooplankton could be greater when predators are present and or when food is absent (Winder et al., 2017). In a recent mesocosm experiment to determine the variations in biological responses to climate change, larvaceans outcompeted copepods in warmer and lower pH scenarios (Wirtz, 2012). Larvaceans can efficiently shunt energy to higher trophic levels (Peijnenburg and Goetze, 2013; Heneghan et al., 2016) and also export considerable amounts of carbon. These experimental studies suggest that changes in food webs are likely to be complex and oversimplification of zooplankton processes and functional groups in the current, similarly-parameterised, biogeochemical models could lead to errors in project magnitude and direction of change in zooplankton biomass (medium confidence).

Moreover, experimental studies on the impact of realistic ocean acidification scenarios for 2100 show zooplankton responses are species-specific and can be both positive and negative. Many experimental studies have shown no ocean acidification impact on the diversity of mesozooplankton (Li et al., 2015) early life stages of copepods (Weydmann et al., 2012; McConville et al., 2013; Cripps et al., 2014; Alguero-Muniz et al., 2016; Bailey et al., 2016) or scyphomedusa ephyrae (Winans and Purcell, 2010; Schulz et al., 2013).

However, some studies have shown a negative impact on zooplankton, including shell degradation in pteropods (Alguero-Muniz et al., 2017), lower abundance of an anthomedusa jellyfish (Cripps et al., 2014), increased mortality of copepod nauplii (Smith et al., 2016a), and a reduction in demersal zooplankton (Alguero-Muniz et al., 2017). By contrast, some experimental studies have shown a benefit of ocean acidification on zooplankton, including a positive effect on overall abundance but with differences in direction of changes between species (Alguero-Muniz et al., 2017; Taucher et al., 2011). For example, in a mesocosm experiment with high pCO2 treatment, *Hybocodon prolifer* (Anthomedusa) reacted negatively by lower abundances, while *Aglantha digitale* (Trachymedusa) became more abundant (Havenhand, 2012). A wide range of studies, from laboratory experiments (Taucher et al., 2015), mesocosm enclosures (Isla et al., 2008), synthesis of observations (Rose and Caron, 2007) to modeling experiments (Bopp et al., 2013) provide insights into how the multi-faceted components of the ‘biological pump’ (the biologically mediated processes responsible for transporting carbon from the upper ocean to depth) are projected to be altered in the coming decades. A synthesis of the individual components reported to both influence the performance of the biological pump, and which are sensitive to changing ocean conditions, is presented in Table 5.1. They include those catalogued by Pörtner et al. (2014) along with additional factors - such as seawater viscosity and its influence on particle sinking rates (Taucher et al., 2014) and the joint influence of warming/acidification on bacterial solubilization of particles (Piontek et al., 2015) - from recent research. Table 5.1 lists the putative controlling environmental factor, such as warming, on each individual component that influences the biological pump, and the reported modification (where available) of each individual factor.
factor by changing ocean conditions for both the epipelagic ocean and the deep ocean. Overall, although models project a small decrease in productivity and export flux (with regional variability) (see section 5.2.2.4), different lines of evidence (including observation, modeling and experimental studies) provide low to medium confidence on how climatic drivers affect different components of the biological pump in the epipelagic ocean, as well as changes in the efficiency and magnitude of carbon export in the deep ocean (see section below and Table 5.1).

**Table 5.1:** Projected future changes to the ocean biological pump (adapted from Boyd, 2015). Environmental controls on individual factors that influence downward POC flux are based on published reports from: experiments (denoted by E), modelling simulations (M), and observations (O). In some cases, due to the paucity and regional specificity of published reports, it has been indicated the sign of the projected change on export (in italics), as opposed to magnitude. NPP: Net Primary Production; POC: Particulate Organic Carbon; DOC: Dissolved Organic Carbon; TEP: Transparent Exopolymer Particles; OA: Ocean Acidification. Climate change denotes multiple controls, such as nutrients, temperature and irradiance, as parameterised in coupled ocean atmosphere models. * denotes observed for low altitudes only. ? represents major uncertainty over environmental modulation of this component of the biological pump. * denotes joint influence of temperature and acidification.

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<tr>
<th>Pump component</th>
<th>Oceanic driver</th>
<th>Projected change (by year 2100)</th>
<th>Confidence</th>
<th>References &amp; Lines of evidence</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Epipelagic Ocean</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Phytoplankton growth</td>
<td>Temperature (warming)</td>
<td>~10% Faster (nutrient-replete) no change (nutrient-deplete)</td>
<td>High</td>
<td>Boyd et al. (2013) E; Marañón et al. (2014) O</td>
</tr>
<tr>
<td>Net Primary Production (NPP)</td>
<td>Climate change (temperature, nutrients, CO₂)</td>
<td>10-20% decrease (low latitudes), 10-20% increase (high latitudes)</td>
<td>Medium</td>
<td>Bopp et al. (2013) M</td>
</tr>
<tr>
<td>Food web retention of NPP</td>
<td>OA</td>
<td>Enhanced transfer of organic matter to higher trophic levels, reduced N and P sedimentation by 10%</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Floristic shifts</td>
<td>Climate change (Warming/OA/Infl. Salinity)</td>
<td>Shift to smaller or larger cells (less export vs more export. Inconclusive)</td>
<td>Low</td>
<td>Minas et al. (2010) O; Li et al. (2009) O; Seet et al. (2018) I</td>
</tr>
<tr>
<td>Differential susceptibility</td>
<td>Temperature (warming)</td>
<td>Growth rate of grazers more temperature dependent than prey (less export)</td>
<td>Low</td>
<td>Rose and Canin (2007) O</td>
</tr>
<tr>
<td>Bacterial hydrolytic effects</td>
<td>Warming/OA</td>
<td>Increase under warming and low pH (variable response in different plankton communities)</td>
<td>Low</td>
<td>Burrell et al. (2017) E</td>
</tr>
<tr>
<td>Grazier physiological responses</td>
<td>Warming</td>
<td>Copepods had faster respiration and ingestion rates, but higher mortality (inconclusive)</td>
<td>Low</td>
<td>Isla et al. (2006) O</td>
</tr>
<tr>
<td>Faunistic shifts</td>
<td>Temperature and subpolar zooplankton species shifts</td>
<td>Temperature (inconclusive)</td>
<td>Low</td>
<td>Edwards et al. (2013) O</td>
</tr>
<tr>
<td><strong>Deep Ocean</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Particle sinking rates (viscosity)</td>
<td>Warming</td>
<td>5% faster sinking/ degree C warming</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Mesozooplankton community composition</td>
<td>Temperature?</td>
<td>Shifts which increase/decrease particle transformations (less/more export, respectively) (more export)</td>
<td>Low</td>
<td>Jackson and Burd (2002) M; Ikeda et al. (2001) O</td>
</tr>
<tr>
<td>Vertical migrants</td>
<td>Climate change (irradiance, temperature)</td>
<td>More export (variable response in different plankton communities)</td>
<td>Low</td>
<td>Almén et al. (2014) O; Berge et al. (2014) O</td>
</tr>
</tbody>
</table>

Under both RCP4.5 and 8.5 there will be significant poleward migrations and phenological shifts in many groups of microorganisms (Baker-Austin et al., 2013), increasing infectious disease outbreaks with pathogens moving polewards (Böttjer et al., 2014; Burge et al., 2014) (medium confidence). Moreover, the poleward shift of zooplankton species is likely to continue in the future (Woodworth-Jefcoats et al., 2017). There is high agreement in model predictions that there will be reduced global zooplankton biomass in the future, consistent with the warming-stratification hypothesis. All 11 biogeochemical models in a recent study project a decline in zooplankton biomass in the future (Stock et al., 2014), similar to other studies that report global declines of 7.9% between 1951-2000 and 2051-2100 under RCP8.5 (Stock et al., 2014) and 11% between 1980 and 2000 and between 2080 and 2100 under SRES A1B (Stock et al., 2014). Some regions could experience >50% declines in biomass in these scenarios (Woodworth-Jefcoats et al., 2017). These same models show that zooplankton amplify the climate change signal that propagates up from phytoplankton, by 10-30% (Stock et al., 2014) to 100% (Lewandowska et al., 2014). Thus, mesocosm experiments and modelling studies also agree in supporting the warming-stratification hypothesis. Warming leads to reduced nutrient flux and lower phytoplankton and zooplankton biomass, and increased zooplankton grazing relative to phytoplankton growth (O'Connor et al., 2009; Alguero-Muniz et al., 2017) (medium confidence).
Changes in oxygen content and ocean acidification exacerbate temperature effects on fishes and invertebrates. Decrease in oxygen content under warming projected to reduce maximum body size (Deutsch et al., 2015; Pauly and Cheung, 2017) and contraction of suitable environmental conditions for fishes (Munday, 2014). Fishes exposed to ocean acidification level expected under RCP8.5 showed impairments of sensory ability and alteration of behaviour including olfaction, hearing, vision, homing and predator avoidance (Kroeker et al., 2013; Heuer and Grosell, 2014; Nagelkerken et al., 2015). A wide variety of biological processes, ranging from growth, survivorship, reproduction, and calcification are impacted by ocean acidification expected under RCP8.5, with crustaceans and calcifying molluscs being most sensitive to such effects (Kroeker et al., 2013; Alava et al., 2017). Early life stages are particularly sensitivity to ocean acidification impacts. Also, ocean acidification and warming have shown to have synergistic effects with warming that exacerbate the risk of population decline. Overall, there is high confidence that warming, decrease in oxygen and acidification will impact growth, reproduction and behaviour of fishes. Adaptation will not always suffice to mitigate the climate impacts, although there will be winners and losers (high confidence, high agreement). The risks of population decrease of marine fishes will be further exacerbated by overfishing and pollution (Munday et al., 2013) (high confidence).

Based on outputs from an ensemble of marine ecosystem and fisheries models, potential total consumer (mainly invertebrate and fish) biomass without fishing is projected to decrease by 4.8±3.5% and 17.2±10.7% under RCP2.6 and 8.5, respectively, by 2090-2099 relative to 1990-1999, while the decrease is around 4.5% by the mid-21st century across all RCPs (likely) (Figure 5.10). Accounting for the removal of biomass by fishing exacerbates the decrease in biomass for large-bodied animals which are particularly sensitive to fishing (likely for the direction of changes). Regionally, total consumer biomass decreases largely in tropical and mid-latitude oceans (likely) and increases in the Arctic ocean (Figure 5.10, as likely as not). The high uncertainty in the Arctic is because of model uncertainties as a result of the ways the model represent the effects of ocean variables (temperature and net primary production) (Lotze et al., in review) and the model outputs (exploited species only versus total biomass). In the Southern Ocean, the decrease in consumer biomass is mainly in the southern Indian Ocean while other parts of the Southern Ocean are projected to have an increase in consumer biomass by 2100 under RCP8.5, reflecting mainly the projected pattern of changes in net primary production from the Earth system models (see Section 5.2.2.5).

Figure 5.10: Projected changes in biomass of fishes and invertebrates (except zooplanktons) based on outputs from the Fisheries and Marine Ecosystem Impact Models Intercomparison Project (FISH-MIP). (a, b) Multi-model mean change (%) in total marine animal biomass in 2090-99 relative to 1990-99 without fishing. (c, d) Variability among different
ecosystem and Earth-system model combinations \( (n = 10) \) expressed as one inter-model standard deviation. \((e, f)\) Model agreement \( (\%) \) on the direction of change.

Projections of range shifts of fishes under changes in temperature and net primary production agree with expectations from theory and observations (Figure 5.9), with a medium rate of range shift of 15.5 km decade\(^{-1}\) and 25.6 km decade\(^{-1}\) between 2000 and 2059 under RCP2.6 and 8.5, respectively (Jones and Cheung, 2015). Range shifts are projected to result in decreases in species richness in tropical oceans, and increases in mid to high latitude regions \( (medium\ confidence) \) (Ben Rais Lasram et al., 2010; Cheung and Pauly, 2016; Molinos et al., 2016). In addition, geographic barriers such as land boundaries in the poleward species range edge in semi-enclosed seas or lower oxygen water in deeper waters are projected to limit range shifts, resulting in larger relative decrease in species richness \( (medium\ confidence) \) (Cheung et al., 2013; Burrows et al., 2014; Jones and Cheung, 2015).

Responses of pelagic ectotherms to warming, ocean acidification, deoxygenation and changing nutrient supplies may be moderated by physiological acclimatization (Munday, 2014), trans-generational and evolutionary adaptation (Clements and Hunt, 2015; Niinemets et al., 2017), and behavioural changes (Poloczanska et al., 2016). Experimental evidence demonstrate that less complex organisms are better in acclimatizing and adapting to warming and deoxygenation because of their wide aerobic scope and short generation time (Seebacher et al., 2014). In contrast, acclimatization to warming and deoxygenation is often limited by one or more physiological processes in higher organisms with complex organization (Poloczanska et al., 2016). However, more complex organisms are able to adjust for the changes in their internal acid-base balance under high CO\(_2\) in waters. Although scope for acclimatizing or adapting to climate change impacts exist in marine organisms, climate change impacts will not be fully eliminated given that species have been responding to changing ocean environment through shifting biogeography and phenology in recent decades (Poloczanska et al., 2016) (see Figure 5.9). For example, despite zooplankton having high evolutionary potential and presumably being well suited to adapt to climate change (Hinder et al., 2014), no thermal adaptation in a 50-year time series of two key copepod species in the North Atlantic was found (Lewis et al., 2013). Some work suggests that sensitivity to ocean acidification is dependent upon the degree of natural exposure, with species that vertically migrate and commonly encounter low pH conditions more able to adapt to lower pH conditions (Gunderson et al., 2016).

Overall, marine ectotherms have some capacity for physiological adjustment and evolutionary adaptation that lowers their sensitivity to warming and decrease in oxygen \( (low\ confidence) \) (Cavallo et al., 2015) (Figure 5.9). However, historical responses in abundance and ranges of marine fishes to ocean warming and decrease in oxygen in the past suggest that adaptation is not always sufficient to mitigate the projected impacts \( (medium\ confidence) \).

5.2.3.2 The Deep Pelagic Ocean

The pelagic realm of the deep ocean represents a key site for remineralisation of organic matter and long-term biological carbon storage and burial in the biosphere (Aristegui et al., 2009). The direct impacts of climate change are however not well understood for their organisms and ecosystems (Pörtner et al., 2014) (Table 5.1).

Global ocean warming between 0.5°C (RCP2.6) and 1.5°C (RCP8.5) is expected to reach the bathypelagic zone by the end of the century (Figure 5.5), with the largest warming of deep waters occurring first in the Southern Ocean (Ciais et al., 2013; Patara and Böning Claus, 2014); Roemmich et al. (2015). There is \( medium\ confidence \) that warming and changes to primary productivity in the upper ocean will reduce the export of organic matter to the deep sea (Section 5.2.2.4). Analyses of long-term trends in primary production and particle export production, as well as model simulations, reveal that increasing temperatures, leading to enhanced stratification, nutrient limitation and shifts towards small phytoplankton, will have the greatest influence on decreasing the flux of particulate organic carbon (POC) to the deep ocean (Bopp et al., 2013; Boyd et al., 2015; Fu et al., 2016; Laufkötter et al., 2016). Observational and model-based methods provide evidence that the transfer efficiency of organic carbon through the mesopelagic zone is partly controlled by temperature and oxygen \( (low\ confidence) \), affecting microbial metabolism and zooplankton community structure, with highest efficiencies for high-latitude and oxygen minimum zones (OMZs) (see
Section 5.2.2.4) for more detail on OMZs), while below 1000 m the highest transfer efficiencies occur in subtropical regions, and are controlled by particle sinking speed (Boyd et al., 2015; Marsay et al., 2015; DeVries and Weber, 2017). There is also low confidence on the effects of increasing temperatures on particulate organic carbon (POC) remineralisation to CO₂ versus POC solubilisation to dissolved organic carbon (DOC) by microbial communities and its storage as refractory DOC (i.e., with life times of >16,000 years) (Legendre et al., 2015). The global magnitude of the ‘active flux’ of organic carbon from the surface to the mesopelagic due to vertical migration of zooplankton and fishes is uncertain, but has been reported to account from 10 to 100% of the gravitational sinking flux (Davison et al., 2013; Hudson et al., 2014; Jónasdóttir et al., 2015) (low confidence).

Predictions based on model studies suggest that mesopelagic zooplankton and fish communities living at deep scattering layers (DSLs) will increase their biomass by 2100, enhancing their trophic efficiency, because of ocean warming and shallowing of DSL (Proud et al., 2017) (low confidence). Recent model projections agree with the earlier results from IPCC AR5 WGI Chapter 3, predicting an average 3.5% loss of total ocean oxygen by 2100 and expansion of OMZs (see Section 5.2.1.4). It will also widen the DSL and increase the exposure of mesopelagic organisms to shallower depths (Gilly et al., 2013; Netburn and Koslow, 2015). In the California Current, the abundance of mesopelagic fishes is closely tied to variations in the OMZ, whose dynamic is linked to the Pacific Decadal Oscillation (PDO) and El Niño-Southern Oscillation (ENSO) cycles (Koslow et al., 2015). Hypoxia-adapted animals, like the Humboldt squid, will benefit from expanding OMZs (Stewart et al., 2014), but many non-adapted fish and invertebrates (like diurnal vertical migrants) will have their depth distributions squeezed, affecting the carbon transport and trophic efficiency of food webs in the mesopelagic (Stramma et al., 2011; Brown and Thatje, 2014; Rogers, 2015) (high confidence). In OMZ waters, where zooplankton is almost absent, like in the Eastern Tropical North Pacific, the microbial remineralisation efficiency of sinking particles would be reduced, eventually increasing the transfer efficiency of organic matter to the deep ocean and thus biological carbon storage (Cavan et al., 2017) (medium confidence). However, increases in ocean temperature may also lead to shallower remineralisation of POC in warm tropical regions, counteracting the storage of carbon in the dark ocean (Marsay et al., 2015).

5.2.4 Impacts on Deep Sea Floor Systems

5.2.4.1 Defining the Deep

The deep sea is assessed here as the vast area of the ocean >200 m deep, beyond most continental shelves (Figure 5.11). Below 200 m changes in light, food supply, and the physical environment lead to altered animal taxonomic composition, morphologies, lifestyles, and body sizes collectively understood to represent the deep sea (Tyler, 2003). Although the AR5 WG2 Chapter 30 (Hoegh-Guldberg et al., 2014) defined the deep sea as below 1000 m, the absence of photosynthetically useful light and ensuing critical ecological changes, biogeochemical transformations, and altered human interactions that occur on much of the sea floor below 200 m have led both pelagic and benthic biologists to include the ocean waters and seafloor below 200 m within the definition of the deep sea (Herring and Dixon, 1998; Gage, 2003).

5.2.4.2 Detection and Attribution of Oceanographic and Biological Changes in the Deep Sea

The deep sea (below 200 m) is vast and remote, difficult to access and expensive to study and as a result hosts relatively few long biological time series or experiments suitable for assessment of climate impacts (Smith et al., 2013). However, there are long-term records that document physical change derived from deep observing programs (see Section 5.2.1.2.1) including from repeat hydrography to > 6000 m (Talley et al., 2016b) and from autonomous float instruments to 2000 m (Roemmich et al., 2015). The paleo record holds information about hydrological, biogeochemical and biological change on historical or geological time scales and contributes understanding valuable for interpreting modern responses to climate change. To approach the ecosystem changes for the deep sea, this section relies on (a) observed and projected changes in environmental exposures to climate drivers relative to natural variation (climate hazard) (Section 5.2.2.2) (b) definition of habitat requirements and tolerance thresholds from current species distributions and habitat suitability modeling, (c) mechanistic understanding derived from experiments with shallow water relatives or with eurybathic deep-sea species reared in the laboratory (Section 5.2.4.1), (d) observed ecological, physiological or trophic responses to drivers based on long-term, time series data or spatial variation across.
natural gradients, (e) ecological insights from fossils in the paleo record, and (f) meta analyses of driver impacts and their interactions. Taken together they provide substantial evidence for future changes to most deep-sea ecosystems.

Figure 5.11: A conceptual diagram illustrating how climate drivers operating as inputs from land, atmosphere, and the crust alter upwelling, surface production, circulation, and fluid flows with the potential to modify deep-sea ecosystems.

5.2.4.3 Changes on the Deep-sea Floor

For all international waters and those subject to management by regional fisheries management organizations (RFMOs), an analysis was conducted to determine a) projected exposure to change in temperature, pH, O2, and POC flux, b) exposure to climate change hazard estimated as the change between future and historical projections divided by the standard deviation of historical projection c) time of emergence (Box 5.1) of seafloor climate change -- the year when accumulated standard deviation of future projections (since 2006) exceeds historical standard deviation (during 1951-2000), d) cumulative impact, the sum of exposure to climate change hazard. For each of the seven RFMOs, these metrics were evaluated at RCP8.5 for two projections (2041-2060 and 2081-2100) for > 200m, > 200-2500 m, bottom fishing areas, vulnerable marine ecosystems (VME) closure areas, seamounts, cold-water corals, and canyons. Areas < 2500 m were targeted as this is where fishing occurs. The general changes in these metrics are discussed in Sec 5.2.2.

Clear projected trends within RFMO jurisdiction suggest that exposures of deep-sea habitats to changing environments are greatest for temperature change in the Mediterranean (GFCM), to pH and dissolved oxygen decline in the North Atlantic (NAFO and NEAFC), and to declines in POC flux (food supply) in the northwest Atlantic (NAFO). Signal emergence is soonest (before 2040) for pH in most habitats in the N. Atlantic, but also for canyons in the Antarctic (CCAMLR) and the South Pacific (SPRFMO). Mean temperature signals also emerge before 2040 in canyons of the Antarctic (CCAMLR), northwest Atlantic (NAFO) and South Pacific (SPRFMO). In contrast POC Flux signal emerges earliest (before 2040) in the North Pacific canyons and slopes (NPFC), and dissolved oxygen emerges earliest (before 2040) for coldwater corals in the Antarctic (CCAMLR), canyons and slopes in the northwest Atlantic (NAFO), and
slopes of the southeast Atlantic (SEAFO). Cumulative impacts are greatest in the Northwest Atlantic,
followed closely by the northeast Atlantic, Antarctic, and southeast Atlantic with slightly lesser cumulative
impacts in the Mediterranean and lowest impacts in the South and North Pacific (Levin et al., 2018).
Uncertainty in these estimates of exposures and signal emergence are introduced by the coarse resolution of
CMIP5 models at the seafloor and by the extrapolation of surface POC production to bottom fluxes using the
Martin Curve.

5.2.4.4 Biological Responses to Stressors

Temperature is understood to influence the biogeography of the deep ocean at large scales (Watling et al.,
2013), and to the extent that it reflects energy, can have a unimodal influence on biodiversity and biomass in
the deep sea (Watling et al., 2013; Yasuhara and Danovaro, 2016) (medium confidence). The underlying
mechanisms involve oxygen-and capacity-limited thermal tolerance (Pörtner, 2012) (Section 5.2.4.1.1).
Pressure tolerance may also be limited by availability of oxygen (Brown and Thatje, 2014), thus changes
brought about by ocean deoxygenation could limit the ability of animals to seek cooler waters in response to
warming by moving deeper (low confidence). Bottom temperature has important implications for estimates
of seasonal and spatial benthic flux variation, benthic–pelagic coupling, and impacts of predicted ocean
warming at high latitudes (Belley and Snelgrove, 2016).

Deoxygenation in the present and past derives largely from warming in the open ocean (Keeling et al., 2010;
Levin, 2018). For example, in the Santa Barbara Basin sediments, O$_2$ changes are associated with Dansgaard
Oeschger events (rapid climate fluctuations) and abrupt deglacial warming (Cannariato and Kennett, 1999;
Moffitt et al., 2015). The warming and oxygen loss coincide with shifts from diverse invertebrates and
multiple feeding guilds to extremophile invertebrates and eventually to extremophile, accompanied by
increased reliance on chemosynthesis (Moffitt et al., 2015). Oxygen loss affects deep seafloor taxonomic
composition, taxon-specific abundances, biomass, body size, diversity, physiology, reproduction and
behavior (Levin, 2003; Breitberg et al., 2015) (high confidence).

Ocean acidification, the availability of carbonate ions, and the aragonite and calcite saturation horizons
(boundaries above which aragonite and calcite carbonate ions are saturated and below which they are
undersaturated, making calcification more energetically costly) exert strong control on the calcification rates
of organisms, the energy required and the dissolution rates of living and dead carbonate skeletons, tests and
sediments which form habitat in the deep sea (medium confidence). While ocean acidification is recognized
to influence a suite of behavioural and physiological responses in shallow-water taxa, including growth rate,
metabolic rates, mode of catabolism, development, gene expression, energetics and more, most of these have
not been studied in deep-sea species. A meta analysis reported in AR5 Chapter 6 (Pörtner et al., 2014), Table
6-3 identifies 10 studies involving 6 species of cold-water corals that suggest low vulnerability to CO$_2$
changes at RCP6.0 and medium vulnerability at RCP8.5, with negative effects starting at pCO$_2$ of 445 µatm.
Nevertheless, oxygen and capacity dependent thermal tolerance is likely to integrate a broad range of
physiological responses to multiple climate drivers: warming-induced CO$_2$ accumulation in body fluids links
to the effects of ocean acidification mediated by the weak acid distribution of CO$_2$, while temperature-
induced hypoxemia links to the hypoxia sensitivity of thermal tolerance (Pörtner, 2012).

Decline in food supply (POC flux) to the deep sea is caused by warming and subsequent stratification of the
ocean, which combined with slowing of thermohaline circulation, reduces the nutrient supply to the surface
ocean; declining surface productivity subsequently causes declines in export of primary production to the
depth sea (Section 5.2.2.5) (high agreement, limited evidence). Reductions in POC flux are predicted at low
and mid latitudes but increases are possible at high latitudes, associated with reduction in sea ice cover (Yool
et al., 2013; Rogers, 2015; Sweetman et al., 2017; Yool et al., 2017; FAO, 2018).

The Coupled Model Intercomparison Project (CMIP5) used in AR5 found that, under a high greenhouse gas
emissions scenario (RCP8.5), declines in primary production will lead to a global fall in export production
from −7% to −16% (Bopp et al., 2013) (Section 5.2.2.5). When linked to ocean acidification, which shoals
the calcite compensation depth and decreases production of biogenic calcium carbonate, the export
production declines can be up to −40.7% at 1000 m (Yool et al., 2013) or up to 38% at the northeast Atlantic
Seafloor (Jones et al., 2014). Those environments already subject to food stress (e.g., abyssal plains beneath
oligotrophic gyres) are likely to be most affected (Smith et al., 2008; Jones et al., 2014) (high likelihood,
Changes in POC flux are likely to alter the relative importance of the microbial loop and benthic invertebrates in carbon transfer (Dunlop et al., 2016) (limited evidence). Deep-sea biota demonstrate increased efficiency (effective use of food energy for growth and metabolism with minimal loss) at low food inputs (due to small size and dominance by prokaryotic taxa), thus they should be better able to adapt than shallow biota to reduced POC flux with climate change (Gambi et al., 2017) (low confidence, based on E Mediterranean Sea ultra-oligotrophic sediments). Changes in the overlying mesopelagic and bathypelagic communities (see Section 5.2.2.2) will affect food flux to the deep sea, as nekton and zooplankton transfer energy through diel vertical migrations, ontogenetic migrations and falls of dead carcasses (Gage, 2003) (medium likelihood, limited evidence). Abyssal responses to changes in POC flux are discussed in Section 5.2.4.4.

5.2.4.4.1 Abyssal plains

There is a strong positive relationship between surface production and seafloor faunal biomass in the Pacific Ocean (Smith et al., 2013) and the Gulf of Mexico (Wei et al., 2011). Thermal stratification associated with rising temperatures will reduce surface production and export flux to the seafloor (Steinacher et al., 2010).

The rapid abyssal response to changes at the surface (Smith et al., 2013) suggest declines in seafloor biomass by the 2090s are highly likely, with projected global losses of 5.2% (Jones et al., 2014) to 17.6% (Yool et al., 2017). Depths below 2000 m will experience the greatest declines in biomass from the 1990s to 2090s (−28.9% at 2000 m, −32% at 5000 m under RCP8.5) with much lesser declines under RCP2.6 (−5.8%, −7.0% respectively) (Yool et al., 2017). By 2091-2100 under RCP8.5, benthic biomass may decline up to 38% in the abyssal NE Atlantic (Jones et al., 2014) (medium confidence). These estimates do not account for changes in the type or quality of the sinking material, to which macrofaunal and meiofaunal invertebrates are highly sensitive (Smith et al., 2008; Smith et al., 2009; Tittensor et al., 2011). Nor do they account for direct faunal responses to changes in temperature, oxygen or the carbonate system, all of which will influence benthic response (medium confidence). Lower food supply to the abyss is expected to result in a size-shift towards smaller organisms (Jones et al., 2014), resulting in rising respiration rates, lower biomass production efficiency, and lesser energy transfer to higher trophic levels (Brown et al., 2004), with changes being largest for macrofauna and lesser and similar for megafauna and meiofauna (Jones et al., 2014) (limited evidence, low confidence). A strong positive relationship between annual POC flux and abyssal sediment community oxygen consumption (with POC flux accounting for 63% of seafloor carbon demand during a 27-year time series at 4000 m) (Smith et al., 2016b), combined with projected changes in biomass suggest that abyssal communities are highly likely to experience structural and functional changes that affect the carbon cycle in this century. The findings speak to the importance of monitoring POC flux to the deep sea across many regions.

5.2.4.4.2 Continental slope sediments.

Seafloor assemblages on continental margins experience strong vertical temperature, oxygen and pH gradients generating sharp ecological zonation (Levin and Sibuet, 2012). Both the extant gradients and past fluctuations in exposures make margins a valuable natural laboratory for understanding ecosystem response to multiple climate drivers (Sperling et al., 2016). Most global climate projections are too coarse (grid cell too large) to resolve changes in gradients of climate drivers on margins, however long-term regional observations (e.g., the California Cooperative Oceanic Fisheries Investigations, or CALCOFI) and modeled projections have highlighted increasing vulnerability of continental slope ecosystem to ocean acidification (Gruber et al., 2012) and ocean deoxygenation (Goericke et al., 2015) as upwelling intensifies in eastern boundary upwelling areas (Sydeman et al., 2014) (medium confidence).

Large oxygen declines are linked to warming events in the past on margins, over multiple time scales (Dickson et al., 2012; Moffitt et al., 2015). Over the past 50 years, deoxygenation is recorded on continental margins of the northeast Pacific and other oxygen minimum zone margins (Stramma et al., 2008; Levin, 2018) and more loss is projected (Bopp et al., 2013; Cocco et al., 2013) (see Section 5.2.2.4). Ecosystem effects of oxygen loss on margins will include loss of fish and invertebrate biodiversity (Levin, 2003; Gallo and Levin, 2016; Sperling et al., 2016) and a simplification of trophic structure reducing energy flows to upper trophic levels (Sperling et al., 2013) and shifting in carbon processing pathways from metazoans to
protozoans (Woulf et al., 2009) *(high likelihood, medium confidence)*. A loss of deep bioturbating faunas at  
lowest oxygen levels is *virtually certain*, and but because they contribute to both carbon burial and  
remineralization (via oxygenation) there is less certainty about the net effect on carbon burial rates, although  
it is *likely* to be positive (Smith et al., 2000; Levin and Dayton, 2009). Hypoxia reduces colonization  
potential and community resilience of slope biota *(low confidence, one study)* (Levin and Sibuet, 2012).  
Carbon sequestration and nitrogen recycling are highly sensitive to small changes in oxygenation (Deutsch  
et al., 2011) *(medium confidence)*.

Benthic foraminifera are more tolerant to changing climate drivers on margins than larger invertebrates.  
Short-term experiments have showed no significant effect of ocean acidification on survival for *Ammonia*  
sp. (Dissard et al., 2010; Haynert et al., 2011; Keul et al., 2013), *Bulimina marginata* and *Bolivia argentea*  
(McIntyre-Wressnig et al., 2014) and *Globobuliminina turgida* (Wit et al., 2016). However, lower pH  
exacerbates shallow foraminiferal sensitivity to warming (Webster et al., 2016) and shallow water (reef)  
foraminifera microbiomes are sensitive to rising temperature (Webster et al., 2016). Additionally, the  
sensitivity of calcification by planktic foraminifera such as *G. bulloides* to ocean acidification can reduce the  
rate to the seabed and contribute to declining POC flux (Davis et al., 2017). Benthic foraminifera also  
exhibit well-recognized changes in abundance and diversity in response to oxygen loss (Bernhard and  
Reimers, 1991; Goody et al., 2000; Moffitt et al., 2014). Hypoxia in combination with elevated pCO₂ favors  
survival of some foraminifera (*Globobuliminina*) (Wit et al., 2016). Combined declines in pH and oxygen may  
lead to increase in some agglutinating taxa and a decrease in the carbonate-producing foraminifera, including  
those using carbonate cement (van Dijk et al., 2017) *(one study, low confidence)*.

Among the metazoan meiofauna, nematodes are sensitive to temperature change; an anomalous temperature  
drop of only 0.4°C in the Mediterranean deep sea caused a significant decrease in nematode abundance and  
functional diversity, concomitant with increased species richness and evenness (Danovaro et al., 2001;  
Danovaro et al., 2004). Nematode abundance and biomass can also decrease with increasing sediment  
temperatures (Yodnarsri et al., 2008). Ice shelf collapse transforms nematode assemblages from  
depauperate, low-diversity communities, to richer and denser communities dominated by opportunistic  
species (Ingels et al., 2012). Elevated CO₂ affects nematode survival (Barry et al., 2004; Fleeger et al., 2006;  
Fleeger et al., 2010) and although tolerant to hypoxia as indicated by high abundances, their diversity  
declines significantly within low-oxygen sediments (Neira et al., 2001). Meiofaunal biomass is sensitive to  
POC flux (Smith et al., 2008). Reduced POC flux with depth causes Mediterranean nematodes to decline in  
body size, conferring efficiency (biomass:respiration) in exploitation of food resources under very low food  
conditions (Gambi et al., 2017). There is *low agreement* about meiofaunal responses among studies,  
sometimes reflecting opposing responses in different regions. However, there is agreement that meiofauna  
are sensitive to change in environment and food supply *(medium confidence)*.

Warming-induced changes in circulation are linked to abrupt changes in benthic ostracod diversity in the  
historical record (Yasuahara et al., 2008) *(medium confidence)*. Gamete synthesis, skeletal structure and  
biological composition of deep-sea echinoderms are sensitive to ocean acidification (deep-sea holothurians  
- Verkaik et al. (2016)) and to ocean deoxygenation (echinoids - Sato et al. (2017)). Shoaling oxyclines can  
cause changes in echinoid vertical distributions (Sato et al., 2017) *(one study, low confidence)*. However  
species in upwelling regions, such as the echinoid *Strongylocentrotus fragilis*, that have adapted to oxygen  
minimum zones with characteristically high CO₂ levels are less vulnerable to the negative impacts of ocean  
acidification (Taylor et al., 2014). Where both oxygen and CO₂ stress occur together, oxygen can be the  
primary driver of change (Taylor et al., 2014; Sato et al., 2017), but this is not always true (Gobler and  
Baumann, 2016).

There is a strong macroecological relationship between depth, export POC flux, biomass and zonation of  
macrobenthos on continental slopes (Wei et al., 2011), such that loss of POC flux will alter seafloor  
community biomass and structure *(high confidence)*. This is modified on the local scale by near-bottom  
currents, which alter sediment grain size, food availability, and larval dispersal (Wei et al., 2011).

### 5.2.4.4.3 Chemosynthetic ecosystems

With nutrition derived largely from chemosynthetic sources fueled by fluids from the earth’s interior,  
hydrothermal vents and seeps are often assumed to be largely insulated from effects of climate change.  
However, there are multiple linkages to broader surface ocean processes that ultimately influence the
productivity, diversity, and health of these systems (medium confidence). Examples include requirements for oxygen (to serve as electron acceptor for aerobic hydrogen-, sulfide- and methane oxidation), links to surface productivity and mesoscale eddies, for growth and transport of larvae (Arellano and Young, 2009; Adams et al., 2011), and for nutrition of mixotrophic species (relying on photosynthesis and chemosynthesis), as well as seasonal/ ENSO influences on these features.

Vent species such as bathymodiolin mussels, alvinocarid shrimp, and some limpets have larvae that develop in surface waters (Herring and Dixon, 1998; Arellano et al., 2014). These will be exposed to warming waters, decreasing pH and carbonate saturation states, and in some places, reduced phytoplankton availability, causing reduced calcification and growth rates (as in shallow water mussel larvae, Frieder et al. (2014)) (medium confidence). Larvae originating at vents or seeps beneath upwelling regions may also be impaired by effects of hypoxia associated with expanding oxygen minimum zones (Stramma et al., 2008) during migration to the surface (low confidence).

Methane seep systems occur almost exclusively on continental margins where they will experience changes in exposure to climate drivers (Sweetman et al., 2017; FAO, 2018) (high confidence). Rising bottom temperatures or shifting of warm currents on margins could increase dissociation of buried gas hydrates on margins (Phrampus and Hornbach, 2012) (low confidence). If this occurs there would be intensified anaerobic methane oxidation (which produces hydrogen sulfide) (Boetius and Wenzhoefer, 2013), and expansion of methane seep communities (medium confidence). Such an expansion would increase seafloor productivity (high confidence) and either locally decrease or regionally increase biodiversity (Cordes et al., 2010) (medium confidence). At the same time, climate-driven expansion of midwater oxygen minima (Stramma et al., 2008; Schmidtko et al., 2017) occurring at depths where many seep ecosystems form (200-1000 m) (Levin, 2005) could reduce the cover of large vent and seep foundation species (mussels, siboglinid tube worms), which are unable to grow under dysoxic conditions (< 5-10 μmol kg⁻¹) (Sweetman et al., 2017) (medium confidence).

At some vents, high CO₂ conditions limit mussel shell calcification, reduce shell weight and slow growth (in Bathymodiolus septendrionis) at low saturation states (Tunnicliffe et al., 2009; Rossi and Tunnicliffe, 2017). However, the absence of predators reduces the need for calcification and CO₂ stress can be relieved by high food supply (through symbiotic autotrophy including hydrogen-based metabolism) (low confidence).

Climate-driven changes can alter the biodiversity associated with mussel bed habitat as well as the food web in which they reside (low confidence). Declines in POC flux to the deep seabed in most areas except the Arctic (see Section 5.2.1.5) would negatively affect vent and seep mussels and other mixotrophs that have the ability to use photosynthetically derived food raining down from the surface to supplement chemosynthetic sources. Reduction in amount and changes in timing of surface production affects vent and seep mussels (e.g., Bathymodiolus azoricus and Bathymodiolus childressi) which show synchronized gametogenesis (sperm and egg production) with spawning occurring loosely around the time phytodetritus from surface production arrives at the seafloor (Dixon et al., 2006; Tyler et al., 2007). These climate-driven changes to seep and vent ecosystems could alter food available for surrounding deep-sea ecosystems which benefit through transport of nutrients and microbes, movement of vagrant predators and scavengers, and plankton interactions (Levin et al., 2016) (limited evidence; low confidence). Warming and its effects on climate cycles have the potential to alter patterns of connectivity through changes in circulation (Fox et al., 2016) (limited evidence; low confidence).

5.2.4.4 Canyons

Canyons are abundant widespread topographic features that deeply incise continental and island margins globally (Harris et al., 2014). As regions of focused particulate organic matter and coarse material transport from the shelf (including terrigenous plant material), intensified mixing and advection of water masses, submarine canyons connect deep-sea and shallow ecosystems and enhance sediment transfer and deposition to the abyssal plain (Levin and Sibuet, 2012). As a result of local upwelling of nutrients to the surface waters induced by canyon topography, these areas exhibit high productivities with enhanced plankton, fish, mammal and invertebrate abundances (De Leo et al., 2010; Levin and Sibuet, 2012). At the seabed and in deep waters, canyons can sustain exceptionally high levels of biomass sometimes including high densities of deep-sea corals, gorgonians and sponges (De Leo et al., 2010; Fernandez-Arcaya et al., 2017). These functionally important habitats are vulnerable to climate-driven changes in physical and biogeochemical
processes that affect food supply. Enhanced stratification and change in the intensity and frequency of
downwelling processes under atmospheric forcing (including storms and density-driven cascading events
(Allen and Durrieu de Madron, 2009) will decrease organic matter transported through canyons (low
confidence). Changes in the quantity and quality of transferred particulate organic matter, as well as physical
disturbance during extreme events cause a complex combination of positive and negative impacts at different
depths along the canyon floor (Canals et al., 2006; Pusceddu et al., 2010). This will be superimposed on
regional stratification, inducing declines in POC flux predicted to affect 85% of the 8637 mapped canyons
and could lead to declines in benthic biomass in the next century (Jones et al., 2014) (medium confidence).
Expansion of oxygen minimum zones beneath upwelling regions at concentrations < 0.7 ml/l would reduce
fish abundance or diversity in central Pacific oceanic canyons (De Leo et al., 2012) (low confidence, one
study) as occurs on continental margins at lower oxygen levels (Gallo and Levin, 2016) (high confidence).
Canyon walls are home to extensive deep-water reef habitat formed of cold water corals, octocorals, sponges
and bryozoans (Williams et al., 2010), and are now recognized to host many methane seeps and other
chemosynthetic habitats supported by massive transport of terrestrial organic matter (Pruski et al., 2017).
Climate effects on these ecosystems are discussed separately in Sections 5.2.3.6, and 5.2.3.9.

5.2.4.4.5 Seamounts
There are 100,000-200,000 seamounts (underwater volcanoes, many of which are inactive) that protrude
above the seafloor. Seamount flanks are exposed to strong gradients in temperature and sometimes oxygen,
and often exhibit sharp faunal zonation reflecting fairly narrow habitat requirements. They can support high
animal densities and biomass, exhibit high secondary production supported by locally enhanced primary
production and intensified flows, function as diversity hotspots and serve as stepping stones for larval
dispersal (Rowden et al., 2010), but are also vulnerable to fishing disturbance, especially from bottom
trawling (Clark and Tittensor, 2010). The majority of seamounts (82% of 33,452 globally mapped
seamounts) are predicted to experience reduced POC flux under RCP8.5 in 2100, resulting in declines in
benthic biomass (Jones et al., 2014). Habitat suitability for reproduction (e.g., skate egg-laying sites) on
seamounts may be disrupted by warming, given apparently narrow ranges reflecting environmental
requirements (Henry et al., 2016). Seamounts have been proposed to serve as refugia for coldwater corals
facing shoaling aragonite saturation horizons (Tittensor et al., 2011), but could become too warm for deep-
water corals in some regions (e.g., projections off Australia) (Thresher et al., 2015). In the North Atlantic
seamounts are an important source of cold-water coral larvae maintaining resilience under shifting NAO
conditions (Fox et al., 2016), so loss of habitat suitability may have far-reaching consequences (Gehlen et
al., 2014b).

5.2.4.4.6 Biogenic structures: cold-water coral ecosystems.
In the years since IPCC AR5, understanding on the vulnerability of cold-water corals to global change has
increased substantially, with new initial findings including information on the impacts of ocean acidification
on skeletal integrity (Hennige et al., 2015) and the implications of altered regional oceanographic regimes
(Fox et al., 2016). While important details in our understanding of cold-water coral physiological response to
ocean acidification are coming into place, great uncertainty remains in our understanding of the adaptive
capacity of cold-water corals or how the connectivity of present-day populations will be affected by altered
regional current patterns (Roberts et al., 2016). However, this awareness is now feeding forward to shape
thinking in policy development, including Area-Based Management Tool approaches (Johnson et al., 2018).

Cold-water coral reef ecosystems consist of a (a) 3-dimensional, living complex of organisms (solitary or
colonial) dominated by one or more species of habitat-forming (foundation) scleractinian coral, (b) transition
zone with carbonate skeletons of dead individuals, and (c) rubble zone, sometimes growing on a carbonate
matrix many hundreds of meters thick (Buhl-Mortensen et al., 2010). Calcium carbonate concentrations and
aragonite saturation influence the distribution of scleractinians and octocorals and the habitats formed by
their structures (Guinotte et al., 2006; Tittensor et al., 2009; Thresher et al., 2015) (medium confidence).
Observed (Byrne et al., 2010) and predicted future acidification and shoaling of the aragonite saturation
horizon, especially in the North Atlantic will alter species habitat suitability and thus distributions (Gehlen et
al., 2014a) (medium confidence).

*Lophelia pertusa* is a widely distributed, habitat-forming, foundation species, and is probably the best
studied with respect to climate tolerances. In many instances *L. pertusa* can continue to calcify at aragonite
undersaturation (Georgian et al., 2016; Kurman et al., 2017) (medium confidence) and can acclimate to rising
CO₂ levels (Hennige et al., 2015) (low confidence). It can also calcify when exposed to multiple environmental stresses in the laboratory (Hennige et al., 2015; Büscher et al., 2017), however acidification exposure over 12 months will cause the biomineralized aragonite skeleton to exhibit less organization, a longer and thinner growth form, and reduced structural integrity of exposed skeletal framework, that forms the base of cold-water coral reefs and a significant proportion of the habitat, (low confidence) (Hennige et al., 2015). Weakening of skeletons by acidification enhances bioerosion of carbonates (e.g., by bacteria, fungi, annelids and sponges) (Schönberg et al., 2017), and bioerosion may exacerbate dissolution; dissolution and bioerosion will cause loss of cold-water coral habitat (virtually certain).

The importance of skeletal dissolution and bioerosion under ocean acidification has largely been overlooked in the discussion on how cold-water coral ecosystems will fare under future climate change. It is of equal importance to the growth of live cold-water corals, as it is the habitat provision of the reef as a whole, made up of live and dead coral framework, that is important for biodiversity support. Importantly, given that no adaptation can happen with regard to dissolution, as it is a biogeochemical response, it is highly likely that increased dissolution of exposed aragonite, further enhanced by bioerosion, rather than a reduction in calcification rates of the live coral, could lead to future net cold-water coral reef loss (high confidence). This is based on current occurrence of 95% of cold water coral reefs above the aragonite saturation horizon (Guinotte et al., 2006), loss of structural integrity of exposed coral framework of dead corals to projected aragonite undersaturation, and understanding of biogeochemical processes of aragonite dissolution in aragonite undersaturated water (Eyre et al., 2014).

Experiments on cold-water coral growth over a range of time scales have now demonstrated positive (but variable) calcification rates under future scenarios of ocean acidification and warming (Maier et al., 2013a; Maier et al., 2013b; Hennige et al., 2015; Rodolfo-Metalpa et al., 2015; Gori et al., 2016; Maier et al., 2016; Büscher et al., 2017) (medium confidence). However, net calcification rates (calcification+dissolution) of L. pertusa exposed to aragonite-undersaturated conditions or close to saturation (Ω< 1) often decrease to close to zero or even become negative (Lunden et al., 2014; Hennige et al., 2015; Büscher et al., 2017). There is genetic variability in L. pertusa’s ability to calcify at low aragonite saturation, and some populations (e.g., in the Gulf of Mexico) may more readily adapt to ocean acidification (Kurman et al., 2017) (single study low confidence). However, hypoxia (1.5 ml L⁻¹) can cause mortality in Gulf of Mexico L. pertusa populations, which already live under low-oxygen, high CO₂ conditions (Lunden et al., 2014) (single study, low confidence).

The occurrence of black corals (Antipathes fiordensis) in the low salinity, shallow waters of New Zealand fjords (< 35 m) indicates that some non-reef forming species can tolerate low pH conditions (Jiang et al., 2015). The solitary coral Desmophyllum dianthus, a species closely related to L. pertusa, also is able to live and thrive under natural low pH conditions (down to 7.4) in fjords of Northern Patagonia, reflecting potential adapt to ocean acidification (Fillinger and Richter, 2013). However, exposure to elevated pCO₂ when combined with elevated temperatures reduces calcification and respiration rates in D. dianthus, and shifts metabolic substrates from a mixture of protein and carbohydrate/lipids to a less efficient protein-dominated catabolism (Gori et al. 2016) (limited evidence, single study). Populations of this species waxed and waned over the last 40,000 years as the global ocean was influenced by glacial advances and retreats and changing concentrations of CO₂, O₂, and surface productivity (Thiagarajan et al., 2013). Coral populations on seamounts in the N. Atlantic and SW Pacific Oceans altered their depth distribution in response to changes in the aragonite saturation depth, which has shoaled since the last glacial maximum (Thiagarajan et al., 2013); future increases in atmospheric CO₂ are expected to further decrease suitable habitat for deep-dwelling hard coral species and their associated benthic organisms (Thiagarajan et al., 2013) (low confidence).

Scleractinian corals have the capacity to acclimate to high CO₂ conditions (Form and Riebesell, 2011; Rodolfo-Metalpa et al., 2015; Gori et al., 2016) due to their capacity to upregulate the pH at the calcification site. However, the synergy of climate stressors enhances their vulnerability by affecting a combination of physiological functions (medium confidence; number of studies limited with respect to spatial variability in stressor combination and species/population genetic variability). An increase of temperature from 13°C to 15°C was shown to affect both the capacity of Desmophyllum dianthus to acquire energy and to calcify at 750 ppm CO₂. Although resistant to low pH scenario conditions, the foundation species Lophelia pertusa does not tolerate warming above 14°C and oxygen concentrations below 1.6 ml l⁻¹ in the Gulf of Mexico.
oxygen minimum zone (3.3 ml l⁻¹ for North Atlantic specimens), highlighting critical thresholds for coral populations living at the edge of their tolerance (e.g., periphery of hypoxic zones, warm Mediterranean deep-sea). The role of temporal dynamics in the combination of stressors is recognized but is still poorly studied under in situ conditions (one study supports species-specific response to seasonal influence in the growth of L. pertusa and M. occulata) (Lartaud et al., 2014).

There is a decrease in the northeast Atlantic suitable habitat for key coral species (Desmophyllum spp. and Lophelia pertusa) predicted by habitat suitability modelling under projected changes in T, O₂, pH and POC flux for 2081-2100, but an increase for 2041-2060. Habitat suitability analyses suggest that food supply (POC flux) is a critical variable for the corals Desmophyllum spp., Lophelia pertusa, Madrepora oculata, Acanella arbuscula, and Paragorgia arboarea, while dissolved oxygen is an important variable in the northwest Atlantic area for Desmophyllum spp., Lophelia pertusa, Madrepora oculata (Morato et al., 2006; Levin, 2018). In the Southern hemisphere, future projections to 2099 of warming and acidification predict extreme loss of habitat for cold water coral (Solenosmilia variabilis) on seamounts off Australia and New Zealand under RCP4.5 and nearly complete loss under RCP8.5, both warming above a tolerance threshold of 7°C and decline of aragonite saturation (Ωarag < 0.84) below that required for survival are responsible (Thresher et al., 2015).

Future mean North Atlantic Oscillation (NAO) state can affect cold-water coral habitat connectivity in the deep North Atlantic (medium confidence), based on mechanistic understanding of the variation in wind-driven, Ekman-layer surface currents in the northeast Atlantic with NAO state (Fox et al., 2016). Modeled variability of connectivity of protected cold-water coral (Lophelia pertusa) populations in the northeast Atlantic over 40 years indicates positive NAO generates overall higher connectivity but lower east-to-west connectivity while lower NAO is associated with decreased connectivity, more isolated populations, and thereby reduced resilience (medium confidence). This study was informed by recent laboratory studies of L. pertusa larval behavior, planktonic durations (8-9 weeks or more) and vertical positioning in the water column (rising from depth to the surface, drifting, then descending to the bed) (Larsson et al., 2014).

Uncertainties lie in future NAO trends (AR5 reported medium confidence in an increase in mean NAO by 2050, especially in autumn and winter), and how laboratory results transfer to the real ocean (e.g., do the larvae reach the surface layer?).

5.2.4.4.7 Biogenic structures: sponges and other taxa.

Habitat-forming, deep-sea sponges (Geodia barrette) originally thought to experience mortality under a 4°C temperature rise (Guihen et al., 2012) were found through experimentation to tolerate 5°C elevation along with their microbiome, albeit with a temporary rise in respiration and nitrogen efflux and cellular stress (Strand et al., 2017). Ocean acidification (pH 7.5) reduces the feeding of deep-sea demosponges (Radiella sp., Polymastia sp.), indicating sensitivity of deep-sea sponges under natural and anthropogenic disturbance (Limited evidence, low confidence) (Robertson et al., 2017). Ocean acidification has adverse effects of acidification on gamete production, embryo morphology and performance as well as structure of ossicles and lipid content of muscles, gonads and oocytes of a widespread, abundant cold-water holothurian (Cucumaria frondosa) (Verkaik et al., 2016) (single study, limited evidence, low confidence). Nevertheless, the geologic record and modern distributions suggest that sponges are more tolerant to warming, acidification (and potentially deoxygenation) than are corals and may be less vulnerable to changes in temperature and pH (Schulz et al., 2013). Giant agglutinated protozoans (xenophyophores), common on the deep sea floor in many settings, are local hotspots of deep-sea biodiversity and are recognized as vulnerable marine ecosystem indicators by several regional fisheries management organizations (RFMOs) (Levin et al., 2018). Habitat suitability analyses reveal thresholds associated with depth, temperature, nitrate (reflecting productivity), oxygen and carbonate chemistry (Ashford et al., 2014) suggesting xenophyophores are potentially vulnerable to changes in these conditions at the seafloor.

5.2.4.5 Gaps

Understanding of climate change impacts on deep-sea ecosystems is challenged by the absence of long-term observations in most systems, and limited numbers of laboratory or in situ studies. Most such studies examine on single species and on single climate drivers. Studies on the effects of ocean acidification in the deep ocean are restricted primarily to calcifying organisms (i.e., cold-water corals) while similar work on non-calcifying ecosystem engineers (i.e., sponges, xenophyophores, sea pens) and the ecosystems that they...
form (e.g., sponge grounds), is almost absent. Although the 300 my geological record contains examples of simultaneous change in temperature, pH and oxygen, paleo research highlights the difficulty of identifying individual drivers from presence/absence of certain fossils in the absence of geochemical data, as well as the uncertainty introduced by short time scales <10,000 years. Modern multiple driver studies are based largely on climate projections combined with habitat suitability assessment for foundation species and key commercial species, but additional groundtruthing is required, ideally through sustained deep-ocean observing programs that incorporate biology across a broader diversity of ecosystems. Baseline assessments are needed to reveal the environmental status of deep-sea ecosystems, with development of spatial and temporal scales to be assessed and abiotic and biotic indicators to be used for measuring response to changing climate drivers (Danovaro et al., 2017; Sweetman et al., 2017).

Some components of the ecosystem are fundamental to deep-sea ecosystem functioning but are severely understudied in the deep sea, including viral and protist communities and their links to higher forms. Viruses are key drivers of microbial metabolism (Dell’Anno et al., 2015; Danovaro et al., 2016). Labile C released by viral cell lysis and virus decomposition stimulates microbial heterotrophs which are coupled to ammonium-dependent archaeal chemosynthesis and nitrogen regeneration (Danovaro et al., 2016). Viral dynamics are key to understanding microbial mediated CO₂ production and consumption and carbon storage capacity of the deep ocean, and cumulative impacts could cause regime shifts that alter life support services provided largely by microbes (Carbon sequestration and Nitrogen regeneration).

5.2.4.6 Summary for the Deep Pelagic Ocean

Despite its seeming remoteness, most of the deep-sea floor and its varied ecosystems already have or are projected to experience rising temperatures and declining oxygen, pH and POC flux beyond natural variability within the next half century (See Section 5.2.1). Much of the abyssal seafloor is expected to experience declines in food supply (Section 5.2.1.5) that will diminish benthic biomass, change community structure and rates of carbon burial (medium confidence). Expansion of oxygen-limited and oxygen-minimum zones on upwelling margins will reduce fish and invertebrate biodiversity, simplify food webs, reduce ecosystem resilience and alter elemental cycling (medium confidence). Even chemosynthetic ecosystems like hydrothermal vents and seeps, once thought to be isolated from surface phenomena, are recognized to be vulnerable to changing ocean conditions through effects on their pelagic larvae (e.g., shrimp and mussels), partial reliance on surface derived particulate organic matter (e.g., mussels), microbial symbionts that require oxygen to fix carbon (e.g., tubeworms, mussels), and reliance on mesoscale circulation features for connectivity (low confidence). Biogenic habitats that form biodiversity hotspots (e.g., corals, sponges, xenophyophores), and the topographic features that support them (seamounts, canyons, slopes) will be exposed to a combination of stressors (warming, reduced food supply, increasing acidity and oxygen loss) (medium confidence). Some of the most cumulative exposures to warming, declining pH, oxygen and food supply, will be in the Arctic and Atlantic Oceans over the next 100 years (medium confidence). Ecological responses are expected to vary regionally, but as critical thresholds of temperature, oxygen and CO₂ are exceeded, changing species depth distributions, dissolution, and bioerosion, exacerbated by lower food supply, will cause loss of biogenic habitat critical to the maintenance of biodiversity and ecosystem health (medium confidence).

5.2.5 Risk Assessment of Open Ocean Ecosystems

Overall, the upper ocean (0–700 m) and 700-2000m layers have both warmed from 2004 to 2016 (virtually certain) and the abyssal ocean continues to warm in the Southern Hemisphere (high confidence). The ocean is stratifying; observed warming and high latitude freshening are both surface intensified trends making the surface ocean lighter at a faster rate than deeper in the ocean (high confidence). It is very likely that stratification in the upper few hundred meters of the ocean will increase significantly in the 21st century. It is virtually certain that ocean pH is declining by ~0.02 pH units per decade where time-series observations exist. The anthropogenic pH signal has already emerged over the entire surface ocean (high confidence) and emission scenarios are the most important control of surface ocean pH relative to internal variability for most of the 21st century at both global and local scale (virtually certain). The oxygen content of the global ocean has declined by about 2%, largely in responses to changes in ocean circulation and respiratory demand (high confidence). Over the next century oxygen declines of 3.5% by 2100 are predicted by CMIP5 models globally (medium confidence), with low confidence at regional scales, especially in the tropics. The largest
changes in the deep sea will occur after 2100 (Section 5.2.2.3). CMIP5 models project a small decrease in
global organic matter production (medium confidence) with increases in high latitude (low confidence) and
decreases in low latitude (medium confidence) in response to changes ocean nutrient supply.

Emergence of novel ocean conditions for organisms from plankton to mammals are driving changes in eco-
physiology, biogeography and ecology and biodiversity (high confidence). Observed and projected
population declines in the equator-ward range boundary (medium confidence), expansion in the poleward
boundary (high confidence), earlier timing of biological events (high confidence), and overall shift in
biomass and species composition (very high confidence) are consistent with expected responses to climate
change. It is likely that increased OA has not yet caused sufficient reduction in fitness to decrease
abundances of calcifying phytoplankton and zooplankton, but is very likely (high confidence) that calcifying
planktonic organisms will experience great decreases in abundance and diversity under high emission
scenarios by the end of the century. Changes in biogeography and community structure are projected to
continue in the 21st century (high confidence), with potential animal biomass projected to decrease (likely).
Open ocean ecosystem functions are at risk of climate change impacts, with elevated risk of changes in
different components of the biological pump in the epipelagic ocean, as well as changes in the efficiency of
ocean carbon export in the deep ocean. However, there is low confidence on the direction and magnitude of these
changes. Biogenic habitats that form biodiversity hotspots in the deep sea floor and the topographic features
that support them (seamounts, canyons, slopes) will be exposed to a combination of stressors (warming, reduced food supply, increasing acidity and oxygen loss) (medium confidence). The strong positive
relationship between annual Particulate Organic Carbon (POC) flux and abyssal sediment community
oxygen consumption combined with projected changes in biomass suggests that benthic communities in
abyssal and deep-sea habitats will experience structural and functional changes that affect the carbon cycle
in this century under all emission scenarios (medium confidence).

There may be some capacity for adjustment and evolutionary adaptation that lowers their sensitivity to
warming and decrease in oxygen (low confidence). However, historical responses in abundance and ranges
of marine fishes to ocean warming and decrease in oxygen in the past suggest that adaptation is not always
sufficient to mitigate the observed impacts (medium confidence). In conclusion there is a consistent theme
with multiple lines of evidence that predicted and observed changes in open ocean ecosystems will occur,
and that there will be both winners and losers (high confidence, high agreement).

5.3 Changing Coastal Seas and Biodiversity

5.3.1 Introduction to Changing Coastal Seas

The world’s coastal and shelf seas (hereafter called the coastal seas collectively) extend from the coastline
towards the 200 m depth (Section 5.3.2) and encompasses diverse coastal ecosystems, including estuaries,
sandy beaches, kelp forest, mangroves and coral reefs (Section 5.3.3). Although they occupy a small part of
the World Ocean (7.6%), coastal seas provide up to 30% of global marine primary production and about
50% of the organic carbon supplied to the deep open ocean (Chen, 2003; Bauer et al., 2013) (Section
5.2.2.5). In addition, these systems harbor 90% of the world’s fish catch and their overall economic value is
estimated to be at least 40% of the value of the world's ecosystem services and natural capital (Costanza et
al., 2014) (Section 5.4.1.3). These marine environments include several frontal and upwelling areas (Box
5.2) that support high plankton productivity and fisheries yields (Scales et al., 2014), while coastal biogenic
structures e.g., mangrove forest, provide physical protection against extreme events such as storms and
floods (Kelleway et al., 2017a). Regional characteristics and habitat heterogeneity of some seas allow the
development of endemic fauna and flora, which makes them particularly vulnerable to climate change
impacts with high risk of diversity loss and alterations in ecosystem structure and functioning (Rilov, 2016;
Chefaoui et al., 2018). Features such as bathymetry and mesoscale processes provide refuge and facilitate the
aggregation of prey and safeguard the feeding and breeding of macrofauna, including marine mammals,
seabirds and reptiles (Section 5.3.3.9).

Unlike the open ocean where detection and attribution of climate driven-physical and chemical changes are
robust (Section 5.2.2), coastal habitats and shelf seas display regional hydrological complexity driven by the
interactions between the land, the sea and seabed structures and substrates, and are exposed to direct human
impacts that can render the conclusive detection and attribution of climate effects difficult (Levin et al., 2015). The high density of human populations along coastal areas in the World implies that most of these ecosystems are exposed to non-climatic disturbances such as nutrient loads (eutrophication), coastline modifications, pollution and overfishing. Climate threats interact with such non-climatic disturbances and pose serious risk to ecosystems goods and services. For example, cumulative exposure to eutrophication exacerbates the vulnerability of coastal ecosystems to warming and deoxygenation, increasing the likelihood of harmful algal blooms (HABs) (Box 5.3). The expansion of human infrastructure limits the adaptation of coastal ecosystems to sea level rise, by preventing the landward migration of coastal forms and reducing the available sites for nesting, feeding and breeding. Projections of the ecological impacts of climate change must thereby deal with a number of emerging complexities such as the differentiation between the long-term climate trends (e.g., progressive acidification) and the short-term natural fluctuations (Boyd et al., 2018), ranging from the seasons to climate oscillations like El Niño. The so-called ‘time of emergence’ for specific drivers beyond background variability is distinct in different ecosystems and is strongly sensitive to projected mitigation scenarios (Henson et al., 2017) (Box 5.1).

Despite the multi-scale environmental and anthropogenic stressors that affect the land-sea continuum, global experimental and long-term observational evidence since AR5 (AR5 WG2 Chapters 5, 6 and 30), allow the attribution of current changes in coastal ecosystems to climate trends. Moreover, the emergent ecosystem responses detected at present day strengthen the projections of their impacts under future emission scenarios (Section 5.3.4). This section assesses the exposure of global coastal systems to multiple climate hazards: warming, acidification, deoxygenation and sea level rise, by addressing the physical and biogeochemical changes in coastal seas (Section 5.3.2), and the sensitivity and vulnerability of each coastal ecosystem, considering geomorphological and biological components (Section 5.3.3). The assessment highlights common ecosystem responses to climate threats: coastal submergence and erosion, reduction in habitat area and complexity, poleward migration of species, biodiversity loss and reduction in carbon sequestration; and compares the adaptive capacity in the context of ecosystem resilience and the applicability of mitigation actions.

### 5.3.2 Oceanographic Drivers

There are several physico-chemical drivers that are either specific to, or more pronounced in, coastal ecosystems. These include:

- Changes in freshwater delivery from rivers and groundwater will directly affect coastal water salinity and suspended sediment loads (Yang et al., 2015; Dunn et al., 2018; Zahid et al., 2018). Such changes may result from increased or decreased precipitation in river catchments, changes in glacial meltwater, or anthropogenically-mediated hydrological changes (e.g., due to river dams for hydropower and/or water extraction) (Chang et al., 2015; Mukhopadhyay and Khan, 2015; Arnell and Gosling, 2016).
- Changes in coastal salinity will subsequently alter local/regional circulation patterns, through stratification, tidal interactions and other mixing processes, and hence both planktonic and benthic organisms (Little et al., 2017).
- Changes in river nutrient delivery (e.g., via storm run-off, as well as changes in agricultural practices and river flows) will also affect primary production, and hence other higher trophic levels (Maavara et al., 2017; Sharples et al., 2017; Chen et al., 2018). River nutrient loading may also affect the local trends, seasonality and the short-term variability of coastal ocean acidification (Duarte et al., 2013; Zhang and Gao, 2016; Laurent et al., 2017).
- Observed rates of warming in coastal seas are generally higher than in the open ocean (Holt et al., 2012; Wu et al., 2017), and that trend is projected to continue in both temperate and polar waters (Hellmer et al., 2017; Bonsell and Dunton, 2018). Other meteorological changes also have more pronounced effects in shallow seas than deeper waters; e.g., greater physical disturbance caused by an increasing likelihood of storm surge events (Rahmstorf, 2017; Trenberth et al., 2018).
- The biological effects of sea level change are primarily experienced by coastal wetlands or intertidal ecosystems (Field et al., 2017), also by benthic organisms, e.g., warm-water corals, in very shallow seas (Perry et al., 2018; Webster et al., 2018).
- Seasonal deoxygenation in coastal seas is likely to become more pronounced under warmer conditions (Laurent et al., 2018); it will also be indirectly affected by changes in primary production (and nutrient loads, both riverine and atmospheric) (Rabalais et al., 2014; Fennel and Testa, 2018; Irby et al., 2018).
• Physico-chemical changes occurring at the seafloor can strongly affect all of water column in coastal waters and shelf seas; e.g., the rate of nutrient re-cycling (Godbold et al., 2017; Snelgrove et al., 2017). Such changes may be climatic (driven by temperature or changes in water circulation and mixing) or anthropogenic (seabed disturbance by trawling, dredging, aggregate extraction, coastal development or the construction of offshore structures) (Cloern et al., 2016; Sciberras et al., 2016).

Coastal waters will also be affected by changes in the open ocean. Such influences include changes in local, regional and global circulation patterns (altering shelf-edge currents and upwelling, and hence nutrient supplies). The relative importance of open ocean and land-based drivers will depend on the width of the continental shelf, that varies from tens to hundreds of kilometres; the size of rivers and drainage basins; and the spatial variability of anthropogenic pressures.

5.3.3 Coastal Ecosystems

Coastal ecosystems are amongst the most diverse due to their complex geomorphology and sharp physicochemical gradients, and are highly sensitive to changes in atmospheric conditions. They are also exposed to other non-climatic human drivers and their potential interactions with climate change because of their close proximity to human populations (Levin et al., 2015; Diop and Scheren, 2016). All coastal ecosystems either classified by their geomorphological structure (i.e., estuaries, sandy beaches, rocky reefs, fjords) or foundation species (i.e., saltmarshes, mangroves, seagrass meadows, kelp forest, coral reefs) are threatened by sea level rise, warming, acidification, deoxygenation and extreme weather events with substantial differences in the type and level of vulnerabilities (Section 5.3.4) (Gattuso et al., 2015). While some ecosystems such as estuaries (Section 5.3.3.1), sandy beaches (Section 5.3.3.5), mangroves (Section 5.3.3.3) and saltmarshes (Section 5.3.3.2) are more vulnerable to sea level rise and erosive processes, others like rocky reefs (Section 5.3.3.7), kelp forests (Section 5.3.3.8), coral and other biogenic reefs (Section 5.3.3.6), are mostly threatened by warming and acidification, and are highly vulnerable to increased storms and heat-weaves. In particular, saltmarshes, mangroves and coral reefs have a disproportional ecological value due to their role in the protection against erosion and the reduction of exposure and sensitivity of biological communities to climatic stressors such as warming (Camp et al., 2016a).

At the species level, the main biological responses to the interactive effects of climate drivers are similar to those in pelagic ecosystems, including shifts in species distribution towards the poles, deeper water or specific physicochemical gradients that result in species range expansion and local extinction (Section 5.2.3.1.1). Unique biological responses in coastal areas include the potential for inshore and inland migration of benthic fauna and littoral vegetation in response to sea level rise. Consequently, the restructuring of coastal ecosystems affects their functioning and services such as carbon storage, productivity and storm protection (Section 5.4.1.3). Furthermore, the growing anthropogenic habitat degradation exacerbates climate vulnerabilities of coastal systems. Common examples of these compounding effects is the expansion of hypoxic zones enhanced by eutrophication in estuaries (Warwick et al., 2018) and low adaptability to sea level rise driven by coastal squeezing due to human barriers in sandy beaches and saltmarshes (Hubbard et al., 2014).

This sub-section summarizes our new understanding of ecological and functional changes that coastal ecosystems are experiencing related to multiple climate and non-climatic human drivers, and their synergies. Our assessment combines global evidence of biotic and abiotic responses in the short- and long-term from observations, experimental studies and numerical modelling. This approach allows current ecosystem alterations to be attributed to dominant climate drivers and facilitates the projection of future changes across latitudinal climatic regimes.

5.3.3.1 Estuaries

Estuarine ecosystems are defined by the river-sea interface that provides high habitat heterogeneity and supports large biodiversity across freshwater and subtidal ecotones (see Glossary); while marshes and mudflats with algal and vascular plant producers are key features of their riverine areas. Estuaries are sources of CO₂ to the atmosphere and emit an average of 0.25 ± 0.25 PgC yr⁻¹ (Regnier et al., 2013).

However, estuarine carbon budgets are also driven by carbon cycling within the coastal vegetated wetlands;
and either saltmarshes (Section 5.3.3.2), mangroves (Section 5.3.3.3) and subtidal seagrasses (Section 5.3.3.4), are important areas of carbon burial (Section 5.5.1.1). Carbon sequestration is threatened by modifications on the drainage area due to e.g., urban settlements, land reclamation and dredging, which has resulted in a 25-50% loss of total estuarine carbon intertidal pools over the past century (Regnier et al., 2013). Accordingly, rising sea levels represent one of the most important threats to estuaries, between 20-60% of coastal wetlands are expected to be lost globally by 2100, depending on emission scenarios (Watson and Hinojosa Corona, 2017), whereas gaining area will be possible if vertical sediment accretion occurs together with lateral re-accommodation (Schuerch et al., 2018). In turn, combined effects of anthropogenic activities and accelerated sea level rise increase the erosion of saltmarshes and the export of carbon and nitrogen into the estuarine waters, causing eutrophication (Duarte et al., 2014).

More observational evidence since AR5 WGII (Wong et al., 2014) supports that climate change modifies the estuarine gradients of salinity, turbidity and nutrients through interactive effects of sea level, precipitation, temperature and estuarine circulation patterns (Robins et al., 2016; Raimonet and Cloern, 2017). The interaction of sea level rise and changes in rainfall, and thereby river flows, will have a more severe impact on shallow estuaries (<10 m) than on deep basin estuaries (>10 m). In a projected sea level rise of 1 m, indices of vulnerability indicate that shallow estuaries will increase the tidal current amplitude by 5%, and the energy dissipation, the vertical mixing, and the salinity intrusion by 25% (Prandle and Lane, 2015). Meso- and macrotidal estuaries with rich sediment areas are more resilient to global climate changes than microtidal estuaries with low sediment supply, which are more vulnerable to sea level rise and changes in river flow (medium confidence) (Warwick et al., 2018). In the AR5 WG1 assessment it was concluded that observed changes in rainfall and the earth water cycle was likely since 1960 and attributable to human influence and an intensification of heavy precipitation (medium confidence) (Stocker et al., 2013). In the Amazon River basin, an important region for the regulation of global climate and hydrological system, more than half of the increasing trend of wet season precipitation over the last 35 years is attributed to warming in the tropical Atlantic (Xin-Yue et al., 2018). Then, the response of estuaries to river floods will vary regionally depending on the infiltration capacity of soils and the mechanisms that generate the floods (rainfall, snow melting and soil saturation), as well as on the balance between rainfall and evaporation rate (Arnell and Gosling, 2016).

The cycling between periods of drought and flood over land in combination with warming will affect the physical-chemical water gradients of ocean properties along the land-sea continuum and thus the distribution and performance of benthic biota. For example, anomalous salinity values exceeding the local threshold of 25 PSS (Practical Salinity Scale) after a dry period in Apalachiocola Bay, interacts with high water temperature and produces an increase of oyster mortality of 15% under warmer conditions (Petes et al., 2012). Conversely, droughts lead to salinization, and this condition can be intensified by climate warming due to evaporation in coastal shallow waters, and by sea level rise that can produce an inward movement of saltwater from the adjacent ocean (Ross et al., 2015; Zhou et al., 2017) (medium confidence). In two macrotidal estuaries in southeast England, climate-driven increases in saline incursion in the long-term (Prandle and Lane, 2015) will modify the distribution of the benthic fauna according to their specific salinity tolerance. The relocation will be further shaped by the boundary imposed by the substratum properties, since substratum type is important in providing habitat diversity, acting as a refuge and retaining food (Little et al., 2017).

Increasing flooding during rainy periods over fertilized agriculture land will enhance estuarine eutrophication and stratification, and consequently will increase the likelihood of phytoplankton blooms (Li et al., 2015). Such bloom events are difficult to synchronize with local grazers (Thackeray et al., 2016) leading to short-term biomass accumulation that ultimately will reduce the water quality by increasing the risk of hypoxia and production of microalgal toxins (Anderson et al., 2015) (medium confidence). The evidence of harmful algal blooms (HABs) has increased in coastal areas worldwide over the last 20 years (Anderson et al., 2015) (see Box 5.3) in relation to interactive effects of climate warming and nutrient inputs such as nitrogen and phosphate from land and wind surge (Paerl et al., 2016). Additionally, interactive effects between human pollution, warming and extreme precipitation events on estuarine turbidity, dissolved organic matter and oxygen concentrations, will increase the occurrence of pathogenic bacteria, e.g., Vibrio species (Baker-Austin et al., 2017; Kopprio et al., 2017) (Section 5.3.2, low confidence). Likewise, in shallow and microtidal estuaries, extreme river runoff increases the inflow of detritus and suspended sediments and reduces the optical properties of sea water. In addition, the exposure of high loads of organic
matter to bacterial degradation (i.e., resource-fueled respiration), increases the area of suboxic or anoxic conditions in bottom layers (Breitberg et al., 2015; Gobler and Baumann, 2016), increasing the likelihood of large fish kills (Warwick et al., 2018).

In particular, expansion of oxygen-depleted dead zones in coastal areas is projected under intensification of co-occurrence of climate threats and eutrophication, with warming as the main climate-related driver that enhances hypoxia through a variety of physical and biological mechanisms (Breitzberg et al., 2018). As the oxygen solubility is greater at lower temperatures, the effect of warming will be more pronounced on high latitude, temperate and arctic estuaries, where the predicted rates of warming are higher and the initial water temperatures are lower. Additionally, hypoxia will be more pronounced in shallow temperate estuaries because they are more responsive to warming due to their limited exchange with the open ocean (e.g., Rio de La Plata Estuary, Baltic Sea and Chesapeake Bay), and seasonality leads to dead zone development when temperature reaches critical values in summer (e.g., Black Sea) (Altieri and Gedan, 2015). Eutrophic and shallow estuaries face magnification on hypoxic conditions due to algae blooms and microbial respiration, which generate CO2 and reduce the pH (Breitberg et al., 2015). The coastal acidification related to the expansion of hypoxic dead zones is of main concern for conservation of sensitive organisms (Beck et al., 2011; Duarte et al., 2013; Feely et al., 2016), such as the pteropod Limacina helicina (Bendræske et al., 2017).

Estuarine gradients are further influenced by large-scale climate signals, such as North Atlantic Oscillation (NAO), El Niño Southern Oscillation (ENSO) and Southwest Atlantic Meridional Overturning Circulation (SAMOC), which have shown persistent anomalies since the 1970s associated with climate change (Wang and Cai, 2013; Delworth and Zeng, 2016; Garcia-Moreiras et al., 2018). In turn, these changes in estuarine gradients affect the phenology and composition of coastal plankton, as displayed by sustained observations (high confidence). For instance, at the temperate Mondego estuary (northeast Atlantic Ocean), higher temperatures driven by NAO fosters the intrusion of marine, cosmopolitan zooplankton species at the expense of native, estuarine species (Marques et al., 2017). Heavy rainfall events driven by ENSO, causes surges in river runoff and nutrient supply in temperate estuaries from the southern hemisphere, which in turn modifies the structure of phytoplankton communities (Sathiq et al., 2015; Thompson et al., 2015; Andrade et al., 2016) (medium confidence). In the temperate Bahía Blanca Estuary (SW Atlantic Ocean), warmer phases of ENSO mediate the emergence of water turbidity (enhanced by dredging) as a dominant driver of phytoplankton phenology, and promoted the decline of chlorophyll concentration at a yearly rate of 1% (López Abbate et al., 2017). Negative resonance effects from ENSO-driven droughts (i.e., increase soil salinity) also occur within salt marshes, where a reduction in plant production affects direct consumers (Pascual et al., 2015). In addition, ENSO exacerbates the effect of estuarine habitat modification by humans (e.g., coastal infrastructure, dredging) on the mobilization of blue carbon (Macleod et al., 2015). Heatwaves associated with La Niña in Shark Bay, Australia, produced nutrient pulses from degraded submerged coastal vegetation and triggered phytoplankton and bacterial blooms (Thompson et al., 2015; Arias-Ortiz et al., 2018). Although these changes in ecosystem components may be attributed to climate variability (Box 5.1), they provide tentative evidence of sensitivity to climate change, given the likely intensification of large-scale climate events (e.g., ENSO) projected for 2100 (Stocker et al., 2013).

Overall, the land-sea interface confers estuaries hydrological complexity and high habitat heterogeneity. Estuaries are commonly eutrophic ecosystems under intense human impacts, and are likely resilient to short-term environmental perturbations. However, growing long-term observational evidence since AR5 allows the attribution of ecosystem modifications to climate change (high confidence). The widespread observational evidence from shallow estuaries has already linked the increased levels of eutrophication and hypoxic conditions to warming (high confidence), and projections show that this processess will be more pronounced in estuaries from high latitudes (high confidence). Future scenarios of sea level rise, warming and extreme precipitation events will affect the balance between river runoff and marine water intrusions, thus modifying the physical and biogeochemical properties of the estuarine gradient. Shallow and microtidal estuaries in those areas exposed to higher probability of flooding will be the most vulnerable (high confidence) to experience transformation in the ecosystem functioning and services.

5.3.3.2 Salt Marshes
Salt marshes are productive ecosystems in the upper intertidal zone that are characterized by salt-tolerant, herbaceous vegetation. They occur in bays and estuaries from tropical to polar regimes, mostly found in mid-latitudes, where they support numerous animal species (Greenberg et al., 2014; Scott et al., 2014). Salt marsh plants commonly define patterns of zonation, with the lower zones dominated by a few hardy pioneer genera such as *Salicornia*, *Suaeda*, *Aster*, and *Spartina*, and then mixed communities of forbs and succulents at higher landward positions. Invertebrates often inhabit in plant culms and roots, gaining relief from physical stresses while facilitating microbially-mediated nutrient turnover via burrowing and storage via biodeposition. Despite their comparatively small area (55,000 km², in 99 countries; Mc Owen et al. (2017b)), tidal saline wetlands play disproportionately important roles in coastal carbon budgets, forming critical linkages between rivers, estuaries, and oceans (Najjar et al., 2018). As a result of tidal flooding, saltmarsh soils do not dry out and high levels of carbon can accumulate under anaerobic conditions. This is coupled with generally low rates of methane emission, which is strongly limited in saline marshes (high confidence) (Poffenbarger et al., 2011; Martin and Moreman-Valtierra, 2015; Kroeger et al., 2017). The carbon burial rate of salt marshes is comparable to mangroves due to high belowground productivity (McLeod et al., 2011). Together with seagrass meadows and mangroves, salt marshes represent ‘blue carbon’ habitats (see Sections 5.3.1.2, 5.4.1.1) (McLeod et al., 2011).

Nearly half of the intertidal area has been lost, due to drainage, agriculture, coastal settlement, hydrological alterations and reductions in sediment supply (Adam, 2002; Wang et al., 2014; Kroeger et al., 2017; Li et al., 2018). Sea level rise is a major threat to salt marshes; salt intrusion results in community restructuring (Janousek et al., 2017) and changing above-and below-ground productivity (McLeod et al., 2011; Watson et al., 2017). Plants with low tolerance to flooding are especially vulnerable. For example in New England salt marshes, *Spartina patens* is being replaced with flood-tolerant *S. alterniflora* or converted to pools of unvegetated standing water (Raposa et al., 2017), while *S. patens* specialized foragers are rapidly declining (Johnson and Williams, 2017). A similar expansion of the flood-tolerant *S. alterniflora* in response to rising sea level occurred in a temperate estuary in the southwestern Atlantic, Argentina. Over 33% of saltmarsh area covered by *Sarcocornia perennis* and 6% of halophytic shrub-like steppes dominated by *Cyclolepis genistoides*, *Allenrolfea patagonica* and *Atriplex undulata*, have been lost in the last 40 years and replaced by either mudflats or *S. alterniflora* marshes (Pratolongo et al., 2013).

Arid, sub-tropical marshes may show a contrasting response to raising sea level (medium confidence). For instance, the area of five of the largest coastal wetlands in Bahia California, Mexico, have increased by nearly 4% (Watson et al., 2017). The encroachment of mangroves into such saltmarshes (in response to warming and rising sea level) alters substantially the structure and function of the ecosystem. Benefits include, increased carbon storage and surface elevation, and improved coastal storm protection. However, there can also be negative ecological implications; the loss of open areas with herbaceous plants reduces food and habitat availability for resident and migratory animals (Kelleway et al., 2017a).

The ability of salt marshes to withstand in situ sea level rise depends on the development of new soil by the external supply of mineral sediments and organic accretion by local biota that builds marsh elevation. At many localities, critical organic accretion rates are declining due to decreases in plant productivity (from stress of inundation) and increases in plant and microbial respiration rates (i.e., from warming), thereby preventing marsh ability to keep pace with rising sea levels (medium confidence) (Carey et al., 2017; Watson and Hinojosa Corona, 2017). This negative climate impact is exacerbated in cases where anthropogenic barriers cause the ‘coastal squeeze’, which impedes the upward migration of less flood-tolerant plant and animal species, causing their loss (high confidence). In several salt marsh plant species in the northeast Pacific, however, inundation does not seem to be an important factor affecting litter decomposition (Janousek et al., 2017). Positive or negative climate feedbacks can result in salt marshes in response to sea level rise (Section 5.3.1.2) depending on the supply of suspended sediments, tidal ranges, subsurface processes (e.g., compaction) and above and belowground plant productivity.

Warming will increase metabolic rates of salt marsh organisms but effects will vary between species (medium confidence). For example, warming experiments in New England salt marshes showed greater increases in shoot productivity of the low marsh dominant grass, *Spartina alterniflora*, than that of the high marsh grasses, *Spartina patens* and *Distichlis spicata* (Charles and Dukes, 2009). The combination of higher CO₂ and warming could increase marsh elevation sufficiently to withstand moderate increases in sea level.
(Langley et al., 2009; Kirwan and Mudd, 2012). However, in models of rising temperatures and higher sea levels, Spartina alterniflora marsh responds initially with increasing accretion rates and carbon burial, therefore having negative feedbacks on climate, while by 2075, larger carbon pools and dominant root zone processes slow down rates of accretion and increase decay rates, switching to a positive climatic feedback (Kirwan and Mudd, 2012). Increased decay rates in response to warming are attributed to increased microbial and fungal activities (Kirwan and Blum, 2011).

Local stressors such as eutrophication, overfishing, and species invasions, accelerate the deterioration of saltmarshes. In New England salt marshes, for example, overfishing has led to uncontrolled grazing by a herbivorous marsh crab, Sesarma reticulatum, that is currently degrading Spartina alterniflora zones. This trophic cascade synergistically interacts with rising sea level because water-induced softening of inundated marsh peats permits new expansion of the crab into high marsh zones (medium confidence) (Crotty et al., 2017).

Excess nutrients may also reduce marsh resistance to sea level rise. Experimental nitrogen additions increased primary productivity and rates of sediment accretion in sediment-rich marshes (Morris et al., 2002) but caused low marsh collapse in sediment-poor marshes over the course of decades (Deegan et al., 2012). In the latter, excess nitrogen increased S. alterniflora shoot biomass at the expense of belowground biomass and also increased decomposition rates, both of which decrease the critical peat accumulation needed to maintain salt marsh elevation. Because salt marshes remove river-borne nutrients from waters entering coastal areas (Negrin et al., 2016), marsh area losses induced by rising sea levels are likely to enhance the nutrient input impacts described above.

Invasive plant species are likely to be favored by multiple factors associated with climate change. In North America, one of the most aggressive invasive species is the cosmopolitan Phragmites australis which is facilitated by excess nitrogen. As a C3 species, P. australis displays strong potential to expand under rising CO₂ levels (Mozdzer and Megenigal, 2012) and is evading many management strategies for its eradication (Martin and Blossey, 2013). However, recent mesocosm experiments revealed that in conditions of warming without excess nutrients, S. alterniflora resists P. australis invasion, thus minimizing nutrient pollution makes native species more competitive as temperatures increase (Legault II et al., 2018). In east China, experimental warming with the invasive S. alterniflora increases net nitrification and mineralization rates compared to soils with native plant species. Thus, invasive species may disproportionately benefit from warming-associated gains in nutrient availability (medium confidence) (Zhang et al., 2016).

Hydrological alteration is a key stressor to salt marshes worldwide; however, this factor is frequently reversible. Tidally-restricted wetlands comprise roughly 20% of coastal wetlands on the Atlantic Coast of the US. In this region, it is likely that significant permanent reductions in methane emissions, and enhanced carbon storage can be achieved by restoring tidal flows (see Section 5.4.1) (Kroeger et al., 2017). Such actions also increase the capacity of the ecosystems to maintain elevation relative to sea level. The scale of benefits depends on how completely tidal restrictions can be reversed as well the duration of prior alteration (Emery and Fulweiler, 2017). Areas for future research include the potential for marsh plants to locally remediate ocean acidification (via CO₂ uptake) or deoxygenation (through O₂ production/transport), as is currently considered for seagrasses (Garrard et al., 2014). Seasonal extremes may, however, be increased, reducing potential benefits (Sabine, 2018).

Substantial evidence supports with high confidence that salt marshes will continue to reduce their area in response to sea level rise in the coming century, especially if landward migration is further constrained by human modification of shorelines. Plants with low tolerance to flooding are particularly vulnerable and they can be locally extirpated. The loss of vegetated coastal ecosystems causes a reduction in carbon storage with positive feedbacks to the climate system (high confidence). Furthermore, the flooded area can become a mudflat or be colonized by more tolerant, invasive species (e.g., Spartina alterniflora, C3 plants), whose expansion is favored by combined effects of warming, rising CO₂ and nutrient enrichment (medium confidence). These shifts in community structure denote severe consequences in the ecosystem functioning and services.

5.3.3.3 Mangroves
Mangroves are vital but fragile coastal ecosystems found within sheltered areas in tropical and sub-tropical coasts around the world (Spalding, 2010) (see also Figure 5.14). Their global distribution falls into two main biogeographical regions namely the Indo West Pacific (IWP - which encompasses eastern Africa and Madagascar, Indo-Malesia, Asia and Australia) and the Atlantic East Pacific (AEP). These two regions are characterized by different species compositions and the dispersal between them has been limited by land barriers and wide expanses of water and the temperature thereof (Duke et al., 2007).

While over-exploitation of mangrove wood products, and conversion of mangrove areas into other land uses are the major mangrove threats in the world, climate change will likely exacerbate mangrove loss (Table 5.2). Mangrove ecosystems are particularly vulnerable to rising sea levels, increasing temperatures, and anoxic conditions brought about by sporadic sedimentation due to extreme rainfall (Alongi, 2008; Bosire et al., 2008; Gilman et al., 2008). Hydro-geomorphological settings play important role in mangrove responses to sea level rise (Sasmito et al., 2015; Eguiguren-Velepucha et al., 2016), where rising sea levels are predicted to increase flooding of the low-lying coastal areas and drawn mangroves (high confidence). In those areas where accretion of the sediments is sufficient and topography ideal (macrotidal, riverine areas), mangroves will migrate towards higher elevations (Cahoon et al., 2006; Lovelock et al., 2010). On the contrary, mangroves located in small islands, with coastal development causing squeezing, lack of rivers, groundwater extraction, steep topography and sediment-starved areas, are the more vulnerable to sea level rise. Thus, mangroves ecosystems are initially resilient (Ward et al., 2017) to increased storms and sea levels under scenario RCP2.6 throughout the 100 years projection period, but only up to mid century under RCP8.5 (Sasmito et al., 2015).

Increased salinity caused by sea-level rise may also result in decreased productivity and stunted growth in vulnerable species. The expected increase in mangrove growth under high CO₂ conditions has been experimentally observed with an unexpected change in salinity optima, shifting the ecological niche of the species towards more saline conditions (Reef et al., 2015). These physiological traits may imply changes in distributional patterns of the species. Likewise, the poleward expansion of mangroves and their encroachment into sub-tropical saltmarshes in response to warming and rising sea level (medium confidence) (Saintilàn et al., 2014) is causing alterations in ecosystem functioning and services (Kelleway et al., 2017a) (Section 5.4.1.3.2).

Mangroves operate in the C₃ pathway of carbon fixation for photosynthesis. Research indicates that increases in atmospheric CO₂ enhance the productivity and efficiency of water use by C₃ plants. Thus it is anticipated that projected increases in CO₂ will enhance mangrove tree growth and litter production. Recent long-term (20 years) experimental evidence, however, suggests that the positive C₃ response to increased CO₂ is a short-term one, as after 12 years exposed to high CO₂ levels, the plants decreased their biomass (Reich et al., 2018).

Inspite of the partial positive effects of rising CO₂ on mangrove growth rates, the overall effects of climate change on mangroves is negative (high confidence, Table 5.2) and will also negatively impact on coastal protection, agriculture, aquaculture and forestry activities (Table 5.2). Fringe mangroves are more vulnerable to rising sea level than basin mangroves, as well as those under microtidal regime with low soil accretion capacity (high confidence). Mangrove forestry activities will for instance be affected by changes in phenology, aridity, salinity and through direct disruption of specific sites by sea level rise and storms. Thus, the close monitoring of the status and responses of mangrove forests to climate change for early indication of change is important for appropriate management interventions.

**Table 5.2:** Predicted impacts of climate change on mangrove ecosystems (Ellison and Zouh, 2012).

<table>
<thead>
<tr>
<th>Factor</th>
<th>Processes Affected</th>
<th>Impacts</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rising sea level</td>
<td>Forest health</td>
<td>Forest mortality, dieback</td>
<td>(Ellison, 1993; Semeniuk, 1994; Cahoon et al., 2006; Gilman et al., 2007; Gilman et al., 2008; Soares, 2009; Ellison and Zouh, 2012)</td>
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<tr>
<td></td>
<td>Forest productivity</td>
<td>from the seaward edge</td>
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<td></td>
<td>Recruitment</td>
<td>Migration landward, but dependent on sediment inputs, topography and human modifications</td>
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<td></td>
<td>Inundation period</td>
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<td></td>
<td>Sedimentation rates</td>
<td></td>
<td></td>
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<tr>
<td>Extreme storms</td>
<td>Forest productivity</td>
<td>Forest damaged or destroyed</td>
<td>(Dahdouh-Guebas et al., 2005; Alongi, 2008; Yanagisawa et al., 2010)</td>
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<tr>
<td></td>
<td>Recruitment</td>
<td>Ground elevation change</td>
<td></td>
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<tr>
<td></td>
<td>Sedimentation rates</td>
<td>Erosion or sediment smothering</td>
<td></td>
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<tr>
<td>Increased waves and wind</td>
<td>Sedimentation rates</td>
<td>Changes in forest cover depending on whether coasts are accreting or eroding</td>
<td>(Semeniuk, 1994)</td>
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<tr>
<td></td>
<td>Recruitment</td>
<td></td>
<td></td>
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<tr>
<td>Increased air and sea temperature</td>
<td>Respiration</td>
<td>Reduced productivity at low latitudes and increased winter productivity at high latitudes.</td>
<td>(Clough and Sim, 1989; Cheeseman et al., 1991; Cheeseman, 1994; Cheeseman et al., 1997)</td>
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<tr>
<td></td>
<td>Photosynthesis</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td>Forest production</td>
<td></td>
<td></td>
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<tr>
<td>Enhanced CO₂</td>
<td>Photosynthesis</td>
<td>Increased productivity, subject to limiting factors of salinity, humidity and nutrients</td>
<td>(Snedaker, 1995; Farnsworth et al., 1996; Ball et al., 1997; Langley et al., 2009)</td>
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<tr>
<td></td>
<td>Respiration</td>
<td>Soil elevation gain</td>
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<td>Biomass allocation</td>
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<td></td>
<td>Forest productivity</td>
<td></td>
<td></td>
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<tr>
<td>UV-B radiation</td>
<td>Morphology</td>
<td>Minor</td>
<td>(Lovelock et al., 1992; Day and Neale, 2002; Caldwell et al., 2007)</td>
</tr>
<tr>
<td></td>
<td>Photosynthesis</td>
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<td>Reduced sediments and relative subsidence</td>
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<td>Reduced humidity</td>
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5.3.3.4 Seagrass Meadows

Seagrass-dominated meadows deliver essential ecological functions and services. The most relevant services they provide are related with carbon storage and the moderation of coastal disturbances. For instance, they offer protection to neighbouring coral reefs by their buffering ability against ocean acidification (Camp et al., 2016b), while seagrass meadows of *Posidonia* spp. form highly refractory carbon deposits that can be stored over thousands of years within sediments (Pergent et al., 2014; Hyndes et al., 2016; York et al., 2017). Efforts to understand the response of seagrass meadows to climatic stressors need to take into account co-occurring local-scale stressors, which are particularly relevant in coastal areas of intense human actions (Tomas et al., 2015). The co-occurrence of climatic and anthropogenic drivers is limiting the tolerance of plants to rising temperature and biological invasions, and is causing a worldwide decline of seagrass area (high confidence) (Tomas et al., 2015; Pagès et al., 2017).

Responses of seagrasses to climatic stressors change among species/genera, according to their life-history strategies (colonizing, opportunistic and perennial species) and climatic affinities (Rasheed et al., 2014; Kilminster et al., 2015; O’Brien et al., 2017; York et al., 2017). Certain seagrass species are expected to colonize new areas, i.e., ‘winners’, while others are expected to retreat, i.e., ‘losers’. Overall, warm-temperate regions, such as Western Australia, are the most vulnerable to lose seagrass area and the associated habitat complexity (Hyndes et al., 2016). Synergistic interactions of climatic stressors, e.g., heat waves and increased turbidity and nutrient levels associated with floods, have been shown to cause drastic changes in the composition and biomass of co-occurring seagrass species, potentially eroding provision of ecosystem services (high confidence) (Fraser et al., 2014; Nowicki et al., 2017; Lin et al., 2018). An example of a seriously damaged seagrass ecosystem following a marine heatwave is in Shark Bay, Australia, which has lost 36% of the meadows dominated by *Amphibolis antarctica* and to a minor extent by *Posidonia australis* (Arias-Ortiz et al., 2018). While Shark Bay seagrasses occupy an area of ~4% of the Australia’s vegetated coasts, they account for a 9% of the C sequestration by seagrasses, mangroves and tidal marshes. The losses
of seagrasses due to thermal stress impacts reduce the sediment C stocks and enhance CO₂ emission due to
oxygenation of the eroded sediments.

In the iconic Mediterranean seagrass meadow formed by *Posidonia oceanica*, warming causes a number of
negative indirect effects (medium confidence) such as increased toxicity of chemical hazards such as
sulphide (Garcia et al., 2013), and intense competition and predation by the arrival of tropical species
(Pergent et al., 2014). It also reduced the fitness of seedlings by increasing necrosis and susceptibility to
consumers and pathogenic pressure, while reducing establishment potential and nutritional reserves (Olsen et
al., 2016b; Hernán et al., 2017). Despite temperature-driven flowering of *P. oceanica* has been advocated as
way of increasing resilience to warming (Ruiz-Frau et al., 2017), severe habitat loss (70%) of this endemic
species is projected by 2050 and potential functional extinction by 2100 under RCP8.5 climate senario.
Similarly, warming will lead to significant reduction of *Cymodocea nodosa* meadows (46%) in the
Mediterranean, which will be compensated in part by future expansion into the Atlantic (Chefàoui et al.,
2018).

Under increased CO₂ levels in the future, seagrasses are predicted to physiologically perform better, as
photosynthetic rates increase (Koch et al., 2013; Borum et al., 2015). However, there is no consensus on
long-term consequences of increased CO₂ on seagrass carbon budgets and the stoichiometric composition of
seagrasses (Russell et al., 2013). Acidification in the future, moreover, will not counterbalance lethal effects
of warming on the physiological fitness of certain temperate species, such as *Zostera noltii* (Repolho et al.,
2017).

Responses of seagrasses to anthropogenic-mediated perturbations (e.g., CO₂ increments and fertilization)
fantherage feeding behaviour of their consumers, causing facilitation or suppression of herbivory
across functional groups (medium confidence) (Tomas et al., 2015). Importantly, the intensity of herbivory
over seagrasses is expected to increase with global warming, particularly in temperate areas, because of the
migration of tropical herbivores into temperate seagrass-dominated environments (Hyndes et al., 2016). In
turn, as herbivores play a key role in modulating the biomass of plant communities, their intense activity
affects the provision of ecosystem services (Scott et al., 2018). For instance, intense activity of tropical
herbivores reduces the production of refractory organic matter and thus the bulk carbon sequestration (Heck
et al., 2015; Hyndes et al., 2016).

Understanding the effects of climatic stressors require holistic approaches to encapsulate the wide range of
processes that may affect the fitness of seagrasses and the structure and functioning of the ecosystems they
support. In spite of these complexities, there is a high confidence that warming and the introduction of
tropical consumers in higher latitudes will continue to reduce seagrass area. Tropical seagrass species with
their low dispersal ability and their high exposure to above-normal temperatures are the most vulnerable to
local extinction in the future decades (medium confidence). This will affect the overall ecosystem diversity
by reducing the nursery area for fish species and the available habitat for species that live on tropical
seagrasses.

### 5.3.3.5 Sandy Beaches

Sandy beaches embrace the world’s longest coastline ecosystems and play an important socio-economical
role by providing area for recreation and tourism, and living or non-living resources for commercial
extraction. As assessed in AR5 WGII (Wong et al., 2014), the worldwide pervasive coastal squeezing, which
constricts sandy beaches between rising sea levels and expanding urbanization, will continue to limit the
ability of these ecosystems to compensate climate threats (high confidence). One of the most important
compensatory responses of sandy beaches to erosive processes is the relocation of sediments that facilitates
profile migration, but this response is constrained by the reduction of effective beach area. Along with sea
level rise, severe erosive events such as storms surges, flooding and onshore winds, challenge the
conservation of sandy beaches, accelerating dune scarping with vegetation loss (Castelle et al., 2015). Also,
excessive precipitation driven by climate change (Westra et al., 2013) will accentuate erosion through the
exportation of sand and therise in groundwater levels.

Identified as a major global-warming hotspot, the Southwestern Atlantic Ocean is experiencing a rise in sea
surface temperature at several times the average global rate (Hobday et al., 2016a). Increasing ocean
temperature produced a constant poleward shift of the warm water front (20°C isotherm) at a rate of ca. 9 km y⁻¹ (Ortega et al., 2016). In addition, an increase in speed and duration of onshore southern winds and storm surges (Escobar et al., 2004; Bischoff, 2005; D’Onofrio et al., 2008; Ortega et al., 2013), generated high energy waves (Codignotto et al., 2012). This climate-driven intensification of waves modified the morphology of sandy beaches through the augmentation of the swash width and the erosion rates (Ortega et al., 2013; Gutierrez et al., 2016). There is high confidence that this erosive process extends along with the current pace of sea level rise, keep causing morphological changes, suggesting a reduced resilience of the subaerial profile of sandy beaches. For the Pacific Ocean basin, a 33-year observational analysis for 48 beaches showed that erosion across Pacific coasts varies with ENSO (Barnard et al., 2015). Independently from sea level rise, the projected increase in the frequency of extreme ENSO events warns on the acceleration of flooding and erosion rates in opposite coastal sides of the Pacific Ocean basin (Barnard et al., 2015).

Evidence from long-term observations and empirical studies of South American sandy beaches allows attributing with high confidence, the reorganization of macrofauna to changing climate (Table 5.3). This is supported by global meta-analyses that disclose consistent responses across taxa and geographical regions (McLachlan and Defeo, 2017). Cool-adapted clam species are especially sensitive to warming, and are already experiencing mass-mortality events with low recovery denoting a poor adaptive capacity of beach-inhabiting populations to climate change. Mass mortalities have profound socio-economic implications by limiting the exploitation of marine resources and recreational activities such as artisanal seafood harvest (Defeo et al., 2013; Aburto et al., 2014; Gianelli et al., 2015; Turra et al., 2016). Opposite demographic responses may also occur in the Pacific and the Atlantic coasts of South America, where more tolerant filter feeding macrofauna are experiencing a tropicalization of species (high confidence) (Table 5.3).

In the east coast of Australia, a similar migration in response to climate was documented in adults of the ghost crab Ocypode cordimanus, which expanded poleward: 270 km south; 0.9° of latitude (Schoeman et al., 2015). This distributional shift is likely related to the regional ocean warming and the displacement southward of the surface isotherms along the coast at a rate of 20–50 km decade⁻¹. Similarly, in California, the upper beach isopod Tylos punctatus is shifting its northern range limit southward in response to intense expansion of human population and habitat degradation (Hubbard et al., 2014). This species will be under higher risk with rising sea level scenarios in these narrow beaches, where isopoda will have little possibility for landward retreat. In fact, 31% to 67% of southern California beaches are predicted to become completely eroded by 2100 under SLR scenarios of 0.93–2.0 m (Vitousek et al., 2017). Simultaneous with biogeographic changes, benthic crustaceans including isopods, crab and amphipod species display a consistent reduction in body size towards warmer latitudes in sandy beaches from eastern Pacific (USA and Chile) (Jaramillo et al., 2017). The fact that these organisms inhabit wide-ranging tidal zones and display contrasting feeding modes and life histories, and that they show a common response in both hemispheres, strength the reliability of body size as a metric of temperature effect (Section 5.2.3.1.1). This evidence suggests that sea surface warming in global sandy beaches will negatively affects trophic webs, with detrimental consequences for ecosystem services provision.

Table 5.3: Lines of evidence summarizing the potential effects of climate change in Western and Eastern American sandy beach macrobenthic communities. Based on the theoretical framework detailed in Parmesan et al. (2013).
Adapted from McLachlan and Defeo (2017). The numbers indicate the source references.

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<th>Lines of evidence</th>
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<td>Long-term observations and resurveys</td>
<td>Mass mortality events of the clam species <em>Mesodesma donacium</em> in the Pacific (Perú and Northern Chile) across the northern portion of its range linked to warm sea surface temperature events (e.g., El Niño), and <em>M. mactroides</em> in the Atlantic, following increasing SST and isotherms moving poleward across its southern range. <em>Southward shift of the isopod Tylos punctatus</em> in the Pacific (California, USA) linked to expansion of human population and habitat degradation, with high risk under sea level rise and coastal squeezing.</td>
<td>1. Riascos et al. (2009) 2. Ortega et al. (2013); Ortega et al. (2016) 3. Herrmann et al. (2009); Adams et al. (2011); Herrmann et al. (2011) 4. (Hubbard et al., 2014) 30 years of systematic sampling in Uruguay and resurveys in Brazil and Argentina across the distribution range of the clam</td>
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Climate events such as El Niño also affect pelagic communities inhabiting surf zones of sandy shores. In the southern east coast of Brazil, long-term changes in phytoplankton composition and abundance in the surf zone of sandy shores have been related to extreme El Niño 1997/1998 (Odebrecht et al., 2010; Odebrecht et al., 2014); while the prevalence of warm waters and increasing onshore winds depleted phytoplankton biomass and thus macrofaunal abundance in a sandy beach in Uruguay (Lercari et al., 2018). Similarly, Californian beaches suffered drastic changes in response to the 1997/98 El Niño event including the reduction in beach width (60%), abundance of macrophytes, species richness of shorebirds and individual size of macroinvertebrates (Revell et al., 2011).

Additionally, the interaction between climate warming and eutrophication are increasing the occurrence of harmful algal blooms (HABs, see Box 5.3), with tropical toxic species being increasingly represented in coastal subtropical ecosystems (Martinez et al., 2017). Exploitation of beach clams is often constrained by HABs, which can cause mass mortalities or render clams unsafe for human consumption (Gianelli et al., 2015). Eutrophication has also led to the mass development of mats of drifting algae along sandy beaches that deteriorated the environment and the resident macrofauna (Quillien et al., 2015; Quillien et al., 2016). Detached algae invade the water column and are able to increase their nutrient supply, building up large biomass and forming massive green (Ulva) and golden (Sargassum) tides (Smetacek and Zingone, 2013). The frequency, intensity and periodicity of these mass stranding events have been increasing worldwide (Ye et al., 2011) and have triggered high economic losses associated with beach deterioration and the consequent impacts on tourism and fisheries.
Habitat modifications due to sea level rise, erosion and coastal squeeze are also reducing the habitat accessibility for macrofauna. For instance, turtle nesting is being threatened by the reduction of suitable sandy beach area. Moreover, because sea turtles have temperature-dependent sex determination, warming could also affect their reproductive output (Hays et al., 2003). As incubation temperatures near lethal levels, growth and survival rates of turtle populations will tend to decrease in the long-term (medium confidence) (Laloë et al., 2016; Laloë et al., 2017). Shorebird richness and abundance may decrease in the long-term due to a combination of habitat loss, decreased accessibility because of sea-level rise and reduced prey availability, particularly when the upper shore and sand dunes are modified by engineering interventions that place armorig structures on beaches (Dugan et al., 2008). Sea-level rise is expected to affect suitable breeding sites for birds, particularly in the case of long coastlines with extensive low-lying coastal areas. Similarly, tropical cyclones (Convertino et al., 2011) and ENSO events (Hubbard and Dugan, 2003), which may become more extreme due to climate change (Cai et al., 2014), will tend to decrease nesting and shorebird abundance (medium confidence).

Overall, warming is promoting a range redistribution of macrofauna in sandy beaches with consequences for food webs, ecosystem structure and function. Cold-water species are highly vulnerable to mass-mortality when exposed to seawater warming (high confidence). The considerable reduction of habitable area driven by coastal squeezing implies a critical reduction on the adaptive capacity of macrofauna and other animals such as turtles and birds (high confidence). Projections of future sea level rise entails a decrease on the buffering capacity and lower recovery potential of these ecosystems to extreme climate signals such as El Niño events (high confidence).

5.3.3.6 Coral Reefs

Anthropogenic activities have resulted in major impacts on shallow water tropical coral reefs that are summarized by wide range of classes of drivers such as overfishing, nutrient enrichment, climate change and ocean acidification, and all eroding reef resilience (IPCC AR5 WGII). Climate change further drives ocean warming, which impacts biological and ecological reef processes, triggers large-scale coral bleaching events, and fuels tropical storms. Ocean acidification impacts overall physiological traits, slows reef calcification and growth, alters competitive interactions, and impairs population replenishment (Anthony, 2016). It is further noted that either one of the stressors have multiple, sometimes conflicting effects on functionally similar groups of reef species and their interactions (Harborne et al., 2017). The literature assessments (Munday et al., 2013; Ban et al., 2014; McClanahan et al., 2014; Rinkevich, 2014; Rinkevich, 2015a; Roff et al., 2015; Wolff et al., 2015; Anthony, 2016; Gunderson et al., 2016; Duvat et al., 2017; Gunderson et al., 2017; Harborne et al., 2017) point to negative consequences of reef ecosystems to warming, ocean acidification, and sea level rise (high confidence). Seawater warming results in bleaching and mass mortality, reduces coral growth rates and leads to changes in host symbiont combinations and physiologies, affecting coral species distribution changes, and causes phase shifts to algal dominated reefs (high confidence). Ocean acidification directly influences coral calcification and can lead to community changes, also enhancing reef dissolution and bioerosion (high confidence). The rate of sea level rise (primarily noticed in small reef islands) may surpass the ability of the coral reefs to grow fast enough to keep up (low confidence) (Brown et al., 2011; Perry et al., 2018). Reefs are further exposed to increased indirect impacts such as enhanced storm intensity, turbidity and increased runoff from the land (including the impacts of sedimentation and human derived toxicants) due to extreme precipitation events (reviews above) (high confidence).

Globally, coral reef communities are likely to change in the future through different modes of actions (IPCC AR5 WGII). Many taxa respond differentially to global threats, such as increased sea water temperatures and enhanced ocean acidification (Rodolfo-Metalpa et al., 2011; Edmunds et al., 2013; Munday et al., 2013; Takahashi and Kurihara, 2013; Wittmann and Poernter, 2013; Palumbi et al., 2014; Kavousi et al., 2015; Putnam et al., 2016; Ramajo et al., 2016; Krueger et al., 2017) (high confidence). In general, multiple stressors act in tandem to increase the risk of species population declines or extinction and such multiple stressors are exacerbated directly or indirectly by climate change impacts (Brook et al., 2008). It includes indirect impacts on physiology and behaviors (Gunderson et al., 2017) and other impacts, such as the decrease in water quality with the increased sedimentation and nutrient concentrations (Fabricius, 2005) (high confidence). The combination of global change drivers and anthropogenic impacts may also lead to
one of the most drastic consequences of coral reef degradation of phase shifts in reef communities (McCook, 1999; Hughes et al., 2010; Graham et al., 2013) (high confidence). A phase shift is characterized by an abrupt decrease in coral abundance or cover and concurrent increase in the dominance of non-reef-building organisms, such as algae and soft corals.

A wealth of empirical data (Wilson et al., 2006; Halpern et al., 2008; Edwards et al., 2011; Harborne et al., 2017) attest for a slow recovery of coral reefs resulting from repeated disturbance events (high confidence). For example, frequent warming events or a cluster of major storms develop into dependent stressors over time in those cases where the reef system’s health that is emerged from a specific impact, influences the vulnerability to, or recovery from, the subsequent event. The cumulative and synergetic impacts of various climate change drivers and anthropogenic impacts are also revealed with the increased rates of coral bleaching vulnerability under nutrient enrichment (Carilli et al., 2009) and results that connect ocean acidification with other stressors such as coral eating organisms (De’ath et al., 2012; Duvat et al., 2017; Harborne et al., 2017).

In contrast, the literature (DeBiasse and Kelly, 2016; Gibbin et al., 2017; Wall et al., 2017; Camp et al., 2018; Donelson et al., 2018; Drake et al., 2018; Veilleux and Donelson, 2018) attest that corals, as other marine organisms, possess the cellular, physiological and molecular machineries that help compensating for the effects of global change (medium confidence). As examples are species that are less impacted by either ocean acidification or highlighted temperature (Cornwall et al., 2018; Gintert et al., 2018). These outcomes and the existing critical gaps in our understanding of the cumulative global change impacts on coral reefs point to the increased uncertainty in predicting future responses by reef corals to climate change (high confidence) (Schulz et al., 2013; McClanahan et al., 2014; Mumby and van Woesik, 2014; Pandolfi, 2015; Folkesen, 2018).

Furthermore, there are clear regional differences for reef vulnerability (high confidence). On the smaller scales, deep reef ecozones are generally less exposed to disturbances (e.g., bleaching, storms, sedimentation, tourism impacts, floods from terrestrial habitats) and could therefore constitute an important refuge to some shallow habitats following disturbances, such as of larval supply (Bridge et al., 2013; Thomas et al., 2015) (medium confidence). Same conclusion is assigned to mesophotic reefs (found in tropica/subtropical regions at 30-150m depth) (Lindfield et al., 2016; Smith et al., 2016c; Bongaerts et al., 2017) (low confidence). Even lagoon and forereef areas of the same reef site may differ significantly in reef vulnerability to global change impacts like water temperatures and bleaching (Tkachenko and Soong, 2017) (high confidence). On the larger scales, in areas like the eastern tropical Pacific Ocean (Smith et al., 2017), the Caribbean (Chollett and Mumby, 2013), the Red Sea (Fine et al., 2013; Osman et al., 2017a), the Great Barrier Reef, Australia (Hughes et al., 2010; Morgan et al., 2017), and more, there are areas or habitats that avoid stress at spatial scales of >100 km, areas that were never bleached or support corals that are more resistant to stress or with the capacity to recover rapidly (medium confidence). Regional differences for reef vulnerability is further implied on large scale latitudinal gradients (van Hooijdonk et al., 2014) and on the global reef system analyses (van Hooijdonk et al., 2013; Heron et al., 2016; Langlais et al., 2017; McClanahan et al., 2017) (high confidence).

Almost all major reef systems that are vulnerable to climate change (IPCC AR5 WGII) (high confidence). An example of reef vulnerability is the Coral Triangle, the global pinnacle of tropical coral biodiversity, which has experienced thermal stress followed by extensive coral bleaching and coral mortality events are directly associated with extremes in the El Niño-Southern Oscillation (ENSO) (i.e., 1998 and 2010). This assessment further supports the conclusion of SR 1.5 °C that coral reefs are projected to decline by 70–90% at 1.5°C atmospheric warming above pre-industrial level (high confidence) with larger losses (>99%) at 2°C (very high confidence). Above all, the coral systems in tropical small islands are at major risk of being already severely affected by current climate change impacts, including both extreme events and gradual environmental changes, such as sea level rise (Duvat et al., 2017; Harborne et al., 2017) (high confidence).

5.3.3.7 Rocky Reefs

Rocky reef ecosystems span the intertidal and shallow subtidal zones of the world’s temperate coastal zones, and are dominated by calcareous mussels to weedy algal-dominated shallow, subtidal ecosystems. Other
organisms that inhabit rocky reefs are coralline algae (i.e., maerl beds), polychaetes, mollusks, bryozoans and sponges.

Intertidal rocky reefs are especially vulnerable to warming, due in part to the potential for extreme heat exposure during low tide emersion. Increased thermal stress during emersion associated with climate change is expected to cause the upper vertical limit of rocky intertidal species to shift lower (high confidence). Coupled with sea level rise and temperature-induced increases in predation by consumers (Sanford, 1999), whose distributions are associated with seawater temperatures, contractions of the vertical habitat available to rocky intertidal communities are expected (Harley, 2011). While previous studies have documented a poleward shift in species distributions of rocky intertidal and other rocky reef algae (Duarte et al., 2013; Nicastro et al., 2013) and faunal species (Barry et al., 1995; Mieszowska et al., 2006; Lima et al., 2007), local extinctions at the equatorial or warm edge of species ranges are increasingly being attributed to climate change (high confidence) (Yeruham et al., 2015; Sorte et al., 2017). Extreme heat waves are increasingly expected to cause mortality among rocky reef species (Gazeau et al., 2014; Jurgens et al., 2015) and the subsequent declines or losses in ecologically important species can have cascading effects on the whole intertidal community and the services it provides (Gatti et al., 2017; Sorte et al., 2017; Sunday et al., 2017). The loss of thermal refugia associated with continued warming could exacerbate the impacts of heat stress on rocky intertidal communities (Lima et al., 2016). At the same time, ocean acidification is expected to decrease the net calcification (high confidence) and abundance (medium confidence) of numerous rocky intertidal and reef-associated species (Kroeker et al., 2013), and the dissolution of calcareous species has already been documented in rocky inter-tidal tidepool communities (Kwiatkowski et al., 2016a). Moreover, the effects of ocean acidification on vulnerable species may be exacerbated by extreme thermal stress associated with heat waves (Rodolfo-Metalpa et al., 2011; Gazeau et al., 2014). Recent experimental and field studies, however, have demonstrated the importance of food resources in mediating the effects of ocean acidification on vulnerable rocky reef species (Ciais et al., 2013; Ramajo et al., 2016), suggesting that species’ vulnerability to ocean acidification may be most pronounced in areas of high heat stress and low food availability (medium confidence) (Kroeker et al., 2017). There is increasing evidence that the interactions between multiple climate drivers will determine species vulnerability and the ecosystem impacts of climate change (Hewitt et al., 2016).

Examples from naturally acidified rocky reef ecosystems suggest ocean acidification will simplify algal-dominated rocky reef ecosystems, due to an overgrowth in weedy algal species, a reduction in biodiversity, and a reduction in the abundance of calcareous species (medium confidence) (Kroeker et al., 2013; Linares et al., 2015). Reductions in the abundance of calcareous herbivores that historically have created space for more rare and competitively inferior species by grazing the dominant algae, are expected to contribute to the overgrowth of fleshy macroalgae on rocky reefs (Baggini et al., 2015). This shift towards macroalgae is associated with a simplification of the food web at lower trophic levels (Kroeker et al., 2011), but the emergent effects of ocean acidification on higher trophic levels (e.g., demersal fish) are less clear.

At a local scale, warming and ocean acidification are expected to change energy flows within rocky reef ecosystems. Experiments have demonstrated that both warming and ocean acidification may boost primary productivity at a local scale, which could lead to increased bottom-up productivity in some cases (Goldenberg et al., 2017). However, increased metabolic demands and consumption of predators under warmer temperature increase the strength of top-down control, which increase predation mortalities of herbivores and eliminate enhanced production from increased bottom-up effects (Goldenberg et al., 2017; Kordas et al., 2017). Increasing evidence suggests that ocean acidification could also increase species energetic costs and grazing rate of herbivores, which could boost ecosystem resistance to shifts associated with increased primary productivity (Ghedini et al., 2015). Although these increasingly complex experiments have highlighted the potential for species interactions to mediate the effects of climate change, our understanding of the effects on intact, functioning ecosystems is more limited. Despite predictions for increased production and herbivory with warming and acidification, a study of a more complex food web revealed an overall reduction in the energy flow to higher trophic levels and a shift towards detritus-based food webs (Ullah et al., 2018).

Overall, rocky reef ecosystems either dominated by calcareous mussels or algae are highly sensitive to ocean warming and acidification and extreme heat exposure during low tide emersion. Since AR5, increasing field and experimental evidence allow the attribution of species relocation towards lower intertidal levels and
mass mortalities, to climate change (high confidence). Ocean acidification also causes depletion of several calcified species (high confidence). Synergism between acidification and warming exacerbates the negative effects on rocky reef communities, causing a shift towards a less diverse ecosystem in terms of species richness and spatial heterogeneity (i.e., rocky reefs dominated by macroalgae) (medium confidence). This will directly affect the lower levels of the food web, with reduction in productivity and trophic energy (medium confidence).

5.3.3.8 Kelp Forest

Kelp forests are three-dimensional highly productive coastal ecosystems with a global net primary production (NPP) between 1.02 and 1.96 PgC yr$^{-1}$ (Krause-Jensen and Duarte, 2016), and an extension of approx. 25% of the worlds coastlines (Filbee-Dexter et al., 2016), from temperate to polar (see Chapter 3) rocky reefs (Steneck et al., 2003). Around 43% of the NPP is exported either as DOC (52%) or POC (48%) to neighbouring soft sediment habitats, deep canyons and the deep sea (see Section 5.2.4) where it provides additional food source for invertebrates and fishes, or it is buried in the sediments and thereby contributes to global CO$_2$ sequestration (Krause-Jensen and Duarte, 2016).

Two global datasets and one dataset spanning European coastlines (Araujo et al., 2016; Krumhans et al., 2016; Poloczanska et al., 2016) identify large local and regional variability of kelp abundance, with a small instantaneous rate of change of $-0.018$ yr$^{-1}$ displaying 38% of decline, 27% of increase and 35% of no change in the ecoregions (Krumhans et al., 2016). The average decline is much smaller than any observed regional loss which is a reflection of the naturally high spatio-temporal variability and resilience of kelp forests (Cavanaugh et al., 2011; Reed et al., 2016). A consistent negative response to regional warming between 1950 and 2009 in the abundance, phenology, demography or calcification of macroalgae on a global scale became evident for the northeast Atlantic and southeast Indian Ocean (Poloczanska et al., 2016).

Declines in kelp forest abundance attributed to climate change and not related to sea urchin overgrazing (which is a major driver of decline; Ling et al. (2015)) have been documented since the late 1980’s and evidence has increased within the last 2 decades. Conversely, climate related cooling events, e.g., along the southern African coastline (Bolton et al., 2012) or the warming of Polar sites (see Chapter 3) led to a spread of kelp beds or algal biomass increase. There is a general lack of data from some regions such as southern America (Pérez-Matus et al., 2017) and also a lack of historical baseline information (Bolton et al., 2012; Poloczanska et al., 2016) but since AR5, substantially more observational evidence supports that loss of kelp forest systems is associated with a phase shift from a 3-dimensional highly diverse kelp habitat to a less diverse and less complex turf forming habitat (high confidence) (Filbee-Dexter et al., 2016). Further evidence supports model predictions (Raybaud et al., 2013; Assis et al., 2016) showing that range contractions of kelps at the warm end of distributional margins and expansions at the poleward end are ongoing at diverse sites globally (high confidence).

A good example for ongoing range changes is documented in the northeastern Atlantic, where the warm-temperate kelp species Laminaria ochroleuca has proliferated at its leading edge in southwest Britain during the last decade corresponding to a period of rapid warming in the western English Channel (Smale et al., 2015), and at the same time has retracted westward along the north coast of Spain (Diez et al., 2012) (high confidence). The most important cold-temperate kelp species of the northeast Atlantic, Laminaria hyperborea, also retracted by 1.7° latitudes northward between the 1980s and 2011 along a gradient of decreasing temperature and increasing nutrients in western Portugal (Tuya et al., 2012).

In southern and southwestern Norway, the sugar kelp Saccharina latissima disappeared on a broad spatial scale (Moy and Christie, 2012). This phenomenon was especially pronounced in the Skagerrak region where heat-waves in 1997, 2002 and 2006 surpassed sub-lethal levels for sugar kelp of approx. 19°C for 5-8 weeks (Moy and Christie, 2012). Thus increase in summer temperature is a major driver for the decline, and synergistic interaction with eutrophication enhances this situation and also partially hinder the re-establishment (Andersen et al., 2013). A heatwave also impacted the infralittoral fringe kelp Laminaria digitata in the southern North Sea, an area of rapid temperature increase (Wiltshire et al., 2008). The species temporarily suffered canopy destruction after a prolonged summer heat wave in 2003 surpassing 19°C for 10 days (Bartsch et al., 2013). Tolerance limits alone do not explain this observation, thus synergistic negative interaction with high irradiances is an alternative explanation (Bartsch et al., 2013) (low confidence). Species distribution modelling applying CMIP5 temperature data predicts the complete loss of this kelp from its current southern distribution edge in Brittany (France) up to Denmark and the southern UK by 2100 with
spreading northward extinction under medium to high warming scenarios (Raybaut et al., 2013) (medium confidence). Along the north coast of the Iberian Peninsula the dominant warm-tolerant kelp *L. ochroleuca*, and the kelp-like *Saccorhiza polyschides* have retreated westward since the 1980’s and have been replaced by turf communities and warm water species (Fernandez, 2011; Diez et al., 2012; Voerman et al., 2013). Limited dispersal capacity in association with warming is identified as major driver for the considerable retreat of *S. polyschides* by species distribution modelling (Assis et al., 2017) (medium confidence).

In the northwest Atlantic off Nova Scotia, Canada, during a period of increasing mean seawater temperature (1.58°C between 1979 and 2014 at 2–6 m depth), kelp biomass has substantially decreased by 85-99% (Filbee-Dexter et al., 2016), while turf forming algae has increased positively related to the amount of inorganic sediment. Threshold temperatures for tissue degradation, breaking stress and mortality (Simonson et al., 2015) of the three dominant kelp species: *Agarum clathratum*, *Saccharina latissima* and *Laminaria digitata*, have significantly increased during this time period (Filbee-Dexter et al., 2016). *L. digitata* does not survive the summer simulation of the predicted 3°C temperature increase for 2100 (23°C) for 2 weeks (Wilson et al., 2015), what corroborates the predictions for this species in the northeast Atlantic (Raybaut et al., 2013). Synergistic effects of temperature with multiple drivers such as physical disturbance by hurricane damage (Filbee-Dexter and Scheibling, 2012) and interaction with enhanced grazing when kelp cover decreases have the potential to further reduce the kelp biomass (O’Brien et al., 2016).

In the northwest Pacific, decline of kelp bed ecosystems have been observed since the 1980s, especially along the coasts of Japan which span from sub-tropical to temperate areas and are considered a global biodiversity hotspot (Tittensor et al., 2011). Along the coast of Japan seawater temperatures have risen by 1.08°C per century (1891–2012) which is more than the double of the mean global ocean temperatures for this period. In 1989–1991 the total area of seaweed beds covered 2012 km² while it decreased by nearly 30% in 1998 (Takao et al., 2015) (medium confidence). The southern kelp beds of Japan are inhabited by one of the most warm adapted kelp species, *Ecklonia radiata*, which can still survive up to 27°C (Takao et al., 2015) although temperatures above 20°C will negatively impact the productivity (Sonisawa et al., 2004). *Ecklonia* beds have rapidly declined since the 1990’s, and partially disappeared by 2000 in southern Japan mostly attributed to the overall increase of seawater temperatures strongly affecting abalone fisheries which depend on *Ecklonia* as food source (Serisawa et al., 2004). Along a coastline of approx. 700 km length in southwest Japan, a poleward expansion of tropical species and a retraction of temperate species follow an increase of mean annual SST of 0.3°C per decade between 1970–2009 (Tanaka et al., 2012) (high confidence). A significant reduction in size of potential habitats is projected under all four RCPs using sea surface temperature projected from CMIP5 models, in combination with a species distribution model of kelp (*Ecklonia*) and its main herbivorous fish species (Takao et al., 2015). The reduction for 2090 is projected to be 15% in the RCP2.6 scenario, but up to 55%, 75% and 85% habitat loss compared to the present-day extent under RCP4.5, 6.0 and 8.5 scenarios, respectively. The loss process is expected to accelerate from 2020 onwards (Takao et al., 2015) (medium confidence). Even the kelp species that is most adapted to warmer waters, *Ecklonia radiosa*, apparently disappeared from its former southern distribution limit in Kagoshima Bay concomitant with the 1°C increase in mean seawater temperatures in the past four decades (Komazawa et al., 2014; Terada et al., 2016).

Increased physical stress by storm events also alters kelp communities. A good example is the considerable decline in the average abundance of the disturbance susceptible ‘warm water’ kelp *L. ochroleuca* during an unusual storm community in 2014 at southern sites of the British Isles (Smale et al., 2016). As kelp species possess a significantly different communities of organisms living on them and productivity (Smale et al., 2013), increase in the intensity of storms as forecasted by IPCC scenarios will change the food-web structure and productivity of future kelp assemblages (Filbee-Dexter and Scheibling, 2012). In addition, the age structure of kelp communities will become younger under increased physical disturbance favouring species with a year-round spore production or an opportunistic life strategy (Pereira et al., 2017).

In summary, kelp forest ecosystems will continue to decline in temperate regions and to expand poleward driven by warming (high confidence). Subtropical invasive kelp species will continue spreading facilitated by ocean warming and rising CO₂ (high confidence). Further observational evidence since AR5 supports that the loss of kelp forests attributed to climate change is followed by the colonization of turfs, which contributes to the reduction in habitat complexity and diversity (high confidence). Synergisms between increased extreme events such as storms, hurricanes and heat waves, and intensified herbivory due to
warming cause physical and physiological stress to kelps, reducing their population fitness, e.g.,
reproduction and productivity (high confidence).

5.3.3.9 Implications for Coastal Marine Megafauna

As endotherms, mammals and birds are more buffered to the effects of temperature changes than reptiles,
which as ectotherms, rely on external heat sources to regulate their body temperature. For reptiles, including
lizards, turtles and snakes, temperature directly effects important life history traits including hatching size,
sex, viability and performance (very high confidence) (Hays et al., 2003; Pike, 2014; Cavallo et al., 2015;
Dudley et al., 2016). This is particularly important for marine turtles as changing temperatures will affect the
hatching sex ratio because sex is determined by nest site temperature (high confidence) (Santidrián Tomillo
et al., 2014; Patricio et al., 2017; Santos et al., 2017). Rising temperature will also lead to a loss of breeding
substrate as the associated rise in sea level can reduce the available nesting or pupping habitat for land
breeding marine turtles, lizards, seabirds and pinnipeds (very likely) (Fish et al., 2005; Fuentes et al., 2010;
Hattfield et al., 2012; Funayama et al., 2013; Reece et al., 2013; Katselidis et al., 2014; Patino-Martinez et al.,
2014; Pike et al., 2015; Reynolds et al., 2015).

Changes in ocean temperature, and pH will also indirectly impact marine mammals, seabirds and reptiles by
changing the abundance and distribution of their prey (high confidence) (Polovina, 2005; Polovina et al.,
2011; Doney et al., 2012; Sydeman et al., 2015; Marshall et al., 2017; Woodworth-Jefcoats et al., 2017).
While indirect effects of changing climate are harder to document, examples are becoming apparent. The
distributions of some of these large animals is determined by the occurrence and persistence of oceanic
bridges and barriers that are related to climate driven processes (Ascani et al., 2016; Briscoe et al., 2017).
For example, the decline of Arctic sea-ice is effecting the range and migration patterns of some species and
is allowing the exchange of species previously restricted to either the Pacific or Atlantic oceans (virtually
or Monodon
Montacore, are avoiding the use of key habitats that are already in decline to avoid predation (medium
confidence) (Sydeman et al., 2015; Breed et al., 2017) (see SROCC Ch. 3).

The habitat requirements of marine mammals, seabirds and sea turtles are associated with bathymetric and
mesoscale features that facilitate the aggregation of their prey (Bost et al., 2009; Kavanagh et al., 2015;
Hindell et al., 2016; Hunt et al., 2016; Santora et al., 2017). The persistence and location of these features are
linked to variations in climate (Crocketer et al., 2006; Baez et al., 2011; Dugger et al., 2014; Bost et al., 2015;
Youngflesh et al., 2017) and to foraging success, juvenile recruitment, breeding phenology, growth rates and
population stability (high confidence) (Ancona and Drummond, 2013; Ducklow et al., 2013; Chambers et al.,
2014; Descamps et al., 2015; Abadi et al., 2017; Abrahms et al., 2017; Bjorndal et al., 2017; Fluhr et al.,
2017; Youngflesh et al., 2017). These changes have been most pronounced in polar environments, where
sea-ice is an important physical feature that is required for both breeding and haul out by birds and
mammals (Ducklow et al., 2006; Siniff et al., 2008; Jenouvrier et al., 2009; Costa et al., 2010; Constable et
al., 2014; Barbraud et al., 2015; George et al., 2015; MacIntyre et al., 2015; Amelineau et al., 2016; Renner
et al., 2016; Ferguson et al., 2017; Fernandez-Arcaya et al., 2017; Hamilton et al., 2017; Hauser et al., 2017;
Stempniewicz et al., 2017) but also forms a critical link in the structure of polar marine habitats (Murphy et
al., 2013; Murphy et al., 2016) (see Chapter 3).

A consistent theme in both predicted as well as observed changes in megafauna attributed to climate change
is that there will be winners and losers (virtually certain) (Costa et al., 2010; Hazen et al., 2013; Clucas et
al., 2014; Constable et al., 2014; Laird et al., 2015). For example, while some Adélie and Chinstrap
Penguin populations have plummeted, Gentoo Penguin colonies have increased (Trivelpiece et al., 2011;
Cimino et al., 2013; Ducklow et al., 2013; Koczak-Abshire et al., 2013; Dunn et al., 2016; Youngflesh et
al., 2017). Such changes are quite regional, Adélie Penguin populations at the southernmost limits of the
range are increasing (LaRue et al., 2013), while those at the northern limits are declining (Trivelpiece et al.,
2011; Cimino et al., 2013; Ducklow et al., 2013). Long term population changes associated with climate
change have also been observed for temperate species of seabirds and shifts in distribution of marine
mammals and seabirds (high confidence) (Henderson et al., 2014; Ramp et al., 2015; Trathan et al., 2015).
Rockhopper, Eudyptes chrysocome, and Yellow Eyed Penguins, Megadyptes antipodes (Hiscock and
Chilvers, 2014; Morrison et al., 2015; Mattern et al., 2017), have declined, while populations of Gentoo,
Pygoscelis papua, and Magellanic Penguins, Spheniscus magellanicus, Imperial Cormorants, Phalacrocorax
atrieps, Dolphin, Larus scoresbii, and Kelp, L. dominicanus, gulls have increased, some significantly (Rey
et al., 2014). However, increases in Magellanic penguin populations occurred at higher latitudes colonies,
whereas they declined at the lower latitude colonies (Boersma and Rebstock, 2014). This pattern is
consistent with a shifting habitat and species range associated with climate change (highly likely). Laysons,
Phoebastria immutabilis, and Wandering, Diomedea exulans, albatross have responded positively to climate
change as they have been able to take advantage of the increased intensity of winds, allowing them to forage
farther and faster making their foraging trips shorter increasing their foraging efficiency and breeding
success (Weimerskirch et al., 2012; Thorne et al., 2016).

Climate change is also associated with an increase in the frequency and intensity of storms and other
extreme weather events (high confidence) (Vargas et al., 2006; Henderson et al., 2014; Descamps et al.,
2015; Shirasago-Germán et al., 2015; Descamps et al., 2017; Pardo et al., 2017) (Chapter 6), causing
increased risk on population viability of marine mammals and seabirds (medium confidence). An increase in
storms between 1983 and 2010 resulted in increased reproductive failure in Magellanic Penguins at Punta
Tombo, Argentina (Boersma and Rebstock, 2014). An unusual ‘warm-water anomaly’ off the Northeast
Pacific Ocean resulted in closures of commercially important fisheries and mass strandings of marine
mammals and seabirds (Cavole et al., 2016) and a reduction in Shag, P. aristotelis, and Kittiwake, Rissa
tridactyla, breeding success in the North Sea (Newell et al., 2015). Depending on the age class of petrel and
a regional effect, there could be both positive and negative effects of tropical cyclones on Round Island
Petrels, Pterodroma arminjoniana (Nicolli et al., 2017).

The future for species that have limited ranges, limited habitat availability or small populations is quite
precarious (very high confidence). Some species like Galapagos Penguins, Spheniscus mendiculus, sea lions,
Zalophus wollebaeki, fur seals, Arctocephalus galapagoensis, and flightless cormorants, Phalacrocorax
harrisi, have very small populations with restricted ranges and do not have alternative habitats to utilize
(Vargas et al., 2005; MacLeod and Steinfartz, 2016; Riofrio-Lazo et al., 2017). In general, specialist species
are more vulnerable than generalists. This can be in terms of diet or habitat requirements, or in terms of
existing life history traits (Gallagher et al., 2015).

5.3.3.10 Summary for Coastal Ecosystems

Overall, coastal ecosystems are under stress from the combination of climate change impacts in the ocean
and on land as well as non-climatic human activities, with observable and projected climate impacts through
modification of geomorphology, reduction in habitat area, and shifting distributions and community structure
(high confidence) (Section 5.2.2). Specifically, the morphology of sandy beach and saltmarshes is changing
as a result of rising sea level and more severe storms that leads to their erosion and retreat (Sections
5.2.2.3.4, 5.2.2.3.5). These consequently lead to losses of vegetation and benthic fauna (high confidence),
with detrimental consequences for saltmarshes carbon storage (high confidence) (Section 5.3.2.3). The
increased levels, frequency and duration of eutrophication and hypoxic conditions in estuaries and coastal
zones worldwide in the last few decades are attributable to warming (high confidence), and these processes
are projected to be more pronounced in the high latitudes (high confidence) (Section 5.2.2.3.1). Synergism
between acidification and warming exacerbates the negative effects on rocky and coral reef communities,
causing a shift towards a less diverse ecosystem in terms of species richness and spatial heterogeneity with
detected climate impacts including the increased dominance of macroalgae in rocky reefs and decreased in
species richness and structural complexity of coral reefs (medium confidence) (Section 5.2.2.3.6). Kelp
forest, mangroves and seagrass meadow ecosystems are shifting poleward with warming and increasing
frequency and intensity of extreme events such as heat waves and storms (high confidence) (Section
5.2.2.3.7). Herbivory intensified by warming will result in physical and physiological stress on coastal
vegetation and reduce their productivity (medium confidence) (Sections 5.2.2.3.4, 5.2.2.3.8).

5.3.4 Assessments for Coastal Ecosystems

This section synthesizes the assessments on coastal ecosystems to evaluate the risk of climate change on
their biodiversity and ecosystem functions (Appendix 5.A, Table 2). The overall global level of risk is
assessed based on the latest understanding of changes in oceanographic drivers in coastal regions (Section
5.3.2). The level of vulnerability of coastal ecosystems to climate hazards depends on their sensitivity and adaptive capacity discussed in Section 5.3.3. The ecosystem sensitivity is assessed by discriminating the distinct responses to climate hazards of the main biotic and abiotic components that define each ecosystem (Section 5.3.3). For example, intertidal ecosystems are more threatened by erosive processes due to sea level rise and storms than subtidal (always submerged) ecosystems; rising sea level affects the geomorphology in estuaries and sandy beaches (e.g., dune scarping), and the plant biomass and soil accretion in saltmarshes and mangroves. Conversely, in ecosystems defined by foundational species such as seagrasses, kelp forest, coral and other biogenic reefs, the physiology, distribution and ecological interactions of the organisms are more sensitive to warming and acidification.

All coastal ecosystems assessed here are exposed to multiple climate hazards (rising sea surface temperature, acidification, deoxygenation, sea level rise, storms and precipitation) that increases with higher carbon emission scenarios. The level of exposure depends on (1) the geographical distribution (Figure 5.12), (2) local interactions between climate and human impacts (e.g., pollution, eutrophication, aquaculture, urbanization), and (3) topographic characteristics (e.g., intertidal or subtidal). For example, estuaries and sandy beaches are distributed all across the global coastlines that are experiencing diverse range of changes in climate hazards, while seagrasses and mangroves are restricted to temperate and tropical/subtropical regions where their exposure to climate hazards is generally similar in nature (Figure 5.14). Likewise, kelp forests and seagrasses meadows are submerged ecosystems, while all the other ecosystems defined here are intertidal (rocky and coral reefs can be both, intertidal and submerged), with exposition to atmospheric-oceanographic conditions. Overall, in all coastal ecosystems, multiple climate drivers will emerge from historical variability in the 21st century under RCP8.5 (time of emergence, Box 5.1), while the time of emergence will be later with less climate drivers under RCP2.6. Non-climatic human drivers such as eutrophication add to, and in some cases, exacerbate these large scale slow climate drivers beyond biological thresholds on local scale, e.g., deoxygenation.

**Figure 5.12:** Risk scenarios for coastal ecosystems based on observed and projected impacts of ocean warming, acidification and sea level rise. ‘Present day’ corresponds to the 2000s, whereas impact levels are for the year 2100 under the different CO₂ emissions scenarios: RCP2.6 (stringent reduction scenario) and RCP8.5 (business-as-usual high emissions scenario), with the intermediate scenario RCP4.5. Impact levels do not consider human risk reduction strategies such as mitigation and societal adaptation.

Some particularly vulnerable coastal ecosystems are already experiencing moderate to high risk under present day conditions. Specifically, coral reefs and seagrass meadows (Section 5.3.3.6 and 5.3.3.4, respectively). Changes in ecosystem structure and functions have been detected at present day and attributed to climate change (see Section 5.3.3). Furthermore, the surviving corals of tomorrow will not resemble the same as at present day, because irreversible changes in habitat structure and functioning, including species extinctions, food web disruptions and shifts in community complexity are currently emerging in some hot spots areas of climate change (e.g. the Caribbean reefs). In consequence, this permanent transition to new ecosystem states driven by unpredictable pulses of disturbance and progressive climate hazards implies detrimental impacts to ecosystem services (Section 5.4.1.2).
All coastal ecosystems are projected to have high to very high risk of impacts under RCP8.5 in the 21st century that varies generally across ecosystems. Particularly, the ecosystems with very high risks are coral reefs (very high confidence), seagrasses meadows (very high confidence), kelp forest (very high confidence) and rocky reef (medium confidence) under RCP8.5. These ecosystems have low to moderate adaptive capacity, as they are highly sensitive to warming and acidification. For example, kelp forests and endemic species of mid latitude seagrasses have low dispersal ability and are particularly sensible to heatwaves. Biogenic shallow reefs with calcified organisms (e.g., corals, mussels, calcified algae), are particular sensible to ocean acidification, which combined with rising temperatures, deoxygenation, sea level rise and increasing extreme events, make these ecosystems highly vulnerable (impede resilience) to future emission scenarios.

The ecosystems with the lower levels of risk under future emission scenarios and more resilient to increasing level of climate change are mangroves (medium confidence), estuaries and sandy beaches (low/medium confidence), and saltmarshes (medium confidence). In general, these ecosystems are initially resilient to climate threats. Estuaries and sandy beaches are highly dynamic in terms of geomorphology, sediment processes, river runoff and biogeochemistry, what confers them more adaptive capacity to climate drivers. Mangroves and saltmarshes can initially cope with sea level rise by plant biomass accumulation and sediment deposition, but saltmarshes will not be able to withstand the sea level rise under RCP8.5.

Across the suite of ecosystems that have been assessed in the coastal seas (Figure 5.12), there is increased risk, and overall none have a reduction of risk in either low or high emission scenarios (RCP2.6 and RCP8.5). In all cases that were assessed, with all of the factors considered (climate drivers and physiological understanding) in Section 5.3.3, RCP2.6 has a lower level of risk than the high emissions scenario RCP8.5 (very high confidence).

5.4 Changing Marine Ecosystem Services and Human Wellbeing

The Fifth Assessment Report concluded that climate change increases the risk of impacts on the goods and services derived from marine biodiversity and ecosystems. These goods and services include fisheries and other provisional services, climate regulation by marine microbes, coastal protection by vegetation and other biogenic habitats, and cultural values such as tourism, heritage and ceremonial importance (Pörtner et al., 2014). This section further assesses the impacts of climate change on ecosystem goods and services in specific coastal, pelagic and deep-sea floor ecosystems. Under the conceptual framework developed by the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Hughes et al., 2017) (Figure 5.13), climate change is considered to be a direct driver that affects ecosystem goods and services (provisioning and others including regulation, supporting and cultural, see Section 5.4.1). This section assesses how climate change impacts on marine biodiversity and ecosystems (Sections 5.2 and 5.3) affect changes in ecosystem services and their consequences for human wellbeing (social, cultural, economic and environmental, see Section 5.4.2). Climate change can also impact ecosystem services through the amplification of the effects of non-climatic drivers on marine ecosystems and biodiversity, such as increasing the variation and magnitude of runoff and its associated impacts of erosion and coastal sedimentation (Singh et al., 2017). Attribution of climate change effects on these marine ecosystems are addressed in Sections 5.2 and 5.3, while Section 5.4 assesses the implications of changing ecosystem goods and services for human wellbeing and their challenges to the United Nations’ Sustainable Development Goals (Section 5.4.2).
Figure 5.13: Conceptual framework of the linkages between climate change, non-climatic human drivers, ecosystems services and human wellbeing (a) and illustrative examples from tropical fisheries (b), deep sea nutrient re-cycling (c) and coral reef tourism (d) (see 5.3.1, 5.3.2, and Singh et al. (in review)).

5.4.1 Changes in Key Ecosystem Services

Ecosystem services are the environment and its processes that render benefits to people and support the wellbeing of people (Figure 5.13) (Tallis et al., 2010). Recently they have also been described as ‘Nature’s Contribution to People’ (Diaz et al., 2018). These services include all benefits that humanity obtains from nature. These ecosystem goods and services can be broadly divided into provisioning services, regulating services, cultural services and supporting/habitat services, although they are interconnected with one another (Figure 5.13) (Leadley et al., 2014a). It also includes diverse values from monetary to intrinsic. Provisioning services include material or energy outputs from ecosystems, while some key ecosystem services that human access directly are in turn supported by biodiversity and other ecosystem services. For example, climate change will reduce the long-term viability and increase the risk of local and global extinction of vulnerable organisms (Section 5.2.3). If the abundance of these species decreases, fisheries resources will also decrease, and the potential reduction in genetic diversity will lead to loss of potential genetic resources. Regulating services are those ecosystem functions that ensure the environment is in a state conducive to human wellbeing and development (Costanza et al., 2017). Climate regulation is a core regulating ecosystem service, and the feedback from a reduction in climate regulation on increased greenhouse gas emissions can further exacerbate climate impacts on other regulating or other types of ecosystem services. For example, projected increases in the intensity of storms in the 21st century under high CO₂ emission scenario can disrupt the ability of local wetlands to regulate and treat waste-water (Erwin, 2009). Climate change also impacts habitats or ecosystem functions that support or maintain biodiversity and other ecosystem services, known as habitat/supporting ecosystem services (Costanza et al., 2017). Climate change will impact biogenic habitats such as coral reefs (Section 5.2.2.3) and mangrove forest (Section 5.2.2.6), consequently reducing their suitability for organisms that are associated with those habitats (Pörtner et al., 2014; Wong et
al., 2014) (Sections 5.2.2, 5.2.3). The final major category of ecosystem services – cultural ecosystem services – includes recreation, tourism, aesthetic and spiritual experiences. These services are a product of both humans experiencing nature and the availability of nature to provide the experiences (Chan et al., 2012); the quality and quantity of the latter can be impacted by climate change. Also, in some cases, climate change will alter the distribution of organisms and ecosystems, and consequently, where these ecosystem services are enjoyed. However, adjustments to how human communities access place-based services may not be possible, particularly for those services that rely on indigenous knowledge and local knowledge (Cross-Chapter Box 3 in Chapter 1). The cultural aspects and linkages to local knowledge and indigenous knowledge of human-ecosystem linkages are further highlighted in the recent suggestion to use ‘Nature’s Contribution to People’ as a new conceptual framing instead of ecosystem services (Díaz et al., 2018).

However, this assessment will be based on the conceptual framing of the relevant existing literature, which is mostly still based on the ecosystem services framing, with the particular focus on the observed and projected impacts and risks of climate change on these services (Sandifer and Sutton-Grier, 2014). Following the overall structure of this assessment, ecosystem services from open-ocean and coastal ecosystems are assessed.

5.4.1.1 Open Ocean Ecosystems

The vast volume of open ocean ecosystems offers the full range of ecosystem services, including provisioning, regulatory, supporting and cultural services, to human societies, although our access to these services are often indirect (Ingels et al., 2016). The open ocean ecosystems are fundamental components of the Earth system that are key to climate regulation (Section 5.2.2). The pelagic ecosystem, where the majority of the open ocean primary production takes place (Section 5.2.2.5), supports biomass production for valuable fisheries as well as food for marine organisms including marine mammals, seabirds and turtles (Sections 5.2.3, 5.3.3) that contribute to eco-tourism. The deep sea floor ecosystems also support biodiversity that is important for fisheries and they recycle nutrients that support the pelagic ecosystem (Section 5.2.4). In addition, open ocean ecosystems and their associated biota are intrinsically valued by people, offer opportunities for scientific discoveries and education, and hold genetic resources that have potential for human uses (Blasiak et al., 2018).

Human access to open ocean ecosystem services is less direct compared to coastal ecosystems, rendering the detection and attribution of climate change impacts more difficult. Thus, in some cases such as for deep sea floor ecosystem services, we must conjecture from biophysical principles and knowledge about human accesses to similar ecosystem goods and services, in order to assess the risk and vulnerability of these ecosystem services to climate change.

5.4.1.1.1 Provisioning service

Fisheries from the open ocean pelagic ecosystems remain important sources of food, nutrition, income and livelihoods for many millions of people around the world (FAO, 2016). Globally, total fishery catches of pelagic species amount to 44–53 million MT annually in the 2000s (based on the data from Pauly and Zeller (2016)) (Figure 5.14), generating 9.2 billion USD of revenue directly from their fisheries (in year 2010 value) contributing to around 12% of the total fishery catch (Sumaila et al., 2015). The majority of the catch from the open ocean pelagic ecosystems is tunas and billfishes (Crespo et al., 2017). Fishery catches from these ecosystems have been stagnant since the 1990s, which is partly attributable to fishing at or above biological capacity for production (Merrie et al., 2014). Fishing has also driven down populations of species with vulnerable life history characteristics for a high extinction risk, such as large sharks (Fowler, 2014).

Based on fishery catch records and model projections, ocean warming, decrease in primary production and expansion of oxygen minimum zones in some ocean regions are shifting patterns of fishery productivity in these pelagic ecosystems towards higher latitudes or longitudinally following temperature gradients, while the distribution of commercially important fish stocks and their fisheries are shifting toward higher latitude. The distribution of fisheries on large pelagic species such as tunas correlate strongly with historical climate variability (high confidence). For example, in intermediate latitudes across the Atlantic, Indian and Pacific oceans, catches of tropical tunas, including skipjack and yellowfin tuna, are significantly positively related to increases in sea surface temperature, although the overall catches across latitudinal zones do not show a significant change (Monllor-Hurtado et al., 2017). Changes in the distribution of oxygen minimum zones also correlated strongly with changes in catches and the fishing effort of pelagic fishing fleets (Stramma et
al., 2011). Moreover, observed seasonal patterns of pelagic fisheries in the open oceans are also strongly and significantly related to oceanographic conditions, particularly sea surface temperature; this relationship is thought to be driven by the relationship between the distribution of pelagic fisheries resources and environmental variables (Crespo et al., 2017). Overall, the poleward shifts in pelagic fisheries agree with the observed and projected range shifts of tunas under ocean warming (Section 5.2.3.4) (Dell et al., 2015; Poloczanska et al., 2016).

Figure 5.14: Global distribution of fisheries catches (average 2010-2014, based on Pauly and Zeller (2016)), coastal habitats including seagrasses (UNEP-WCMC and FT, 2017) saltmarshes (Mcownen et al., 2017b), mangroves (Spalding, 2010), coral reef (UNEP-WCMC and WRI, 2010) and an index (called Marine Focus Factor) of inclusion of the ocean in the Nationally Determined Contributions (NDCs) published by countries (Gallo et al., 2017). The higher the Marine Focus Factor, the more frequent use of ocean in the country’s NDCs.

Globally, the potential fisheries catch in the open pelagic oceans is projected to decrease by 2100 relative to the 2000s under RCP8.5, RCP4.5, and RCP2.6 scenarios with regional differences (Cheung et al., 2016a; Blanchard et al., 2017). Based on projections from an ensemble of three CMIP5 earth system models and a fisheries impact model, there is a projected significant (more than -10% by 2050 relative to present day) decrease in catch potential under both RCP4.5 and RCP8.5 forcing (Cheung et al., 2016a). Here, potential fisheries catch or maximum catch potential refers to the biological potential productivity of the fish stocks to provide long-term fisheries catches; it is considered a proxy of maximum sustainable yield. However, the actual catches realized by fisheries will depend strongly on past and present fishing effort and exploitation status of the resources (Cheung et al., 2018a). Ensemble projections including multiple fisheries models generally support the direction of such projections (Blanchard et al., 2017). Results from these models are largely driven by projected changes in temperature and net primary production. Regionally, the tropical Ocean is projected to have substantially larger impacts (three times or more decrease in catch potential) than the global average, particularly in the western central Pacific Ocean, eastern central Atlantic Ocean and the western Indian Ocean under RCP8.5. In contrast, the catch potential in the Arctic is projected to increase, although there is high intermodel variability in these projections. In addition to poleward shifts in fisheries catches, selected open ocean pelagic resources are also projected to show a longitudinal shift, such as eastward shifts of skipjack tuna under ocean warming (Lehodey et al., 2011; Schulz et al., 2013). However, the actual catches and profitability of the fisheries interact strongly with the level of exploitation of the resources (Cheung and Pauly, 2016; Dueri et al., 2016). In addition, fisheries in the open ocean are predominantly operated by industrial fishing from a few countries (Sumaila et al., 2015), which may be more adaptable to changes in resource distributions, although analysis of the adaptive capacity of distant water
fishing to climate change is limited. Currently, knowledge is limited regarding how climate change would affect the large but unexploited biomass resources in the mesopelagic ecosystems; commercial interests to develop fisheries to exploit such resources are rising (St. John et al., 2016).

Overall, fisheries are currently the most important provisional service in the open ocean pelagic ecosystems. Fishery resources in this ecosystem are sensitive to oceanographic conditions; in particular, changes in their distributions follow sea surface temperature closely (robust evidence, high agreement, very high confidence). Historical patterns of fishing activities in the ocean pelagic ocean have also followed fishery resource distributions (medium confidence). Potential fisheries catch from currently exploited fish stocks are projected to decrease by mid-century under all greenhouse gas emission scenarios, with the tropical oceans being projected to have the highest impacts (medium confidence), partly as a result of the projected poleward shifts in resource distributions (high confidence) and a decrease in net primary production in low latitude regions (medium confidence). The realized fisheries catch in the future will depend strongly on the way that fishing activities are managed in the open ocean pelagic ecosystems (very high confidence).

[START OF BOX 5.2 HERE]

Box 5.2: Responses of Coupled Human-Natural Eastern Boundary Upwelling Systems to Climate Change

Eastern Boundary Upwelling Systems (EBUS) are among the most productive ocean ecosystems in the world (Kämpf and Chapman, 2016), supporting not only the livelihood of coastal communities and sectors that are directly dependent on EBUS, but also contributing to benefits that much broader human societies enjoy (García-Reyes et al., 2015; Levin and Le Bris, 2015) (Appendix 5.A, Table 3). The high productivity of EBUS is supported by the upwelling of cold and nutrient-rich waters from depth, driven by equatorward alongshore winds that cause the displacement of surface waters offshore and the replacement by deeper waters. Supported by the high productivity, total annual fisheries catch from the four main EBUS in the world (California Current, Humboldt Current, Canary Current and Benguela Current) amount to 16-24 MT per year in the 2000s, contributing to approximately 17% of the global catches (based on data from Pauly and Zeller (2016)). These catches are consumed locally as well as processed and exported as seafood, fish meals and oils to support aquaculture and livestock production. Upwelling of cold deeper water increases condensation of wet air in coastal areas that brings in moisture and precipitation that are important for the maintenance of coastal vegetation and agriculture, as well as suppressing of forest fires (Black et al., 2014). The high concentration of marine mammals attracted by the productive upwelling ecosystem support lucrative eco-tourism such as whale-watching in the California Current (Kämpf and Chapman, 2016). A total economic value calculation of the goods and services provided by the Humboldt Current alone is estimated to be US$19.45 billion per annum (Gutiérrez et al.). Thus, although their area is small compared to other pelagic ecosystems, the impacts of climate change on EBUS are likely to have disproportionately large consequences for human societies (very high confidence).

The coupled human-natural EBUS are vulnerable to the multiple effects of climate change with large regional variation (Blasiak et al., 2017). Observational and modelling analysis suggests that there has been an intensification of wind with global warming in three of the four main EBUS (except the Canary Current EBUS) during the last 60 years, with a number of hypotheses explaining the mechanisms of such intensification (Sydeman et al., 2014; García-Reyes et al., 2015; Rykaczewski et al., 2015). Moreover, coastal warming and wind intensification may lead to variable countervailing responses to upwelling intensification at local scales (García-Reyes et al., 2015; Wang et al., 2015; Oyarzún and Brierley, 2018; Xiu et al., 2018). Local winds and mesoscale oceanographic features (not resolved in most coupled climate models), rather than large-scale wind patterns, are thought to have a greater impact on regional productivity (Renaut et al., 2016; Xiu et al., 2018). Changes in SST in EBUS in recent decades are substantial but with conflicting evidence because of the differences in resolution of the data and their high temporal variability (Sydeman et al., 2014; Xiu et al., 2018). Specifically, cooling of coastal areas is observed in Humboldt Current, Benguela Current and California Current EBUS but not in the Canary Current EBUS. However, temperature trends revealed from recent records may not have emerged from the large natural variabilities of some EBUS. Larger-scale data that include both coastal and offshore waters around EBUS suggest that there have been decreases in pH (Turi et al., 2016; Chavez et al., 2017) and dissolved oxygen (Levin, 2018),
although biogeochemical changes are highly variable in coastal waters of EBUS. At present, global climate
models lack the necessary spatial resolution to represent upwelling processes adequately.

The climate change impacts on ecosystem services from EBUS vary regionally depending on both the
biophysical and the social and economic characteristics of the upwelling systems (Garcia-Reyes et al., 2015)
(Box 5.2, Appendix 5.A, Table 3; Box 5.2, Figure 1). The fisheries in the upwelling systems are highly
sensitive to upwelling conditions that is exacerbated by fishing effects on the exploited populations. For
example, the anchoveta population collapsed in the Humboldt Current after the El Niño in the 1970s that
impacted the fishing industries largely (Gutiérrez et al., 2017). Since the small pelagic fisheries from
upwelling regions are the main source of the global fishmeal market, the decrease of its production leads to
increase in international fishmeal price, that consequently affects the price of other food commodities (like
aquaculture-derived fish) that rely on fishmeal for their production (Merino et al., 2010; Carlson et al.,
2017). Moreover, the decrease in catches would also affect regional food security as catches from the
upwelling region is an important source of nutrients. For example, in the Canary Current, coastal fisheries
are important source of micronutrients to the nearby West African countries (Golden et al., 2016), whose
susceptibility to climate change impacts are particularly high and adaptive capacity is low because of their
strong dependence on the fisheries resources, a rapidly growing population and regional conflicts (Box 5.2,
Figure 1). Decreased small pelagic fish stocks also increase the mortality and reduce reproduction of large
vertebrates such as hake (Guevara-Carrasco and Lleonart, 2008), whales and seabirds (Essington et al.,
2015). Impacts on these organisms affect other non-fishing sectors that are dependent on EBUS, such as
whale-watching in the California Current EBUS, and generally degrade their intrinsic values.

Overall, EBUS have been changing with intensification of wind that drives the upwelling, leading to changes
in water temperature and other ocean biogeochemistry (high confidence); however, the direction and
magnitude of observed changes vary between and within EBUS with mechanisms explaining these
variations not fully understood yet. Moreover, the high natural variability of EBUS and their insufficient
representation by global climate models renders attribution to climate change having low confidence. Given
the high sensitivity of the coupled human-natural EBUS to oceanographic changes, the future sustainable
delivery of key ecosystem services from EBUS is at risk under climate change; those that are most at risk in
the 21st century include fisheries (high confidence), aquaculture (medium confidence), coastal tourism (low
confidence) and climate regulation (low confidence). For vulnerable human communities with a strong
dependence on EBUS services and low adaptive capacity, such as those along the Canary Current system,
unmitigated climate change effects on EBUS (complicated by other non-climatic stresses such as social
unrest) have a high risk of altering the development pathways of these vulnerable societies (high confidence,
Box 5.2, Figure 1).
**Box 5.2. Figure 1:** Projected climate hazards, sensitivity and adaptive capacity of the coupled natural-human EBUS:


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5.4.1.2 Regulating, supportive and cultural services

Climate change is threatening other key ecosystem services from the pelagic open ocean including carbon storage and other aspects of climate regulation, tourism, and biodiversity in general for culture and supporting ecosystem functions for other services (Martin et al., 2016).

The open ocean is estimated to have taken up 32 ± 5 PgC over the period 1994–2007 (Section 5.2.2.3) (IPCC, 2007; IPCC, 2014) thus making the biggest contribution in carbon sequestration relative to coastal and deep sea ecosystems. Assuming a carbon price of US$ 7 mt⁻¹ (World Bank, 2014), the value of the uptake of 32 PgC could be worth over USD200 billion. Net ocean global carbon uptake by the ocean projected by CMIP5 earth system models for 2080 is between 1.0–5.5 PgC yr⁻¹ under RCP2.6 and 8.5, with considerable variability between models (Lovenduski et al., 2016). The upper range of the projected changes exceeds the current variability of global ocean carbon uptake from observations that varies between 1.0–2.5 PgC yr⁻¹ between 2000 and 2012. Because of saturation effects, ocean carbon uptake is projected to decrease even under increased atmospheric CO₂ under RCP2.6. Thus, higher CO₂ emission scenarios will further accelerate the carbon concentration in the atmosphere and thus intensity of climate change. In addition, changes in upwelling patterns will speed up the shift to microclimate-loss in California under climate change, consequently affecting agriculture and water resources (Li et al., 2014).

Recreation and tourism in the open ocean are popular in many regions. For example, total expenditure on whale watching operations worldwide is estimated to be USD 413 million, employing 5762 people (Cisneros-Montemayor et al., 2010). Sport fishing on large pelagic species such as tuna, billfish and sharks also contribute to value and livelihood (Gallagher et al., 2017). These large pelagic species have been observed to respond to warming through shift in distribution towards higher latitude (see Section 5.2.4), impacting fisheries catches (see Section 5.3.1.2.1). Such impacts are expected to affect the potential catch rate and fishing locations of recreational fishing, and consequently the recreational fishing experience; however, observations and projections of such impacts are not available (low confidence).
The open ocean pelagic ecosystem also supports nutritional health and culture of coastal indigenous communities through migratory species. Particularly, many marine species that are important to coastal indigenous communities migrate between coastal waters and the open ocean, such as tunas and sea turtles (Harrison et al., 2018). These coastal communities access the open ocean biodiversity when they migrate to coastal waters. Coastal indigenous communities have higher than the average level of seafood consumption (Cisneros-Montemayor et al., 2016). Marine life is also strongly linked to many traditional cultures. For example, sea turtles for Fijians are both subsistence food and have strong association to hierarchical obligations among the chiefs and clans (Morgan, 2007). The challenges faced by indigenous coastal communities in adjusting their culture in response to climate change are also particularly high. Thus, climate change impacts on these species in the open ocean (5.2.2) are expected to have implications for the nutritional health and culture of coastal dependent communities such as indigenous people (5.4.3).

Overall, climate change impacts on a broad range of ecosystem services have been observed and/or projected for the open ocean pelagic ecosystems (high confidence). Particularly, the observed and projected impacts on carbon sequestration will intensify future climate change from greenhouse gas emission (high confidence). Also, coastal communities will be affected by climate impacts on open ocean biodiversity through migratory species that straddle between the high seas and coastal waters (high confidence). Specifically, the cultural values of these animals to indigenous coastal communities are particularly at risk to climate change (medium confidence).

### 5.4.1.2 Deep Sea Floor Ecosystems

Deep-ocean processes and organisms (Table 5.4) generate many types of provisioning, regulating, supporting and cultural ecosystem services (Figure 5.13) (Armstrong et al., 2012; Thurber et al., 2014), most of which are poorly studied and rarely quantified. The stakeholders for the deep ocean, which covers 54% of the surface area of the planet and most of the habitable volume, are all the people on the earth. The deep sea holds a wealth of living and non-living resources and provides services that are required to sustain humans for centuries to come. Understanding and assessing the impacts of climate change on deep-sea ecosystem services is particularly challenging as the deep sea is vast, remote and only partially explored. Many deep-sea habitats are unfamiliar and the functions of organisms are poorly known.

Increasing demands on deep-sea resources such as fish/shellfish, oil and gas and seabed minerals are creating the need to identify, understand and valuate the services provided by vulnerable deep-sea ecosystems and raising interest in ecosystem services in the context of environmental and financial management (Le et al., 2017). In shallow water, there is an appreciation that ecosystem-based approaches can contribute to climate solutions as well as biodiversity and conservation goals (Wüstemann et al., 2017), but similar efforts for the deep ocean are just beginning, despite its major role in climate mitigation through heat and CO₂ uptake. New approaches such as the Ecosystem Principles Approach are being developed or applied to draw the link between the ecological and socio-economic dimensions of the deep sea (Table 5.4) (Jobstvogt et al., 2014). When Willingness to Pay (WTP) methods were applied to cold-water corals (CWCs) the possibility that CWCs play an important role as habitat for fish was the single most important variable to explain respondents' WTP for CWC protection (Aanesen et al., 2015). There is a need for advanced thinking about ecosystem services in the deep ocean, going beyond intrinsic and instrumental values to incorporate relational values (Eudaimonia) that collectively acknowledge human associations with nature in multiple forms across different cultural practices (Chan et al., 2016).

### Table 5.4: Examples of detected or projected climate change impacts on key deep-sea floor ecosystems. Yellow – low confidence; orange – medium confidence; Mode of attribution: *theoretical inference; + model projection; # field observation  ³laboratory experiment; $ paleo record

<table>
<thead>
<tr>
<th>Ecosystem services</th>
<th>Canyon</th>
<th>Seamounts</th>
<th>Seeps and vents</th>
<th>Cold water coral and sponge reefs</th>
<th>Abyssal Plains</th>
<th>Continental Slopes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Provisioning</td>
<td>Reduced food provision due to increased stratification, loss of organic matter export</td>
<td>Reduced food provision due to loss of fish/shellfish</td>
<td>Unknown effect: species reliant on methane</td>
<td>Loss/dissolution of nursery habitat could translate into loss of fish</td>
<td>Limited food provision from this setting</td>
<td>Transient increased catch may occur as a result of</td>
</tr>
</tbody>
</table>

Do Not Cite, Quote or Distribute 5-82

Total pages: 224
<table>
<thead>
<tr>
<th>organic matter export and benthic production (+)</th>
<th>and benthic production (+)</th>
<th>seeps could benefit if warming induces additional methane release ($); loss of chemosynthetic foundation species (mussels, tubeworms) expected on upwelling margins as OMZs expand (#)</th>
<th>production and fisheries (+)</th>
<th>habitat compression above expanding oxygen minimum zones. Catches of hypoxia tolerant species could increase (tuna crabs, giant squid) and intolerant species could decrease (rockfish) or redistribute (#,+)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Genetic resources and chemical compounds for biotechnological, pharmaceutical or industrial applications</td>
<td>No assessment</td>
<td>No assessment</td>
<td>Loss of resources associated with foundation species (*)</td>
<td>Habitat loss could cause loss of genetic resources (*)</td>
</tr>
<tr>
<td>Regulating</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Carbon sequestration and storage</td>
<td>Reduced carbon sequestration and storage (#,+)</td>
<td>Reduced carbon sequestration and storage (#,+)</td>
<td>Limited effect (*)</td>
<td>Increased dissolution and reduced C sequestration (*,+</td>
</tr>
<tr>
<td>Waste absorption and detoxification</td>
<td>No assessment</td>
<td>Unknown effect; could interact with physical disturbance from mining (*)</td>
<td>Unknown effect; could interact with physical disturbance from mining (*)</td>
<td>No assessment</td>
</tr>
<tr>
<td>Disease and invasive species</td>
<td>No assessment</td>
<td>Unknown effect; could interact with physical disturbance from mining (*)</td>
<td>Unknown effect; could interact with physical disturbance from mining (*)</td>
<td>No assessment</td>
</tr>
<tr>
<td>Supporting (intermediate)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habitat and trophic support</td>
<td>Habitat loss, reduced biodiversity, loss of</td>
<td>Habitat loss, reduced biodiversity, loss of</td>
<td>Some commercial species affiliate</td>
<td>Habitat loss inducing loss of nursery grounds, Unknown. Potential loss of VMEs such xenophyophor</td>
</tr>
<tr>
<td>Nutrient cycling</td>
<td>Reduced via stratification, OM export and seafloor biomass (+)</td>
<td>Reduced via stratification, OM export and seafloor biomass (+)</td>
<td>No assessment</td>
<td>Reduced via stratification, OM export and seafloor biomass (+)</td>
</tr>
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<tr>
<td>Chemosynthetic primary production</td>
<td>No assessment</td>
<td>No assessment</td>
<td>No assessment</td>
<td>Productuion could expand at methane seeps if warming increases gas hydrate dissociation (*)</td>
</tr>
<tr>
<td>Resilience and resistance</td>
<td>Reduced via decline in food supply, warming, deoxygenation, acidification (*)</td>
<td>Reduced via decline in food supply, warming, deoxygenation, acidification (*)</td>
<td>Reduced via deoxygenation, effects on planktonic larvae in surface waters (*).</td>
<td>Reduced via warming, deoxygenation, acidification (*)</td>
</tr>
<tr>
<td>Cultural</td>
<td>Aesthetic and spiritual</td>
<td>No assessment</td>
<td>No assessment</td>
<td>No assessment</td>
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<td>Bequest and existence</td>
<td>No assessment</td>
<td>No assessment</td>
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<td></td>
<td>Scientific and educational</td>
<td>No assessment</td>
<td>No assessment</td>
<td>No assessment</td>
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</tbody>
</table>

5.4.1.2.1 Provisioning services

Living marine resources derived from deep sea organisms include fisheries, genetic resources that yield pharmaceuticals, industrial agents, and biomaterials, as well as non-living resources such as oil and gas, gas hydrates or minerals. Deep-sea organisms living near hydrothermal vents offer promising genetic resources for the development of novel enzymes to be used in varying pharmaceutical and industrial processes, the full potential of this enzyme market is valued at a minimum of $50 billion per year (Arico and Salpin, 2005; Leary et al., 2009). Fishing activities have expanded into deep water (Watson and Morato, 2013; Thurber et al., 2014; Victorero et al., 2018), facilitated by sophisticated technology, greater fleet efficiency and power and a desire for new resources (Kleisner et al., 2015). Thus the deep-sea fish that were once considered supporting services (i.e., as part of biodiversity) or part of the carbon pump are now valued as a provisioning service, despite the fact that their slow growth and longevity make them an unsustainable fishery resource (Norse et al., 2012). A number of deep-sea fishy species, such as the Antarctic toothfish (*Dissostichus mawsoni*), yellowtail flounder (*Limanda ferruginea*) and Golden redfish (*Sebastes norvegicus*) are highly vulnerable to climate change by virtue of their large body size and narrow thermal tolerance (Cheung et al.,
2018c; Levin, 2018). The North Atlantic, Indo-Pacific, West Africa and South Pacific regions host the most vulnerable deep-sea species (Cheung et al., 2018c; Levin, 2018). Of equal or greater concern is the fact that climate change will exacerbate physical disturbance from bottom trawling, reducing habitat complexity and resilience, affecting the provision of other ecosystem services (Levin and Le Bris, 2015; Sweetman et al., 2017). The movement to manage fish stocks collectively using an ecosystem-based approach has led to development of indicators of fish community change (survey-based mean trophic level, proportion of predatory fish, and mean lifespan (Kleisner et al., 2015), as well as indicators of supporting habitat status. These could be applied to deep-sea fish assemblages in the context of climate change for some of the better-studied ecosystems.

The effects of warming and ocean deoxygenation on deep sea organisms and ecosystems (Section 5.2.3) affect the quality and quantity of deepwater fisheries catches through similar pathways as in the open ocean pelagic ecosystems (medium confidence). Warming may reduce body size (medium confidence, Section 5.2.3) and lead to range shifts of exploited vertebrate and invertebrate species to deeper depths and higher latitude (habitat compression) (high confidence), with effects on availability, accessibility, costs of fishing and transport (medium confidence) (Cheung and Pauly, 2016; Lam et al., 2016). Habitat compression above oxyclines can increase catchability but may lead to overfishing or increased vulnerability to natural predators (Prince et al., 2010; Stramma et al., 2011). Ocean deoxygenation is leading to expansion of oxygen minimum zones in the tropics and subtropics (Stramma et al., 2008; Stramma et al., 2010). On the US West coast, near-bottom oxygen concentration is positively associated with both the biomass of commercially harvested fish species (Keller et al., 2010) and catch per unit effort (Keller et al., 2015). Deoxygenation is predicted to reduce available rockfish habitat (McClatchie et al., 2010). Crabs (Cancer magister) are also likely to be negatively impacted by expanded hypoxia (Gallo and Levin, 2016). In Peru, India, and Pakistan, the catch per unit of effort (CPUE) of demersal fisheries decrease when bottom oxygen levels decline (Banse, 1968; Rosenberg et al., 1983), and some species such as hake, octopus and scallops only appear on the Peru margin during oxygenation events (associated with El Niño) (Arntz et al., 2006). Beyond direct effects, deoxygenation can reduce the diversity (Sperling et al., 2016), size and abundance (Levin, 2003) of infaunal prey, as well as suppress benthic-pelagic trophic interactions. However, hypoxia-tolerant species such as tuna crabs (Pleuroncodes planipes) (Pineda et al., 2016; Seibel et al., 2018), Humboldt squid (Dosidicus gigas) (Stewart et al., 2014) and the pink urchin Strongylocentrotus fragilis (Sato et al., 2017) appear to be expanding their ranges (medium confidence) and could offer new fishery opportunities (Sato et al., 2018). As OMZs expand on margins, waters also become more acidic (Paulmier et al., 2011), with potential effects on calcifying taxa (see Section 5.2.4.4) (medium confidence).

5.4.1.2.2 Regulating, supporting and cultural services

The large areas and long residence times over which regulating services are provided in the deep sea mean that they are massive (high confidence), although the processes are often far removed from the humans that receive them (Thurber et al., 2014). The deep ocean ecosystem service that is most directly linked to climate is the sequestration of carbon, both through storage in the deep-sea pool and through long-term burial in sediments (high confidence). A net flow of approximately 1.6 GtC yr\(^{-1}\) from the atmosphere into the deep sea, of which 0.2 GtC is buried, represents a value of USD108 billion yr\(^{-1}\) if valued at a conservative USD19 per tCO\(_2\)-eq (which is USD62 per tC assuming all the carbon is in the form of CO\(_2\)) (Armstrong et al., 2010) (medium confidence). The value of carbon export to the deep ocean was estimated at USD12.9 billion yr\(^{-1}\) (using average European C market prices) for the Eastern Tropical Pacific covering 21 million km\(^2\) with waters of 12 nations + international commons (Martin et al., 2016) (medium confidence). Only one study has examined the economic cost of climate driven changes in carbon fluxes, focusing on the North Atlantic. Declines in C flux at 1000 m in the North Atlantic, estimated between 27–43% by 2100 depending on RCP scenario, were valued at USD170–300 billion in mitigation costs and 23–401 billion in adaptation costs over the 21st century (Barange et al., 2017) (low confidence). Carbon sequestration by the biological pump over the Mediterranean Basin was estimated at 281 million Euros per year (=USD 29 billion over the same period) (Melaku Canu et al., 2015) (low confidence). Uncertainties in these values derive from uncertainties in the ecosystem models (mechanisms, scenario pathways) and in carbon price estimates and discounting rates (Her Majesty’s Treasury, 2011; van den Bergh and Botzen, 2015). Other forms of carbon sequestration in the deep sea merit valuation, such as the removal of methane released from the beneath the seabed through microbial anaerobic oxidation and the sequestration in carbonate as well as aerobic methane oxidation by free-living microbes and symbionts (in mussels, sponges, ciliates) (Marlow et al., 2014;
Thurber et al., 2014). These services could become increasingly important as warming accelerates methane release (low confidence).

Cycling of nitrogen, silica, phosphorus, hydrogen and sulfur also occur in the deep sea, mediated by nitrogen fixation, denitrification, nitrification and anaerobic ammonium oxidation in deep waters and sediments (Marlow et al., 2014) (high confidence). Warming, acidification and deoxygenation will influence nitrogen sources, sinks and transformations (low confidence), but the extent of change and the value of these services have yet to be quantified.

Option value (the value of maintaining a resource, i.e., function, due to uncertainty regarding supply or demand) and bequest value (the value of maintaining a resource for availability to future generations) can be associated with all deep-sea ecosystem structures and functions discussed in Section 5.2.2.4 (high confidence). Particularly, many of the potential values and services from deep-sea ecosystems are currently not realized, but are increasingly recognized to be crucial for human wellbeing (Aanesen et al., 2015). The importance of these services in the deep ocean will increase as climate change degrades resources and services in shallow water (medium confidence), and can motivate further conservation of ecosystem services. Climate change-induced degradation of deep-sea ecosystems and biodiversity could also reduce the availability of such option value.

Supporting services such as primary and secondary production, habitat provision for feeding, spawning or nursery grounds, refugia, and biodiversity are all essential functions that ultimately contribute to the provisioning, regulating or cultural services (Haines-Young and Potschin, 2013; Bopp et al., 2017). Vertically migrating fish, which carry organic matter and carbon into deep-water perform functions that support carbon sequestration, fisheries and more. Habitat support functions in the deep sea are well recognized for canyon and seamount environments as these highly productive systems are often targeted by fishers. Coral and sponge assemblages provide food, refuge and nursery grounds for key fishery species, and host a wealth of biodiversity (Buhl-Mortensen et al., 2010), much as in shallow water systems. Deep-water coral habitats, especially the non-living matrix, are highly vulnerable to ocean acidification (as shallow reefs; Section 5.2.3.3) and thus the supporting services they provide are at risk (low confidence). Seeps and vents serve as nursery grounds for elasmobranchs (Treude et al., 2011) and cephalopods (Drazen et al., 2003) and as aggregation sites for adults of a surprising number of commercial species (Sellanes et al., 2008; Grupe et al., 2015), while sea pen and octocoral gardens provide nursery habitat for fish embryos and larvae (Baillon et al., 2012). The continuing discovery of new faunal associations with deep-sea habitats referenced in the previous sentence suggest that we most likely have yet to uncover many of the habitat support services provided by the deep sea (high confidence).

Biodiversity, which is understood to be high in the deep sea but remains mostly undescribed (Ramirez-Llodra et al., 2010), has existence value itself. It has optional values for the potential use of genetic resources to generate products for human uses such as pharmaceuticals industrial agents, cosmetics, biofuels or biomaterials (Skropeta and Wei, 2014; Harden-Davies, 2017). Deep sea biodiversity also contributes significantly to overall ecosystem functioning that supports other services as well. Ocean deoxygenation will reduce biodiversity of fauna at intermediate depths (see Section 5.2.4.3) but valuation of deep-sea biodiversity is in its infancy and translation of climate-induced loss into services has yet to occur. The greatest effect of biodiversity loss is likely to be loss of support functions. Nearly all deep-sea supporting services (e.g., secondary production, nutrient remineralization, trophic support, habitat provision, nursery and spawning grounds) that support fisheries production and climate regulation (Thurber et al., 2014; Bopp et al., 2017) will be affected by changes in temperature, oxygen, pH and POC flux, as discussed in (Section 5.2.4) on impacts on ecosystems.

5.4.1.3 Coastal Ecosystems

Coastal and shelf sea ecosystems, including, saltmarshes, mangroves, coral reefs, and seagrass beds are some of the most valuable, yet heavily used and threatened natural systems globally, affecting the ecosystem goods and services they deliver (Worm et al., 2006; Blanchard et al., 2017) (see Section 5.2.2). Coastal ecosystems are in close proximity to human populations, with one third of the world’s population living in coastal zones. Thus, coastal ecosystems are important to human society and contribute large benefits to support our well-being. For example, a study estimated based on total economic valuation that the values of
coastal ecosystems are $352,249/ha for coral reefs, and $193,843/ha for tidal marshes and mangroves (Pendleton et al., 2016), although the values are not static and have wide uncertainties (Hilmi et al., 2013b). Coastal ecosystems such as coral reef are also providing ecosystem services to half a billion of people, worldwide (Hughes et al., 2012). In Southeast Asia the total number employed in fisheries and aquaculture is estimated at over 20 million people and the net annual economic benefits of healthy coral reefs in the region to shoreline protection, fisheries, and tourism are estimated to range between $23,000 to $270,000 per km² of reef (Whittingham et al., 2003; Burke et al., 2006). However, the deterioration of coastal ecosystems due to human activities is intense and increasing; 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and 29% of seagrasses worldwide have been lost or degraded over the last 50–100 years (Valiela et al., 2001; Duarte et al., 2005; Spalding, 2010; Giri et al., 2011). The main activities that contribute to such deterioration are coastal development, over-exploitation, pollution and climate change (Section 5.2.2.2). Over the 20th century, losses of coastal ecosystems such as mangroves, seagrass and saltmarshes were estimated to be 25–50% of their area (McLeod et al., 2011), with a recent loss rate of approximately 0.5–3.0% per year (Pendleton et al., 2012b). At such rates, 30–40% of tidal marshes and seagrasses (IPCC, 2007) and nearly 100% of mangroves (Duke et al., 2007) could be lost in the next 100 years. Globally, 61% of reefs are classified as threatened with 100% of reefs expected to be threatened by 2050 if steps are not taken to reduce local pressure and reduce the emission of greenhouse gases. Quantitative attribution of the relative contribution of climatic and non-climatic human stressors observed changes in coastal ecosystems is difficult because of the interaction between these stressors and the lack of long-term datasets (Burke et al., 2011).

Global loss and degradation of coastal ecosystems is known to affect at least three critical ecosystem services (Worm et al., 2006; Inniss et al., 2017). Reefs subject to land-based pollution showed a reduction of 30-50% in diversity at a 3m depth, and a 40–60% reduction in diversity at a 10 m depth compared to unpolluted reefs (Cesar et al., 2003a). A meta-analysis of global datasets suggest that the loss of coastal biodiversity impacts the number of viable fisheries (33% decline); the provision of nursery functions (69% decline); and filtering services provided by coastal wetlands (63% decline) (Worm et al., 2006). The loss of biodiversity, ecosystem functions, and coastal vegetation in coastal ecosystems have also contributed to impacts on their recreational and cultural values, biological invasions, carbon capture and sequestration, declining water quality, and decreased shoreline protection (Braatz et al., 2007; Cochard et al., 2008; Koch et al., 2009). More recent studies provide further evidence to the significant relationship between coastal biodiversity and ecosystem functions that support these services (Gamfeldt et al., 2015). Thus, human activities and their consequences, including climate change, are substantially degrading all coastal ecosystem services that are important to human wellbeing (very high confidence). For example, loss of coral reefs may cost around USD 1 trillion globally (Hoegh-Guldberg, 2015). This sub-section focuses on assessing observed and projected impacts of climate change on coastal ecosystem functions and services.

5.4.1.3.1 Provisioning services Coastal and shelf seas ecosystems provide the largest contribution to global seafood production from capture fisheries and aquaculture (FAO, 2016). These ecosystems provide nurseries, breeding, spawning and hatching sites and through transportation of organic matter to the marine environment, thus providing nutrients for marine and estuarine fauna (Lee et al., 2015). Based on global fisheries catch data from the Sea Around Us project (Pauly and Zeller, 2016), in the 2000s, 67 - 74 million MT of fishes and invertebrates are caught in waters with depth less than 200 m per year. This amounts to >60% of total fisheries catches while marine aquaculture is almost exclusively produced from coastal and shelf waters. Most of these catches are supported by coastal habitat such as coral reefs (Pratchett et al., 2014), seagrass bed (de la Torre-Castro et al., 2014) and mangroves (Carraquilla-Henao and Juanes, 2017). For example, coral reef fishes are an important nutrient source to many coastal human communities worldwide (Cinner, 2014; Micheli et al., 2014; Hoegh-Guldberg et al., 2017). In the Caribbean, coral reef associated fisheries provide net annual revenues estimated at over $300 million while providing a source of protein for millions and employment for hundreds of thousands of full- and part-time workers (Burke et al., 2004).
a. Mean projection under RCP8.5
b. Projection range under RCP8.5
c. Mean projection under RCP2.6
d. Projection range under RCP2.6
e. Maximum catch potential
f. Species turnover (composition change)

Figure 5.15: Projections of changes in maximum fisheries catch potential by Exclusive Economic Zones by 2050 (average between 2041 – 2060) relative to 2000 (1991 – 2010): A. mean change between fisheries and marine ecosystem models (Dynamic Bioclimatic Envelope Model and Dynamic Size-based Foodweb model) under RCP8.5; B. the range (difference) in projections between the two fisheries and marine ecosystem models; C. mean change among the two models under RCP2.6; D. the range in projections between the two models (Cheung et al., 2018a); and scaling between projected global atmospheric warming (relative to 1950-1961) and (E) changes in maximum fisheries catch potential and (F) species turnover projected using the Dynamic Bioclimatic Envelope Model and outputs from three CMIP5 Earth System Models (Cheung et al., 2016b).

The impacts of ocean warming and changes in net primary production on invertebrates and fishes are affecting species composition and biomass production in coastal and shelf sea ecosystems with regional differences in the magnitude and direction of impacts (likely, see Sections 5.3.2, 5.3.3, Figure 5.15). Temperature and net primary production explain regional (Halpern et al., 2008; McOwen et al., 2015) and temporal (Britten et al., 2016) differences in fisheries catches in contemporary shelf seas. Trends of recent (during the 2000s period) reported fisheries catches in majority of the coastal and shelf seas are significantly related to temperature (indicated by SST) and primary production (indicated by Chlorophyll a concentration). Recent catch trends in some areas such as shelf seas around Australia and New Zealand are mainly related to changes in fishing effort. Analysis of historical catch records since AR5 continue to support a change in species composition of fisheries catches that is related to warming, with increased dominance of warm-water associated species in the coastal and shelf seas regions (Kesk and Pauly, 2014; Tsikliras et al., 2014; Maharaj et al., 2018) (high confidence).
Loss of critical coastal habitats is a major driver of loss of coastal fisheries productivity. Loss of coral reef habitats driven by climate change (see Section 5.3.3) and the subsequent declines in the physical reef structures have marked effects on the abundance of coral reef fishes and invertebrates, thus impacting ecosystem functions such as resilience and stability, as well as fisheries productivity (Prather et al., 2013; Pratchett et al., 2014) (high confidence). For mangrove forest, spatial comparison between Indo-West Pacific and Atlantic eastern Pacific suggest that higher floral and faunal diversity in the former is correlated with a greater range of species exploited for fuel, timber and crustaceans (Huxham et al., 2017). Global meta-analysis of mangrove-fishery linkages show a significant relationship between mangrove area and fishery catches (Carrasquilla-Henao and Juanes, 2017). For example, one hectare of mangrove has been valued at $37,500 per year when used as a nursery for commercial fishes that will later recruit into adjacent fisheries (Sala et al., 2013). Thus, if climate change reduces the area and/or quality of mangrove forest (see Section 5.2.2), potential fisheries catches will also decrease (medium confidence).

An ensemble of simulations from multiple models driven by projected changes in temperature, net primary production (and in some models, oxygen, salinity and sea ice extent) from CMIP5 Earth System Models projects a decrease in maximum catch potential by 2.8–8.7% and 2.8–9.1% under RCP2.6 by 2050 and 2095 relative to 2000 respectively (FAO, 2018). By contrast, the decrease was larger, 7.0–11.8% and 16.2–25.5% by 2050 and 2095 relative to 2000, respectively, under RCP8.5. These impacts will result in a loss of potential catches of 3.4 million tonnes and changes of 6.4% of the exploited species per degree Celsius atmospheric warming relative to 1951–1960 level (Cheung et al., 2016b) (medium confidence) (Figure 5.15). The interactions between temperature, net primary production and transfer efficiency of energy across the foodweb are projected to amplify these trends, producing changes that may exceed 50% in some regions by the end of the 21st century under a high-emissions scenarios (Stock et al., 2017). Climate change impacts on biological productivity of fish stocks have been shown to interact with fishing (Pershing et al., 2015; Britten et al., 2016) resulting in decline in a decline fish stocks that further reduces fisheries productivity (medium confidence).

Regionally, tropical and shallower waters are projected to have the largest decrease in maximum catch potential by 2100 relative to the 2000s levels, while catch potential in deeper water of the shelf seas is projected to increase because of a shift in distribution towards deeper, cooler waters (Section 5.2.3; Figure 5.15). As a result of changes in species distribution ranges, coastal fisheries resources are also projected to shift geographically towards higher latitude or deeper waters (Figure 5.15). For example, around the Pacific Islands, with projected changes in SST, decreases in surface dissolved oxygen and NPP by ≥3°C, ≥0.01 ml L⁻¹, 0.5 g m⁻² d⁻¹ by 2100 under RCP8.5, more than 50% of exploited fishes and invertebrates are projected to become locally-extinct in many regions; these factors contribute 74% of the area to be projected to experience a loss in catch potential of more than 50%. The area of large projected catch loss is projected to be halved under RCP2.6 (Asch et al., 2017). In West Africa, one model predicts that by 2050, due to climate change the annual landed value of marine fisheries will decline by 21% equating to an annual loss of $311 million and a 50% loss in fishery related employment (FAO, 2016). Fisheries in vulnerable ecosystems such as coral reef are expected to be particularly impacted by climate change. Healthy reefs in the Caribbean can support a maximum sustained yield (MSY) of 4 tons of fish per km² while degraded reefs were estimated to support a MSY between 0.7 and 2.9 tons per km² (Whalley, 2011). Globally, within 50 years the effects of temperature induced coral bleaching alone is estimated to costs $105 billion (Cesar et al., 2003b).

Marine aquaculture (i.e., mariculture; marine and brackish) exerts a profound influence as part of global food systems (high confidence), will impact about 27.6 million tonnes to total aquaculture production in 2016 (FAO, 2016), making a large contribution to food and nutrition security (Béné et al., 2015). Climate change and ocean acidification will impact the sustainability of mariculture production (high confidence). Most mariculture activities take place mainly in sheltered areas of the coastal zone and includes habitats such as estuaries, salt marshes, mud flats and more recently the open ocean. A study on the growth of farmed Atlantic salmon, Cobia and seabream evaluated the effect of future climate trends on optimal growth, concluding that climate change will decrease the species growth potential in many ocean areas where temperature will increase to levels that is outside the thermal tolerance range of the species (Klinger et al., 2017). Consequently, the decrease in growth could translate into a decrease in the general productivity of the sector. Disease is another major threat that may be exacerbated by extreme weather events and other ocean changes associated with climate change (low confidence). Karvonen et al. (2010) show that increasing water temperature is associated with higher prevalence of the parasite Ichthyophthirius multifiliis and bacteria.
Flavobacterium columnare that can cause disease and mortality of farmed fishes such as salmon. Changes due to climate change in global fisheries abundance and catch of forage fish, such as anchovy and sardine (Merino et al., 2012; Barange et al., 2014) (Sections 5.2.3, Box 5.2) will further impact mariculture productivity as fishmeal and fish oil production is reduced with consequences for the supply of aqua feeds (medium confidence).

Besides fisheries, coastal ecosystems also provide a range of other products that people use including timber, firewood and local medicine (Kairo et al., 2008; Spalding, 2010). The loss of biodiversity contributed by climate change in coastal habitats (Section 5.2.2) results in reduction in biomass productivity as well as other important biological resources. Although quantitative attribution and projection of such losses in these non-fisheries provisional services are limited (low confidence), such impacts are expected to cause substantial impacts on these ecosystem services under ‘business-as-usual’ climate change in the 21st century (high confidence).

5.4.1.3.2 Regulating, supporting and cultural services

Climate change affects coastal ecosystem services in addition to fisheries that connect directly to human well-being, health, livelihoods and survival (Cinner et al., 2016). Coastal ecosystems benefit human communities by protecting them from flooding and storm surges and thus preventing loss of life, damages to housing, infrastructure and by preventing saltwater intrusion (Rao et al., 2015). Coastal ecosystems are also an important sink of carbon and play a significant role in carbon sequestration and thus climate regulation. The biogenic structures of coastal ecosystems offer habitats that support rich biodiversity and biological productivity within and outside the ecosystems, and subsequently supporting wide range of provisional and cultural services.

Mangroves, coral reefs and saltmarshes worldwide regulate the impacts of storm and protect coastline from erosion (Rao et al., 2015) and thus their losses or degradation under climate change (Sections 5.3.3.2, 5.3.3.6) increase the risk of impacts on the security of coastal communities (Perry et al., 2018). The roots of mangrove plants help to hold the sediment in place (Orth et al., 2006) while coral reef reduce wave energy on average by 97% (Ferrario et al., 2014) thus substantially protecting coastal populations from natural hazards. Mangrove forests protect inland communities and freshwater resources from saltwater intrusion during storms, and they protect near shore settlements from erosion and reduce flooding. Coastal communities experienced greater damage and higher mortality rates from many types of natural disasters when mangroves had been removed, and the value of these damages ranges widely (Rao et al., 2015). Reduction in storm impacts regulation by coastal ecosystems will also indirectly affect other services as well; for example, the loss of fishing days because of reduction in the number of days with sea conditions that are suitable for fishing is projected to reduce recreational fishing by 10-18% by 2050 in the Southern USA relative to present day under RCP8.5 (Bowker et al., 2014). These regulating services are particularly important under climate change because coastal ecosystems help buffer the impacts of sea level rise and wave action (Duarte et al., 2013) (Section 5.5.2.2).

Carbon sequestration by coastal ecosystems is a key regulatory service that is vulnerable to climate change (Table 5.5). Carbon is biologically sequestered in coastal sediments, commonly known as ‘blue carbon’ (see Section 5.5.1). Atmospheric carbon is captured by marine plants and stored, primarily in the soil and sediment of coastal ecosystems such as mangroves, salt marshes, and seagrasses (Pendleton et al., 2012b). Carbon storage capacity in the top meter of soil is estimated at 280 Mg C ha⁻¹ for mangroves, 250 Mg C ha⁻¹ for salt marshes, and 140 Mg C ha⁻¹ for seagrass meadows, equivalent to 1,030 Mg CO₂eq ha⁻¹ for mangroves, 920 Mg CO₂eq ha⁻¹ for salt marshes, and 520 Mg CO₂eq ha⁻¹ for seagrass bed. The mean carbon storage is 1,494, 951 and 607 Mg CO₂eq ha⁻¹ for mangroves, salt marshes and seagrass beds, respectively. The carbon sequestration ecosystem service of mangroves has been valued at $190 per ha per year and $220 per ha per year for seagrasses (Trégarot et al., 2017). Carbon burial rates are also affected by variability in hydrological regimes, salinity, nutrient status, and management status. Long-term rates of carbon accumulation in sediments of salt marshes, mangroves, and seagrasses range from 18–1713 g C m⁻² yr⁻¹. Overall, McLeod et al. (2011) estimated blue carbon uptake at about 0.2 Gt C yr⁻¹.

Table 5.5: Estimates of carbon released by loss and degradation of coastal ecosystems globally and associated economic impact Note: 1pg = 1 billion metric tons. To obtain values per km², multiply by 100 (Pendleton et al., 2012a).
Given the substantial contribution of coastal blue carbon to carbon budget, loss of these coastal ecosystems because of climate change (Section 5.3.3) will also directly impact such service. When coastal ecosystems are lost or degraded the huge stocks of blue carbon in the soils are exposed and released as CO$_2$ into the atmosphere contributing to global warming. Current rates of loss of blue carbon ecosystems may result in 0.15–1.02 billion tons of CO$_2$ released annually. These emissions are equivalent to 3–19% of those from deforestation globally (Table 5.5), and result in economic damages of USD6–42 billion annually. For example, current releases from mangrove deforestation are estimated at 0.02–0.12 GtC yr$^{-1}$, accounting for up to 10% of global deforestation emissions (Donato et al., 2011). Tidal wetlands play disproportionately important roles in coastal carbon budgets, forming critical linkages between rivers, estuaries, and oceans (Najjar et al., 2018). As a result of tidal flooding, saltmarsh soil does not dry out and high levels of carbon can accumulate under anaerobic conditions (Figure 5.16). This is coupled with generally low rates of methane emission, which is strongly limited in saline marshes (medium confidence) (Poffenbarger et al., 2011; Abella Perez et al., 2016). The carbon burial rate of salt marshes is comparable to mangroves due to high belowground productivity (218 ± 24 gC m$^{-2}$ yr$^{-1}$) (McLeod et al., 2011). Despite their importance for coastal protection and biodiversity, the global area of saltmarshes has only recently been properly quantified, at around 550 ha, in 99 countries (Mcown et al., 2017b). Climate change and the temperature-driven displacement of saltmarsh plants by mangrove trees in the mangrove-saltmarsh ecosystem are predicted to increase carbon sequestration in coastal wetlands (Mgonigal et al., 2016) (medium confidence).

Recent expansion of mangroves into saltmarshes are likely to have large impacts on the structure, function and service provisioning of coastal wetlands (Kelleway et al., 2017b). Although mangrove encroachment may increase nutrient storage and improve storm protection (Sheng and Zou, 2017), the declines in saltmarshes will reduce habitat availability for fauna requiring open vegetation structure, and consequently the recreational and cultural activities associated with this fauna (Kelleway et al., 2017b). The impact on provisional services such as fisheries productivity and cultural services is projected to be site-specific and dependent on the species involved (Kelleway et al., 2017b).
Global change impacts dramatically reduce coral reef ecosystems goods and services because of the deterioration of biodiversity and ecosystem functions (Elliff and Silva, 2017; Hoegh-Guldberg et al., 2017) (high confidence). Ecosystem services provided by reefs have different sensitivities to a range of future climate change scenarios (Rogers, 2015) and to the combinations of local stressors and climate change (Elliff and Silva, 2017). The loss of live coral results in decrease in biodiversity and abundance of coral reef specialized species while generalist species that are less dependent on coral reefs are potentially less sensitive to reef degradation (Pratchett et al., 2014) (high confidence). The cluster of drivers that are associated with climate change profiles (primarily global warming, ocean acidification, and sea level rise) increase the risk of marine regime shifts (Chen et al., 2011; Leadley et al., 2014b; Rocha et al., 2015), with different biodiversity impacts and downstream reduction in coral reef services (high confidence). Thus, global change impacts will inevitably alter reef biodiversity, resulting in new reef assemblages that consequently degrade critical ecosystem functions and services. Although it is difficult to predict how the loss of particular species or several species will affect ecosystem services (Bellwood et al., 2006; Balmford et al., 2011; Hoegh-Guldberg et al., 2017), it is broadly predicted that novel reefs assemblages under climate change would not necessarily deliver the same ecosystem services such as fisheries, tourism and coastal protection that were provided by the past and current coral reefs (Pandolfi et al., 2011; Dornelas et al., 2014; Graham et al., 2014; Rinkevich, 2014; Rinkevich, 2015a) (high confidence).

The value of global coral reefs under various climate change scenarios range by an order of magnitude, 3.95–23.78 US$ billion annually (Hicks et al., 2015; Schuhmann and Mahon, 2015; Pendleton et al., 2016), with 23% reduction in coral reef values for each 0.6% diminution in coral cover projected under a scenario of 1% increase in carbon dioxide concentration (Chen et al., 2015) (medium confidence)—a level that will be achieved in the next decade under all RCPs (Seneviratne et al., 2016). These include commercial and recreational values as well as the valuation of supporting and regulatory services (Chen et al., 2015). In a

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**Figure 5.16:** Biogeomorphic feedbacks involving plant biomass, sediment accretion and inundation that control the response of vegetated coastal ecosystems to rising sea levels. (A) Under high rate of soil formation, plants are able to offset sea level rise and may produce a negative feedback by increasing the uptake of atmospheric CO₂. In addition, belowground root production contributes to the formation of new soils and consolidates the floor line. (B) Under low rate of soil formation, and when sea level rise exceeds critical thresholds, plants become severely stressed by inundation leading to less organic accretion, belowground subsidence and decay and a positive feedback of sea level rise by the net CO₂ outgassing. This figure does not consider landward movements, controlled by topography and human land-use.
recent study by Deloitte Access Economics, Australia’s Great Barrier Reef contributed $29 billion through tourism, $24 billion through non-use values, and $3 billion to recreation activities while also supporting 64,000 jobs (O’Mahoney et al., 2017). Globally, around 30% of the world’s reefs are of value in the tourism sector, with a total value estimated at nearly US$36 billion, or over 9% of all coastal tourism value in the world’s coral reef countries (Spalding et al., 2017). In the Caribbean 2.4 million divers per year contributed $2.1 billion to the region’s economy (Burke et al., 2004). Further, mangroves in Southeast Asia have been estimated to provide $10,821 per hectare per year in storm protection services (Sala et al., 2013). It should also be taken into consideration that in many instances the values for environmental services are estimated in a simplified way, thus not reflecting their real values (Rinkevich, 2015b). Moreover, most of the services provided by any specific ecosystem are usually not considered when employing conventional macro-economic indicators (such as the Gross Domestic Product) (TEEB, 2010), since many of the goods and services are not traded in markets.

Another important class of reef services is coral reef tourism marked with a global value of US$ 36 billion per year (Spalding et al., 2017). This is a reef service highly impacted by global changes (high confidence). A direct relationship between coral cover and recreational use (a subsector in the tourism industry) can be drawn (while varied between sites) revealing that all types of tourist activities are highly vulnerable to climate change impacts on the coral reefs (Biggs et al., 2015; Lane et al., 2015). All above ecosystem services in the coral reefs can be significantly impacted from the increased frequency and intensity of extreme climatic events that have emerged from global warming. These impacts are not easy to model, since current impact assessment models are largely based on gradual warming trends (Wernberg et al., 2012).

Overall, climate change impacts on coastal ecosystems are impacting all ecosystem services offered by coastal ecosystems (high confidence). Particularly, loss and degradation of coastal ecosystems under climate change impacts the availability and quality of habitats of fish and invertebrates populations (very high confidence), consequently reducing the diversity and productivity of fisheries resources that support commercial, subsistence and recreational fishing (high confidence). However, the level of impacts varies regionally, with tropical coastal ecosystems being the most at risk to such impacts (high confidence) relative to those in mid-latitude regions (medium confidence). Loss coastal ecosystems, particularly coral reef, mangroves and saltmarshes, as a result of sea level rises and increased extreme events increases the exposure of coastal communities to storms, erosion and saltwater intrusion because of the loss of their coastal protection services (high confidence) and contributes to the loss of carbon sink (medium confidence). The climate change-induced degradation of these regulating services that are important for moderating climate impacts represent a positive feedback that further exacerbates the risk of climate change (medium confidence). Intrinsic values and cultural importance of coastal ecosystems, such as indigenous culture, recreational fishing and tourism, that are dependent on their biodiversity and ecosystem functions as well as the regulating and supportive services provided by these ecosystems are also at risk to climate change impacts (high confidence).

### 5.4.2 Climate Risk, Vulnerability and Exposure of Human Communities and their Wellbeing

Human communities are heavily dependent on the oceans through the goods and services that marine ecosystems offer (Section 5.4.1) (Hiilm et al., 2015a). A recent report estimated that the total asset from these ocean-based economic sectors is at least US$24 trillion, generating hundreds of millions of jobs (WWF 2015). As climate change is impacting marine biodiversity and ecosystem services (Section 5.3.1), various dimensions of benefits that contribute to sustaining human communities and their wellbeing will also be affected. However, detection and attribution of climate change impacts on the wellbeing of human communities are confounded by wide-ranging non-climatic factors that strongly affect the exposure, sensitivity and adaptive capacity of human communities to climate impacts.

Therefore, the assessment in this section are based on studies that provide diverse types of information from quantitative modelling to qualitative studies using expert opinion, local experience and Indigenous knowledge. Projection and assessment of risk and vulnerabilities are also strongly dependent on assumptions about future social-economic development. Specifically, the risk of climate impacts on human communities are based on assessments on the level of dependence of ecosystem services, and the risks of climate impacts of these ecosystem services (Section 5.4.1). The Shared Socio-economic Pathways (SSPs) is referred to for assumption of future societal development where appropriate (O’Neill et al., 2014).
This assessment sub-divides the linkages between ecosystem services and human communities and their well-being into the three pillars of sustainable development, as laid out in the 1987 report from the UN and introduced by the World Business Council for Sustainable Development. These three pillars are social and cultural (Section 5.4.2.1), economic (Section 5.4.2.2) and environmental (Section 5.4.2.3), encompassing the materials, health (including physical and psychological), and other more subjective aspects of well-being (Table 5.6). This section also assesses the implications of climate change impacts on sustainable development through the ocean contextualized by evaluating the climate effects on the possibility of achieving the United Nations’ Sustainable Development Goals (SDGs) (Section 5.4.2.4).

**Table 5.6:** The social, economic and environmental dimensions of sustainable development and examples of their linkages to marine ecosystem services.

<table>
<thead>
<tr>
<th>Pillars of sustainable development</th>
<th>Dimensions assessed</th>
<th>Key linkages to ecosystem services</th>
<th>High-dependent communities</th>
<th>Key hazards</th>
</tr>
</thead>
<tbody>
<tr>
<td>Social and cultural</td>
<td>Human diseases (5.4.2.1), pollution (5.4.2.2), security and conflicts (5.4.2.3), Indigenous culture (5.4.2.4) and education (5.4.2.5)</td>
<td>Fisheries and aquaculture, medicinal resources, water supply, distribution and prevalence of water-borne pathogens, Indigenous knowledge and local knowledge, spiritual experience and sense of place</td>
<td>Coastal Indigenous communities and the developing countries with lack of access to education, health care and infrastructure</td>
<td>Increased exposure to pathogens, disputes in resources sharing and competitions, loss of habitats with important intrinsic and educational values, increased inequality between and within political entities</td>
</tr>
<tr>
<td>Economic</td>
<td>Fisheries (5.4.2.2.1), food security (5.4.2.2.2), tourism (5.4.2.2.3)</td>
<td>Fisheries and aquaculture, coastal protection, climate regulation, tourism</td>
<td>Fishing, aquaculture and ecotourism sectors, coastal Indigenous communities and the global south, vulnerable population of mega-cities, poorest strata of society</td>
<td>Decrease in fisheries and aquaculture profits, decrease in the quantity and quality of seafood supply, degradation or loss of tourist attributions</td>
</tr>
<tr>
<td>Environments</td>
<td>Aesthetic aspects, property values, development of coastal cities (5.4.2.3.1), energy and mining (5.4.2.3.2)</td>
<td>Coastal protection, climate regulation, intrinsic values of biodiversity</td>
<td>Coastal Indigenous people coastal populations in poverty</td>
<td>Increased exposure to storm, flooding and coastal erosion, loss of biodiversity and habitats.</td>
</tr>
</tbody>
</table>

### 5.4.2.1 Social and Cultural Dimensions

The sustainable development paradigm includes domains at the intersection between the societal and the natural realms, focusing on human engagement with and within nature, including the built environment. Further, the linkages between the health of the environment, ecosystem health and human health are considered. This sub-section will focus on different dimensions of risks on human health and security as a result of climate change effects on the oceans, biodiversity and ecosystem services. Specifically, the assessment focuses on ecosystem-mediated impacts of climate change, impacts of pollutants other than CO₂ and climate impacts on human security and conflicts. The cultural aspects concern indigenous culture and education. Climate change will also impact cultural dimensions.

#### 5.4.2.1.1 Ecosystem-mediated impacts of climate change on water-borne diseases

The risk of human diseases caused by water-borne pathogens such as *Vibrio* bacteria and Harmful Algal Blooms (HABs), are closely related to environmental conditions (*medium agreement, medium evidence*). This assessment focuses on the detection, attribution and projection of risk of diseases caused by *Vibrio*.
bacteria, which is one of the most common water-borne human diseases and are considered as major health
concerns (Kirk et al., 2015). Assessment of HABs is described in Box 5.3.

Around twelve out of over 100 species of *Vibrio* have caused human infections, mainly via exposure to
seawater or consumption of raw or undercooked seafood with the bacteria (Austin, 2005). Common diseases
caused by *Vibrio* include necrotizing wound infections, primary septicemia, ear infections, and seafood-
related food-poisoning. Elderly and other individuals with suppressed immune system are particularly
vulnerable to such infection (Daniels, 2011). Globally, *Vibrio* is causing substantial epidemiological risk.
For example, *Vibrio cholera* (causing Cholera) is estimated to be responsible for around 760,000 and
650,000 cases of human illness and death in the world in 2010 (Kirk et al., 2015). Naturally, the
nontoxigenic *Vibrio* species are prevalent worldwide in warm, nutrient-rich and low salinity coastal waters.
AR5 concludes that warming, excessive nutrient and seawater inundation due to sea level rise are projected
to exacerbate the expansion and threat of cholera (*medium confidence*, AR5 WGII Ch. 6).

New evidence since AR5 continues to support the linkages between climate change and increased risk of
diseases caused by *Vibrio* bacteria. Further analysis of time-series data confirms an increase in the
prevalence of *Vibrio* that is related to warming (Baker-Austin et al., 2017). For example, analyzing the
Continuous Plankton Recorder dataset (Section 5.2.3) with new molecular techniques detected a significant
increase in *Vibrio* abundance in the North Sea since the 1980s that is related to sea surface warming
(Vezzulli et al., 2016). Recent review also suggests a poleward expansion of *Vibrio* pathogens to areas with
no previous incidence, particularly in mid- to high- latitude regions; such expansion is suggested to be partly
driven by climate change, particularly in areas where rapid warming is taking place (Baker-Austin et al.,
2013; Baker-Austin et al., 2017). Simultaneously, extreme weather events such as flooding and tropical
cyclones are linked to increased incidences of *Vibrio*-related diseases; Such increases in diseases are
suggested to be caused by the increased exposure of human populations to the pathogens during these
extreme events (Baker-Austin et al., 2017). Particularly, coastal floods are associated with intrusion of
seawater and wastewater into fresh water, leading to water shortage and risks of water-borne diarrheal
disease (Kim et al., 2014). Floods are also related to displacement of human populations, which is known to
increase vulnerability of populations and to contribute to the spread of infectious diseases. Therefore, recent
ocean warming has led to the increase prevalence and expansion of *Vibrio* pathogens (*very high confidence*).

Extreme weather events increase the exposure of human populations to *Vibrio*, potentially contributing to the
increased incidence of *Vibrio* diseases (*medium confidence*). However, the global surveillance of the disease
in human population is limited and the nontoxigenic Vibrio infection is rarely notifiable to the health
agencies, leading worldwide to limited understanding of the disease epidemiology and risks, and their links
to climate change (*low confidence*).

Climate change is expected to exacerbate the risk of *Vibrio* diseases particularly in coastal regions.
Coastal areas that are projected to experience warming, changes in precipitation and increases in nutrient
inputs would have an increase in prevalence of *Vibrio* pathogens. For example, in the coastal Baltic Sea,
*Vibrio* is expected to grow faster in coastal areas where, under climate change projections, waters are
expected to be warmer, increase in salinity due to reduced precipitation and have higher chlorophyll
concentrations (Escobar et al., 2015) resulting in an elevated risk of *Vibrio* infections (Semenza et al., 2017).
Areas that are projected to have elevated frequency of coastal floods from storm surges and sea level rise in
response to future climate change (Section 5.2.2) will also increase the risk of *Vibrio* disease as a result of
human exposure to the pathogen (*medium confidence*). However, uncertainty in the social and economic
developments shaping vulnerabilities of populations makes estimates of absolute mortality highly uncertain
(Lloyd et al., 2016).

New evidence since AR5 also confirms that the increased risks of other non-Vibrio ocean-related diseases
are attributable to climate change. Increases in the incidence of Harmful Algal Blooms (HABs) are related to
climate change and are projected to increase in the 21st century, and thus increasing the risk of associated
diseases (see Box 5.3). For example, increased reported calls to the poison control centre in the United States
that were correlated with warmer sea surface temperature and increased tropical storm frequency. Changes
in ocean wind patterns may also have also affected the dispersal of viruses of infectious diseases through the
transport of dust particles (Wu et al., 2016).
Overall, increases in the prevalence of pathogens for some water-borne human diseases that have large public health impacts are related to climate change (high confidence), and their risks are projected to increase in the 21st century (medium confidence). However, health risks of these pathogens are strongly influenced by economic and social factors, and thus posing a major uncertainty for the projection of the future risk of these diseases. The risk of these diseases may also interact indirectly with other climate change impacts on marine biodiversity and ecosystems. For example, because of the loss of marine biodiversity due to climate change (Sections 5.2.3, 5.3.3), the chances to discover new bioactive compounds will decrease (Section 5.4.1). Many traditional medicines coming from the sea may also disappear. This can potentially have an impact on human health in general, and more specifically, the discovery for effective prevention and treatment for water-borne diseases (Malve, 2016) (low confidence).

[START BOX 5.3 HERE]

Box 5.3: Harmful Algal Blooms and Climate Change

Harmful algal blooms (HABs) are proliferations of phytoplankton species that have negative effects on marine environments and associated biota, with impacts ranging from water discoloration and foam accumulation, anoxia, contamination of seafood with toxins, disruption of food webs, and massive invertebrate and fish kills (Hallegraeff, 2010). Since the 1980s, the HABs have increasingly affect human society (Anderson et al., 2015). For example, the economic cost of HABs in the United States alone is estimated to be $100 million per year (in 2012 dollars). These costs mainly resulted from impacts on public health, commercial fisheries and recreation and tourism (Berdalet et al., 2017). In AR5 it was concluded that ‘harmful algal outbreaks are increasing their frequency and intensity stimulated by warming, nutrient fluctuations in upwelling areas, and coastal eutrophication (medium confidence); however, there is limited evidence and low confidence in how global climate change will affect HABs’ (AR5 WGII Ch. 5, 6).

Since AR5, more observational and experimental evidence allows for improved detection and attribution of HABs to climate change. The main hypotheses explaining the increased HAB-related impacts include increased human influences on the marine environment through nutrient enrichment, pollution, aquaculture, shipping, and climate change (Lassus et al., 2016). A Global HAB Status Report initiative by the International Oceanographic Commission and The United Nations Educational, Scientific and Cultural Organization (IOC UNESCO) and hosted within the International Oceanographic Date Exchange (IODE), has generated global datasets on HABs (Moestrup 2009; http://www.marinespecies.org/hab) to unify toxic species nomenclature and track their geographical prevalence. Emerging climate-driven responses identified after field and experimental work include range expansion of warm-water HAB-related phytoplankton (e.g., Gambierdiscus, producer of ciguatoxins that causes ciguatera fish poisoning, CFP) and contraction of cold-water species (medium confidence) (Tester et al., 2010; Rodriguez et al., 2017), regional rises in abundance and seasonal bloom window (medium confidence), and increased cellular toxins of key species (Alexandrium, Pseudo-nitzschia, Karlodinium) due to combined effects of warming and acidification; low confidence (Fu et al., 2012). Recent experiments with large mesocosms enclosing natural planktonic communities in subtropical waters, simulating CO2 emission scenarios between RCP2.6 and RCP8.5, found that acidification promoted the development of the toxic microalgae Vicicitus globosus, with severe negative impacts over other co-existing phytoplankton species and in general over the whole planktonic food web (Riebsell, in press). Furthermore, the implementation of precise techniques for the quantification and identification of novel phyco toxins and toxic species in underexplored areas of the Ocean (e.g., in the South Atlantic: Akselman et al., 2015; Tillmann et al., 2016; Guinder et al., 2018), and experiments to evaluate the production of toxins in the presence of grazers (Tammilehto et al., 2015; Xu and Kiorboe, 2018) and changing nutrient levels (Van de Waal et al., 2013; Brunson et al., 2018), provide evidence for the significant role of climatic and oceanographic drivers in inducing HABs and their degree of impact (medium confidence). For example, the biosynthesis of domoic acid by some Pseudo-nitzschia species is induced by combined phosphate limitation and high CO2 conditions (Brunson et al., 2018). The growth rate of the toxic dinoflagellates Alexandrium (producer of paralytic shellfish poisoning, PSP toxins) and Dinophysys (producer of okadaic acid) are enhanced by warmer conditions. Therefore, the biogeographic expansion of these dinoflagellates’s ecological niche has been observed in the North Atlantic and North Pacific and are related to increasing sea surface temperatures (Gobler et al., 2017). Similarly, in the Northern California Current, environmental conditions linked with warm phases of ENSO are associated with the production of
the neurotoxin domoic acid (DA) by species of *Pseudo-nitzschia* (McKibben et al., 2017), with devastating
effects on coastal ecosystems (McCabe et al., 2016). Finally, in the coastline of the Canary Islands, modelled
distribution of the harmful benthic cyanobacteria *Lyngbia majuscula* predicts expansion of the population
linked to changes in topographical and wave exposure. The development of dense mats of *Lyngbia* implies
high risk for local rocky and sandy ecosystems as well as seagrass meadows, as they affect the sedimentation
processes and the structure of benthic communities (Martín-García et al., 2014). Overall, these observed
changes in biogeography and phenology of HAB-related species are consistent with general responses of
phytoplankton to climate change (Section 5.2.2) (*medium confidence*), while physiological studies confirm
elevated cellular toxins production by HAB species under high CO$_2$ conditions and nutrient-limited
conditions (*medium confidence*) (Fu et al., 2012; Brunson et al., 2018).

Regional variations of the trends in the occurrences of HABs could be explained by the spatial differences in
key climate drivers of temperature, water column stratification, ocean acidification (increased CO$_2$) and
extreme weather events (e.g., precipitation) (Hallegraeff, 2010). Particularly, these algal phenomena are
common in coastal areas where the exposure of organisms to climate threats is more variable due to
interactive effects with human-induced environmental changes. Thus, the occurrences of HABs are largely
dependent on local-scale emergence of changing ocean conditions (Box 5.1).

HABs occurrences, their toxicity and risk on natural and human systems are expected to continue to increase
under climate change in the 21st century. Based on observations and simulation models, occurrence of HABs
is projected to continue with warming and rising CO$_2$ (*high confidence*) (Gilbert et al., 2014; Martín-García
et al., 2014; McCabe et al., 2016; Paerl et al., 2016; Gobler et al., 2017; McKibben et al., 2017; Rodríguez
et al., 2017; Riebsell, in press). The worldwide distribution of the key toxic species of *Alexandrium* and
*Pseudo-nitzschia* and the projected increase of anomalous warming events, suggest an intensification of the
HAB-related risk for coastal biodiversity and ecosystems services particularly those that are naturally
sensitive to the impacts of HABs. For instance, more severe risk is expected for estuarine organisms such as
oysters, shrimp and fishes exposed to both low pH and low dissolved oxygen, as these conditions make them
more vulnerable to toxins (*medium confidence*) (Gobler and Baumann, 2016). The increasing risk of
occurrences of HABs under climate change also elevates their risks on ecosystem services such as fisheries,
aquaculture and tourism as well as public health (Section 5.4.1, 5.4.2).

Overall, improved observational and experimental evidence and projection support increasing incidence and
intensity of HABs that could be attributed to climate change and its interactions with other human drivers
(*high confidence*). Global predictive capability is hindered by contradictory species and strain-specific
responses, lack of insights into evolutionary adaptation, long-term (>30 years) data streams and climate
variability at a regional scale. The greatest problems for human society will be caused by being unprepared
for range expansions or increase of biotoxins in poorly monitored areas. Therefore, action for adaptation and
mitigation relies on sustained monitoring programs at a local scale of phytoplankton at the species level, and
of chemical and physical conditions (Anderson et al., 2015). Early warning systems of potential harmful
outbreaks can alert resource managers and stakeholders and ameliorate the risk for human wellbeing. In
addition, the reduction of coastal pollution and eutrophication can mitigate the synergistic effects with
climate hazards and partially reduce the severity of the ecosystem damage produced by HABs, but whether
the reduction of anthropogenic stressors will be enough to adapt the risk of climate hazards on HABs
remains uncertain (Section 5.5.2).

5.4.2.1.2 Interactions between climate change and non-CO$_2$ pollutants

Climate change–contaminant interactions may alter the bioaccumulation and biomagnification of two
contaminant classes: the toxic and fat-soluble persistent organic pollutants (POPs), such as polychlorinated
biphenyls (PCBs), as well as the neurotoxic and protein-binding organic form of mercury, methylmercury
(MeHg) (Alava et al., 2017) (*medium confidence*). POPs are bioaccumulated by marine organisms and
biomagnified in food webs, reaching exposure concentrations that become harmful and toxic to populations
of apex predators such as marine mammals (Desforges et al., 2016) (Figure 5.17). Exposure to POPs can
lead to serious health effects including certain cancers (i.e., by PCBs and dioxins), birth defects, and
impairments to the immune and reproductive systems, as well as greater susceptibility to disease and
damages to the central and peripheral nervous systems (Lallas, 2001). MeHg is highly neurotoxic and
nephrotoxic and bioaccumulates and biomagnifies throughout the food web via dietary uptake (Fort et al.,
2015). Consumption of mercury-contaminated fish is linked to disease in humans, including the neurological
disorders due to MeHg poisoning (i.e., Minamata disease) (Ishikawa and Ikegaki, 1980; UNEP, 2013). Of
particular concern is the pollution risks influenced by climate change in the Arctic ecosystems and
indigenous communities because of the protracted bioamplification of POPs and MeHg with associated
long-term contamination of their traditional foods (Marques et al., 2010; Tirado et al., 2010; Alava et al.,
2017).

In the face of ocean warming and acidification, the exposure, persistence, bioaccumulative capacity and
toxic nature of these pollutants can be further exacerbated by climate-change pollutants interactions
(McKinney et al., 2015; Alava et al., 2017) (medium confidence). Scientific evidences for human and
ecotoxicological risk assessments highlight that climate change increases the exposure and bioaccumulation
of contaminants and thus the risk of impacts of POPs and MeHg on marine ecosystems and their dependent
human communities (Marques et al., 2010; Tirado et al., 2010; Alava et al., 2017) (high agreement, medium
evidence). For example, mercury methylation processes in aquatic environments is expected to be
exacerbated by ocean warming coupled with more acidic and anoxic sediments. Mercury and some metals
(e.g., Al, Cu, Fe, Pb, Zn) are generally more bioavailable in acidified aquatic habitats (Scheuhammer, 1991;
Celo et al., 2006; López et al., 2010; Roberts et al., 2013; de Orte et al., 2014). Increased global temperatures
may also increase the metabolic activity of bacteria which will augment the cycling and conversion of
mercury into MeHg (Macdonald and Loseto, 2010; Corbitt et al., 2011; Krabbenhoft and Sunderland, 2013).
Furthermore, increased melting of snow and ice from alpine ecosystems and mountains (Chapter 2) can also
increase the release of POPs and mercury from land-based sources into coastal ecosystems (Morrissey et al.,
2005). Ultimately, mercury concentrations in fish are inversely related to pH in acidic water, which would be
due to the higher solubility of mercury in acidic water and enhancement of mercury methylation rates at
lower pH (Lean, 2003) as well as increased mercury uptake under warming (Riget et al., 2010; McKinney et
al., 2015). Modelling projection suggest increased bioaccumulation of MeHg in the Faroe Islands region
under climate change, with an associated increase in human exposure to this metal from the consumption of
pilot whale (Globicephala melas) (Booth and Zeller, 2005).

Climate change alters the degree of human exposure to pollutants and the response of human populations to
this exposure (Balbus et al., 2013) (medium confidence). Changing human behaviour will also affect how
humans come into contact with contaminated air, water, and food in an era of climate change. Climate
change and regional climate variability may have an impact on the occurrence of food safety hazards,
including changes in transport pathways for contaminants, at various stages of the food chain, from primary
production to consumption (Marques et al., 2010; Tirado et al., 2010). While species inhabiting
industrialized areas may be most at risk for contaminants, and the vulnerability of the Arctic climate
suggests that the latter region may be particularly sensitive and susceptible to climate–contaminant
interactions (Chapter 3). In this context, contaminant concentrations in marine mammal species and wildlife
populations from the Arctic already exceed overall biological levels of concern for POP exposure, i.e., 1 mg
kg⁻¹ or ppm (Letcher et al., 2010). The impact of POPs on top predators such as marine mammals leads to
changes in their population density, which then propagate through the entire Arctic food web and ecosystem,
ultimately affecting global fisheries. This leads to potential health impacts for indigenous communities in a
warming Arctic, who rely strongly on traditional seafood and marine mammals (Alava et al., 2017).
Figure 5.17: The biophysical, eco-physiological and eco-toxicological pathways of impacts of climate change through climate change-induced pollutant sensitivity on contaminant bioaccumulation in marine food webs and effects on hierarchical levels of biological organization, with implications for public health and socio-economic dimensions. The urgent need of climate change and pollutant policies to address anthropogenic emissions of pollutants is also featured. Adapted from Booth and Zeller (2005).

The high exposure to climate-pollution risks is not limited to Arctic communities, but is also a concern for indigenous communities from other ocean and coastal regions of the world (i.e., temperate, sub-tropical and tropical regions) in general (medium confidence, medium evidence). Globally, average seafood consumption by coastal indigenous communities is 15 times that of average consumers (Cisneros-Montemayor et al., 2016). Thus, these communities are intimately connected with the ocean and rely strongly on the harvest and consumption of traditional seafood and fish products. As a result, these communities are also exposed to higher pollution risks and their potential exacerbation from climate change effects.

5.4.2.1.3 Human security and conflicts

Climate change may generally increase conflict risks (Gilmore, 2017) and there is emerging empirical evidence that relates climate change and conflicts to fisheries and oceans. Conflicts can be defined as events involving individuals or groups of individuals (organized or disorganized) in which disputes fail to be resolved or social orders are changed. This assessment identified risks to human communities and their wellbeing that emerge from climate change effects on the ocean through conflicts.

Climate change effects on ocean conditions (Section 5.2.2) can create or exacerbate conflicts over fisheries as a result of shifts in distribution of fish stocks and the fluctuations and decreases in their abundance and potential catches (Spijkers and Boonstra, 2017), as well as the movements of fishers (Belhabib et al., 2016; Pomeroy et al., 2016; Blasiak et al., 2017) (Section 5.4.1.1.1, 5.4.1.2.1, 5.4.1.3.1). Shifts in distribution of transboundary fish stocks under climate change alter the sharing of fisheries resources between countries (Miller and Russ, 2014). Under climate change, new fishing opportunities will be created when exploited...
Fish stocks shift their distribution into a country’s waters where they did not previously occur in sufficient abundance to support viable fisheries before (Pinsky et al., 2018). Such shifts in transboundary resources have occurred before and led to international disputes in some cases (Spijkers and Boonstra, 2017; Pinsky et al., 2018). For example, shifting transboundary stock distributions of Pacific salmons and Atlantic mackerel contributed to international conflicts in exploitation and setting catch quotas for these fisheries. Moreover, the number of new transboundary stocks occurring in exclusive economic zones was projected to be around 46 and 60 under RCP2.6 and RCP8.5, respectively, by 2060 relative to 1950-2014 (Pinsky et al., 2018).

However, most existing international fisheries governance legal framework does not have explicit provisions to directly account for these shifting fish distributions and abundance. Without a climate-considered governance framework, chances for disputes in the sharing and exploitation of these fisheries resources are higher (high confidence), consequently increasing the risk of overfishing (high confidence) and fractured international relationships, and political conflicts that could spill over into other, non-fishery areas of international politics (low confidence).

Extreme events associated with climate change, such as hurricanes and floods, can exacerbate tensions among ‘trapped populations’ unable to afford a move out of vulnerable areas (McGranahan et al., 2007; Adger et al., 2014). Climate change is projected to increase the risk of coastal communities to extreme events such as hurricanes and coastal flooding (Chapter 6). A substantial contribution of such risk is through increased exposure due to human populations settling in vulnerable areas, particularly the poorer or marginal groups in the society (Adger et al., 2014). These vulnerable populations have lower capacity to adapt through migration and the increased impacts of extreme events can exacerbate inequality within the society, and consequently, increase the risk of social unrest and conflicts (medium confidence). In addition, competing use of natural resources internationally may also increase the risk of these conflicts. For example, increased exploitation of marine resources such as fisheries sometimes offer a coping mechanism to communities affected by extreme events; however, opportunistic expansion of foreign fishing into impacted areas may impact fisheries and exacerbate social tensions (Belhabib et al., 2018) (low confidence). Overall, high vulnerability and lack of adaptive capacity to climate change impacts (including fisheries-dependent livelihoods, attachment to place, and pre-existing tensions) increase the risk of conflicts, including among fishery area users and authorities (Ndhlovu et al., 2017; Shaffril et al., 2017a; Spijkers and Boonstra, 2017).

Population movements associated with climate change impacts can result in conflicts in areas receiving displaced fishing communities (Reuveny, 2008). Negative inland climate effects can push some communities to coastal areas, exacerbating pressure on local fisheries and social tensions between long-term coastal residents and in-coming populations (Bohra-Mishra et al., 2014). Climate change impacts on rural communities may displace people towards cities, although such migration may not occur for low-income countries (Cattaneo and Peri, 2016), and while many cities are located in coastal areas the impact of outmigration to coastal fishing communities is not well documented. Also, the relationship between climate change, migration and conflicts in unstable urban and rural demographics may be weak and short-term (Fort et al., 2008) and affected from sampling and reporting biases (Adams et al., 2018). The impacts of these conflicts on communities and their wellbeing can include loss of livelihood and access to fish, forced displacement and community breakdown, as well as the physical and psychological harms of violence (medium confidence).

Other pathways of increased risk of conflicts under climate change include territorial claims and mal-adaptation policies. Sea level rise can also lead to conflicts over the redefinition of territorial claims between countries, especially in areas where large areas of the exclusive economic zones or the very existence of countries are at stake (Maas and Carius, 2012; Stoutenburg, 2015), or maritime sovereignty is frequently disputed (Nyman, 2013). Climate protection, exogenous adaptation policies, and marine conservation measures for coastal populations can induce social disruptions, exacerbate inequalities and delegitimize authorities, thereby increasing the risk of conflicts (Bennett and Dearden, 2014; Kamat, 2014; Sovacool, 2018).

In summary, climate change will increase the risk of conflicts within and between countries and regions. The increase in risk is a result of shifts in living marine resources abundance under climate change that results in dispute between countries or regions (medium confidence, low agreement) and lack of adaptive capacity of vulnerable or marginal human populations driving increases in exposure to marine extreme events, and consequently, increased inequality and social unrest (low confidence). These risks are further exacerbated.
Indigenous cultures

Indigenous communities can be found in all continents except Antarctica (UN, 2009). For Indigenous communities along the coast, recent estimates suggest that there are more than 1900 groups with around 27 million people across 87 countries (Cisneros-Montemayor et al., 2016). The cultures and languages of Indigenous peoples are responsible for much of the cultural diversity that continues to exist (UN, 2009).

Many of these communities exist on their traditional lands. The Declaration on the Rights of Indigenous Peoples outlines minimum standards ‘for the survival, dignity and well-being of the Indigenous peoples of the world’ (UNPFII, 2007), including the right of self-determination. This enables Indigenous peoples to ‘freely determine their political status and freely pursue their economic, social and cultural development’.

Particularly, a fundamental component of human well-being is culture, particularly Indigenous culture. However, marginalisation makes their lives, values and cultures vulnerable to changes in natural resources (Wessendorf and Garcia-Alix, 2009). At the same time, Indigenous values and culture are an important enabling component for sustainable development, particularly in offering different foundations for and perspectives of what sustainable development means (Watene and Yap, 2015). However, climate change is affecting different aspects of Indigenous culture that would also affect sustainable development (Cross-Chapter Box 3 in Chapter 1). This assessment particularly focuses on the perspectives for coastal Indigenous people in general. Specific aspects related to Arctic Indigenous people are discussed in SROCC Chapter 3.

Because Indigenous culture varies greatly between regions and the assessment requires specific context, therefore, we draw experience Indigenous culture under the changing ocean and marine ecosystems from the perspective of the Pacific.

The case of the Pacific is a good example for illustrating the effects of climate change on Indigenous knowledge, the transmission thereof, and ocean education. This is because in the Pacific the (climatic) changes are different in nature and refer to different time frames (see also Cross-Chapter Box 3 in Chapter 1). Geological and climatic history refers to time intervals of thousands or even millions of years. Traditional knowledge, on the other hand, relates to shorter periods ranging from several generations to a few centuries.

On the other hand, Indigenous knowledge, even if it has been transmitted for several generations, can be appreciated over shorter periods ranging from a few centuries to several generations. Thus, the adjustment of the transmission and of the network of Indigenous and local knowledge on the ocean, of the related perceptions and practice, implies a reworking of these knowledge systems where the individuals and the groups are actors in a narrative and historical construction (Roué, 2012; Alderson-Day et al., 2015). In this sense, the latest surveys conducted in Kiribati and the Tuamotu archipelago in French Polynesia (Tamatoa and Latouche, 2016) highlight the gap between the scientific discourse predicting unprecedented climate change and the local perceptions that consider it to be an ‘outside’ discourse in a context where science has often been considered as the ally of a historic colonial power (Tuihiwai, 2013). Nevertheless, climate change is already transforming the shape of seashores in many low islands. Some sandbank and rocks representing gods and mythological ancestors are disappearing (Camus, 2017). Other studies in Tuvalu show that during the past four decades, sea level rise and climate change have resulted in a net decrease in land area (Kench et al., 2018) affecting the mobility of people, residence patterns, overcrowding, and the structure of Indigenous knowledge and local knowledge in the sense that more mythological syncretism may be produced by the concentration of people in the safest places.

Thus, the great myths of foundation and the ocean that play a central role in these communities are re-elaborated, and the knowledge and practices are modified under climate change. From the perspective of these communities, these changes do not come about because of science prophesying a major climate change, but because the local populations observe that climate change induces change in their environment. For example, the scarcity of pelagic resources and lagoons, and the disappearance of a mythical sandbank at Tabiteua (Kiribati) altered the historical trajectory of a population and its relationship to the ocean forever (Camus, 2017).

In addition, in the Pacific as elsewhere, there is a shift in the transmission of local knowledge about the ocean, from a family monopoly to a plurality of actors including compulsory schooling. The fear of sea level rise and climate change encourage security measures and the grouping of local people to the safest places.

This plurality of modes of transmission and such concentration of the population not only contribute to the
erosion of local knowledge about the ocean (Bambridge and Le Meur, 2018), but has also almost totally impaired any opportunity for social rise for most people (Ali, 2016). It should be noted, however, that the traditional lunar calendars, which predict fishing and farming seasons by species and guide the choice of techniques, are still used (Bambridge and Le Meur, 2018), and that the validity of their prediction is often discussed and compared to the supposed effects of climate change.

This paradoxical situation, however, has encouraged Pacific peoples to embrace climate change not in the terms of science, but according to their cultural categories and patterns (low confidence). For example, many Polynesian and Micronesian societies have been involved in the construction of double canoes reminiscent of major exploration voyages. Today, they contribute to re-enactment of old sailing techniques while at every stage of their journey they strive for a respectful relationship between society, the ocean and the climate. In the same way, the last ten years have seen the introduction of ancestral techniques for the conservation of lagoon resources, called rahui in eastern Polynesia (D’Arcy and Bambridge, 2014), now extended to previously unknown scales. The Cook Islands protect their exclusive economic zone, with the establishment of a rahui, marae o te moana—the sacred temple of the ocean—over an area of 1.9 million km². French Polynesia has pledged to do the same on an area of 5.5 million km². This vast historical movement goes beyond a simple goal of adaptation to climate change. It is also a question of restoring an Oceanian identity to a space in which they consider themselves to be the historical guardians (Hilmi et al., 2016). Finally, ‘environmentally displaced’ Pacific Islanders are acquiring land in the Pacific regions bordering their areas (New Zealand, Australia, and the USA) to secure the future of next generations.

AR5 concluded that climate change will affect the harvests of marine species with spiritual and aesthetic importance to indigenous cultures. Recent research has further demonstrated that Indigenous fishing communities that depend on traditional marine resources for food and economic security are particularly vulnerable to climate change through a reduced capacity to conduct traditional harvests because of limited access to, or availability of, resources (Larsen et al., 2014; Weatherdon et al., 2016) (medium confidence). For instance, shellfish and traditional clam beds—which form an integral part of the culture, economy and diet of many Indigenous communities situated along the Pacific Coast of North America—may be affected detrimentally by increased SST, sea level rise, and changes in ocean chemistry and circulation patterns (Lynn et al., 2013). Indigenous people can also be highly dependent on various marine mammals that are vulnerable to climate change impacts. Any change in numbers, quality or access to these mammals (for example through die-off of walrus from St. Lawrence Island) increases the vulnerability of local Indigenous communities (Lynn et al., 2013) (medium confidence).

Overall, indigenous communities may be impacted by climate change because of their vulnerabilities and because of their reliance on the ocean for their livelihood and their cultural beliefs (medium agreement, medium evidence). Climate change will impact Indigenous communities through the displacement to new locations and the decrease in marine species. This will be a loss for the human cultural heritage (low to medium confidence).

5.4.2.1.5 Other cultural dimensions

Among the aspects of cultural wellbeing supported by the ocean, education based on scientific knowledge of the marine environments, and the economic and technical benefits generated by exploration and discovery play an important role. There are also aesthetic and inspirational values of marine biodiversity and ecosystems that are important to the psychological and spiritual well-being of people, including film, literature and art, and recreation (Pescaroli and Magni, 2015). The appreciation of scientific, artistic, spiritual, and health opportunities, as well as the appreciation of biodiversity, lifestyle and aesthetics, as well as intrinsic values, are other cultural dimensions becoming more widely acknowledged as key cultural dimensions potentially disturbed by climate change (Marshall et al., 2018). While climate change is impacting marine ecosystems (Sections 5.2.3, 5.3.3), their support to human well-being through the cultural dimensions could also be affected.

Education can increase knowledge and awareness of climate change impacts and the efficacy of their mitigation (Meadows, 2011). It can influence the extent to which stewardship activities are adopted (von Heland et al., 2014; Wynveen et al., 2015). It can also help to develop new networks between coastal people and environmental managers for the purposes of developing and implementing new adaptation strategies (Wynveen et al., 2015) Research suggests that a lack of education causes misconceptions about the scale of
climate change (temporally and spatially), and the causes of climate change (Tapsuwan and Rongrongmuang, 2015). Critical element to reducing vulnerability to climate change is to educate people that they are core to the Earth system and have a huge influence on the balance of the system. An important service of marine biodiversity and ecosystems is to support such education (Section 5.4.1). Thus, education can play a pivotal role in how climate change is perceived and experienced and marine biodiversity and ecosystems play an important role in this. At the same time, climate change impacts on marine ecosystems (Sections 5.2.3, 5.2.4) can affect the role of the ocean in support such public education.

Climate change will affect the perception of local communities about the ocean (low confidence). For many island nations, their deep ocean is hundreds to thousands of times larger than their land, and most of their natural capital lies in the deep sea. Some such as Rapa Nui (Easter Island) have a strong spiritual connection with this element. A reaction common to many low islands Pacific countries such as Kiribati, Cook island, Pascua Island and French Polynesia, is to create huge large marine protected areas as an adaptation measure to climate change (Bambridge and Le Meur, 2018). Marine Protected Areas also offer a cultural opportunity in impacted areas to constructively and collaboratively discuss adaptation plans and plans to increase resilience, whilst simultaneously reinforcing cultural identity and pride. This in return will affect the catch in the Pacific and the economy of long distant fleets for example.

Climate change threatens many cultural dimensions of lives and livelihoods (Adger et al., 2012) (low agreement, low confidence). People develop strong cultural ties and associate distinctive meanings with many natural places in the form of traditions, customs and ways of life (Marshall et al., 2018). Nature presents a place within which people can belong, whilst offering other places to visit and be inspired by. Nature provides a day-to-day sense of meaning through providing opportunities for intellectual engagement, shared recreation, economic well-being and support for livelihoods. The cultural opportunities provided by ecosystems are many, and include concepts such as attachment to place, heritage, rootedness, spirituality, tranquility, escape, togetherness, discovery, knowledge, health, judgement, aesthetics, recreation, education, community development, lifestyle, identity, and emotional/psychological well-being (Chan et al., 2012; Fish et al., 2016; Marshall et al., 2018). Ecosystems not only make human life possible, they also contribute to making life worth living (Costanza et al., 1997). Cultural losses or impacts associated with climate change on human well-being can be attributed as a factor contributing to mental health issues, such as depression (Fritze et al., 2008; Cunsolo Willox et al., 2015; Helm et al., 2018). However, recent review suggests that cultural aspects of climate change are largely under-represented in the literature (Fish et al., 2016). This may be because cultural impacts are largely ‘invisible’ (Turner et al., 2008) where losses associated with culture are less tangible than those associated with economic losses. Given the observed impacts and high risks of impacts from climate change on most coastal ecosystems where people are living and interacting closely with (Section 5.3.2), it could be postulated that the consequential climate impacts on the cultural values that marine ecosystems offer would elevate the risk of impacts on mental and psychological aspects of human-wellbeing (low agreement, low evidence).

The relationship between people and nature is continually changing, with corresponding and constantly evolving opportunities for cultural enrichment (low evidence, medium agreement). Climate change, however, is likely to alter the rate of environmental change to the extent that cultural adaptation may be unable to keep pace. This is because the culture that forms around a natural environment can be so integral to people’s lives that disassociation from that environment can induce a sense of disorientation and disempowerment (Fisher and Brown, 2014). For example, when a resource user such as a fisher, farmer, or forester is suddenly faced with the prospect that their resource-based occupation is no longer viable, they lose not only a means of earning an income but also an important part of their identity (Marshall et al., 2012; Tidball, 2012). Loss of identity can, in turn, have severe economic, psychological, and cultural impacts (Turner et al., 2008). Climate change can quickly alter the quality of, or access to, a natural resource through degradation or coastal inundation, where livelihoods and lifestyles are no longer able to be supported by the resource. When people are displaced from places that they value, there is strong evidence to suggest that their cultures are diminished, and in many cases endangered. There are no effective substitutions for, or adequate compensation for, lost sites of significance (Adger et al., 2012).

The vulnerability of human communities to cultural impacts depends on the nature of the cultural relationship with a climate sensitive resource and the extent to which culture can absorb or cope with such changes and adapt (low confidence). For example, Marshall et al. (2018) assessed the importance of identity,
pride, place, aesthetics, biodiversity, lifestyle, scientific value and wellbeing within the Great Barrier Reef region by 8,300 people across multiple cultural groups. These groups included indigenous and non-indigenous local residents, Australians (non-local), international and domestic tourists, tourism operators, and commercial fishers. They found that whilst some groups granted higher levels of importance to some values, all groups highly rated all (listed) cultural values, suggesting that at least these cultural values are critically associated with iconic ecosystems. Climate change impacts upon the Great Barrier Reef through increased temperatures, cyclones and sea level rise that cumulatively degrade the quality of the Reef, are liable to result in severe cultural impacts within these cultural groups.

Climate change affects differently the human cultures and well-being: it depends on the importance of the ocean in their life. The adaptive capacity of people to moderate or influence cultural impacts, and thereby reduce vulnerability to cultural impacts is also culturally determined (Cinner et al., 2018). The capacity to consider climate risks, share learnings and reorganize, as well as the capacity to mitigate the impacts of climate change reflect the cultural tendencies of groups of people. Contemporary responses to climate change have been labelled as barriers to adaptation given that they fail to address the cultural dimensions of climate risk, suggesting that considerable investment in the adaptive capacity of people to consider cultural impacts is critical (Adger et al., 2012). Vulnerability to climate change, and the adherence to the Sustainable Development Goals, will be correspondingly affected once cultural dimensions are carefully considered in strategic developments.

5.4.2.2 Economic Dimension

5.4.2.2.1 Monetary and material wealth generated from fisheries

Marine biodiversity and ecosystem services support the income and livelihood of coastal communities and marine-related economic sectors such as fisheries, aquaculture, forestry and tourism (World Ocean Assessment, Section 5.4.1). Marine fisheries and forestry are directly dependent on provisioning service (biomass production of fish stocks and mangroves). Marine aquaculture is partly dependent on capture fisheries production (as feed to supply animal protein and nutrients for the farmed animals) and regulatory service through provision of suitable farming habitat (nutrients, oxygen, current) (Oyinlola et al., 2018).

Global gross revenues from marine fisheries is around US$ 100 billion annually (Swartz et al., 2013). Capture fisheries sectors provide full-time and part-time jobs for an estimated 260 ± 6 million people, 22 ± 0.45 million of whom are small-scale fishers (Teh et al., 2011). Specifically, small-scale fisheries are important for the livelihood and viability of coastal communities worldwide (Chuenpagdee, 2011). The oyster industry in the Pacific Northwest of the United States has already lost nearly US$110 million in annual revenue and some 3,200 jobs due to ocean acidification. AR5 concludes with low confidence that climate change will lead to a global decrease in revenue with regional differences that are driven by spatial variations of climate impacts on and the flexibility and capacities of food production systems. It also highlights the high vulnerability of mollusk aquaculture to ocean acidification. This assessment examines the rapidly growing literature on the economics of climate change impacts of marine fisheries from global to regional scales.

Since AR5, global projections on the economics of climate change impacts on marine fisheries have incorporated and related to a broader range of social-economic factors. Driven by shifts in species distributions and maximum catch potential of fish stocks (Section 5.4.1) as well as projected changes in the prices of fisheries landings, marine fisheries revenues are projected to be negatively impacted in 89% of the world’s fishing countries under the RCP8.5 scenario in the 2050s relative to the current status, with projected global decreases of 10.4% ±4.2% and 7.1% ±3.5% under RCP8.5 and RCP2.6, respectively (Lam et al., 2016). The projected changes in the potential catch in a country’s Exclusive Economic Zones (EEZs) do not directly translate to the change in the fisheries revenues of a country because the catch composition change and vessels of some countries do not only fish in their own EEZs, but also in the high seas and other countries’ EEZs.

Climate change may affect the subsequent benefits of the indirect sectors (secondary and ancillary) that relate to fisheries, such as fish canning, processing industries and boat repair. Accounting for the subsequent impacts on the dependent communities leads to a projected decrease in seafood workers’ incomes of US$ 3.7 billion and an increase in total households’ seafood expenditures by the global population of US$6.3 billion annually under a 3.5°C atmospheric warming scenario relative to a 1.5°C warming scenario (Sumaila et al., in
Adaptation strategies to fisheries productivity changes and effective transboundary institutions can offset these losses; for the RCP6.0 scenario, adaptation is projected to lead to substantial higher global profits (154%), harvest (34%), and biomass (60%) in the future when compared to projections without adaptation. However, the total profit, harvest and biomass have negative impacts even with full adaptation scenario under the RCP8.5 scenario (Gaines et al., 2018). Climate change will adversely impact the monetary wealth generated from fisheries and such impacts will be higher under higher carbon emission scenarios (high confidence). Higher carbon emissions under ‘business-as-usual’ may also impose substantial limits to adaptation (medium confidence).

The implications of the effects of climate changes on the fisheries sector vary among different regions and countries (high confidence) (Hilmi et al., 2015a), with the most profound implications in the coastal low-income food deficit countries including small island countries, African countries and tropical Asian countries (Lam et al., 2016). For example, using geo-spatially referenced malnutrition and infant mortality data as a proxy for poverty, 15 developing countries are found to contain over 90 percent of the world’s low-elevation coastal zone rural poor (Barbier, 2015). These least developed countries usually rely heavily on fish and fisheries as a major source of animal proteins, nutritional needs, and income and job opportunities, and these countries have a limited availability of alternative livelihoods and sources of food. Therefore, negative impacts on the catch and total fisheries revenues obtained by these countries may have greater implications for jobs, economies, food and nutritional security than the impacts on high Human Development Index (HDI) countries (Srinivasan et al., 2010; Golden et al., 2016). These least developed countries are highly vulnerable to the impacts of climate change but have relatively low capacity to adapt (Allison et al., 2009; Blasiak et al., 2017). Climate change may also worsen the non-climate-related socio-economic shocks and stresses, and hence is an obstacle to poverty reduction (Hallegraeff et al., 2015). Hence, climate change adaptation and mitigation strategies must be human-centered, with an emphasis on the need for poverty eradication, food security, empowerment of local stakeholders, and addressing power imbalances and inequity in fishing and aquaculture communities. (Dasgupta et al., 2017; Kalikoski et al., 2018a) investigate the possible impacts of climate change on coastal communities in Bangladesh using the salinity tolerance range of a variety of fish species they consume, and find that areas with poor populations that will lose species are about six times more prevalent than areas gaining species.

The vulnerability of coastal communities is increased because of their dependence on some specific ecosystems, which are very sensitive to climate change. For example, pelagic and coastal fisheries provide vital services to Pacific island communities. Oceanic fisheries in the region rely on four species of tuna: skipjack (Katsuwonus pelamis), yellowfin (Thunnus albacares), albacore (Thunnus alalunga) and bigeye (Thunnus obesus). In 2014, total regional offshore fishery production amounted to 1,823,561mt with license fees from foreign fleets generating US$349,335,572 across all 22 PICTs (Pacific island countries and communities; Gillett (2009)). In Tokelau and Kiribati for example, in 2014, the sale of foreign fishing vessel access and licensing fees generated 98% and 84% of government revenue, respectively (Bell et al., 2018a); they represented 52% and 40% of government revenue in Tuvalu and FSM for that same year (Johnson and Williams, 2017). While accounting for much less volume than oceanic fisheries, coastal fisheries have a high unit value, provide important local livelihoods, and are critical to local food security and nutrition, representing between 40–90% of dietary animal protein (Bell et al., 2009; Bell et al., 2011). Overall, fisheries and aquaculture production in the region amounted to about 2 million mt in 2014, worth an estimated USD3.2 billion (Gillett, 2009). Fishing contributions to GDP – which do not include post-harvest activities – were re-estimated in a consistent manner by Gillett (2009) and found to account for between 7% and up to 30% of GDP for Solomon Islands, FSM, Kiribati and Marshall Islands. This heavy economic reliance on fisheries make Pacific island communities particularly sensitive to the effects of climate change.

Amongst different ecosystems, coral reefs are an important asset from an economic point of view. Climate change impacts to coral reefs and other fish habitats, as well as to targeted fish and invertebrate species themselves, are likely to reduce harvests from small-scale, coastal fisheries by up to 20% by 2050, and by up to 50% by 2100, under the RCP8.5 emissions scenario (Bell et al., 2018a). Changes to the ocean and to coral reefs are likely to reduce catches of reef fish by 20% by 2050. However, national governments lack the capacity to respond effectively to major events, as current patterns and projected changes require policy solutions and practices that are trans-sectoral and span levels of governance (Cashman and Nagdée, 2017). The net benefits accrued by the global economy of coral reefs reach US$30 billion yr⁻¹ from food security, coastal protection against natural hazards, tourism, financial incomes, sources for bio-prospecting, the
development of novel pharmaceuticals and other types of goods and services (Cesar et al., 2003b), without
even considering the capitalization of newly recognized reef economic assets and without the valuation of
environmental services (Rinkevich, 2015b) (high confidence). It should also be noted that ecosystem
services define up to 90% of poor rural communities' incomes, worldwide (Ring et al., 2010).

Within the fisheries sector, there may be gender differences in the level of vulnerability and risk of climate
change impacts on people’s income and livelihood. Women are heavily engaged in the post-harvesting
sectors, mainly involving selling and processing fish in the fish landing sites and local markets. Often it is
mostly women working in canning, processing and other ancillary economic activities related to marine
resources. Women are also involved in the bookkeeping operations and taking care of the home while the
fishermen are away. However, women in the post-harvest sector are often in a disadvantaged position
because of the marginalization of the fishing communities and their lack of access to credits and capacity
development. Thus, women may be disproportionately affected relative to men when climate change impacts
fisheries. Gender equity and empowering women's participation in climate change discourse and actions
have been included in the Gender Action Plan of the United Nations Framework Convention on Climate
Change (UNFCCC) (Kalikoski et al., 2018b).

Overall, climate change impacts on abundance, distribution and potential catches of fish stocks (see Section
5.3.1) are expected to reduce revenues and economic values of the global fisheries (medium agreement
medium evidence). These impacts on fisheries sectors will increase the risk of impacts on the income and
livelihood of people working on these economic sectors (medium confidence). However, these impacts have
not been well addressed in recent literature (Lacoue-Labarthe et al., 2016). More effort will be needed to
understand the effects of climate change on the dynamics of the fish prices and costs of fishing. Both of
these changes would lead to the adjustment of the fishing effort and hence intensify or lessen the
overcapacity issue. Studies have attempted to project how fishers may respond to changes in fish distribution
and abundance by incorporating different management systems (Haynie and Pfeiffer, 2012; Galbraith et al.,
2017). However, the impacts of climate change on the management effectiveness and trade practices is still
inadequately understood (Galbraith et al., 2017) and needs further exploration.

5.4.2.2 Impacts of climate change on food security

Around 140 million tonnes of fish and seafood per year are used for human consumption. Fish is a major
contributor to a healthy diet. It supplies proteins, healthy fatty acids, vitamins and other elements essential
for health such as iodine and selenium (medium agreement). In developing countries fish is often the only
affordable and relatively easily available source of animal protein (FAO, 2017). Since AR5, extensive new
literature provides additional evidence to support the assessment of the linkages between climate change and
food security.

Climate change impacts on marine ecosystem services will lead to a high risk of impacts on the nutritional
health of dependent communities with high seafood-dependence, particularly for Indigenous and other
coastal communities in the global south; through decreases in seafood availability (Hilmi et al., 2014) (low
confidence; Table 5.6, Figure 5.18). Marine ecosystems, through fisheries and aquaculture, support the
global population with proteins and vital micronutrients, such as iron, zinc, omega-3 fatty acids and vitamins
(Golden et al., 2016). In particular, fish are important sources of micronutrients, especially for some
populations that are largely dependent on fisheries for subsistence and livelihood. Deficiencies of some of
these micronutrients can increase the risk of perinatal and maternal mortality, contribute to growth
retardation, child mortality and cognitive impairments (Black et al., 2013). Climate change is projected to
reduce fisheries catch potential by the 2050s relative to the 2000s, particularly for regions where dependence
on micronutrients from fish is high (medium confidence, Section 5.3.1). The projected decrease in fish
supply is expected to exacerbate health risks over the next decades from a reduction in the intake of
micronutrients and fatty acids. In particular, low- and middle- income coastal communities around the
equator where large decrease in potential fisheries catches are projected (>20% by the 2050s relative to the
2000s under RCP8.5) are at a higher risk of such impacts on nutritional health (Golden et al., 2016) (medium
confidence; Figure 5.18). It has been estimated that 1.39 billion people are vulnerable to deficiencies of
micronutrients if fish catches continue to decline (Golden et al., 2016).

Decreases in fish supply under climate change will also contribute to a decrease in overall protein supply to
these coastal communities (Lam et al., 2012). Many areas with strong fish dependence and a high risk of
climate change impacts on fisheries production coincide with areas where land-based food production is projected to decrease due to climate change (Blanchard et al., 2017), further exacerbating the risk of climate change impacts on nutritional health of people living in these areas. Hilmi et al. (2017) conclude that climate change and ocean acidification will impact fisheries and aquaculture in the Mediterranean countries, but the impacts are difficult to evaluate because of uncertainties in the projected physical and ecological changes (Hilmi et al., 2013a). The distribution of tuna in the tropical Pacific Ocean will shift and abundance decline under climate change, because of projected increases in sea surface temperature, reduced oxygen, changes in the strength of the region’s major currents, and declines in primary production (Boyce et al., 2010; Lehodey et al., 2011). For yellowfin and skipjack tuna, countries in the eastern Pacific are likely to gain from such projected shifts, particularly if prices increase due to declines in catches elsewhere (Johnson and Williams, 2017). However, such shifts in tuna distribution would see countries in the west suffer from declining stock biomass (Bell et al., 2018b). Based on skipjack only, preliminary model simulations show changes in percentage contribution to PICTs’ GDP, to range between -0.4% and +6% by 2050, and from -2% to +2% by 2100 (Bell et al., 2013). The same study estimated changes in average percent contribution to government revenue to vary up to between -0.3% and +21% by 2050, and from -2% to +12% by 2100. The abundance of bigeye and south Pacific albacore is likely to decline overall (Lehodey et al., 2015), exceeding 60% in several EEZs by 2100 for bigeye (Bell et al., 2018a).

**Figure 5.18:** Projected changes in catch potential and the reliance of the human population on fish and their vulnerability of micronutrient malnutrition (Golden et al., 2016). The colour scale on land represents the proportion of fish micronutrient intake relative to the total animal-sourced food (ASF) for the country’s population; the scale on sea represents projected changes in maximum catch potential under RCP8.5 by 2100 relative to the 2000s.

Other non-climatic factors may interact with climate change effects on food resources, resulting in complex patterns of risk for vulnerable coastal communities to climate impacts on food security. For example, the shift from traditional nutritious wild caught seafood-based diets of coastal Indigenous communities, towards increased consumption of processed (and other) energy-dense foods that are high in fat, refined sugar, and sodium, due to social and economic changes (Kuhnlein and Receveur, 1996; Shannon, 2002; Charlton et al., 2016; Batal et al., 2017), may reduce communities’ exposure (Quinn et al., 2012) to climate change effects on fish stocks. Such changes, however, have important consequences on diet quality and nutritional status (Thaman, 1982; Luick et al., 2014), and have led to an increased prevalence of obesity, diabetes, and other
diet-related chronic diseases (Gracey, 2007; Sheikh et al., 2011) as well as the related decrease in access to culturally or religiously significant food items.

5.4.2.2.3 Monetary and material wealth generated from coastal and marine tourism sector

Tourism is one of the largest sectors in the global economy, with coastal tourism comprising a significant part of global revenue derived from the tourism sector. The development of the tourism sector has permitted economic gains for coastal countries and islands. For example, about 30% of world tourism is concentrated in the Mediterranean Basin (World Tourism Organisation data). The wealth in natural resources of the Mediterranean coastline include unique landscapes, climate favourable for tourism, important biodiversity which makes it a leading world tourist destination (Hilmi et al., 2015b). Climate change will lead to changes in the basic conditions for coastal tourism, including climatic parameters (air temperature, precipitations, wind speed, etc.), underwater parameters (SST, invasive species such as jellyfishes, etc.), and coastal hazards (erosion, marine flooding, etc.) (Safa and Hilmi, 2012). Such changes will affect the main components of the tourism industry—such as tourists’ preferences, tourism operators’ strategies, and the transportation market—and thus the tourism supply and demand balance at the global scale. The impacts will vary between ecosystems (high confidence).

Climate change impacts on coral reefs and other coastal ecosystems (Section 5.2.1) may affect the quality of the dive sites, beaches and the coast (Hilmi et al., 2012) (high confidence). Coral reefs and coastal ecosystems draw significant tourism each year, supporting recreational activities, such as diving and snorkeling, and supporting coastal communities and auxiliary sectors (Spalding et al., 2017). Reef tourism is increasing in popularity, with some parts of the world showing growth in the number of visitor in recent years, increasing the pressure to develop tourist facilities and accommodation. Beach recreation values are also expected to decline in response to narrower or vanished beaches resulting from sea level rise and erosion, with considerable losses for local economies (e.g., $1 billion loss between 2006 and 2080 in North Carolina (Whitehead et al., 2009; Nelson et al., 2013). As principal assets of coastal tourism, the loss of beaches could significantly impact local economies in subtropical and topical regions, and could incur additional costs associated with coastal protection and infrastructure repairs (DiSegni and Shechter, 2013). The impacts are exacerbated in SIDS because of the increasing demography which carries an increasing pressure on the reef resources mainly used for self-consumption. This trend is exacerbated by the disappearance of traditional ways of management of the reef resources (such as taboos and religious beliefs translating into informal temporary no-take zones) which were quite effective (Caillaud et al., 2004).

Climate change will cause global economic losses in tourism sector (medium agreement, medium evidence). The climate change projections show that coral reef coverage declines between 6.65% (under the RCP2.6 scenario) and as much as 27.58% by 2095 under RCP8.5, in which case a meta-analysis by Chen et al. (2015) estimates a coral reef recreational and tourism value loss ranging from US$1.88 to US$12.02 billion annually. These economic damage estimates only account for recreational and tourism loss, although there would be other losses in the services provided by coral reefs including coastal protection, fish habitats, and marine biodiversity. Cesar et al. (2003) estimate the total costs of coral bleaching over a 50-year time horizon with a 3% discount rate at over US$ 84 billion in net present value. For moderate bleaching, this number is US$ 20 billion. The tourism value is nearly US$ 40 billion in the ‘severe’ bleaching case, followed by fisheries (US$ 23 billion) and biodiversity (US$ 22 billion) (Cesar et al., 2003b). Their regional distribution shows that the largest costs are in Australia (US$ 28.4 billion) and Southeast Asia (US$ 38.3 billion). A recent study by The Australia Institute (2016) showed that if coral bleaching persists, tourism areas adjacent to the Great Barrier Reef risk declines in visitors from 2.8 million visitors to around 1.7 million per year. This the equivalent of more than $1 billion in tourism expenditures, supporting around 10,000 tourism jobs in regional Queensland (The Australia Institute, 2016) (Hughes et al., 2017).

Tourism is an important sector in the Pacific region (www.wttc.org/). For many PICTs, given limited options, tourism is one of the main industries that provides opportunities for social and economic development (Jiang and DeLacy, 2014). With a projected additional 1 million visitors to the region by 2040, tourism could further accelerate these opportunities for growth and employment across the region by generating additional spending of more than US$1.6 billion, and creating an additional 110,000 jobs (World Bank, 2017). However, tourism destinations in the Pacific region, which tend to focus on nature-based and predominantly marine activities, have also been identified as being most at risk of climate change (Klint et al., 2015). Sea level rise, rising sea temperatures, cyclones and storm surges are the most important projected
impacts likely to affect the sector through for example loss of beaches, degradation of coastal ecosystems, damage to critical infrastructure, and loss of reef attractiveness for tourists due to bleaching. Droughts, which are projected to be more frequent, are also likely to have an impact on the industry (and local islanders’ food security) through water and food shortages (Pearce et al., 2018)

The main economic activity in the Small Island Developing countries of the Caribbean region is tourism. In addition, fisheries provide important livelihoods as well as food security and nutrition opportunities for local communities, especially of low income. Extreme weather events, sea level rise and increasing sea surface temperatures are three of the main related climate change impacts that appear to be of particular concern to Caribbean islands. A USAID Rapid climate change vulnerability assessment states that ‘The total capital cost of a 1-meter rise in sea level by 2080 in the 10 countries is predicted to be US$20.6 billion to rebuild or relocate infrastructure due to the direct damage as well as lost land-value.’ Primarily this comes from losses in annual tourism revenues, as well as capital costs related to damage to tourism resorts.

Caribbean tourism is particularly vulnerable to climate change due to its concentration in low-lying coastal areas. According to the AR5 report, the Caribbean region will be highly impacted by climate change effects on the tourism sector. The report highlighted the extent of economic losses to coastal tourism in the Caribbean due to hurricanes. For example, in the Caribbean community countries, rebuilding costs of tourist resorts are estimated at US$10 to US$ 23.3 billion annually by 2050. A hypothetical 1-m sea level rise would result in the loss or damage of 21 airports, inundation of the land surrounding 35 ports, and at least 149 multi-million dollar tourism resorts damaged or lost from erosion to the coastal beach areas (Simpson, 2010). The model developed by Cashman et al. (2012) suggests that, given the proximity of most hotels to the beach, a 0.2 meter rise in sea levels could likely impact on about 12 hotels or about 7 percent of the total number of hotels in Barbados. These hotels account for 122 rooms or about 3 percent of the room stock.

Assuming about a 56 percent occupancy rate, this translates into an annual reduction in revenue of about US$3.5 million. The long-term projected rise in sea levels, about 0.5 and 1 meter, is likely to have more severe effects: both scenarios are likely to impact over 40 percent of hotels and lead to a reduction in revenue of around US$100 million, or about 3 percent of GDP, over the long-run (Cashman et al., 2012).

North Atlantic hurricanes and tropical storms have increased in intensity over the last 30 years, with climate projections indicating the trend is for a continued increase in hurricane intensity (Chapter 6). An analysis of climate change’s impact on Jamaica’s coastal transportation infrastructure (two airports, a port and a container terminal) estimated current cumulative losses due to damage associated with natural disasters to be around USD120 billion, roughly 7% of GDP, and rising to as high as 56% of GDP by 2025 if climate projections are accurate (UNCTAD, 2017). In 2015, Tropical storm Erika devastated Dominica causing USD483 million in total damages and losses (mostly related to transportation, housing and agriculture damages), equivalent to 90 percent of Dominica’s GDP (World Bank, 2017). 2017 saw three major storms, Harvey, Irma and Maria, destroy parts of the region. Total damages and losses caused by the latter two storms to the tourism industry of Dominica, the British Virgin Islands, and Antigua and Barbuda were estimated at USD2.2 billion, with recovery costs in the environment evaluated at USD6.8 million (UNDP, 2017). Total recovery needs for Dominica, Barbuda, and the British Virgin Islands alone were estimated at over USD5 billion (UNDP, 2017). Based on information collected on damages to fishing operations and fishing-related businesses for Florida, Puerto Rico and the US Virgin Islands, NOAA estimated total damages of $117 million and revenue losses of $106 million.

Recent studies provide additional evidence that species range shifts (Sections 5.2.3, 5.3.3) are likely to alter tourism flows. For instance, poleward range shifts of marine fishes in the United States could yield new opportunities for recreational fishing in temperate regions, while warmer conditions could benefit some species purposely introduced for recreational fishing (DiSegni and Shechter, 2013). Conversely, increases in the prevalence of tropical box jellyfish through climate induced poleward range shifts could detrimentally affect coastal tourism and water-related activities in subtropical and temperate regions (Klein et al., 2014).

Increased prevalence and transmission of diseases are also likely to occur under warmer ocean temperatures, with empirical evidence of poleward shifts in disease outbreaks under warmer conditions (Burge et al., 2014). Importantly, adverse publicity accompanying an increased risk of disease transmission in tropical and subtropical regions could further have deleterious impacts on coastal tourism (DiSegni and Shechter, 2013).

If climate change and ocean acidification reduce the seafood supply (Wabnitz et al., 2017), the attractiveness for tourists will also decrease (low confidence). This will have an economic impact on jobs, revenue and

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income, and thus on the GDP (growth domestic product) of the countries. This is more deleterious for small
countries relying on tourism for foreign currency to pay for imports of subsistence products.

Overall, climate change will have many deleterious impacts on coastal tourism (medium confidence). As
tourism is a source of revenue for many countries, their economies will be impacted (medium confidence),
especially for developing countries where tourism is the main source of entry of foreign currencies (medium
to high evidence).

5.4.2.3 Environmental Dimension

5.4.2.3.1 Aesthetic aspects, property values and coastal infrastructure

The aesthetic appreciation of natural places is one of the most fundamental ways in which people relate to
their environment. However, climate change threatens these aspects through degrading aesthetic elements of
climate sensitive natural resource (medium agreement, low confidence). Positive externalities generated from
ecosystems, or cultural ecosystem services, include a range of values such as appreciation of biodiversity
and scientific opportunities, as well as aesthetic stimulus or aesthetic appreciation. Aesthetically beautiful
places are critically important for supporting local and international economies (especially through tourism),
aesthetic values within the Great Barrier Reef (GBR) by indigenous and non-indigenous local residents,
Australians (non-local), international and domestic tourists, tourism operators, and commercial fishers and
found that aesthetic values are a critically important cultural value for all cultural groups, and are important
for maintaining sense of place, pride, identity and opportunities for inspiration, spirituality, recreation and
wellbeing. However, climate change has and will destroy many of the features of regional aesthetical
ecosystem services and possible creating new aesthetical ecosystem services.

Along with negative changes in aesthetically pleasing ecosystems, climate change has and will affect real
estate values both in coastal areas as well as inland through direct physical impacts. Flooding along coasts,
drought and fires in inland areas have large economic impacts. Putra et al. (2015) study the vulnerable
coastal areas of the New Jersey real estate market by categorizing types of real estate parcels with flooding
risk. They simulate their results and calibrate the model with actual data of real estate parcels sold and
examine different insurance policies offered to the coastal flood zones. They found that property owners that
were well informed about climate change would invest heavily in defensives in the near term but then
abandon coastal real estate at some later time when the costs become too risky. McNamara and Keeler
(2013) study barrier island communities in the Atlantic Coast of US, where rising sea levels and coastal
erosion can damage real estate values. They find through their models that informed property owners invest
in defensive expenditures in the short run (medium confidence). Moreover, most urban area and mega-cities
are found along the coast. The combined effects of urbanization and climate change on near-surface
temperatures predict urban areas to be warmer and less comfortable than surrounding unaltered areas in the
future (Argüeso et al., 2014). Fischer et al. (2012) predict potential increases in morbidity and mortality with
increased temperatures and high humidity in urban areas. People may prefer to pay to compensate the
climate change impacts. Costello et al. (2010) study risk aversion and probability distribution of future
temperature changes by placing an upper bound in the fat upper tail of the distribution to find that
willingness to pay to avoid climate change is not sensitive to bound value.

One cannot avoid the issue that climate change will alter or decimate coastal and inland real estate and land
values across many regions in a variety of ways and degrees (medium agreement, medium evidence). As
regional information accumulates on the occurrence and frequency of extreme events such as flood risks,
drought duration and intensity, fires, etc., data will be incorporated into risk assessments and asset pricing.
Decreases in the real estate values, and lack of rental incomes from jeopardized properties are going to
trigger a chain of events which will be unavoidable (McNamara and Keeler, 2013). Loss in household asset
values are crucial since homes or farmland tend to be the largest form of savings for most households in
many regions. Spending patterns and amounts will be affected where lower long-term spending will lead to
decreased economic business activity and may lead to closure of many small business firms, further leading
into increased local unemployment (Rubin and Hilton, 1996). If house or land values decrease below
outstanding mortgages, defaults on debt payments may lead to regional bank crisis. In 2050, in the Tampa
Bay region, Florida, it is estimated that the inundation of 3-foot sea level rise could cost Hillsborough and
Pinellas County over 300 and 900 million dollars respectively for the real estate market alone (Fu et al., 2016).

Climate change is expected to impact coastal real estate values and incur substantial costs on coastal infrastructure due to increased frequency and intensity of floods and other disasters driven by extreme weather events, particularly in tropical SIDS. For example, much of the estimated damages, losses and associated costs incurred by Pacific SIDS from climate change will stem from sea level rise, an increase in temperature as well as the severity of cyclones, floods, and drought. Natural disasters already cost PICTs an average of between 0.5 - 6.6 % of GDP per year (World Bank, 2017). Cyclone intensity may increase in the future, by 2 – 11% by 2100, while the frequency of cyclones could fall by 6-34% over the same period (Knutson et al., 2010). Cyclones Pam and Winston, which hit Vanuatu and Fiji in 2015 and 2016 respectively, can be seen as indications of what the future may hold. For Fiji, average asset losses due to tropical cyclones and floods were estimated at more than USD226 million per year, representing more than 5 percent of Fiji’s GDP (Esler, 2016). The same study found that the fraction of GDP lost every year due to tropical cyclones and floods could increase by up to 50% by 2050 (i.e., 6.5 percent of GDP), and that almost 100 percent of GDP in investments are required over the next 10 years to strengthen Fiji’s resilience to future climate change impacts and natural hazards. The total economic value of damages and losses from tropical cyclone Winston in Fiji was estimated at US$ 0.9 billion (Esler, 2016), equivalent to 20% of the country’s GDP. If the value of destroyed environmental assets and losses in environmental services are accounted for, the total impact of cyclone Winston amounts to US$ 1.3 billion - with US$ 0.7 billion in damages and USD0.6 billion in losses. Projected production losses to the fishery sector were estimated at about USD14 million annually until 2021; and total costs to rebuild, recover and build resilience at USD0.9 billion (Esler, 2016). The total economic value of damages and losses sustained by Vanuatu during tropical cyclone Pam were estimated at US$ 449.4 million, equivalent to 64.1% of GDP. Losses to the agricultural and tourism sectors made up 33% and 26%, respectively, of total losses (Esler, 2015). Ecosystem service losses while acknowledged to be substantial were not accounted for within the impacts to GDP. Recovery and reconstruction needs were estimated at US$ 316 million.

In developed countries’ context, Ackerman and Stanton (2007) forecast that annual real estate losses due to climate change will be from 0.17% of USA GDP in 2025 to 0.36% in 2100, with Atlantic and Gulf Coast states being the most vulnerable. Bunten and Kahn (2014) build a model for regions in which capital and labor are mobile and in which there are variations in the emerging climate change risks between different areas. They state that if the population has homogenous preferences, real estate prices will drop when negative news about climate change is revealed, but not if the population is heterogeneous with respect to incomes, networks, tastes and the ability to pay for risks. Hallegatte et al. (2007) simulate several scenarios for European cities and estimate climate change adaptation costs and investments for most of the 21st century. They state that ‘As a consequence (of uncertainty), no anticipated adaptation is possible and the reactive adaptation leads to sub-optimal solutions, costly over the long-term, or to huge short-term costs of adaptation. The static and enumerative approach to monetary damage valuation tends to underestimate their actual costs for future generations because it neglects important mechanisms such as a crowding-out of productive investment or a depreciation of real estate assets which can dampen economic growth.’ (p. 14). Masson et al. (2014) discuss how joint efforts by all planners in the urban areas are needed to plan adaptation strategies to climate change. Time will show whether the potentially catastrophic loss in real estate and consumer wealth will be addressed through more insurance or through costlier defensive measures, which all depend on the asset holders and cities willingness to pay. How to discount for potential losses that may occur in the future (with an unknown occurrence) are areas of research that need to be addressed on a regional basis to forecast real estate loss estimates (low confidence).

5.4.2.3.2 Risk and opportunities for blue economy

Ocean is a source of renewable energy and marine renewable energy is still underdeveloped compared to other renewable energy and technology (Section 5.5.1). However, the potential economic benefits generated from the development of marine renewable energy can be positive effects of engaging in carbon mitigation.

Ocean Renewable Energy (ORE) provides an emerging alternative to fossil fuels and comprises energy extraction from tides, wind-waves, ocean thermal gradients, currents and salinity gradients (O Rourke et al., 2010; Kempener and Neumann, 2014a; Kempener and Neumann, 2014b; Borthwick, 2016; Manasseh et al., 2017b) (see Secton 5.4.1). The vast global resource of OCE is abundant (Borthwick 2016) and is less
variable than resources such as wind and solar, and therefore offers the potential benefit of providing more continuous supply within the energy network (Foteinis and Tsoutsos, 2017; Hemer et al., 2017). Tidal energy is the most advanced technology with notable commercial installations of tidal barrage technology in France (La Rance River, 240 MW), Canada (Bay of Fundy, 20 MW) (O Rourke et al., 2010), and most recently South Korea (Silhwa Lake, 254 MW) (Kempener and Neumann, 2014a). Tidal stream technology has seen significant advances in the development and testing of turbines, but has not yet advanced to the stage of large-scale arrays. The complex nature of ocean waves has given rise to a great diversity in the device designs, most of which can be broadly categorised as oscillating water columns, oscillating body converters or overtopping converters (Kempener and Neumann, 2014a; Kempener and Neumann, 2014b; Manassee et al., 2017a). Other potential ORE include aglae biofuels. However, few designs have progressed beyond the technology demonstration stage (Hemer et al., 2018). The remaining ORE technologies are generally at lower levels of technical development (Kempener and Neumann, 2014a; Kempener and Neumann, 2014b; Borthwick, 2016).

Direct effects of climate change which are relevant for maritime transport include ocean warming and related melting of sea ice, sea level rise and floodings in coastal areas. Sea level rise, storms and associated floodings will cause direct costs due to infrastructure damages in ports and other coastal facilities and costs for measures for flood-defenses (Becker et al., 2018; Ng et al., 2018). The damage caused by a changing climate may present more of a threat for developing than for developed economies. This needs to be considered for adaptation measures and investments ‘to build resilient infrastructure’ (SDG 9) for maritime industry of developing countries and Small Island Developing States whose economies depend on ports (Koetse and Rietveld, 2009). Additionally, short-notice port closures, delays, detours and trip cancellations could create indirect costs for maritime transportation (Koetse and Rietveld, 2009; Jaroszewske et al., 2010).

Global warming and changing weather patterns may have a substantial impact on global trade and transport pathways (Koetse and Rietveld, 2009). The reduction in sea ice in the Arctic Ocean during summer opens up the possibility for sea transport on the Northwest or Northeast Passage during at least several months per year (Ng et al., 2018). Both routes may provide opportunities for more efficient transport between North America, Europe, Russia and China for fleets with established Arctic equipment and may open up access to known natural resources which were so far covered by ice (Guy and Lasserre, 2016). However, whether the Arctic shipping routes may or may not be a realistic alternative will depend on political regulations, economic aspects such as infrastructure along and reliability of the routes but also societal trends, demographics, and tourism demand (Prowse et al., 2009; Wassmann et al., 2010; Pelletier and Guy, 2012; George, 2013; Hodgson et al., 2016; Pizzolato et al., 2016; Dawson, 2017). Increased traffic in either the Arctic or Antarctic would contribute to additional pollutants in the region, having a direct impact on the climate and exacerbating localized warming (Wan et al., 2016).

The shipping industry is also facing emission controls of greenhouse gases and pollutants. The combustion of low-grade, high-sulphur content fuel in ships emits significant amounts of other greenhouse gases and pollutants, such as sulphur oxides, nitrogen oxides, aerosols (containing organic carbon, black carbon, polycyclic hydrocarbons (PAH) and heavy metals) to the atmosphere (Eyring et al., 2005; Corbett et al., 2007). These have substantial consequences for the atmosphere, the marine environment and human health (Corbett et al., 2009). Some compounds are deposited relatively close to the source vessel and become dissolved or suspended in the surface ocean. Several measures were implemented by the IMO through the MARPOL Convention and other treaties to ‘conserve and sustainably use the marine ecosystems and resources’ (SDG 14). This includes well-known ship-related measures to prevent atmospheric and seawater pollution from ships (by oil, chemicals, harmful substances, sewage, garbage) and new technological advances to reduce underwater noise, avoid transfer of harmful invasive species by ballast water, and recycle ships in an environmentally sound way. Recently implemented, the IMO ‘Global sulphur cap’ limits the release of sulphur oxides due to fuel combustion. Strict CO2 emission controls in maritime transport are expected in future to be comparable to recent regulatory developments in aviation or land-based transport. In order to reduce greenhouse gas emissions by ships, two mandatory energy efficiency measures, known as the Energy Efficiency Design Index (EEDI) for new ships and the Ship Energy Efficiency Management Plan (SEEMP) for existing ships, were introduced by the UN International Maritime Organization (IMO) in 2011 (Resolution MEPC 203(62)). A CO2 reduction strategy has been developed and adopted at the IMO in April 2018 (RESOLUTION MEPC.304(72), 2018). It aims to reducing total annual greenhouse gas emissions from international shipping by 50% by 2050 compared to reference year 2008. In the meanwhile, rising oil
prices increase shipping costs significantly and drive economical decisions. Until technological measures are in place, slow steaming may be a preferred option by ship operators in order to save costs but also to reduce CO2 emissions (Bouman et al., 2017). During recent years, there have been large advances in technologies and operational practices with high potential to cut CO2 emissions and fuel consumption by as much as 84%. Those measures include the use of alternative fuels, such as LNG and biofuels, developing better power and propulsion systems, more efficient ship hull design, improved equipment, and advanced weather routing (Bouman et al., 2017). However, no single measure is itself sufficient to achieve significant emission reductions in a cost effective manner within the next decades (Corbett et al., 2007). In order to minimize ship pollution of this pristine environment, special measures were implemented under the IMO Polar Code (IMO 1 January 2017) for ships operating in both Antarctic and Arctic waters. The Arctic Council recommended, that in the future, IMO should extend the Polar Code to include a ban on heavy fuel oil (HFO) (Wan et al., 2016).

Overall, a growing number of shipping companies are intending to drastically reduce CO2 emissions from ships over the next decade hence contributing to ‘combat climate change and its impacts’ (SDG 13) and ‘ensure sustainable consumption and production’ (SDG 12) (*medium agreement, limited evidence*). In addition, the shipping industry aims to reduce shipping-related pollution in the wider oceans but also in ports and coastal regions where ship-derived emissions may contribute to tens of thousands of premature mortality every year (Bonnet et al., 2005; Sofiev et al., 2018). The use of new emission reduction technologies, such as scrubbers, to comply with a recent environmental regulation (in this case the IMO ‘Global sulfur cap’) will benefit the environment by significantly reducing the release of pollutants to the atmosphere. However, there is incomplete understanding of the impact of, e.g., scrubber wash water discharge on marine chemistry, biodiversity and biogeochemical processes (Stips et al., 2016; Endres et al., 2018). In the global shipping network, new strategies were established such as the management of ship routing and establishment of Special Areas and Particularly Sensitive Sea Areas to avoid collisions between ships and marine mammals and to protect vulnerable marine ecosystems. Despite existing guidelines for levels of monitoring and compliance of ship pollution, there is still the risk for acidification, eutrophication and accumulation of pollutants in the marine environment (CGDD and CETE, 2012).

### 5.4.2.4 Impacts of Changing Ocean on Sustainable Development Goals

Climate change impacts will have consequences for the ability of human society to achieve sustainable development. The SR 1.5 °C concludes that ‘Limiting global warming to 1.5°C rather than 2°C would make it markedly easier to achieve many aspects of sustainable development, with greater potential to eradicate poverty and reduce inequalities (*medium evidence, high agreement*).’ This assessment focuses more specifically on how climate change impacts on marine ecosystems would challenge sustainable development. Specifically, this assessment uses the United Nations Sustainable Development Goas (SDGs) as a framework to discuss the linkages between marine ecosystems and sustainable development, and how climate change would affect such linkages.

Sections 5.4.2.1, 5.4.2.2 and 5.4.2.3 highlights that climate change impacts on the ocean affect people directly and indirectly. Climate impacts on marine ecosystems affects its ability to provide seafood, raw materials, and support biodiversity, habitats and other regulating processes (Section 5.4.1). Thus, the most direct impact on sustainable development is through the SDG on sustainable ocean (‘life under water’ SDG 14).

Impacts to other SDGs were considered more indirect in nature, where impacted ecosystems are mediated through social and economic factors first before affecting an SDG targets (Singh et al., in review). For example, climate impacts on marine ecosystem services related to primary industries that provide food, income and livelihood to people have direct implications for a range of SDGs. These SDGs include ‘no poverty’ (SDG 1), ‘zero hunger’ (SDG 2), ‘decent work and economic growth’ (SDG 8), ‘reduced inequalities’ (SDG 10) and ‘Responsible consumption and production’ (Singh et al., in review) (Figure 5.19). These impacts relate to changing ocean under climate change that affect the pathways to build sustainable economies and eliminate poverty (Sections 5.4.2.2.1, 5.4.2.2.3), eliminate hunger and achieve food security (Section 5.4.2.1.2), reduce inequalities (Sections 5.4.2.1.3, 5.4.2.2) and achieve responsible consumption and production (Sections 5.4.2.2.1, 5.4.2.1.5) (Carvalho et al., 2017; Castells-Quintana et al., 2017). Climate change is also creating living conditions in coastal areas that are less suitable to human
settlement and changing distributions of marine disease vector (5.3.2.1.1), reducing our chances of achieving the goal for good health and wellbeing (SDG 3) (Pearse, 2017; Wouters et al., 2017). Also, women are often engaged in jobs and livelihood sources that are more exposed to climate change impacts from the ocean such as impacts on fisheries (Section 5.4.2.2.1) and impacts of sea level rise on coastal regions (Chapter 6). For example, in Senegal, women disproportionately cultivate rice crops (Linares, 2009), and in coastal areas they may be more affected by rising sea levels and resulting salinization (Dennis et al., 1995). Flooding in Bangladesh has increased the vulnerability of women to harassment and abuse as the flooding upends normal life and increases crime rates (Azad et al., 2013). As such, climate change may negatively affect our ability to achieve ‘gender equality’ (SDG 5) (Salehyan, 2008). Impacts to living conditions as well as changing recreational, aesthetic, and spiritual experiences also affects our ability to achieve ‘sustainable cities and communities’ (SDG 11).

The consequences of climate change in the oceans to achieving the remaining SDGs are less clear. However, the SDGs are interlinked, and achieving SDG 14, and especially the targets of increasing economic benefits to Small Island Developing States and Least Developed Countries, as well as eliminating illegal fishing and overfishing, will benefit all other SDGs (Singh et al., 2017). Achieving these targets is sometimes necessary to achieve other SDGs.

Overall, climate change will impact the chance of achieving the Sustainable Development Goals and sustaining their benefits (medium confidence). Particularly, climate change will have direct consequences for the goals that consider sustainable oceans (SDG 14) (high confidence), achieving food security (SDG 2) (medium confidence), achieving good health and wellbeing (SDG 3) (low confidence) as well as all other SDGs indirectly (low confidence).

**Figure 5.19:** Summary of the types of relationships between impacted marine ecosystem services and the Sustainable Development Goals based on literature review and expert-based analysis (Singh et al., in review).
5.5 Risk-reduction Responses and their Governance

5.5.1 Ocean-based Mitigation

IPCC AR5 and UNFCC definitions of climate mitigation cover both the reduction of greenhouse gas emissions and the active removal of greenhouse gases from the atmosphere, and that usage is followed here (SROCC Glossary). Article 4.1 of the Paris Agreement recognizes the need for both actions in its requirement for a ‘balance between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century’. That balance (net zero emissions) is how dangerous climate change can be avoided by ‘holding the increase in the global average temperature to well below 2.0°C’ Article 2.1 (Rogelj et al., 2015; Geden, 2016). To reduce the scale of the physical changes to the ocean and their environmental impacts (Sections 5.2, 5.3), transformative and near-total emission reductions are therefore needed in all sectors of human activity, not just energy, and for all greenhouse gases (Myhre et al., 2013) not just CO₂ (UNEP, 2017a).

Detailed assessment of marine renewables, both from physical processes and biofuels, is expected to be given by WG III in the IPCC Sixth Assessment Report (AR6), in the context of other renewable energy sources and a comprehensive assessment of all risk-reduction responses. In accordance with the approved scoping of this report, as discussed in Section 1.5, ocean-based mitigation through renewable energy is not considered here, nor are ocean-based albedo enhancement methods, also known as solar radiation management or sunlight reflection methods (SRM).

Instead the focus here is on the effectiveness, risk trade-offs and societal feasibility of two other ocean-based mitigation approaches. First, the scope to decrease anthropogenically-driven releases of greenhouse gases from the marine environment, primarily by maintaining the integrity of natural carbon stores in the coastal zone. Second, the potential to promote additional carbon uptake by marine ecosystems, both in the coastal zone and the open ocean, as a contribution to the additional gigaton-scale uptake (negative emissions) of CO₂ that is needed in most emission pathways that limit global warming to 1.5°C-2.0°C (Fuss et al., 2014; Anderson and Peters, 2016). Any ocean-based removals of CO₂ from the atmosphere could only be as an addition to, and not as a substitute for, very rapid reductions in greenhouse gas emissions (UNEP, 2017a). Under most international definitions, including those of IPCC, such negative emissions (CO₂ removal) are considered to be a form of climate geoengineering, as well as part of climate mitigation (Williamson and Bodle, 2016).

The term ‘blue carbon’ was originally defined in UN reports to cover all carbon captured by marine ecosystems, both coastal and oceanic (Nellemann and Corcoran, 2009). Subsequent usage has focused on coastal carbon-rich habitats structured by rooted plants, such as mangroves, seagrasses and saltmarshes, since those components are most amenable to management that might increase the security of carbon storage (reducing emissions) and also potentially enhancing sinks (Kuwae and Hori; McLeod et al., 2011; Pendleton et al., 2012b; Thomas, 2014; Macreadie et al., 2017). The global burial carbon fluxes from all vegetated coastal ecosystems is estimated at 0.1 Gt C yr⁻¹ to 0.2 Gt C yr⁻¹ (Alongi and Mukhopadhyay, 2015; Howard et al., 2017). However, there are major uncertainties regarding these estimates, as discussed below.

Carbon uptake in the open ocean is driven by three processes: the biological pump, involving particulate organic carbon export, the microbial carbon pump, based on microbial transformation to refractory dissolved organic carbon (RDOC) (Jiao et al., 2010) and the physical pump (Ciais et al., 2013). Estimates of the global carbon ‘export’ flux from the euphotic zone due to the biological pump range from 4 Gt C yr⁻¹ to 12 Gt C yr⁻¹ (Henson et al., 2010; DeVries et al., 2017), augmented by a dissolved organic carbon flux of around 0.2 Gt C yr⁻¹ (Jiao et al., 2010; Jiao et al., 2014a; Legendre et al., 2015; Jiao et al., 2018a). Under the relatively stable climatic conditions of most of the Holocene, such inputs have been balanced by losses from CO₂ outgassing, maintaining the long residence times (hundreds to thousands of years) of dissolved organic and inorganic carbon in the ocean (Jiao et al., 2010; Jiao et al., 2014a; Legendre et al., 2015; Jiao et al., 2018a).
Figure 5.20: Marine legal zoning based on United Nations Convention on Law of the Sea (UNCLOS) relevant to potential ocean-based mitigation of CO₂ (excluding renewable energy). Mitigation measures in pale green font are those involving increased primary production to enhance CO₂ removal; those in pale brown involve other processes, mostly geochemical. NM = nautical miles. Note that: i) measurements are made from a coastal baseline (not shown); ii) the limits of EEZ may be greater than 200 NM if there is an extension of the shelf sea beyond that distance; and iii) impacts of actions may be broader than the sea areas indicated. Image by Glynn Gorick with zoning based on UNEP (2017b). [Further minor edits likely to improve image quality]

The scope for climatically-significant, effective and societally-acceptable ocean-based mitigation depends on ecosystem-specific marine carbon stocks (storage) and flows (fluxes), and on the relative importance of biological, physical, chemical and anthropogenic drivers on those carbon flows. Note that in some global carbon budgets, coastal blue carbon is considered as a land-based carbon sink, and it is included under the heading of ‘wetlands’ in the IPCC Special Report on Climate Change and Land. Accounting issues for forest-based mitigation (Grassi et al., 2017) are also relevant to vegetation-based blue carbon.

Many uncertainties currently relate to the effectiveness and feasibility of ocean-based mitigation (and adaptation), involving a wide range of environmental, technological and socio-economic considerations. Eleven ocean-based responses that involve ocean processes and ecosystems, i.e. not including societal adaptation, were reviewed by Gattuso et al. (2018), using eight criteria: maximum potential effectiveness in reducing climatic drivers (ocean warming, ocean acidification and sea level rise) by 2100, assuming full theoretical implementation; technological readiness and lead time to full potential effectiveness (subsequently combined as technical feasibility); duration of benefits; co-benefits; dis-benefits; cost-effectiveness; and governability (considered as capability of implementation, and management of any associated conflicts). These criteria were considered at either global or local scales, or both.

Effectiveness scores by Gattuso et al. (2018) are summarized in Figure 5.21. Those grouped as ‘global’ (circles) can be regarded primarily as mitigation (reducing drivers), those as ‘local’ (rectangles), as ecosystem-based adaptation (reducing impacts); two were considered at both scales. Ocean-based albedo enhancement techniques are excluded here, being outside the scope of the current report. However, physically-based marine renewables are included for comparison with the other approaches across all of the criteria.
Figure 5.21: Summary of ocean-based risk-reduction options and their potential benefits and constraints, from literature-based expert assessments by Gattuso et al. (2018), excluding solar radiation management and societal adaptation responses. Mitigation effectiveness was quantified relative to RCP8.5, assuming maximum theoretical implementation, with reduction of climate-related drivers considered at either global or local (<100 km²) scale, shown as circles or rectangles respectively. Impact reduction, co-benefits and dis-benefits are in the context of eight sensitive marine ecosystems and ecosystem services. ‘Technical issues to overcome’ is based on scores for technological readiness, lead time for full implementation and duration of effects. Cost is based on US$ per tonne of CO₂ either not released or removed from the atmosphere (for global measures) or per hectare of coastal area with action implemented (for local measures). Governability was considered in terms of the potential capability of the international community to implement each measure. NA, not assessed.

5.5.1.1 Climate Mitigation in the Coastal Ocean

5.5.1.1.1 Coastal carbon cycle: opportunities and challenges

The coastal ocean connects the mountain cryosphere (and other terrestrial systems) with the open ocean, through estuaries, shelf seas and a wide range of intertidal and shallow-water habitats. These systems play a highly dynamic and important role in the global carbon cycle, not only through in situ primary production, by macrophytes, macroalgae and phytoplankton, but also by processing riverine organic carbon. Coastal carbon flows and storage can be strongly affected by human activities, indirectly in river catchments (Jiao et
al., 2011; Regnier et al., 2013) and directly by coastal land-use changes and marine resource exploitation in shelf seas. Despite intensive study, significant uncertainties remain (Cai, 2010; Bauer et al., 2013), with many knowledge gaps arising from the complexity of bio-physical interactions and their local and regional spatial heterogeneity.

The relatively recent appreciation of the scale of carbon storage (and its apparent vulnerability) in coastal carbon ecosystems has focused attention on the opportunities and challenges for targeted ‘blue carbon’ policies that are focused on intertidal and near-shore vegetation (Crooks et al., 2011; Herr and Landis, 2016; Macreadie et al., 2017). For climate mitigation purposes, the need is for cost-effective management actions that significantly either reduce greenhouse gas sources or enhance sinks, implemented at the local, national and regional scale within an international framework. Such actions should be amenable to national carbon accounting within emission inventories, with associated requirements for quantitative verification of long-term carbon removal, together with new regulations and financial incentives, similar to those developed for managing terrestrial carbon sinks (Crooks et al., 2011; Hejnowicz et al., 2015). Proposals to extend the blue carbon concept to other components of the coastal ocean, e.g., shelf sea sediments and the water column (Zhang et al., 2017) are being developed.

5.5.1.1.2 Coastal blue carbon

The global importance of three kinds of coastal vegetation has recently been recognised: mangroves, salt marshes and seagrasses (Chmura et al., 2003; Duarte et al., 2005; Kennedy et al., 2010; McLeod et al., 2011). Although covering only ~0.1% of the Earth’s surface, it has been estimated that these three systems together support 1–10% of global marine primary production, and may make an even larger contribution to total oceanic CO₂ uptake (Duarte et al., 2017). Some 151 countries around the world contain at least one blue carbon ecosystem (seagrass, saltmarshes or mangroves) and 71 countries contain all three (Herr and Landis, 2016). Seaweeds (macroalgae) are considered separately in Section 5.5.1.1.3 below, since their long-term carbon storage does not occur in situ, and is inherently more uncertain.

The order-of-magnitude range of values for the role of vegetated coastal ecosystems in carbon processing is primarily due to uncertainties in quantifying two key factors: the areal extent of these habitats, and their rates of carbon burial. Differences in estimates for areal coverage depend on whether values are determined empirically, or on theoretical grounds of environmental suitability (substrate, light availability and temperature). Observation-based mapping provides global area estimates of ~140,000 km² for mangroves (Giri et al., 2011), ~55,000 km² for saltmarshes (McOwen et al., 2017a) and ~177,000 km² for seagrasses (Green et al., 2003); however, theoretically-based estimates can be much higher, e.g., 500,000 km² for seagrasses (Green et al., 2003).

Uncertainties in rates of carbon uptake and long-term removal mostly relate to assumptions made regarding associated sediment accumulation rates, occurring over decadal to millennial timescales (McKee et al., 2007; Lo Iacono et al., 2008). Application of geochemical-based approaches to blue carbon sediments has indicated that many previously-published values for carbon burial may be overestimates, by a factor of 11 - 3,100 for seagrasses (Johannessen and Robie, 2018). Such issues are contentious (Johannessen and Robie, 2018; Macreadie et al., 2018; Matthew et al., 2018); their scientific resolution is therefore highly desirable. Additional considerations are discussed below.

Despite the low confidence regarding values for their carbon fluxes, there is high confidence that carbon storage, per unit area, in vegetated marine habitats can be much greater than for terrestrial habitats. Mangroves are particularly effective in retaining carbon: their above- and below-ground woody structures and associated sediments typically contain 1000 t C ha⁻¹, providing global total storage of 5-6 GtC (Donato et al., 2011; Alongi and Mukhopadhyay, 2015; Howard et al., 2017). Factors affecting the mangrove carbon storage are discussed by Adame et al. (2017); Schile et al. (2017); for seagrasses, by Lavery et al. (2013) and for salt marshes, by Kelleway et al. (2017b).

There is also high confidence that coastal wetlands can significantly assist with climate adaptation. For example, by absorbing incoming wave energy, providing hurricane and storm surge protection, and preventing erosion (Costanza et al., 2008; Spalding et al., 2014). Coastal ecosystems may keep pace with sea level rise and, in some instances, are more cost-effective than hard civil infrastructure like seawalls (Temmerman et al., 2013). Healthy coastal ecosystems provide other benefits, including fishery habitat,
control of water quality, and local livelihoods (Sections 5.3.3, 5.4.1.3). Thus, blue carbon protection has multiple co-benefits through supporting biodiversity and a wide range of ecosystem services (high confidence).

When degraded, these co-benefits are greatly diminished along with the ecosystems’ capacity to sequester carbon, and stored carbon can be released back to the atmosphere, along with other greenhouse gases (Marba and Duarte, 2009; Duarte et al., 2010; Pendleton et al., 2012b; Lovelock et al., 2017). Around 25–50% of mangrove forests and other vegetated coastal habitats have already been lost or degraded due to coastal developments and other land-use change during the past 100 years (McLeod et al., 2011). Current losses are estimated at 0.2–3.0% yr\(^{-1}\), depending on vegetation type and location (Hiraishi et al., 2014; Alongi and Mukhopadhyay, 2015; Atwood et al., 2017) (Section 5.3.3, 5.4.1.3). Associated global carbon emissions due to these habitat losses have been estimated at 0.04 GtC yr\(^{-1}\) to 0.28 GtC yr\(^{-1}\) (Pendleton et al. 2012); 0.06 GtC yr\(^{-1}\) to 0.61 GtC yr\(^{-1}\) (Howard et al., 2017) and 0.1 GtC yr\(^{-1}\) to 1.46 GtC yr\(^{-1}\) (Lovelock et al., 2017). The highest of these estimates exceeds the most recent global estimate of carbon emissions from all land-use change: 1.3 ± 0.7 GtC yr\(^{-1}\) (Le Quéré et al., 2018). However, Lovelock et al. (2017) note that several processes could reduce their maximum value. These factors include organic matter remaining in anoxic conditions after vegetation is removed; interactions between CO\(_2\) and carbonates in sediments; and the offshore transport and reburial of organic carbon.

Awareness of the scale of potential CO\(_2\) emissions from the continued loss of coastal vegetated habitats (together with increased appreciation of other ecosystem services that they provide) has resulted in the protection and restoration of coastal blue carbon ecosystems being recognized as a national priority for both climate change mitigation and adaptation (Martin et al., 2016; Gallo et al., 2017). In their Nationally Determined Contribution (NDC) to the Paris Agreement, 28 countries include a reference to coastal wetlands in terms of mitigation and 9 countries include coastal ecosystems and the coastal zone in their adaptation strategies (Figure 5.14).

The following management measures have been proposed to maintain and promote carbon storage in coastal ecosystems, although not all are specifically included in NDCs:

- **Improved regulatory protection.** Legal structures already exist in many countries to protect mangrove forests. The main need is for much increased enforcement, for both nation-wide regulation and site-specific marine protected areas (Miteva et al., 2015; Howard et al., 2017). The lack of such capacity to date has allowed loss and degradation to continue (Herr et al., 2017). There is currently much less regulatory protection for salt marshes and seagrass habitats, not yet well-mapped in many regions (Duarte et al., 2017). Whilst some salt marsh protection does now exist in Europe and North America, it may be ‘too little, too late’ for significant climate mitigation; in NE Europe, ~90% of estuarine wetlands have already been lost (Jickells et al., 2017).

- **Restoration and rehabilitation.** Around 90 restoration and rehabilitation projects for mangroves have been documented (López-Portillo et al., 2017). Not all were successful; the need for improved, science-based guidelines has been identified. Coastal wetland restoration is generally unlikely to succeed unless the original environmental and human factors causing loss and degradation have been properly addressed (high confidence) (Zhao, 2016). Natural regeneration has greatest chance of long-term survival.

- **Specific measures to increase carbon sequestration.** Macreadie et al. (2017) identify three catchment-level management measures to increase carbon uptake and storage for coastal ecosystems: reducing anthropogenic nutrient inputs; reinstating predators (to reduce carbon loss caused by some bioturbators); and restoring hydrology, by removing barriers to tidal flow.

Successful large-scale implementation of the above measures would assist in global climate mitigation, and hence in reaching the net zero emissions target of the Paris Agreement (high confidence). For some countries, e.g., Indonesia, Malaysia and small island states, the contribution to climatic benefits could be substantial (Murdiyarso et al., 2015; Atwood et al., 2017). Nevertheless, the associated uncertainties in quantifying relevant carbon storage and flows, as discussed above, are problematic for reliable measuring, reporting and verifying (MRV). Additional considerations with regard to the complexity of accounting for coastal blue carbon include the following:
• Significant CO₂ release from carbon-rich sediments may continue for 10–40 years after vegetation loss, with the rate of that release depending on local environmental factors (Lovelock et al., 2017).

- Emissions of methane and nitrous oxide, from both intact and degraded blue carbon ecosystems, can significantly alter the scale of climatic benefits (Adams et al., 2012; Chen and Ganapin, 2016; Chmura et al., 2016; Rosentretre et al., 2018).
- Carbonate formation in some blue carbon systems may release CO₂, offsetting the benefits of carbon storage (Howard et al., 2017; Macreadie et al., 2017).
- Some of the carbon stored in marine sediments may be recalcitrant carbon from terrestrial or atmospheric sources, and should therefore be excluded (Chew and Gallagher, 2018).
- The permanence of vegetated coastal systems, even if well-protected, cannot be assumed under future temperature regimes (Ward et al., 2015; Duke et al., 2017; Jennerjahn et al., 2017; Nowicki et al., 2017) and sea level rise (Kirwan and Megenigal, 2013; Spencer et al., 2016). However, carbon sequestration capacity may not necessarily decrease; e.g., if mangroves replace salt marshes (Kelleway et al., 2016), or where totally new habitats might be created (Barnes, 2017).

The above complexities currently make it difficult to provide accurate estimates of the climate mitigation achieved through blue carbon management measures. Furthermore, even if all coastal blue carbon ecosystems could be restored to their 1990 extent (unlikely, because of the semi-permanent nature of most coastal land-use change, such as human settlement and conversion to agriculture), their contribution to the ‘emissions gap’ (UNEP, 2017a) would be relatively modest at the global scale (high confidence). For example, Griscom et al. (2017) estimated that maximum global mitigation benefits of cost-effective coastal wetland restoration could be ~0.05 GtC yr⁻¹ by 2030, compared to current net emissions from all sources of 10.0 GtC yr⁻¹ (Le Quéré et al., 2018). Gattuso et al. (2018) similarly concluded that coastal restoration and protection only offered a very limited (and relatively unreliable) ‘solution’ to climate change. Nevertheless, measures to protect and, where practicable, re-instate such habitats provide many other societal benefits through ecosystem services; in particular, benefitting local fisheries, enhancing biodiversity, giving storm protection and reducing coastal erosion. Therefore, coastal blue carbon can be considered as a ‘no regrets’ option at the national level in many countries, in addition to other, higher magnitude, mitigation measures. Additional process studies are needed over the full range of environmental conditions to improve the low confidence in current estimates of carbon loss from degraded vegetated coastal ecosystems, covering more than an order of magnitude range.

5.5.1.1.3 Seaweeds and shelf sea sediments

Seaweeds (macroalgae) do not directly transfer carbon to marine sediments, unlike rooted coastal vegetation (macrophytes, as considered above) (Howard et al., 2017). Nevertheless, seaweed detritus can deliver carbon to sedimentary sites (Hill et al., 2015) and may also provide a source of refractory dissolved organic carbon (RDOC) (Hughes et al., 2012). Recent studies indicate that globally-important amounts of carbon may be involved in these processes (Krause-Jensen and Duarte, 2016; Krause-Jensen et al., 2018; Smale et al., 2018). Nevertheless, there is low confidence that seaweed provides a significant natural contribution to long-term permanent carbon removal, due to large uncertainties relating to transport pathways and the fate of material transported to deeper water. The balance between carbon uptake and the scale of its subsequent releases to the atmosphere is the key issue, rather than uptake per se. Large-scale and multi-year tracking studies would be needed to provide reliable data on carbon sequestration over decadal to century timescales.

Seaweed aquaculture is inherently more manageable for mitigation purposes (N’Yeurt et al., 2012; Chung et al., 2013; Chung et al., 2017; Duarte et al., 2017). If linked to biofuel/biogas production (N’Yeurt and Iese, 2014; Moreira and Pires, 2016; Sondak et al., 2017), there would be potential to reduce emissions (as an alternative to fossil fuels); if also linked to carbon capture and storage (Hughes et al., 2012), there could be potential to achieve negative emissions (net CO₂ removal from the atmosphere). Full life cycle analyses are needed to assess the energy efficiency of such approaches, and the viability of their scaling-up to climatically-significant levels. N’Yeurt et al. (2012) envisage 9% of the global ocean used for seaweed aquaculture; however, they did not assess the associated environmental and socio-economic implications. A different mitigation option using seaweeds relates to their use as a dietary supplement for ruminants, to suppress methane production. In vitro studies have given promising results (Dubois et al., 2013; Machado et al., 2016; Machado et al., 2018); the need is to quantify the potential scale of real-world benefits. Until that has been done, there is only low confidence in this approach having significant mitigation potential.
Actions to increase the security of carbon storage in shelf sea sediments primarily relate to the management of human activities that cause sediment disturbance, through fishing, dredging and the installation of offshore structures, hence affecting carbon storage and cycling both directly (physically) and indirectly (biologically, through effects on benthic organisms) (Hale et al., 2017). Carbon retention, over relatively large areas of shelf seas, might therefore be increased (Avelar et al., 2017). However, the effects involved are complex (van de Velde et al., 2018), with lack of data on the natural variability of the processes involved. In particular, there have been few large-scale studies on ‘baseline’ carbon sequestration rates in shelf sea sediments, and of the factors affecting storage in the potentially-mobile fraction; exceptions are provided by Hu et al. (2015) and Diesing et al. (2017). Due to these uncertainties, there is currently low confidence that shelf sea sediments can be used for climate mitigation purposes.

5.5.1.1.4 Alkalinization of coastal seas and other non-biological CO₂ removal methods

There is theoretical potential for substantive enhancement of coastal (and open ocean) geochemical processes that remove carbon from the atmosphere (Rau et al., 2012; Renforth and Henderson, 2017). These approaches may either initially involve extensive chemical manipulations (e.g., to generate CaOH), or be based on relatively abundant mineral resources, primarily carbonates (e.g., calcite and dolomite) or silicate minerals (such as olivine) that neutralize acidity and absorb CO₂.

Such ocean alkalinization methods might be used to reduce local impacts, e.g., for coral reefs (Albright et al., 2016; Feng et al., 2016), as well as contributing to wider counteraction of climate change. Rock-weathering with associated permanent CO₂ uptake is a natural process, although occurring relatively slowly: it currently sequesters ~0.25 GtC yr⁻¹, on land and at sea (Taylor et al., 2015). The main constraints on the development and deployment of ocean alkalinization as a mitigation method relate to the governance, cost, and uncertain environmental impacts (on land as well as at sea) of climatically-significant application (Gattuso et al., 2018). The combination of these factors results in low confidence that seawater alkalinization can provide a viable and acceptable climate mitigation approach.

CO₂ can also be extracted from coastal seawater electrochemically (Eisaman et al., 2011), with potential for the co-production of hydrogen (Willauer et al., 2017). The CO₂ would need to be safely stored through carbon capture and storage (CCS) technologies. Detailed assessment of the feasibility of electrochemical and geochemical methods for ocean-based mitigation is not in scope for this Special Report.

5.5.1.1.5 Refractory dissolved organic carbon ('land-sea integrated eco-engineering')

Land-based measures have also been proposed, affecting marine carbon storage in the water column through nutrient management and effects on dissolved organic carbon. Measurements of air-sea CO₂ fluxes (Laruelle et al., 2015) and modelling studies (Bauer et al., 2013) indicate that most shelf seas are currently a net carbon sink, with anthropogenic nutrients implicated through their role in stimulating shelf primary production, hence CO₂ drawdown (Bauer et al., 2013; Regnier et al., 2013). However, eutrophic waters can also be carbon sources, with rapid re-mineralisation of organic material by microbially-driven decomposition, returning CO₂ to the atmosphere (Jiao et al., 2011). If land-based nutrient inputs can be reduced, a higher proportion of refractory DOC may be produced (Jiao et al., 2014b). This idea is supported by a statistical analysis of the relationship between organic carbon and nitrate in various natural environments (Taylor and Townsend, 2010) as well as by experimental results in estuarine and offshore waters (Yuan et al., 2010; Jiao et al., 2011; Jiao et al., 2014b).

Improved management of chemical fertilizers may promote DOC formation. Chemical fertilizers have been excessively applied in farming for decades especially in developing countries. Excessive N and P are then washed out into rivers and ultimately discharged into the coastal waters, causing eutrophication and algal bloom (both red-and green tide) (Zhang et al., 2018). Although algal blooms seemingly produce more organic carbon, this carbon is basically labile and can be respired rapidly. In addition, the labile DOC produced by autochthonous primary producers has priming effects on the river discharged terrestrial RDOC, i.e., remobilize RDOC for microbial uptake and respiration (Liu et al., 2014), which can create high CO₂ concentration causing acidification in ambient water, and excess CO₂ can escape from water to atmosphere as outgassing. That is why productive estuarine and coastal waters are often sources rather than sinks of atmospheric CO₂ (Jiao et al., 2018a; Jiao et al., 2018b). Meanwhile this process consumes large quantity of oxygen resulting in hypoxia. Anoxic conditions could cause massive death of macro- and micro- biomes, resulting in the breeding of anaerobic bacteria that transform organics into CH₄, H₂S, NH₃ and other toxic...
substances, which in turn are destructive for the ecosystem (Jiao et al., 2018a; Jiao et al., 2018b). On top of that, excess discharge of nutrients (nitrogen, N and phosphorus, P) shapes the C/N and C/P elemental ratios in favour of remobilization of RDOC for respiration, lowering the microbial carbon pump (MCP) efficiency and carbon sequestration (Jiao et al., 2010).

Therefore, reducing terrestrial input of inorganic nutrients could be a countermeasure for the enhancement of carbon sequestration in coastal waters (Jiao et al., 2011). Land-ocean integrated management and engineering could help reduce the application of chemical fertilizers in farming and eliminate sewage discharge into the river so as to reduce the N, P inputs into the sea. Such eco-engineering could have the co-benefits of protect coastal ecosystems by reducing eutrophication and occurrence of red-tides while increasing carbon sequestration through the MCP. Land-ocean eco-engineering may also be more effectively implemented in association with reward-based carbon trading policy to replace the penalty-based pollution policy that is currently being in practice in many countries (Jiao et al., 2018b).

5.5.1.2 Climate Mitigation in the Open Ocean

5.5.1.2.1 Open ocean carbon cycle: opportunities and challenges

Current net primary production by marine phytoplankton is estimated to be 58 ± 7 GtC yr⁻¹, similar to terrestrial primary production (Buitenhuis et al., 2013). However, only ~1% of that marine production is removed annually from circulation by the combined effects of the three biologically-driven ocean carbon pumps (soft-tissue pump, carbonate pump and microbial carbon pump) (Legendre et al., 2015). It is the concentration gradient between (increasing) atmospheric CO₂ and dissolved CO₂ in surface waters that is primarily responsible for the global net removal of 2.6 GtC yr⁻¹ from the atmosphere in the decade 2006–2015 (~25% of anthropogenic CO₂ emissions; Le Quere et al. (2016)). The residence time of dissolved inorganic carbon (DIC, arising from atmospheric uptake and the relatively rapid remineralization of most organic material) depends on ocean circulation, varying from days to centuries, that of refractory dissolved organic carbon (RDOC) may be as long as 5000 years, while the organic carbon buried in marine sediments could be permantly removed from circulation.

Two main methods have been proposed to increase CO₂ uptake by biological processes in the open ocean: chemically, by adding terrestrial-derived nutrients that may currently limit primary production, and physically, by enhanced upwelling, to increase the supply of a wider range of naturally-occurring nutrients from deeper water. The latter technique would also release additional CO₂, thereby greatly reducing its potential for climate mitigation (Bauman et al., 2014), and with risk of other undesirable climatic consequences (Kwiatkowski et al., 2016b). Because of these uncertainties, enhanced upwelling is not considered further as a mitigation measure.

5.5.1.2.2 Ocean fertilization

The potential mitigation benefits of ocean iron fertilization have been investigated experimentally, also by modelling and by observations of natural system behaviour (Keller et al., 2014; Bowie et al., 2015; Kearney et al., 2015; Tagliabue et al., 2017). The 13 experimental studies to date (7 in the Southern Ocean, 5 in the Pacific, and one in the sub-tropical Atlantic) have shown that primary production can be greatly enhanced by the addition of iron to certain ocean regions (Yoon et al., 2016). However, most of the additional carbon uptake is only temporarily removed from the atmosphere. Iron fertilisation implemented on a climate-scale, if allowed, would involve many technical difficulties in verification of the enhanced carbon sequestration, and in monitoring its potentially-adverse impacts (Bowie et al., 2015; Williamson and Bodle, 2016). There are also many contested governance issues (Williamson et al., 2012; Boyd and Bressac, 2016; Fuentes-George, 2017; McGee et al., 2018), particularly for the Southern Ocean where ocean iron fertilization is considered to be most effective (Robinson et al., 2014). Ocean iron fertilization is regulated by the London Protocol, with amendments prohibiting such action unless constituting legitimate scientific research, authorized under permit (see Section 5.5.4.1).

Open ocean fertilization by macro-nutrients (e.g., nitrate) has also been proposed, with modelled potential for gigaton-scale carbon removal (Harrison, 2017). Similar technical and governance considerations would, however, apply with regard to the quantification of mitigation benefits, the monitoring of potential adverse impacts, and the political acceptability of large-scale deployment.
5.5.2 Ocean-based Adaptation

The IPCC Fifth Assessment Report (AR5) concluded, with high agreement but limited evidence, that climate change impacts on human settlements and communities could be reduced through coastal protection activities (Wong et al., 2014). Although an assessment was made on reported implementation of adaptation practices, there was limited evidence of the context-specific application of principles or approaches, and this was considered a key research gap. Furthermore, adaptation activities remained aspirational, whilst inappropriate coastal development and ecosystem degradation continued.

In this special report, Section 5.5.2 assesses human adaptation response to climate change within the framework that includes risk, its reduction, and adaptation pathways towards resilience as outlined in Cross-Chapter Box 1 in Chapter 1. Residual risk, however remains where hazard, vulnerability and exposure intersect, subsequent to the reduction of the latter two components through an adaptation pathway response.

For an introduction to climate change adaptation within this special report, including climate change adaptation in natural and human systems, see Section 1.5.2. Section 5.5.2 begins with an assessment of adaptation frameworks (Section 5.5.2.1), followed by assessments of the following three main focus areas of adaptation response measures: ecosystem-based (5.5.2.2), built infrastructure (5.5.2.3) and socio-institutional (5.5.2.4) adaptation (see Table 5.7 for a description of these categories). Finally, the costs of, and limits to, adaptation is assessed in Section 5.5.2.5. Section 5.5.2 avoids, where possible, focused discussion of response to sea level rise and extreme events (including heat waves, and compound and cascading events) to avoid duplication with Chapter 4 (see Table 4.6 for examples) and Chapter 6, respectively.

<table>
<thead>
<tr>
<th>Section</th>
<th>SROCC description</th>
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<tbody>
<tr>
<td>5.5.2.2 Nature-based/Ecosystem-based Adaptation</td>
<td>The use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change. Ecosystem-based adaptation uses the range of opportunities for the sustainable management, conservation, and restoration of ecosystems to provide services that enable people to adapt to the impacts of climate change. (Narayan et al., 2016b; Moosavi, 2017)</td>
</tr>
<tr>
<td>5.5.2.3 Built Environment Adaptation Solutions</td>
<td>Adaptation solutions pertaining to coastal built infrastructure and the systems that support such infrastructure. (Mutombo and Ölçer, 2016b; Forzieria et al., 2018)</td>
</tr>
<tr>
<td>5.5.2.4 Socio-Insitutional Adaptation Solutions</td>
<td>Adaptation responses within human social, governance and economic systems and sectors, including community-based adaptation. (Oswald Beiler et al., 2016; Thorne et al., 2017)</td>
</tr>
<tr>
<td>5.5.2.4.1 Community-based Adaptation</td>
<td>Local, community-driven adaptation. Community-based adaptation focuses attention on empowering and promoting the adaptive capacity of communities. It is an approach that takes context, culture, knowledge, agency, and preferences of communities as strengths. (Archer et al., 2014a; Shaffir et al., 2017a)</td>
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5.5.2.1 Ocean-based Climate Change Adaptation Frameworks

As is outlined in Cross-Chapter Box 1 in Chapter 1, adaptation action in pursuit of a climate resilient development pathway is likely to have a deeper transformative outcome than stepwise or piecemeal responses. Climate change adaptation responses are more effective when developed within institutional frameworks that include effective planning and across-sector integration. In Section 5.5.2.1, an assessment is made of the current status of the development of adaptation frameworks for marine systems and their implementation, and conditions that could support their further development.

Since AR5, a substantial body of literature has developed that reports on research support for adaptation planning. This includes vulnerability assessments on ecosystems (Kuhfuss et al., 2016), species (Cheung et al., 2015; Cushing et al., 2018), communities (Islam et al., 2013; Himes-Cornell and Kasperski, 2015), and integrated assessments of all of the above (Peirson et al., 2015; Kaplan-Hallam et al., 2017; McNeeley et al.,...
2017; Ramm et al., 2017a; Mavromatidi et al., 2018). Seasonal and decadal forecasting tools have improved rapidly since AR5, especially in supporting management of living marine resources (Payne et al., 2017; Tommasi et al., 2017) and modelling to support decision-making processes (Čerkesova et al., 2016; Chapman and Darby, 2016; Jiang et al., 2016; Justic et al., 2016; Joyce et al., 2017; Mitchell et al., 2017). Decision-making processes are supported by economic evaluations (Bujosa et al., 2015; Jones et al., 2015), evaluations for ecosystem services (MacDonald et al., 2017; Micallef et al., 2018) and participatory processes (Byrne et al., 2015). A growing body of literature is emerging on the development of adaptation pathways, frameworks and decision making (Buurman and Babovic, 2016; Dittrich et al., 2016; Michailidou et al., 2016; Cumiskey et al., 2018; Osorio-Cano et al., in press), as well as the development of indicators to support evaluation of adaptation actions (Carapuço et al., 2016; Nguyen et al., 2016) through monitoring frameworks (Huxham et al., 2015).

There is high agreement on the need for the integration of scientific research into management decision-making for climate adaptation (Endo et al., 2017; Thorne et al., 2017), particularly through better understanding coastal ecosystems and human adaptation processes (Dutra et al., 2015; Cvitanovic et al., 2016). This includes consideration of non-climate change-related factors. Areas of scientific research that are considered useful to facilitate the use of scientific knowledge for climate adaptation include multiple stakeholders participatory planning (Archer et al., 2014b; Abedin and Shaw, 2015), transboundaries ocean management (Gormley et al., 2015; Williams et al., 2016), scientific research to deepen understanding of ecosystem- (Hobday et al., 2015b; Dalyander et al., 2016; McNeely et al., 2017; Osorio-Cano et al., in press), community-based adaptation with socio-economic outcomes (Merkens et al., 2016) and development of climate resilient development pathways (high confidence). In addition, research on applying ‘big data’ and high end computational capabilities could help develop a comprehensive understanding of climate and non-climate variables in planning for coastal adaptation (Rumson et al., 2017). Knowledge from these research areas could improve the planning, implementation and monitoring of climate adaptation actions for marine systems (high confidence).

Despite the increased development of scientific research to inform climate adaptation since AR5, examples of evaluations of the planning, implementation and monitoring of adaptation actions remain scarce (Miller et al., 2017). In a global analysis of 401 local governments, 15% reported on adaptation actions, and 18% reported on planning towards adaptation policy (Araos et al., 2016). The 15% was dominated by large cities in high income countries (Araos et al., 2016). Particularly, integrated adaptation planning with non-climate change related impacts remains an under-achieved ambition, especially in developing countries (Finkbeiner et al., 2018).

Overall, challenges with adaptation planning include uncoordinated, top-down approaches, a lack of political will and insufficient resources (Elias and Omojola, 2015). For example, in some countries, despite investment in research and science having improved local government practitioners’ technical and cognitive ability, implementation of adaptation actions has been impeded by budget cuts and lack of national level political support (Porter et al., 2015). In a survey of 150 Pacific Northwest coastal natural resource managers, time, money and staff resources were their largest adaptation obstacles. Also, different levels of access to information corresponded with differences in preparedness and willingness to plan for adaptation (Thorne et al., 2017).

A number of key principles are important in developing adaptation frameworks. A robust but flexible approach can deliver adaptation responses across a range of projected climate scenarios. This can be achieved by accounting for deep uncertainty through well-coordinated participatory processes (Jiao et al., 2014b; Dutra et al., 2015; Dittrich et al., 2016; Buurman and Babovic, 2017). Monitoring programmes are important for effective implementation of adaptation actions (Barrett et al., 2015; Bell et al., 2018b). Other principles include taking a whole system approach that addresses underlying socio-ecological compounding factors (Sheaves et al., 2016b), including cost-benefit analyses (André et al., 2016) and providing co-benefits for human development and the environment (Wise et al., 2016).

There is a notable dearth of studies from African and Caribbean least developed countries and small island developing states (Kuruppu and Willie, 2015) and few locally-focused coastal climate change adaptation planning tools (Torresan et al., 2016). This is partly due to the highly complex nature of adaptation response in these countries. In contrast, the coastal adaptation framework literature is dominated by Australian, North
American and European cities. There are large sub-national variations even in developed countries. For example, in Australian coastal areas, the small number of available case studies suggests that only a few coastal local governments have begun adaptation planning (Ramm et al., 2017b). Interest in adaptation effort is more likely in individuals experiencing extreme weather (Ray et al., 2017). For example, nature-based solutions and improved coordination effort increased in USA cities located in Atlantic coastal regions following Hurricane Sandy (Oswald Beiler et al., 2016). Positive responses, reported from developing countries, were expressed in terms of successes with community- and ecosystem-based adaptation approaches. The difference in adaptation assessments between developed and developing economies and cities suggests some divergence in approach to addressing climate change (medium confidence).

To pursue a climate resilient development pathway, more extensive learning processes are necessary to build decision-makers’ capacity to tackle systemic drivers, and to scrutinize potentially maladaptive infrastructural investments (Wise et al., 2016). This will require effective coordination across a range of stakeholders within and between organizations, and substantial effort is required in developing countries to close the gap in adaptation planning with developed countries (high confidence). In contrast with the many examples of proposed frameworks for adaptation, there remains relatively little literature that has assessed the success of implementing such frameworks, possibly due to the time-lag between implementation, monitoring and evaluation and reporting. In terms of developing climate resilient development pathways (see Cross-Chapter Box 1 in Chapter 1), there was substantial support for ‘no regrets’ approaches addressing both proximate and systematic underlying drivers of vulnerability (Sánchez-Arcilla et al., 2016; Pentz and Klenk, 2017; Zandvoort et al., 2017) with leadership, adaptive management, capacity and the monitoring and evaluation of actions deemed necessary governance responses (Dutra et al., 2015; Doherty et al., 2016).

Sections 5.5.2.2 to 5.5.2.4 assess three categories of adaptation: nature-based, built environment and socio-institutional solutions (including community-based adaptation) respectively. The categorization used here is consistent with recent treatment of coastal adaptation issues in AR4 and AR5 (Wong et al., 2014). Community-based adaptation typically refers to bottom-up, community-driven adaptation responses in which public participation processes are central to their implementation. Examples of reported adaptation solutions are given in Table 5.8.

Table 5.8: Summary table of some climate change impacts upon ecosystems and human systems, and an assessment of adaptation responses, reported in Secton 5.5.2. See Section 5.3.3 for a discussion of key coastal ecosystems and their services vulnerabilities to multiple climate hazards for an in-depth assessment of the hazards, and see Sections 5.4.1.3 and 5.4.2 for a discussion on the impact of climate hazards on ecosystems and human wellbeing:

<table>
<thead>
<tr>
<th>Ecosystems/ Human systems</th>
<th>Hazards</th>
<th>Biophysical drivers</th>
<th>Adaptation response</th>
<th>Assessment (Confidence)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coral reefs</td>
<td>Bleaching, Inundation, Dissolution, Borealosion, Structural damage</td>
<td>Ocean warming, Sea level rise, Ocean acidification, Extreme storm events</td>
<td>Preventative or adaptive restoration, assisted evolution</td>
<td>Active restoration, still in its infancy, can be applied in a wide range of ecological and socio-economic applications (high). Coral epigenetics, ‘assisted colonization’, ‘assisted evolution’, novel coral–symbiont associations and coral microbiome manipulations as adaptive management tools for reef rehabilitation (low)</td>
</tr>
<tr>
<td>Mangroves, coastal wetlands, seagrass beds and saltmarshes</td>
<td>Storm damage, Inundation</td>
<td>Extreme storm events, sea level rise</td>
<td>Synergistic adaptation responses, socio-institutional responses</td>
<td>Employing synergistic ecosystem-based adaptation responses are a cost-efficient and effective response, especially when employed alongside socio-institutional responses (high).</td>
</tr>
<tr>
<td>Category</td>
<td>Example</td>
<td>Example</td>
<td>Example</td>
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<td>---------------------------------------</td>
<td>--------------------------------------------------------------------------</td>
<td>--------------------------------------------------------------------------</td>
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<td></td>
</tr>
<tr>
<td>Coastal dunes and sandy beaches</td>
<td>Erosion, Inundation, Fire damage</td>
<td>Extreme storm events, sea level rise, heat waves/ drought</td>
<td>Rate of loss of sand and level of participation of stakeholders are important factors (medium)</td>
<td></td>
</tr>
<tr>
<td>Species</td>
<td>Direct physiological impacts, Habitat transformation, Foraging success decline, Reproductive failure</td>
<td>Various climate change driven variables (direct and indirect)</td>
<td>Species range shift, behavioural change, human facilitated responses including governance, conservation etc.</td>
<td></td>
</tr>
<tr>
<td>Integrated coastal zone management</td>
<td>Loss of biodiversity and ecosystem change, Loss of economically important species, Infrastructure damage</td>
<td>Climate drivers of ecosystem change, sea level rise and storm surge</td>
<td>Virtually certain to be winners and losers, knowledge gaps need to be filled</td>
<td></td>
</tr>
<tr>
<td>Tourism</td>
<td>Tourism industry decline (changing behaviour and choices), Infrastructure loss, Resource declines</td>
<td>CO₂ emissions, changes to ecosystems, sea level rise/ storm surge, climate impacts on resources e.g., water availability</td>
<td>High confidence in efficacy of stakeholder participation and MPAs, but some resistance from conservative administrations to the former, medium confidence in IK/LK (Cross-Chapter Box 3 in Chapter 1), high agreement on need for transboundary agreements</td>
<td></td>
</tr>
<tr>
<td>Fisheries</td>
<td>Reduction in landings and revenues, Changes in species composition, Ecosystem change</td>
<td>Climate drivers of changes in species distributions</td>
<td>High confidence that climate change will impact upon tourism, high agreement, limited evidence that a range of adaptation responses will be required locally, likely winners and losers</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Optimal fisheries management including adaptive management, seasonal-to-decadal forecasting, capacity development, target fishery switching, fish aggregating devices, transnational agreements, knowledge development and community responses</td>
<td>High agreement that optimal fisheries management integrated with community responses will be critical to offsetting global fisheries landings declines, high agreement that knowledge gaps in fisheries management need to be filled for successful adaptation, medium confidence in ability of community-based responses to offset social impacts</td>
<td></td>
</tr>
</tbody>
</table>
### Aquaculture
- **Biophysical impacts upon farmed species**
- **Disruption to operations**
- **Salinity intrusion**
  
### Physiological climate change
- **Drivers**, **Extreme weather events**, **Sea level rise**

### Ecosystem approach to aquaculture
- **(EAA)**, **Integrated Coastal Zone Management**, **translocation and polyculture**, **socio-ecological resilience**

### High confidence in EAA and ICZM approaches, high confidence for translocation and polyculture, resilience and climate compatible development very important

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**Notes:**
- (a) synergistic adaptation response: multiple adaptation actions that complement and act synergistically with each other.

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### 5.5.2.2 Nature-based / Ecosystem-based Adaptation

Section 1.5.2.1 introduced the concept of evolutionary adaptation as a species response to climate change, which can cascade to impact upon ecosystem structure and function. An assessment of species and ecosystem responses to climate, and other drivers of change, is provided in Sections 5.2 and 5.3 with some discussion in 5.5.2.4. Section 1.5.2.2 introduces human systems’ adaptation responses and Section 4.4.3.3 provides an overview of Ecosystem-based Adaptation (EbA), covering global distribution of coastal ecosystems, different types of EbA responses and governance mechanisms to advance these responses, including international funding programs, national policies and community responses. Section 5.5.2.2 assesses human adaptation response to climate change using ecosystem- and nature-based approaches within key coastal ecosystems.

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### 5.5.2.2.1 Adaptation in coral reefs

The continuous global degradation of warm-water coral reef ecosystems and the failure of traditional conservation acts to revive most of the degrading reefs (Rinkevich, 2008; Miller and Russ, 2014), all call for novel adaptive management. Rehabilitation of reef ecosystems may progress by implementing either preventive ‘passive’ restoration or adaptive active restoration that together lead to a suite of diverse restoration scenarios (Miller and Russ, 2014; Rinkevich, 2017) (high confidence) (see Box 5.4).

Driven by the qualities of the silviculture (forestation) approaches, studies (Rinkevich, 1995; Rinkevich, 2005; Rinkevich, 2006; Rinkevich, 2008; Bongiorni et al., 2011) have proposed a two-step restoration strategy termed gardening of denuded coral reefs. In the first step, a large pool of coral colonies (derived from coral nubbins and fragments, and from sexually derived spat) are farmed in underwater nurseries, preferably on mid-water floating devices. These mid-water floating nurseries are installed in sheltered zones, in which coral material can be cultured for up to several years. In the second step, nursery-grown coral colonies, together with recruited associated biota, are transplanted to degraded reef sites (Shafrir and Rinkevich, 2008; Mbie et al., 2010; Shaish et al., 2010b; Shaish et al., 2010a; Bongiorni et al., 2011; Horoszowski-Fridman et al., 2011; Linden and Rinkevich, 2011; Mbie et al., 2013; Cruz et al., 2014; Chavanich et al., 2015; Horoszowski-Fridman et al., 2015; Lirman and Schopmeyer, 2016; Montoya Maya et al., 2016; Ng Chin Soon et al., 2016; Lohr and Patterson, 2017; Rachmilovitz and Rinkevich, 2017). Active restoration of coral reefs, while still in its infancy and facing a variety of challenges (Rinkevich, 2015b; Hein et al., 2017), is now capable to handle a wide range of ecological and socio-economic applications (Rinkevich, 2014; Rinkevich, 2015b; Rinkevich, 2017) (high confidence).

Recent studies focusing on ecological engineering approaches (that also include the gardening approach) (Rinkevich, 2014; Forsman et al., 2015; Coelho et al., 2017; Horoszowski-Fridman and Rinkevich, 2017; Linden and Rinkevich, 2017; Rachmilovitz and Rinkevich, 2017), species interactions that reef managers may utilize to facilitate the restoration of corals and/or the augmentation of functional diversity (including microbiome) (Casey et al., 2015; Horoszowski-Fridman and Rinkevich, 2017; Shaver and Silliman, 2017), the transplantation of whole niches (Shaish et al., 2010b; Gómez et al., 2014) and keeping/enhancing the genetic diversity (Iwao et al., 2014; Drury et al., 2016; Horoszowski-Fridman and Rinkevich, 2017). Active restoration becomes a major tool for reef rehabilitation in many countries worldwide (representing all major...
reef regions) (Rinkevich, 2014; Rinkevich, 2015b), particularly in Southeast Asian region (Hilmi et al., 2018). However, there is little data revealing how resistant these corals are to global change drivers (Shaish et al., 2010b; Shaish et al., 2010a) or how the nursery time improves biological traits in coral transplants e.g., reproduction (Horoszowski-Fridman et al., 2011).

While the coral gardening approach has been tested in the last two decades, several new approaches are now discussed in the literature (Box 5.4). Additionally, coral epigenetics as an adaptive management tool for reef rehabilitation (low confidence) has gained some support from studies showing coral adaptation correlating to epigenetics (Brown et al., 2002; Horoszowski-Fridman et al., 2011; Palumbi et al., 2014; Putnam and Gates, 2015; Putnam et al., 2016). Also, a recent study in the Great Barrier Reef (Drake et al., 2018) reveals that coral cells possess the molecular machinery that helps compensate for the effects of global change, such as ocean acidification.

While predictions for the future reef statuses under climate change are gloomy (high confidence), contemporary active reef restoration (Box 5.4) has a promise to be used as a major management and adaptive tool for the rehabilitation of the global change impacts on coral reefs (high confidence). The new suggested adaptive management tools (conducted to harness basic science to develop corals with capacity to withstand climate change impacts; Box 5.4) may further improve the active reef restoration outcomes (low confidence). Results will be intensified with the employment of ecological engineering approaches (high confidence). It is envisaged that the ‘reefs of tomorrow’ will not resemble the reefs of today or the pristine reefs (Box 5.4) (high confidence) (Section 5.3.3.7), thus it is suggested to prepare a coral restoration toolbox (Rinkevich, 2014; Rinkevich, 2015a; Rinkevich, 2015b) for cases and the time it is needed.

**Box 5.4: Coral Reef Restoration as Ocean-based Adaptation**

Global change induced by human activities is impacting all warm-water corals (and the reef structures they form) (Section 5.2.2.3.3). Impacts are rapidly increasing in scale and intensity, exposing coral reefs to enhanced degradation rates and diminishing capacities to maintain ecological resilience, to absorb disturbances, and to adapt in time to the expected changes (Box 5.1) (Graham et al., 2014; Rinkevich, 2015a; Harborne et al., 2017). However, traditional coral reef conservation measures, aimed at protecting reefs from human activities and allowing natural processes to mitigate human impacts, are increasingly being acknowledged as insufficient to address climate change, Section 5.2.3.2). Instead, restoration measures are being called for (Rinkevich, 1995; Rinkevich, 2000; Barton et al., 2017). However, various recently suggested approaches for rebuilding coral reef resilience such as the ‘assisted colonization’ (actively moving species that are confined to disappearing habitats) (Hoegh-Guldberg et al., 2008; Chauvenet et al., 2013), the ‘assisted evolution’ (aiming to develop corals resistant to climate change via accelerated natural evolution processes), novel coral–symbiont associations (McIlroy and Coffroth, 2017) and coral microbiome manipulation (Bourne et al., 2016; Sweet and Bulling, 2017; van Oppen et al., 2017) are still at the ‘proof-of-concept’ stage. Others, such as the use of artificial reefs (Ng Chin Soon et al., 2017) are limited in impacts, and all are also revealing considerable challenges (Riegler et al., 2011; Coles and Riegler, 2013; Ferrario et al., 2014).

Today, the most tested reef restoration approach involves ‘coral gardening’, a two-step process, similar to silviculture concepts and practices (Rinkevich, 2006). About 90 coral species and over 100,000 colonies that were farmed in various archetypical nurseries across the world, followed by a broad range of transplantation tactics showed high survivability, growth rates and reproduction (Rinkevich, 2014). Whereas the ‘coral gardening’ restoration techniques are still lagging behind restoration practices developed for terrestrial habitats, reef restoration has advanced greatly in the last two decades (Rinkevich, 2014; Lirman and Schompmeyer, 2016). This advancement includes economic considerations (Rinkevich, 2015b; Barton et al., 2017; Flores et al., 2017; Hein et al., 2017) and citizen science involvement (Cruz et al., 2014; Hesley et al., 2017). Recent studies (Frias-Torres and van de Geer, 2015; Lirman and Schompmeyer, 2016; Montoya Maya et al., 2016; Jacob et al., 2017; Rachmilovitz and Rinkevich, 2017) have added ecological engineering perspectives, while also targeting the enhanced conscription of reef dwelling invertebrates and fish and
augmented recruitment of coral spats. The addition of the assisted evolution tools, when developed, may improve coral restoration outcomes (van Oppen et al., 2015; van Oppen et al., 2017).

However, the ‘coral gardening’ approach is facing two foreseen challenges. The first is the scaling up issue, the viability of this approach as a large-scale restoration measure at the level of hundreds of thousands/millions of coral colonies/site (aiming at the elucidating of the sum ecological attributes at large-scale approaches) (Rinkevich, 2014). The second challenge (Box 5.4, Figure 1), is the effectiveness of active reef restoration to mitigate or rehabilitate global change impacts (Shaish et al., 2010a; Schopmeyer et al., 2012; Coles and Riegler, 2013; Hernández-Delgado et al., 2014; Rinkevich, 2015a; Wilson and Forsyth, 2018) and whether it is applicable for tracking future increases in sea level (Perry et al., 2018), primarily in low lying ocean states. Also, the literature reveals a wide range of calculated restoration costs that did not consider the high monetary value of the coral reefs’ ecosystem services (Bayraktarov et al., 2016; Rinkevich, 2017).

Yet, it has been proposed (Rinkevich, 2015a) that ‘coral reef gardening’ has the capacity to enhance adaptability of coral reef organisms to climate change. The implementation of reef restoration approaches and the climate change induces poleward range expansion of corals’ reef habitats (Precht and Aronson, 2004; Yara et al., 2011; Madin et al., 2016), altogether the evidence suggests that the ‘reefs of tomorrow’ would not resemble the ‘reefs of today’ (Box 5.4, Figure 1) (Rinkevich, 2008; Ban et al., 2014) (high confidence). The very high vulnerability of coral reefs to warming over reefs, ocean acidification, the increase in storms’ intensity and severity and sea level rise under climate change (AR5 WG2;), including enhanced bioerosin (Schönberg et al., 2017) (high confidence) point to the importance of considering both mitigation (Section 5.5.1) and adaptation for coral reefs, the latter includes active restoration measures and for improved cultivation and transplantation methodologies.

**Box 5.4, Figure 1:** Coral reef restoration as an ocean-based adaptation tool to climate change. The figure depicts 5 reef states (in varying ecological complexity [x-axis] and service levels [y-axis]; circles 1-5) including two extreme statuses (the pristine versus and the highly degraded states, circles 1 and 2, respectively), as two ‘restored reef-state’ scenarios (circles 3, 4), leading to the state of the ‘reef of tomorrow’ (circle 5). The route from the state of the ‘reef of tomorrow’
(circle 5) to a pristine state (circle 1) is doubtful and is still at a theoretical level. Based on Rinkevich (2014). A-C
represent different reef statuses. A = a denuded knoll at the Dekel Beach, Eilat, Israel before reef transplantation was
taken (November 2005; Photo: Y. Horoszowski-Fridman); B = the same knoll, restored. More than 300 nursery-grown
colonies of 7 coral species were transplanted during three successive transplantation (2005, 2007, 2009) on this knoll,
11 years later (June 2016; photo by Shai Shafir). The knoll is surrounded by reef inhabiting schools of fish. C = a
pristine reef, not existing under current and anticipated reef conditions. Restoration scenarios are developed along paths
from a degraded reef (low ecological complexity, minimal reef services) toward a healthy ‘reef of tomorrow’, passing
through two restored reef states that are impacted by climate change (CC) (Shaish et al., 2010a; Schopmeyer et al.,
2012; Hernández-Delgado et al., 2014; Rinkevich, 2015a). The employment of ecological engineering approaches may
help in moving the ecological states from either restored reef to the ‘reef of tomorrow’ status.

[END BOX 5.4 HERE]

5.5.2.2.2 Mangrove and other coastal ecosystems

Mangrove swamps provide significant coastal protection services from extreme storm events, providing
supporting services through increased sedimentation rate (Hayden and Granek, 2015) and provisioning
services for local communities (Section 5.4.1.3). Mangroves can also provide carbon mitigation and job
creation co-benefits (for examples through Reducing Emissions from Deforestation and Forest Degradation
(REDD+) programs) when managed properly (5.5.1.1), and there is evidence of their value for aquaculture
initiatives (Huxham et al., 2015; Ahmed and Glaser, 2016). When used in combination with other adaptation
responses, their value increases. For example in Bangladesh sedimentation rates increased when mangroves
were planted on the lee side of an oyster farm, with the latter serving to reduce wave energy reaching the
mangroves (Ahmed and Glaser, 2016). On the other hand, loss of mangrove swamps and fringing coral reefs
would result in the loss of coastal protection services by these ecosystems against future cyclones in
Bangladesh (Ataur Rahman and Rahman, 2015; Ahmed and Glaser, 2016), so effective management is
important.

There are many examples of coastal protection services provided by mangrove forests, which have been
restored through community-based mangrove restoration programs, and such programs can provide
additional community and carbon sequestration benefits. There are also, however, examples where such
initiatives have resulted in maladaptation, generally as a result of poor governance processes or lack of
community compliance with restoration plans. This emphasizes the need for effective implementation of
adaptation responses with broad community participation. In a specific example of mangrove management
for climate adaptation that included community-focused mangrove management plans in Vietnam, the
importance of providing technical guidance to local communities as part of the adaptation action was noted
(Nguyen et al., 2017), along with the need for the incorporation of mangrove regulations into local planning
instruments (Sierra-Correa and Cantera Kintz, 2015). Community participation in the development climate
change adaptation for mangroves could improve its outcomes and co-benefits (medium evidence, high
agreement).

Coastal developments and coastal squeeze transform wetlands and marshlands which reduces the ability of
these ecosystems to provide protection from wave and storm impacts, whilst releasing carbon into the
atmosphere (Wong et al., 2014). The level of protection seagrass beds provide depends partly upon their
ecological health, so their optimal management is an important adaptation response, especially when paired
with synergistic ecosystems like mussel beds on the seaward side of seagrass beds to reduce wave energy
and erosion (Ondiviela et al., 2014).

Coastal dune systems are widely transformed globally; human disturbance and a low stabilizing ability of
vegetation are key causes of degradation (Ciccarelli et al., 2017) with restoration effort supported by dune
builder species and sand replenishment schemes. Section 4.4.4.1 provides an overview of sediment-based
adaptation response measures, including cost estimates for beach nourishment and dune maintenance, a
discussion of co-benefits and drawbacks of combining hard and soft infrastructure measures, and challenges
with sourcing sediment for beach replenishment. There are a number of examples of sand replenishment
projects as a soft measures approach to combatting erosion; see Vikolainen et al. (2017) for an example.
Such projects have, however not always been successful due to, for example fire damage (Shumack and
Hesse, 2017) or the rapid loss of sand within replenishment schemes due to coastal processes, and in some
cases due to stakeholder rejection of adaptation activities (Pranzini, 2017), with the latter suggesting
insufficient stakeholder engagement prior to project operationalization. An example of an intermediate adaptation approach (between natural and built infrastructure) was reported in a Mauritian demonstration project, where a gravel beach was constructed to protect low-lying developments (Onaka et al., 2015).

Combining engineered and sand replenishment responses are considered complementary approaches (Martínez et al., 2017).

5.5.2.2.3 Ecosystem-based adaptation assessment

This ecosystem-based adaptation (EBA) assessment was supported by a growing body of literature and increased international funding for this approach, yet a synthesis of context-specific application and cost-effectiveness is a remaining gap (Narayan et al., 2016a). ‘Soft’ engineering options have a range of co-benefits like increasing ecological complexity with multiple services provided and a wide range of economic benefits for society and resilience to climate change (Perkins et al., 2015; Perry, 2015; Moosavi, 2017; Scarano, 2017). In their analysis, Narayan et al. (2016a) found that coral reefs and salt-marshes performed best at reducing wave heights, whilst salt-marshes and mangroves were two to five times cheaper than submerged breakwaters for wave heights less than half a meter. The application of EBA approaches can be made more effective by incorporating traditional practices and knowledge (Ataur Rahman and Rahman, 2015).

The application of synergistic combinations of ecosystems can provide a range of co-benefits, and this approach is strengthened when combined with socio-institutional approaches (Kochnower et al., 2015; MacDonald et al., 2017) (medium evidence, high agreement). As important as the outcomes of implementation of EBA projects is, the learning process to improve and refine approaches used, given the context-specific nature of their application, is crucial (Sutton-Grier et al., 2015). The integration of traditional and natural infrastructure as an adaptation response is strongly supported by the literature (Perkins et al., 2015; Sutton-Grier et al., 2015; Sánchez-Arcilla et al., 2016; van der Nat et al., 2016).

The lack of the inclusion, and economic undervaluation, of intertidal ecosystem services undermines coastline management, and this is compounded by ecological data gaps (Perkins et al., 2015). In an analysis of urban climate planning, a good awareness of EBA measures was reported in Europe, but baseline information was lacking, along with convincing implementation actions (Geneletti and Zardo, 2016). In summary, the benefits of EBA are widely acknowledged in the literature, and the sum of these benefits strongly suggests that they should be a key tool used for coastal climate change adaptation, but with an acknowledgement of the limitations inherent in this adaptation response (high confidence). As with AR5, further research evaluating natural infrastructure is required (Roberts et al., 2017).

Ecosystem-based Adaptation (EBA) is considered to be a cost-effective coastal protection tool that has multiple co-benefits. Management of ecosystems provides the opportunity for job creation, as communities living adjacent to ecosystems, like mangrove forests, are incorporated into mangrove rehabilitation and maintenance programs. Mangrove restoration and conservation also creates carbon sequestration opportunities, and importantly, such ecosystems under effective management can recover from damage after extreme events (whereas built infrastructure requires rebuilding). This effectivly gives natural-based adaptation solutions a cost-effective advantage over built infrastructure, besides providing other co-benefits, including biodiversity conservation, utility and recreational value.

5.5.2.2.4 Species adaptation

There is a substantial body of peer review literature reporting marine species’ climate change adaptation responses (Crozier and Hutchings, 2014; Miller et al., 2017; Diamond, 2018), and further examples of individual species’ responses to climate change (Section 5.2.3, 5.3.3). Sections 5.2.3 and 5.3.3 discuss three main types of adaptation responses: genetic adaptation to changed conditions, phenotypic plasticity and mobility towards more favourable conditions. This assessment concludes that whilst these forces may already be driving species adaptation, climate risk on marine species cannot be eliminated through adaptation and that there are large variations between species and populations (Gienapp and Merilä, 2018).

An accurate understanding of climate change impacts upon species, their sensitivity and adaptive capacity and consequent ecological effects is required to estimate extinction risk, so that an appropriate management response can be developed (Butt et al., 2016). It is particularly important to consider both indirect as well as direct physiological impacts. For example, warming of the Arctic might be beneficial in terms of
physiological tolerance, but loss of sea-ice habitat could reduce (or alternatively increase) foraging habitat (Castellini, 2017). Human management of ecosystems for climate change will, therefore, require a focus beyond traditional mandates (Hobday et al., 2015a). Species adaptation responses that address both the direct and indirect impacts of climate change, as well as changes to ecosystems supporting species, for example transformation through alien invasive species (Whitfield et al., 2016), will require broader consideration than traditional species management practices (medium evidence, good agreement). This will include the development of international collaborations and databases to improve ocean-scale understanding of climate change impacts (Okey et al., 2014; Young et al., 2015). A key knowledge gap of where the critical thresholds are for irreversible change for species remains, and this should be prioritized (Powell et al., 2017).

5.5.2.3 Built Environment and Engineered Climate Change Adaptation Responses

Many of the world’s great cities lie within the coastal region, and climate change impacts put these cities, their inhabitants and their economic activities, at risk. WG2 AR5 Chapter 14 reported that integration of climate risk into built infrastructure design in newer projects was emerging in the literature, whilst describing two adaptation limitations; uncertainty around climate projections and the risk of maladaptation. Section 5.5.2.3 assesses the impacts of climate change upon the built environment and adaptation response and considers these two adaptation limitations.

5.5.2.3.1 Climate impacts upon built infrastructure

Changes in the following physical and climatic variables are most likely to affect the built environment: sea level, wave regime, salinity, wind, rainfall and humidity (Colin et al., 2016; Mutombo and Ölçer, 2016c; Antwi-Agyeia et al., 2018; Forzieria et al., 2018). The effect of these variables depends upon type of infrastructure, geographical location and level of economic and technological development (Carter, 2018).

Understanding the effect of these climate variables upon built infrastructure should dictate the adaptation course taken. In this section, an assessment is made of the impacts of the above variables on the built environment.

Sea level rise (SLR) causes, projections, impacts and adaptation response is dealt with in detail in SROCC Chapter 4. Wave regime and salinity act synergistically with SLR in coastal locations and are considered next. Increases in extreme ocean wave heights will affect coastal infrastructure and impact many vulnerable populations of low-lying islands (Vanem, 2015; Shope et al., 2016; Simon et al., 2016; Wadey et al., 2017). In the Western Pacific Solomon Islands, shoreline recession as a result of extreme events, seawalls and inappropriate development has badly damaged infrastructure in villages (Simon et al., 2016). Shoreline recession is positively associated with areas of high wave energy, so understanding the relationship between wave surge, wave energy and development is critical. Large wave events in the tropical Pacific are likely to decrease from December to February, but the top 5% of significant wave heights will increase from June to August (Shope et al., 2016). This increase, accompanied by changes in direction, will increase flooding and shoreline change. Wave height will increase and swell direction is projected to shift in a counter-clockwise direction in the North Pacific above 50° (Erikson et al., 2015). Climate variability, including the El Niño–Southern Oscillation (ENSO), North Atlantic Oscillation (NAO), and Pacific decadal oscillation (PDO), will have further, region specific impacts upon wave height, wind speed and mean sea level pressure gradient amplitude (Kumar et al., 2016).

Storm surge levels are projected to increase at locations along the North European coastline by the end of the century for both RCP4.5 and RCP8.5. South of 50 °N change is only projected towards the end of the century under RC 8.5 (Vousdoukas et al., 2016). Reduction in sea ice may result in an increase in wave height in the Gulf of St Lawrence due to reduced attenuation during the 21st Century, but no effects are expected from changes in wind regime (Ruest et al., 2016).

Much of the literature on wave studies is associated with ship movements and maneuverability (Kim et al., 2017; Hizir et al., 2018) rather than with built infrastructure. Coastal infrastructure on shorelines is designed to withstand specific wave-induced load based on historical wave data. These structures may therefore not be able to perform optimally under changing wave conditions. Furthermore, excessive deterioration of infrastructure may be experienced due to the overflow of sea water onto infrastructure as a result of overtopping of waves (Kong et al., 2013). As with sea level rise, to reduce the risk of material displacement, inundation and overtopping, adaptation to wave climate should be underpinned by the principles of retreat,
accommodate or protect (Demirbilek, 2013; Hunter et al., 2013; Gibbs, 2016; Ruleau and Rey-Valette, 2017).

Section 5.3.2 concludes that it is very likely that regional changes in coastal salinity correspond with the dominance of either evaporation or precipitation, with surface freshening at high latitudes due to ice melt. Increased salinity may accelerate deterioration of infrastructure erected in areas exposed to sea water, for example coastal heritage structures (Kaja and Mallic, 2017) or affect operations of infrastructure. Salinity affects built infrastructure mostly through inundation, which is closely related to sea level rise and/or storm surge, but infrastructure may also increase inundation through subsidence (Aslam et al., 2015). High salinity levels can affect the operation of wastewater treatment plants and reduce the quality of treated water, corrode subsurface pipes resulting in spillages or impact upon electrical systems and energy transmission (Rotzoll and Fletcher, 2012; Osman et al., 2017b). All underground pipelines and manholes below the 2 m contour in Durban, South Africa are considered to be vulnerable to saline intrusion (Friedrich and Kretzinger, 2012).

Corrosion of steel infrastructure, or deterioration of concrete through chloride ingress could both be made worse through the interactive effect of pollution (Hunting, 2016; Peng et al., 2017), with these authors suggesting that further research on such interactive effects is required. Despite this, salinity is globally identified as a low risk to the built environment (Escobar et al., 2016; Girjatowicz and Świątek, 2016).

Higher concentrations of CO₂ in the atmosphere affects temperature and increased temperatures affect humidity/amount of water evaporated per unit air. Despite some evidence that the performance, serviceability and safety of reinforced concrete and steel structures are affected by corrosion-induced deterioration as a result of increased humidity, among other climate factors, humidity is considered to be a relatively low risk to the built environment. Changing localized wind patterns and speeds impact mainly on operations and maneuverability of vehicles. Prevailing wind patterns based on available recorded data are traditionally used to account for wind in infrastructure design during development stage. Given uncertainty around downscaled climate projections, accurately accounting for changes in wind during design stage has now become very challenging with potential cost and maladaptation impacts (Hdidouan and Staffell, 2017).

It is very likely that coastal drainage infrastructure design based on current rainfall records will be unsuitable for future rainfall patterns (Villatoro et al., 2014; Elshobbagy et al., 2018). Storm water infrastructure is traditionally designed with a probability of extreme rainfall being statistically stationary. However, precipitation patterns are changing, with increasing impacts such as inundation, flooding, erosion, and deterioration of structures. Consideration of such changes is required by storm water design engineers to develop cost effective solutions for extreme rainfall events (Kong et al., 2013). Uncertainty regarding precipitation variability remains a major impediment for storm water infrastructure adaptation response.

Within the built environment, exposure to climate risks differs substantially between geographical locations, in-situ weather conditions and the nature and strategic objectives of the infrastructure in question (Colin et al., 2016; Mushtaq, 2018). Significant knowledge gaps remain in terms of the uncertainty in local impact projections, and its interaction with complex adaptive decisions being made (Mostafavi, 2018). The interaction between complex human systems and the variable nature of climate risks continues to present a major challenge in respect of developing a one-size-fits-all solution for infrastructure adaptation (Sekimoto et al., 2013; Rahimi et al., 2014; Mutombo and Ölçer, 2016a) (high confidence).

5.5.2.3.2 Decision making processes for built infrastructure adaptation

Within the built environment, adaptation seeks to moderate or avoid harm, or exploit beneficial opportunities (McCurdy and Travis, 2018). There are numerous politically-oriented-response methodologies (Colin et al., 2016; Azhoni et al., 2018; Forzieria et al., 2018) which aim at analysing and reviewing governance policies, legislative frameworks and institutional capacity. Perceptions around infrastructure management are shifting from the purely physical towards socio-technical systems (Chappin and Van der Lei, 2014). Many authors recommend the need for two levels of adaptation, technical and organisational (Antwi-Agyeia et al., 2018; McCurdy and Travis, 2018). Further, concerns have been raised about the ineffectiveness of the ‘predict-then-act’ approach, due to the large uncertainties with climate predictions; thereby raising the need for system flexibility when pursuing climate resilient development pathways (Antwi-Agyeia et al., 2018; McCurdy and Travis, 2018), also see Cross-Chapter Box: Risk and Resilience. In pursuit of the latter, built infrastructure adaptation becomes much more nuanced and steeped in human systems analysis (Bles et al., 2016; Hauge et al., 2017; Forzieria et al., 2018).
5.5.2.3.3 Built infrastructure adaptation: approaches and options

Given their location at the interface between sea and land, coastlines are vulnerable to climate change and other human impacts. In many countries, Integrated Coastal Management is widely implemented to address issues relating to coastal development and climate change (Rosendo et al., 2018). Strategies for coastal management generally revolve around an action-reaction approach on a post-disaster basis (Gibbs, 2016; Gracia et al., 2018) and it often entails the installation of hard engineering structures such as gabions, revetments, seawalls and breakwaters. Meanwhile, there is increasing evidence of the negative effect that hard engineering has on coastal ecosystems (Sheaves et al., 2016a) and a substantial body of literature is reporting implementation of an ecosystem-based adaptation approach to supporting built infrastructure (Gracia et al., 2018; Rosendo et al., 2018). For examples of the creation and restoration of wetlands, seagrass beds and dune vegetation as effective long term responses in combating erosion in built infrastructure see (see Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; Rosendo et al., 2018). This approach is considered more sustainable, adaptable, multi-functional, economically viable and has self-repair and recovery ability (robust evidence, good agreement) (Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; Rosendo et al., 2018).

In addressing climate change, there is an increasing focus on the concept of interdependencies in the built environment rather than quantifying specific risk factors (Chappin and Van der Lei, 2014; Antwi-Agyeia et al., 2018). Evidence is emerging that suggests that interdependencies during catastrophic events are a major cause of disruptions in societal processes (Higgins, 2013; Rodríguez et al., 2013). Minor disturbances originating from the environment can cascade, through a series of non-climate change-related interdependent impacts, to create major visible damage in an infrastructure system. As a result, the need for a systematic, cross-sector approach to addressing climate risk in built infrastructure systems is acknowledged (Rahimi et al., 2014; Araos et al., 2016); these may also be referred to as intra- and inter-organizational networks (Azhoni et al., 2018). Recent literature trends reveal a gap in understanding of how climate variables are linked to each other and to the built environment (Chappin and Van der Lei, 2014; Forzieri et al., 2018), and how they can be harmonized. Effective and informed decision-making takes into consideration a range of climate change and other human system factors within the built environment, including the complex interrelationship between the built environment and economic activities (Chappin and Van der Lei, 2014; Antwi-Agyeia et al., 2018; Venghaus and Hake, 2018).

Despite the risks posed by climate change in the built environment (Araos et al., 2016), the trade-off between costs and benefits of adaptation solutions (Baum, 2012; Ölçer and Ballini, 2015) is recognised as a major impediment. There is compelling evidence that this trade-off has led to maladaptation through pursuing suboptimal solutions (Azhoni et al., 2018), especially in developing countries with limited resources and competing developmental priorities (Hoggart et al., 2014). As a response, concepts of incremental and transformational adaptation actions are emerging in literature (Felgenhauer, 2015; Bosomworth et al., 2017; Mushtaq, 2018), and as outlined in Cross-Chapter Box 1. Incremental adaptation is defined as actions taken...
Resistance and resilience are considered to be two effective adaptation response strategies for the built environment (Lebaka et al., 2016; Mutombo and Ölçer, 2016b; Pursiainen, 2018). Resistance, which is analogous with incremental adaptation (Azhoni et al., 2018) consists of identifying various thresholds of resistance within the infrastructure networks and developing initiatives to increase such thresholds. Under the current condition of high uncertainty, responding by resistance is known to have limitations at some point along a scale of climate impact severity. A complementary approach is resilience (Lebaka et al., 2016; Pursiainen, 2018), which Azhoni et al. (2018) describes as transformational adaptation.

Understanding, at large scales, the interaction of resilience and resistance in built environment systems is considered an important knowledge gap. This includes studying cyclical patterns of growth, collapse, and renewal of resilience and resistance within the built environment (Pursiainen, 2018). Robust dynamic models for built environment resilience and resistance systems have been strongly advocated in the literature. Such models would enable the various role players to better prepare for extreme disruptions. Actions will require holistic implementation under three main pillars: Technology, Management, and Policy (Mutombo and Ölçer, 2016a). While it was found that technology and management actions are adequate to generate individual benefits at small scale, efforts to implement policy will need to tackle issues of interdependency on a national scale (Mileski et al., 2016), but with a cascade effect at local level. Cross-sectoral climate regulations may be necessary to ensure long term regional economic sustainability. There is an optimum level of regulation, given that excess regulation may potentially have an adverse effect (Chappin and Van der Lei, 2014). Moreover, stakeholders may present discriminative risk exposure to climate events, and this has led to an increasing recognition of the need to explore the cross scale dynamics for effectively implementing these strategies. Coastal governance and policy are dealt with in Section 5.4.4.

Because of the physical nature of the built environment, the literature has focused more on ‘hard’ measures (Chappin and Van der Lei, 2014; Taneja et al., 2014; Forzieria et al., 2018), as they are more visible, evident and easily measurable in demonstrating performance than behavioural, institutional and policy measures which are increasingly referred to as adaptive capacity (Carter, 2018; Woodruff, 2018). Neglecting potentially critical ‘soft’ measures may lead to inappropriate and costly adaptation actions (Barbier, 2015; Perkins et al., 2015). In this light, the concept of soft engineering is rapidly emerging as a valuable alternative within the built environment (de Vriend et al., 2014; Dyson and Yocom, 2015; Loke et al., 2015) to ensure that adaptation solutions sustain multiple services, provide greater economic benefits for society, and resilience to climatic change.

Paradoxically, concern is emerging about the high level of sophistication of built infrastructure. The more technologically sophisticated an infrastructure system is, the more sensitive it could become to disruptions (Rodrigue et al., 2013; Payne et al., 2017), thereby lowering its resilience. There is a gap in the literature on how balancing sophistication with resilience may be feasible, depending on management’s appetite and preferences for risk. Contracting insurance against extreme events is regarded as an effective risk transfer strategy for some (Fiksel et al., 2014) while others argue that insurance should never constitute a comprehensive risk treatment due to its limitations in scale coverage and its slow payment process (Müller et al., 2017). A need to enhance insurance solutions to deal with extreme events within the built environment sector is, therefore, becoming increasingly necessary (Bles et al., 2016; Colin et al., 2016; Forzieria et al., 2018).

Given that people naturally act on the basis of perceptions (Hopkins et al., 2016), there is high agreement that failure of existing tools that deal with climate uncertainty within the built environment is due to their inability to represent a person’s state of knowledge and preferences. This sentiment is strongly reflected in the literature (Antwi-Agyeia et al., 2018; Azhoni et al., 2018), emphasising with high confidence the need
for individual adaptive capacity and to factor environmental psychology into adaptation planning. This
underlies a strong need to assign higher value to intangible initiatives, such as climate education and
capacity building. Recent emerging literature strongly advocates for a shift to soft adaptation which implies
focussing on issues of psychology, culture, perceptions and education (Hauge et al., 2017; Azhoni et al.,
2018).

Meanwhile, in spite of the consensus on the need for adaptation, defining successful adaptation still remains
a challenge given that perceptions and risk appetites may substantially differ. Also, in view of the long term
nature of climate processes, adaptation is continuous. As new challenges emerge, questions of what actually
counts as successful adaptation remain. It is worth noting that despite all the guidelines, research,
recommendations and available tools for adaptation, the ultimate call as to whether or not to adapt often
remains the prerogative of an individual or manager who will act based on his perceptions, beliefs, culture
and willingness. Environmental psychological perspective is currently receiving very little attention, yet it
fully dictates the outcome of any climate initiatives.

5.5.2.4 Socio-institutional Adaptation Responses

This section focusses on community and institutional adaptation responses to climate change, providing an
update on AR5. The section begins by giving an overview of types of socio-institutional responses, and
assessing their implementation within the tourism, fisheries and aquaculture sectors, before assessing policy,
agreement and cooperative adaptation responses.

5.5.2.4.1 Community-based adaptation

Community-based adaptation (CBA) refers here to coastal communities’ and non-government organizations’
climate change adaptation responses. It includes participatory decision-making and bottom-up approaches to
implementing climate change adaptation, community organization for action, communities’ engagements
with local management authorities, individual level responses to climate change at a homestead scale,
livelihood challenges and responses. This assessment is based on reports of climate change impacts,
community vulnerability and responses from around the globe, in both developed and developing nations.
The literature from the latter focuses more on serious threats to livelihoods (Islam et al., 2013; Ahmed and
Diana, 2015b; Kuruppu and Willie, 2015; Lohmann, 2016; Smith et al., 2016a; Esteban et al., 2017; Koya et
al., 2017; Shaffril et al., 2017a), whereas the focus from developed nations has a greater focus on threats to
infrastructure and the disturbance of community processes. (Serrao-Neumann et al., 2014; Dutra et al., 2015;
Colburn et al., 2016; Elrick-Barr et al., 2016; Richards et al., 2016; Jurjonaas and Seekamp, 2018). When
comparing CBA approaches in communities from developed and developing economy nations, access to
resources is the clear defining difference (robust evidence, medium agreement).

Climate change awareness, community mobilization through collective action and the ability to engage with
governance structures provided successful examples of community-based adaptation (Dutra et al., 2015; Ray
et al., 2017). In Sri Lanka, shrimp farmers demonstrated adaptive and transformative capacity through the
understanding of their stressors and their willingness to engage in collective action (Galappaththi et al.,
2017). This example demonstrates the value of coherence in communities, awareness and collective
initiative (high confidence). Other characteristics associated with effective CBA include strong leadership,
policy integration (Serrao-Neumann et al., 2014), integrating CBA with other approaches e.g., integrated
coastal zone management (Ahmed and Diana, 2015b; Mitchell et al., 2017), acknowledging and addressing
tradeoffs (Finkbeiner et al., 2018), enhancing access to credit and switching skills sets (Shaffril et al.,
2017a).

Community-based adaptation requires the support of stakeholders with a sufficient level of awareness about
climate change, supported by policy and government institutions, to improve decision-making processes to
avoid climate maladaptation (Tapsuwan and Rongrongmuang, 2015). For example, the climate change
adaptation response of participants in a Southeast Asia dive industry was reported to be based on
misconceptions about climate change and personal observations (Tapsuwan and Rongrongmuang, 2015). In
another case, a community’s engagement with local authorities in a community mangrove forest
management project resulted in the deterioration of the mangrove forest due to poor management practices,
with a corresponding increase in community vulnerability as coastal protection services and economic
benefits declined (Nguyen et al., 2017). Despite good intentions with this project, a lack of enforcement,
weak governance processes and insufficient technical guidance drove the project towards a maladaptive outcome. These examples support the importance for communities to be served by clear policy and government institutions with accountable and firm governance processes and enforcement (high confidence).

Local knowledge and Indigenous knowledge can provide a source of experience from which climate change adaptation actions can draw, for example see the inuit of the Canadian Arctic (Panikkar et al., 2018) and Cross-Chapter Box 3. It will be important to commit resources (Alam et al., 2016) for the inclusion of cultural practices (Audefroy and Sánchez, 2017; Fatorić and Seekamp, 2017) and Indigenous knowledge systems (Kuruppu and Willie, 2015; von Storch et al., 2015) as part of nuanced adaptation planning at the local level (Novak Colwell et al., 2017).

Besides inclusion in adaptation programs, non-climate stressors will need to be addressed, including rural poverty levels (Jur Jonas and Seekamp, 2015), job opportunities and aging communities. In Canada, this includes the promotion of ‘age friendliness’ programs (Krawchenko et al., 2016). In East Africa, self-organization through negotiated social practices were deemed to be suitable, context-specific local responses to climate change (Kithiia, 2015). There is substantial evidence underlining the importance of social capital for increasing resilience in communities (Petzold and Ratter, 2015; Salik et al., 2015; Chen and Ganapin; Sakakibara, 2017; Triyanti et al., 2017). The substantial body of literature assessed suggests that community-based adaptation will continue to be a very important component of coastal adaptation planning and implementation going forward (high confidence).

5.5.2.4.2 Integrated coastal zone management and marine protected areas
The Fifth Assessment Report concluded that Integrated Coastal Zone Management (ICM), the coordination and integration of management efforts for coastal and marine regions, was an effective framework for addressing coastal climate change adaptation needs (Wong et al., 2014).

The incorporation of Marine Protected Areas (MPAs) into coastal and oceanic zone management is considered an important strategy in adapting to climate change (Hopkins et al., 2016) (medium agreement, medium evidence), but management of such zones is, itself likely to be impacted by climate change (Johnson et al., 2018). Management approaches need to accommodate a ‘shifting baseline’ as climate change becomes the new normal, with novel dispersal movements of highly-mobile species (Elliott et al., 2015). Where such shifts occur across national boundaries, management success will require transnational agreements (Gormley et al., 2015). Transboundary ICM is likely to become more important with ecosystem shifts due to climate change (medium evidence, high agreement).

An advantage of ICM is that it helps manage the interactions between multiple climate and non-climatic drivers of coastal ecosystems and sectors. For example, sea level rise is interacting with non-climate change impacts including illegal sand mining, inappropriate development and habitat destruction (e.g., mangroves), impacting upon tourism (Rangel-Buitrago et al., 2015) (Section 5.4.2.2.3). Adaptation responses to coastal erosion is thus marine spatial planning and coastal zone management informed by scientific, local knowledge and indigenous knowledge, and with sustainable funding mechanisms and support networks for decision-making. Overall, participatory spatial planning processes and ecosystem-based adaptation responses could improve tourism adaptation efforts (medium confidence).

Trade-offs associated with ICM and MPAs that may result in maladaptive outcomes; such that consideration of such trade-offs is important in effective climate-adaptive ICM and MPAs. For example, in Brazil, poorly managed MPAs did not provide increased benefits for small-scale fishers adjacent to the MPAs, with a high reliance upon access to mangroves whereby the restriction in access without increased benefit was considered maladaptive (Luiz Francisco Ditzel et al., 2016). Incorporating stakeholder participation processes as a form of bottom-up co-management could bring local knowledge and indigenous knowledge (Serrao-Neumann et al., 2013) to reduce the risk of mal-adaptation, and increase buy-in for implementation (Section 5.5.2.4.1) (medium evidence, high agreement). Some institutions with a strong top-down institutional framework may, however be resistant to changing to this approach (Abelshauser et al., 2015).

5.5.2.4.3 Adaptation in fisheries and aquaculture
In AR5 (WG2 Chapter 6), improved fisheries and aquaculture management is highlighted as a key opportunity to reduce climate risks and impacts on these sectors (Section 5.4.1). This assessment evaluates
further how different aspects of fisheries and aquaculture management could play a role in climate
adaptation, as well as an assessment of some other non-management adaptation interventions for these
sectors.

Management to reduce existing non-climatic stressors was suggested to be an effective climate risk reduction
adaptation measures for fisheries and aquaculture with large co-benefits (Gattuso et al. 2018). Specifically,
as overfishing is one of the most important non-climatic drivers affecting the sustainability of fisheries,
eliminating overfishing could help rebuild fish stocks, reduce ecosystem impacts and increase the adaptive
capacity of fisheries. In an analysis, 60% of assessed species are projected to be at high risk from both
overfishing and climate change by 2050 (RCP8.5), particularly tropical and sub-tropical species (Cheung et
al., 2018b). Pursuing sustainable fisheries practices under a low emissions scenario would decrease risk by
63%, highlighting the importance of effective fisheries management (Gaines et al., 2018). However, as
eliminating overfishing would require reducing current levels of fishing capacity, there are uncertainties
associated with whether the short-term reduction in catches may impact livelihood and food security of local
communities (Hobday et al., 2015b; Dey et al., 2016; Rosegrant et al., 2016; Campbell, 2017; Finkbeiner et
al., 2018). Despite the high agreement on the significant effectiveness of eliminating overfishing in climate
change adaptation for fisheries, available evidence remains aspirational rather than presenting evidence of
successful adaptation outcomes.

Range shift of fish stocks under ocean warming (Section 5.2.3) will alter distribution of fish stocks across
political boundaries, thus demand for consideration of transboundary fisheries management in adaptation for
fisheries will increase. Re-distribution of transboundary fish stocks between countries will destabilize
existing international fisheries agreements and increase the risk of international conflicts (Section 5.4.2).
Adaptation of international fisheries management to reduce such risk would involve improving planning for
cooperative management between countries informed by reliable projections of species shifts and associated
uncertainties (Pinsky et al., 2018). Regular monitoring and performance reviews of fisheries could also
facilitate the management adjustment towards considering species shifts. Other international fisheries
arrangements, such as flexible fishing effort allocation with opportunities for countries to increase
exploitation of resources by providing payment to other countries, may also improve the robustness of
fisheries management (Miller et al., 2013). Success from existing arrangements to adapt to climate
variability suggests that these measures can be effective. However, the ability to fully adapt to climate
change beyond historical variability is uncertain. Thus, although range shift poses a significant challenge to
transboundary fisheries management, proactive planning and adjustment of fisheries management
arrangements, informed by scientific projections, could help improve their adaptive capacity (high
confidence).

Fisheries management strategies depend heavily upon the accuracy of data collected in respect of predicting
environmental conditions, over time scales from months to decades (Dunstan et al., 2017), controlling for
aspects of fish population dynamics like recruitment success and fish movement. Seasonal to decadal climate
prediction systems allow for skillful predictions of climate variables relevant to fisheries management
strategies (Hobday et al., 2016b; Payne et al., 2017; Tommasi et al., 2017). A planning response in the U.S.
National Marine Fisheries Service has been to conduct vulnerability assessments, improve monitoring of
ecosystem indicators and evaluate management strategies (Busch et al., 2016). Himes-Cornell and Kasperski
(2015) provided a framework of indicators for Alaskan fishing communities’ vulnerability to climate change
that includes exposure to climate change bio-physical effects, resource dependency and adaptive capacity.
Community response as a part of climate change adaptation for local fisheries is an important element in
assessing adaptive capacity (medium evidence, good agreement), and this response is affected by socio-
economic factors like alternative income, gender and religion, which collectively shape a community’s
adaptation response (Arroyo Mina et al., 2016). In West Africa, the industrial fishery response to climate
change-induced reduction in landings was the expansion of fishing grounds, which increased operational
costs (Belhabib et al., 2016), although this response was not available to artisanal and local fishing
communities, who are considered highly vulnerable (Kais and Islam, 2017). Other adaptation responses
included improved fishing gear and technology, use of Fish Aggregating Devices and uptake of insurance
products (Zougmoré et al., 2016).

In considering both an ecosystem and participatory decision-making approach for climate change-
appropriate fisheries management, Heenan et al. (2015b) provided a number of key essential elements like
expert knowledge of climate change threats to fish habitats, stocks and landings, the necessity of
transdisciplinary collaboration and stakeholder participation, broadening the range of scope of fisheries
systems and increased commitment of resources and capacity. This was considered in the context of the
ability of developing countries to manage ecosystems and existing over-exploitation of resources. More
research is required on socio-ecological responses to climate change-impacts on fishery communities. This
includes aspects like risk reduction, adaptive capacity through knowledge attainment and social networks,
developing alternative skills and participatory approaches to decision-making (Dubey et al., 2017; Shaffril et
al., 2017c; Shaffril et al., 2017b; Finkbeiner et al., 2018).

Less still is known about how climate change will impact upon the deep oceans and fisheries therein,
especially those resources that are not yet being harvested. Johnson et al. (2018) concluded that in a 20 year
to 50 year timeframe, virtually all North Atlantic deep-water and open ocean area-based management tools
will likely be affected. They concluded that more precise and detailed oceanographic data are needed to
determine possible refugia, and more research on adaptation and resilience in the deep sea is needed to
predict ecosystem response times. Until such analyses can be made, additional human impacts could be
avoided using a more precautionary approach, potentially setting aside more extensive conservation areas
and strictly limiting human uses and/or adopting high protection thresholds (Johnson et al., 2018).

As with fisheries, community- and ecosystem-based adaptation responses, within an integrated coastal zone
management framework, is considered necessary to cope with anticipated challenges for aquaculture
(Ahmed and Diana, 2015a). Where in-situ adaptation is not possible, translocation and polyculture (Ahmed
and Diana, 2015b; Bunting et al., 2017) have been suggested as appropriate responses (robust evidence,
good agreement), but this would suit commercial rather than subsistence interests. Policy, economic,
knowledge and other types of support are required to build socio-ecological resilience of vulnerable coastal
communities (Harkes et al., 2015; Bunting et al., 2017; Rodriguez-Rodriguez and Bande Ramudo, 2017),
which requires a deep understanding of the nature of stressors and a commitment for collective action
(Galappaththi et al., 2017). Climate resilient pathway development (CRPD, see Cross-Chapter Box 1) is
considered a useful framework for Sri Lankan shrimp aquaculture (Harkes et al., 2015). Another successful
example of aquaculture adaptation is the employment of near-real time monitoring technology to track the
carbonate chemistry in water to reduce bioerosion in shellfish from acidification (Barton et al., 2015b;
Cooley et al., 2016).

An example of an engineering-based adaptation option that has the potential to increase productivity of
seaweed aquaculture under climate change is artificial upwelling (Jiao et al., 2014a). Nutrient availability is
the major limiting factor for open ocean primary productivity (see section 5.2.2.2). Upwelling can bring up
high-nutrient deep water to the euphotic zone and enhance primary production (Mackenzie et al., 2002;
Farias et al., 2015). However, upwelling can also bring high dissolved inorganic carbon (DIC) and low
oxygen water to the surface, which can sometimes cause acidification and hypoxia (Demarcq, 2009; Narayan
et al., 2010; Bauer et al., 2013; Miranda et al., 2013). Artificial upwelling powered by green energy (such as
solar energy, wind/wave/tidal energy) (Zhang et al., 2016) to seaweeds culture (Jiao et al., 2014b; Zhang et
al., 2015; Pan and Schimel, 2016) may help moderate the amount of deep water upwelled to the euphotic
zone. The aim of the adaptation intervention is then to control the degree of such upwelling to meet the
demands of nutrients and DIC by the cultured seaweed. This could theoretically reduce acidification and
hypoxia in the upwelling area as DIC is mostly used by the seaweed while oxygen is produced by
photosynthesis. Such artificial upwelling operations may also gradually release the ‘bomb’ of rich nutrients
and hypoxia in the bottom water, which could breakthrough when storms take place otherwise (Daneri et al.,
2012). In situ monitoring of physical and chemical variables such as temperature, nutrients, oxygen, DIC
and pH etc. are used for adjustment of the upwelling extent for adaptive management. This type of
interventions would require high level of scientific knowledge, capital investment and potential unexpected
ecological consequences. Thus, even if such intervention is proven to be practical at large-scale, there may
be substantial limits to its suitability for different upwelling areas in the world.

5.5.2.4.4 Institutional responses
Coastal and oceanic adaptation responses are greatly complicated by the presence of competing interests
(either between user-groups, communities or nations), where considerations other than climate change need
to be incorporated into cooperation agreements and policy (Wong et al., 2014). Climate change adaptation
planning is required in local land-use planning and other processes (Sano et al., 2015; Elsharouny, 2016).
The deployment of either built or natural protection systems, or adopting a ‘wait and see’ approach, is subject to the social acceptance of these approaches in communities, amongst others (Poumadère et al., 2015; Sherren et al., 2016; Torabi et al., 2017). Similarly, the willingness to move away from climate change-impacted zones is dependent upon a range of other socio-economic factors like age, access to resources and crime (Bukvic et al.; Rulleau and Rey-Valette, 2017). Adaptation to climate change will require consideration of a range of non-climatic, social variables that will likely complicate implementation of adaptation plans (robust evidence, high agreement).

In coastal communities, there is often consensus on the importance of cooperation in tackling climate change (Elrick-Barr et al., 2016), but adaptation progress is hampered by competing economic interests and worldviews (Hamilton and Safford, 2015) as well as lack of knowledge (Nanlohy et al., 2015), whilst factors like home ownership and a general future planning ability are important (Elrick-Barr et al., 2016). Another important factor is the level of trust communities have in their local governance processes, causing (Massuanganhe et al., 2015; Wynveen and Sutton, 2015) to conclude that relationship-building actions are an important part of an adaptation response. In a survey to understand community perception of financing adaptation actions, British coastal communities displayed a high level of resistance to the implementation of a coastal defense tax (Alderson-Day et al., 2015). Local collective action, using legal arguments around the right to protection from environmental change, may be a useful mechanism of securing state support for adaptation (Karlsson and Hovelsrud, 2015).

Climate change adaptation capacity is shaped by historical path dependencies, local context and international linkages, while action should be shaped by science, research partnerships and citizen participation (Hernández-Delgado, 2015; Sheller and León, 2016). Locally-context-specific data to guide appropriate adaptation response remains a knowledge gap (Abedin and Shaw, 2015; Hobday et al., 2015b; Lirman and Schompeyer, 2016; Williams et al., 2016), whilst transnational cooperation in marine regions is essential for robust adaptive management (Gormley et al., 2015).

Technology for environmental monitoring, for example using drones (Clark, 2017) and web-based coastal information systems (Mayerle et al., 2016; Newell and Canessa, 2017) promise to improve the local scale knowledge base, which should improve climate adaptation planning effort and environmental management decisions (Conde et al., 2015). Where such knowledge gaps persist, the implementation of climate change adaptation measures could proceed on the basis of a set of general principals of best practice (Sheaves et al., 2016b; Thorne et al., 2017).

There remains a paucity of evaluation literature pertaining to the effectiveness of adaptation efforts. In evaluations of socio-institutional approaches, the importance of partnerships have been emphasized; for example, Rutherford et al. (2016) reported that the length of history of collaboration in Europe was positively associated with adaptation planning progress. In the UK, better access to information from government investment has resulted in better adaptation planning and local government capacity, but not in the implementation of adaptation plans (Porter et al., 2015). This was considered to be due to budget cuts while best adaptation progress has been made in local governments addressing resilience to extreme weather events, and this is consistent with other reports of political focus on more immediate pressures (Barbier, 2015; Gray et al., in press) (robust evidence, high agreement).

5.5.2.4.5 The role of education and local knowledge in adapting to climate change.

It is essential that people are informed that local and small scale behaviours accumulate very quickly and contribute to the global scale phenomenon of climate change. People need very clear messages and consistent messages that reiterate that new behaviours are very much needed at the individual, household and local business scale as well as when individuals travel and participate in tourism opportunities. People urgently need to be educated about cumulative impacts and feel empowered and inspired to take positive actions. The growing popularity of outreach initiatives and scientific communication to the general audience can be a critical instrument to increase awareness and concern (medium confidence).

Education can increase knowledge and awareness of climate change impacts and the efficacy of their mitigation (Meadows, 2011). It can influence the extent to which stewardship activities are adopted (von Heland et al., 2014; Wynveen et al., 2015). It can also help to develop new networks between coastal people and environmental managers for the purposes of developing and implementing new adaptation strategies
(Wynveen et al., 2015) Research suggests that a lack of education causes misconceptions about the scale of climate change (temporally and spatially), and the causes of climate change (Tapsuwand and Rongrongmuang, 2015). For example, ocean acidification may be an excellent educational tool to address climate change issues (Fauville et al., 2011) because the acidification process can be clearly explained. A critical element to reducing vulnerability to climate change is to educate people that they are core to the Earth system and have a huge influence on the balance of the system. An important service of marine biodiversity and ecosystems is to support such education (Section 5.4.1). Thus, education can play a pivotal role in how climate change is perceived and experienced and marine biodiversity and ecosystems plays an important role in this.

A lack of education clearly influences the resilience and/or vulnerability of communities to climate change (Qin et al., 2017). This occurs in two ways; through increasing sensitivity to climate change, and through decreasing adaptive capacity. Sensitivity to climate change can be increased through increasing the dependency that people have on a single natural resource. People in rural coastal areas have less education and thus less options for the future, increasing their dependency on the natural resource (low confidence). The greater the dependency, the greater the sensitivity to climate change. Unless resource-dependent people have a correspondingly high level of adaptive capacity to diminish any social and economic impacts associated with their dependency, they remain highly vulnerable (Marshall et al., 2017; Marshall et al., 2018). Adaptive capacity can be increased through education, where people are able to convert their newly founded knowledge and information into successful adaptive strategies (Gladstone, 2009). Marshall et al. (2013) show that climate change awareness is associated with enhanced adaptive capacity. Additionally, education can increase the likelihood that existing technologies which aid the community to better cope and adapt to climate change. For example, without education, social and behavioral patterns are more likely to influence the effectiveness and use of warning systems, regardless of the technology involved (Pescaroli and Magni, 2015). Education needs to be combined with other institutional support for the full benefits to emerge and for education to be meaningful. For example, indigenous communities can be very aware and educated about climate change, but can be unempowered to make decisions that would protect their communities without the appropriate institutional support.

Local knowledge of environments that are under threat from climate change can both act to promote adaptation and act as a barrier to adaptation. Local knowledge is a measure of the human capital that has been developed by individuals to better understand their local environment (Andrachuk and Armitage, 2015). For example, some commercial fishers or marine-based tourism operators may have spent a significant proportion of their career towards developing a deep familiarity with their local fishing grounds or marine environment for the purposes of economic gain. Their local knowledge makes them particularly dependent upon that area, and they are more likely to be successful within that local area when conditions are stable (Marshall et al., 2007). However, this dependency makes them particularly sensitive to climate events such as bleaching or a cyclone (Marshall et al., 2013). Fishers and tourism operators with higher local knowledge will be more sensitive to change than those that have invested less in developing their local knowledge. The local knowledge that has been developed may be so important to fishers or tourism operators that they are likely to be particularly reluctant to move away from their familiar environment and develop new local knowledge elsewhere. In these instances, local knowledge can act as a barrier to adaptation (Metcalf et al., 2015).

There is evidence, however, to suggest that fishers and tourism operators with high levels of local knowledge (whilst sensitive) are not necessarily vulnerable to climate change if they have correspondingly high level of adaptive capacity (Marshall et al., 2013). Resource users such as commercial fishers with high levels of local knowledge are also likely to have already noticed subtle changes within the environment, and recognise the need to change and adapt. This is because they recognise ‘feedbacks’ within the system (Adger et al., 2008). In these instances, fishers with higher local knowledge are more likely to demonstrate a higher adaptive capacity than fishers with lower local knowledge, and will more likely progress towards developing new strategies to combat the impacts of climate change (Kittinger et al., 2012). In these instances, local knowledge acts to promote adaptation (medium confidence).

5.5.2.5 Costs and Limits for Coastal Climate Change Adaptation
Chapter 5 of AR5 assessed costs of adaptation to sea level rise with estimates varying between tens to hundreds of billions of US dollars, dependent upon the sea level rise scenario. Recent estimates suggest that annual costs of coastal adaptation and adaptation of infrastructure to changes in rainfall and temperature range from US$400 million to US$1.2 billion by 2040 (World Bank, 2017). Average annual direct loss due to cyclone-associated flooding in 15 South Pacific countries was estimated at up to US$ 80 million (2009 prices), with 60% of the damage resulting from loss of residential buildings, 30% from loss of cash crops and 10% from damage to infrastructure (PCRAFI, 2011). A key research gap identified was comparison with natural based solutions, with consideration of the co-benefits the latter option provides. In the recent literature, estimating adaptation costs is still challenging because of wide ranging regional responses and uncertainty (Dittrich et al., 2016). Despite these challenges, the protection from flooding and frequent storms that coral reefs provide has been quantified (Beck et al., 2018). They estimated that without reefs, damage from flooding and costs from frequent storms would double and triple respectively, while countries from Southeast/ East Asia and Central America could each save in excess of US$ 400 million through good reef management.

In West African fisheries, loss of coastal ecosystems and productivity are estimated to require 5–10% of countries’ Gross Domestic Product in adaptation costs (Zougmoré et al., 2016). Similarly, for Pacific Islands and Coastal Territories, fisheries adaptation will require significant investment from local governments and the private sector (Rosegrant et al., 2016), with adaptation costs considered beyond the means of most of these countries (Campbell, 2017). In Small Island Developing States, tourism could provide the funding for climate change adaptation, but concerns with creating investment barriers, assumptions around cost-effectiveness and consumer driven demand remain barriers (Hess and Kelman, 2017). Marine Protected Areas, with multiple co-benefits, are considered a cost-effective strategy (Byrne et al., 2015), but, as was reported in AR5, further research evaluating natural infrastructure is required (Roberts et al., 2017).

There are still a lot of challenges in conducting economic assessments for built infrastructure adaptation due to the associated, complicated uncertainties such as accuracy of climate projections, limited information regarding paths for future economic growth and adaptation technologies. A global estimation of coastal flood damage and adaptation costs due to sea level rise by the end of this century projected that 0.2–4.6% of the global population would be flooded annually with expected annual losses of 0.3–9.3% of global Gross Domestic Product (Hinkel et al., 2014). Annual investment and maintenance costs of protecting coasts were projected to be US$ 12 billion to US$ 71 billion, which was considered significantly less than damage costs in the absence of such action. In an analysis of twelve Pacific island countries, 57% of assessed built infrastructure was located within 500m of coastlines, requiring a replacement value of US$ 21.9 billion. Substantial coastal adaptation costs (and international financing) are required in these countries.

An extreme event that exceeds an infrastructure’s maximum tolerance level can result in failure of the infrastructure. Changes in climatic patterns can slowly affect the service life and performance of infrastructure. Unfortunately, there is little literature bridging climate change and infrastructure deterioration (Ha et al., 2017). The need to understand the economy-wide aspects of climate change and its adaptation has been widely recognised in literature (Fisher-Vanden et al., 2013; Fankhauser, 2017) and there is high agreement on the need to shift towards climate-resilient economies (Robinson et al., 2012; Fisher-Vanden et al., 2013). In addition, given the economy-wide effects captured implicitly in econometric models, various authors have highlighted the relevance of systemic adaptation models (Conway et al., 2015; Mullner and Dietz, 2015; Fankhauser, 2017) despite difficulties faced in structuring all variables onto input-output tables or in factoring social accounting matrices.

Due to interdependency of economic systems, it is very likely that indirect risks, and therefore adaptation needs, are dominant contributing factors to direct effects of climate change for some sectors. Meanwhile, understanding adaptation deficit is increasingly receiving attention. Factors such as literacy, income, income distribution, institutional quality, health spending, and access to finance are linked to adaptive capacity, or the ability to respond to climate risk (Fankhauser and McDermott, 2014; Carleton and Hsiang, 2016; Fankhauser, 2017). There are strong interlinkages between adaptation and economic development, such that future vulnerability to climate change will drive decisions on infrastructure adaptation investments, industrial strategy and urban planning (Fankhauser, 2017). For effective adaptation, knowledge, planning, coordination, and foresight are required. This necessitates multifaceted decision-making which makes adaptation an interesting economic problem.
Within the literature, there are a broad range of reported barriers and limits to climate change adaptation for each of the three themes discussed in Section 5.5.2. For shoreline stabilisation, there is a key research need to improve understanding of limitations in built-infrastructure, beach nourishment and nature-based adaptation responses, especially in respect of cost effectiveness and resilience. (Mackey and Ware, 2018).

Whilst using ecosystems is considered to be a cost-effective and sustainable approach for coastal erosion management (Adriana Gracia et al., 2018), limitations have been acknowledged with coastal ecosystem based adaptation (Sussams et al., 2015). Limitations include the space that ecosystems require (which may not be available in an urban environment), non-climate change co-drivers like human impacts that need to be addressed as part of the adaptation response (Ahmed et al., 2017; Peña-Alonso et al., 2017; Triyanti et al., 2017), the severity of climate impact may exceed the adaptive capacity of the ecosystem, for example, the type of rainfall being experienced affects the efficacy of storm water associated green infrastructure (Joyce et al., 2017) or acidification affecting coral reef viability, and, potentially slower recovery rates within ecosystems than the recurrence of climate impacts, including the forcing of ecosystems into novel dynamic equilibrium states that may not deliver the level of service required. Given these limitations, it is important that EBA decision-makers are better informed about its context and site specific application (Sutton-Grier et al., 2015) to avoid maladaptation (high confidence). A barrier to achieving this is the fragmented nature of EBA implementation across disciplinary approaches (Brink et al., 2016). Other barriers include governance structures, including incorporating longer-term scales, how to measure effectiveness, developing appropriate financial mechanisms and dealing with uncertainty (Ojea, 2015).

Built infrastructure may negatively impact upon adjacent ecosystems as coastal processes can be disrupted, and result in unintended consequences. Examples include New Orleans during Hurricane Katrina and Sendai during the 2010 Tohoku earthquake, where the need for a ‘design storm’ in the engineering design process was considered a critical adaptation limit (Mackey and Ware, 2018). This is particularly the case when design practice does not consider climate change impacts in all design storm event characteristics (Watt and Marsalek, 2013). Understanding when and how to implement a suite of built and natural infrastructure adaptation options is considered essential to achieving an optimal adaptation response (Depietri and McPhearson, 2017).

For artisanal fisheries, a range of physical and socio-institutional limits and barriers to adaptation have been reported, including increasing occurrence and severity of storms, technologically poor boats and fishing equipment and lack of access to credit and markets, among others (Islam et al., 2013). Conflicting interests and values of stakeholders in great barrier reef-associated industries were deemed to limit adaptation response (Evans et al., 2016), whereas the path-dependent nature of organisations and resistance to change, reported in Australia, were considered socio-institutional barriers, (Evans et al., 2016). Governance barriers to adaptation in Vancouver include inadequate collaboration, political leadership, public awareness, finance and capacity (Oulahen et al., 2018). There is a key research need to understand how such limits and barriers interact to suppress adaptation response.

In Tanzania, barriers to adaptation were place-specific, highlighting inequality in response, but also the importance of including local residents within site-specific adaptation planning efforts (Armah et al., 2015). Similarly, in the Maldives, conflict is arising out of the government’s centrist approach to fortifying just a few central islands, owing to economic limitations (Hinkel et al., 2018). Barriers to coastal adaptation in South Africa and Mozambique were chiefly finance, expertise and capacity and knowledge (Rosendo et al., 2018).

Defining limits to social adaptation through a stakeholder-focussed, risk-based approach can help inform debate about societal response to climate change (Dow et al., 2013). In some communities, climate change may not be prioritised. For example, Fischer (2018) presented evidence in six Oregon, USA communities where response to potential climate change impacts like disasters are not prioritised in the face of chronic, daily challenges to secure livelihoods. In coastal Jakarta, Indonesia, a survey was conducted in poor communities about subsidence and sea level rise risks. While local inhabitants appear to be aware of the hazards they face, many seem to underestimate their severity, possibly due to a high frequency of exposure in the recent past (Esteban et al., 2017). In a world with competing risks and urgent priorities, some local inhabitants appear to be unable to avoid, or are willing to carry, the risk associated with a climate impact in order to meet other, more pressing needs. This example reflects the reality of many poor, informal settlement
Adaptation barriers, which are both internal and external, are progressively being discovered and are mainly socio-economic, cultural and psychological. In most cases these factors are fuelled by climate uncertainties and scepticism. Inter-organisational networks enable decision makers and actors to acquire additional adaptive capacity from beyond the organisation to overcome barriers, and thus enable the translation of adaptive capacity into adaptation manifestation. The need for adaptation at all scales is widely emphasised.

5.5.2.6 Summary

There has been a substantial amount of literature focused on coastal and oceanic adaptation since AR5. Socio-institutional adaptation responses are the more numerous of the three types of adaptation responses assessed in this chapter. There is broad agreement that hard engineering responses are optimally supported by ecosystem-based adaptation approaches, and both approaches should be augmented by socio-institutional approaches for adaptation (high confidence) (Nicholls et al., 2015; Peirson et al., 2015; Sánchez-Arcilla et al., 2016; van der Nat et al., 2016; Francesch-Huidobro et al., 2017; Khamis et al., 2017). In planning adaptation responses, awareness-raising and stakeholder engagement processes are important for buy-in and ownership of responses (robust evidence, high agreement) as is institutional capacity within local government organizations, whose importance in coastal adaptation initiatives has been emphasized in the recent literature (robust evidence, high agreement). With all three types of adaptation, it is clear that getting basic good governance and effective implementation of service delivery processes correct is a prerequisite for successful adaptation planning and response.

5.5.3 Governance Across All Scales

5.5.3.1 Ocean and Coastal Governance

Global, regional, national and subnational governance structures in response to climate driven ocean acidification and ocean warming and their impact on marine ecosystems and dependent communities is presumably highly fragmented (Galland et al., 2012; Stephens, 2015; Fennel and VanderZwaag, 2016; Diamond, 2018). Legal, policy and institutional response is shared by the institutions developed for a number distinct but inter-related fields including inter alia governance regimes for climate change, ocean and marine environment, fisheries and the environment generally.

Existing ocean governance structure for the ocean is facing or will face multi-dimensional challenges because of the climate change (Galaz et al., 2012). Considering climate change issues in the global and regional ocean, environmental and fisheries governance structure is needed. Current international governance regime yet to adequately address the issues of ocean warming, acidification and deoxygenation. As mentioned by Oral, legal framework for ocean acidification ‘appears to fall between the cracks of the two principal regimes for climate change, the UNFCCC and ocean governance under UNCLOS’ (Oral, 2018). While adoption of these legal and governance regimes, climate change or carbon dioxide emissions related changes of the ocean, particularly, the interaction of physical, chemical, biological properties of the ecosystems with risks and vulnerabilities of the dependent communities was either not considered or did not get proper attention. There is a need for a comprehensive assessment of exiting international legal regimes for above-mentioned areas considering the changing ocean and its ecosystems and their impact on dependent communities.

5.5.3.2 Climate Change Governance Regime

International climate change governance regime is naturally the most relevant global governance apparatus for combating ocean acidification and ocean warming. However, how far current climate change legal and policy framework is adequate for dealing with the issue of the ocean acidification is questionable (Stephens, 2015). Existing climate change related legal instruments do not adequately address the challenges the ocean
and coastal areas will face. Nevertheless, their overall goal for reduction of emissions will be positive for
mitigation of climate change impact on the ocean and coastal areas (Galland et al., 2012; Gallo et al., 2017).
Ocean acidification and ocean warming did not get full attention during the negotiation of the United
Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol in the 1990s. These
legal instruments are mainly for combating global warming with a clear atmospheric focus and ocean did not
get proper attention (Galland et al., 2012). In 2015, parties to UNFCCC adopted the Paris Agreement,
according to which state parties will address climate change mitigation and adaptation as per their Nationally
Determined Contributions (NDCs). Unlike Kyoto Protocol, the ocean is explicitly mentioned in the Paris
Agreement’s preamble and there may be implications from the Paris Agreement for ocean (Magnan et al.,
2016). However, a study on 161 NDCs submitted where national pledges for climate change mitigation and
adaptation show that there are ‘gaps between scientific [understanding] and government attention, including
on ocean deoxygenation, which is barely mentioned’ (Gallo et al., 2017). Therefore, greater emphasis to the
ocean related issues in the climate change governance regime is needed.

5.5.3.2.1 Environmental governance of ocean
The United Nations Convention on the Law of the Sea (UNCLOS) elaborates the jurisdiction, duties and
obligation of coastal, flag and port states in different maritime zones including internal waters, territorial sea,
exclusive economic zone, continental shelf, and high seas. Two aspects of UNCLOS are specifically
important in the context of this assessment: the provisions relating to prevention of marine pollution and
provisions relating to the conservation and sustainable utilization of marine living resources. UNCLOS
imposes obligation for the conservation and sustainable utilization of marine living resources within national
jurisdiction (UNCLOS 1982, Articles 56, 6, 62, 63, 64, 65 and 67). UNCLOS imposes obligations on state
parties to take action for combating six main sources of ocean pollution including land-based and coastal
activities, continental shelf drilling, seabed mining, ocean dumping, vessel-source pollution, and pollution
from or through the atmosphere. It has been suggested that tools and techniques in UNCLOS may need
adjustment in response to the emerging challenges created by climate change for ocean (Redgwell, 2012).
However, success of the umbrella regulatory framework of UNCLOS is heavily depending on further
development, modification and implementation of detail regulations by relevant international, regional and
national institutions (Karim, 2015).

One of most relevant legal instruments in this regard is the Convention on the Prevention of Marine
Pollution by Dumping of Wastes and Other Matter, 1972 (London Convention). In 1996, the London
Protocol was adopted to modernize and eventually replace this convention. 1996 London Protocol prohibits
dumping of any wastes or other matter to the sea unless it is listed in the Annex 1 of protocol. Through an
amendment in 2006, ‘carbon dioxide streams from carbon dioxide capture processes for sequestration’ has
been included in the Annex 1 of the Protocol. These amendments then followed by further two amendments
on sharing transboundary sub-seabed geological formations for sequestration projects and ocean fertilization
and other marine geoengineering. One of these new amendment prohibits ocean fertilization except for
research purposes (Dixon et al., 2014). Nevertheless, there are concern for these activities. International
Convention for the Prevention of Pollution from Ships, 1973 (MARPOL) is also relevant for climate change
and GHG emissions. In 2011, this convention was amended to include some technical and operational
measures for reduction of GHG emissions from ships. However, effectiveness of these provisions is
questionable (Karim, 2015).

A set of international regulations that will be crucial for combating the challenges pertaining to climate
change and changing ocean are the regional seas conventions adopted under the auspices of different
regional seas programs. For example, the issue of ocean acidification has been considered within the
framework of the Convention for the Protection of the Marine Environment of the North-East Atlantic
(OSPAR Convention) and the Convention on the Conservation of Antarctic Marine Living Resources
(CCAMLR) (Herr et al., 2014).

Major international environmental convention systems are considering the issue of climate change (Morgera,
2011). For example, CBD COP adopted specific decision regarding technical and regulatory issues on
geoengineering. Other major conventions also increasing considering the issue of climate change. However,
there are huge rooms for further synchronization (Proelss and Krivickaitė, 2009). Several sectoral
international environmental conventions are relevant directly or indirectly in respect of changing ocean. For
example, Convention Concerning the Protection of the World Cultural and Natural Heritage may play a role
in respect of world heritage coral reefs. Most of the 29 world heritage listed coral reefs are facing severe heat stress (Heron, 2017). The global marine environmental governance system needs to give more attention to the climate change related issues (Redgwell, 2012; Herr et al., 2014; Heron, 2017).

5.5.3.2.2 Fisheries governance

The impact of climate change on marine fisheries will be very significant and it may have an adverse impact on food security, livelihoods and national development in many coastal countries, and the least developed countries will me particularly vulnerable (Blasiak et al., 2017). As outlined in this report, evidence shows that warming of the oceans has emerged and can be attributed to human influence (Section 5.2.2), that the oceans are acidifying, the oceans are losing oxygen particular in the equatorial and eastern boundaries (Section 5.2.2) and these effects are projected to affect fish distributions and biomass with consequent flow on effects to human communities (Section 5.3 and 5.4). Changing ocean may also create serious problem in the governance of regional fisheries agreements (Brandt and Kronbak, 2010). Regional fisheries management systems also need to address the emerging challenges (Brooks et al., 2013). A study on seven international case studies shows that fisheries governance system will face further uncertainties because of climate change necessitating adoption of progressive policy initiatives introducing flexible management structure, capacity development, sustainable exploitation system and altertaine arrangement for livelihood security of dependent communities (McIlgorm et al., 2010). The same report also indicates that the adaptation to climate change will be more difficult if there is an absence of good existing fisheries governance structure (McIlgorm et al., 2010). A review published in 2017, identifies nine major ecological and socio-ecological criteria for climate resilient regulatory approach including ‘sustainable & age-diverse target populations’, ‘conserving biodiversity & habitats’, ‘managing existing stressor’, ‘adaptive management’, ‘diversified livelihoods’, ‘promoting longterm stewardship’ ‘multi-level governance’, ‘Fisher mobility’ and ‘community based management’ (Ojeda et al., 2017). However, the effectiveness of these will diverge depending on the characteristics of regulatory instruments and other factors (Ojeda et al., 2017). A climate-informed ecosystem-based fisheries governance approach has been suggested for climate change resilience of marine fisheries in the developing world (Heenan et al., 2015a). In producing a strategy and roadmap for coastal fisheries, within a participatory workshop in the Pacific concluded that robust and effective management policy, legislation and planning will be required for coastal fisheries based on flexibility and scientific understanding to support management decisions (Gourlie et al., 2017). An integrative science-based approach has been suggested as ‘a vehicle (1) to examine policy options with respect to their robustness to uncertainty, particularly to climate-related regime shifts and (2) to allow better assessments of behavioral responses of fish, humans and institutions.’ (Miller et al., 2010). Badjeck et al. (2010) indemnified the following is needed in the context of challenges in global fisheries governance: (1) management approaches and policies that build the livelihood asset base, reducing vulnerability to multiple stressors, including climate change; (2) an understanding of current response mechanisms to climate variability and other shocks in order to inform planned adaptation; (3) a recognition of the opportunities that climate change could bring to the sector; (4) adaptive strategies designed with a multi-sector perspective; and (5) a recognition of fisheries potential contribution to mitigation efforts.’ (Badjeck et al., 2010).

The above discussion shows that greater emphasis and comprehensive reform within fisheries governance in all levels is needed for overcoming climate change related challenges for fisheries-based livelihood and conservation of global fisheries resources. Apart from the above sectoral areas of governance, governance of areas beyond national jurisdiction in the context of climate change is another major challenge (Levin and Le Bris, 2015). According to Warner ‘collaborative structures and mechanisms for environmental assessment in ABNJ are still fragmentary and underdeveloped, with limited sectoral involvement. Establishing these governance structures in ABNJ involves multiple stakeholders, including states, global and regional organizations, marine industries, and nongovernmental organizations focused on conservation of marine biodiversity.’ (Warner, 2018). Currently a negotiation is going on in the United Nations for a new agreement under UNCLOS for marine biodiversity of areas beyond national jurisdiction. Climate change related challenges need to be addressed in the future legal framework for areas beyond national jurisdiction.

5.5.3.3 Private Sector Stakeholders

The success of ocean-based climate initiatives, and marine conservation more generally, is constrained by disjoined and inadequate ocean governance, conflicts between stakeholders, limited enforcement capacity and inadequate finance (Gill et al., 2017). Many recent studies have confirmed that funding for protected
areas and biodiversity conservation must increase significantly to achieve targets set at national or international levels (e.g., Aichi targets). A recent global top-down assessment conducted by the Convention on Biological Diversity High Level Panel estimated the global investment required is estimated to be up to 5 times the current budgets (CBD, 2013).

It has been estimated that market-based mechanisms could generate up to 50% of conservation finance for coral reef in 2020, but long-term, reliable instruments need to be established and strengthened (The Katoomba Group, 2010). Five areas of financial innovations have been set out concerning the private sector: schemes for payment for ecosystem services; biodiversity offset mechanisms; markets for green products; public-private partnerships and new forms of charity; development of new and innovative sources of international development finance.

Public-private partnerships (PPPs) are presented as a type of arrangement that addresses the conservation financing gap. The main advantages of PPPs include: their flexibility to set fees and charges, establish funding mechanisms such as concessions, respond to customer needs, their ability to retain the money they earn (which gives a resulting incentive to generate funds through greater entrepreneurship), the implication of local communities in the activities and their freedom to implement staffing policies based on efficiency and market salaries (World Bank, 2014).

PPPs have taken a wide range of forms, which vary in the degree of involvement of the private entity in a traditionally public infrastructure (European Commission, 2003). Five main categories of agreements have been observed for nature conservation in the region: a parastatal agency, management contracts, leases, concessions and joint ventures. All these categories are described more precisely in (European Commission, 2003). Private investment in marine biodiversity and ecosystem services is in its early stage of development, and for the majority of economic instruments practical experiences are very limited (The Katoomba Group, 2010; Crédit Suisse, 2016). There are few investments that have been designed to produce benefits to the marine environment with financial returns, by the name of marine impact investments or otherwise. In the same way, very few private equity and venture capital firms are specialized in marine projects.

PPPs may further be used for marine conservation through active restoration approaches, where ecosystem services and biodiversity are to be improved through the capitalization of the marine ecosystem (Bull et al., 2013; Penca, 2013). This is based on the concept to use market-based incentive mechanisms to empower biodiversity (Rinkevich, 2015b). In the coral reef ecosystem, for example, the structural complexities of reef habitats are strongly associated with the biodiversity of scleractinian corals (De’ath et al., 2012) (high confidence), a property that may be enhanced through the employment of active reef restoration initiatives (Rinkevich, 2015b). This conclusion is further noticed in cases where traditionally used biodiversity offsets do not compensate for habitat destruction (Fujita et al., 2012). Harnessing tradable rights in active restoration (such as reef restoration; Rinkevich (2015b)) by non-public stakeholders may add real compensation for losses and on the other hand may be used as an efficient tool for a long-term financial support of marine ecosystem restoration, as it is done by the logging industry in forest habitats.

One of the main investment vehicles in marine conservation is the Entrepreneurial Marine Protected Area (EMPA) (Cred Suisse AG, 2016). An EMPA is a management area that is primarily funded by a profit-bearing business model, typically associated with nature tourism. EMPAs belong to marine impact investments as they are designed to produce environmental and social impacts on coastal communities, and they primarily employ business models instead of grants to achieve those outcomes. When properly managed, MPAs have proven to control overfishing, reduce user conflicts, increase community support and improve enforcement of water quality regulations.

Benefits include food and incomes for local communities, opportunities for tourism businesses, protection from coastal erosion and a country more resilient to climate change (Halpern, 2003). While many terrestrial protected areas have private sector involvement in the region (Juffe-Bignoli et al., 2014), significantly fewer MPAs include the private sector. A potential obstacle in the further development of this tool are results showing that the management instruments employed in MPAs, in general, often fail to achieve their conservation objectives (Miller and Russ, 2014; Gill et al., 2017).
5.5.3.4 Conflicting and Ineffective Governance

The changes in the marine environment due to the acidification of the ocean is having significant impacts on marine ecosystems, particularly on the calcareous organisms like molluscs and corals which play important ecological roles and on the health of organisms (like fish) that depend on the building blocks of the ecosystems (like corals in the coral reefs). The consequences on the livelihood of millions of people who are dependent on fisheries for their living, are very substantial. The ocean acidification could undermine the social, economic and environmental pillars of sustainable development.

There are no current international instruments which specifically address ocean acidification or its impacts on the marine environment. At the global and at the regional level there exists, however, a number of international instruments aiming at combating different sources of pollution which affect the marine environment. This could be of relevance in addressing some ways to prevent the increase of the acidification of the ocean. In addition, a number of declarations or instruments of a voluntary nature, in which States have committed themselves in taking actions or meeting policy goals and targets, are also significant.

[START BOX 5.5 HERE]

Box 5.5: Policy Responses to Ocean Acidification: Is there a Governance Gap?

Ocean acidification is not mentioned in the Paris Agreement on climate change, and has not been given detailed attention in other UNFCCC discussions. Is that because the topic is not considered to be part of climate change, or because it does not warrant distinct governance arrangements? If climate is strictly defined as the long-term descriptor of meteorological conditions, then changes in ocean chemistry are out of scope. However, physical changes in the ocean, such as sea level rise, are uncontroversially considered part of climate change, and the CO₂-driven change in ocean pH is an inevitable, although indirect, chemical consequence of human perturbations of atmospheric composition – the driver of all other changes to the climate system. The close association between ocean acidification and other components of climate change was recognised in IPCC AR4 and AR5, with Working Groups I and II (but not III) giving consideration to the ecological and economic impacts of ocean acidification that are already occurring, and that are projected to greatly increase under future emission scenarios.

Most literature on ocean acidification is relatively recent, with a five-fold increase during the past decade (Riebesell and Gattuso, 2014; Browman, 2016). Nevertheless, IPCC AR5 WG I considered that there was high confidence that anthropogenic CO₂ had already caused global surface ocean pH to emerge from natural variability, and that pH will continue to decrease with increasing CO₂ emissions, whilst AR5 WG II considered there was high confidence that ocean acidification will increasingly affect marine organisms, ecosystems, and the goods and services they provide. Such assessments are confirmed in this report (Sections 5.2.2, 5.2.3, 5.3.2, 5.3.3, 5.4.1). Concerns regarding ocean acidification have been raised by many international bodies, including the Convention on Biological Diversity (CBD, 2016), the United Nations Convention on the Law of the Sea (UNCLOS), the United Nations Environment Programme (UNEP), the Intergovernmental Oceanographic Commission (IOC) of the United Nations Educational, Scientific and Cultural Organization (UNESCO), the World Meteorological Organization (WMO), and the International Atomic Energy Agency (IAEA); the interests in ocean acidification of these and other bodies are summarised in CBD (2014). Public awareness of ocean acidification is, however, low (e.g., only 14% of Europeans consider that they are ‘informed’ on the topic; Buckley et al. (2017)) and the topic has been neglected to date from a regulatory perspective (Fennel and VanderZwaag, 2016). The most substantive governance outcome to date is UN Sustainable Development Goal 14.3 (Section 5.3.2), with its non-binding and relatively general commitment to ‘minimise and address the impacts of ocean acidification, including through enhanced scientific cooperation at all levels’.

Whilst a new UN mechanism specifically to address ocean acidification has been proposed (Kim, 2012), there are pragmatic arguments favouring strengthened UNFCCC involvement (Harrould-Kolieb and Herr, 2012). Action to reduce CO₂ emissions will necessarily, to some degree, address ocean acidification, although there are topic-specific considerations. Governance issues not only relate to pH-associated regime changes, thresholds or tipping-points (Hughes et al., 2013; Good et al., 2018), but also with regard to ocean
adaptation impacts in scenario-modelling for emission reductions (Steinacher et al., 2013), and specific
consequences for ocean acidification of several climate policy responses. In particular, any leakage from
sub-seafloor carbon storage is very likely to produce local pH impacts, similar to natural CO2 vents
(Blackford et al., 2014), whilst some CO2 removal techniques (negative emissions) such as ocean
fertilization, are likely to worsen acidification for the ocean as a whole (Cao and Caldeira, 2010; Williamson
and Turley, 2012). If policy interventions were to involve solar radiation management, the climate might be
stabilized, yet acidification would essentially continue unabated (high confidence) (Williamson and Turley,
2012; Keller et al., 2014).

Adaptation to climate change could also include a more integrated approach to reduce ocean acidification
impacts (Section 5.5.2). Proposed actions for ocean acidification (Kelly et al., 2011; Billé et al., 2013; Strong
et al., 2014; Albright et al., 2016) include reduction of pollution and other stressors (thereby strengthening
resilience); water treatment, e.g., for high-value aquaculture; adapting human activities; and repairing
damages. Seaweed cultivation and seagrass restoration may slow long term changes at the local level, but can
also exacerbate short-term variability (Sabine, 2018). These measures are generally applicable to relatively
limited spatial scales: whilst they may succeed in ‘buying time’, their future effectiveness will decrease,
unless underlying global drivers are also addressed (high confidence).

Well-coordinated and fit for purpose ocean acidification monitoring is being developed through the Global
Ocean Acidification Observing Network (Newton et al., 2015), integrated with other marine monitoring at
local, national and global levels. Such action can be considered an essential part of the governance response,
and is now used operationally by the US oyster cultivation industry (Barton et al., 2015a). Good progress has
been made in characterising seasonal patterns (Sutton et al., 2016) and identifying the many other factors
that can cause high pH variability in coastal waters (Duarte et al., 2013). In the US, the 2009 Federal Ocean
Acidification Research and Monitoring Act (FOARAM) provided the basis for coordinated national
monitoring of ocean acidification and its consequences (IWGOA, 2014). The FOARAM Act also required:
the development of adaptation strategies to conserve organisms and ecosystems vulnerable to ocean
acidification; the consideration of associated socio-economic impacts; and the development of standardised
protocols for natural values and their variation. In the European Union; measurement of ocean acidification
conditions is encouraged, but is not statutory – since pH change is considered to be a ‘prevailing condition’,
outside national management control (European Union, 2008).

[END BOX 5.5 HERE]

5.5.3.5 Policy Framework and Voluntary Instruments

With regard to policy framework, each year the General Assembly adopts resolutions on the law of the sea
dealing with all issues of marine affairs in an integrated approach. They highlight the problems that require
the mobilization of the international community with a view to finding specific solutions. Their
programmatic character places them in the category of soft law which, in the legal arsenal applied to the
oceans, is beginning to play a widely recognized role.

The question of acidification of the ocean was first introduced in resolution A/RES/62/215 adopted in 2007
in these terms:

‘81. Notes the work of the Intergovernmental Panel on Climate Change, including the finding
that, while the effects of observed ocean acidification on the marine biosphere are as yet
undocumented, the progressive acidification of oceans is expected to have negative impacts on
marine shell-forming organisms and their dependent species, and in this regard encourages States
to urgently pursue further research on ocean acidification, especially programmes of observation
and measurement.’

Hence, each year, the General Assembly continued in the omnibus resolutions to deal with this troubling
phenomenon in order to make States more and more aware of it and of its dangerous consequences to the
marine ecosystems (A/RES/63/111 para.99; A/RES/64/71, para.113; A/RES/65/37A, para.129;
With the Rio + 20 summit and the recommendations in the ‘Future we want’, the resolutions devoted more paragraphs and requested more action, effort and initiatives to further study, to minimize impacts as well as to enhance international cooperation.

It is on the basis of Rio + 20 summit and the resolutions of the General Assembly requesting initiatives that the IAEA and the Centre Scientifique de Monaco organized the 3rd international workshop which took place in Monaco in January 2015. It aimed to provide policy makers and marine resource managers with information and recommendations to prepare for social and economic impacts of ocean acidification on coastal communities.

With regard to voluntary instruments, land-based activities are responsible for most of the pollution of the oceans and affect the most productive areas of the marine environment. The 1982 Convention sets out specific provisions for the prevention, control and reduction of pollution from land-based sources. However, it gives to States a certain degree of discretion with respect to the implementation of international rules, in deference to the sovereignty that a State exercises over the territory where such land-based pollution may arise. As seen above States, however, have accepted to be bound by applicable rules at the regional level.

The Governing Council of UNEP identified in 1982 the question of land-based source of pollution as one of the three priorities for the development of environmental law. In 1985 the Montreal Guidelines for the Protection of the Marine Environment from Land-based Sources were adopted. They were guided by the regional examples as the North-East Atlantic, the Baltic and the Mediterranean Sea. The Guidelines have set the stage for a more comprehensive and global approach to addressing the causes of marine and coastal degradation.

In November 1995, two international documents were adopted by an intergovernmental conference: The Washington Declaration on the Protection of the Marine Environment from Land-based Activities and The Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA) (See UN document A/51/116).

The GPA provides guidance to national and/or regional authorities to devise and to implement sustained action in order to prevent, reduce, control and/or eliminate marine degradation from land-based activities.

The GPA addresses the impacts of land-based activities on the marine and coastal environment, including contaminants, physical alteration, point and non-point sources of pollution and areas of concern such as critical habitats, habitats of endangered species and protection of ecosystem components, such as breeding and feeding grounds.

From a strictly legal perspective, both the Washington Declaration and the GPA have no binding force and are considered to be ‘soft law’. They have, however, both obtained an enormous amount of support from States and intergovernmental and non-governmental organizations that view the GPA as an essential instrument for combating marine environmental degradation from land-based sources of pollution.

Since an estimated 3 billion people depend on marine and coastal biodiversity for their livelihoods, the disruptions to the delivery of marine ecosystem services caused by climate change and ocean acidification will seriously affect the economy of coastal communities and could also impact food security and could in turn result in increased poverty (see Report of UNGA on Oceans and the law of the sea A/70/74, paras 67-70) and as discussed in Section 5.3. To consider, however, fishing activities, particularly Illegal, Unreported and Unregulated fishing, and associate it with acidification is not entirely correct. If there is at all a link it is that it adds to the problems of overfishing which affect sustainable fisheries.

To combat overfishing and particularly IUU fishing, several instruments were adopted under the auspices of FAO. They deal either with the respect of international conservation measures by fishing vessels like the 1993 Agreement on compliance with conservation and management measures, or the increase control by port States to ensure that fishing vessels entering in a port did not violate the measures taken at the regional

or sub regional level. In this case two major instruments can be mentioned: i) the 1995 Agreement on straddling fish stocks and highly migratory fish stocks, and ii) the most recent one, far reaching in the fight of IUU fishing that is the 2009 Agreement on port state measures to prevent, deter and eliminate IUU fishing, in force in 2016 (FAO, C2009/REP et Corr.1 et 3, Annexe E). It will enhance regional and international cooperation and block the flow of IUU caught fish into national and international markets.

5.6 Synthesis

This chapter has documented an extra-ordinary array of changes that have been observed in the open ocean and coastal seas. The physical and ecological state of the ocean is clearly changing, and human communities are being affected by these changes. These changes occur in all of the regions and all depths of the ocean from the surface to the abyssal plains of the Southern Ocean (Section 5.2.2). All of the marine systems (natural and human) that have been covered in this chapter have changed, some very significantly, some less so that are related directly or indirectly to ocean changes. All of these lines of evidence from the literature point to a profound and pervasive change through the whole ocean.

The level of knowledge and confidence of the changes in the marine environment is uneven. Many of the observed changes can be directly attributed to human influence from rising greenhouse gases and other anthropogenic forcings (e.g., aerosols and ozone depletion). On the other variables and systems the evidence is less clear, but the relationship between rising greenhouse gases, changing ocean, marine ecosystems and dependent communities is still surprisingly robust when considered together. In many cases, the assessments are undertaken using a risk assessment framework. By combining ecological and physiological knowledge (from experiment or direct observations) with the climate drivers, the change in risk level for the recent past, and long term future are assessed (e.g., Sections 5.2.5 and 5.3.4). Taken globally, all the assessed marine ecosystems have elevated risk of impacts on biodiversity, ecosystem structure and services with increasing greenhouse gas emission. Specifically, all the coastal ecosystems that were assessed, where linkages between natural systems and human communities are the strongest, all had increased risk, and none saw a risk reduction (or beneficial change) from a warming climate.

The observed and projected changes in the ocean systems that are covered are consistent with our understanding of the ocean chemistry and circulation, and our knowledge of the ecosystems responses. These changes in the open ocean and coastal seas have consequences on human communities and affect all aspects of wellbeing and have economic and environmental costs (Section 5.4). The range and diversity of impacts is remarkable with varying consequences for the wider community when analysed across the key marine ecosystems services. This range of impacts substantially raises questions about the ability for human society to achieve the Sustainable Development Goals (e.g., Figure 5.19).

Risk-reduction responses and their governance through adaptation at the local scale is the most common approach to responding to climate change from ocean systems. It is clear that there are many choices for reducing risk of climate change. Many of the actions have benefits and relatively few dis-benefits, while others have large dis-benefits and marginal effectiveness (Figure 5.21). Many of the risk reduction approaches are limited in their capacity to reduce the risks of climate change, or are at best temporary solutions, and this is significant challenge to adapting to climate change. The assessment points to the increased effectiveness and importance of a portfolio of different types of mitigation and adaptation options. Governance is also critical element in the portfolio of options and occurs at a variety of scales, local, national and international. Such responses can be more effective with the support of scientific information, Local knowledge and Indigenous knowledge, and the consideration of local context and inclusion of stakeholders.

5.7 Key Uncertainties and Gaps

As stated in the introduction, this chapter was designed around a set of guiding questions (Section 5.1). These guiding questions mean that the report covers both regional and global scales and across aspects of human social systems, including governance and institutions, and adaptation pathways for dependent communities. This assessment is new in terms of linking together a broad and diverse set of ocean disciplines and therefore also provides a unique perspective on where the key uncertainties appear in these
systems. It is these key uncertainties that limit the extent of the assessments that were possible in this report. There are surprising gaps in our knowledge that potentially hamper clear decision making by policymakers. We note the outstanding ones from this assessment.

- **Variance in human systems and effectiveness of responses:** the wide range of contributing factors (physical, social and economic) that interact with localised climate projections (which typically have high levels of uncertainty) make projecting site-specific costs of impacts and benefits of adaptation difficult. There were few examples in the literature of assessments of implemented adaptation actions, and there was low confidence in their reliability and provenance, thus largely precluding any assessments of their cost effectiveness. This lack of evidence on cost/benefit particularly affected assessments in Section 5.5. Adaptation responses to climate change have been undertaken by communities and governments. However, their effectiveness for mitigating the risks of climate change (e.g., different types of adaptation response on the coasts, Section 5.5.2) is largely unassessed here, and consequently precludes a global understanding of the capacity in the world to address the risks of climate change on our coastal seas and open ocean. A partial solution would be establishing an appropriate ocean and coasts database for these types of studies.

- **Biological processes and monitoring:** while some biological processes in the ocean are well understood there are a surprising number of marine environments where the lack of scientific understanding frequently precludes assessments of risks and the capacity to project their response to climate change: these include incomplete ‘models’ for quantitative projections of the future of key organisms and ecosystems, gaps in our knowledge of feedbacks in biological systems particularly in coastal ecosystems (Section 5.3.4), the capacity and limits of biological adaptation for many organisms and ecosystems, and the very sparse knowledge of the sensitive deep ecosystems that may be uniquely sensitive to climate change (Section 5.2.2 and 5.2.4).

- **Physical and biogeochemical processes:** while the Earth system is better monitored and the relevant data are more accessible than the first two bullets described above, significant gaps remain. We wish to highlight deep ocean temperature, salinity measurements for sea-level and closure of the energy budget, oxygen and carbon measurements dense enough to measure de-oxygenation of the world ocean and track the mechanisms driving the ocean carbon cycle; and the rates of carbon burial in coastal sediments as significant omissions. Projections of future changes in the Earth system depend on the use of coupled models. While large-scale climate changes are believed to be well described by these models, there are still uncertainties in the timing, magnitude and patterns of the projected changes arising from physical and ecological processes that are omitted or incompletely understood, from intrinsic biases arising from the models’ formulation, and from resolutions that are still too coarse to capture important regional processes and their interactions with the global ocean. Other examples of under-assessed biogeochemical process in the ocean that may have implications for the Earth system under climate change include methane in deep ocean sediments (Section 5.2.4). Open ocean primary productivity and its projections still needs critical corroborating measurements to track the potential changes in ocean productivity with higher confidence (Sections 5.2.2 and 5.2.3).

See Cross-Chapter Box 5 ‘Southern Ocean Circulation: Drivers Changes and Implications’ in Chapter 3.

[START FAQ5.1 HERE]

**FAQ5.1: How is life in the sea affected by climate change?**

Life in most of the global ocean, from pole to pole and from sea surface to the abyssal depths, is now experiencing higher temperatures due to human-driven climate change. In many places, that increase may be barely measurable; in others, particularly in near-surface waters, warming has already had dramatic impacts on marine animals, plants and microbes. Due to closely-linked changes in sea water chemistry, less oxygen remains available (the process is called ocean deoxygenation) and more carbon dioxide is dissolved, causing ocean acidification. Non-climatic effects of human activities are also ubiquitous, including over-fishing and pollution. Whilst the combination of these stressors is likely to be harmful to almost all marine organisms, food-webs and ecosystems, some are at greater risk (FAQ5.1, Figure 1). Consequences for human society can be serious if insufficient action is taken to constrain future climate change.
Warm-water coral reefs are particularly vulnerable, since they can suffer high mortalities when water temperatures persist at between 1°C–2°C above the normal range. Such conditions occurred in many tropical seas between 2015–2017 and resulted in extensive coral bleaching, when the coral animal hosts ejected the algal partners upon which they depend. After mass coral mortalities due to bleaching, reef recovery typically takes at least 10–15 years. Accelerating sea level rise, reef erosion and acidification are slowing down reef recovery rates, especially when bleaching recours. Whilst some coral species are more resilient than others, further reef degradation due to future climate change now seems inevitable. This can have serious implications, like loss of coastal protection for many islands and low-lying areas and loss of the high biodiversity these reefs host. Coral habitats can also occur in deeper waters and cooler seas. Although these are not at risk from bleaching, they may weaken or dissolve under ocean acidification, and other ocean changes.

More mobile species, such as fish, may respond to climate change by moving to more favourable regions/areas, with populations moving poleward, or to deeper water, to find their preferred range of water temperatures or oxygen levels. As a result, projections of total future fishery yields under different climate change scenarios only show a moderate decrease of around 4% (~3.4 million tonnes) per degree Celsius warming. However, there are dramatic regional variations. Under scenarios of high greenhouse gas emissions, fish biomass and maximum catch potentials could be up to 50% lower in tropical regions by the end of the century. Losses may be partly offset by increases in some fish stocks in polar regions. The extent of such gains is uncertain, as fish populations depleted by over-fishing, and subject to other stressors, may not be capable of making the migrations that are assumed in model simulations.

In polar seas, species adapted to life on or under sea-ice are directly threatened by habitat loss due to climate change. The Arctic and Southern Oceans are home to a rich diversity from fish, krill and seafloor invertebrates to whales, seals, polar bears or penguins. Their complex interactions may be altered if new warmer-water species extend their ranges as sea temperatures rise. The impacts of acidification affecting smaller and larger shelled organisms as well as increased human activities (e.g., shipping) in ice-free waters can add to this.

Whilst a few of the above climate change impacts may benefit humans, most will be disruptive for societies, economies and ecosystems. The most damaging economic effects will be in tropical regions, affecting people and societies with least adaptive capacity. The scale of that damage can, however, be much reduced if the world as a whole keeps to the Paris Agreement, limiting warming to well below 2°C.

FAQ5.1. Figure 1: Summary schematic of the impacts and resulting consequences of climate change (warming, acidification, storminess and deoxygenation) and other human impacts, upon coral reefs, polar seas and fisheries, discussed in this FAQ.


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### Appendix 5.A: Supplementary Material

#### Appendix 5.A, Table 1: Compiled information on the rate of pH change from various time series and ship reoccupations. Modified after Williams et al. (2015).

<table>
<thead>
<tr>
<th>Region</th>
<th>pH change (pH units decade(^{-1}))</th>
<th>Uncertainty</th>
<th>Study</th>
<th>Study period</th>
<th>Study type</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irminger Sea</td>
<td>-0.026</td>
<td>0.006</td>
<td>Bates et al. (2014)</td>
<td>1983–2012</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Atlantic (BATS)</td>
<td>-0.017</td>
<td>0.001</td>
<td>Bates et al. (2014)</td>
<td>1983–2012</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Atlantic (BATS)</td>
<td>-0.018</td>
<td>0.002</td>
<td>Takahashi et al. (2014)</td>
<td>1983–2010</td>
<td>Time series</td>
</tr>
<tr>
<td>Iceland Sea</td>
<td>-0.023</td>
<td>0.003</td>
<td>Olafsson et al. (2009)</td>
<td>1985–2008</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Pacific (ALOHA)</td>
<td>-0.018</td>
<td>0.001</td>
<td>Dore et al. (2009)</td>
<td>1988–2007</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Pacific (HOT)</td>
<td>-0.016</td>
<td>0.001</td>
<td>Bates et al. (2014)</td>
<td>1988–2012</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Pacific (HOT)</td>
<td>-0.018</td>
<td>0.001</td>
<td>Takahashi et al. (2014)</td>
<td>1988–2009</td>
<td>Time series</td>
</tr>
<tr>
<td>N.W. Pacific</td>
<td>-0.02</td>
<td></td>
<td>Ishii et al. (2011)</td>
<td>1994–2008</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Atlantic (ESTOC)</td>
<td>-0.018</td>
<td>0.002</td>
<td>Bates et al. (2014)</td>
<td>1995–2012</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Atlantic (ESTOC)</td>
<td>-0.017</td>
<td>0.001</td>
<td>González-Dávila et al. (2010)</td>
<td>1995–2004</td>
<td>Time series</td>
</tr>
<tr>
<td>Caribbean (CARIACO)</td>
<td>-0.025</td>
<td>0.004</td>
<td>Bates et al. (2014)</td>
<td>1995–2012</td>
<td>Time series</td>
</tr>
<tr>
<td>N. Atlantic (ESTOC)</td>
<td>-0.02</td>
<td>0.004</td>
<td>Takahashi et al. (2014)</td>
<td>1996–2010</td>
<td>Time series</td>
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<td>S.W. Pacific (Munida)</td>
<td>-0.013</td>
<td>0.003</td>
<td>Bates et al. (2014)</td>
<td>1998–2012</td>
<td>Time series</td>
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<tr>
<td>E. Equatorial Indian</td>
<td>-0.016</td>
<td>0.001</td>
<td>Xue et al. (2014)</td>
<td>1962–2012</td>
<td>Merged ship occupations</td>
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<td>Polar Zone Southern Ocean</td>
<td>-0.02</td>
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<td>0.005</td>
<td>Midorikawa et al. (2010)</td>
<td>1983–2007</td>
<td>Merged ship occupations</td>
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<td>Byrne et al. (2010)</td>
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<td>Pacific Southern Ocean (S4P)</td>
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<td>Williams et al., 2015</td>
<td>1992–2011</td>
<td>Merged ship occupations</td>
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<td>S. Pacific</td>
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<td></td>
<td>Waters et al. (2011)</td>
<td>1994–2008</td>
<td>Merged ship occupations</td>
</tr>
<tr>
<td>Pacific Southern Ocean (P16S)</td>
<td>-0.024</td>
<td>-0.009</td>
<td>Williams et al., 2015</td>
<td>1995–2011</td>
<td>Merged ship occupations</td>
</tr>
<tr>
<td>Drake (PZ)</td>
<td>-0.015</td>
<td>0.008</td>
<td>Takahashi et al. (2014)</td>
<td>2002–2012</td>
<td>Merged ship occupations</td>
</tr>
<tr>
<td>Drake (SAZ)</td>
<td>-0.023</td>
<td>0.007</td>
<td>Takahashi et al. (2014)</td>
<td>2002–2012</td>
<td>Merged ship occupations</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Coastal Ecosystems</th>
<th>Tidal exposure</th>
<th>Ecosystem component</th>
<th>Sensitivity</th>
<th>Adaptive capacity</th>
<th>Vulnerability (Risk)</th>
<th>Other hazards</th>
<th>Impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salt marshes</td>
<td>IT</td>
<td>V, SE, F, SB</td>
<td>SST–V, F: medium (obs-proj): but positive/negative effects. pH/CO2: -V, low (exp); SLR: -V, SE, SB: high (obs-proj). Increased storm disturbance (but positive/negative effects depending on sediment supply) &gt; V, SE.</td>
<td>Saltmarshes are initially resilient to SLR due to soil accretion (biomass accumulation and sediment deposition) but system weak to withstand SLR after 60 years under RCP 4.5.</td>
<td>Substantial (medium confidence)</td>
<td>Coastal development causing squeeze, altered flushing regimes. Species invasions. Mangrove encroachment. Eutrophication. Land use change for agriculture. Dredging. Overfishing.</td>
<td>Reduction in above- and belowground plant biomass, carbon storage and soil elevation due to SLR (tolerance to inundation and salinization depends on plant species and organic accretion). Shifts in plant species, local extinctions. Habitat reconfiguration from intertidal to subtidal, biodiversity loss. The projected loss in global coastal wetlands (2040-50) is in part counterbalanced by some increase in arid and sub-tropical marshes (3-6%) under low SLR scenario and lateral reconfiguration of sediments (if not constructed by human constructions) (some projections up to 60 % gain).</td>
</tr>
<tr>
<td>Mangroves</td>
<td>IT</td>
<td>V, SE, F, SB</td>
<td>SST–V, F: low-medium (obs-proj): but positive/negative effects. pH/CO2: -V, low (obs-proj): but positive/negative effects SLR: -V, SE, SB: high (obs-proj). Increased storm disturbance (but positive/negative effects depending on sediment supply) &gt; V, SE.</td>
<td>Long-living, large plants. Increased storm disturbance (but positive/negative effects depending on sediment supply) &gt; V, SE.</td>
<td>Low/Moderate (Long-living, large plants. Initially resilient to SLR due to soil accretion. Ecosystem under intense human impacts. Rehabilitation practices can stimulate soil elevation).</td>
<td>Low (medium confidence)</td>
<td>Coastal development causing squeezing. Habitat degradation due to human activities: Deforestation, aquaculture, agriculture, urbanization.</td>
</tr>
<tr>
<td>Seagrass meadows</td>
<td>S</td>
<td>V, F, MM, SB, T</td>
<td>SST–V, F, MM, T: high (obs-proj): but positive/negative effects. pH/CO2: -V, high (exp); SLR: -GM, BI, V, T: high (obs-proj). Increased storm disturbance &gt; V.</td>
<td>Very high (very high confidence)</td>
<td>Low/Moderate (high confidence)</td>
<td>Eutrophication, habitat degradation, biological invasions.</td>
<td>Reduction in plant fitness due to temperature stress and reduction in underwater light levels due to SLR. Mass mortality events, spread of invasive tropical species. Severe habitat loss of the endemic Posidonia oceanica in the Mediterranean, by 2050 and potential extinction by 2100 under RCP 8.5. Warming will lead to significant reduction of Cymodocea nodosa meadows (46 %) in the Mediterranean, and expansion into the Atlantic. Increased herbivory by tropical consumers on temperate seagrasses, ecosystem biodiversity loss.</td>
</tr>
</tbody>
</table>
### Appendix 5.A, Table 3: Review of literature on linkages between ecosystem and ecosystem services in eastern boundary upwelling systems.

<table>
<thead>
<tr>
<th>System components</th>
<th>Observed</th>
<th>Projected</th>
<th>Confidence and Uncertainty</th>
<th>References &amp; Lines of evidence</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Coastal temperature and stratification</strong></td>
<td>Complex integration of global (climate change) and local (coastal upwelling) processes. Differences in global and EBUS SST trends are observed.</td>
<td>Ocean warming and stratification might be ameliorated by increased upwelling. At local scales however coastal upwelling may be reduced by ocean stratification.</td>
<td>Low confidence; Trends are sensitive to location (nearshore vs. offshore), resolution, period and dataset considered, as well as to decadal variability.</td>
<td>Brochier et al. (2013) M Schroeder et al. (2013) O Oyarzun &amp; Brierley (2018) M Xiu et al. (2018) M</td>
</tr>
</tbody>
</table>
5.1 Projection of Time of Emergence of Habitat Suitability Index of Marine Species

The calculation of time of emergence followed the procedure described in Frölicher et al. (2016).

Appendix 5.A, Figure 1: Schematic diagram explaining the calculation of time of emergence of habitat suitability index of marine species.
Step 1: Computing the Habitat suitability index (HSI) of the species using BIOMOD2 (ref thuillier). The HSI is computed using Boosted regression tree and maxent for IPSL, MPI and GFDL model under RCP8.5.

Step 2: For each geographical cell, we export the yearly time series of HSI for each ESM.

Step 3: We compute an average HSI between each ESM.

Step 4: We compute the standard deviation for the reference period 1970-2000 for each ESM (internal variability) and between ESM (intermodal variability).

Step 5: We set the boundary limit as the maximal and minimal average value within the reference period (1970-2000) +/- the internal variability and intermodal variability. If the minimum boundary is computed under 0, we set it at HSI=0;

Step 6: We identified the first year when the average model is over or under the boundary defined and defined it as the time of emergence.

Step 7: We compute the total area per time of emergence and plot the area vs time.