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

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Executive Summary

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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.

7 8 The oceans have warmed unabated since 2004, continuing the multi-decadal ocean warming trends 9 documented in the IPCC Fifth Assessment Report (AR5). Ocean warming dominates the increase in heat 10 energy stored in the climate system and pervades the ocean from the surface to bottom. The upper ocean 11 (0-700 m) and 700-2000 m layers have both warmed from 2004 to 2016 (virtually certain') and the abyssal 12 ocean (below 4000 m) continues to warm in the Southern Hemisphere (high confidence²). The observed 13 upper ocean temperature or heat changes are very likely to have a substantial contribution from 14 anthropogenic forcings since 1970. There is increased agreement between coupled model simulations of 15 anthropogenic climate change and observations of changes in ocean heat content (high confidence). The 16 overall warming of the ocean will continue to increase this century even if radiative forcing stabilizes (e.g., 17 RCP2.6, high confidence). Under the RCP2.6 and 8.5 scenarios, by 2100 the ocean is likely to take up about 18 3 and 6 times, respectively, the roughly 500×1021 J that the oceans have already taken up since the start of 19 the 20th century. $\{5.2.2.2\}$ 20

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}

27 Multiple datasets and models show that the oceanic sink for atmospheric CO₂ has continued to

28 strengthen in the recent two decades in response to the growing burden of CO₂ in the atmosphere 29 (virtually certain), while an important degree of temporal variability has emerged. A consistent picture 30 emerges from independent ocean surface and interior datasets (covering the last few decades and the period 31 since the industrial revolution), as well as ocean models that ocean carbon uptake has continued to increase 32 commensurate with the atmospheric carbon dioxide growth rate (high confidence). {5.2.2.3} It is very likely 33 that the ocean has taken up about ~ $25\pm5\%$ of total anthropogenic emissions in the recent two decades. 34 Evidence is growing that the ocean carbon sink is dynamic on decadal timescales, especially in the Southern 35 Ocean, which affects the total global ocean carbon sink (*medium confidence*) .{5.2.2.3} Despite our 36 improved understanding of the magnitude of ocean carbon uptake and its variability, there is still substantial 37 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}
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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 typeset 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 typeset 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).

1 The ocean is observed to be losing oxygen and oxygen minimum zones have expanded. Overall, the 2 oxygen content of the global ocean has declined by about 2%, largely in responses to changes in ocean 3 circulation and respiratory demand (high confidence). {5.2.2.4} The largest reductions in oxygen have been 4 observed in the Southern Ocean, South Atlantic and North Pacific (medium confidence), but there is low 5 confidence for changes in the tropical ocean due to natural variability and limited agreement across studies. 6 Over the next century oxygen declines of 3.5% by 2100 are predicted by CMIP5 models globally (medium 7 confidence), with low confidence at regional scales, especially in the tropics. The largest changes in the deep 8 sea will occur after 2100. {5.2.2.4}. Where oxygen is already low, even very small declines in oxygen 9 availability can lead to decreases in biodiversity, nutrient cycling, and ecosystem productivity. {5.2.2.5, 10 5.2.3 and Box 5.1} 11 12 Ocean primary productivity is projected to show regional variability in response to climate driven 13 changes in nutrient supply with implications for pelagic and seafloor ecosystems. There is high 14 confidence that future changes to ocean primary productivity will be driven by region specific changes in 15 magnitude and ratio of nutrient supply. {5.2.2.5} In general, CMIP5 models project a small decrease in 16 global organic matter production (medium confidence) with increases in high latitude (low confidence) and 17 decreases in low latitude (medium confidence) in response to changes ocean nutrient supply. There is 18 currently low confidence in appraising past open ocean productivity trends, including those determined by 19 satellites, due to newly identified region-specific drivers and the lack of corroborating in situ time series 20 datasets. {5.2.2.5} 21 22 Emergence of novel ocean conditions for organisms from plankton to mammals are driving changes in 23 eco-physiology, biogeography and ecology and biodiversity (high confidence). Observed and projected 24 population declines in the lower-latitude range boundary (medium confidence), expansion in the poleward 25 boundary (high confidence), earlier timing of biological events (high confidence), and overall shift in 26 biomass and species composition (very high confidence) are consistent with expected responses to climate 27 change. {5.2.2, 5.2.3, 5.3.2, 5.3.3} Changes in biogeography and community structure are projected to 28 continue in the 21st century (high confidence), with potential global animal biomass projected to decrease 29 by $4.8\% \pm 3.6\%$ s.d. and $17.2\% \pm 11.1\%$ s.d. under RCP2.6 and 8.5, respectively, by 2090–2099 relative 30 to 1990–1999 (likely). {5.2.3.1} Scope for adaptation for many organisms to cope with novel environmental

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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

conditions is limited (medium confidence), particularly those higher up in the ocean foodweb and for high

36 and retreat from rising sea level and more severe storms are leading to losses of vegetation and benthic fauna 37 e.g., in sandy beaches and saltmarshes (*high confidence*). {5.3.3.4, 5.3.3.5} Extreme events such as heat 38 waves and storms are exacerbating the rate of ecosystem changes, such as those observed in kelp forest and 39 seagrass meadow (high confidence). {5.3.3.4, 5.3.3.9} Herbivory intensified by warming will result in 40 physical and physiological stress on coastal vegetation and reduce their productivity (*medium confidence*). 41 {5.3.3} Compounding effects of warming, deoxygenation, acidification and changes in nutrient supplies 42 exacerbates the decrease in species richness and spatial heterogeneity in coastal ecosystems (medium 43 confidence). $\{5.3.3\}$

44

45 Almost every major coral reef system (shallow and deep) are vulnerable to climate change with clear 46 regional differences in their sensitivities (high confidence). Ocean warming, acidification, rising sea level 47 and more severe storms impede reef resilience on a global level and augment reef destruction (high 48 confidence). {5.3.3.6, SROCC Chapter 6} Shallow coral reefs that are not degraded by other impacts such as 49 overfishing and nutrient enrichment could constitute an important refuge to reefs degraded by climate 50 change. However, loss of deep-water coral reef habitat is virtually certain under projected ocean 51 acidification (aragonite under-saturation) through dissolution and intensified bio-erosion of the non-living 52 matrix. {5.3.3.6} 53

54 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

carbon emission scenarios. {5.2.3, 5.3.3}

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 (*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 (*medium confidence*) {5.2.4}.

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6 Seafood provision from fisheries and aquaculture are impacted by climate change (high confidence), 7 reducing the revenue from fisheries sectors (medium confidence), and impacting the livelihood of the 8 dependent communities (medium confidence) and food security of vulnerable people (medium 9 confidence). Fisheries catches and their composition are already affected by the effects of warming, 10 deoxygenation and changes in primary production on growth, reproduction and survival of fish stock (high 11 *confidence*) {5.4.1}. Shellfish aquaculture is also sensitive to ocean acidification (*high* confidence) 12 {5.4.2.2.1}. In the 21st century, potential global fisheries catches are projected by multiple models to 13 decrease with increasing dominance of warmer water species with increasing CO₂ emission (medium 14 *confidence*), although the changes in realized catch will depend strongly on fishing intensity {5.4.1.3.1}. The 15 decrease in catch in the tropical ocean is projected to be among the largest regional declines (very likely). 16 Farmed species will be exposed to increased risk of disease and harmful algal blooms {5.4.1.3.1}. 17 Consequently, people who depend on fisheries and related-sectors will experience substantial decline in their 18 income, livelihood and availability of animal-sourced nutrients (medium confidence) {5.4.2.1.2}.

19

20 Ocean ecosystems' role in climate regulation, supporting culture and recreation and their intrinsic

21 values that are important for human wellbeing are being threatened by climate change because of the

22 loss and degradation of biodiversity and ecosystems functions (high confidence). This is evidenced from 23 reduction in nutrient cycling in the deep sea floor ecosystems through warming- and hypoxia-induced 24 reduction in biodiversity (medium confidence), {5.4.1.2.2} reduction in the quality and quantity of tourist 25 attractions as coral reefs are degraded from warming and increased severity of storm events (high 26 confidence), {5.4.1.3.2} reduction in carbon stock and sequestration in salt marshes through reduced habitat 27 availability for fauna requiring open vegetation structure as a result of sea level rise (high confidence), 28 {5.4.1.3.2} and potential loss of opportunities for using ocean ecosystems for education and the relationship 29 with Indigenous knowledge and culture (medium confidence). {5.4.1.1.2, 5.4.1.2.2, 5.4.1.3.2, 5.4.2.1}

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31 Climate change impacts on the Ocean is expected to substantially increase the risks on human health 32 and conflicts within and between political entities in the 21st century (*medium confidence*).

33 Elevated risks of water-borne disease, food poisoning and pollutant contamination on human health are 34 directly related to climate change through increase prevalence of pathogens and harmful algal bloom, and 35 increased contaminant bioaccumulation under warming and high CO₂ conditions (high confidence). {Box 36 5.3, 5.4.2.1.1} The projected decline in fish supply and key fish micronutrients threaten food security of 37 coastal communities that are strongly dependent on seafood, such as coastal Indigenous people and many 38 tropical countries. {5.4.2.1.2} Redistribution of fish stocks under climate change increases the risk of dispute 39 between countries or regions (high confidence) while reduced food security and livelihood of socially 40 marginalized human populations exacerbates inequality and social unrest (medium confidence). {5.4.2.1.3}

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42 Blue carbon ecosystems, such as mangroves, salt marshes and seagrasses, can help provide a nature-43 based solution with multiple co-benefits. Some 151 countries around the world contain at least one of 44 these coastal blue carbon ecosystems and 71 countries contain all three. Carbon storage, per unit area, in 45 vegetated marine habitats can be much greater than for terrestrial habitats (high confidence). {5.5.1.1} 46 Successful implementation of measures to maintain and promote carbon storage in coastal ecosystems 47 should significantly assist several countries in reaching the net zero emissions target of the Paris Agreement 48 (high confidence). {5.5.1.1} Conservation of these habitats would also sustain the wide range of ecosystem 49 services they provide and assist with climate adaptation through improving critical habitats for biodiversity, 50 enhancing local fisheries production, and protecting coastal communities from sea level rise and extreme 51 weather events (*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 CO_2 removal methods are smaller or currently have low feasibility currently. $\{5.5.1.1.4\}$
- 54

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

Chapter 5

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 alkalinisation (enhanced weathering), would also have contested governance, with risk of undesirable non-climatic consequences. {5.5.1.2}

9 Given strong carbon emission mitigation, management of other non-climatic human stressors on 10 ocean ecosystems and improving the resilience and resistance of built infrastructure offers cost-11 effective opportunities for climate risk reduction for the ocean (high confidence). Managing existing 12 non-climatic human drivers such as overfishing, habitat degradation, pollution, demographic changes and 13 poverty can substantially reduce climate risk on ocean ecosystems and dependent human communities with 14 large co-benefits in supporting sustainable development (high confidence). {5.5.2.2} Built-infrastructure 15 adaptation are optimally by integration with natural infrastructure and ecosystem-based approaches (high 16 *confidence*). {5.5.2} The effectiveness of adaptation approaches becomes low under high greenhouse gas 17 emission scenario (*high confidence*). {5.5.2.5} Integrated adaptation planning, with non-climate change 18 related co-impacts, remains an under-achieved ambition, especially in developing countries (high 19 *confidence*). {5.5.2.1, 5.5.2.4}

20 21 The effectiveness of coastal ecosystem-based adaptation is supported by a growing body of literature 22 and increased international funding for this approach. Adaptation through ecological engineering such 23 as restoration of natural habitats can support biodiversity and enhances ecosystem services that could 24 provide support livelihood, economic benefits and help build resilience in vulnerable dependent 25 communities to climate change (high confidence). {5.5.2.1} Ecosystem-based approaches are more effective 26 if supported with other adaptation approaches like community-based adaptation, adaptation planning within 27 frameworks and supported by good governance and effective implementation (*high confidence*). {5.5.2.1, 28 5.5.2.2}

29 30 Existing international and national legislation and related processes are not sufficient to provide an 31 effective adaptation response to reduce risk of climate change in the marine system (high confidence). 32 Adjustment of legal tools and techniques of international conventions such as the United Nations Convention 33 on the Law of the Sea (UNCLOS) could facilitate more effective responses to challenges created by climate 34 change for the Ocean. {5.5.4} However, such adjustment is heavily dependent on further development, 35 modification and implementation of detailed regulations e.g., fisheries management or marine spatial 36 planning by relevant international, regional and national institutions. Climate risks can also be further 37 reduced by shortening the lead time before implementation of these adjustments relative to the time of 38 emergence of climate stressors and their impacts in the marine system (medium confidence). {Box 5.1, 39 5.5.4.1} 40

5.1 Introduction

1

2 3 The ocean is a key component of the Earth System. It stores the heat trapped in the atmosphere resulting 4 from rising greenhouse gases, it stores carbon dioxide in its ocean depths and it masks and slows surface 5 warming of the Earth. It is the home to the largest continuous ecosystem, providing habitats for rich marine 6 biodiversity, and is an essential source of food and livelihood that contributes to the world's food, human 7 health and security. The ocean also supports other essential services to humans through, for example, climate 8 regulation, tourism, renewable energy, aesthetic appeal and sustaining local and traditional knowledge. 9 Governance in the ocean also has a different set of approaches, challenges and opportunities from other 10 systems that requires special treatment particularly under a changing climate.

11 12 The IPCC Fifth Assessment (AR5) reports from Working Group I (WG1) have shown from the available 13 research that there are ongoing changes to the physical and chemical state of the ocean. AR5 WG1 report 14 (IPCC, 2013) concluded that (1) 'ocean warming dominates the increased energy stored in the climate 15 system with more than 90% the energy accumulated since 1971'; (2) 'the ocean has absorbed about 30% of 16 the emitted anthropogenic carbon causing ocean acidification' and (3) it is 'extremely likely that human 17 influence has been the dominant cause of warming since mid 20th century'. AR5 WGI Chapter 3 18 summarizes observed changes in the oceans (Rhein et al., 2013), Chapter 10 addresses the detection and 19 attribution of human influence (Bindoff et al., 2013) and Chapter 12 provides projections of the physical 20 system (Collins et al., 2013).

21 22 The Working Group II of IPCC AR5 shows that marine ecosystems are changing with implications for 23 human dependent society through changes in ecosystem services, with Chapter 5 covering coastal 24 ecosystems (Wong et al., 2014), Chapter 6 covering ocean processes (Pörtner et al., 2014), Chapter 30 25 (Hoegh-Guldberg et al., 2014) covering the regional differences between ocean basins, and Chapters 14-20 26 covering general aspects related to adaptation and impacts, risks, vulnerabilities, and opportunities of 27 different societal sectors that are also applicable to the ocean. These assessments concluded that ocean 28 warming, acidification and deoxygenation are affecting marine life from molecular processes to organisms 29 and ecosystems, with major impacts on the use of marine systems by human societies. Specifically, IPCC 30 AR5 WGII highlights that 'marine species have shifted their ranges, seasonal activities, migration patterns, 31 abundance, and demonstrated altering of species interactions in response to ongoing climate change' (high 32 confidence, Chapter 6 and 30). AR5 WGII also concludes that 'future climate change will cause species 33 richness to increase at mid and high latitudes (high confidence) and to decrease at tropical latitudes (medium 34 confidence), resulting in global redistribution of catch potential for fishes and invertebrates, and implications 35 for food security (medium confidence)'. Also, 'ocean acidification poses risks to ecosystems, especially 36 sensitive ones such as coral reefs (medium to high confidence)'.

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38 This report updates these earlier assessments with new evidence on the changing ocean and marine 39 environment. Unlike previous assessment reports, this ocean focused chapter undertakes an integrated 40 assessment of the coupled human-natural marine system, with the current research and knowledge of how 41 changes in physical, chemical and biological properties of the ecosystems interact with risks and 42 vulnerabilities of the dependent communities under climate change. We also specifically assess methods and 43 approaches that have been taken to reduce risk and the governance options that might mitigate the risks of 44 anthropogenic climate change. Radiation management techniques (also known as sunlight reflection 45 methods) are excluded here. Such approaches are direct geo-engineering that are addressed in the IPCC 46 Special Report on Global Warming of 1.5°C (SR1.5) and are outside the scope of the current report. 47 However, two mitigation measures are considered, first by maintaining carbon stores in the marine 48 environment and secondly the potential to enhance carbon uptake are included (Section 5.5.1) and compared 49 with the efficacy of natural marine systems.

50

51 The chapter design is structured around a set of guiding questions. What are the key changes in the physical 52 and biogeochemical properties of the ocean and major habitats and their taxonomic groups (Section 5.2 and

- 53 5.3)? How have these changes impacted key ecosystems, risks to ecosystems services and human wellbeing
- 54 (Section 5.4)? And finally, how effective are coastal blue carbon and open/deep ocean initiatives for
- 55 mitigating carbon emissions and providing pathways for adaptation for marine dependent communities

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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

14 15 The open ocean extends from a water depth of 200 metres into the deeper regions and is the subject of this 16 section. The world's coastal and shelf seas (extending from the coasts to depths of 200 metres) are discussed 17 in Section 5.3. The assessment of the open ocean includes the changing physical and chemical properties 18 (Section 5.2.2) and their impacts on the pelagic ecosystems (Section 5.2.3). The physical and chemical 19 changes include the observed and projected changes from the available climate models for two scenarios 20 (Representative Concentration Pathway or RCP2.6 and RCP8.5) to 2100. The drivers of changes in the 21 pelagic ecosystems are the main focus of Section 5.2.2. The section includes changing temperature and 22 circulation, oxygen, ocean acidification, nutrients and net primary productivity. The pelagic ecosystems are 23 further sub-divided into epipelagic (water column that is exposed sunlight, Section 5.2.3.1), the deep pelagic 24 (water column that is not exposed to sunlight, Section 5.2.3.2) and the deep sea floor ecosystems (5.2.4). The 25 open ocean has perhaps the clearest signals of climate change in the observations of the physical 26 environment and these have been attributed to rising greenhouse gases. This relative simplicity contrasts 27 with the complexity of the coastal and shelf seas. 28

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).

40 41 The impacts on ocean ecosystems and human societies are primarily driven by regional trends and by the 42 local manifestation of the global-scale changes. At these smaller scales, the temperature, salinity, nutrient 43 and oxygen concentrations in the ocean are also expected to exhibit well-understood basin and local-scale 44 changes. However, the ocean also has significant natural variability at local and basin scales with timescales 45 from days to decades and longer (Rhein et al., 2013). This variability can mask the underlying observed and 46 projected trends (see Box 5.1). The impact of multiple stressors on marine ecosystems, which can compound 47 risks, is also less well understood. There is new evidence and understanding about multiple stressors which 48 has accumulated since the last assessment report (e.g., Gunderson et al., 2016). The most severe impacts of a 49 changing climate will typically be experienced when conditions are driven outside of the range of previous 50 experience at rates that are faster than human or ecological systems can adapt (Pörtner et al., 2014; Box 5.1). 51

52 This section summarizes our emerging understanding of the primary changes to the ocean, along with an 53 assessment of several key areas of scientific uncertainty about the changes in the ocean. Because many of 54 these long-term trends have already been extensively discussed in previous assessments (IPCC, 2013), much 55 of this summary of the physical changes will be brief except where there are significant new findings. The 56 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

6 7 The ocean is warming as a direct result of anthropogenic changes to the radiative properties of the 8 atmosphere and the heat budget of the Earth (very likely) (Bindoff et al., 2013). Over each of the past few 9 decades, our ocean observing system has documented an increase in ocean temperature (Figure 5.1) that 10 corresponds with an uptake of about 93% of the excess heat accumulated in the Earth system over this time 11 (Bindoff et al., 2013; Rhein et al., 2013). The ocean heat uptake of 274 ZJ³ during 1970-2010 period is equivalent to an energy imbalance of 0.42 W m⁻² (with respect to the Earth surface area $5.10 \times 10^{14} \text{ m}^2$) (Rhein 12 13 et al., 2013). Thermal expansion has contributed about half of the observed global mean sea level rise during 14 this period (Church et al., 2013; Table, 13.1).

15 16 To understand the recent and future climate, we use coupled ocean-atmosphere models with the full time 17 history of atmospheric forcing (greenhouse gases, aerosols, solar radiation and volcanic eruptions) for the 18 historical period and projections of the concentrations or emissions of these forcing agents to 2100. For these 19 projections the RCPs of atmospheric emissions scenarios are used as specified by the Climate Model Inter-20 comparison Project, Phase 5 (CMIP5) (see Section 1.8.2.3, and also IPCC AR5). This chapter focuses on the 21 low and high emissions scenarios RCP2.6 and RCP8.5, respectively. When these scenarios are used to drive 22 climate models, it is possible to simulate the recent and future patterns of changes in the ocean temperature, 23 salinity and circulation (and other oceanic properties such as ocean oxygen concentration and ocean 24 acidification, Section 5.2.2.3 and 5.2.2.4). The direct comparison of the simulated historical changes with 25 observations is undertaken to detect climate change, to attribute the causes of climate change to the forcings 26 in the system, and to establish the credibility of the model. Attribution studies also reject competing 27 hypotheses to explain the global ocean changes such as natural forcing such as solar variability and volcanic 28 eruptions (refer to Chapter 1) Bindoff et al. (2013). Finally, the projections of ocean changes also inform the 29 detection, attribution and projection of risks and impacts of ecosystems (Sections 5.2.3 and 5.3.4) and human 30 wellbeing (Section 5.4) under climate change.

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³ FOOTNOTE: ZJ is Zettajoule and is equal to 10²¹ joules.

Chapter 5



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1\\2\\3\\4\\5\\6\\7\\8\\9\\10\\11\\12\\13\\14\end{array}$

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.

1 The globally integrated ocean heat content changes from reanalysis of ocean observations and as projected 2 with an ensemble of CMIP5 climate models⁴ are shown in Figure 5.1a. These two estimates are consistent in 3 the historical period, once the limitations of the historical ocean observing network and the internally 4 generated variability with a single realization of the real world are taken into account (see Section 5.2.2.2). 5 The ensemble of climate models project that under the RCP2.6, the ocean will *likely* take up an additional 6 1500 ZJ of heat (Figure 5.1), or about 3 times the roughly 500 ZJ that the oceans have already taken up since 7 1900 (Roemmich et al., 2012). With the RCP8.5 scenario, the oceans will very likely take up about twice as 8 much more heat as RCP2.6 (Figure. 5.1). Even for a low carbon emissions pathway of RCP2.6 there is only 9 a 66% chance of avoiding 2°C of surface air temperature warming relative to preindustrial times and the 10 ocean will continue to warm for several centuries to come (Collins et al., 2013). It is virtually certain that the 11 ocean will continue to take up heat throughout the 21st century, at a rate that depends upon which radiative 12 forcing scenario we collectively choose to follow. 13 14 The regional patterns of the ocean temperature and salinity changes (Figures 5.1 and 5.2) associated with

15 climate change reflect both increased net downward heat fluxes due to the evolving atmospheric composition 16 and related changes in the net fresh-water forcing from the atmosphere, land and cryosphere, as carried by 17 the ocean circulation. These patterns also reflect changes in the ocean circulation itself, driven by changes in 18 wind, heat, and fresh water forcing of the ocean. The ocean circulation plays a major role in redistributing 19 heat laterally and in regulating the exchange between the surface and ocean interior. While ocean circulation 20 plays a critical role in shaping large-scale distributions of temperature and salinity, most of the changes in 21 ocean circulation are relatively modest, and the most obvious signatures of climate change in the ocean are 22 in changing temperature and salinity distributions. 23

24 The ensemble average of the CMIP5 climate models projects widespread ocean warming over the coming 25 century, concentrated in the upper ocean (Figures 5.1f and 5.2) (Kuhlbrodt and Gregory, 2012). The 26 anthropogenic heat will penetrate into the ocean following well-established circulation pathways (Jones et 27 al., 2016). The greatest vertically integrated heat uptake occurs where there is already the formation of 28 interior waters, such as Antarctic Intermediate Water along the Antarctic Circumpolar Current (Frölicher et 29 al., 2015) or North Atlantic Deep Water in the Nordic Seas (Figure 5.1f), but all watermasses⁵ that are 30 subducted over decades are expected to experience significant warming (see Figure 5.2). The warming in the 31 subtropical gyres penetrates deeper into the ocean than other gyres (roughly 15 to 45°N and 15 to 45°S in 32 Figure 5.2), following the wind-driven bowing down of the density surfaces (the solid lines in Figure 5.2) in 33 these gyres (Terada and Minobe, 2018). The greater warming at 700-2000 m in the Atlantic than the Pacific 34 or Indian Oceans (Figure 5.1g) reflects the strong southward transport of recently formed North Atlantic 35 Deep Water at these depths by the Atlantic Meridional Overturning Circulation (AMOC). Two areas that 36 commonly exhibit substantially reduced near-surface warming over the course of the 21st century are the 37 northern North Atlantic, where a slowing AMOC (see chapter 6) reduces the northward heat transport and 38 brings the surface temperatures closer to what is found in other ocean basins at these latitudes (Collins et al., 39 2013), and the southern side of the Southern Ocean, where water upwells that has been submerged for so 40 long that it has not yet experienced significant anthropogenic climate change (Armour et al., 2016). Most of 41 these projected warming patterns are broadly consistent across the current and previous generations of 42 climate models (Mitchell et al., 1995; Collins et al., 2014) as well as observations and understood 43 theoretically. These evidences give *high confidence* that the projections describe the changes the real world 44 will experience (high agreement, robust evidence).

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⁴ 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 ACCESS1.0, BCC-CSM1-1, CCSM4, 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 1981to 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

(Temperature, In-situ Density, Salinity). Shaded areas show where the projected changes are not statistically significant at the 95% level. Solid lines show present contours of these fields; the curious structure in the global-zonal mean contours of density and salinity are due to the relatively salty Mediterranean and fresh Black seas.

4 5 6 The near surface salinity of the ocean is both observed and projected to evolve in ways that reflect the 7 increased intensity of the Earth's hydrologic cycle (Durack, 2015). Generally speaking, the ocean surface in 8 areas of the ocean that currently have net evaporation are expected to become saltier, while areas with net 9 precipitation are expected to get fresher (Rhein et al., 2013), as the current patterns of precipitation and 10 evaporation are generally expected to be amplified (Held and Soden, 2006). At longer timescales of decades 11 to centuries, the larger scale changes in the ocean circulation and basin-integrated fresh water imbalances 12 emerge in the near-surface salinity changes, as shown in Figure 5.2, with an increasingly salty tropical and 13 subtropical Atlantic and Mediterranean contrasting with a freshening Pacific and polar Arctic emerging as 14 robust signals across the suite of climate models (Collins et al., 2013). The freshening of the high latitudes in 15 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 16 17 (discussed in Chapter 3).

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19 Deeper in the ocean, salinity changes reflect changes in the rates of formation of water-masses or their newly 20 formed properties. Due to their various origins, ocean water-masses tend to be relatively fresh (like Antarctic 21 Bottom Water, Antarctic Intermediate Water or North Atlantic Intermediate water) or salty (like 22 Mediterranean Water, Red Sea Overflow Water, or North Atlantic Deep Water) (Figure 5.2). The dominant 23 ocean heat uptake by the southern hemisphere in the early 21st century is expected to become more balanced 24 between the hemispheres as the asymmetric cooling by aerosols is reduced (Shi et al., 2018). Over time, as 25 these water-masses are transported away from their sources in marginal seas or at the ocean's surface, these 26 distinctive properties are diluted by mixing with ambient interior ocean waters. Changes in the interior ocean 27 salinities can reflect either changes in the rates of formation of these water-masses or changes in the 28 properties of their source (Purich et al., 2018). Thus, projected freshening of the Southern Ocean surface 29 leads to a freshening of the Antarctic Intermediate Water that is formed and subducted here, flowing 30 northward from the Southern Ocean as a relatively fresh water-mass at depths of 500-1500 m (Figure 5.2). 31 Increased surface salinity in the Atlantic subtropical gyres are pumped into the interior by the winds, leading 32 to an increased salinity of the interior subtropical gyres, along with contributions from increasingly salty 33 Mediterranean water (Jordà et al., 2017). Conversely, fresh-water capping of the northwestern North Atlantic 34 inhibits deep convection in the Labrador Sea and the consequent production of Labrador Sea Water in some 35 models (Collins et al., 2014), and contributes to the increased salinity of the North Atlantic between 1000 m 36 and 2000 m depths by the reduction of the signal from a relatively fresh water-mass (Figure 5.2). 37

38 The ocean circulation at the end of the 21st century is expected to be qualitatively similar to that today. The 39 projections suggest that some major ocean current transports will exhibit a modest increase (such as the 40 Kuroshio Extension) (Terada and Minobe, 2018) or a small decrease (such as for the Indonesian 41 Throughflow) (Sen Gupta et al., 2016); many current transports are expected to exhibit smaller than 20% 42 changes by 2100 with RCP8.5 forcing. The heat transported by the AMOC, in particular, contributes to the 43 relatively clement climate of northern Europe and the North Atlantic Basin as a whole, although the wind-44 driven ocean gyres also contribute to the meridional ocean heat transport (see the review by Buckley and 45 Marshall (2015)). As a result, there is a concern that significant changes in ocean circulation could lead to 46 localized climate changes that are much larger than the global mean. Projected and observed changes in the 47 AMOC and the rates of formation of deep water-masses in the North Atlantic are discussed in Chapter 6, 48 along with the possibility of abrupt or enduring changes as a result of forcing from Greenland. A significant 49 reduction in the Atlantic Meridional Overturning circulation would, in turn, modestly weaken the Gulf 50 Stream transport, which also has a substantial wind driven component (Frajka-Williams et al., 2016). The 51 Antarctic Circumpolar Current is projected to be subject to strengthening westerly winds and substantially 52 reduced rates of bottom water formation, as discussed in the Cross-Chapter Box 5 in Chapter 3. Climate-53 change induced changes of the circulation in other mid-latitude basins may be difficult to detect or reliably 54 project because of significant natural variability at inter-annual (e.g., El Niño) to decadal (e.g., the Pacific 55 Decadal Oscillation) timescales. Most aspects of the large-scale wind-driven ocean circulation are very likely 56 to be qualitatively similar to the circulation today, with only modest changes in transports and current 57 locations.

1 2 Mesoscale eddies (geostrophic rotating vortices with spatial scales of 10-100 km that penetrate deeply into 3 the water column, and are often described as the ocean's weather, analogous to the midlatitude atmosphere's 4 high and low pressure systems) play an important role in regulating the changes to the larger scale ocean 5 circulation, especially in the Antarctic Circumpolar current, as is discussed more fully in the Box CC-6 Southern Ocean. In addition, sub-mesoscale eddies (surface intensified, rotationally influenced motions with 7 smaller horizontal scales of 100s of m to about 10 km and intrinsic timescales of a few days that especially 8 arise in association with fronts in the ocean's surface properties) are known to be particularly important in 9 the dynamics of the near-surface ocean boundary layer (see the review by Mahadevan (2016)). Sub-10 mesoscale instabilities are associated with restratifying overturning circulations that can limit the thickness 11 of the well-mixed ocean surface boundary layer near fronts (Bachman et al., 2017). Moreover, submesoscale 12 motions generate strong vertical velocities that drive fluxes of nutrients from the interior ocean into the 13 euphotic zone or create pockets of reduced mixing with increased phytoplankton residency time within the 14 euphotic zone (Lévy et al., 2012), so that the sub-mesoscale structures are often readily visible in images of 15 ocean colour. Intense mesoscale eddies are known to create favourable conditions for submesoscale 16 instabilities both observationally (Bachman et al., 2017) and in numerical studies (Brannigan et al., 2017). 17 Intensifying Southern Ocean eddy fields will *likely* have a significant impact on biological productivity,

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19 20 The exact mechanisms driving the abyssal ocean variability (below 4000 m) are still poorly understood 21 owing to limited spatial and temporal coverage of oceanic datasets and significant differences between 22 various climate models, but climate models consistently predict a climate-change induced long-term abyssal 23 warming trend originating in the Southern Ocean due to a reduction in the formation rates of cold Antarctic 24 Bottom Water (Heuzé et al., 2015). Although the abyssal modes of natural variability are not as pronounced 25 as closer to the surface, deep ocean heat content can vary on relatively short time scales through the 26 communication of topographic and planetary waves driven by changes in the rate of deep water formation at 27 high latitudes (Kawase, 1987; Masuda et al., 2010; Spence et al., 2017). Both North Atlantic Deep water 28 (NADW) and Antarctic Bottom Water (AABW) have shown variability in production rates over the past half 29 century. The observed southern intensified global deep warming may be driven by a slow down in AABW 30 formation rates owing to freshening of shelf waters, changes in local winds driving cross shelf mixing, or 31 larger scale dynamics controlling the spin up or down of Southern Ocean gyres influencing the density of 32 outflowing waters over deep sills. It may also reflect changes in deep Southern Ocean convection. The 33 Weddell Polynya is a large opening in the wintertime ice of the Weddell Sea that is kept ice-free despite 34 intense cooling of the ocean by convective mixing bringing up warm and salty water from the deep ocean. 35 (See Box 3.2 for a more extensive discussion of polynyas and the Weddell Polynya in particular.) The 36 Weddell Polynya was present in the first three years of infrared satellite observations of wintertime sea-ice 37 concentrations in the early 1970s, but it has been closed since 1974, only to reopen for the first time in 38 decades in 2016. The prominent Weddell Polynya in the early 1970s greatly increased the volume of the 39 coldest waters in the deep Weddell Sea. Weddell Polynyas are documented to drive abyssal cold and salty 40 signals and can spread /thermal signals as waves further and faster than could be explained by slow 41 advective signals (Martin et al., 2015; Zanowski and Hallberg, 2017); these waves do not directly heat 42 individual water parces, but instead warm the ocean where they cause the coldest deep layers to spread 43 laterally and thin. However, recovery from the large Weddell polynya of the early 1970s can only explain 44 about 20% of the observed abyssal warming trend (Zanowski et al., 2015).

ecosystem structure, and carbon uptake, both directly and via submesoscale processes.

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46 The ocean's properties are changing most rapidly in the near surface waters that are more immediately 47 exposed to atmospheric forcing. As a result of the surface-intensified warming, the upper few hundred 48 meters of the ocean are becoming more stably stratified (Helm et al., 2011; Talley et al., 2016b). The 49 combination of surface intensified warming and near-surface freshening at high latitudes leading to a 50 projection of more intense near-surface stratification (the downward-increasing vertical gradient of density 51 after effects of compressibility have been corrected for) across all ocean basins (Figure 5.2) is a robust result 52 with a high agreement across successive generations of coupled climate models (Capotondi et al., 2012; 53 Bopp et al., 2013). By contrast, the bottom intensified warming in the abyss (see Figure 5.3) which is 54 consistent with a slowing in the rate of bottom water formation, is also associated with a reduction in the 55 abyssal stratification of the ocean (Johnson et al., 2014; Desbruyères et al., 2016b). Both of these changes 56 have consequences for the evolving turbulence and ocean water-mass structure. Based on observational 57 evidence, theoretical understanding and robust climate model projections, it is very likely that stratification in

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11 12 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.

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Figure 5.3: Observed rates of warming from 1981 to 2018 (left panel) as a function of depth globally (orange) and south of the Sub-Antarctic Front or about 55S (purple) with 90% confidence intervals and (right panel) average warming rate (colors) in the abyss (below 4000 m) over various ocean basins (whose boundaries are shown in grey lines), with stippling indicating basins with no significant changes.

13 The dynamical consequences of increased stratification are understood with very high confidence (see, for 14 instance, Gill (1982) or Vallis (2017)). The projected amount of increase (or decrease) in stratification will 15 vary by region, with the time-horizon in question, between forcing scenarios, and to some extent across 16 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 18 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).

31 5.2.2.2.2 Observing and attributing anthropogenic climate changes in the ocean

32 Identifying the patterns of anthropogenic climate changes in oceanic observations is complicated by the 33 presence of basin-scale natural variability with a range of timescales, from tidal to multi-decadal, and due to 34 the difficulties associated with maintaining high-precision observing systems spanning the ocean basins. 35 Historically, scientific research expeditions starting in the 19th century have provided occasional sections 36 measuring deep ocean properties (Roemmich et al., 2012). Greater spatial and temporal coverage of 37 temperatures down to about 700 m from expendable bathythermographs (XBTs) along commercial shipping 38 tracks started in the 1970s (Abraham et al., 2013). Since the early 2000s, thousands of autonomous profiling 39 floats (Argo floats) have provided high-quality temperature and salinity profiles of the upper 2000 m of ice-40 free regions of the ocean (Abraham et al., 2013; Riser et al., 2016). Further advances in autonomous floats 41 have been developed that now allow these floats to operate in seasonally ice-covered oceans (Wong and 42 Riser, 2011: Wong and Riser, 2013), and more recently to profile the entire depth of the water column down 43 to 4000 or 6000 m (Johnson et al., 2015; Zilberman, 2017) and to include biogeochemical properties 44 (Johnson et al., 2017). Autonomous floats have revolutionized our sampling and accuracy of the global 45 ocean temperature and salinity records and increased certainty and confidence in global estimates of the 46 earth heat (temperature) budget, particularly since 2004 (Von Schuckmann et al., 2014; Roemmich et al., 47 2015; Riser et al., 2016), as demonstrated by the convergence of various observational estimates of the

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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.

3 4 Observationally based estimates of global ocean heat content changes prior to the Argo era are complicated 5 by systematic instrumental errors and limited data coverage. Since AR5, there have been improvements in 6 our ability to understand and correct for XBT instrumental errors. New estimates also attempt to avoid a 7 tendency to minimize temperature changes arising from traditional data-void filling strategies that effectively 8 revert to the mean in the absence of observations. Supplementing *in-situ* observations with satellite or using 9 model-based covariances of oceanic variables in time and space, have been found to provide more accurate 10 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 11 12 during the 1971-2010 period (Rhein et al., 2013) (Figure 5.1). However, the three more recent independent 13 estimates that do a better job of accounting for instrumental biases and the sparsity of interior ocean 14 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 15 rapid ocean heat uptake in recent decades, with estimates of between 8.9 ZJ yr⁻¹ and 9.5 ZJ yr⁻¹ ocean heat 16 17 uptake in the upper 2000 m from 1991 to 2017. 18

19 These updated estimates of ocean heat uptake are consistent with CMIP5 ensemble of climate models (see 20 the inset panel in Figure 5.1a). While the collection of the worlds' climate models have been criticized for 21 having an ensemble mean that does not exhibit the observed 'hiatus' or 'slowdown' of global mean surface 22 temperature increase in the early 21st century (Meehl et al., 2011; Trenberth et al., 2016), it is increasingly 23 clear that this is at least in part due to the redistribution of heat within the climate system from the surface 24 into the interior ocean and between ocean basins. Individual realizations of climate models do show decades 25 with slow increases in mean surface temperature comparable to what was observed, even though these cases 26 exhibit continued interior ocean heat uptake, and every ensemble member exhibits surface warming closer to 27 the ensemble-mean over multidecadal timescales (Meehl et al., 2011; England et al., 2015; Knutson et al., 28 2016). Recent Ocean Heat Content (OHC) estimates show warming quite similar to the average of CMIP5 29 models, both during the 1971-2010 period highlighted in AR5, and over the 2004-present Argo era (Figure 30 5.1a). OHC changes in CMIP5 models closely match observational estimates over the past 50 years; the 31 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 32 the upper 2000m from 1971 to 2010 compared to recent observations ranging from 5.8 ZJ yr⁻¹ to 6.3 ZJ yr⁻¹ 33 (Cheng and Chen, 2017; Ishii et al., 2017). This overall consistency between observationally-based estimates 34 of ocean heat uptake and climate model simulations of the historical period gives greater confidence in the 35 accuracy of the climate model projections for changes in the coming century. 36

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).

44 Large scale patterns of natural variability at interannual to decadal time scales can mask the long-term 45 warming trend in the upper 700 m (Figure 5.1b), particularly in the tropical Pacific and Indian Ocean 46 (England et al., 2014; Liu et al., 2016). The most significant upper 700 m warming occurred in a large 47 extratropical band of the Southern Hemisphere between 30° S and 60° S mostly centred around 40° S, in the 48 tropical and northern Indian Ocean, and in the Northern Pacific. Warming of the extra-tropical Southern 49 Ocean is mainly concentrated in the southern Indian and Pacific subtropical gyres. Marginal seas, such as the 50 Mediterranean and Red seas have also exhibited notable warming. Conversely, over the past decade there 51 have also been regions of cooling in the upper 700 m, notably in the North Atlantic around 40° N-60° N. In 52 the North Atlantic, extremely cold and relatively fresh surface and subsurface conditions have persisted 53 since the year 2014 (Josey et al., 2015) linked both to circulation changes (Robson et al., 2014; Smeed et al., 54 2014; Yeager et al., 2015) and anomalous atmospheric conditions (McCarthy et al., 2015; Josey et al., 2018), 55 which together have caused a cooling pattern in regional ocean heat content trends (Figure 5.1b). This robust 56 cooling of the North Atlantic is consistent with a the persistent reduced intensity and heat transport of the 57 North Atlantic Current (Smeed et al., 2018) that very likely contributes to record low densities in the deep

1 Labrador Sea due to deep ocean warming since 1995 and the role played by long-term freshening (Robson et 2 al., 2014), and may have contributed to a subsequent intensification of deep convection in the Labrador Sea 3 since 2012 (Yashayaev and Loder, 2017). Cooling trends over the past decade in the upper 700 m also found 4 in the south-western Indian Ocean (near Madagascar) and within most of the northern subtropics (10° N and 5 30° N) and eastern basins of the Pacific Ocean. Observations show that most of the global ocean heat uptake 6 over the past decade has been stored in the southern hemisphere subtropical gyres. These southern 7 subtropical gyres are driven, in part, by the surface winds, which have intensified in recent decades, 8 facilitating the penetration of heat to deeper depths (Gao et al., 2018). The ensemble of freely running 9 CMIP5 climate model simulations also shows that internal variability can be expected to mask the long-term 10 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 11 12 significant at the 95% confidence level compared with natural variability in only a limited part of the ocean 13 basins. Averaged over the whole 21st century, by contrast, the ensemble of CMIP5 models show statistically 14 regional significant upper 700 m heat content trends almost everywhere (Figure 5.1f). 15

At depths of 700–2000 m, observations in all of the ocean basins show broadly warming trends in the wellobserved 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)).

23 24 Despite the patches of cooling water in the upper 700m described above, every one of the world's ocean 25 basins volume averaged over depths of 0 m to 2000 m has experienced significant warming over the last 26 decade (Figure 5.1 b and c, Figure 5.2, and also Desbruyères et al. (2016a)). The greatest warming has been 27 observed in the Southern Ocean, the tropical and southern subtropical Pacific Ocean, and the 28 tropical/subtropical Atlantic Ocean (Figure 5.2). Indeed, the Southern Hemisphere extratropical ocean 29 accounted for 67%-98% of the total ocean heat increase for the period of 2006 to 2013 (Roemmich et al., 30 2015). The warming of the ocean in the instrumental record since 1970 has continued unabated (Figure 31 5.1a). For the recent period (2004 to 2014) analyses are all in agreement and show that the oceans have 32 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), 33 and thus are in the same range as reported by the IPCC AR5 WGI (Rhein et al., 2013). 34

35 There are many papers that describe the observed change in ocean temperature and ocean salinity, and it is 36 clear that the ocean state has changed (Abraham et al., 2013; Ishii et al., 2017). The main difficulty of the 37 ocean temperature (and salinity) measurements in the global oceans is the presence of internal variability. 38 Internal variability in short records and on regional scales can swamp the underlying climate change signal. 39 As can be seen in Figure 5.1 (Panels d and e) the historical periods 2003 to 2016 are very smooth and have a 40 relative different pattern of change compared with the corresponding observed changes (Figure 5.1 Panels b 41 and c). Detection and Attribution studies take the internal variability into account and separate the 42 underlying climate signals with the same spatio-temporal sampling as the observations, and apply a range of 43 statistical tests to determine the coherence of the observations with the co-sampled observations (Bindoff et 44 al., 2013; AR5 WG1 Box 10.1).

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46 Since AR5, the use of different and updated oceanographic data sets and increase in the number of 47 ensembles of the CMIP5 simulations has improved the overall detection and attribution of human influence. 48 The newer CMIP5 simulations have better and more complete climate forcing (over earlier simulations) and 49 there are a greater number of ensembles. Together these measures increase the coherence of the simulations 50 and reduce noise. For example, an isotherm approach used to reduce the noise from the displacement of 51 isotherms in the upper water column allowing detection in each of the mid-latitude ocean basins was 52 achieved on 60 year time series (Weller et al., 2016). Using all the available ocean temperature and salinity 53 profiles the Southern Ocean showed that the warming and freshening patterns were consistent primarily with 54 increased human induced greenhouse gases and secondarily from ozone depletion in the stratosphere, but 55 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.

7 The global ocean below 2000 m has warmed significantly between the 1980s and 2010s (Figure 5.3), 8 contributing to ocean heat uptake and through thermal expansion to sea level rise (Purkey and Johnson, 9 2010; Desbruyères et al., 2016b). The observed deep warming rate varies regionally and by depth reflecting 10 differences in the waters influencing particular regions. The deep and abyssal North Atlantic, fed by North 11 Atlantic Deep Water (NADW), has reversed from warming to cooling over the past decade, possibly 12 associated with the North Atlantic Oscillation (e.g., Yashayaev, 2007; Desbruyères et al., 2014) or longer-13 term weakening in North Atlantic overturning circulation (Caesar et al., 2018; Thornalley et al., 2018). The 14 strongest warming is observed in regions of the deep ocean Antarctic bottom water (AABW) (Purkey Sarah 15 et al., 2014). Regions of the ocean fed by AABW from the Weddell Sea have exhibited a possible slowdown 16 in local AABW warming rates (Johnson et al., 2014), while the Pacific, fed by AABW from the shelves 17 along the Ross and Adelie Coast, has continued to warm at an accelerating rate between 1990 and 2018 18 (Desbruyères et al., 2016b). 19

20 To date, assessment of deep ocean (below 2000 m) heat content has mostly been from ship-based data 21 collected along decadal repeats of oceanographic transects (Figure 5.2b) (Talley et al., 2016b). While 22 relatively sparse in space and time compared to the upper ocean, these transects were positioned to optimize 23 sampling of most deep ocean basins and provide the highest quality of salinity, temperature and pressure 24 data. Argo floats capable of sampling to 6000 m have just started to populate select deep ocean basins. At 25 present this Deep Argo data is still in its infancy and has not yet been incorporated into the deep ocean 26 warming estimates, but are near-certain to significantly advance our understanding and ability to monitor the 27 deep ocean warming on smaller spatial and temporal scales in years to come, once decadal information is 28 available from the full global array (Johnson et al., 2015). The development of the deep Argo program, 29 complemented by indirect estimates from space (Llovel et al., 2014; Von Schuckmann et al., 2014), will 30 strongly reduce the current large uncertainties of deep ocean heat content change estimates in the future. 31

32 Although climate models are able to capture the dynamics of the climate system, all numerical models have 33 approximations and biases. The most commonly used type of ocean component in climate models is known 34 to exhibit numerically induced diapycnal mixing that can be a significant fraction of the physical mixing 35 (Ilicak et al., 2012; Megann, 2018). Because so many ocean models exhibit the same sign of bias, there is a 36 systematic warming of the lower-main thermocline is not cancelled out when taking the average over the 37 ensemble of all the models in CMIP5. These biases are widely known within the ocean modelling 38 community, and various groups are working to reduce these biases in future climate models with better 39 ocean model numerics and parameterizations. To correct for any linear consequences of model biases, 40 climate model projections are always taken as the difference from a control run without the anomalous 41 forcing. However, some aspects of the ocean response to climate change are nonlinear, and model biases can 42 introduce uncertainties into climate projections. In the case of heat uptake, this is of the order of 10% 43 uncertainty, while for the rate of steric sea level rise (which depends on the nonlinear equation of state of 44 seawater) the uncertainty in CMIP5 models is of the order of 20% (Ilicak et al., 2012).

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46 *5.2.2.2.3 Tides in a changing climate*

47 Both human structures and ecological systems in the coastal zone are directly impacted by tidal amplitudes, 48 which contribute to high-water levels. The tides are the response of a forced-damped-resonance system 49 (Arbic et al., 2009). The M_2 tide is the dominant tidal constituent in most places, with a period of $\frac{1}{2}$ lunar 50 day, or 12 hours, 25 minutes; the M₂ tides are created by the differential motion of the solid Earth and oceans 51 in response to the gravitational attraction of the moon (Newton, 1687; Laplace, 1799). The astronomical 52 forcing evolves only slowly, however the tidal damping and basin resonance at tidal frequencies can change 53 in response to climate change. Tides are well documented to have changed dramatically over geological 54 timescales, with much larger amplitude global-scale tides during the last glacial maximum (e.g., Arbic et al., 55 2004; Egbert et al., 2004; Arbic et al., 2008; Griffiths and Peltier, 2009) when shelf areas were greatly 56 reduced by the lower sea level, and tidally driven mixing was also more intense, amplifying the overturning 57 circulation (Schmittner et al., 2015). The tides have also been changing on more recent time scales. For

1 example, Colosi and Munk (2006) found an increase in the amplitude of the principal lunar semidiurnal tide 2 M_2 in Honolulu of about 1 cm over the past 100 years, which they attributed primarily to changes in oceanic 3 stratification that bring about changes in the internal M₂ tide phase relative to the external M₂ tide, increasing 4 constructive interference. Several recent studies have analyzed historical coastal tide gauge data and found 5 amplitude trends of order 1-4% per century (Flick et al., 2003; Ray, 2006; Jay, 2009; Ray, 2009; 6 Woodworth, 2010; Müller et al., 2011). In some locations, the changes in the tides have been of comparable 7 importance to changes in mean sea level for explaining changes in high water levels (Jay, 2009). For many 8 individual tide gauges, the trends in tidal amplitude are strongly positively or negatively correlated with local 9 time-mean sea level trends (Devlin et al., 2017). Müller et al. (2011) model the response of tides to changes 10 in water column thickness arising from sea level rise and glacial isostatic adjustment; the water column 11 thickness changes explain some of the tidal changes, but by no means all of it. Another source of secular 12 tidal changes, changes in oceanic stratification, modifies the rate of energy conversion from the barotropic 13 tides to the internal tides (Jayne and St. Laurent, 2001) the vertical profile of turbulent viscosity on shelves 14 (Müller, 2012), and the propagation speed of the internal tides (Zhao, 2016). Both sea level and stratification 15 are expected to exhibit robust secular positive trends in the coming century due to climate change, at rates 16 that are significantly larger than historical trends. As a result, it is very likely that majority of coastal regions 17 will experience statistically significant changes in tidal amplitudes over the course of the 21st century. 18

19 Because coastal tides are near resonance in many locations, small changes in sea level and bay shape can 20 change the local tides significantly. For example, the insertion of tidal power plants can have a significant 21 impact on the local tides (Ward et al., 2012). Various observational and modeling studies demonstrate that 22 sea level rise has spatial heterogeneous impacts on the tides, with some locations experiencing decreased 23 tidal amplitudes and others experiencing increased tidal amplitudes (Pickering et al., 2012; Devlin et al., 24 2017). Such tidal changes have implications for designing flood defenses, for tidal renewable energy, for 25 tidal flushing timescales of estuaries and embayments, and for navigational dredging requirements 26 (Pickering et al., 2012). Pelling et al. (2013) demonstrate that the rapid coastline changes in China's Bohai 27 Sea have altered the tides in that region and throughout the Yellow Sea (Hwang et al., 2014). Pelling and 28 Green (2014) examine the impact of flood defenses as well as sea level rise on tides on the European Shelf. 29 Local changes to tides are *very likely* to result in measureable changes in many places from climate drivers 30 (listed above) as well as from direct human coastal adaptation measures.

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32 5.2.2.2.4 Ocean turbulence in a changing climate

33 Ocean turbulent mixing is a key process regulating the ocean circulation and climate. Turbulent mixing is 34 important for the uptake and redistribution of heat, carbon, nutrients, oxygen and other tracers (properties 35 that are carried along with the flow of water) in the ocean. Due to the importance of turbulent mixing on the 36 lateral and vertical distribution of ocean properties, it is anticipated that ocean mixing changes can have 37 significant impacts on marine ecosystems and on the rates of heat and carbon uptake by the ocean 38 (Schmittner et al., 2009; MacKinnon et al., 2017). It has become increasingly clear that turbulent mixing in 39 the ocean is not constant in space or time. Global estimates of both the turbulent kinetic energy dissipation 40 rate and the diapycnal diffusivity, two measures of ocean turbulence, vary over several orders of magnitude 41 throughout the ocean (Figure 5.4) (Polzin et al., 1997; Waterman et al., 2012; Whalen et al., 2012; Alford et 42 al., 2013; Hummels et al., 2013; Sheen et al., 2013; Waterhouse et al., 2014; Kunze, 2017). Turbulent 43 mixing both dissipates energy and diffuses tracers across isopycnals (Fischer et al., 2013). For a given 44 energy dissipation rate, the turbulent diffusivities of heat, salinity, nutrients and other tracers tend to be 45 smaller with stronger stratification. This dependency on stratification helps explain why the observationally 46 inferred diffusivity in the heavily stratified main thermocline (250 m-1000 m depth) is of similar magnitude 47 to those deeper in the water column, while the turbulent energy density and dissipation rate are much 48 stronger at the shallower depths. Consistent global patterns in the variability of turbulent mixing are found 49 using a range of techniques, including microstructure measurements, tracer release experiments, and indirect 50 inference techniques, reflecting global patterns of the underlying physical processes that produce turbulence 51 (Ledwell et al., 2010; Waterhouse et al., 2014; Kunze, 2017). Turbulence also fluctuates in time, is 52 modulated by tidal cycles (Klymak et al., 2008) and seasonal changes (Sloyan et al., 2010; Whalen et al., 53 2012). In the mixed layer and directly below, turbulence changes according to local conditions, such as the 54 winds, heating rates and local stratification (Sloyan et al., 2010; Moum et al., 2013; D'Asaro, 2014; Tanaka 55 et al., 2015) at diurnal to seasonal and longer timescales. These variations in near-surface turbulence must be 56 taken into account for climate models to reproduce the observed seasonal cycle of surface properties and 57 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 26 most of the Southern Ocean and a more intense mesoscale eddy field (Hogg et al., 2015). These intensified 27 eddies will energize the internal wave field where they flow over rough bathymetry, in turn increasing 28 topographically generated mixing in the Southern Ocean (Melet et al., 2015). Changes in the near-bottom 29 stratification will alter the rate that the barotropic tides generate internal waves, thereby altering the strength 30 and distribution of the tidally generated mixing. Given the sparsity of the ocean observing system and the 31 large spatial and temporal variability of ocean mixing, climate-change related changes in the overall levels 32 and patterns of ocean turbulence are only just now beginning to be documented observationally (Dosser and 33 Rainville, 2016).

34

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 Section 5.2.2.2 as a *very likely* consequence of surface-intensified climate change are *likely* to lead to a net

flux of nutrients into the euphotic zone and increasing the gradient in oxygen concentrations between the 2 near surface ocean and the interior. Dramatic seasonal reductions in Arctic sea ice are virtually certain to 3 increase wave-driven mixing in the upper Arctic Ocean and at rough topography surrounding the Arctic Ocean basin.

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6 Interactions between ocean turbulent mixing and other parts of the climate system can play a significant role 7 in amplifying climate changes and must be taken into account in projecting climate changes. For example, 8 changes in internal waves in the Arctic Ocean can in turn drive mixing that warms the surface by drawing 9 upon large subsurface reservoirs of warmer, saltier water, either locally or over rough topography at the 10 basin margins (Carmack et al., 2015; Rippeth et al., 2015; Meyer et al., 2017); this positive feedback 11 mechanism can contribute to rapid Arctic Ocean warming, loss of multiple year ice and increased periods of 12 ice-free conditions. These changes impact societies around the Arctic basin (as discussed in Chapter 3 13 Sections 3.5 and 3.3, Box 3.3) and trigger further climate feedbacks due to changes in the ocean circulation 14 and Earth's albedo. Climate-induced changes to winds, tides and ocean stratification have the potential to 15 change ocean turbulence leading to potentially significant impacts on the marine ecosystem and the oceans 16 ability to mitigate/moderate climate change. 17

18 5.2.2.3 Changes in Ocean Carbon 19

20 The concentration of dissolved inorganic carbon (DIC) in the sea changes in response to (i) the exchange of 21 CO_2 across the air-sea interface, (ii) consumption of seawater CO_2 during organic matter production via 22 photosynthesis, (iii) CO₂ production during the respiration of ocean organic matter, (iv) the formation of 23 mineral forms of calcium carbonate $(CaCO_3)$ by marine biota, (v) the (mostly abiotic) dissolution of these 24 carbonates, and (vi) ocean transport and mixing (Sarmiento and Gruber, 2002). These processes and their 25 interactions lead to a complex spatio-temporal pattern of variability in DIC in the sea and since these 26 processes are of 'natural' orgin, one often refers to their impact as the 'natural' carbon cycle (Sarmiento and 27 Gruber. 2002: McNeil and Matear. 2013). 28

29 The human-induced increase in atmospheric CO₂ perturbs the natural carbon cycle through three means: 30 First, it adds additional, i.e., anthropogenic CO₂ to the ocean via enhanced air-sea gas exchange. This 31 anthropogenic CO₂ accumulates first in the surface ocean and is then transported to depth by ocean currents 32 and mixing. This is the primary process that governs the evolution of the net global uptake of CO_2 by the 33 ocean (Sarmiento et al., 1992). Second, newly accumulated CO₂ alters the CO₂ chemistry in the ocean 34 (Zeebe, 2012) causing 'ocean acidification' (see below). Third, the global climate change induced by the 35 increase in the atmospheric concentration of CO₂ and other greenhouse gases alters main ocean carbon cycle 36 processes in response to the changing ocean temperature, circulation and biological processes, with potential 37 feedbacks back to atmospheric CO₂ (Joos et al., 1999; Ciais et al., 2013). 38

39 Since AR5, new global-scale data synthesis products, novel methods for their analyses, as well as progress in 40 modeling have substantially increased our quantitative understanding of each of the above processes, 41 improving also the confidence levels. The most important progress concerns the data-based quantification of 42 the temporal variability of the ocean carbon sink. While AR5 assessed primarily the climatological mean 43 processes governing the ocean carbon cycle, the most recent work now permits us to assess how these 44 processes have changed in recent decades in response to climate variability and change. Here we focus 45 specifically on the open ocean carbon cycle.

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47 The analyses of the steadily growing number of surface ocean CO₂ observations (now more than 20 million 48 observations, SOCATv6 (https://www.socat.info/index.php/2018/06/19/v6-release/) demonstrate that the net 49 ocean uptake of CO_2 from the atmosphere has increased from around 1.2 ± 0.5 Pg C yr⁻¹ in the early 1980s to 50 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 51 52 for, these new observations imply that the global ocean uptake of anthropogenic CO_2 increased from 1.7 ± 0.5 53 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 54 et al., 2016; Le Quéré et al., 2018). This increase is supported by the current generation of ocean carbon 55 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 $25\pm5\%$ of the global emissions of CO₂ since the mid 1980s from the burning of fossil fuels, cement production, and land use change.

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6 Alongside the global scale picture, these newly available surface ocean observations also reveal a substantial 7 degree of variability at inter annual and decadal scales (Rödenbeck et al., 2015; Landschützer et al., 2016; Le 8 Quéré et al., 2018). Most notable are the air-sea CO₂ flux variations in the tropics linked to ENSO variations 9 (Rödenbeck et al., 2015; Landschützer et al., 2016), as well as the strong decadal variations in the high 10 latitudes, especially the Southern ocean (Landschützer et al., 2015; Munro et al., 2015; Ritter et al., 2017). 11 Although the data scarcity in the Southern Ocean remains a challenge for identifying such decadal variations 12 and trends (Fay et al., 2014; McKinley et al., 2017), the analysis of multiple data sets using different 13 methods support the existence of this variations (see also chapter 3). Fluctuations in the Southern Ocean CO_2 14 flux are important as they impart a substantial imprint also on the global uptake fluxes with reduced 15 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 16 17 primarily caused by a southward shift and intensification of the westerly winds over the Southern Ocean that 18 enhanced the outgassing of natural CO2, the subsequent strengthening of the sink from 2000 onwards 19 appears to arise from a combination of changes in wind, temperature, and circulation. The causes for the 20 decadal changes in the high latitudes of the northern hemisphere are less clear (Landschützer et al., 2016). 21 The majority of ocean carbon cycle models do not capture this decadal variability (Le Quéré et al., 2018), for 22 reasons presently not well understood. This uncertainty precludes a formal attribution study at this stage, but 23 the majority of the analyses suggest that decadal scale fluctuations in ocean-atmosphere CO₂ fluxes are more 24 likely the result of natural (unforced) variations in the climate system and to lesser degree a result of human-25 induced climate change (McKinley et al., 2017). That said, the weakening trend of the Southern Ocean sink 26 in the 1990s was consistent with the existence of the stratospheric ozone hole, which helped to drive the 27 southward shift of the westerly wind belt over the Southern Ocean (Gillett and Thompson, 2003). 28

29 There is growing evidence that the ocean carbon sink exhibits decadal variability at regional scales that 30 significantly alter the globally integrated sink (*medium confidence*), but there is limited understanding of the 31 role of specific driving processes.

33 Substantial progress since AR5 was also made with regard to the determination of the oceanic uptake and 34 storage of anthropogenic CO_2 within the ocean's interior. This is largely a result of the continuing efforts to 35 re-measure DIC along many of the repeat hydrographic lines that were occupied during the 1980s and 1990 36 (Talley et al., 2016a), alongside the preparation of a global quality-controlled database of ocean interior 37 observations (Olsen et al., 2016a). Several studies analyzed the changes in the amount of anthropogenic CO₂ 38 that have accumulated between different occupations in the different ocean basins (Wanninkhof et al., 2010; 39 Pérez et al., 2013; Woosley et al., 2016; Carter et al., 2017), confirming that the anthropogenic CO₂ taken up 40 from the atmosphere is transported to depth, where most of it is stored. Using a newly developed 41 reconstruction method, Gruber et al. (2018) extended these results to the globe. They find that between 1994 42 and 2007 the global ocean has accumulated an additional $32 \pm 5 \text{ Pg C}$ of anthropogenic CO₂ taken up from 43 the atmosphere, bringing the total inventory for the year 2007 to 150±20 Pg C. Extrapolating this estimate to 44 the year 2010 gives an inventory of 158±18 Pg C, which is statistically indistinguishable from the 'best' 45 estimate provided by Khatiwala et al. (2013) of 155±31 Pg C and more recently also found from a steady-46 state ocean model (DeVries, 2014). This estimate also confirms the model-based estimate of AR5, which 47 suggested an ocean inventory of anthropogenic CO₂ in the year 2011 of 155 Pg \pm 31 Pg C (AR5, chapter 6).

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49 Thus, there is *high confidence* that over the period 1750 until 2011, about 29±5% of the global

50 anthropogenic emissions have accumulated in the ocean. The consistency between surface ocean

51 observations and the ocean interior data-based reconstructions provide *robust evidence* that this uptake 52 fraction has not changed in a statistically significant manner in the last few decades and remains consistent 53 with AR5.

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55 Detailed analyses of the spatial structure of the change in storage of anthropogenic CO_2 confirm the variable 56 nature of the ocean carbon sink suggested by the surface observations. This is the case especially in the 57 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 CO₂ 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).

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Figure 5.5: Vertical sections of the change in anthropogenic CO₂ 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 μ mol kg⁻¹. From Gruber et al. (2018).

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 latitides, 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 Tipputra, 25 2018), or the long-term evolution, i.e., extending the simulations beyond 2100 (Randerson et al., 2015). 26 Although the evidence is still rather limited, the existing results suggest that global warming will lead to a 27 sustained reduction in the global ocean uptake of CO₂ from the atmosphere, and that this carbon-climate 28 feedback might be larger than that for the land. 29

30 The oceanic uptake of anthropogenic carbon not only increases the ocean's pool of DIC, but alters also its 31 chemistry. The DIC pool is made up of three species of inorganic carbon: (i) bicarbonate (HCO₃⁻), (ii) 32 carbonate (CO_3^{2-}) and (iii) aqueous carbon dioxide $(CO_{2(aq)})$, which are typically found at a ratio of 100:10:1. 33 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 $CO_{2(aq)}$ 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 (ie 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.

9 Analyses of direct measurements of ocean acidity trends from ocean time-series and derived pH changes 10 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 11 12 places and pH declines are found everywhere. Trends calculated from direct measurements from repeat 13 occupations of specific locations show a similar value of around -0.02 pH units decade⁻¹ for diverse oceanic 14 regions (Appendix 5.A, Table 1). At larger spatial scales (e.g., for broad ocean basins and regions), pH 15 trends are assessed using shipboard observations of the fugacity of CO₂ and estimates of ocean alkalinity 16 (Takahashi et al., 2014; Lauvset et al., 2015). Between 1991–2011, mean ocean pH has declined by 0.018 ± 17 $0.004 \text{ decade}^{-1}$ in 70% of ocean biomes, with the largest declines in the Indian Ocean ($-0.027 \text{ decade}^{-1}$), 18 eastern Equatorial Pacific (-0.026 decade⁻¹) and the South Pacific subtropical (-0.022 decade⁻¹) biomes, 19 with slightly lower rates of change in the Atlantic and Southern Oceans (Lauvset et al., 2015). In general, 20 trends in the stability of mineral forms of aragonite and calcite (known as the 'saturation state') follow those 21 of pH, with high latitude regions most vulnerable to under-saturation as they naturally have lower mean 22 values. In addition to long term trends that are derived by excluding the seasonal cycle, the amplitude of 23 seasonal changes in pCO₂, and hence also pH, have increased by 2.2 ± 0.4 µatm decade⁻¹ between 1982-2014, 24 which would enhance the acidification exposure to organisms (Landschützer et al., 2018). 25

26 pH trends are primarily attributed to the rising DIC levels due to the increased flux of anthropogenic carbon 27 into surface waters from the atmosphere (Lauvset et al., 2015). Observations suggest that these lowered pH 28 waters are being transferred into the ocean interior by overturning circulation in the north and south Atlantic 29 (Rios et al., 2015), as well as the North Pacific (Watanabe and Kawamiya, 2017). However, for both the 30 surface and in the ocean interior, natural variability is an important local player alongside secular changes 31 (Lauvset et al., 2015; Rios et al., 2015). For example, the seasonal cycle in carbon speciation variables (pH, 32 calcite and aragonite saturation states) is equivalent to around 10 years of the long-term trend at some sites 33 (Takahashi et al., 2014; Sutton et al., 2016). Equally, oscillations linked to natural climate variability 34 associated with El Nino, Pacific Decadal Oscillation and North Pacific Gyre Oscillation and other climate 35 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 36 37 saturation states (Lenton et al., 2012; Takahashi et al., 2014). 38

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 CO₂ 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.

45 In parallel to the ocean uptake of anthropogenic carbon, open ocean pH is projected to decline by 0.07 and 46 0.31 pH units for RCP2.6 and RCP8.5 scenarios, respectively (Bopp et al., 2013; Gattuso et al., 2015), with 47 good agreement across the range of CMIP5 models (Figure 5.6a). Projected changes in pH are quite uniform 48 with a relatively weak spatial variations and are greatest in the Arctic Ocean and the high latitudes of the 49 Atlantic and Pacific Oceans due to their lower buffering capacity and are lowest in contemporary upwelling 50 systems (Figure 5.6b). Other ocean regions show changes similar to the global average. Similar declines in 51 the concentration of $CO_3^{2^2}$ ions is predicted by the CMIP5 models (Bopp et al., 2013; Gattuso et al., 2015), 52 with high latitude and Arctic Ocean regions first to be undersaturated due to their systemic vulnerability (Orr 53 et al., 2005). Models also predict that the seasonal cycle in pH will be amplified in the future. In the ocean 54 interior, the progressive decline in pH due to ongoing anthoropogenc carbon uptake will be modulated by 55 changes to ocean overturning and water mass subduction (Resplandy et al., 2013) and may be augmented by 56 modifications to interior ocean respiration rates (Chen et al., 2017). Seafloor changes in pH over the next 57 century are highly localized and are linked to transport of surface anomalies to depth, with over 20% of the

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1 North Atlantic sea floor deeper than 500m experiencing pH reductions greater than 0.2 units by 2100 under 2 the RCP8.5 scenario, including canyons and seamounts designated as marine protected areas (Gehlen et al., 3 2014a). Changes in pH in the abyssal ocean (>3,000 m) deep, are greatest in the Atlantic and Arctic Oceans, 4 with lesser impact in the Southern and Pacific Oceans by 2100, mainly due to circulation timescales 5 (Sweetman et al., 2017). Model skill is generally good for trends in carbon speciation variables, such as pH, 6 thus increasing confidence in their projections over the coming century (Bopp et al., 2013; Frölicher et al., 7 2016) (Figure 5.6a), although persistent model biases in the simulation of seasonal cycles should be noted, 8 especially in the Southern Ocean (Mongwe et al., 2018). 9

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 the 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 relience potential for parallel changes in ocean circulation, that the Arctic and North Atlantic seafloors will experience the largest pH changes over the next century.





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6 7 changes in surface pH, upper 100 m nitrate, O_2 averaged over 100–600 m, and NPP integrated over top 100 m. Figure adapted after (Frölicher et al., 2016). The observation-based estimated for surface pH are from Lauvset et al. (2015), for sea surface temperature from Smith et al. (2008), for O_2 at 300 dbar from Stramma et al. (2011) and for NPP from Behrenfeld et al. (2006). Please note that only linear trends are given for surface pH and O_2 observations, and that reference period for observed pH, O_2 and NPP change are arbitrarily chosen.

8 Anthropogenic trends in carbon speciation variables (pH, saturation of calcite and aragonite) in the open 9 ocean emerge from the background variability by 2010 or 2020 for the tropical ocean or regions of low and 10 high latitude upwelling, respectively (Keller et al., 2014; Rodgers et al., 2015), with the detection of open 11 ocean pH trends requiring around 15 years of data is needed for the resolving of a pH trend (Henson et al., 12 2016). For aragonite saturation, the anthropogenic signal has already emerged from the background over 13 much of the ocean (Rodgers et al., 2015). By the end of century, the anthropogenic pH signal is projected to 14 have emerged over the entire surface ocean (Frölicher et al., 2016). While internal variability and model 15 uncertainty are most important for the next few decades, the largest source of uncertainty in the projection 16 for end-of-century open ocean pH changes (Figure 5.6c) at both global and local scale is the emissions 17 scenario (Frölicher et al., 2016), with large changes in the trajectory and magnitude of pH changes between 18 the RCP8.5 and RCP2.6 scenarios (Figure 5.6a and c). Time of Emergence is longer in coastal regions with 19 higher variability of the carbonate species (Sutton et al., 2018). 20

21 5.2.2.4 Changing Ocean Oxygen

22 23 Ocean oxygen (O_2) levels at the surface are controlled by the balance between oxygen production during 24 photosynthesis and temperature controlled equilibrium with the atmosphere through air-sea exchange. 25 Deeper in the water column, consumption of oxygen during respiration and redistribution by ocean 26 circulation and mixing are dominant. Due to the localisation of photosynthetic activity and relatively short 27 timescales of atmospheric equilibrium, surface waters are close to atmospheric equilibrium and relatively 28 high in oxygen. In contrast, little in situ production means that O₂ consumption during respiration and the 29 interior transport of oxygen rich surface water by ocean circulation dominate in the interior ocean. Oxygen 30 minimum zones (OMZs) arise in the interior ocean where the circulation cannot supply enough oxygen 31 enriched waters to meet the respiratory demand, especially in the shadow zones in the eastern sides of 32 tropical oceans where the time-mean advective pathways of the circulation are blocked by topography rather 33 than originating in the directly ventilated surface ocean. Changes to the extent of low oxygen regions in 34 response to climate change can promote the ocean souce of nitrous oxide (Martinez-Rey et al., 2015; 35 Battaglia and Joos, 2018), which is a potent greenhouse gas, and have occurred in the geologic past in 36 response to climate warming (Praetorius et al., 2015). 37

38 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 decade⁻¹, or by 2% in total since the 1960s 39 40 (Schmidtko et al., 2017). By region, the largest trends are seen in the Equatorial Oceans that contain the 41 lowest oxygen waters, North Pacific, Southern Ocean and South Atlantic (from 210 to 119 Tmol decade⁻¹ overall or 1-3 umol kg⁻¹ decade⁻¹ in the upper 1,200m (Figure 5.7), while the Arctic Ocean shows strong a 42 43 strong decline (Schmidtko et al., 2017). While other studies agree in the bulk trends and at high latitudes. 44 there are differences at the regional scales in the tropics related to data merging and analysis methods that 45 affects confidence locally (Helm et al., 2011; Ito et al., 2017). Syntheses of datasets from local time series 46 tend to document stronger trends, with oxygen declines of over 20% at sites in the Northeastern Pacific 47 between 1956-2006 and the California Current between 1964-2006 (Levin, 2017). Despite holding the 48 highest inventory of oxygen in the ocean, oxygen levels in Southern Ocean contributed 25% to the global 49 decline between 1970–1992 (Helm et al., 2011) and have fallen by over 150 Tmol decade⁻¹ from the 1960s 50 to present (Schmidtko et al., 2017). Observations along ocean cruises as part of the CLIVAR programme 51 have also documented broad thermocline oxygen declines in the northern hemisphere oceans, accompanied 52 by oxygen increases in subtropical and southern hemispheres (Talley et al., 2016b). 53

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 Chapter 5

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).

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Figure 5.7: 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.

13 Oxygen is also declining in the deepest parts of the ocean, where a global oxygen decline of 70 Tmol yr⁻¹ 14 below 1,200 m has been determined. By region, the strongest declines in deep ocea oxygen occur in the 15 Arctic, North Pacific and Southern Oceans, while the North Atlantic has experienced a moderate oxygen 16 increase below 1,200 m (Figure 5.7). Regional changes in the deep ocean are attributed to changes in large 17 scale ocean circulation that affects the ventilation of the deep ocean by Pacific, Atlantic and Antarctic deep 18 waters (Watanabe et al., 2003; Stendardo and Gruber, 2012). Indeed, globally, the observed oxygen decline 19 is negatively correlated with ocean heat content changes (Ito et al., 2017). Changes to respiration rates, either 20 due to temperature enhancement or in the amount/quality of organic material can also be important and the 21 enhanced respiratory demand associated with an intensified Monsoon has been invoked as a driver of the 22 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*.

29 Ocean oxygen changes are affected by climate variability on interannual and decadal timescales, especially 30 for the tropical ocean OMZs (Deutsch et al., 2011). ENSO variability affects the heat content and respiration 31 rates in the tropical Pacific, which then regulates oxygen solubility and respiratory demand (Ito and Deutsch, 32 2013; Eddebbar et al., 2017). These drivers may then be combined with modifications to overturning and 33 and ventilation of OMZs by lateral jets (Duteil et al., 2012). Centennial scale studies based on isotope 34 proxies for low oxygen regions have demonstrated fluxtuations in OMZ extent linked to decadal changes in 35 tropical trade winds that affects interior ocean respiratory oxygen demand, suggesting that it is difficult to 36 attribute recent changes in the Pacific OMZ to anthropogenic forcing alone (Deutsch et al., 2015). Parallel 37 work based on oxygen observations (Czeschel et al., 2012), as well as modelling (Duteil et al., 2018) 38 supports the importance of decadal scale variability in the eastern tropical Pacific OMZ. There is some 39 evidence for the potential of a modulating impact on tropical Pacific oxygen from atmospheric deposition of 40 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.

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7 Future changes in oxygen can be appraised from CMIP5 models that account for the combined effects of 8 ocean physics and biogeochemistry. Globally, oxygen is predicted to decline by 3.45±0.44% across the 9 range of CMIP5 models (Bopp et al., 2013) (Figure 5.6d). Increased tropical ocean stratification reduces 10 interior ocean oxygen by diminishing pathways of ventilation in the subtropical gyres and by inhibiting 11 turbulent mixing with the oxygen-rich surface ocean (see Section 5.2.2.2.4). This relatively robust global 12 modelled trend (Figure 5.6d) however masks important uncertainties in the projection of regional trends 13 (Figure 5.6e), particularly in the tropical ocean OMZs (Bopp et al., 2013; Cocco et al., 2013; Cabré et al., 14 2015). In these regions, an overall projection of oxygen increases is more uncertain as the inter-model 15 standard deviation is over four times greater than the projected trend (Bopp et al., 2017). The uncertainty in 16 the trends in tropical ocean OMZs arises due to the fact that oxygen depletion due to warming induced 17 reductions in oxygen saturation are opposed by oxygen enrichment due to reduced oxygen consumption 18 during respiration in response to predicted declines in marine export production, as well as biases due to 19 model resultion in the tropics and the length of the model spin up (Bopp et al., 2017). At the seafloor, bathyl 20 (200-3000 m depth) regions of the North Pacific, North Atlantic, Arctic and Southern Oceans may see 21 oxygen declines by 0.3% to 3.7% by 2100 (relative to 2005), while abyssal ocean changes are lower and 22 localised around regions in the North Atlantic and Southern Ocean affected by deep water formation 23 (Sweetman et al., 2017). There is *high confidence* that the largest changes in deep sea systems will occur 24 after 2100 (Battaglia and Joos, 2018). 25

26 Accordingly, there is medium confidence in the projected 3.5% loss of total ocean oxygen by 2100, relative 27 to 2005, across CMIP5 models, due to the combination of warming and increased stratification and low 28 confidence at regional scales (especially the tropical ocean) due to inter-model variability and reduced skill 29 (Bopp et al., 2013). Simulations extended to 2300 suggest that by 2150 the trend of declining tropical ocean 30 oxygen (both in terms of concentrations and volume of low oxygen waters) may reverse itself, mainly due to 31 the effect of strong declines in primary production and organic matter fluxes to the ocean interior (Fu et al., 32 2018), but with low confidence due to limited evidence. At the global scale, 10,000 year intermediate 33 complexity model simulations find that ocean oxygen loss shows near linear relationships to equilibrium 34 temperature, itself linearily related to cumulative emissions, and any climate mitigation scenario will reduce 35 peak oxygen loss by 4.4% per degree Celsius of avoided warming (Battaglia and Joos, 2018). 36

37 A median of just over 25 years of data is required for an interior ocean oxygen trend (average concentration 38 over 200-600 m depth) to emerge from the natural background, but more than 40 years data is needed in 39 specific dynamic regions (Henson et al., 2016). By 2100, the anthropogenic signal in interior ocean oxygen 40 is projected to have emerged above uncertainties associated with internal and inter-model variability for 41 around 25% of the ocean area in the 100-600m depth range (Frölicher et al., 2016) (Box 5.1). In particular, 42 the oxygen signal in the northern subtropical Pacific is predicted to emerge by 2030 to 2040 (Frölicher et al., 43 2016; Long et al., 2016). As seen for pH, the scenario uncertainty is also an important component of the total 44 uncertainty in projection of interior ocean oxygen by 2100 (Figure 5.6f), but the contributions of internal 45 variability and model uncertainty remain important, especially at more regional scales, by 2100 (Frölicher et 46 al., 2016). At the zonal scale, robust detection of anthropogenic trends in oxygen emerge from climate 47 models much faster, due to reduced noise (Andrews et al., 2013). Overall, it is *likely* that time of emergence 48 estimates are conservative given that models tend to underestimate the low frequency decadal scale 49 variability in the tropical Pacific Ocean. 50

51 Taken as a whole, we assess that there is *high confidence* that ocean oxygen loss is occurring today and will 52 increase in the future in response to climate change. While projections at regonal and decadal scales may be 53 uncertain due to local processes and natural variability, there is a robust link between global warming and 54 ocean oxygen loss.

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5.2.2.5 Changing Ocean Nutrients and Primary Productivity

1 Light and nutrients support the base of marine food webs and nutrient levels in the surface waters of 2 different parts of the open ocean result from the interplay of external nutrient supply (e.g., from dust and 3 ocean mixing/upwelling) and internal cycling (consumption by primary producers and replenishment during 4 degradation of organic matter). Nutrients that are remineralized in the interior ocean need to be advected or 5 mixed back into the surface ocean before they can again support phytoplankton growth. The depth of ocean 6 mixing also controls the light environment that phytoplankton are exposed to, but outside of dark polar 7 regions during winter, spatial variations in phytoplankton activity reflect changes in nutrient levels. Indeed, 8 the overall production of particulate organic matter by phytoplankton primary production is regulated by 9 available nutrient resources and is an important component of ocean biogeochemical cycling and 10 atmospheric CO_2 uptake (see Section 5.2.2.3), as well as being a source of food for both pelagic and benthic 11 ecosystems. The greatest rates of phytoplankton productivity occur in regions where nutrient supply is 12 maximised by upwelling and upwards mixing of deep nutrient reserves (Falkowski, 1998) (Figure 5.8). 13 Equally, the distribution of specific forms of plankton is closely linked to the match between the relative 14 supply of different nutrients and their requirements (Hagstrom et al., 2017), which themselves can be 15 strongly sensitive to warming (Yvon-Durocher et al., 2015; Yvon-Durocher et al., 2017). For example, the 16 competitive success of nitrogen fixing plankton in the future ocean will be controlled by warming (Fu et al., 17 2016), the relative supply of iron, nitrogen and phosphorus (Ward et al., 2013) and ocean acidification 18 (Hutchins et al., 2013; Hutchins et al., 2015). 19

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.

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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.

1 Since AR5, our understanding regarding the role of different nutrient resoures in different regions has 2 grown. Diverse studies (including shipboard experiments and use of protein biomarkers) have highlighted 3 nitrogen and phosphorus limitation in the stratified tropical ocean regions accompanied by widespread iron 4 limitation at high latitudes and in upwelling regions that typically have elevated levels of productivity 5 (Figure 5.8) (Moore et al., 2013; Saito et al., 2014; Browning et al., 2017; Tagliabue et al., 2017). Moreover, 6 more extensive experimental work has demonstrated nitrogen-iron co-limitation at the boundary of the south 7 Atlantic gyre system (Figure 5.8) that is linked to the common occurrence of simultaneous drawdown of 8 nitrogen (as nitrate) and iron levels that is typical of all gyre/upwelling boundaries, driving greater microbial 9 diversity (Browning et al., 2017). These findings echo the substantial overlap in nutrient stress biomarkers 10 seen in transition between upwelling and gyre systems in the Pacific (Saito et al., 2014). 11

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*).

17 Nutrient changes in the surface ocean over the last few decades reflect modifications to physical supply by 18 mixing and upwelling, as well as external input, typically from dust and internal cycling. In the open ocean, 19 increasing stratification is occurring in the major tropical regions (Dave and Lozier, 2013; Talley et al., 20 2016b; Kwiatkowski et al., 2017; and see also Section 5.2.2.2). Increased stratification is expected to retard 21 nutrient supply to the surface by suppressing upwelling and turbulent mixing (see also Section 5.2.2.2.4). 22 Supply of iron at in the Southern Ocean is mostly regulated by winter mixing (Tagliabue et al., 2014), as 23 well as changes linked to modifications to iron supply linked to sea ice, icebergs and glacial melt. Increasing 24 inputs of anthropogenic nitrogen from the atmosphere are perturbing ocean nutrient levels (Jickells et al., 25 2017). In the western Pacific in particular, additional atmospheric nitrogen input has enriched the system in 26 nitrogen, which raises the nitrogen to phosphorus ratio and induces a progressive shift towards phosphorus 27 limitation in this region (Kim et al., 2014). Enhanced microbial turnover of dissolved organic nitrogen and 28 phosphorus may act to counter balance reduced vertical supply due to enhanced stratification, but this 29 requires microbial turnover to mainly occur in the upper ocean (Letscher et al., 2013; Letscher and Moore, 30 2015). Microbial turnover of dissolved organic nitrogen and phosphorus in the upper ocean depends 31 critically on their lability (readiness to change) (Banse, 1968; Letscher et al., 2013) and the presence of the 32 necessary metal cofactors for the hydrolysing enzymes (Mahaffey et al., 2014; Browning et al., 2017). 33

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.

37 38 Much effort has been made to extract information on the trends in primary production over the recent past 39 using satellite datasets, which have the potential to provide a global picture or change. These datesets rely on 40 mathematical algorithms to convert ocean colour into ocean chlorophyll (an index of phytoplankton 41 biomass), phytoplankton carbon or primary productivity. Globally, a range of studies have reported 42 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 43 44 change estimated globally may mask much larger changes at regional scales (Colella et al., 2016; Hammond 45 et al., 2017; Mélin et al., 2017), with recent datasets merged from different satellites reporting trends of $\pm 4\%$ 46 in different regions with increases at high latitudes and moderate decreases at low latitudes (Mélin et al., 47 2017), but corroboration with *in situ* time series is lacking. Attempts to compare satellite trends with *in situ* 48 data from open ocean time series sites have found substantial mismatches (Saba et al., 2010), but display 49 reasonable correlation in higher biomass coastal regions (Kahru et al., 2009) and, at present, satellite records 50 are not yet long enough to unambiguously isolate long term climate related trends from natural variability 51 (Beaulieu et al., 2013). Indeed, satellite derived trends associated with natural climate variability, e.g., El 52 Nino or the Indian Ocean Dipole can be substantial (Brewin et al., 2012; Currie et al., 2013; Racault et al., 53 2017). While some studies report good intercomparability across different satellite datasets (Mélin et al., 54 2017), others highlight mismatches between various satellite algorithms for primary productivity in absolute 55 values and decadal trends (Gómez-Letona et al., 2017). 56

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.

5 6 Globally, CMIP5 models project overall declines in surface ocean nitrate concentrations (Figure 5.6g) that 7 are localised in tropical upwellings, subpolar gyres and other frontal systems (Figure 5.6h). Iron 8 concentrations are projected to increase in the future, due to changes in ocean physical lateral transport and 9 reduced consumption (Misumi et al., 2013). Currently, projections from CMIP5 model simulations do not 10 account for the changing supply of nitrogen or iron from the atmosphere (e.g., accounting for anthropogenic 11 sources), which lowers confidence in these projections - especially in regions experiencing high levels of 12 atmospheric deposition. 13

14 Globally, net primary productivity trends in CMIP5 models mirror the decline in nutrients, with a decline in 15 the low latitudes and an increase at higher latitudes that results in a low global overall decline (Figure 5.6j). 16 The apparent virtual compensation in the effect of primary production is underpinned by substantial spatial 17 and inter-model heterogeneity in the projected changes (Figure 5.6k). For example, in the tropical ocean a 18 decline of $11 \pm 24\%$ in primary production is projected across the suite of CMIP5 models by 2100 under 19 RCP8.5, with lesser changes under more moderate scenarios (Laufkötter et al., 2015). However, this large 20 inter-model range can be restricted to \pm 6% if emergent constraints from the historical record that link 21 variability productivity to temperature anomalies are used, a four-fold decline in inter-model uncertainty 22 (Kwiatkowski et al., 2017). Via this emergent constraint approach, there is a projected decline in global 23 primary production of $6 \pm 3\%$ (Kwiatkowski et al., 2017). Projected declines in primary production in 24 different biomes across CMIP5 models are generally attributed to increased stratification that reduces in 25 surface nutrient concentrations in low latitude biomes (Cabré et al., 2014; Laufkötter et al., 2015; Fu et al., 26 2016) (Figure 5.8), but in some models the temperature enhancement of growth is the dominant driver 27 (Laufkötter et al., 2015). Interacting controls by iron and light drive the response of primary productivity in 28 the Southern Ocean (Laufkötter et al., 2015; Leung et al., 2015). On multi-century timescales, it is projected 29 that enhanced physical-biogeochemical feedbacks act to supress biological activity for a millennium (Moore 30 et al., 2018). This arises due to the retention of essential nutrients in the ocean interior as changes to winds, 31 sea ice and ongoing warming reduce the usual resupply to the surface ocean, which exacerbates the declining 32 trend in CMIP5 model projections to 2100.

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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).

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41 Changes in organic matter flux from surface waters (known as export production) in CMIP5 models are 42 generally closely linked to changes in primary production in the surface layer. In general, CMIP5 models 43 predict declines in export production of around 1-20% from 2000-2100 for the RCP8.5 scenario (Bopp et 44 al., 2013; Fu et al., 2016; Laufkötter et al., 2016). Models with a more detailed representation of the marine 45 food web tend to predict a stronger decline in export production than total organic matter production. 46 highlighting the role of food web changes in driving changes to organic matter supply to the ocean interior 47 (Fu et al., 2016). Parallel to these declines in export production, the flux of organic material to sea floor is 48 also predicted to decline globally, with increases projected only in the polar regions (Sweetman et al., 2017). 49 The projected changes in export production can be larger than global primary production because they are 50 affected by both the magnitude of organic matter production in the surface ocean, and also how food web 51 structure modulates the 'transfer efficiency' of particulate organic material, which then affects the sinking 52 speed and lability of exported particles through the ocean interior to the sea floor (Bopp et al., 2013; Fu et 53 al., 2016; Laufkötter et al., 2016).

54 55 Confidence in model projections can be appraised using the model skill in reproducing distributions of the 56 limiting nutrient in specific ocean regions (Figure 5.8) as this will underpin projected changes in primary 57 productivity. High model skill in reproducing surface distributions of nitrate and phosphate (Laufkötter et

1 al., 2015), raises confidence in projections in nitrogen and phosphorus limited systems. In contrast, poor 2 model skill in reproducing iron dynamics (Tagliabue et al., 2016) lowers confidence in projections in iron 3 limited regions (Figure 5.8). Despite apparently high skill in the reproduction of bulk nitrate and phosphate 4 fields, skill is reduced when underlying biogeochemical and physical mechanisms (Duteil et al., 2012) or the 5 ratio of resources is assessed. Indeed, properly representing the ratio of different resources correctly has been 6 shown to have significant impacts on the response of primary productivity to environmental change (Moreno 7 et al., 2017). Model skill in reproducing patterns of primary production and export from satelitte derived 8 estimates range from poor to average (correlation coefficients of 0.1-0.6 across different models (Laufkötter 9 et al., 2016; Moreno et al., 2017)), but it should be noted that complete comprehensive observational datasets 10 do not exist for these metrics with very few in situ observations. Improving model skill in ocean 11 biogeochemical processes is important as trends in the future magnitude of changes in nutrient supply and 12 associated impacts on organic matter production by primary production are related to the magnitude of 13 model biases in contemporary ocean biogeochemistry (Fu et al., 2016). 14 15 Trends in surface nitrate, primary production and export production emerge from background variability

16 after a median of around 30 years, but can take >50 years in some regions (Henson et al., 2016). Using 17 CMIP5 models, it is suggested that detecting trends using ocean observing systems will be challenging 18 because they often require a long time series (>~35 years) and the results are only statistically representative 19 of a small area (Henson et al., 2016). The anthropogenic trend in primary production does not emerge from 20 background uncertainty at any geographic point due to internal and model dynamics by 2100 (Frölicher et 21 al., 2016). Unlike pH and oxygen, scenario uncertainty is not a dominant component of the uncertainty in 22 future projections of nutrient levels and organic matter production, which is instead dominated by model 23 uncertainty, especially at regional scales (Figure 5.6 i and 1) (Frölicher et al., 2016). That the between model 24 uncertainty in the modelled trends in nitrate levels and productivity are larger in the uncertainty between the 25 RCP2.6 and RCP8.5 scenarios (Figure 5.6 g and j) highlights the impact of the incomplete representation of 26 these processes in climate models. 27

28 In summary, there is high confidence that future modifications to ocean temperatures, stratification and 29 circulation will affect nutrient supply to the ocean surface, with *medium confidence* in the trends in nitrogen 30 and phosphorus, but low confidence in projections of changes to iron supply. Incomplete representations of 31 the major processes in climate models and in some cases, poor model skill, undermine confidence in 32 projections. There is medium confidence (high agreement, limited evidence) from CMIP5 models that 33 changes in nutrient supply will lead to modest decline in global productivity and a larger decline in export 34 production by 2100. At regional scales, there is high confidence in strong spatial variability emerging, but 35 low confidence in the direction of change. In general, uncertainty increases at each step along the cascade 36 from changes to ocean physics, to nutrient supply, to primary production and finally to export production as 37 our understanding worsens concomitantly. 38

40 [START BOX5.1 HERE]

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

43 44 The concept of time of emergence (ToE) is defined as the time at which the *signal* of climate change in a 45 given variable emerges from a measure of the background variability or *noise* (AR5 WGI 11.3.2.1, SROCC 46 Glossary). In associating a calendar date with the detection, attribution and projection of climate trends, the 47 concept of a ToE has proved useful for policy and planning particularly through informing important 48 climatic thresholds and the uncertainties associated with past and future climate change (Hawkins and 49 Sutton, 2012). However, there is not a single agreed metric of ToE and the ToE for a given variable thus 50 depends on choices regarding the space and time scale, the threshold at which emergence is defined and the 51 reference period (AR5 WGI 11.3.2.1). Recently, the ToE concept has been expanded to include variables 52 that describe ocean ecosystems, considering essential variables such as pH, carbonate ion concentrations, 53 aragonite and calcite saturation states, nutrient levels and marine primary productivity (Box 5.1, Figure 1) 54 (Ilyina et al., 2009; Friedrich et al., 2012; Keller et al., 2014; Lovenduski et al., 2015; Rodgers et al., 2015).

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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

1 (Stocker et al., 2013) is that organisms respond to climate hazards that emerge locally, rather than to the 2 global and basin-scale averages reported in IPCC AR5 WGI. As the noise from internal variability is greater 3 at local scales than for bulk averaged quantities, emergence is much later. Moreover, variability is also much 4 greater in the coastal ocean than for the open ocean considered here, which will be important for discussing 5 hazard exposure for coastal species. For example, although signals of anthropogenic influences have already 6 emerged from internal variability in the late 20th century for global and basin-scale averaged ocean surface 7 and sub-surface temperature (very likely) (AR5 WGI SPM), their ToE and level of confidence vary greatly at 8 local scales and in coastal seas (Frölicher et al., 2014). Pelagic organisms with small range size may thus be 9 more (or less) at risk to warming with earlier (or later) ToE at the scale of the area that they inhabit. 10

11 ToE assessments typically quantify the internal variability using the standard deviation of the detrended data 12 over a given time period (Keller et al., 2014; Rodgers et al., 2015; Henson et al., 2016; Henson et al., 2017), 13 the scenario and model uncertainty associated with different climate scenarios and across available climate 14 models (Frölicher et al., 2016), and in some cases the autocorrelation of noise (Weatherhead et al., 1998). As 15 more components of 'noise' are accounted for, the ToE lengthens. Here we show the local ToE for five key 16 ocean condition variables (Box 5.1, Figure 1). Calculated using the preindustrial period and presented 17 relative to the period 1986-2005 as representative of the observation period, this accounts for internal and 18 model variability (Frölicher et al., 2016). Interior ocean oxygen depletion emerges from local background 19 variability only in the north Pacific subtropical gyre, Arabian Sea, southeastern Indian Ocean, north and 20 central eastern Atlantic Ocean, whereas nitrate and net primary productivity will not generally emerge, 21 except at some local regions, until the next century. From an observational standpoint, similar analyses that 22 account for autocorrelation of noise suggest time series of around a decade are sufficient to detect a trend in 23 pH or SST, whereas datasets spanning 30 years or longer are typically needed for detection of emergence at 24 local scales for oxygen, nitrate and primary productivity (Henson et al., 2016).

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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 (O_2) , (D) surface nitrate (NO_3) , 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).

35 The rapidity of change, encompassed in the ToE, can be linked to concepts of exposure to hazard and 36 vulnerability of biota. As organisms have evolved to be adaptable to natural variations in the environmental 37 conditions of their habitats, changes to their habitat conditions larger than that typically experienced or 38 specific biological thresholds such as upper temperature tolerance may become hazardous (Mora et al., 39 2013). Thus, the ToE of biological response is dependent on the species' biology and biogeography as 40 illustrated by the differences in ToE of habitat suitability between Skipjack tuna (larger home range with 41 narrower temperature tolerance) and Atlantic cod (small range size with wider temperaturel tolerance) (Box 42 5.1, Figure 2). Generally, large proportion of the habitats of both species will experience ToE with the 21st

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century under RCP8.5. Also, Atlantic cod may experience ToE of the change in their habitat suitability approximately a decade earlier than that of skipjack tuna (Box 5.1, Figure 2).

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4 Species with fast generation times relative to the ToE of key habitat conditions (e.g., phytoplankton) may
5 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

8 facilitate adaptation to environmental change, it does not necessarily result in successful adaptation of

9 organisms (5.2.3.1). Moreover, there are species-specific thresholds of changes in ocean variables of

- 10 organisms' habitats above which large biological impacts occur, e.g., warming threshold for coral bleaching
- 11 (Pendleton et al., 2016) and temperature and oxygen threshold for fishes such as Atlantic cod and tunas
- 12 (Deutsch et al., 2015; Box 5.1, Figure 2). These thresholds increase the biological sensitivity to ToE and

13 shorten the time of organismal response relative to ToE of the respective ocean variables. 14



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 morhus*). (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).

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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).

19 As was discussed in AR5 WGII (Pörtner et al., 2014) and assessed in Section 5.2.2.5, short time series of 20 two decades or less in length are too short for determining trends in the ocean (Henson et al., 2010; Chavez 21 et al., 2011), and longer time-series are needed to establish the relative influences of climate change and 22 variability and other human non-climatic influences (Edwards et al., 2013). To overcome this drawback, 23 multiple lines of evidence -ranging from experiments to models and field observations- are combined for the 24 detection and attribution of biological changes, future projections and risk assessments. This section will 25 follow the subdivision of the pelagic system into the epipelagic ocean (<200 m, the uppermost part of the 26 ocean that receives enough sunlight to allow photosynthesis), and the deep pelagic ocean, containing the 27 'mesopelagic' (200–1000 m depth) and 'bathypelagic' (>1000 m depth) zones. 28

29 5.2.3.1 The Epipelagic Ocean30

31 Since the AR5, there have been a wide range of advances in understanding the effect of climate change on 32 the surface, epipelagic ecosystem, across observations, experimental and modelling studies. The surface 33 ocean is much more exposed to climatic and non-climatic influences from human than other open ocean 34 ecosystems due to its close interaction with the lower atmosphere and the impact of human activities. The 35 following assessments include the interactions between multiple climate and non-climate stressors, instead 36 of the effects of individual drivers; the latter was solidly established in AR5. These updates from AR5 result 37 in more comprehensive understanding of the future risks of the epipelagic organisms (spanning from 38 microbes to marine mammals) and ecosystems, as well as their scope and limitation of adaptation, including 39 shifts in distribution, phenology and biodiversity. 40

41 5.2.3.1.1 Detection and attribution of biological changes in the epipelagic ocean

42 Recent evidence from mechanistic understanding, physiological experiments and observations since AR5 43 continue to support the effects of changing temperature and oxygen level on pelagic marine ectotherms 44 (organisms that are dependent on external sources of body heat). Changes in temperature beyond the thermal 45 preferences of the organism affects body functions, impacting growth and reproduction. The dominant 46 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 47 48 explaining temperature-sensitivity of some marine organisms may exist (Lefevre, 2016), the conclusion that 49 biological functions are impacted by warming beyond species' thermal preferences and tolerances remain 50 robust with high agreement of evidence. Anoxic conditions hinder marine life that obtain oxygen from the 51 ambient waters, while hypoxic conditions reduce their metabolic scope for growth, and consequently limits 52 the ability of animals to forage, avoid predation or fend off diseases (Kroeker et al., 2013). Since AR5, 53 experiments continue to support that increasing CO_2 and decreasing pH in the ocean disrupts organism's 54 vital biological processes such as growth, reproduction and calcification through increased corrosion of 55 calcium carbonate exoskeletons, elevated metabolic demand from disruption of the intra-cellular and inter-56 cellular acid-base balance, and neuro-sensory and behaviour such as foraging and predatory avoidance, with 57 the precise effects varying between species and ecosystems (Sett et al., 2014; Cattano et al., 2018).

1 2 Organisms' responses to multiple climate drivers -ocean warming, acidification, deoxygenation, and changes 3 in nutrient or food supplies- have also been observed in experimental and field studies (Boyd and Bressac, 4 2016; Breitburg et al., 2018) (Figure 5.9). Sensitivity to ocean acidification varies across species and such 5 variations may interact with the effects of warming and low oxygen directly and indirectly. Warming may 6 exacerbate the effects of ocean acidification on the rate of photosynthesis in phytoplankton (Lefevre, 2016). 7 However, the effects of the interactions between warming and ocean acidification on phytoplankton 8 production varies between the phyoplankton's climatic zones. In a recent meta-analysis of published 9 experimental studies, primary production by temperate noncalcifying plankton increases with elevated 10 temperature and CO₂, whereas tropical plankton decreases productivity because of acidification 11 (Nagelkerken and Connell, 2015). Also, temperature increases consumption by and metabolic rates of 12 herbivores but not on secondary production; the latter decreases with acidification in calcifving and 13 noncalcifying species. This effect creates a mismatch with carnivores whose metabolic and foraging costs 14 increase with temperature (Nagelkerken and Connell, 2015). Reported physiological responses to multiple 15 climatic drivers include the decrease in aerobic scope for growth as demand for metabolic oxygen increases, 16 neuroendocrine responses that affect the behaviour of the organisms, immune responses that affect the 17 organisms' defence against parasites and pathogens, disruption of iono- and osmoregulation, and changes in 18 reproductive biology (Whitney et al., 2013). reducing development of larvae of some sea urchins 19 (echinoderms) (Ferrari Maud et al., 2014), or increasing predation rate of reef fishes (Nagelkerken and 20 Munday, 2016) and behaviour in general (Nagelkerken and Munday, 2016), although interactions vary 21 strongly between species and biological processes (Gobler and Baumann, 2016; Lefevre, 2016) (medium 22 confidence). Hypoxia and acidification can also limit the thermal reaction norm of organisms and exacerbate 23 their sensitivity to warming (Mackenzie et al., 2014; Pörtner et al., 2017). Limitation of nutrient and food 24 availability and predation pressures can further increase the sensitivity of organismal groups to climate 25 change in specific ecosystems (Riebesell et al., 2017). Climate change also affect organisms indirectly 26 through the impacts on competitiveness between organisms that favour those that are more adaptive to the 27 changing environmental conditions (Alguero-Muniz et al., 2017) and trophic interactions (Seebacher et al., 28 2014). 29

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*).

37 New evidence since AR5 continues to support that shifts in distribution range and phenology from 38 phytoplankton, zooplankton to fishes are consistent with expectations from physiological understanding of 39 the organisms' temperature preferences and changes in environmental temperature and other oceanographic 40 conditions (Pörtner et al., 2014; Britten et al., 2016; Poloczanska et al., 2016) (Figure 5.9). Specifically, 41 spatial and temporal occurrences of marine biota follow changes in environmental conditions. Warming is 42 related to expansion of leading edge of their geographic range and contraction of trailing edge in the past 43 (Feary et al., 2014). Timing of biological events of marine biota are also shifting to earlier in the years under 44 warming, which is consistent with the expectation from the close relationship between temperature and these 45 biological events (Poloczanska et al., 2016). 46

47 Overall, the rate of historical geographic range shift in the last century, towards cooler higher latitude waters 48 and deeper layers, is partly correlated with the rate of ocean warming, the slope of their habitat's temperature 49 gradient and species' biological characteristics, but with group-specific differences (Pinsky et al., 2013; 50 Jones and Cheung, 2015; Poloczanska et al., 2016). For example, analysis of the Continuous Plankton 51 Recorder data-series from the North Atlantic show that the range of dinoflagellates and copepods tended to 52 closely track the velocity of climate change (the rate of isotherm movement). In contrast, the distribution 53 range of the diatoms shifted much more slowly (Chivers et al., 2017) and its distribution seems to be primary 54 influenced by multi-decadal variability rather than from the average temperature trend (Edwards et al., 2013; 55 Harris et al., 2014).
1 In the North Atlantic, zooplankton are advancing northwards by 11.6 ± 2.9 days per decade faster than any other marine group (Richardson, 2008; Schlueter 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).

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> c. Range shift a. Physiological impacts Fishes (N = 185) Zooplankton (N = 43) F Others **A**O °0+ **A**O shift ange Ζ X** X** Seabirds (N = 4) В В X** X*** X** X*** hift +* X*** X** X** 0 X** **Physiological level Population level** b. Change in body size d. Change in phenology on (N = 115) Ζ dat ٩ Experi E E ZB *** *** Sea turtles, seabirds, mammals (N = 66) Fishes (N = *** *** *** F 0 F O **Organism level** Ρ Phytoplankton Consistent with expectation Zooplankton Inconsistent with expectation or inconclusive **Benthic invertebrates Fishes** High confidence Medium confidence 0 Others, including marine reptiles, Low confidence mammals and seabirds

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, O_2 – ocean deoxygenation, and others – other variables mostly include salinity, nutrient and primary production.

4 5 6 7 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 8 temperature change with the temperature of the environment) are more sensitive to climate change than 9 endothermal organisms (those that generate heat to maintain their body temperature). Species that evolved in 10 fluctuating and variable environments (e.g., seasonally variable temperate environment) often become 11 eurythermal (i.e., the ability to tolerate or adjust to a wide range of temperature) while those that evolved in 12 more stable environments become stenothermal (i.e., living within a narrow temperature range) (Cheung and 13 Pauly, 2016; Portner et al., 2017). Based on existing observations and ecological theories, larger body size, 14 higher dispersal ability, availability of suitable habitats and food also contribute to the successfulness of 15 range expansion (Sunday et al., 2011). 16

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.

24 25 AR5 concluded that there had been limited field observations that robustly demonstrate biotic responses 26 attributable to anthropogenic ocean acidification, partly because of the large natural variability (Pörtner et al. 27 2014). More recent evidence since AR5 continues to suggested mixed results on whether impacts of ocean 28 acidification on calcifying zooplankton are already observable (Howes et al., 2015). Attempts were made to 29 tease apart the relative roles of regional and multi-decadal variability in the effect of ocean acidification. 30 (Rivero-Calle et al., 2015) reported, using Continuous Plankton Recorder (CPR) archives, that stocks of 31 coccolithophores (a group of phytoplankton that forms calcium carbonate plateles) over the last five decades 32 have increased by 20%, and that this increase is linked (by statistical analysis) to higher CO₂ concentrations. 33 Their results agree with other recent observational study in the subtropical North Atlantic that attributes an 34 increase from 1990 to 2014 in coccolithophore abundance to ocean acidification. However, the outcome of 35 (Rivero-Calle et al., 2015) analysis depends on whether global (Mauna Loa archive) or regional (Takahashi 36 archive) CO_2 time-series are employed in the statistical analysis, with global CO_2 and the Atlantic Multi-37 decadal Oscillation (AMO), respectively, being the main drivers attributed to this floristic shift (medium 38 confidence). 39

- 40 Overall, detectable changes in abundance of califying phytoplankton and zooplankton have observed in
 41 some regions of the epipelagic ocean (*medium confidence*), but increased ocean acidification may not have
 42 caused sufficient biological changes beyond natural variations (*medium confidence*).
 43
- 44 The differences in rate of responses warming between biota and regions result in re-organization of pelagic 45 communities, with increasing dominance of warmer water species (Stuart-Smith et al., 2015) and potential 46 mismatch of biotic interactions (Thackeray et al., 2016; Pecl et al., 2017). Distribution shifts of copepods are 47 accompanied by a decrease in their mean size, leading to a system with higher metabolism, more recycling 48 and less export (Attrill et al., 2007; Poloczanska et al., 2013). The warming-stratification hypothesis suggests 49 warming decreases of phytoplankton and zooplankton biomass in the tropical ocean because of nutrient 50 limitation associated with enhanced stratification and a shallower mixed layer, and increases biomass in 51 polar regions through greater water column stability and faster growth (Richardson and Schoeman, 2004; 52 Boyce et al., 2010; Laufkötter et al., 2013). There is some support for this hypothesis. Although there were 53 methodological challenges, a data synthesis from the tropical Atlantic showed a 10-fold drop in biomass 54 from 1950 to 2000, related to reduced phosphate concentration and thinning of the thermocline. Similarly, a 55 recent modelling study showed a 5% decline in zooplankton biomass in the past 50 years (Steinberg et al., 56 2012). However, the best available tropical data (from the Bermuda Atlantic Time Series) shows a 61% 57 increase in zooplankton biomass associated with sea surface warming and increased stratification

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(Havenhand, 2012), which is *likely* to be driven by the increase in smaller phytoplankton translating up the microbial food web to zooplankton.

3 4 As endotherms, mammals and birds are more buffered to the effects of temperature changes than reptiles, 5 which as ectotherms, rely on external heat sources to regulate their body temperature. For reptiles, including 6 lizards, turtles and snakes, temperature directly effects important life history traits including hatchling size, 7 sex, viability and performance (high confidence) (Hays et al., 2003; Pike, 2014; Dudley et al., 2016; Santos 8 et al., 2017). This is particularly important for marine turtles as changing temperatures will affect the 9 hatchling sex ratio because sex is determined by nest site temperature (high confidence) (Hatfield et al., 10 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 11 12 land breeding marine turtles, lizards, seabirds and pinnipeds (very likely) (Fish et al., 2005; Fuentes et al., 13 2010; Funayama et al., 2013; Reece et al., 2013; Katselidis et al., 2014; Patino-Martinez et al., 2014; Pike et 14 al., 2015; Reynolds et al., 2015; Marshall et al., 2017).

15 16 Changes in ocean temperature, and pH will also indirectly impact marine mammals, seabirds and reptiles by 17 changing the abundance and distribution of their prey (high confidence) (Polovina, 2005; Polovina et al., 18 2011; Doney et al., 2012; Sydeman et al., 2015; Briscoe et al., 2017; Woodworth-Jefcoats et al., 2017). 19 While indirect effects of changing climate are harder to document, examples are becoming apparent. The 20 distributions of some of these large animals is determined by the occurrence and persistence of oceanic 21 bridges and barriers that are related to climate driven processes (Ascani et al., 2016; McKeon et al., 2016). 22 For example, the decline of Arctic sea-ice is effecting the range and migration patterns of some species and 23 is allowing the exchange of species previously restricted to either the Pacific or Atlantic oceans (Virtually 24 certain) (Alter et al., 2015; George et al., 2015; Laidre et al., 2015; MacIntyre et al., 2015; McKeon et al., 25 2016; Breed et al., 2017; Hauser et al., 2017). It is hard to predict how these new invasions will change the 26 existing food webs and species interactions. For example, as killer whales, Orcinus orca, inhabit Arctic 27 waters, narwhale's, Monodon monoceros, are avoiding the use of key habitats that are already in decline to 28 avoid predation (Medium confidence) (Bost et al., 2009; Sydeman et al., 2015) (see Chapter 3). 29

30 The habitat requirements of marine mammals, seabirds and sea turtles are associated with bathymetric and 31 mesoscale features that facilitate the aggregation of their prey (Bost et al., 2015; Kavanaugh et al., 2015; 32 Hindell et al., 2016; Hunt et al., 2016; Santora et al., 2017). The persistence and location of these features are 33 linked to variations in climate (Crocker et al., 2006; Baez et al., 2011; Dugger et al., 2014; Abrahms et al., 34 2017; Youngflesh et al., 2017) and to foraging success, juvenile recruitment, breeding phenology, growth 35 rates and population stability (high confidence) (Costa et al., 2010; Ancona and Drummond, 2013; Ducklow 36 et al., 2013; Chambers et al., 2014; Descamps et al., 2015; Abadi et al., 2017; Bjorndal et al., 2017; Fluhr et 37 al., 2017; Youngflesh et al., 2017). These changes have been most pronounced in polar environments, where 38 sea-ice is an important physical feature that is required for both breeding and hauling out by birds and 39 mammals (Ducklow et al., 2006; Siniff et al., 2008; Jenouvrier et al., 2009; Costa et al., 2010; Murphy et al., 40 2013; Constable et al., 2014; Barbraud et al., 2015; George et al., 2015; MacIntyre et al., 2015; Amelineau et 41 al., 2016; Renner et al., 2016; Ferguson et al., 2017; Fernandez-Arcaya et al., 2017; Hamilton et al., 2017; 42 Hauser et al., 2017; Stempniewicz et al., 2017) but also forms a critical link in the structure of polar marine 43 habitats (Costa et al., 2010; Murphy et al., 2016) (see Chapter 3).

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45 A consistent theme in both predicted as well as observed changes in megafauna attributed to climate change 46 is that there will be winners and losers (virtually certain) (Trivelpiece et al., 2011; Hazen et al., 2013; Clucas 47 et al., 2014; Constable et al., 2014; George et al., 2015). For example, while some Adélie and Chinstrap 48 Penguin populations have plummeted, Gentoo Penguin colonies have increased (Cimino et al., 2013; 49 Ducklow et al., 2013; Korczak-Abshire et al., 2013; LaRue et al., 2013; Dunn et al., 2016; Youngflesh et al., 50 2017). Such changes are quite regional, Adélie Penguin populations at the southernmost limits of the range 51 are increasing (Trivelpiece et al., 2011), while those at the northern limits are declining (Cimino et al., 2013; 52 Ducklow et al., 2013; Trathan et al., 2015). Long term population changes associated with climate change 53 have also been observed for temperate species of seabirds and shifts in distribution of marine mammals and 54 seabirds (high confidence) (Henderson et al., 2014; Hiscock and Chilvers, 2014; Ramp et al., 2015). 55 Rockhopper, Eudyptes chrysocome, and Yellow Eyed Penguins, Megadyptes antipodes (Rey et al., 2014; 56 Morrison et al., 2015; Mattern et al., 2017), have declined, while populations of Gentoo, Pygoscelis papua,

Chapter 5

1 Larus scoresbii, and Kelp, L. dominicanus, gulls have increased, some significantly (Boersma and Rebstock, 2 2014). However, increases in Magellanic penguin populations occurred at higher latitudes colonies, whereas 3 they declined at the lower latitude colonies (Weimerskirch et al., 2012). This pattern is consistent with a 4 shifting habitat and species range associated with climate change (highly likely). Laysan, Phoebastria 5 immutabilis, and Wandering, Diomedea exulans, albatross have responded positively to climate change as 6 they have been able to take advantage of the increased intensity of winds, allowing them to forage farther 7 and faster making their foraging trips shorter increasing their foraging efficiency and breeding success 8 (Descamps et al., 2015; Thorne et al., 2016). 9

10 Climate change is also associated with an increase in the frequency and intensity of storms and other 11 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 12 13 increased risk on population viability of marine mammals and seabirds (medium confidence). An increase in 14 storms between 1983 and 2010 resulted in increased reproductive failure in Magellanic Penguins at Punta 15 Tombo, Argentina (Cavole et al., 2016). An unusual 'warm-water anomaly' off the Northeast Pacific Ocean 16 resulted in closures of commercially important fisheries and mass strandings of marine mammals and 17 seabirds (Newell et al., 2015) and a reduction in Shag, P. aristotelis, and Kittiwake, Rissa tridactyla, 18 breeding success in the North Sea (Nicoll et al., 2017). Depending on the age class of petrel and a regional 19 effect, there could be both positive and negative effects of tropical cyclones on Round Island Petrels, 20 Pterodroma arminioniana (Riofrio-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*).

33 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).

38 39 Planktonic microbes (bacteria, archaea, unicellular algae, and protozoans), with short turnover times (hours 40 to days), would adapt faster than larger animals, due to their phenotypic plasticity (Schaum et al., 2013). 41 Microbes are thus expected to adapt quickly to rapidly rising of CO₂ or warming, partly compensating for 42 the negative effects (Schaum and Collins, 2014). The response of microbes to multiple drivers is complex, 43 however, since it may involve interactive effects (i.e., synergisms and antagonisms) between changing 44 biologically influential properties (Breitberg et al., 2015; Hutchins and Boyd, 2016; O'Brien et al., 2016) 45 (medium confidence). The majority of studies suggest that reduced pH will result in a shift in the microbial 46 communities associated with invertebrate hosts, with direct impacts on host health and disease susceptibility, 47 although the magnitude of effects will vary among species (Lee et al., 2015) (medium confidence). Another 48 indirect CO₂ effect on the bacterial and phytoplankton community could be through viral lysis, which is 49 reported to respond positively to pCO₂ (Brussard et al., 2013; Crawfurd et al., 2017) (low confidence).

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51 A wide range of N_2 fixation responses of diazotrophic cyanobacteria (ranging from no change to increases or 52 decreases in N_2 fixation rates) have been recently observed in laboratory experiments under temperature and

 CO_2 conditions that are similar to RCP8.5 projected for the end of the 21st century. These variable responses

- 54 are possibly due to differences in the mechanisms of N_2 fixation between strains and species (Eichner et al.,
- 55 2014; Gradoville et al., 2014) (*medium confidence*). Moreover, nitrogen fixation by *Trichodesmium* appears
- 56 to be significantly influenced by interactions between elevated CO_2 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*).

3 4 In situ experiments with natural plankton communities revealed some consistent patterns in phytoplankton 5 responses to ocean acidification (summarised in Pörtner et al., 2014). Across a wide range of climate zones 6 and ecosystem types, autotrophic standing stocks (chlorophyll a) were consistently higher at high pCO_2 , 7 more pronounced in smaller-sized taxa, with variable responses depending on nutrient availability. 8 Haptophytes (coccolithophores and *Phaeocystis* sp.) are the only phytoplankton class known to respond 9 negatively to ocean acidification (Hoogstraten et al., 2012; Bermúdez et al., 2016; Riebesell et al., 2017). 10 Coccolithophores are vulnerable to projected end of century pH (Sett et al., 2014) (medium confidence). 11 However, new evidence, based on long-term experiments of acclimation or adaptation to increasing 12 temperatures in combination with elevated CO₂, show that growth, calcification and carbon fixation rates of 13 coccolithophores at high CO₂ are modulated by temperature, are species-specific, and could increase 14 calcification (Khanna et al., 2013). Calcification of planktonic foraminifera will be negatively affected by 15 acidification (Roy et al., 2015) (medium to high confidence), and their populations are predicted to 16 experience the greatest decrease in diversity and abundance in sub-polar and tropical areas, under RCP8.5 17 (Brussaard et al., 2013).

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19 At the level of mesozooplankton, experimental studies simulating future scenarios of warming and 20 acidification are contradictory. Some studies suggest that acidification and warming act synergistically to 21 increase the impact on zooplankton (Garzke et al., 2016), whereas other studies suggest that acidification 22 effects partially counteract some temperature impacts on abundance and body size (Mayor et al., 2015). 23 Other evidence suggests that the impact of ocean acidification on non-calcifying zooplankton could be 24 greater when predators are present and or when food is absent (Winder et al., 2017). In a recent mescosm 25 experiment to determine the variations in biological responses to climate change, larvaceans outcompeted 26 copepods in warmer and lower pH scenarios (Wirtz, 2012). Larvaceans can efficiently shunt energy to 27 higher trophic levels (Peijnenburg and Goetze, 2013; Heneghan et al., 2016) and also export considerable 28 amounts of carbon. These experimental studies suggest that changes in food webs are *likely* to be complex 29 and oversimplification of zooplankton processes and functional groups in the current, similarly-30 parameterised, biogeochemical models could lead to errors in project magnitude and direction of change in 31 zooplankton biomass (medium confidence). 32

33 Moreover, experimental studies on the impact of realistic ocean acidification scenarios for 2100 show 34 zooplankton responses are species-specific and can be both positive and negative. Many experimental 35 studies have shown no ocean acidification impact on the diversity of mesozooplankton (Li et al., 2015) early 36 life stages of copepods (Weydmann et al., 2012; McConville et al., 2013; Cripps et al., 2014; Alguero-Muniz 37 et al., 2016; Bailey et al., 2016) or schyphomedusa ephyrae (Winans and Purcell, 2010; Schulz et al., 2013). 38 However, some studies have shown a negative impact on zooplankton, including shell degradation in 39 pteropods (Alguero-Muniz et al., 2017), lower abundance of an anthomedusa jellyfish (Cripps et al., 2014), 40 increased mortality of copepod nauplii (Smith et al., 2016a), and a reduction in demersal zooplankton 41 (Alguero-Muniz et al., 2017). By contrast, some experimental studies have shown a benefit of ocean 42 acidification on zooplankton, including a positive effect on overall abundance but with differences in 43 direction of changes between species (Alguero-Muniz et al., 2017; Taucher et al., 2017). For example, in a 44 mesocosm experiment with high pCO2 treatment, Hybocodon prolifer (Anthomedusa) reacted negatively by 45 lower abundances, while Aglantha digitale (Trachymedusa) became more abundant (Havenhand, 2012).

47 A wide range of studies, from laboratory experiments (Taucher et al., 2015), mesocosm enclosures (Isla et 48 al., 2008), synthesis of observations (Rose and Caron, 2007) to modeling experiments (Bopp et al., 2013) 49 provide insights into how the multi-faceted components of the 'biological pump' (the biologically mediated 50 processes responsible for transporting carbon from the upper ocean to depth) are projected to be altered in 51 the coming decades. A synthesis of the individual components reported to both influence the performance of 52 the biological pump, and which are sensitive to changing ocean conditions, is presented in Table 5.1. They 53 include those catalogued by Pörtner et al. (2014) along with additional factors - such as seawater viscosity 54 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 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)

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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: expertiments (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 indiated the sign of the projected change on export (in italics), as opposed to magnitude. NPP: Net Pimary 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 termperature and acidification.

Pump component	Oceanic driver	Projected change (by year 2100)	Confidence	References & Lines of evidence
Epipelagic Ocean				
Phytoplankton growth	Temperature (warming)	~10% Faster (nutrient-replete) no change	High	Boyd et al. (2013) E; Marañón et al. (2014) O&
Net Primary Production (NPP)	Climate change (temperature, nutrients, CO ₂)	10-20% decrease (low latitudes); 10-20% increase (high latitudes)	Medium	Bopp et al. (2013) M
Partitioning of NPP (POC, TEP, DOC)	OA	~20% increase in TEP production (Engel, 2002)	Medium	Engel (2002) E ; Engel et al., 2014 E ; Riebesell et al. (2007) Seebah et al. (2014) E :
Food web retention of NPP	OA	Enhanced transfer of organic matter to higher trophic levels, reduced N and P sedimentation by 10%	Low	Boxhammer et al. (2018) E
Floristic shifts	Climate change (Warming/OA/Iron/ Salinity	Shift to smaller or larger cells (less export vs more export: Inconclusive)	Low	Morán et al. (2010) O ; Li et al. (2009) O ; Sett et al. (2018) I
Differential susceptibility	Temperature (warming)	Growth-rate of grazers more temperature dependent than prey	Low	Rose and Caron (2007) O
Bacterial hydrolytic effects	Warming/OA	(veriable response in different plankton communities)	Low	Burrell et al. (2017) E
Grazer physiological responses	Warming	Copepods had faster respiration and ingestion rates, but higher mortality (inconclusive)	Low	Isla et al. (2008) O
Faunistic shifts	Temperate and subpolar zooplankton species shifts	(inconclusive)	Low	Edwards et al. (2013) O
Deep Ocean				
Bacterial hydrolytic enzyme activity	Temperature	20% increase (resource-replete) to no	Low	Wohlers-Zollner et al. (2011) E; Endres et al. (2014) E; Bendtsen et al. (2015) E: Piontek et al. (2015) E*
Particle sinking rates (viscosity)	Warming	5% faster sinking/ degree C warming	Low	Taucher et al. (2014) M
Mesozooplankton community composition	Temperature?	Shifts which increase/decrease particle transformations (less/more export respectively)	Low	Jackson and Burd (2002) M; Ikeda et al. (2001) \boldsymbol{O}
Vertical migrators	Climate change (irradiance, temperature)	(more export)	Low	Almén et al. (2014) O ; Berge et al. (2014) O
Deoxygenation	Climate change	(more export)	Low	Dunne (2010) M ; Cocco et al. (2013) O Hoffman and Schellnhuber (2009) M

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 35 grazing relative to phytoplankton growth (O'Connor et al., 2009; Alguero-Muniz et al., 2017) (medium 36 confidence). 37

1 Changes in oxygen content and ocean acidification exacerbate temperature effects on fishes and 2 invertebrates. Decrease in oxygen content under warming projected to reduce maximum body size (Deutsch 3 et al., 2015; Pauly and Cheung, 2017) and contraction of suitable environmental conditions for fishes 4 (Munday, 2014). Fishes exposed to ocean acidification level expected under RCP8.5 showed impairments of 5 sensory ability and alteration of behaviour including olfaction, hearing, vision, homing and predator 6 avoidance (Kroeker et al., 2013; Heuer and Grosell, 2014; Nagelkerken et al., 2015). A wide variety of 7 biological processes, ranging from growth, survivorship, reproduction, and calcification are impacted by 8 ocean acidification expected under RCP8.5. with crustaceans and calcifying molluses being most sensitive to 9 such effects (Kroeker et al., 2013; Alava et al., 2017). Early life stages are particularly sensitivity to ocean 10 acidification impacts. Also, ocean acidification and warming have shown to have synergistic effects with 11 warming that exacerbate the risk of population decline. Overall, there is *high confidence* that warming, 12 decrease in oxygen and acidification will impact growth, reproduction and behaviour of fishes. Adaptation 13 will not always suffice to mitigate the climate impacts, although there will be winners and losers (high 14 confidence, high agreement). The risks of population decrease of marine fishes will be further exacerbated 15 by overfishing and pollution (Munday et al., 2013) (high confidence).

16 17 Based on outputs from an ensemble of marine ecosystem and fisheries models, potential total consumer 18 (mainly invertebrate and fish) biomass without fishing is projected to decrease by 4.8±3.5% and 17.2±10.7% 19 under RCP2.6 and 8.5, respectively, by 2090-2099 relative to 1990-1999, while the decrease is around 4.5% 20 by the mid-21st century across all RCPs (likely) (Figure 5.10). Accounting for the removal of biomass by 21 fishing exacerbates the decrease in biomass for large-bodied animals which are particularly sensitive 22 to fishing (likely for the direction of changes). Regionally, total consumer biomass decreases largely in 23 tropical and mid-latitude oceans (*likely*) and increases in the Arctic ocean (Figure 5.10, as likely as not). The 24 high uncertainty in the Arctic is because of model uncertainties as a result of the ways the model represent 25 the effects of ocean variables (temperature and net primary production) (Lotze et al., in review) and the 26 model outputs (exploited species only versus total biomass). In the Southern Ocean, the decrease in 27 consumer biomass is mainly in the southern Indian Ocean while other parts of the Southern Ocean are 28 projected to have an increase in consumer biomass by 2100 under RCP8.5, reflecting mainly the projected 29 pattern of changes in net primary production from the Earth system models (see Section 5.2.2.5). 30



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.

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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).

15 Responses of pelagic ectotherms to warming, ocean acidification, deoxygenation and changing nutrient 16 supplies may be moderated by physiological acclimatization (Munday, 2014), trans-generational and 17 evolutionary adaptation (Clements and Hunt, 2015; Niinemets et al., 2017), and behavioural changes 18 (Poloczanska et al., 2016). Experimental evidence demonstrate that less complex organisms are better in 19 acclimatizing and adapting to warming and deoxygenation because of their wide aerobic scope and short 20 generation time (Seebacher et al., 2014). In contrast, acclimatization to warming and deoxygenation is often 21 limited by one or more physiological processes in higher organisms with complex organization (Poloczanska 22 et al., 2016). However, more complex organisms are able to adjust for the changes in their internal acid-base 23 balance under high CO₂ in waters. Although scope for acclimatizing or adapting to climate change impacts 24 exist in marine organisms, climate change impacts will not be fully eliminated given that species have been 25 responding to changing ocean environment through shifting biogeography and phenology in recent decades 26 (Poloczanska et al., 2016) (see Figure 5.9). For example, despite zooplankton having high evolutionary 27 potential and presumably being well suited to adapt to climate change (Hinder et al., 2014), no thermal 28 adaptation in a 50-year time series of two key copepod species in the North Atlantic was found (Lewis et al., 29 2013). Some work suggests that sensitivity to ocean acidification is dependent upon the degree of natural 30 exposure, with species that vertically migrate and commonly encounter low pH conditions more able to 31 adapt to lower pH conditions (Gunderson et al., 2016). 32

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*).

39 5.2.3.2 The Deep Pelagic Ocean 40

The pelagic realm of the deep ocean represents a key site for remineralisation of organic matter and longterm biological carbon storage and burial in the biosphere (Arístegui 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).

46 Global ocean warming between 0.5°C (RCP2.6) and 1.5°C (RCP8.5) is expected to reach the bathypelagic 47 zone by the end of the century (Figure 5.5), with the largest warming of deep waters occurring first in the 48 Southern Ocean (Ciais et al., 2013; Patara and Böning Claus, 2014); Roemmich et al. (2015). There is 49 medium confidence that warming and changes to primary productivity in the upper ocean will reduce the 50 export of organic matter to the deep sea (Section 5.2.2.4). Analyses of long-term trends in primary 51 production and particle export production, as well as model simulations, reveal that increasing temperatures, 52 leading to enhanced stratification, nutrient limitation and shifts towards small phytoplankton, will have the 53 greatest influence on decreasing the flux of particulate organic carbon (POC) to the deep ocean (Bopp et al., 54 2013; Boyd et al., 2015; Fu et al., 2016; Laufkötter et al., 2016). Observational and model-based methods 55 provide evidence that the transfer efficiency of organic carbon through the mesopelagic zone is partly 56 controlled by temperature and oxygen (low confidence), affecting microbial metabolism and zooplankton 57 community structure, with highest efficiencies for high-latitude and oxygen minimum zones (OMZs) (see

1 Section 5.2.2.4) for more detail on OMZs), while below 1000 m the highest transfer efficiencies occur in 2 subtropical regions, and are controlled by particle sinking speed (Boyd et al., 2015; Marsay et al., 2015; 3 DeVries and Weber, 2017). There is also low confidence on the effects of increasing temperatures on 4 particulate organic carbon (POC) remineralisation to CO₂ versus POC solubilisation to dissolved organic 5 carbon (DOC) by microbial communities and its storage as refractory DOC (i.e., with life times of >16,000 6 years) (Legendre et al., 2015). The global magnitude of the 'active flux' of organic carbon from the surface 7 to the mesopelagic due to vertical migration of zooplankton and fishes is uncertain, but has been reported to 8 account from 10 to 100% of the gravitational sinking flux (Davison et al., 2013; Hudson et al., 2014; 9 Jónasdóttir et al., 2015) (low confidence).

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11 Predictions based on model studies suggest that mesopelagic zooplankton and fish communities living at 12 deep scattering layers (DSLs) will increase their biomass by 2100, enhancing their trophic efficiency, 13 because of ocean warming and shallowing of DSL (Proud et al., 2017) (low confidence). Recent model 14 projections agree with the earlier results from IPCC AR5 WGI Chapter 3, predicting an average 3.5% loss of 15 total ocean oxygen by 2100 and expansion of OMZs (see Section 5.2.1.4). It will also widen the DSL and 16 increase the exposure of mesopelagic organisms to shallower depths (Gilly et al., 2013; Netburn and 17 Koslow, 2015). In the California Current, the abundance of mesopelagic fishes is closely tied to variations in 18 the OMZ, whose dynamic is linked to the Pacific Decadal Oscillation (PDO) and El Niño-Southern 19 Oscillation (ENSO) cycles (Koslow et al., 2015). Hypoxia-adapted animals, like the Humboldt squid, will 20 benefit from expanding OMZs (Stewart et al., 2014), but many non-adapted fish and invertebrates (like 21 diurnal vertical migrators) will have their depth distributions squeezed, affecting the carbon transport and 22 trophic efficiency of food webs in the mesopelagic (Stramma et al., 2011; Brown and Thatje, 2014; Rogers, 23 2015) (high confidence). In OMZ waters, where zooplankton is almost absent, like in the Eastern Tropical 24 North Pacific, the microbial remineralisation efficiency of sinking particles would be reduced, eventually 25 increasing the transfer efficiency of organic matter to the deep ocean and thus biological carbon storage 26 (Cavan et al., 2017) (medium confidence). However, increases in ocean temperature may also lead to 27 shallower remineralisation of POC in warm tropical regions, counteracting the storage of carbon in the dark 28 ocean (Marsay et al., 2015). 29

30 5.2.4 Impacts on Deep Sea Floor Systems31

32 *5.2.4.1 Defining the Deep* 33

34 The deep sea is assessed here as the vast area of the ocean ≥ 200 m deep, beyond most continental shelves 35 (Figure 5.11). Below 200 m changes in light, food supply, and the physical environment lead to altered 36 animal taxonomic composition, morphologies, lifestyles, and body sizes collectively understood to represent 37 the deep sea (Tyler, 2003). Although the AR5 WG2 Chapter 30 (Hoegh-Guldberg et al., 2014) defined the 38 deep sea as below 1000 m, the absence of photosynthetically useful light and ensuing critical ecological 39 changes, biogeochemical transformations, and altered human interactions that occur on much of the sea floor 40 below 200 m have led both pelagic and benthic biologists to include the ocean waters and seafloor below 41 200 m within the definition of the deep sea (Herring and Dixon, 1998; Gage, 2003).

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43 5.2.4.2 Detection and Attribution of Oceanographic and Biological Changes in the Deep Sea 44

45 The deep sea (below 200 m) is vast and remote, difficult to access and expensive to study and as a result 46 hosts relatively few long biological time series or experiments suitable for assessment of climate impacts 47 (Smith et al., 2013). However, there are long-term records that document physical change derived from deep 48 observing programs (see Section 5.2.1.2.1) including from repeat hydrography to > 6000 m (Talley et al., 49 2016b) and from autonomous float instruments to 2000 m (Roemmich et al., 2015). The paleo record holds 50 information about hydrological, biogeochemical and biological change on historical or geological time 51 scales and contributes understanding valuable for interpreting modern responses to climate change. To 52 approach the ecosystem changes for the deep sea, this section relies on (a) observed and projected changes in 53 environmental exposures to climate drivers relative to natural variation (climate hazard) (Section 5.2.2) (b) 54 definition of habitat requirements and tolerance thresholds from current species distributions and habitat 55 suitability modeling, (c) mechanistic understanding derived from experiments with shallow water relatives 56 or with eurybathic deep-sea species reared in the laboratory (Section 5.2.4.1), (d) observed ecological, 57 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

2 impacts and their interaction3 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

Chapter 5

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.

8 5.2.4.4 Biological Responses to Stressors 9

10 Temperature is understood to influence the biogeography of the deep ocean at large scales (Watling et al., 11 2013), and to the extent that it reflects energy, can have a unimodal influence on biodiversity and biomass in 12 the deep sea (Watling et al., 2013; Yasuhara and Danovaro, 2016) (medium confidence). The underlying 13 mechanisms involve oxygen-and capacity-limited thermal tolerance (Pörtner, 2012) (Section 5.2.4.1.1). 14 Pressure tolerance may also be limited by availability of oxygen (Brown and Thatje, 2014), thus changes 15 brought about by ocean deoxygenation could limit the ability of animals to seek cooler waters in response to 16 warming by moving deeper (low confidence). Bottom temperature has important implications for estimates 17 of seasonal and spatial benthic flux variation, benthic-pelagic coupling, and impacts of predicted ocean 18 warming at high latitudes (Belley and Snelgrove, 2016). 19

20 Deoxygenation in the present and past derives largely from warming in the open ocean (Keeling et al., 2010; 21 Levin, 2018). For example, in the Santa Barbara Basin sediments, O₂ changes are associated with Dansgaard 22 Oeschger events (rapid climate fluctuations) and abrupt deglacial warming (Cannariato and Kennett, 1999; 23 Moffitt et al., 2015). The warming and oxygen loss coincide with shifts from diverse invertebrates and 24 multiple feeding guilds to extremophile invertebrates and eventually to extremophile, accompanied by 25 increased reliance on chemosynthesis (Moffitt et al., 2015). Oxygen loss affects deep seafloor taxonomic 26 composition, taxon-specific abundances, biomass, body size, diversity, physiology, reproduction and 27 behavior (Levin, 2003; Breitberg et al., 2015) (high confidence). 28

29 Ocean acidification, the availability of carbonate ions, and the aragonite and calcite saturation horizons 30 (boundaries above which aragonite and calcite carbonate ions are saturated and below which they are 31 undersaturated, making calcification more energetically costly) exert strong control on the calcification rates 32 of organisms, the energy required and the dissolution rates of living and dead carbonate skeletons, tests and 33 sediments which form habitat in the deep sea (medium confidence). While ocean acidification is recognized 34 to influence a suite of behavioural and physiological responses in shallow-water taxa, including growth rate, 35 metabolic rates, mode of catabolism, development, gene expression, energetics and more, most of these have 36 not been studied in deep-sea species. A meta analysis reported in AR5 Chapter 6 (Pörtner et al., 2014), Table 37 (6-3) identifies 10 studies involving 6 species of cold-water corals that suggest low vulnerability to CO_2 38 changes at RCP6.0 and medium vulnerability at RCP8.5, with negative effects starting at pCO₂ of 445 µatm. 39 Nevertheless, oxygen and capacity dependent thermal tolerance is *likely* to integrate a broad range of 40 physiological responses to multiple climate drivers: warming-induced CO₂ accumulation in body fluids links 41 to the effects of ocean acidification mediated by the weak acid distribution of CO₂, while temperature-42 induced hypoxemia links to the hypoxia sensitivity of thermal tolerance (Pörtner, 2012). 43

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 deep 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).

51 The Coupled Model Intercomparison Project (CMIP5) used in AR5 found that, under a high greenhouse gas 52 emissions scenario (RCP8.5), declines in primary production will lead to a global fall in export production 53 from -7% to -16% (Bopp et al., 2013) (Section 5.2.2.5). When linked to ocean acidification, which shoals

54 the calcite compensation depth and decreases production of biogenic calcium carbonate, the export

55 production declines can be up to -40.7% at 1000 m (Yool et al., 2013) or up to 38% at the northeast Atlantic

56 Seafloor (Jones et al., 2014). Those environments already subject to food stress (e.g., abyssal plains beneath

limited evidence). Arctic and Southern Ocean POC flux at the seafloor may increase by 60% and 53%,
 respectively by 2100 under RCP8.5 (*high confidence*); upwelling regions may also experience increases (*low confidence*) (Jones et al., 2014; Sweetman et al., 2017).

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5 Changes in POC flux are *likely* to alter the relative importance of the microbial loop and benthic 6 invertebrates in carbon transfer (Dunlop et al., 2016) (limited evidence). Deep-sea biota demonstrate 7 increased efficiency (effective use of food energy for growth and metabolism with minimal loss) at low food 8 inputs (due to small size and dominance by prokaryotic taxa), thus they should be better able to adapt than 9 shallow biota to reduced POC flux with climate change (Gambi et al., 2017) (low confidence, based on E 10 Mediterranean Sea ultra-oligotrophic sediments). Changes in the overlying mesopelagic and bathypelagic 11 communities (see Section 5.2.2.2) will affect food flux to the deep sea, as nekton and zooplankton transfer 12 energy through diel vertical migrations, ontogenetic migrations and falls of dead carcasses (Gage, 2003) 13 (medium likelihood, limited evidence). Abyssal responses to changes in POC flux are discussed in Section 14 5.2.4.4.

16 5.2.4.4.1 Abyssal plains

17 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 18 19 rising temperatures will reduce surface production and export flux to the seabed (Steinacher et al., 2010). 20 The rapid abyssal response to changes at the surface (Smith et al., 2013) suggest declines in seafloor biomass 21 by the 2090s are highly likely, with projected global losses of 5.2% (Jones et al., 2014) to 17.6% (Yool et al., 22 2017). Depths below 2000 m will experience the greatest declines in biomass from the 1990s to 2090s (-23 28.9% at 2000 m, -32% at 5000 m under RCP8.5) with much lesser declines under RCP2.6 (-5.8%, -7.0% 24 respectively) (Yool et al., 2017). By 2091-2100 under RCP8.5, benthic biomass may decline up to 38% in 25 the abyssal NE Atlantic (Jones et al., 2014) (medium confidence). These estimates do not account for 26 changes in the type or quality of the sinking material, to which macrofaunal and meiofaunal invertebrates are 27 highly sensitive (Smith et al., 2008; Smith et al., 2009; Tittensor et al., 2011). Nor do they account for direct 28 faunal responses to changes in temperature, oxygen or the carbonate system, all of which will influence 29 benthic response (medium confidence). Lower food supply to the abyss is expected to result in a size-shift 30 towards smaller organisms (Jones et al., 2014), resulting in rising respiration rates, lower biomass production 31 efficiency, and lesser energy transfer to higher trophic levels (Brown et al., 2004), with changes being largest 32 for macrofauna and lesser and similar for megafauna and meiofauna (Jones et al., 2014) (limited evidence, 33 low confidence). A strong positive relationship between annual POC flux and abyssal sediment community 34 oxygen consumption (with POC flux accounting for 63% of seafloor carbon demand during a 27-year time 35 series at 4000 m) (Smith et al., 2016b), combined with projected changes in biomass suggest that abyssal 36 communities are highly likely to experience structural and functional changes that affect the carbon cycle in 37 this century. The findings speak to the importance of monitoring POC flux to the deep sea across many 38 regions. 39

40 5.2.4.4.2 Continental slope sediments.

41 Seafloor assemblages on continental margins experience strong vertical temperature, oxygen and pH 42 gradients generating sharp ecological zonation (Levin and Sibuet, 2012). Both the extant gradients and past 43 fluctuations in exposures make margins a valuable natural laboratory for understanding ecosystem response 44 to multiple climate drivers (Sperling et al., 2016). Most global climate projections are too coarse (grid cell 45 too large) to resolve changes in gradients of climate drivers on margins, however long-term regional 46 observations (e.g., the California Cooperative Oceanic Fisheries Investigations, or CALCOFI) and modeled 47 projections have highlighted increasing vulnerability of continental slope ecosystem to ocean acidification 48 (Gruber et al., 2012) and ocean deoxygenation (Goericke et al., 2015) as upwelling intensifies in eastern 49 boundary upwelling areas (Sydeman et al., 2014) (medium confidence).

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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 1 protozoans (Woulds et al., 2009) (high likelihood, medium confidence). A loss of deep bioturbating faunas at 2 lowest oxygen levels is virtually certain, and but because they contribute to both carbon burial and 3 remineralization (via oxygenation) there is less certainty about the net effect on carbon burial rates, although 4 it is *likely* to be positive (Smith et al., 2000; Levin and Dayton, 2009). Hypoxia reduces colonization 5 potential and community resilience of slope biota (low confidence, one study) (Levin and Sibuet, 2012). 6 Carbon sequestration and nitrogen recycling are highly sensitive to small changes in oxygenation (Deutsch et 7 al., 2011) (medium confidence).

8 9 Benthic foraminifera are more tolerant to changing climate drivers on margins than larger invertebrates. 10 Short-term experiments have showed no significant effect of ocean acidification on survival for Ammonia spp. (Dissard et al., 2010; Haynert et al., 2011; Keul et al., 2013), Bulimina marginata and Bolivina argentea 11 12 (McIntyre-Wressnig et al., 2014) and Globobulimina turgida (Wit et al., 2016). However, lower pH 13 exacerbates shallow foraminiferal sensitivity to warming (Webster et al., 2016) and shallow water (reef) 14 for a microbiomes are sensitive to rising temperature (Webster et al., 2016). Additionally, the 15 sensitivity of calcification by planktic foraminifera such as G. bulloides to ocean acidification can reduce the 16 rain rate to the seabed and contribute to declining POC flux (Davis et al., 2017). Benthic foraminifera also 17 exhibit well-recognized changes in abundance and diversity in response to oxygen loss (Bernhard and 18 Reimers, 1991; Gooday et al., 2000; Moffitt et al., 2014). Hypoxia in combination with elevated pCO₂ favors 19 survival of some foraminifera (*Globobulimina*) (Wit et al., 2016). Combined declines in pH and oxygen may 20 lead to increase in some agglutinating taxa and a decrease in the carbonate-producing foraminifera, including 21 those using carbonate cement (van Dijk et al., 2017) (one study, low confidence).

22 23 Among the metazoan meiofauna, nematodes are sensitive to temperature change; an anomalous temperature 24 drop of only 0.4°C in the Mediterranean deep sea caused a significant decrease in nematode abundance and 25 functional diversity, concomitant with increased species richness and evenness (Danovaro et al., 2001; 26 Danovaro et al., 2004). Nematode abundance and biomass can also decrease with increasing sediment 27 temperatures (Yodnarasri et al., 2008). Ice shelf collapse transforms nematode assemblages from 28 depauperate, low-diversity communities, to richer and denser communities dominated by opportunistic 29 species (Ingels et al., 2012). Elevated CO₂ affects nematode survival (Barry et al., 2004; Fleeger et al., 2006; 30 Fleeger et al., 2010) and although tolerant to hypoxia as indicated by high abundances, their diversity 31 declines significantly within low-oxygen sediments (Neira et al., 2001). Meiofaunal biomass is sensitive to 32 POC flux (Smith et al., 2008). Reduced POC flux with depth causes Mediterranean nematodes to decline in 33 body size, conferring efficiency (biomass:respiration) in exploitation of food resources under very low food 34 conditions (Gambi et al., 2017). There is low agreement about meiofaunal responses among studies, 35 sometimes reflecting opposing responses in different regions. However, there is agreement that meiofauna are sensitive to change in environment and food supply (medium confidence).

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38 Warming-induced changes in circulation are linked to abrupt changes in benthic ostracod diversity in the 39 historical record (Yasuhara et al., 2008) (medium confidence). Gamete synthesis, skeletal structure and 40 biochemical composition of deep-sea echinoderms are sensitive to ocean acidification (deep-sea holothurians 41 - Verkaik et al. (2016)) and to ocean deoxygenation (echinoids - Sato et al. (2017)). Shoaling oxyclines can 42 cause changes in echinoid vertical distributions (Sato et al., 2017) (one study, low confidence). However 43 species in upwelling regions, such as the echinoid Strongylocentrotus fragilis, that have adapted to oxygen 44 minimum zones with characteristically high CO₂ levels are less vulnerable to the negative impacts of ocean 45 acidification (Taylor et al., 2014). Where both oxygen and CO_2 stress occur together, oxygen can to be the 46 primary driver of change (Taylor et al., 2014; Sato et al., 2017), but this is not always true (Gobler and

47 Baumann, 2016).

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49 There is a strong macroecological relationship between depth, export POC flux, biomass and zonation of 50 macrobenthos on continental slopes (Wei et al., 2011), such that loss of POC flux will alter seafloor 51 community biomass and structure (high confidence). This is modified on the local scale by near-bottom 52 currents, which alter sediment grain size, food availability, and larval dispersal (Wei et al., 2011). 53

54 5.2.4.4.3 Chemosynthetic ecosystems

55 With nutrition derived largely from chemosynthetic sources fueled by fluids from the earth's interior,

56 hydrothermal vents and seeps are often assumed to be largely insulated from effects of climate change. 57

However, there are multiple linkages to broader surface ocean processes that ultimately influence the

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productivity, diversity, and health of these systems (medium confidence). Examples include requirements for 2 oxygen (to serve as electron acceptor for aerobic hydrogen-, sulfide- and methane oxidation), links to surface 3 productivity and mesoscale eddies, for growth and transport of larvae (Arellano and Young, 2009; Adams et 4 al., 2011), and for nutrition of mixotrophic species (relying on photosynthesis and chemosynthesis), as well 5 as seasonal/ ENSO influences on these features. 6

7 Vent species such as bathymodiolin mussels, alvinocarid shrimp, and some limpets have larvae that develop 8 in surface waters (Herring and Dixon, 1998; Arellano et al., 2014). These will be exposed to warming 9 waters, decreasing pH and carbonate saturation states, and in some places, reduced phytoplankton 10 availability, causing reduced calcification and growth rates (as in shallow water mussel larvae, Frieder et al. 11 (2014)) (medium confidence). Larvae originating at vents or seeps beneath upwelling regions may also be 12 impaired by effects of hypoxia associated with expanding oxygen minimum zones (Stramma et al., 2008) 13 during migration to the surface (low confidence).

14 15 Methane seep systems occur almost exclusively on continental margins where they will experience changes 16 in exposure to climate drivers (Sweetman et al., 2017; FAO, 2018) (high confidence). Rising bottom 17 temperatures or shifting of warm currents on margins could increase dissociation of buried gas hydrates on 18 margins (Phrampus and Hornbach, 2012) (low confidence). If this occurs there would be intensified 19 anaerobic methane oxidation (which produces hydrogen sulfide) (Boetius and Wenzhoefer, 2013), and 20 expansion of methane seep communities (medium confidence). Such an expansion would increase seafloor 21 productivity (high confidence) and either locally decrease or regionally increase biodiversity (Cordes et al., 22 2010) (medium confidence). At the same time, climate-driven expansion of midwater oxygen minima 23 (Stramma et al., 2008; Schmidtko et al., 2017) occurring at depths where many seep ecosystems form (200-24 1000 m) (Levin, 2005) could reduce the cover of large vent and seep foundation species (mussels, siboglinid 25 tube worms), which are unable to grow under dysoxic conditions ($< 5-10 \mu mol kg^{-1}$) (Sweetman et al., 2017) 26 (medium confidence).

28 At some vents, high CO₂ conditions limit mussel shell calcification, reduce shell weight and slow growth (in 29 Bathymodiolus septemdierum) at low saturation states (Tunnicliffe et al., 2009; Rossi and Tunnicliffe, 2017). 30 However, the absence of predators reduces the need for calcification and CO₂ stress can be relieved by high 31 food supply (through symbiont autotrophy including hydrogen-based metabolism) (low confidence).

32 33 Climate-driven changes can alter the biodiversity associated with mussel bed habitat as well as the food web 34 in which they reside (low confidence). Declines in POC flux to the deep seabed in most areas except the 35 Arctic (see Section 5.2.1.5) would negatively affect vent and seep mussels and other mixotrophs that have 36 the ability to use photosynthetically derived food raining down from the surface to supplement 37 chemosynthetic sources. Reduction in amount and changes in timing of surface production affects vent and 38 seep mussels (e.g., Bathymodiolus azoricus and Bathymodiolus childressi) which show synchronized 39 gametogenesis (sperm and egg production) with spawning occurring loosely around the time phytodetritus 40 from surface production arrives at the seafloor (Dixon et al., 2006; Tyler et al., 2007). These climate-driven 41 changes to seep and vent ecosystems could alter food available for surrounding deep-sea ecosystems which 42 benefit through transport of nutrients and microbes, movement of vagrant predators and scavengers, and 43 plankton interactions (Levin et al., 2016) (*limited evidence; low confidence*). Warming and its effects on 44 climate cycles have the potential to alter patterns of connectivity through changes in circulation (Fox et al., 45 2016) (limited evidence; low confidence).

47 5.2.4.4.4 Canyons

48 Canyons are abundant widespread topographic features that deeply incise continental and island margins 49 globally (Harris et al., 2014). As regions of focused particulate organic matter and coarse material transport 50 from the shelf (including terrigenous plant material), intensified mixing and advection of water masses, 51 submarine canyons connect deep-sea and shallow ecosystems and enhance sediment transfer and deposition 52 to the abyssal plain (Levin and Sibuet, 2012). As a result of local upwelling of nutrients to the surface waters 53 induced by canyon topography, these areas exhibit high productivities with enhanced plankton, fish, 54 mammal and invertebrate abundances (De Leo et al., 2010; Levin and Sibuet, 2012). At the seabed and in 55 deep waters, canyons can sustain exceptionally high levels of biomass sometimes including high densities of 56 deep-sea corals, gorgonians and sponges (De Leo et al., 2010; Fernandez-Arcaya et al., 2017). These 57 functionally important habitats are vulnerable to climate-driven changes in physical and biogeochemical

1 processes that affect food supply. Enhanced stratification and change in the intensity and frequency of 2 downwelling processes under atmospheric forcing (including storms and density-driven cascading events 3 (Allen and Durrieu de Madron, 2009) will decrease organic matter transported through canyons (low 4 confidence). Changes in the quantity and quality of transferred particulate organic matter, as well as physical 5 disturbance during extreme events cause a complex combination of positive and negative impacts at different 6 depths along the canyon floor (Canals et al., 2006; Pusceddu et al., 2010). This will be superimposed on 7 regional stratification, inducing declines in POC flux predicted to affect 85% of the 8637 mapped canyons 8 and could lead to declines in benthic biomass in the next century (Jones et al., 2014) (medium confidence). 9 Expansion of oxygen minimum zones beneath upwelling regions at concentrations < 0.7 ml/l would reduce 10 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). 11 12 Canyon walls are home to extensive deep-water reef habitat formed of cold water corals, octocorals, sponges 13 and bryozoans (Williams et al., 2010), and are now recognized to host many methane seeps and other 14 chemosynthetic habitats supported by massive transport of terrestrial organic matter (Pruski et al., 2017). 15 Climate effects on these ecosystems are discussed separately in Sections 5.2.3.6, and 5.2.3.9.

17 5.2.4.4.5 Seamounts

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18 There are 100,000-200,000 seamounts (underwater volcanoes, many of which are inactive) that protrude 19 above the seafloor. Seamount flanks are exposed to strong gradients in temperature and sometimes oxygen, 20 and often exhibit sharp faunal zonation reflecting fairly narrow habitat requirements. They can support high 21 animal densities and biomass, exhibit high secondary production supported by locally enhanced primary 22 production and intensified flows, function as diversity hotspots and serve as stepping stones for larval 23 dispersal (Rowden et al., 2010), but are also vulnerable to fishing disturbance, especially from bottom 24 trawling (Clark and Tittensor, 2010). The majority of seamounts (82% of 33,452 globally mapped 25 seamounts) are predicted to experience reduced POC flux under RCP8.5 in 2100, resulting in declines in 26 benthic biomass (Jones et al., 2014). Habitat suitability for reproduction (e.g., skate egg-laying sites) on 27 seamounts may be disrupted by warming, given apparently narrow ranges reflecting environmental 28 requirements (Henry et al., 2016). Seamounts have been proposed to serve as refugia for coldwater corals 29 facing shoaling aragonite saturation horizons (Tittensor et al., 2011), but could become too warm for deep-30 water corals in some regions (e.g., projections off Australia) (Thresher et al., 2015). In the North Atlantic 31 seamounts are an important source of cold-water coral larvae maintaining resilience under shifting NAO 32 conditions (Fox et al., 2016), so loss of habitat suitability may have far-reaching consequences (Gehlen et 33 al., 2014b).

35 5.2.4.4.6 Biogenic structures: cold-water coral ecosystems.

36 In the years since IPCC AR5, understanding on the vulnerability of cold-water corals to global change has 37 increased substantially, with new initial findings including information on the impacts of ocean acidification 38 on skeletal integrity (Hennige et al., 2015) and the implications of altered regional oceanographic regimes 39 (Fox et al., 2016). While important details in our understanding of cold-water coral physiological response to 40 ocean acidification are coming into place, great uncertainty remains in our understanding of the adaptive 41 capacity of cold-water corals or how the connectivity of present-day populations will be affected by altered 42 regional current patterns (Roberts et al., 2016). However, this awareness is now feeding forward to shape 43 thinking in policy development, including Area-Based Management Tool approaches (Johnson et al., 2018).

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45 Cold-water coral reef ecosystems consist of a (a) 3-dimensional, living complex of organisms (solitary or 46 colonial) dominated by one or more species of habitat-forming (foundation) scleractinian coral, (b) transition 47 zone with carbonate skeletons of dead individuals, and (c) rubble zone, sometimes growing on a carbonate 48 matrix many hundreds of meters thick (Buhl-Mortensen et al., 2010). Calcium carbonate concentrations and 49 aragonite saturation influence the distribution of scleractinians and octocorals and the habitats formed by 50 their structures (Guinotte et al., 2006; Tittensor et al., 2009; Thresher et al., 2015) (medium confidence). 51 Observed (Byrne et al., 2010) and predicted future acidification and shoaling of the aragonite saturation 52 horizon, especially in the North Atlantic will alter species habitat suitability and thus distributions (Gehlen et 53 al., 2014a) (medium confidence).

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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

1 CO₂ levels (Hennige et al., 2015) (low confidence). It can also calcify when exposed to multiple 2 environmental stresses in the laboratory (Hennige et al., 2015; Büscher et al., 2017), however acidification 3 exposure over 12 months will cause the biomineralized aragonite skeleton to exhibit less organization, a 4 longer and thinner growth form, and reduced structural integrity of exposed skeletal framework, that forms 5 the base of cold-water coral reefs and a significant proportion of the habitat, (low confidence) (Hennige et 6 al., 2015). Weakening of skeletons by acidification enhances bioerosion of carbonates (e.g., by bacteria, 7 fungi, annelids and sponges) (Schönberg et al., 2017), and bioerosion may exacerbate dissolution; 8 dissolution and bioerosion will cause loss of cold-water coral habitat (virtually certain). 9

10 The importance of skeletal dissolution and bioerosion under ocean acidification has largely been overlooked 11 in the discussion on how cold-water coral ecosystems will fare under future climate change. It is of equal 12 importance to the growth of live cold-water corals, as it is the habitat provision of the reef as a whole, made 13 up of live and dead coral framework, that is important for biodiversity support. Importantly, given that no 14 adaptation can happen with regard to dissolution, as it is a biogeochemical response, it is highly likely that 15 increased dissolution of exposed aragonite, further enhanced by bioerosion, rather than a reduction in 16 calcification rates of the live coral, could lead to future net cold-water coral reef loss (high confidence). This 17 is based on current occurrence of 95% of cold water coral reefs above the aragonite saturation horizon 18 (Guinotte et al., 2006), loss of structural integrity of exposed coral framework of dead corals to projected 19 aragonite undersaturation, and understanding of biogeochemical processes of aragonite dissolution in 20 aragonite undersaturated water (Eyre et al., 2014). 21

22 Experiments on cold-water coral growth over a range of time scales have now demonstrated positive (but 23 variable) calcification rates under future scenarios of ocean acidification and warming (Maier et al., 2013a; 24 Maier et al., 2013b; Hennige et al., 2015; Rodolfo-Metalpa et al., 2015; Gori et al., 2016; Maier et al., 2016; 25 Büscher et al., 2017) (medium confidence). However, net calcification rates (calcification+dissolution) of L. 26 *pertusa* exposed to aragonite-undersaturated conditions or close to saturation ($\Omega < 1$) often decrease to close 27 to zero or even became negative (Lunden et al., 2014; Hennige et al., 2015; Büscher et al., 2017). There is 28 genetic variability in L. pertusa's ability to calcify at low aragonite saturation, and some populations (e.g., in 29 the Gulf of Mexico) may more readily adapt to ocean acidification (Kurman et al., 2017) (single study low 30 *confidence*). However, hypoxia (1.5 ml L⁻¹) can cause mortality in Gulf of Mexico *L. pertusa* populations, 31 which already live under low-oxygen, high CO₂ conditions (Lunden et al., 2014) (single study, *low* 32 confidence). 33

34 The occurrence of black corals (Antipathes fiordensis) in the low salinity, shallow waters of New Zealand 35 fjords (< 35 m) indicates that some non-reef forming species can tolerate low pH conditions (Jiang et al., 36 2015). The solitary coral *Desmophyllum dianthus*, a species closely related to *L. pertusa*, also is able to live 37 and thrive under natural low pH conditions (down to 7.4) in fjords of Northern Patagonia, reflecting potential 38 to adapt to ocean acidification (Fillinger and Richter, 2013). However, exposure to elevated pCO₂ when 39 combined with elevated temperatures reduces calcification and respiration rates in D. dianthus, and shifts 40 metabolic substrates from a mixture of protein and carbohydrate/lipids to a less efficient protein-dominated 41 catabolism (Gori et al. 2016) (limited evidence, single study). Populations of this species waxed and waned 42 over the last 40,000 years as the global ocean was influenced by glacial advances and retreats and changing 43 concentrations of CO₃⁻², O₂, and surface productivity (Thiagarajan et al., 2013). Coral populations on 44 seamounts in the N. Atlantic and SW Pacific Oceans altered their depth distribution in response to changes 45 in the aragonite saturation depth, which has shoaled since the last glacial maximum (Thiagarajan et al., 46 2013); future increases in atmospheric CO_2 are expected to further decrease suitable habitat for deep-47 dwelling hard coral species and their associated benthic organisms (Thiagarajan et al., 2013) (low 48 confidence).

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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 1 oxygen minimum zone $(3.3 \text{ ml l}^{-1}$ for North Atlantic specimens), highlighting critical thresholds for coral 2 populations living at the edge of their tolerance (e.g., periphery of hypoxic zones, warm Mediterranean deep-3 sea). The role of temporal dynamics in the combination of stressors is recognized but is still poorly studied 4 under in situ conditions (one study supports species-specific response to seasonal influence in the growth of 5 *L. pertusa* and *M. occulata*) (Lartaud et al., 2014).

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7 There is a decrease in the northeast Atlantic suitable habitat for key coral species (*Desmophyllum* spp. and 8 Lophelia pertusa) predicted by habitat suitability modelling under projected changes in T, O₂, pH and POC 9 flux for 2081-2100, but an increase for 2041-2060. Habitat suitability analyses suggest that food supply 10 (POC flux) is a critical variable for the corals Desmophyllum spp., Lophelia pertusa, Madrepora oculata, 11 Acanella arbuscula, and Paragorgia arborea, while dissolved oxygen is an important variable in the 12 northwest Atlantic area for Desmophyllum spp., Lophelia pertusa, Madrepora oculata (Morato et al., 2006; 13 Levin, 2018). In the Southern hemisphere, future projections to 2099 of warming and acidification predict 14 extreme loss of habitat for cold water coral (Solenosmilia variabilis) on seamounts off Australia and New 15 Zealand under RCP4.5 and nearly complete loss under RCP8.5, both warming above a tolerance threshold of 16 7°C and decline of aragonite saturation ($\Omega_{arag} < 0.84$) below that required for survival are responsible 17 (Thresher et al., 2015). 18

19 Future mean North Atlantic Oscillation (NAO) state can affect cold-water coral habitat connectivity in the 20 deep North Atlantic (medium confidence), based on mechanistic understanding of the variation in wind-21 driven, Ekman-layer surface currents in the northeast Atlantic with NAO state (Fox et al., 2016). Modeled 22 variability of connectivity of protected cold-water coral (Lophelia pertusa) populations in the northeast 23 Atlantic over 40 years indicates positive NAO generates overall higher connectivity but lower east-to-west 24 connectivity while lower NAO is associated with decreased connectivity, more isolated populations, and 25 thereby reduced resilience (medium confidence). This study was informed by recent laboratory studies of L. 26 pertusa larval behavior, planktonic durations (8-9 weeks or more) and vertical positioning in the water 27 column (rising from depth to the surface, drifting, then descending to the bed) (Larsson et al., 2014). 28 Uncertainties lie in future NAO trends (AR5 reported medium confidence in an increase in mean NAO by 29 2050, especially in autumn and winter), and how laboratory results transfer to the real ocean (e.g., do the 30 larvae reach the surface layer?).

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32 5.2.4.4.7 Biogenic structures: sponges and other taxa.

33 Habitat-forming, deep-sea sponges (Geodia barrette) originally thought to experience mortality under a 4°C 34 temperature rise (Guihen et al., 2012) were found through experimentation to tolerate 5°C elevation along 35 with their microbiome, albeit with a temporary rise in respiration and nitrogen efflux and cellular stress 36 (Strand et al., 2017). Ocean acidification (pH 7.5) reduces the feeding of deep-sea demosponges (Radiella 37 sp., Polymastia sp.), indicating sensitivity of deep-sea sponges under natural and anthropogenic disturbance 38 (Limited evidence, low confidence) (Robertson et al., 2017). Ocean acidification has adverse effects of 39 acidification on gamete production, embryo morphology and performance as well as structure of ossicles and 40 lipid content of muscles, gonads and oocytes of a widespread, abundant cold-water holothurian (Cucumaria 41 frondosa) (Verkaik et al., 2016) (single study, limited evidence, low confidence). Nevertheless, the geologic 42 record and modern distributions suggests that sponges are more tolerant to warming, acidification (and 43 potentially deoxygenation) than are corals and may be less vulnerable to changes in temperature and pH 44 (Schulz et al., 2013). Giant agglutinated protozoans (xenophyophores), common on the deep sea floor in 45 many settings, are local hotspots of deep-sea biodiversity and are recognized as vulnerable marine ecosystem 46 indicators by several regional fisheries management organizations (RFMOs) (Levin et al., 2018). Habitat 47 suitability analyses reveal thresholds associated with depth, temperature, nitrate (reflecting productivity), 48 oxygen and carbonate chemistry (Ashford et al., 2014) suggesting xenophyophores are potentially vulnerable 49 to changes in these conditions at the seafloor. 50

51 *5.2.4.5 Gaps* 52

53 Understanding of climate change impacts on deep-sea ecosystems is challenged by the absence of long-term 54 observations in most systems, and limited numbers of laboratory or *in situ* studies. Most such studies 55 examine on single species and on single climate drivers. Studies on the effects of ocean acidification in the 56 deep ocean are restricted primarily to calcifying organisms (i.e., cold-water corals) while similar work on 57 non-calcifying ecosystem engineers (i.e., sponges, xenophyophores, sea pens) and the ecosystems that they

1 form (e.g., sponge grounds), is almost absent. Although the 300 my geological record contains examples of 2 simultaneous change in temperature, pH and oxygen, paleo research highlights the difficulty of identifying 3 individual drivers from presence/absence of certian fossils in the absence of geochemical data, as well as the 4 uncertainty introduced by short time scales <10,000 years, Modern multiple driver studies are based largely 5 on climate projections combined with habitat suitability assessment for foundation species and key 6 commercial species, but additional groundtruthing is required, ideally through sustained deep-ocean 7 observing programs that incorporate biology across a broader diversity of ecosystems. Baseline assessments 8 are needed to reveal the environmental status of deep-sea ecosystems, with development of spatial and 9 temporal scales to be assessed and abiotic and biotic indicators to be used for measuring response to 10 changing climate drivers (Danovaro et al., 2017; Sweetman et al., 2017).

12 Some components of the ecosystem are fundamental to deep-sea ecosystem functioning but are severely 13 understudied in the deep sea, including viral and protist communities and their links to higher forms. Viruses 14 are key drivers of microbial metabolism (Dell'Anno et al., 2015; Danovaro et al., 2016). Labile C released 15 by viral cell lysis and virus decomposition stimulates microbial heterotrophs which are coupled to 16 ammonium-dependent archaeal chemosynthesis and nitrogen regeneration (Danovaro et al., 2016). Viral 17 dynamics are key to understanding microbial mediated CO₂ production and consumption and carbon storage 18 capacity of the deep ocean, and cumulative impacts could cause regime shifts that alter life support services 19 provided largely by microbes (Carbon sequestration and Nitrogen regeneration). 20

21 5.2.4.6 Summary for the Deep Pelagic Ocean

22 23 Despite its seeming remoteness, most of the deep-sea floor and its varied ecosystems already have or are 24 projected to experience rising temperatures and declining oxygen, pH and POC flux beyond natural 25 variability within the next half century (See Section 5.2.1). Much of the abyssal seafloor is expected to 26 experience declines in food supply (Section 5.2.1.5) that will diminish benthic biomass, change community 27 structure and rates of carbon burial (medium confidence). Expansion of oxygen-limited and oxygen-28 minimum zones on upwelling margins will reduce fish and invertebrate biodiversity, simplify food webs, 29 reduce ecosystem resilience and alter elemental cycling (medium confidence). Even chemosynthetic 30 ecosystems like hydrothermal vents and seeps, once thought to be isolated from surface phenomena, are 31 recognized to be vulnerable to changing ocean conditions through effects on their pelagic larvae (e.g., 32 shrimp and mussels), partial reliance on surface derived particulate organic matter (e.g., mussels), microbial 33 symbionts that require oxygen to fix carbon (e.g., tubeworms, mussels), and reliance on mesoscale 34 circulation features for connectivity (low confidence). Biogenic habitats that form biodiversity hotspots (e.g., 35 corals, sponges, xenophyophores), and the topographic features that support them (seamounts, canyons, 36 slopes) will be exposed to a combination of stressors (warming, reduced food supply, increasing acidity and 37 oxygen loss) (medium confidence). Some of the greatest cumulative exposures to warming, declining pH, 38 oxygen and food supply, will be in the Arctic and Atlantic Oceans over the next 100 years (medium 39 confidence). Ecological responses are expected to vary regionally, but as critical thresholds of temperature, 40 oxygen and CO₂ are exceeded, changing species depth distributions, dissolution, and bioerosion, exacerbated 41 by lower food supply, will cause loss of biogenic habitat critical to the maintenance of biodiversity and 42 ecosystem health (medium confidence). 43

44 5.2.5 Risk Assessment of Open Ocean Ecosystems45

46 Overall, the upper ocean (0–700 m) and 700-2000m layers have both warmed from 2004 to 2016 (virtually 47 certain) and the abyssal ocean continues to warm in the Southern Hemisphere (high confidence). The ocean 48 is stratifying; observed warming and high latitude freshening are both surface intensified trends making the 49 surface ocean lighter at a faster rate than deeper in the ocean (high confidence). It is very likely that 50 stratification in the upper few hundred meters of the ocean will increase significantly in the 21st century. It is 51 virtually certain that ocean pH is declining by ~0.02 pH units per decade where time-series observations 52 exist. The anthropogenic pH signal has already emerged over the entire surface ocean (high confidence) and 53 emission scenarios are the most important control of surface ocean pH relative to internal variability for most 54 of the 21st century at both global and local scale (virtually certain). The oxygen content of the global ocean 55 has declined by about 2%, largely in responses to changes in ocean circulation and respiratory demand (high 56 confidence). Over the next century oxygen declines of 3.5% by 2100 are predicted by CMIP5 models 57 globally (medium confidence), with low confidence at regional scales, especially in the tropics. The largest

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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.

4 5 Emergence of novel ocean conditions for organisms from plankton to mammals are driving changes in eco-6 physiology, biogeography and ecology and biodiversity (high confidence). Observed and projected 7 population declines in the equator-ward range boundary (medium confidence), expansion in the poleward 8 boundary (high confidence), earlier timing of biological events (high confidence), and overall shift in 9 biomass and species composition (very high confidence) are consistent with expected responses to climate 10 change. It is *likely* that increased OA has not yet caused sufficient reduction in fitness to decrease 11 abundances of calcifying phytoplankton and zooplankton, but is very likely (high confidence) that calcifying 12 planktonic organisms will experience great decreases in abundance and diversity under high emission 13 scenarios by the end of the century. Changes in biogeography and community structure are projected to 14 continue in the 21st century (high confidence), with potential animal biomass projected to decrease (likely). 15 Open ocean ecosystem functions are at risk of climate change impacts, with elevated risk of changes in 16 different components of the biological pump in the epipelagic ocean, as well as changes in the efficiency of 17 carbon export in the deep ocean. However, there is low confidence on the direction and magnitude of these 18 changes. Biogenic habitats that form biodiversity hotspots in the deep sea floor and the topographic features 19 that support them (seamounts, canyons, slopes) will be exposed to a combination of stressors (warming, 20 reduced food supply, increasing acidity and oxygen loss) (medium confidence). The strong positive 21 relationship between annual Particulate Organic Carbon (POC) flux and abyssal sediment community 22 oxygen consumption combined with projected changes in biomass suggests that benthic communities in 23 abyssal and deep-sea habitats will experience structural and functional changes that affect the carbon cycle 24 in this century under all emission scenarios (medium confidence). 25

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*).

34 5.3 Changing Coastal Seas and Biodiversity35

36 5.3.1 Introduction to Changing Coastal Seas 37

38 The world's coastal and shelf seas (hereafter called the coastal seas collectively) extend from the coastline 39 towards the 200 m depth (Section 5.3.2) and encompasses diverse coastal ecosystems, including estuaries, 40 sandy beaches, kelp forest, mangroves and coral reefs (Section 5.3.3). Although they occupy a small part of 41 the World Ocean (7.6%), coastal seas provide up to 30% of global marine primary production and about 42 50% of the organic carbon supplied to the deep open ocean (Chen, 2003; Bauer et al., 2013) (Section 43 5.2.2.5). In addition, these systems harbor 90% of the world's fish catch and their overall economic value is 44 estimated to be at least 40% of the value of the world's ecosystem services and natural capital (Costanza et 45 al., 2014) (Section 5.4.1.3). These marine environments include several frontal and upwelling areas (Box 46 5.2) that support high plankton productivity and fisheries yields (Scales et al., 2014), while coastal biogenic 47 structures e.g., mangrove forest, provide physical protection against extreme events such as storms and 48 floods (Kelleway et al., 2017a). Regional characteristics and habitat heterogeneity of some seas allow the 49 development of endemic fauna and flora, which makes them particularly vulnerable to climate change 50 impacts with high risk of diversity loss and alterations in ecosystem structure and functioning (Rilov, 2016; 51 Chefaoui et al., 2018). Features such as bathymetry and mesoscale processes provide refuge and facilitate the 52 aggregation of prey and safeguard the feeding and breeding of macrofauna, including marine mammals, 53 seabirds and reptiles (Section 5.3.3.9).

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55 Unlike the open ocean where detection and attribution of climate driven-physical and chemical changes are 56 robust (Section 5.2.2), coastal habitats and shelf seas display regional hydrological complexity driven by the 57 interactions between the land, the sea and seabed structures and substrates, and are exposed to direct human

1 impacts that can render the conclusive detection and attribution of climate effects difficult (Levin et al., 2 2015). The high density of human populations along coastal areas in the World implies that most of these 3 ecosystems are exposed to non-climatic disturbances such as nutrient loads (eutrophication), coastline 4 modifications, pollution and overfishing. Climate threats interact with such non-climatic disturbances and 5 pose serious risk to ecosystems goods and services. For example, cumulative exposure to eutrophication 6 exacerbates the vulnearibility of coastal ecosystems to warming and deoxygenation, increasing the likelihood 7 of harmful algal blooms (HABs) (Box 5.3). The expansion of human infrastructure limits the adaptation of 8 coastal ecosystems to sea level rise, by preventing the landward migration of coastal forms and reducing the 9 available sites for nesting, feeding and breeding. Projections of the ecological impacts of climate change 10 must thereby deal with a number of emerging complexities such as the differentiation between the long-term 11 climate trends (e.g., progressive acidification) and the short-term natural fluctuations (Boyd et al., 2018), 12 ranging from the seasons to climate oscillations like El Niño. The so-called 'time of emergence' for specific 13 drivers beyond background variability is distinct in different ecosystems and is strongly sensitive to 14 projected mitigation scenarios (Henson et al., 2017) (Box 5.1).

15 16 Despite the multi-scale environmental and anthropogenic stressors that affect the land-sea continuum, global 17 experimental and long-term observational evidence since AR5 (AR5 WG2 Chapters 5, 6 and 30), allow the 18 attribution of current changes in coastal ecosystems to climate trends. Moreover, the emergent ecosystem 19 responses detected at present day strengthen the projections of their impacts under future emission scenarios 20 (Section 5.3.4). This section assesses the exposure of global coastal systems to multiple climate hazards: 21 warming, acidification, deoxygenation and sea level rise, by addressing the physical and biogeochemical 22 changes in coastal seas (Section 5.3.2), and the sensitivity and vulnerability of each coastal ecosystem, 23 considering geomorphological and biological components (Section 5.3.3). The assessment highlights 24 common ecosystem responses to climate threats: coastal submergence and erosion, reduction in habitat area 25 and complexity, poleward migration of species, biodiversity loss and reduction in carbon sequestration; and 26 compares the adaptive capacity in the context of ecosystem resilience and the applicability of mitigation 27 actions.

5.3.2 Oceanographic Drivers

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

- 34 Changes in freshwater delivery from rivers and groundwater will directly affecting coastal water salinity 35 and suspended sediment loads (Yang et al., 2015; Dunn et al., 2018; Zahid et al., 2018). Such changes 36 may result from increased or decreased precipitation in river catchments, changes in glacial meltwater, 37 or anthropogenically-mediated hydrological changes (e.g., due to river dams for hydropower and/or 38 water extraction) (Chang et al., 2015; Mukhopadhyay and Khan, 2015; Arnell and Gosling, 2016). 39 Changes in coastal salinity will subsequently alter local/regional circulation patterns, through 40 stratification, tidal interactions and other mixing processes, and hence both planktonic and benthic 41 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).

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• 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).

8 Coastal waters will also be affected by changes in the open ocean. Such influences include changes in local, 9 regional and global circulation patterns (altering shelf-edge currents and upwelling, and hence nutrient 10 supplies). The relative importance of open ocean and land-based drivers will depend on the width of the 11 continental shelf, that varies from tens to hundreds of kilometres; the size of rivers and drainage basins; and 12 the spatial variability of anthropogenic pressures.

5.3.3 Coastal Ecosystems

16 Coastal ecosystems are amongst the most diverse due to their complex geomorphology and sharp 17 physicochemical gradients, and they are highly sensitive to changes in atmospheric conditions. They are also 18 exposed to other non-climatic human drivers and their potential interactions with climate change because of 19 their close proximity to human populations (Levin et al., 2015; Diop and Scheren, 2016). All coastal 20 ecosystems either classified by their geomorphological structure (i.e., estuaries, sandy beaches, rocky reefs, 21 fjords) or foundation species (i.e., saltmarshes, mangroves, seagrass meadows, kelp forest, coral reefs) are 22 threatened by sea level rise, warming, acidification, deoxygenation and extreme weather events with 23 substantial differences in the type and level of vulnerabilities (Section 5.3.4) (Gattuso et al., 2015). While 24 some ecosystems such as estuaries (Section 5.3.3.1), sandy beaches (Section 5.3.3.5), mangroves (Section 25 5.3.3.3) and saltmarshes (Section 5.3.3.2) are more vulnerable to sea level rise and erosive processes, others 26 like rocky reefs (Section 5.3.3.7), kelp forests (Section 5.3.3.8), coral and other biogenic reefs (Section 27 5.3.3.6), are mostly threatened by warming and acidification, and are highly vulnerable to increased storms 28 and heat-weaves. In particular, saltmarshes, mangroves and coral reefs have a disproportional ecological 29 value due to their role in the protection against erosion and the reduction of exposure and sensitivity of 30 biological communities to climatic stressors such as warming (Camp et al., 2016a). 31

32 At the species level, the main biological responses to the interactive effects of climate drivers are similar to 33 those in pelagic ecosystems, including shifts in species distribution towards the poles, deeper water or 34 specific physicochemical gradients that result in species range expansion and local extinction (Section 35 5.2.3.1.1). Unique biological responses in coastal areas include the potential for inshore and inland migration 36 of benthic fauna and littoral vegetation in response to sea level rise. Consequently, the restructuring of 37 coastal ecosystems affects their functioning and services such as carbon storage, productivity and storm 38 protection (Section 5.4.1.3). Furthermore, the growing anthropogenic habitat degradation exacerbates 39 climate vulnerabilities of coastal systems. Common examples of these compounding effects is the expansion 40 of hypoxic zones enhanced by eutrophication in estuaries (Warwick et al., 2018) and low adaptability to sea 41 level rise driven by coastal squeezing due to human barriers in sandy beaches and saltmarshes (Hubbard et 42 al., 2014). 43

44 This sub-section summarizes our new understanding of ecological and functional changes that coastal 45 ecosystems are experiencing related to multiple climate and non-climatic human drivers, and their synergies. 46 Our assessment combines global evidence of biotic and abiotic responses in the short- and long-term from 47 observations, experimental studies and numerical modelling. This approach allows current ecosystem 48 alterations to be attributed to dominant climate drivers and facilitates the projection of future changes across 49 latitudinal climatic regimes.

51 *5.3.3.1* Estuaries

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52 53 Estuarine ecosystems are defined by the river-sea interface that provides high habitat heterogeneity and 54 supports large biodiversity across freshwater and subtidal ecotones (see Glossary); while marshes and 55 mudflats with algal and vascular plant producers are key features of their riverine areas. Estuaries are 56 sources of CO_2 to the atmosphere and emit an average of 0.25 ± 0.25 PgC yr⁻¹ (Regnier et al., 2013). 57 However, estuarine carbon budgets are also driven by carbon cycling within the coastal vegetated wetlands;

1 and either saltmarshes (Section 5.3.3.2), mangroves (Section 5.3.3.3) and subtidal seagrasses (Section 2 (5.3.3.4), are important areas of carbon burial (Section (5.5.1.1)). Carbon sequestration is threatened by 3 modifications on the drainage area due to e.g., urban settlements, land reclamation and dredging, which has 4 resulted in a 25-50 % loss of total estuarine carbon intertidal pools over the past century (Regnier et al., 5 2013). Accordingly, rising sea level represents one of the most important threats to estuaries, between 20-60 6 % of coastal wetlands are expected to be lost globally by 2100, depending on emission scenarios (Watson 7 and Hinojosa Corona, 2017), whereas gaining area will be possible if vertical sediment accretion occurs 8 together with lateral re-accommodation (Schuerch et al., 2018). In turn, combined effects of anthropogenic 9 activities and accelerated sea level rise increase the erosion of saltmarshes and the export of carbon and 10 nitrogen into the estuarine waters, causing eutrophication (Duarte et al., 2014). 11 12 More observational evidence since AR5 WGII (Wong et al., 2014) supports that climate change modifies the 13 estuarine gradients of salinity, turbidity and nutrients through interactive effects of sea level, precipitation, 14 temperature and estuarine circulation patterns (Robins et al., 2016; Raimonet and Cloern, 2017). The 15 interaction of sea level rise and changes in rainfall, and thereby river flows, will have a more severe impact 16 on shallow estuaries (<10 m) than on deep basin estuaries (>10 m). In a projected sea level rise of 1 m, 17 indices of vulnerability indicate that shallow estuaries will increase the tidal current amplitude by 5%, and

- 18 the energy dissipation, the vertical mixing, and the salinity intrusion by 25% (Prandle and Lane, 2015).
- 19 Meso- and macrotidal estuaries with rich sediment areas are more resilient to global climate changes than
- 20 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).
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31 The cycling between periods of drought and flood over land in combination with warming will affect the 32 physical-chemical water gradients of ocean properties along the land-sea continuum and thus the distribution 33 and performance of benthic biota. For example, anomalous salinity values exceeding the local threshold of 34 25 PSS (Practical Salinity Scale) after a dry period in Apalachicola Bay, interacts with high water 35 temperature and produces an increase of oyster mortality of 15% under warmer conditions (Petes et al., 36 2012). Conversely, droughts lead to salinization, and this condition can be intensified by climate warming 37 due to evaporation in coastal shallow waters, and by sea level rise that can produce an inward movement of 38 saltwater from the adjacent ocean (Ross et al., 2015; Zhou et al., 2017) (medium confidence). In two macro-39 tidal estuaries in southeast England, climate-driven increases in saline incursion in the long-term (Prandle 40 and Lane, 2015) will modify the distribution of the benthic fauna according to their specific salinity 41 tolerance. The relocation will be further shaped by the boundary imposed by the substratum properties, since 42 substratum type is important in providing habitat diversity, acting as a refuge and retaining food (Little et al., 43 2017).

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45 Increasing flooding during rainy periods over fertilized agriculture land will enhance estuarine 46 eutrophication and stratification, and consequently will increase the likelihood of phytoplankton blooms (Li 47 et al., 2015). Such bloom events are difficult to synchronize with local grazers (Thackeray et al., 2016) 48 leading to short-term biomass accumulation that ultimately will reduce the water quality by increasing the 49 risk of hypoxia and production of microalgae toxins (Anderson et al., 2015) (medium confidence). The 50 evidence of harmful algal blooms (HABs) has increased in coastal areas worldwide over the last 20 years 51 (Anderson et al., 2015) (see Box 5.3) in relation to interactive effects of climate warming and nutrient inputs 52 such as nitrogen and phosphate from land and wind surge (Paerl et al., 2016). Additionally, interactive 53 effects between human pollution, warming and extreme precipitation events on estuarine turbidity, dissolved 54 organic matter and oxygen concentrations, will increase the occurrence of pathogenic bacteria, e.g., Vibrio 55 species (Baker-Austin et al., 2017; Kopprio et al., 2017) (Section 5.3.2, low confidence). Likewise, in 56 shallow and microtidal estuaries, extreme river runoff increases the inflow of detritus and suspended 57 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).

4 5 In particular, expansion of oxygen-depleted dead zones in coastal areas is projected under intensification of 6 co-occurrence of climate threats and eutrophication, with warming as the main climate-related driver that 7 enhances hypoxia through a variety of physical and biological mechanisms (Breitburg et al., 2018). As the 8 oxygen solubility is greater at lower temperatures, the effect of warming will be more pronounced on high 9 latitude, temperate and arctic estuaries, where the predicted rates of warming are higher and the initial water 10 temperatures are lower. Additionally, hypoxia will be more pronounced in shallow temperate estuaries 11 because they are more responsive to warming due to their limited exchange with the open ocean (e.g., Río de 12 La Plata Estuary, Baltic Sea and Chesapeake Bay), and seasonality leads to dead zone development when 13 temperature reaches critical values in summer (e.g., Black Sea) (Altieri and Gedan, 2015). Eutrophic and 14 shallow estuaries face magnification on hypoxic conditions due to algae blooms and microbial respiration, 15 which generate CO₂ and reduce the pH (Breitberg et al., 2015). The coastal acidification related to the 16 expansion of hypoxic dead zones is of main concern for conservation of sensitive organisms (Beck et al., 17 2011; Duarte et al., 2013; Feely et al., 2016), such as the pteropod *Limacina helicina* (Bednaršek et al., 18 2017).

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20 Estuarine gradients are further influence by large-scale climate signals, such as North Atlantic Oscillation 21 (NAO), El Niño Southern Oscillation (ENSO) and Southwest Atlantic Meridional Overturning Circulation 22 (SAMOC), which have shown persistent anomalies since the 1970s associated with climate change (Wang 23 and Cai, 2013; Delworth and Zeng, 2016; García-Moreiras et al., 2018). In turn, these changes in estuarine 24 gradients affect the phenology and composition of coastal plankton, as displayed by sustained observations 25 (high confidence). For instance, at the temperate Mondego estuary (northeast Atlantic Ocean), higher 26 temperatures driven by NAO fosters the intrusion of marine, cosmopolitan zooplankton species at the 27 expense of native, estuarine species (Margues et al., 2017). Heavy rainfall events driven by ENSO, causes 28 surges in river runoff and nutrient supply in temperate estuaries from the southern hemisphere, which in turn 29 modifies the structure of phytoplankton communities (Sathicq et al., 2015; Thompson et al., 2015; Andrade 30 et al., 2016) (medium confidence). In the temperate Bahía Blanca Estuary (SW Atlantic Ocean), warmer 31 phases of ENSO mediate the emergence of water turbidity (enhanced by dredging) as a dominant driver of 32 phytoplankton phenology, and promoted the decline of chlorophyll concentration at a yearly rate of 1% 33 (López Abbate et al., 2017). Negative resonance effects from ENSO-driven droughts (i.e., increase soil 34 salinity) also occur within salt marshes, where a reduction in plant production affects direct consumers 35 (Pascual et al., 2015). In addition, ENSO exacerbates the effect of estuarine habitat modification by humans 36 (e.g., coastal infrastructure, dredging) on the mobilization of blue carbon (Macreadie et al., 2015). 37 Heatwaves associated with La Niña in Shark Bay, Australia, produced nutrient pulses from degraded 38 submerged coastal vegetation and triggered phytoplankton and bacterial blooms (Thompson et al., 2015; 39 Arias-Ortiz et al., 2018). Although these changes in ecosystem components may be attributed to climate 40 variability (Box 5.1), they provide tentative evidence of sensitivity to climate change, given the *likely* 41 intensification of large-scale climate events (e.g., ENSO) projected for 2100 (Stocker et al., 2013). 42

43 Overall, the land-sea interface confers estuaries hydrological complexity and high habitat heterogeneity. 44 Estuaries are commonly eutrophic ecosystems under intense human impacts, and are likely resilient to short-45 term environmental perturbations. However, growing long-term observational evidence since AR5 allows 46 the attribution of ecosystem modifications to climate change (high confidence). The widespread 47 observational evidence from shallow estuaries has already linked the increased levels of eutrophication and 48 hypoxic conditions to warming (*high confidence*), and projections show that this processess will be more 49 pronounced in estuaries from high latitudes (high confidence). Future scenarios of sea level rise, warming 50 and extreme precipitation events will affect the balance between river runoff and marine water intrusions, 51 thus modifying the physical and biogeochemical properties of the estuarine gradient. Shallow and microtidal 52 estuaries in those areas exposed to higher probability of flooding will be the most vulnerable (high 53 *confidence*) to experience transformation in the ecosystem functioning and services.

5.3.3.2 Salt Marshes

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1 Salt marshes are productive ecosystems in the upper intertidal zone that are characterized by salt-tolerant, 2 herbaceous vegetation. They occur in bays and estuaries from tropical to polar regimes, mostly found in mid-3 latitudes, where they support numerous animal species (Greenberg et al., 2014; Scott et al., 2014). Salt marsh 4 plants commonly define patterns of zonation, with the lower zones dominated by a few hardy pioneer genera 5 such as Salicornia, Suaeda, Aster, and Spartina, and then mixed communities of forbs and succulents at 6 higher landward positions. Invertebrates often inhabit in plant culms and roots, gaining relief from physical 7 stresses while facilitating microbially-mediated nutrient turnover via burrowing and storage via biodeposition. Despite their comparatively small area (55,000 km², in 99 countries; Mcowen et al. (2017b)), 8 9 tidal saline wetlands play disproportionately important roles in coastal carbon budgets, forming critical 10 linkages between rivers, estuaries, and oceans (Najjar et al., 2018). As a result of tidal flooding, saltmarsh 11 soils do not dry out and high levels of carbon can accumulate under anaerobic conditions. This is coupled 12 with generally low rates of methane emission, which is strongly limited in saline marshes (high confidence) 13 (Poffenbarger et al., 2011; Martin and Moseman-Valtierra, 2015; Kroeger et al., 2017). The carbon burial 14 rate of salt marshes is comparable to mangroves due to high belowground productivity (McLeod et al., 15 2011). Together with seagrass meadows and mangroves, salt marshes represent 'blue carbon' habitats (see 16 Sections 5.3.1.1.2, 5.4.1.1) (McLeod et al., 2011). 17

18 Nearly half of the pre-industrial, natural extent of salt marshes has been lost, due to drainage, agriculture, 19 coastal settlement, hydrological alterations and reductions in sediment supply (Adam, 2002; Wang et al., 20 2014; Kroeger et al., 2017; Li et al., 2018). Sea level rise is a major threat to salt marshes; salt intrusion 21 results in community restructuring (Janousek et al., 2017) and changing above-and below-ground 22 productivity (McLeod et al., 2011; Watson et al., 2017). Plants with low tolerance to flooding are especially 23 vulnerable. For example in New England salt marshes, Spartina patens is being replaced with flood-tolerant 24 S. alterniflora or converted to pools of unvegetated standing water (Raposa et al., 2017), while S. patens 25 specialized foragers are rapidly declining (Johnson and Williams, 2017). A similar expansion of the flood-26 tolerant S. alterniflora in response to rising sea level occurred in a temperate estuary in the southwestern 27 Atlantic, Argentina. Over 33 % of saltmarsh area covered by Sarcocornia perennis and 6 % of halophytic 28 shrub-like steppes dominated by Cyclolepis genistoides, Allenrolfea patagonica and Atriplex undulata, have 29 been lost in the last 40 years and replaced by either mudflats or S. alterniflora marshes (Pratolongo et al., 30 2013). 31

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 Bahía 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).

40 The ability of salt marshes to withstand *in situ* sea level rise depends on the development of new soil by the 41 external supply of mineral sediments and organic accretion by local biota that builds marsh elevation. At 42 many localities, critical organic accretion rates are declining due to decreases in plant productivity (from 43 stress of inundation) and increases in plant and microbial respiration rates (i.e., from warming), thereby 44 preventing marsh ability to keep pace with rising sea levels (medium confidence) (Carey et al., 2017; Watson 45 and Hinojosa Corona, 2017). This negative climate impact is exacerbated in cases where anthropogenic 46 barriers cause the 'coastal squeeze', which impedes the upward migration of less flood-tolerant plant and 47 animal species, causing their loss (*high confidence*). In several salt marsh plant species in the northeast 48 Pacific, however, inundation does not seem to be an important factor affecting litter decomposition 49 (Janousek et al., 2017). Positive or negative climate feedbacks can result in salt marshes in response to sea 50 level rise (Section 5.3.1.1.2) depending on the supply of suspended sediments, tidal ranges, subsurface 51 processes (e.g., compaction) and above and belowground plant productivity. 52

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

1 (Langley et al., 2009; Kirwan and Mudd, 2012). However, in models of rising temperatures and higher sea 2 levels, Spartina alterniflora marsh responds initially with increasing accretion rates and carbon burial, 3 therefore having negative feedbacks on climate, while by 2075, larger carbon pools and dominant root zone 4 processes slow down rates of accretion and increase decay rates, switching to a positive climatic feedback 5 (Kirwan and Mudd, 2012). Increased decay rates in response to warming are attributed to increased 6 microbial and fungal activities (Kirwan and Blum, 2011). 7 8 Local stressors such as eutrophication, overfishing, and species invasions, accelerate the deterioration of 9 saltmarshes. In New England salt marshes, for example, overfishing has led to uncontrolled grazing by a 10 herbivorous marsh crab, Sesarma reticulatum, that is currently degrading Spartina alterniflora zones. This 11 trophic cascade synergistically interacts with rising sea level because water-induced softening of inundated 12 marsh peats permits new expansion of the crab into high marsh zones (medium confidence) (Crotty et al., 13 2017). 14 15 Excess nutrients may also reduce marsh resistance to sea level rise. Experimental nitrogen additions 16 increased primary productivity and rates of sediment accretion in sediment-rich marshes (Morris et al., 2002) 17 but caused low marsh collapse in sediment-poor marshes over the course of decades (Deegan et al., 2012). In 18 the latter, excess nitrogen increased S. alterniflora shoot biomass at the expense of belowground biomass 19 and also increased decomposition rates, both of which decrease the critical peat accumulation needed to 20 maintain salt marsh elevation. Because salt marshes remove river-borne nutrients from waters entering 21 coastal areas (Negrin et al., 2016), marsh area losses induced by rising sea levels are *likely* to enhance the 22 nutrient input impacts described above. 23 24 Invasive plant species are *likely* to be favored by multiple factors associated with climate change. In North 25 America, one of the most aggressive invasive species is the cosmopolitan *Phragmites australis* which is 26 facilitated by excess nitrogen. As a C3 species, P. australis displays strong potential to expand under rising 27 CO₂ levels (Mozdzer and Megonigal, 2012) and is evading many management strategies for its erradication

(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).

35 Hydrological alteration is a key stressor to salt marshes worldwide; however, this factor is frequently 36 reversible. Tidally-restricted wetlands comprise roughly 20% of coastal wetlands on the Atlantic Coast of 37 the US. In this region, it is *likely* that significant permanent reductions in methane emissions, and enhanced 38 carbon storage can be achieved by restoring tidal flows (see Section 5.4.1) (Kroeger et al., 2017). Such 39 actions also increase the capacity of the ecosystems to maintain elevation relative to sea level. The scale of 40 benefits depends on how completely tidal restrictions can be reversed as well the duration of prior alteration 41 (Emery and Fulweiler, 2017). Areas for future research include the potential for marsh plants to locally 42 remediate ocean acidification (via CO₂ uptake) or deoxygenation (through O₂ production/transport), as is 43 currently considered for seagrasses (Garrard et al., 2014). Seasonal extremes may, however, be increased, 44 reducing potential benefits (Sabine, 2018). 45

46 Substantial evidence supports with high confidence that salt marshes will continue to reduce their area in 47 response to sea level rise in the coming century, especially if landward migration is further constrained by 48 human modification of shorelines. Plants with low tolerance to flooding are particularly vulnerable and they 49 can be locally extirpated. The loss of vegetated coastal ecosystems causes a reduction in carbon storage with 50 positive feedbacks to the climate system (high confidence). Furthermore, the flooded area can become a 51 mudflat or be colonized by more tolerant, invasive species (e.g., Spartina alterniflora, C3 plants), whose 52 expansion is favored by combined effects of warming, rising CO2 and nutrient enrichment (medium 53 confidence). These shifts in community structure denote severe consequences in the ecosystem functioning 54 and services.

- 55
- 56 5.3.3.3 Mangroves 57

SECOND ORDER DRAFT

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).

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While over-exploitation of mangrove wood products, and conversion of mangrove areas into other land uses 9 are the major mangrove threats in the world, climate change will *likely* exacerbate mangrove loss (Table 10 5.2). Mangrove ecosystems are particularly vulnerable to rising sea levels, increasing temperatures, and 11 anoxic conditions brought about by sporadic sedimentation due to extreme rainfall (Alongi, 2008; Bosire et 12 al., 2008; Gilman et al., 2008). Hydro-geomorphological settings play important role in mangrove responses 13 to sea level rise (Sasmito et al., 2015; Eguiguren-Velepucha et al., 2016), where rising sea levels 14 arepredicted to increase flooding of the low-lying coastal areas and drawn mangroves (high confidence). In 15 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 16 17 contrary, mangroves located in small islands, with coastal development causing squeezing, lack of rivers, 18 groundwater extraction, steep topography and sediment-starved areas, are the more vulnerable to sea level 19 rise. Thus, mangroves ecosystems are initially resilient (Ward et al., 2017) to increased storms and sea levels 20 under senario RCP2.6 throughout the 100 years projection period, but only up to mid century under RCP8.5 21 (Sasmito et al., 2015).

22 23 Increased salinity caused by sea-level rise may also result in decreased productivity and stunted growth in 24 vulnerable species. The expected increase in mangrove growth under high CO₂ conditions has been 25 experimentally observed with an unexpected change in salinity optima, shifting the ecological niche of the 26 species towards more saline conditions (Reef et al., 2015). These physiological treats may imply changes in 27 distributional patters of the species. Likewise, the poleward expansion of mangroves and their encroachment 28 into sub-tropical saltmarshes in response to warming and rising sea level (medium confidence) (Saintilan et 29 al., 2014) is causing alterations in ecosystem functioning and services (Kelleway et al., 2017a) (Section 30 5.4.1.3.2).

Mangroves operate in the C_3 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).

39 Inspite of the partial positive effects of rising CO_2 on mangrove growth rates, the overall effects of climate 40 change on mangroves is negative (high confidence, Table 5.2) and will also negatively impact on coastal 41 protection, agriculture, aquaculture and forestry activities (Table 5.2). Fringe mangroves are more vulnerable 42 to rising sea level than basin magroves, as well as those under microtidal regime with low soil accretion 43 capacity (high confidence). Mangrove forestry activities will for instance be affected by changes in 44 phenology, aridity, salinity and through direct disruption of specific sites by sea level rise and storms. Thus, 45 the close monitoring of the status and responses of mangrove forests to climate change for early indication of 46 change is important for appropriate management interventions. 47

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

Factor	Processes Affected	Impacts	References
Rising sea level	Forest health	Forest mortality, dieback	(Ellison, 1993;
	Forest productivity	from the seaward edge	Semeniuk, 1994; Cahoon
	Recruitment	Migration landward, but	et al., 2006; Gilman et
	Inundation period	dependent on sediment	al., 2007; Gilman et al.,
	Sedimentation rates	inputs, topography and	2008; Soares, 2009;
		human modifications	Ellison and Zouh, 2012)

Chapter 5

Extreme storms	Forest productivity Recruitment Sedimentation rates	Forest damaged or destroyed Ground elevation change Erosion or sediment smothering	(Dahdouh-Guebas et al., 2005; Alongi, 2008; Yanagisawa et al., 2010)
Increased waves and wind	Sedimentation rates Recruitment	Changes in forest cover depending on whether coasts are accreting or eroding	(Semeniuk, 1994)
Increased air and sea temperature	Respiration Photosynthesis Forest production	Reduced productivity at low latitudes and increased winter productivity at high latitudes.	(Clough and Sim, 1989; Cheeseman et al., 1991; Cheeseman, 1994; Cheeseman et al., 1997)
Enhanced CO ₂	Photosynthesis Respiration Biomass allocation Forest productivity	Increased productivity, subject to limiting factors of salinity, humidity and nutrients Soil elevation gain	(Snedaker, 1995; Farnsworth et al., 1996; Ball et al., 1997; Langley et al., 2009)
UV-B radiation	Morphology Photosynthesis Forest productivity	Minor	(Lovelock et al., 1992; Day and Neale, 2002; Caldwell et al., 2007)
Increased rainfall	Sediment inputs Ground water Salinity	Reduced sediments and relative subsidence Migration landward Reduced ground water Reduced photosynthesis Reduced productivity Species turnover Reduced diversity	(Smith and Duke, 1987; Rogers et al., 2005; Whelan et al., 2005)
Reduced humidity	Photosynthesis Forest productivity	Reduced productivity Species turnover Reduced diversity	(Clough and Sim, 1989; Cheeseman et al., 1991; Cheeseman, 1994; Ball et al. 1997)

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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).

16 Responses of seagrasses to climatic stressors change among species/genera, according to their life-history 17 strategies (colonizing, opportunistic and perennial species) and climatic affinities (Rasheed et al., 2014; 18 Kilminster et al., 2015; O'Brien et al., 2017; York et al., 2017). Certain seagrass species are expected to 19 colonize new areas, i.e., 'winners', while others are expected to retract, i.e., 'losers'. Overall, warm-temperate 20 regions, such as Western Australia, are the most vulnerable to lose seagrass area and the associated habitat 21 complexity (Hyndes et al., 2016). Synergistic interactions of climatic stressors, e.g., heat waves and 22 increased turbidity and nutrient levels associated with floods, have been shown to cause drastic changes in 23 the composition and biomass of co-occurring seagrass species, potentially eroding provision of ecosystem 24 services (high confidence) (Fraser et al., 2014; Nowicki et al., 2017; Lin et al., 2018). An example of a 25 seriously damaged seagrass ecosystem following a marine heatwave is in Shark Bay, Australia, which has 26 lost 36% of the meadows dominated by Amphibolis antarctica and to a minor extent by Posidonia australis 27 (Arias-Ortiz et al., 2018). While Shark Bay seagrasses occupy an area of ~4% of the Australia's vegetated 28 coasts, they account for a 9% of the C sequestration by seagrasses, mangroves and tidal marshes. The losses

- 1 of seagrasses due to thermal stress impacts reduce the sediment C stocks and enhance CO₂ emission due to 2 oxygenation of the eroded sediments. 3 4 In the iconic Mediterranean seagrass meadow formed by Posidonia oceanica, warming causes a number of 5 negative indirect effects (medium confidence) such as increased toxicity of chemical hazards such as 6 sulphide (García et al., 2013), and intense competition and predation by the arrival of tropical species 7 (Pergent et al., 2014). It also reduced the fitness of seedlings by increasing necrosis and susceptibility to 8 consumers and pathogenic pressure, while reducing establishment potential and nutritional reserves (Olsen et 9 al., 2016b; Hernán et al., 2017). Despite temperature-driven flowering of P. oceanica has been advocated as 10 way of increasing resilience to warming (Ruiz-Frau et al., 2017), severe habitat loss (70%) of this endemic 11 species is projected by 2050 and potential functional extinction by 2100 under RCP8.5 climate senario. 12 Similarly, warming will lead to significant reduction of Cymodosea nodosa meadows (46%) in the 13 Mediterranean, which will be compensated in part by future expansion into the Atlantic (Chefaoui et al., 14 2018). 15 16 Under increased CO₂ levels in the future, seagrasses are predicted to physiologically perform better, as 17 photosynthetic rates increase (Koch et al., 2013; Borum et al., 2015). However, there is no consensus on 18 long-term consequences of increased CO2 on seagrass carbon budgets and the stoichiometric composition of 19 seagrasses (Russell et al., 2013). Acidification in the future, moreover, will not counterbalance lethal effects 20 of warming on the physiological fitness of certain temperate species, such as Zostera noltii (Repolho et al., 21 2017). 22 23 Responses of seagrasses to anthropogenic-mediated perturbations (e.g., CO₂ increments and fertilization)
- 24 further change the feeding behaviour of their consumers, causing facilitation or suppression of herbivory 25 across functional groups (medium confidence) (Tomas et al., 2015). Importantly, the intensity of herbivory 26 over seagrasses is expected to increase with global warming, particularly in temperate areas, because of the 27 migration of tropical herbivores into temperate seagrass-dominated environments (Hyndes et al., 2016). In 28 turn, as herbivores play a key role in modulating the biomass of plant communities, their intense activity 29 affects the provision of ecosystem services (Scott et al., 2018). For instance, intense activity of tropical 30 herbivores reduces the production of refractory organic matter and thus the bulk carbon sequestration (Heck 31 et al., 2015; Hyndes et al., 2016). 32
- 33 Understanding the effects of climatic stressors require holistic approaches to encapsulate the wide range of 34 processes that may affect the fitness of seagrasses and the structure and functioning of the ecosystems they 35 support. In spite of these complexities, there is a high confidence that warming and the introduction of 36 tropical consumers in higher latitudes will continue to reduce seagrass area. Tropical seagrass species with 37 their low dispersal ability and their high exposure to above-normal temperatures are the most vulnerable to 38 local extinction in the future decades (medium confidence). This will affect the overall ecosystem diversity 39 by reducing the nursery area for fish species and the available habitat for species that live on tropical 40 seagrasses.

42 5.3.3.5 Sandy Beaches

- 43 44 Sandy beaches embrace the world's longest coastline ecosystems and play an important socio-economical 45 roleby providing area for recreation and tourism, and living or non-living resources for commercial 46 extraction. As assessed in AR5 WGII (Wong et al., 2014), the worldwide pervasive coastal squeezing, which 47 constricts sandy beaches between rising sea levels and expanding urbanization, will continue to limit the 48 ability of these ecosystems to compensate climate threats (*high confidence*). One of the most important 49 compensatory responses of sandy beaches to erosive processes is the relocation of sediments that facilitates 50 profile migration, but this response is constrained by the reduction of effective beach area. Along with sea 51 level rise, severe erosive events such as storms surges, flooding and onshore winds, challenge the 52 conservation of sandy beaches, accelerating dune scarping with vegetation loss (Castelle et al., 2015). Also, 53 excessive precipitation driven by climate change (Westra et al., 2013) will accentuate erosion through the 54 exportation of sand and therise in groundwater levels.
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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

1 temperature produced a constant poleward shift of the warm water front (20°C isotherm) at a rate of ca. 9 km 2 v^{-1} (Ortega et al., 2016). In addition, an increase in speed and duration of onshore southern winds and storm 3 surges (Escobar et al., 2004; Bischoff, 2005; D'onofrio et al., 2008; Ortega et al., 2013), generated high 4 energy waves (Codignotto et al., 2012). This climate-driven intensification of waves modified the 5 morphology of sandy beaches through the augmentation of the swash width and the erosion rates (Ortega et 6 al., 2013; Gutierrez et al., 2016). There is high confidence that this erosive processes along with the current 7 pace of sea level rise, keep causing morphological changes, suggesting a reduced resilience of the subaerial 8 profile of sandy beaches. For the Pacific Ocean basin, a 33-year observational analysis for 48 beaches 9 showed that erosion across Pacific coasts varies with ENSO (Barnard et al., 2015). Independently from sea 10 level rise, the projected increase in the frequency of extreme ENSO events warns on the acceleration of 11 flooding and erosion rates in opposite coastal sides of the Pacific Ocean basin (Barnard et al., 2015). 12 13 Evidence from long-term observations and empirical studies of South American sandy beaches allows 14 attributing with *high confidence*, the reorganization of macrofauna to changing climate (Table 5.3). This is 15 supported by global meta-analyses that disclose consistent responses across taxa and geographical regions 16 (McLachlan and Defeo, 2017). Cool-adapted clam species are especially sensitive to warming, and are 17 already experiencing mass-mortality events with low recovery denoting a poor adaptive capacity of beach-18 inhabiting populations to climate change. Mass mortalities have profound socio-economic implications by 19 limiting the exploitation of marine resources and recreational activities such as artisanal seafood harvest 20 (Defeo et al., 2013; Aburto et al., 2014; Gianelli et al., 2015; Turra et al., 2016). Opposite demographic 21 responses may also occur in the Pacific and the Atlantic coasts of South America, where more tolerant filter 22 feeding macrofauna are experiencing a tropicalization of species (*high confidence*) (Table 5.3). 23 24 In the east coast of Australia, a similar migration in response to climate was documented in adults of the 25 ghost crab Ocypode cordimanus, which expanded poleward: 270 km south; 0.9° of latitude (Schoeman et al., 26 2015). This distributional shift is *likely* related to the regional ocean warming and the displacement 27 southward of the surface isotherms along the coast at a rate of 20–50 km decade⁻¹. Similarly, in California, 28 the upper beach isopod Tylos punctatus is shifting its northern range limit southward in response to intense 29 expansion of human population and habitat degradation (Hubbard et al., 2014). This species will be under 30 higher risk with rising sea level scenarios in these narrow beaches, where isopoda will have little possibility 31 for landward retreat. In fact, 31% to 67% of southern California beaches are predicted to become completely

32 eroded by 2100 under SLR scenarios of 0.93-2.0 m (Vitousek et al., 2017). Simultaneous with 33 biogeographic changes, benthic crustaceans including isopods, crab and amphipod species display a 34 consistent reduction in body size towards warmer latitudes in sandy beaches from eastern Pacific (USA and 35 Chile) (Jaramillo et al., 2017). The fact that these organisms inhabit wide-ranging tidal zones and display 36 contrasting feeding modes and life histories, and that they show a common response in both hemispheres, 37 strength the reliability of body size as a metric of temperature effect (Section 5.2.3.1.1). This evidence 38 suggests that sea surface warming in global sandy beaches will negatively affects trophic webs, with 39 detrimental consequences for ecosystem services provision.

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42 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).

14	Addapted from McLachlan and Defeo ((2017). The numbers indicate the source references.	

Lines of evidence	Attribution process	References
Long-term observations and resurveys	 Mass mortality events of the clam species <i>Mesodesma donacium</i> 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)¹, and <i>M. mactroides</i> in the Atlantic , following increasing SST and isotherms moving poleward² across its southern range. Sowthward shift of the isopod <i>Tylos punctatus</i> in the Pacific (California, USA) linked to expansion of human population and habitat degradation, with high risk under sea level rise and coastal squeezing⁴. 30 years of systematic sampling in Uruguay² and resurveys in Brazil and Argentina³ across the distribution range of the clam 	 Riascos et al. (2009) Ortega et al. (2013); Ortega et al. (2016) Herrmann et al. (2009); Adams et al. (2011); Herrmann et al. (2011) (Hubbard et al., 2014)

Fin	gerprints	<i>M. mactroides</i> document declines and local extirpations related to increasing SST and associated factors. Population extirpations along the northern (trailing) range edge and lowest levels of impact along southern (leading) range boundaries ^{1, 2, 5} , uniquely consistent with regional warming in the Pacific and Atlantic oceans and not with local human-caused stresses.	5. Riascos et al. (2011) 6. Fiori et al. (2004)
Meta-analyses: global coherence of responses across taxa and regions		Mass mortalities of yellow clams occurred during warmer seasons throughout its entire distribution range ⁶ , concurrently with accelerating SST warming. Mass mortalities observed across Pacific and Atlantic sandy beaches of South America, particularly during El Niño events and recorded oceanographic shifts ^{1, 2} . Drastic changes in the composition of the suspension-feeder assemblages, with coherent responses across taxa and regions. <i>Mesodesma</i> clams were virtually extirpated at their northern range edges, being partially replaced by <i>Donax</i> clams and <i>Emerita</i> mole crabs (tropical affinities) in both Pacific ^{7, 8} and Atlantie ^{9, 10, 11} sandy beaches. Impoverished macrofaunal communities and <i>Mesodesma</i> species never reached biomasses at pre-mortality levels ^{6, 10, 12}	 Riascos et al. (2011) Arntz et al. (1987); Arntz et al. (2006) Celentano and Defeo (2016) Dadon (2005) Herrmann et al. (2009) Lercari et al. (2018)
		Parasites were found in clams, and also necrosis in gills and stomachs, during mortality events in the Atlantic (Argentina ⁵ and Uruguay ²) and in the Pacific during strong El Niño events ⁷	
		Body downsizing towards warmer latitudes was consistently observed for the guilds of intertidal suspension feeders and benthic crustaceans including isopods, crab and amphipod species in sandy beaches from the Atlantic ^{2, 9} of South America and the Pacific (USA and Chile) ^{1, 5, 12} .	
Exp	periments	Controlled in-vitro experiments showed that <i>D. obesulus</i> (tropical affinities) was tolerant to El Niño temperatures, whereas <i>M. donacium</i> (temperate) was sensitive to both El Niño and La Niña extreme conditions, with sublethal and lethal effects ¹³	12. (Jaramillo et al., 2017)13. Carstensen et al. (2010)

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11 12 Additionally, the interaction between climate warming and eutrophication are increasing the occurrence of 13 harmful algal blooms (HABs, see Box 5.3), with tropical toxic species being increasingly represented in 14 coastal subtropical ecosystems (Martínez et al., 2017). Exploitation of beach clams is often constrained by 15 HABs, which can cause mass mortalities or render clams unsafe for human consumption (Gianelli et al., 16 2015). Eutrophication has also led to the mass development of mats of drifting algae along sandy beaches 17 that deteriorated the environment and the resident macrofauna (Quillien et al., 2015; Quillien et al., 2016). 18 Detached algae invade the water column and are able to increase their nutrient supply, building up large 19 biomass and forming massive green (Ulva) and golden (Sargassum) tides (Smetacek and Zingone, 2013). 20 The frequency, intensity and periodicity of these mass stranding events have been increasing worldwide (Ye 21 et al., 2011) and have triggered high economic losses associated with beach deterioration and the consequent 22 impacts on tourism and fisheries.

Climate events such as El Niño also affect pelagic communities inhabiting surf zones of sandy shores. In the

zone of sandy shores have been related to extreme El Niño 1997/1998 (Odebrecht et al., 2010; Odebrecht et

southern east coast of Brazil, long-term changes in phytoplankton composition and abundance in the surf

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,

reduction in beach width (60%), abundance of macrophytes, species richness of shorebirds and individual

Californian beaches suffered drastic changes in response to the 1997/98 El Niño event including the

size of macroinvertebrates (Revell et al., 2011).

2 Habitat modifications due to sea level rise, erosion and coastal squeeze are also reducing the habitat 3 accessibility for macrofauna. For instance, turtle nesting is being threaten by the reduction of suitable sandy 4 beach area. Moreover, because sea turtles have temperature-dependent sex determination, warming could 5 also affect their reproductive output (Hays et al., 2003). As incubation temperatures near lethal levels, 6 growth and survival rates of turtle populations will tend to decrease in the long-term (medium confidence) 7 (Laloë et al., 2016; Laloë et al., 2017). Shorebird richness and abundance may decrease in the long-term due 8 to a combination of habitat loss, decreased accessibility because of sea-level rise and reduced prey 9 availability, particularly when the upper shore and sand dunes are modified by engineering interventions that 10 place armoring structures on beaches (Dugan et al., 2008). Sea-level rise is expected to affect suitable 11 breeding sites for birds, particularly in the case of long coastlines with extensive low-lying coastal areas. 12 Similarly, tropical cyclones (Convertino et al., 2011) and ENSO events (Hubbard and Dugan, 2003), which 13 may become more extreme due to climate change (Cai et al., 2014), will tend to decrease nesting and 14 shorebird abundance (medium confidence).

15 16 Overall, warming is promoting a range redistribution of macrofauna in sandy beaches with consequences for 17 food webs, ecosystem structure and function. Cold-water species are highly vulnerable to mass-mortality 18 when exposed to seawater warming (high confidence). The considerable reduction of habitable area driven 19 by coastal squeezing implies a critical reduction on the adaptive capacity of macrofauna and other animals 20 such as turtles and birds (high confidence). Projections of future sea level rise entails a decrease on the 21 buffering capacity and lower recovery potential of these ecosystems to extreme climate signals such as El 22 Niño events (high confidence). 23

24 *5.3.3.6 Coral Reefs* 25

26 Anthropogenic activities have resulted in major impacts on shallow water tropical coral reefs that are 27 summarized by wide range of classes of drivers such as overfishing, nutrient enrichment, climate change and 28 ocean acidification, and all eroding reef resilience (IPCC AR5 WGII). Climate change further drives ocean 29 warming, which impacts biological and ecological reef processes, triggers large-scale coral bleaching events, 30 and fuels tropical storms. Ocean acidification impacts overall physiological traits, slows reef calcification 31 and growth, alters competitive interactions, and impairs population replenishment (Anthony, 2016). It is 32 further noted that either one of the stressors have multiple, sometimes conflicting effects on functionally 33 similar groups of reef species and their interactions (Harborne et al., 2017). The literature assessments 34 (Munday et al., 2013; Ban et al., 2014; McClanahan et al., 2014; Rinkevich, 2014; Rinkevich, 2015a; Roff et 35 al., 2015; Wolff et al., 2015; Anthony, 2016; Gunderson et al., 2016; Duvat et al., 2017; Gunderson et al., 36 2017; Harborne et al., 2017) point to negative consequences of reef ecosystems to warming, ocean 37 acidification, and sea level rise (high confidence). Seawater warming results in bleaching and mass 38 mortality, reduces coral growth rates and leads to changes in host symbiont combinations and physiologies, 39 affecting coral species distribution changes, and causes phase shifts to algal dominated reefs (high 40 confidence). Ocean acidification directly influences coral calcification and can lead to community changes, 41 also enhancing reef dissolution and bioerosion (high confidence). The rate of sea level rise (primarily noticed 42 in small reef islands) may surpass the ability of the coral reefs to grow fast enough to keep up (low 43 confidence) (Brown et al., 2011; Perry et al., 2018). Reefs are further exposed to increased indirect impacts 44 such as enhanced storm intensity, turbidity and increased runoff from the land (including the impacts of 45 sedimentation and human derived toxicants) due to extreme precipitation events (reviews above) (high 46 confidence). 47

48 Globally, coral reef communities are *likely* to change in the future through different modes of actions (IPCC 49 AR5 WGII). Many taxa respond differently to global threats, such as increased sea water temperatures and 50 enhanced ocean acidification (Rodolfo-Metalpa et al., 2011; Edmunds et al., 2013; Munday et al., 2013; 51 Takahashi and Kurihara, 2013; Wittmann and Poertner, 2013; Palumbi et al., 2014; Kavousi et al., 2015; 52 Putnam et al., 2016; Ramajo et al., 2016; Krueger et al., 2017) (high confidence). In general, multiple 53 stressors act in tandem to increase the risk of species population declines or extinction and such multiple 54 stressors are exacerbated directly or indirectly by climate change impacts (Brook et al., 2008). It includes 55 indirect impacts on physiology and behaviors (Gunderson et al., 2017) and other impacts, such as the 56 decrease in water quality with the increased sedimentation and nutrient concentrations (Fabricius, 2005) 57 (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.

6 A wealth of empirical data (Wilson et al., 2006; Halpern et al., 2008; Edwards et al., 2011; Harborne et al., 7 2017) attest for a slow recovery of coral reefs resulting from repeated disturbance events (*high confidence*). 8 For example, frequent warming events or a cluster of major storms develop into dependent stressors over 9 time in those cases where the reef system's health that is emerged from a specific impact, influences the 10 vulnerability to, or recovery from, the subsequent event. The cumulative and synergetic impacts of various 11 climate change drivers and anthropogenic impacts are also revealed with the increased rates of coral 12 bleaching vulnerability under nutrient enrichment (Carilli et al., 2009) and results that connect ocean 13 acidification with other stressors such as coral eating organisms (De'ath et al., 2012; Duvat et al., 2017; 14 Harborne et al., 2017).

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16 In contrast, the literature (DeBiasse and Kelly, 2016; Gibbin et al., 2017; Wall et al., 2017; Camp et al., 17 2018; Donelson et al., 2018; Drake et al., 2018; Veilleux and Donelson, 2018) attest that corals, as other 18 marine organisms, possess the cellular, physiological and molecular machineries that help compensating for 19 the effects of global change (*medium confidence*). As examples are species that are less impacted by either 20 ocean acidification or highlighted temperature (Cornwall et al., 2018; Gintert et al., 2018). These outcomes 21 and the existing critical gaps in our understanding of the cumulative global change impacts on coral reefs 22 point to the increased uncertainty in predicting future responses by reef corals to climate change (high 23 confidence) (Schulz et al., 2013; McClanahan et al., 2014; Mumby and van Woesik, 2014; Pandolfi, 2015; 24 Folkersen, 2018). 25

26 Furthermore, there are clear regional differences for reef vulnerability (high confidence). On the smaller 27 scales, deep reef ecozones are generally less exposed to disturbances (e.g., bleaching, storms, sedimentation, 28 tourism impacts, floods from terrestrial habitats) and could therefore constitute an important refuge to some 29 shallow habitats following disturbances, such as of larval supply (Bridge et al., 2013; Thomas et al., 2015) 30 (medium confidence). Same conclusion is assigned to mesophotic reefs (found in tropica/subtropical regions 31 at 30-150m depth) (Lindfield et al., 2016; Smith et al., 2016c; Bongaerts et al., 2017) (low confidence). Even 32 lagoon and forereef areas of the same reef site may differ significantly in reef vulnerability to global change 33 impacts like water temperatures and bleaching (Tkachenko and Soong, 2017) (high confidence). On the 34 larger scales, in areas like the eastern tropical Pacific Ocean (Smith et al., 2017), the Caribbean (Chollett and 35 Mumby, 2013), the Red Sea (Fine et al., 2013; Osman et al., 2017a), the Great Barrier Reef, Australia 36 (Hughes et al., 2010; Morgan et al., 2017), and more, there are areas or habitats that avoid stress at spatial 37 scales of >100 km, areas that were never bleached or support corals that are more resistant to stress or with 38 the capacity to recover rapidly (medium confidence). Regional differences for reef vulnerability is further 39 implied on large scale latitudinal gradients (van Hooidonk et al., 2014) and on the global reef system 40 analyses (van Hooidonk et al., 2013; Heron et al., 2016; Langlais et al., 2017; McClenachan et al., 2017) 41 (high confidence).

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43 Almost all major reef systems that are vulnerable to climate change (IPCC AR5 WGII) (high confidence). 44 An example of reef vulnerability is the Coral Triangle, the global pinnacle of tropical coral biodiversity, 45 which has experienced thermal stress followed by extensive coral bleaching and coral mortality events are 46 directly associated with extremes in the El Niño-Southern Oscillation (ENSO) (i.e., 1998 and 2010). This 47 assessment further supports the conclusion of SR 1.5 °C that coral reefs are projected to decline by 70-90% 48 at 1.5°C atmospheric warming above pre-industrial level (high confidence) with larger losses (>99%) at 2°C 49 (very high confidence). Above all, the coral systems in tropical small islands are at major risk of being 50 already severely affected by current climate change impacts, including both extreme events and gradual 51 environmental changes, such as sea level rise (Duvat et al., 2017; Harborne et al., 2017) (high confidence).

53 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.

3 4 Intertidal rocky reefs are especially vulnerable to warming, due in part to the potential for extreme heat 5 exposure during low tide emersion. Increased thermal stress during emersion associated with climate change 6 is expected to cause the upper vertical limit of rocky intertidal species to shift lower (high confidence). 7 Coupled with sea level rise and temperature-induced increases in predation by consumers (Sanford, 1999), 8 whose distributions are associated with seawater temperatures, constrictions of the vertical habitat available 9 to rocky intertidal communities are expected (Harley, 2011). While previous studies have documented a 10 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; Mieszkowska et al., 2006; Lima et al., 2007), 11 12 local extinctions at the equatorial or warm edge of species ranges are increasingly being attributed to climate 13 change (high confidence) (Yeruham et al., 2015; Sorte et al., 2017). Extreme heat waves are increasingly 14 expected to cause mortality among rocky reef species (Gazeau et al., 2014; Jurgens et al., 2015) and the 15 subsequent declines or losses in ecologically important species can have cascading effects on the whole 16 intertidal community and the services it provides (Gatti et al., 2017; Sorte et al., 2017; Sunday et al., 2017). 17 The loss of thermal refugia associated with continued warming could exacerbate the impacts of heat stress 18 on rocky intertidal communities (Lima et al., 2016). At the same time, ocean acidification is expected to 19 decrease the net calcification (high confidence) and abundance (medium confidence) of numerous rocky 20 intertidal and reef-associated species (Kroeker et al., 2013), and the dissolution of calcareous species has 21 already been documented in rocky inter-tidal tidepool communities (Kwiatkowski et al., 2016a). Moreover, 22 the effects of ocean acidification on vulnerable species may be exacerbated by extreme thermal stress 23 associated with heat waves (Rodolfo-Metalpa et al., 2011; Gazeau et al., 2014). Recent experimental and 24 field studies, however, have demonstrated the importance of food resources in mediating the effects of ocean 25 acidification on vulnerable rocky reef species (Ciais et al., 2013; Ramajo et al., 2016), suggesting that 26 species' vulnerability to ocean acidification may be most pronounced in areas of high heat stress and low 27 food availability (medium confidence) (Kroeker et al., 2017). There is increasing evidence that the 28 interactions between multiple climate drivers will determine species vulnerability and the ecosystem impacts 29 of climate change (Hewitt et al., 2016).

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31 Examples from naturally acidified rocky reef ecosystems suggest ocean acidification will simplify algal-32 dominated rocky reef ecosystems, due to an overgrowth in weedy algal species, a reduction in biodiversity, 33 and a reduction in the abundance of calcareous species (medium confidence) (Kroeker et al., 2013; Linares et 34 al., 2015). Reductions in the abundance of calcareous herbivores that historically have created space for 35 more rare and competitively inferior species by grazing the dominant algae, are expected to contribute to the 36 overgrowth of fleshy macroalgae on rocky reefs (Baggini et al., 2015). This shift towards macroalgae is 37 associated with a simplification of the food web at lower trophic levels (Kroeker et al., 2011), but the 38 emergent effects of ocean acidification on higher trophic levels (e.g., demersal fish) are less clear. 39

40 At a local scale, warming and ocean acidification are expected to change energy flows within rocky reef 41 ecosystems. Experiments have demonstrated that both warming and ocean acidification may boost primary 42 productivity at a local scale, which could lead to increased bottom-up productivity in some cases 43 (Goldenberg et al., 2017). However, increased metabolic demands and consumption of predators under 44 warmer temperature increase the strength of top-down control, which incease predation mortalities of 45 herbivores and eliminate enhanced production from increased bottom-up effects (Goldenberg et al., 2017; 46 Kordas et al., 2017). Increasing evidence suggests that ocean acidification could also increase species 47 energetic costs and grazing rate of herbivores, which could boost ecosystem resistance to shifts associated 48 with increased primary productivity (Ghedini et al., 2015). Although these increasingly complex 49 experiments have highlighted the potential for species interactions to mediate the effects of climate change, 50 our understanding of the effects on intact, functioning ecosystems is more limited. Despite predictions for 51 increased production and herbivory with warming and acidification, a study of a more complex food web 52 revealed an overall reduction in the energy flow to higher trophic levels and a shift towards detritus-based 53 food webs (Ullah et al., 2018). 54

55 Overall, rocky reef ecosystems either dominated by calcareous mussels or algae are highly sensitive to ocean 56 warming and acidification and extreme heat exposure during low tide emersion. Since AR5, increasing field 57 and experimental evidence allow the attribution of species relocation towards lower intertidal levels and

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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 4 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

10 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 11 approx. 25% of the worlds coastlines (Filbee-Dexter et al., 2016), from temperate to polar (see Chapter 3) 12 13 rocky reefs (Steneck et al., 2003). Around 43% of the NPP is exported either as DOC (52%) or POC (48%) 14 to neighbouring soft sediment habitats, deep canyons and the deep sea (see Section 5.2.4) where it provides 15 additional food source for invertebrates and fishes, or it is buried in the sediments and thereby contributes to 16 global CO₂ sequestration (Krause-Jensen and Duarte, 2016).

17 18 Two global datasets and one dataset spanning European coastlines (Araujo et al., 2016; Krumhans et al., 19 2016; Poloczanska et al., 2016) identify large local and regional variability of kelp abundance, with a small 20 instantaneous rate of change of -0.018 yr⁻¹ displaying 38% of decline, 27% of increase and 35% of no 21 change in the ecoregions (Krumhans et al., 2016). The average decline is much smaller than any observed 22 regional loss which is a reflection of the naturally high spatio-temporal variability and resilience of kelp 23 forests (Cavanaugh et al., 2011; Reed et al., 2016). A consistent negative response to regional warming 24 between 1950 and 2009 in the abundance, phenology, demography or calcification of macroalgae on a global 25 scale became evident for the northeast Atlantic and southeast Indian Ocean (Poloczanska et al., 2016). 26 Declines in kelp forest abundance attributed to climate change and not related to sea urchin overgrazing 27 (which is a major driver of decline; Ling et al. (2015)) have been documented since the late 1980's and 28 evidence has increased within the last 2 decades. Conversely, climate related cooling events, e.g., along the 29 southern African coastline (Bolton et al., 2012) or the warming of Polar sites (see Chapter 3) led to a spread 30 of kelp beds or algal biomass increase. There is a general lack of data from some regions such as southern 31 America (Pérez-Matus et al., 2017) and also a lack of historical baseline information (Bolton et al., 2012; 32 Poloczanska et al., 2016) but since AR5, substantially more observational evidence supports that loss of kelp 33 forest systems is associated with a phase shift from a 3-dimensional highly diverse kelp habitat to a less 34 diverse and less complex turf forming habitat (high confidence) (Filbee-Dexter et al., 2016). Further 35 evidence supports model predictions (Raybaud et al., 2013; Assis et al., 2016) showing that range 36 contractions of kelps at the warm end of distributional margins and expansions at the poleward end are 37 ongoing at diverse sites globally (high confidence). 38

39 A good example for ongoing range changes is documented in the northeastern Atlantic, where the warm-40 temperate kelp species Laminaria ochroleuca has proliferated at its leading edge in southwest Britain during 41 the last decade corresponding to a period of rapid warming in the western English Channel (Smale et al., 42 2015), and at the same time has retracted westward along the north coast of Spain (Diez et al., 2012)(high 43 confidence). The most important cold-temperate kelp species of the northeast Atlantic, Laminaria 44 hyperborea, also retracted by 1.7° latitudes northward between the 1980s and 2011 along a gradient of 45 decreasing temperature and increasing nutrients in western Portugal (Tuya et al., 2012). 46 In southern and southwestern Norway, the sugar kelp Saccharina latissima disappeared on a broad spatial 47 scale (Moy and Christie, 2012). This phenomenon was especially pronounced in the Skagerrak region where 48 heat-waves in 1997, 2002 and 2006 surpassed sub-lethal levels for sugar kelp of approx. 19°C for 5-8 weeks 49 (Moy and Christie, 2012). Thus increase in summer temperature is a major driver for the decline, and 50 synergistic interaction with eutrophication enhances this situation and also partially hinder the re-51 establishment (Andersen et al., 2013). A heatwave also impacted the infralittoral fringe kelp Laminaria 52 *digitata* in the southern North Sea, an area of rapid temperature increase (Wiltshire et al., 2008). The species 53 temporarily suffered canopy destruction after a prolonged summer heat wave in 2003 surpassing 19°C for 10 54 days (Bartsch et al., 2013). Tolerance limits alone do not explain this observation, thus synergistic negative 55 interaction with high irradiances is an alternative explanation (Bartsch et al., 2013) (low confidence). Species 56 distribution modelling applying CMIP5 temperature data predicts the complete loss of this kelp from its 57 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 (Raybaud 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*).

8 In the northwest Atlantic off Nova Scotia, Canada, during a period of increasing mean seawater temperature 9 (1.58°C between 1979 and 2014 at 2–6 m depth), kelp biomass has substantially decreased by 85-99% 10 (Filbee-Dexter et al., 2016), while turf forming algae has increased positively related to the amount of 11 inorganic sediment. Threshold temperatures for tissue degradation, breaking stress and mortality (Simonson 12 et al., 2015) of the three dominant kelp species: Agarum clathratum, Saccharina latissima and Laminaria 13 digitata, have significantly increased during this time period (Filbee-Dexter et al., 2016). L. digitata does not 14 survive the summer simulation of the predicted 3°C temperature increase for 2100 (23°C) for 2 weeks 15 (Wilson et al., 2015), what corroborates the predictions for this species in the northeast Atlantic (Raybaud et 16 al., 2013). Synergistic effects of temperature with multiple drivers such as physical disturbance by hurricane 17 damage (Filbee-Dexter and Scheibling, 2012) and interaction with enhanced grazing when kelp cover 18 decreases have the potential to further reduce the kelp biomass (O'Brien et al., 2016).

19 20 In the northwest Pacific, decline of kelp bed ecosystems have been observed since the 1980s, especially 21 along the coasts of Japan which span from sub-tropical to temperate areas and are considered a global 22 biodiversity hotspot (Tittensor et al., 2011). Along the coast of Japan seawater temperatures have risen by 23 1.08°C per century (1891–2012) which is more than the double of the mean global ocean temperatures for 24 this period. In 1989–1991 the total area of seaweed beds covered 2012 km² while it decreased by nearly 30% 25 in 1998 (Takao et al., 2015) (medium confidence). The southern kelp beds of Japan are inhabited by one of 26 the most warm adapted kelp species, *Ecklonia radiata*, which can still survive up to 27°C (Takao et al., 27 2015) although temperatures above 20°C will negatively impact the productivity (Serisawa et al., 2004). 28 Ecklonia beds have rapidly declined since the 1990's, and partially disappeared by 2000 in southern Japan 29 mostly attributed to the overall increase of seawater temperatures strongly affecting abalone fisheries which 30 depend on Ecklonia as food source (Serisawa et al., 2004). Along a coastline of approx. 700 km length in 31 southwest Japan, a poleward expansion of tropical species and a retraction of temperate species follow an 32 increase of mean annual SST of 0.3°C per decade between 1970-2009 (Tanaka et al., 2012) (high 33 confidence). A significant reduction in size of potential habitats is projected under all four RCPs using sea 34 surface temperature projected from CMIP5 models, in combination with a species distribution model of kelp 35 (Ecklonia) and its main herbivorous fish species (Takao et al., 2015). The reduction for 2090 is projected to 36 be 15% in the RCP2.6 scenario, but up to 55%, 75% and 85% habitat loss compared to the present-day 37 extent under RCP4.5, 6.0 and 8.5 scenarios, respectively. The loss process is expected to accelerate from 38 2020 onwards (Takao et al., 2015) (medium confidence). Even the kelp species that is most adapted to 39 warmer waters, *Ecklonia radicosa*, apparently disappeared from its former southern distribution limit in 40 Kagoshima Bay concomitant with the 1°C increase in mean seawater temperatures in the past four decades 41 (Komazawa et al., 2014; Terada et al., 2016).

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43 Increased physical stress by storm events also alters kelp communities. A good example is the considerable 44 decline in the average abundance of the disturbance susceptible 'warm water' kelp L. ochroleuca during an 45 unusual storm event in 2014 at southern sites of the British Isles (Smale et al., 2016). As kelp species possess 46 a significantly different communities of organisms living on them and productivity (Smale et al., 2013), 47 increase in the intensity of storms as forecasted by IPCC scenarios will change the food-web structure and 48 productivity of future kelp assemblages (Filbee-Dexter and Scheibling, 2012). In addition, the age structure 49 of kelp communities will become younger under increased physical disturbance favouring species with a 50 year-round spore production or an opportunistic life strategy (Pereira et al., 2017).

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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

6 As endotherms, mammals and birds are more buffered to the effects of temperature changes than reptiles, 7 which as ectotherms, rely on external heat sources to regulate their body temperature. For reptiles, including 8 lizards, turtles and snakes, temperature directly effects important life history traits including hatchling size, 9 sex, viability and performance (very high confidence) (Hays et al., 2003; Pike, 2014; Cavallo et al., 2015; 10 Dudley et al., 2016). This is particularly important for marine turtles as changing temperatures will affect the 11 hatchling sex ratio because sex is determined by nest site temperature (high confidence) (Santidrián Tomillo 12 et al., 2014; Patricio et al., 2017; Santos et al., 2017). Rising temperature will also lead to a loss of breeding 13 substrate as the associated rise in sea level can reduce the available nesting or pupping habitat for land 14 breeding marine turtles, lizards, seabirds and pinnipeds (very likely) (Fish et al., 2005; Fuentes et al., 2010; 15 Hatfield et al., 2012; Funayama et al., 2013; Reece et al., 2013; Katselidis et al., 2014; Patino-Martinez et al., 16 2014; Pike et al., 2015; Reynolds et al., 2015).

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18 Changes in ocean temperature, and pH will also indirectly impact marine mammals, seabirds and reptiles by 19 changing the abundance and distribution of their prey (high confidence) (Polovina, 2005; Polovina et al., 20 2011; Doney et al., 2012; Sydeman et al., 2015; Marshall et al., 2017; Woodworth-Jefcoats et al., 2017). 21 While indirect effects of changing climate are harder to document, examples are becoming apparent. The 22 distributions of some of these large animals is determined by the occurrence and persistence of oceanic 23 bridges and barriers that are related to climate driven processes (Ascani et al., 2016; Briscoe et al., 2017). 24 For example, the decline of Arctic sea-ice is effecting the range and migration patterns of some species and 25 is allowing the exchange of species previously restricted to either the Pacific or Atlantic oceans (virtually 26 certain) (Alter et al., 2015; George et al., 2015; MacIntyre et al., 2015; McKeon et al., 2016; Hauser et al., 27 2017). It is hard to predict how these new invasions will change the existing food webs and species 28 interactions. For example, as killer whales, Orcinus orca, inhabit Arctic waters, narwhal's, Monodon 29 Monoceros, are avoiding the use of key habitats that are already in decline to avoid predation (medium 30 confidence) (Sydeman et al., 2015; Breed et al., 2017) (see SROCC Ch. 3). 31

32 The habitat requirements of marine mammals, seabirds and sea turtles are associated with bathymetric and 33 mesoscale features that facilitate the aggregation of their prev (Bost et al., 2009; Kavanaugh et al., 2015; 34 Hindell et al., 2016; Hunt et al., 2016; Santora et al., 2017). The persistence and location of these features are 35 linked to variations in climate (Crocker et al., 2006; Baez et al., 2011; Dugger et al., 2014; Bost et al., 2015; 36 Youngflesh et al., 2017) and to foraging success, juvenile recruitment, breeding phenology, growth rates and 37 population stability (high confidence) (Ancona and Drummond, 2013; Ducklow et al., 2013; Chambers et al., 38 2014; Descamps et al., 2015; Abadi et al., 2017; Abrahms et al., 2017; Bjorndal et al., 2017; Fluhr et al., 39 2017; Youngflesh et al., 2017). These changes have been most pronounced in polar environments, where 40 sea-ice is an important physical feature that is required for both breeding and hauling out by birds and 41 mammals (Ducklow et al., 2006; Siniff et al., 2008; Jenouvrier et al., 2009; Costa et al., 2010; Constable et 42 al., 2014; Barbraud et al., 2015; George et al., 2015; MacIntyre et al., 2015; Amelineau et al., 2016; Renner 43 et al., 2016; Ferguson et al., 2017; Fernandez-Arcaya et al., 2017; Hamilton et al., 2017; Hauser et al., 2017; 44 Stempniewicz et al., 2017) but also forms a critical link in the structure of polar marine habitats (Murphy et 45 al., 2013; Murphy et al., 2016) (see Chapter 3).

46 47 A consistent theme in both predicted as well as observed changes in megafauna attributed to climate change 48 is that there will be winners and losers (virtually certain) (Costa et al., 2010; Hazen et al., 2013; Clucas et 49 al., 2014; Constable et al., 2014; Laidre et al., 2015). For example, while some Adélie and Chinstrap 50 Penguin populations have plummeted, Gentoo Penguin colonies have increased (Trivelpiece et al., 2011; 51 Cimino et al., 2013; Ducklow et al., 2013; Korczak-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 52 53 range are increasing (LaRue et al., 2013), while those at the northern limits are declining (Trivelpiece et al., 54 2011; Cimino et al., 2013; Ducklow et al., 2013). Long term population changes associated with climate 55 change have also been observed for temperate species of seabirds and shifts in distribution of marine 56 mammals and seabirds (high confidence) (Henderson et al., 2014; Ramp et al., 2015; Trathan et al., 2015). 57 Rockhopper, Eudyptes chrysocome, and Yellow Eyed Penguins, Megadyptes antipodes (Hiscock and
1 Chilvers, 2014; Morrison et al., 2015; Mattern et al., 2017), have declined, while populations of Gentoo, 2 Pygoscelis papua, and Magellanic Penguins, Spheniscus magellanicus, Imperial Cormorants, Phalocrocorax 3 atriceps, Dolphin, Larus scoresbii, and Kelp, L. dominicanus, gulls have increased, some significantly (Rey 4 et al., 2014). However, increases in Magellanic penguin populations occurred at higher latitudes colonies, 5 whereas they declined at the lower latitude colonies (Boersma and Rebstock, 2014). This pattern is 6 consistent with a shifting habitat and species range associated with climate change (highly likely). Laysan, 7 Phoebastria immutabilis, and Wandering, Diomedea exulans, albatross have responded positively to climate 8 change as they have been able to take advantage of the increased intensity of winds, allowing them to forage 9 farther and faster making their foraging trips shorter increasing their foraging efficiency and breeding 10 success (Weimerskirch et al., 2012; Thorne et al., 2016).

11 12 Climate change is also associated with an increase in the frequency and intensity of storms and other 13 extreme weather events (high confidence) (Vargas et al., 2006; Henderson et al., 2014; Descamps et al., 14 2015; Shirasago-Germán et al., 2015; Descamps et al., 2017; Pardo et al., 2017) (Chapter 6), causing 15 increased risk on population viability of marine mammals and seabirds (medium confidence). An increase in 16 storms between 1983 and 2010 resulted in increased reproductive failure in Magellanic Penguins at Punta 17 Tombo, Argentina (Boersma and Rebstock, 2014). An unusual 'warm-water anomaly' off the Northeast 18 Pacific Ocean resulted in closures of commercially important fisheries and mass strandings of marine 19 mammals and seabirds (Cavole et al., 2016) and a reduction in Shag, P. aristotelis, and Kittiwake, Rissa 20 tridactyla, breeding success in the North Sea (Newell et al., 2015). Depending on the age class of petrel and 21 a regional effect, there could be both positive and negative effects of tropical cyclones on Round Island 22 Petrels, Pterodroma arminjoniana (Nicoll et al., 2017).

23 24 The future for species that have limited ranges, limited habitat availability or small populations is quite 25 precarious (very high confidence). Some species like Galapagos Penguins, Spheniscus mendiculus, sea lions, 26 Zalophus wollebaeki, fur seals, Arctocephalus galapagoensis, and flightless cormorants, Phalacrocorax 27 harrisi, have very small populations with restricted ranges and do not have alternative habitats to utilize 28 (Vargas et al., 2005; MacLeod and Steinfartz, 2016; Riofrio-Lazo et al., 2017). In general, specialist species 29 are more vulnerable than generalists. This can be in terms of diet or habitat requirements, or in terms of 30 existing life history traits (Gallagher et al., 2015). 31

32 5.3.3.10 Summary for Coastal Ecosystems33

34 Overall, coastal ecosystems are under stress from the combination of climate change impacts in the ocean 35 and on land as well as non-climatic human activities, with observable and projected climate impacts through 36 modification of geomorphology, reduction in habitat area, and shifting distributions and community structure 37 (high confidence) (Section 5.2.2). Specifically, the morphology of sandy beach and saltmarshes is changing 38 as a result of rising sea level and more severe storms that leads to their erosion and retreat (Sections 39 5.2.2.3.4, 5.2.2.3.5). These consequently lead to losses of vegetation and benthic fauna (high confidence), 40 with detrimental consequences for saltmarshes carbon storage (high confidence) (Section 5.3.2.3). The 41 increased levels, frequency and duration of eutrophication and hypoxic conditions in estuaries and coastal 42 zones worldwide in the last few decades are attributable to warming (*high confidence*), and these processes 43 are projected to be more pronounced in the high latitudes (high confidence) (Section 5.2.2.3.1). Synergism 44 between acidification and warming exacerbates the negative effects on rocky and coral reef communities. 45 causing a shift towards a less diverse ecosystem in terms of species richness and spatial heterogeneity with 46 detected climate impacts including the increased dominance of macroalgae in rocky reefs and decreased in 47 species richness and structural complexity of coral reefs (medium confidence) (Section 5.2.2.3.6). Kelp 48 forest, mangroves and seagrass meadow ecosystems are shifting poleward with warming and increasing 49 frequency and intensity of extreme events such as heat waves and storms (high confidence) (Section 50 5.2.2.3.7). Herbivory intensified by warming will result in physical and physiological stress on coastal 51 vegetation and reduce their productivity (medium confidence) (Sections 5.2.2.3.4, 5.2.2.3.8).

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5.3.4 Assesments for Coastal Ecosystems

55 This section synthesizes the assessments on coastal ecosystems to evaluate the risk of climate change on 56 their biodiversity and ecosystem functions (Appendix 5.A, Table 2). The overall global level of risk is 57 assessed based on the latest understanding of changes in oceanographic drivers in coastal regions (Section 1 5.3.2). The level of vulnerability of coastal ecosystems to climate hazards depends on their sensitivity and 2 adaptive capacity discussed in Section 5.3.3. The ecosystem sensitivity is assessed by discriminating the 3 distinct responses to climate hazards of the main biotic and abiotic components that define each ecosystem 4 (Section 5.3.3). For example, intertidal ecosystems are more threatened by erosive processes due to sea level 5 rise and storms than subtidal (always submerged) ecosystems; rising sea level affects the geomorphology in 6 estuaries and sandy beaches (e.g., dune scarping), and the plant biomass and soil accretion in saltmarshes 7 and mangroves. Conversely, in ecosystems defined by foundational species such as seagrasses, kelp forest, 8 coral and other biogenic reefs, the physiology, distribution and ecological interactions of the organisms are 9 more sensible to warming and acidification.

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11 All coastal ecosystems assessed here are exposed to multiple climate hazards (rising sea surface temperature, 12 acidification, deoxygenation, sea level rise, storms and precipitation) that increases with higher carbon 13 emission scenarios. The level of exposure depends on (1) the geographical distribution (Figure 5.12), (2) 14 local interactions between climate and human impacts (e.g., pollution, eutrophication, aquiculture, 15 urbanization), and (3) topographic characteristics (e.g., intertidal or subtidal). For example, estuaries and 16 sandy beaches are distributed all across the global coastlines that are experiencing diverse range of changes 17 in climate hazards, while seagrasses and mangroves are restricted to temperate and tropical/subtropical 18 regions where their exposure to climate hazards is generally similar in nature (Figure 5.14). Likewise, kelp 19 forests and seagrasses meadows are submerged ecosystems, while all the other ecosystems defined here are 20 intertidal (rocky and coral reefs can be both, intertidal and submerged), with exposition to atmospheric-21 oceanographic conditions. Overall, in all coastal ecosystems, multiple climate drivers will emerge from 22 historical variability in the 21st century under RCP8.5 (time of emergence, Box 5.1), while the time of 23 emergence will be later with less climate drivers under RCP2.6. Non-climatic human drivers such as 24 eutrophication add to, and in some cases, exacerbate these large scale slow climate drivers beyond biological

thresholds on local scale, e.g., deoxygenation.

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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.

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35 Some particularly vulnerable coastal ecosystems are already experiencing moderate to high risk under 36 present day conditions. Specifically, coral reefs and seagrass meadows (Section 5.3.3.6 and 5.3.3.4, 37 respectively). Changes in ecosystem structure and functions have been detected at present day and attributed 38 to climate change (see Section 5.3.3). Furthermore, the surviving corals of tomorrow will not resemble the 39 same as at present day, because irreversible changes in habitat structure and functioning, including species 40 extinctions, food web disruptions and shifts in community complexity are currently emerging in some hot 41 spots areas of climate change (e.g the Caribbean reefs). In consequence, this permanent transition to new 42 ecosystem states driven by unpredictable pulses of disturbance and progressive climate hazards implies

43 detrimental impacts to ecosystem services (Section 5.4.1.2).

2 All coastal ecosystems are projected to have high to very high risk of impacts under RCP8.5 in the 21st 3 century that varies generally across ecosystems. Particularly, the ecosystems with very high risks are coral 4 reefs (very high confidence), seagrasses meadows (very high confidence), kelp forest (very high confidence) 5 and rocky reef (medium confidence) under RCP8.5. These ecosystems have low to moderate adaptive 6 capacity, as they are highly sensitive to warming and acidification. For example, kelp forests and endemic 7 species of mid latitude seagrasses have low dispersal ability and are particularly sensible to heatwaves. 8 Biogenic shallow reefs with calcified organisms (e.g., corals, mussels, calcified algae), are particular 9 sensible to ocean acidification, which combined with rising temperatures, deoxygenation, sea level rise and 10 increasing extreme events, make these ecosystems highly vulnerable (impede resilience) to future emission 11 scenarios. 12

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

29 30 The Fifth Assessment Report concluded that climate change increases the risk of impacts on the goods and 31 services derived from marine biodiversity and ecosystems. These goods and services include fisheries and 32 other provisional services, climate regulation by marine microbes, coastal protection by vegetation and other 33 biogenic habitats, and cultural values such as tourism, heritage and ceremonial importance (Pörtner et al., 34 2014). This section further assesses the impacts of climate change on ecosystem goods and services in 35 specific coastal, pelagic and deep-sea floor ecosystems. Under the conceptual framework developed by the 36 Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Hughes et al., 2017) (Figure 37 5.13), climate change is considered to be a direct driver that affects ecosystem goods and services 38 (provisioning and others including regulation, supporting and cultural, see Section 5.4.1). This section 39 assesses how climate change impacts on marine biodiversity and ecosystems (Sections 5.2 and 5.3) affect 40 changes in ecosystem services and their consequences for human wellbeing (social, cultural, economic and 41 environmental, see Section 5.4.2). Climate change can also impact ecosystem services through the 42 amplification of the effects of non-climatic drivers on marine ecosystems and biodiversity, such as 43 increasing the variation and magnitude of runoff and its associated impacts of erosion and coastal 44 sedimentation (Singh et al., 2017). Attribution of climate change effects on these marine ecosystems are 45 addressed in Sections 5.2 and 5.3, while Section 5.4 assesses the implications of changing ecosystem goods 46 and services for human wellbeing and their challenges to the United Nations' Sustainable Development 47 Goals (Section 5.4.2).

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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' (Díaz et al., 2018). These services include all benefits that humanity obtains from nature. These ecosystem goods and services can be broadly divided into provisional 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

1 al., 2014) (Sections 5.2.2, 5.2.3). The final major category of ecosystem services – cultural ecosystem 2 services –includes recreation, tourism, aesthetic and spiritual experiences. These services are a product of 3 both humans experiencing nature and the availability of nature to provide the experiences (Chan et al., 4 2012); the quality and quantity of the latter can be impacted by climate change. Also, in some cases, climate 5 change will alter the distribution of organisms and ecosystems, and consequently where these ecosystem 6 services are enjoyed. However, adjustments to how human communities access place-based services may not 7 be possible, particularly for those services that rely on indigenous knowledge and local knowledge (Cross-8 Chapter Box 3 in Chapter 1). The cultural aspects and linkages to local knowledge and indigenous 9 knowledge of human-ecosystem linkages are further highlighted in the recent suggestion to use 'Nature's 10 Contribution to People' as a new conceptual framing instead of ecosystem services (Díaz et al., 2018). 11 However, this assessment will be based on the conceptual framing of the relevant existing literature, which is 12 mostly still based on the ecosystem services framing, with the particular focus on the observed and projected 13 impacts and risks of climate change on these services (Sandifer and Sutton-Grier, 2014). Following the 14 overall structure of this assessment, ecosystem services from open-ocean and coastal ecosystems are 15 assessed. 16

17 5.4.1.1 Open Ocean Ecosystems

18 19 The vast volume of open ocean ecosystems offers the full range of ecosystem services, including 20 provisioning, regulatory, supporting and cultural services, to human societies, although our access to these 21 services are often indirect (Ingels et al., 2016). The open ocean ecosystems are fundamental components of 22 the Earth system that are key to climate regulation (Section 5.2.2). The pelagic ecosystem, where the 23 majority of the open ocean primary production takes place (Section 5.2.2.5), supports biomass production for 24 valuable fisheries as well as food for marine organisms including marine mammals, seabirds and turtles 25 (Sections 5.2.3, 5.3.3) that contribute to eco-tourism. The deep sea floor ecosystems also support 26 biodiversity that is important for fisheries and they recycle nutrients that support the pelagic ecosystem 27 (Section 5.2.4). In addition, open ocean ecosystems and their associated biota are intrinsically valued by 28 people, offer opportunities for scientific discoveries and education, and hold genetic resources that have 29 potential for human uses (Blasiak et al., 2018).

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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.

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37 5.4.1.1.1 Provisioning service

38 Fisheries from the open ocean pelagic ecosystems remain important sources of food, nutrition, income and 39 livelihoods for many millions of people around the world (FAO, 2016). Globally, total fishery catches of 40 pelagic species amount to 44-53 million MT annually in the 2000s (based on the data from Pauly and Zeller 41 (2016)) (Figure 5.14), generating 9.2 billion USD of revenue directly from their fisheries (in year 2010 42 value) contributing to around 12% of the total fishery catch (Sumaila et al., 2015). The majority of the catch 43 from the open ocean pelagic ecosystems is tunas and billfishes (Crespo et al., 2017). Fishery catches from 44 these ecosystems have been stagnant since the 1990s, which is partly attributable to fishing at or above 45 biological capacity for production (Merrie et al., 2014). Fishing has also driven down populations of species 46 with vulnerable life history characteristics for a high extinction risk, such as large sharks (Fowler, 2014).

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48 Based on fishery catch records and model projections, ocean warming, decrease in primary production and 49 expansion of oxygen minimum zones in some ocean regions are shifting patterns of fishery productivity in 50 these pelagic ecosystems towards higher latitudes or longitudinally following temperature gradients, while 51 the distribution of commercially important fish stocks and their fisheries are shifting toward higher latitude. 52 The distribution of fisheries on large pelagic species such as tunas correlate strongly with historical climate 53 variability (high confidence). For example, in intermediate latitudes across the Atlantic, Indian and Pacific 54 oceans, catches of tropical tunas, including skipjack and yellowfin tuna, are significantly positively related to 55 increases in sea surface temperature, although the overall catches across latitudinal zones do not show a 56

Chapter 5

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).

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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 (Mcowen 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.

16 17 Globally, the potential fisheries catch in the open pelagic oceans is projected to decrease by 2100 relative to 18 the 2000s under RCP8.5, RCP4.5, and RCP2.6 scenarios with regional differences (Cheung et al., 2016a; 19 Blanchard et al., 2017). Based on projections from an ensemble of three CMIP5 earth system models and a 20 fisheries impact model, there is a projected significant (more than -10% by 2050 relative to present day) 21 decrease in catch potential under both RCP4.5 and RCP8.5 forcing (Cheung et al., 2016a). Here, potential 22 fisheries catch or maximum catch potential refers to the biological potential productivity of the fish stocks to 23 provide long-term fisheries catches; it is considered a proxy of maximum sustainable yield. However, the 24 actual catches realized by fisheries will dependent strongly on past and present fishing effort and 25 exploitation status of the resources (Cheung et al., 2018a). Ensemble projections including multiple fisheries 26 models generally support the direction of such projections (Blanchard et al., 2017). Results from these 27 models are largely driven by projected changes in temperature and net primary production. Regionally, the 28 tropical Ocean is projected to have substantially larger impacts (three times or more decrease in catch 29 potential) than the global average, particularly in the western central Pacific Ocean, eastern central Atlantic 30 Ocean and the western Indian Ocean under RCP8.5. In contrast, the catch potential in the Arctic is projected 31 to increase, although there is high intermodel variability in these projections. In addition to poleward shifts 32 in fisheries catches, selected open ocean pelagic resources are also projected to show a longitudinal shift, 33 such as eastward shifts of skipjack tuna under ocean warming (Lehodey et al., 2011; Schulz et al., 2013). 34 However, the actual catches and profitability of the fisheries interact strongly with the level of exploitation 35 of the resources (Cheung and Pauly, 2016; Dueri et al., 2016). In addition, fisheries in the open ocean are 36 predominantly operated by industrial fishing from a few countries (Sumaila et al., 2015), which may be more 37 adaptable to changes in resource distributions, although analysis of the adaptive capacity of distant water

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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).

4 5 Overall, fisheries are currently the most important provisional service in the open ocean pelagic ecosystems. 6 Fishery resources in this ecosystem are sensitive to oceanographic conditions; in particular, changes in their 7 distributions follow sea surface temperature closely (robust evidence, high agreement, very high confidence). 8 Historical patterns of fishing activities in the ocean pelagic ocean have also followed fishery resource 9 distributions (medium confidence). Potential fisheries catch from currently exploited fish stocks are projected 10 to decrease by mid-century under all greenhouse gas emission scenarios, with the tropical oceans being 11 projected to have the highest impacts (medium confidence), partly as a result of the projected poleward shifts 12 in resource distributions (high confidence) and a decrease in net primary production in low latitude regions 13 (medium confidence). The realized fisheries catch in the future will depend strongly on the way that fishing 14 activities are managed in the open ocean pelagic ecosystems (very high confidence). 15

[START OF BOX 5.2 HERE]

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

22 Eastern Boundary Upwelling Systems (EBUS) are among the most productive ocean ecosystems in the 23 world (Kämpf and Chapman, 2016), supporting not only the livelihood of coastal communities and sectors 24 that are directly dependent on EBUS, but also contributing to benefits that much broader human societies 25 enjoy (García-Reyes et al., 2015; Levin and Le Bris, 2015) (Appendix 5.A, Table 3). The high productivity 26 of EBUS is supported by the upwelling of cold and nutrient-rich waters from depth, driven by equatorward 27 alongshore winds that cause the displacement of surface waters offshore and the replacement by deeper 28 waters. Supported by the high productivity, total annual fisheries catch from the four main EBUS in the 29 world (California Current, Humboldt Current, Canary Current and Benguela Current) amount to 16-24 MT 30 per year in the 2000s, contributing to approximately 17% of the global catches (based on data from Pauly 31 and Zeller (2016)). These catches are consumed locally as well as processed and exported as seafood, fish 32 meals and oils to support aquaculture and livestock production. Upwelling of cold deeper water increases 33 condensation of wet air in coastal areas that brings in moisture and precipitation that are important for the 34 maintenance of coastal vegetation and agriculture, as well as suppressing of forest fires (Black et al., 2014). 35 The high concentration of marine mammals attracted by the productive upwelling ecosystem support 36 lucrative eco-tourism such as whale-watching in the California Current (Kämpf and Chapman, 2016). A total 37 economic value calculation of the goods and services provided by the Humboldt Current alone is estimated 38 to be US\$19.45 billion per annum (Gutiérrez et al.). Thus, although their area is small compared to other 39 pelagic ecosystems, the impacts of climate change on EBUS are *likely* to have disproportionately large 40 consequences for human societies (very high confidence). 41

42 The coupled human-natural EBUS are vulnerable to the multiple effects of climate change with large 43 regional variation (Blasiak et al., 2017). Observational and modelling analysis suggests that there has been 44 an intensification of wind with global warming in three of the four main EBUS (except the Canary Current 45 EBUS) during the last 60 years, with a number of hypotheses explaining the mechanisms of such 46 intensification (Sydeman et al., 2014; García-Reyes et al., 2015; Rykaczewski et al., 2015). Moreover, 47 coastal warming and wind intensification may lead to variable countervailing responses to upwelling 48 intensification at local scales (García-Reyes et al., 2015; Wang et al., 2015; Oyarzún and Brierley, 2018; Xiu 49 et al., 2018). Local winds and mesoscale oceanographic features (not resolved in most coupled climate 50 models), rather than large-scale wind patterns, are thought to have a greater impact on regional productivity 51 (Renault et al., 2016; Xiu et al., 2018). Changes in SST in EBUS in recent decades are substantial but with 52 conflicting evidence because of the differences in resolution of the data and their high temporal variability 53 (Sydeman et al., 2014; Wang et al., 2015). Specifically, cooling of coastal areas is observed in Humboldt 54 Current, Benguela Current and California Current EBUS but not in the Canary Current EBUS. However, 55 temperature trends revealed from recent records may not have emerged from the large natural variabilities of 56 some EBUS. Larger-scale data that include both coastal and offshore waters around EBUS suggest that there 57 have been decreases in pH (Turi et al., 2016; Chavez et al., 2017) and dissolved oxygen (Levin, 2018),

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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.

3 4 The climate change impacts on ecosystem services from EBUS vary regionally depending on both the 5 biophysical and the social and economic characteristics of the upwelling systems (García-Reyes et al., 2015) 6 (Box 5.2, Appendix 5.A, Table 3; Box 5.2, Figure 1). The fisheries in the upwelling systems are highly 7 sensitive to upwelling conditions that is exacerbated by fishing effects on the exploited populations. For 8 example, the anchoveta population collapsed in the Humboldt Current after the El Niño in the 1970s that 9 impacted the fishing industries largely (Gutiérrez et al., 2017). Since the small pelagic fisheries from 10 upwelling regions are the main source of the global fishmeal market, the decrease of its production leads to 11 increase in international fishmeal price, that consequently affects the price of other food commodities (like 12 aquaculture-derived fish) that rely on fishmeal for their production (Merino et al., 2010; Carlson et al., 13 2017). Moreover, the decrease in catches would also affect regional food security as catches from the 14 upwelling region is an important source of nutrients. For example, in the Canary Current, coastal fisheries 15 are important source of micronutrients to the nearby West African countries (Golden et al., 2016), whose 16 susceptibility to climate change impacts are particularly high and adaptive capacity is low because of their 17 strong dependence on the fisheries resources, a rapidly growing population and regional conflicts (Box 5.2, 18 Figure 1). Decreased small pelagic fish stocks also increase the mortality and reduce reproduction of large 19 vertebrates such as hake (Guevara-Carrasco and Lleonart, 2008), whales and seabirds (Essington et al., 20 2015). Impacts on these organisms affect other non-fishing sectors that are dependent on EBUS, such as 21 whale-watching in the California Current EBUS, and generally degrade their intrinsic values.

22 23 Overall, EBUS have been changing with intensification of wind that drives the upwelling, leading to changes 24 in water temperature and other ocean biogeochemistry (high confidence); however, the direction and 25 magnitude of observed changes vary between and within EBUS with mechanisms explaining these 26 variabilities not fully understood yet. Moreover, the high natural variability of EBUS and their insufficient 27 representation by global climate models renders attribution to climate change having *low confidence*. Given 28 the high sensitivity of the coupled human-natural EBUS to oceanographic changes, the future sustainable 29 delivery of key ecosystem services from EBUS is at risk under climate change; those that are most at risk in 30 the 21st century include fisheries (high confidence), aquaculture (medium confidence), coastal tourism (low 31 confidence) and climate regulation (low confidence). For vulnerable human communities with a strong 32 dependence on EBUS services and low adaptive capacity, such as those along the Canary Current system, 33 unmitigated climate change effects on EBUS (complicated by other non-climatic stresses such as social 34 unrest) have a high risk of altering the development pathways of these vulnerable societies (high confidence, 35 Box 5.2, Figure 1).



1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33

Box 5.2, Figure 1: Projected climate hazards, sensitivity and adaptive capacity of the coupled natural-human EBUS: (A) (A) California Current, (B) Humboldt Current, (C) Canary Current and (D) Benguela Current. Data from vulnerability assessment by Blasiak et al. (2017). L – Low, M – Medium, H – High.

[END OF BOX 5.3 HERE]

5.4.1.1.2 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*).

1 The open ocean pelagic ecosystem also supports nutritional health and culture of coastal indigenous 2 communities through migratory species. Particularly, many marine species that are important to coastal 3 indigenous communities migrate between coastal waters and the open ocean, such as tunas and sea turtles 4 (Harrison et al., 2018). These coastal communities access the open ocean biodiversity when they migrate to 5 coastal waters. Coastal indigenous communities have higher than the average level of seafood consumption 6 (Cisneros-Montemayor et al., 2016). Marine life is also strongly linked to many traditional cultures. For 7 example, sea turtles for Fijians are both subsistence food and have strong association to hierarchical 8 obligations among the chiefs and clans (Morgan, 2007). The challenges faced by indigenous coastal 9 communities in adjusting their culture in response to climate change are also particularly high. Thus, climate 10 change impacts on these species in the open ocean (5.2.2) are expected to have implications for the 11 nutritional health and culture of coastal dependent communities such as indigenous people (5.4.3). 12

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

23 Deep-ocean processes and organisms (Table 5.4) generate many types of provisioning, regulating, 24 supporting and cultural ecosystem services (Figure 5.13) (Armstrong et al., 2012; Thurber et al., 2014), most 25 of which are poorly studied and rarely quantified. The stakeholders for the deep ocean, which covers 54% of 26 the surface area of the planet and most of the habitable volume, are all the people on the earth. The deep sea 27 holds a wealth of living and non-living resources and provides services that are required to sustain humans 28 for centuries to come. Understanding and assessing the impacts of climate change on deep-sea ecosystem 29 services is particularly challenging as the deep sea is vast, remote and only partially explored. Many deep-30 sea habitats are unfamiliar and the functions of organisms are poorly known.

31 32 Increasing demands on deep-sea resources such as fish/shellfish, oil and gas and seabed minerals are creating 33 the need to identify, understand and valuate the services provided by vulnerable deep-sea ecosystems and 34 raising interest in ecosystem services in the context of environmental and financial management (Le et al., 35 2017). In shallow water, there is an appreciation that ecosystem-based approaches can contribute to climate 36 solutions as well as biodiversity and conservation goals (Wüstemann et al., 2017), but similar efforts for the 37 deep ocean are just beginning, despite its major role in climate mitigation through heat and CO₂ uptake. New 38 approaches such as the Ecosystem Principles Approach are being developed or applied to draw the link 39 between the ecological and socio-economic dimensions of the deep sea (Table 5.4) (Jobstvogt et al., 2014). 40 When Willingness to Pay (WTP) methods were applied to cold-water corals (CWCs) the possibility that 41 CWCs play an important role as habitat for fish was the single most important variable to explain 42 respondents' WTP for CWC protection (Aanesen et al., 2015). There is a need for advanced thinking about 43 ecosystem services in the deep ocean, going beyond intrinsic and instrumental values to incorporate 44 relational values (Eudaimonia) that collectively acknowledge human associations with nature in multiple 45 forms across different cultural practices (Chan et al., 2016).

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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 ^laboratory experiment; \$ paleo record

Ecosystem services	Canyons	Seamounts	Seeps and vents	Cold water coral and sponge reefs	Abyssal Plains	Continental Slopes
Provisioning						
Food Provision	Reduced food provision due to increased stratification, loss of	Reduced food provision due to loss of organic matter export	Unknown effect: Species reliant on methane	Loss/dissolution of nursery habitat could translate into loss of fish	Limited food provision from this setting	Transient increased catch may occur as a result of

	organic matter export and benthic production (+)	and benthic production (+)	seeps could benefit if warming induces additional methane release (\$); loss of chemosynt hetic foundation species (mussels, tubeworms) expected on upwelling margins as OMZs expand (#)	production and fisheries (+)		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 (#,+)
Genetic resources and chemical compounds for biotechnologi cal, pharmaceutic al or industrial applications	No assessment	No assessment	Loss of resources associated with foundation species (*)	Habitat loss could cause loss of genetic resources (*)	Unknown effect; could interact with physical disturbance from mining	Expanding hypoxia may diversify anaerobic microbial genetic resources (#)
Regulating			.			
Carbon sequestration and storage	Reduced carbon sequestration and storage (#,+)	Reduced carbon sequestration and storage (#,+)	Limited effect (*)	Increased dissolution and reduced C sequestration	Reduced carbon sequestration and storage	Reduced carbon sequestration and storage
Waste				(*,+)	(#,+)	(*;+)
absorption and detoxification	No assessment	Unknown effect; could interact with physical disturbance from mining (*)	Unknown effect; could interact with physical disturbance from mining (*)	(*,+) No assessment	(#,+) Unknown effect; could interact with physical disturbance from mining (*)	(*; +) No assessment
absorption and detoxification Disease and invasive species	No assessment No assessment	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	No assessment	(#,+) Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	(*; +) No assessment No assessment
absorption and detoxification Disease and invasive species Supporting (intermediate)	No assessment No assessment	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	No assessment	(#,+) Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	(*; +) No assessment No assessment
absorption and detoxification Disease and invasive species Supporting (intermediate) Habitat and	No assessment No assessment Habitat loss,	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*) Habitat loss,	Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*)	(*,+) No assessment No assessment Habitat loss	(#,+) Unknown effect; could interact with physical disturbance from mining (*) Unknown effect; could interact with physical disturbance from mining (*) Unknown.	(*; +) No assessment No assessment Warming,

	spawning and nursery grounds (*,+)	spawning and nursery grounds (*, +)	with methane seeps; elasmobran chs use vents and seeps as nursery grounds. Could expand services. (#)	trophic support (*, +)	es; could interact with physical disturbance from mining (*, +)	acidification causing habitat loss and reduced trophic support for hypoxia/acidi fication/war ming- intolerant species (* #,+)
Nutrient cycling	Reduced via stratification, OM export and seafloor biomass (+)	Reduced via stratification, OM export and seafloor biomass (+)	No assessment	Reduced via stratification, OM export and seafloor biomass (+)	Reduced via stratification, OM export and seafloor biomass (+)	Reduced via stratification, OM export and seafloor biomass (+)
Chemosynthe tic primary production	No assessment	No assessment	Produducti on could expand at methane seeps if warming increases gas hydrate dissociation (*)	No assessment	No assessment	No assessment
Resilience and resistence	Reduced via decline in food supply, warming, deoxygenatio n, acidification (*)	Reduced via decline in food supply, warming, deoxygenatio n, acidification (*)	Reduced via deoxygenat ion, effectcs on planktonic larvae in surface waters (*).	Reduced via warming, deoxygenation, acidification (*)	Reduced via decline in food supply, warming and acidification (*)	Reduced via decline in food supply, warming, deoxygenatio n, acidification (*)
Cultural						
Aesthetic and spiritual	No assessment	No assessment	No assessment	No assessment	No assessment	No assessment
Bequest and existence	No assessment	No assessment	No assessment	No assessment	No assessment	No assessment
Scientific and educational	No assessment	No assessment	No assessment	No assessment	No assessment	No assessment

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 fishery 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., 2014).

1 2018c; Levin, 2018). The North Atlantic, Indo-Pacific, West Africa and South Pacific regions host the most 2 vulnerable deep-sea species (Cheung et al., 2018c; Levin, 2018). Of equal or greater concern is the fact that 3 climate change will exacerbate physical disturbance from bottom trawling, reducing habitat complexity and 4 resilience, affecting the provision of other ecosystem services (Levin and Le Bris, 2015; Sweetman et al., 5 2017). The movement to manage fish stocks collectively using an ecosystem-based approach has led to 6 development of indicators of fish community change (survey-based mean trophic level, proportion of 7 predatory fish, and mean lifespan (Kleisner et al., 2015), as well as indicators of supporting habitat status. 8 These could be applied to deep-sea fish assemblages in the context of climate change for some of the better-9 studied ecosystems.

- 11 The effects of warming and ocean deoxygenation on deep sea organisms and ecosystems (Section 5.2.3) 12 affect the quality and quantity of deepwater fisheries catches through similar pathways as in the open ocean 13 pelagic ecosystems (medium confidence). Warming may reduce body size (medium confidence, Section 14 5.2.3) and lead to range shifts of exploited vertebrate and invertebrate species to deeper depths and higher 15 latitude (habitat compression) (high confidence), with effects on availability, accessibility, costs of fishing 16 and transport (medium confidence) (Cheung and Pauly, 2016; Lam et al., 2016). Habitat compression above 17 oxyclines can increase catchability but may lead to overfishing or increased vulnerability to natural predators 18 (Prince et al., 2010; Stramma et al., 2011). Ocean deoxygenation is leading to expansion of oxygen 19 minimum zones in the tropics and subtropics (Stramma et al., 2008; Stramma et al., 2010). On the US West 20 coast, near-bottom oxygen concentration is positively associated with both the biomass of commercially 21 harvested fish species (Keller et al., 2010) and catch per unit effort (Keller et al., 2015). Deoxygenation is 22 predicted to reduce available rockfish habitat (McClatchie et al., 2010). Crabs (*Cancer magister*) are also 23 *likely* to be negatively impacted by expanded hypoxia (Gallo and Levin, 2016). In Peru, India, and Pakistan, 24 the catch per unit of effort (CPUE) of demersal fisheries decrease when bottom oxygen levels decline 25 (Banse, 1968; Rosenberg et al., 1983), and some species such as hake, octopus and scallops only appear on 26 the Peru margin during oxygenation events (associated with El Niño) (Arntz et al., 2006). Beyond direct 27 effects, deoxygenation can reduce the diversity (Sperling et al., 2016), size and abundance (Levin, 2003) of 28 infaunal prey, as well as suppress benthic-pelagic trophic interactions. However, hypoxia-tolerant species 29 such as tuna crabs (*Pleuroncodes planipes*) (Pineda et al., 2016; Seibel et al., 2018), Humboldt squid 30 (Dosidicus gigas) (Stewart et al., 2014) and the pink urchin Strongylocentrotus fragilis (Sato et al., 2017) 31 appear to be expanding their ranges (medium confidence) and could offer new fishery opportunities (Sato et 32 al., 2018). As OMZs expand on margins, waters also become more acidic (Paulmier et al., 2011), with 33 potential effects on calcifying taxa (see Section 5.2.4.4) (medium confidence).
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35 5.4.1.2.2 Regulating, supporting and cultural services

36 The large areas and long residence times over which regulating services are provided in the deep sea mean 37 that they are massive (high confidence), although the processes are often far removed from the humans that 38 receive them (Thurber et al., 2014). The deep ocean ecosystem service that is most directly linked to climate 39 is the sequestration of carbon, both through storage in the deep-sea pool and through long-term burial in 40 sediments (high confidence). A net flow of approximately 1.6 GtC yr⁻¹ from the atmosphere into the deep 41 sea, of which 0.2 GtC is buried, represents a value of USD108 billion yr⁻¹ if valued at a conservative USD19 42 per tCO₂-eq (which is USD62 per tC assuming all the carbon is in the form of CO₂) (Armstrong et al., 2010) 43 (*medium confidence*). The value of carbon export to the deep ocean was estimated at USD12.9 billion yr^{-1} 44 (using average European C market prices) for the Eastern Tropical Pacific covering 21 million km² with 45 waters of 12 nations + international commons) (Martin et al., 2016) (medium confidence). Only one study 46 has examined the economic cost of climate driven changes in carbon fluxes, focusing on the North Atlantic. 47 Declines in C flux at 1000 m in the North Atlantic, estimated between 27–43% by 2100 depending on RCP 48 scenario, were valued at USD170–300 billion in mitigation costs and 23–401 billion in adaptation costs over 49 the 21st century (Barange et al., 2017) (low confidence). Carbon sequestration by the biological pump over 50 the Mediterranean Basin was estimated at 281 million Euros per year (=USD 29 billion over the same 51 period) (Melaku Canu et al., 2015) (low confidence). Uncertainties in these values derive from uncertainties 52 in the ecosystem models (mechanisms, scenario pathways) and in carbon price estimates and discounting 53 rates (Her Majesty's Treasury, 2011; van den Bergh and Botzen, 2015). Other forms of carbon sequestration 54 in the deep sea merit valuation, such as the removal of methane released from the beneath the seabed 55 through microbial anaerobic oxidation and the sequestration in carbonate as well as aerobic methane 56 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.

10 Option value (the value of maintaining a resource, i.e., function, due to uncertainty regarding supply or 11 demand) and bequest value (the value of maintaining a resource for availability to future generations) can be 12 associated with all deep-sea ecosystem structures and functions discussed in Section 5.2.2.4 (high 13 *confidence*). Particularly, many of the potential values and services from deep-sea ecosystems are currently 14 not realized, but are increasingly recognized to be crucial for human wellbeing (Aanesen et al., 2015). The 15 importance of these services in the deep ocean will increase as climate change degrades resources and 16 services in shallow water (medium confidence), and can motivate further conservation of ecosystem services. 17 Climate change-induced degradation of deep-sea ecosystems and biodiversity could also reduce the 18 availability of such option value.

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20 Supporting services such as primary and secondary production, habitat provision for feeding, spawning or 21 nursery grounds, refugia, and biodiversity are all essential functions that ultimately contribute to the 22 provisioning, regulating or cultural services (Haines-Young and Potschin, 2013; Bopp et al., 2017). 23 Vertically migrating fish, which carry organic matter and carbon into deep-water perform functions that 24 support carbon sequestration, fisheries and more. Habitat support functions in the deep sea are well 25 recognized for canyon and seamount environments as these highly productive systems are often targeted by 26 fishers. Coral and sponge assemblages provide food, refuge and nursery grounds for key fishery species, and 27 host a wealth of biodiversity (Buhl-Mortensen et al., 2010), much as in shallow water systems. Deep-water 28 coral habitats, especially the non-living matrix, are highly vulnerable to ocean acidification (as shallow reefs; 29 Section 5.2.3.3) and thus the supporting services they provide are at risk (low confidence). Seeps and vents 30 serve as nursery grounds for elasmobranchs (Treude et al., 2011) and cephalopods (Drazen et al., 2003) and 31 as aggregation sites for adults of a surprising number of commercial species (Sellanes et al., 2008; Grupe et 32 al., 2015), while sea pen and octocoral gardens provide nursery habitat for fish embryos and larvae (Baillon 33 et al., 2012). The continuing discovery of new faunal associations with deep-sea habitats referenced in the 34 previous sentence suggest that we most likely have yet to uncover many of the habitat support services 35 provided by the deep sea (high confidence). 36

37 Biodiversity, which is understood to be high in the deep sea but remains mostly undescribed (Ramirez-38 Llodra et al., 2010), has existence value itself. It has optional values for the potential use of genetic resources 39 to generate products for human uses such as pharmaceuticals industrial agents, cosmetics, biofuels or 40 biomaterials (Skropeta and Wei, 2014; Harden-Davies, 2017). Deep sea biodiversity also contributes 41 significantly to overall ecosystem functioning that supports other services as well. Ocean deoxygenation will 42 reduce biodiversity of fauna at intermediate depths (see Section 5.2.4.3) but valuation of deep-sea 43 biodiversity is in its infancy and translation of climate-induced loss into services has yet to occur. The 44 greatest effect of biodiversity loss is *likely* to be loss of support functions. Nearly all deep-sea supporting 45 services (e.g., secondary production, nutrient remineralization, trophic support, habitat provision, nursery 46 and spawning grounds) that support fisheries production and climate regulation (Thurber et al., 2014; Bopp 47 et al., 2017) will be affected by changes in temperature, oxygen, pH and POC flux, as discussed in (Section 48 5.2.4) on impacts on ecosystems. 49

50 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

1 coastal ecosystems are \$352,249/ha for coral reefs, and \$193,843/ha for tidal marshes and mangroves 2 (Pendleton et al., 2016), although the values are not static and have wide uncertainties (Hilmi et al., 2013b). 3 Coastal ecosystems such as coral reef are also providing ecosystem services to half a billion of people, 4 worldwide (Hughes et al., 2012). In Southeast Asia the total number employed in fisheries and aquaculture is 5 estimated at over 20 million people and the net annual economic benefits of healthy coral reefs in the region 6 to shoreline protection, fisheries, and tourism are estimated to range between \$23,000 to \$270,000 per km² 7 of reef (Whittingham et al., 2003; Burke et al., 2006). However, the deterioration of coastal ecosystems due 8 to human activities is intense and increasing; 50% of salt marshes, 35% of mangroves, 30% of coral reefs, 9 and 29% of seagrasses worldwide have been lost or degraded over the last 50–100 years (Valiela et al., 2001; 10 Duarte et al., 2005; Spalding, 2010; Giri et al., 2011). The main activities that contribute to such 11 deterioration are coastal development, over-exploitation, pollution and climate change (Section 5.2.2.2). 12 Over the 20th century, losses of coastal ecosystems such as mangroves, seagrass and saltmarshes were 13 estimated to be 25–50% of their area (McLeod et al., 2011), with a recent loss rate of approximately 0.5– 14 3.0% per year (Pendleton et al., 2012b). At such rates, 30–40% of tidal marshes and seagrasses (IPCC, 2007) 15 and nearly 100% of mangroves (Duke et al., 2007) could be lost in the next 100 years. Globally, 61% of 16 reefs are classified as threatened with 100% of reefs expected to be threatened by 2050 if steps are not taken 17 to reduce local pressure and reduce the emission of greenhouse gases. Quantitative attribution of the relative 18 contribution of climatic and non-climatic human stressors observed changes in coastal ecosystems is difficult 19 because of the interaction between these stressors and the lack of long-term datasets (Burke et al., 2011). 20

21 Global loss and degradation of coastal ecosystems is known to affect at least three critical ecosystem 22 services (Worm et al., 2006; Inniss et al., 2017). Reefs subject to land-based pollution showed a reduction of 23 30-50% in diversity at a 3m depth, and a 40- 60% reduction in diversity at a 10 m depth compared to 24 unpolluted reefs (Cesar et al., 2003a). A meta-analysis of global datasets suggest that the loss of coastal 25 biodiversity impacts the number of viable fisheries (33% decline); the provision of nursery functions (69% 26 decline); and filtering services provided by coastal wetlands (63% decline) (Worm et al., 2006). The loss of 27 biodiversity, ecosystem functions, and coastal vegetation in coastal ecosystems have also contributed to 28 impacts on their recreational and cultural values, biological invasions, carbon capture and sequestration, 29 declining water quality, and decreased shoreline protection (Braatz et al., 2007; Cochard et al., 2008; Koch 30 et al., 2009). More recent studies provide further evidence to the significant relationship between coastal 31 biodiversity and ecosystem functions that support these services (Gamfeldt et al., 2015). Thus, human 32 activities and their consequences, including climate change, are substantially degrading all coastal ecosystem 33 services that are important to human wellbeing (very high confidence). For example, loss of coral reefs may 34 cost around USD 1 trillion globally (Hoegh-Guldberg, 2015). This sub-section focuses on assessing 35 observed and projected impacts of climate change on coastal ecosystem functions and services. 36

37 5.4.1.3.1 Provisioning services

38 Coastal and shelf seas ecosystems provide the largest contribution to global seafood production from capture 39 fisheries and aquaculture (FAO, 2016). These ecosystems provide nurseries, breeding, spawning and 40 hatching sites and through transportation of organic matter to the marine environment, thus providing 41 nutrients for marine and estuarine fauna (Lee et al., 2015). Based on global fisheries catch data from the Sea 42 Around Us project (Pauly and Zeller, 2016), in the 2000s, 67 - 74 million MT of fishes and invertebrates are 43 caught in waters with depth less than 200 m per year. This amounts to >60% of total fisheries catches while 44 marine aquaculture is almost exclusively produced from coastal and shelf waters. Most of these catches are 45 supported by coastal habitat such as coral reefs (Pratchett et al., 2014), seagrass bed (de la Torre-Castro et 46 al., 2014) and mangroves (Carrasquilla-Henao and Juanes, 2017). For example, coral reef fishes are an 47 important nutrient source to many coastal human communities worldwide (Cinner, 2014; Micheli et al., 48 2014; Hoegh-Guldberg et al., 2017). In the Caribbean, coral reef associated fisheries provide net annual 49 revenues estimated at over \$300 million while providing a source of protein for millions and employment for 50 hundreds of thousands of full- and part-time workers (Burke et al., 2004).



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 Bioclimate 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 Bioclimate 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

differences in the magnitude and direction of impacts (likely, see Sections 5.3.2, 5.3.3, Figure 5.15).

related to temperature (indicated by SST) and primary production (indicated by Chlorophyll a

affecting species composition and biomass production in coastal and shelf sea ecosystems with regional

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

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

 $2 \\ 3 \\ 4 \\ 5 \\ 6 \\ 7 \\ 8 \\ 9 \\ 10 \\ 11 \\ 12 \\ 13 \\ 14 \\ 15 \\ 16 \\ 17 \\ 18 \\ 19 \\ 20 \\ 21 \\ 22$

- dominance of warm-water associated species in the coastal and shelf seas regions (Keskin and Pauly, 2014;
 Tsikliras et al., 2014; Maharaj et al., 2018) (*high confidence*).
- $\frac{22}{23}$

1 Loss of critical coastal habitats is a major driver of loss of coastal fisheries productivity. Loss of coral reef 2 habitats driven by climate change (see Section 5.3.3) and the subsequent declines in the physical reef 3 structures have marked effects on the abundance of coral reef fishes and invertebrates, thus impacting 4 ecosystem functions such as resilience and stability, as well as fisheries productivity (Prather et al., 2013; 5 Pratchett et al., 2014) (high confidence). For mangrove forest, spatial comparison between Indo-West Pacific 6 and Atlantic eastern Pacific suggest that higher floral and faunal diversity in the former is correlated with a 7 greater range of species exploited for fuel, timber and crustaceans (Huxham et al., 2017). Global meta-8 analysis of mangrove-fishery linkages show a significant relationship between mangrove area and fishery 9 catches (Carrasquilla-Henao and Juanes, 2017). For example, one hectare of mangrove has been valued at 10 \$37,500 per year when used as a nursery for commercial fishes that will later recruit into adjacent fisheries 11 (Sala et al., 2013). Thus, if climate change reduces the area and/or quality of mangrove forest (see Section 12 5.2.2), potential fisheries catches will also decrease (medium confidence).

14 An ensemble of simulations from multiple models driven by projected changes in temperature, net primary 15 production (and in some models, oxygen, salinity and sea ice extent) from CMIP5 Earth System Models 16 projects a decrease in maximum catch potential by 2.8-8.7% and 2.8-9.1% under RCP2.6 by 2050 and 2095 17 relative to 2000 respectively (FAO, 2018). By contrast, the decrease was larger, 7.0-11.8% and 16.2-25.5% 18 by 2050 and 2095 relative to 2000, respectively, under RCP8.5. These impacts will result in a loss of 19 potential catches of 3.4 million tonnes and changes of 6.4% of the exploited species per degree Celsius 20 atmospheric warming relative to 1951–1960 level (Cheung et al., 2016b) (medium confidence) (Figure 5.15). 21 The interactions between temperature, net primary production and transfer efficiency of energy across the 22 foodweb are projected to amplify these trends, producing changes that may exceed 50% in some regions by 23 the end of the 21st century under a high-emissions scenarios (Stock et al., 2017). Climate change impacts on 24 biological productivity of fish stocks have been shown to interact with fishing (Pershing et al., 2015; Britten 25 et al., 2016) resulting in decline in a decline fish stocks that further reduces fisheries productivity (medium 26 confidence). 27

28 Regionally, tropical and shallower waters are projected to have the largest decrease in maximum catch 29 potential by 2100 relative to the 2000s levels, while catch potential in deeper water of the shelf seas is 30 projected to increase because of a shift in distribution towards deeper, cooler waters (Section 5.2.3; Figure 31 5.15). As a result of changes in species distribution ranges, coastal fisheries resources are also projected to 32 shift geographically towards higher latitude or deeper waters (Figure 5.15). For example, around the Pacific 33 Islands, with projected changes in SST, decreases in surface dissolved oxygen and NPP by $\geq 3^{\circ}C$, ≥ 0.01 ml 34 L^{-1} , 0.5 g m⁻² d⁻¹ by 2100 under RCP8.5, more than 50% of exploited fishes and invertebrates are projected 35 to become locally-extinct in many regions; these factors cause 74% of the area to be projected to experience 36 a loss in catch potential of more than 50%. The area of large projected catch loss is projected to be halved 37 under RCP2.6 (Asch et al., 2017). In West Africa, one model predicts that by 2050, due to climate change 38 the annual landed value of marine fisheries will decline by 21% equating to an annual loss of \$311 million 39 and a 50% loss in fishery related employment (FAO, 2016). Fisheries in vulnerable ecosystems such as coral 40 reef are expected to be particularly impacted by climate change. Healthy reefs in the Caribbean can support a 41 maximum sustained yield (MSY) of 4 tons of fish per km² while degraded reefs were estimated to support a 42 MSY between 0.7 and 2.9 tons per km² (Whalley, 2011). Globally, within 50 years the effects of temperature 43 induced coral bleaching alone are estimated to costs \$105 billion (Cesar et al., 2003b).

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45 Marine aquaculture (i.e., mariculture; marine and brackish) exerts a profound influence as part of global food 46 systems (high confidence), adding about 27.6 million tonnes to total aquaculture production in 2016 (FAO, 47 2016), making a large contribution to food and nutrition security (Béné et al., 2015). Climate change and 48 ocean acidification will impact the sustainability of mariculture production (high confidence). Most 49 mariculture activities take place mainly in sheltered areas of the coastal zone and includes habitats such as 50 estuaries, salt marshes, mud flats and more recently the open ocean. A study on the growth of farmed 51 Atlantic salmon, Cobia and seabream evaluated the effect of future climate trends on optimal growth, 52 concluding that climate change will decrease the species growth potential in many ocean areas where 53 temperature will increase to level that is outside the thermal tolerance range of the species (Klinger et al., 54 2017). Consequently, the decrease in growth could translate into a decrease in the general productivity of the 55 sector. Disease is another major threat that may be exacerbated by extreme weather events and other ocean 56 changes associated with climate change (low confidence). Karvonen et al. (2010) show that increasing water 57 temperature is associated with higher prevalence of the parasite Ichthyophthirius multifiliis and bacteria

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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

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 nonfisheries 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*).

15 5.4.1.3.2 Regulating, supporting and cultural services

16 Climate change affects coastal ecosystem services in addition to fisheries that connect directly to human 17 well-being, health, livelihoods and survival (Cinner et al., 2016). Coastal ecosystems benefit human 18 communities by protecting them from flooding and storm surges and thus preventing loss of life, damages to 19 housing, infrastructure and by preventing saltwater intrusion (Rao et al., 2015). Coastal ecosystems are also 20 important sink of carbon and play a significant role in carbon sequestration and thus climate regulation. The 21 biogenic structures of coastal ecosystems offer habitats that support rich biodiversity and biological 22 productivity witin and outside the ecosystems, and subsequently supporting wide range of provisional and 23 cultural services. 24

25 Mangroves, coral reefs and saltmarshes worldwide regulate the impacts of storm and protect coastline from 26 erosion (Rao et al., 2015) and thus their losses or degradation under climate change (Sections 5.3.3.2, 27 5.3.3.6) increase the risk of impacts on the security of coastal communities (Perry et al., 2018). The roots of 28 mangrove plants help to hold the sediment in place (Orth et al., 2006) while coral reef reduce wave energy 29 on average by 97% (Ferrario et al., 2014) thus substantially protecting coastal populations from natural 30 hazards. Mangrove forests protect inland communities and freshwater resources from saltwater intrusion 31 during storms, and they protect near shore settlements from erosion and reduce flooding. Coastal 32 communities experienced greater damage and higher mortality rates from many types of natural disasters 33 when mangroves had been removed, and the value of these damages ranges widely (Rao et al., 2015). 34 Reduction in storm impacts regulation by coastal ecosystems will also indirectly affect other services as 35 well; for example, the loss of fishing days because of reduction in the number of days with sea conditions 36 that are suitable for fishing is projected to reduce recreational fishing by 10-18% by 2050 in the Southern 37 USA relative to present day under RCP8.5 (Bowker et al., 2014). These regulating services are particularly 38 important under climate change because coastal ecosystems help buffer the impacts of sea level rise and 39 wave action (Duarte et al., 2013) (Section 5.5.2.2).

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41 Carbon sequestration by coastal ecosystems is a key regulatory service that is vulnerable to climate change 42 (Table 5.5). Carbon is biologically sequestered in coastal sediments, commonly known as 'blue carbon' (see 43 Section 5.5.1). Atmospheric carbon is captured by marine plants and stored, primarily in the soil and 44 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 MgC ha⁻¹ for mangroves, 250 MgC ha⁻¹ 45 for salt marshes, and 140 MgC ha⁻¹ for seagrass meadows, equivalent to 1,030 MgCO₂eq ha⁻¹ for 46 47 mangroves, 920 MgCO₂-eq ha⁻¹ for salt marshes, and 520 MgCO₂-eq ha⁻¹ for seagrass bed. The mean carbon storage is 1,494, 951 and 607 MgCO₂-eq ha⁻¹ for mangroves, salt marshes and seagrass beds, 48 49 respectively. The carbon sequestration ecosystem service of mangroves has been valued at \$190 per ha per 50 year and \$220 per ha per year for seagrasses (Trégarot et al., 2017). Carbon burial rates are also affected by 51 variability in hydrological regimes, salinity, nutrient status, and management status. Long-term rates of 52 carbon accumulation in sediments of salt marshes, mangroves, and seagrasses range from 18-1713 gC m⁻² 53 yr^{-1} . Overall, McLeod et al. (2011) estimated blue carbon uptake at about 0.2 GtC 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, multiple by 100 (Pendleton et al., 2012a).

Inputs		Results		
Ecosystem	Global extent (Mha)	Current conversion rate (% yr ⁻¹)	Near-surface C susceptible (top meter sediment + biomass, Mg CO2 ha ⁻¹)	C emissions (Pg CO2 yr ⁻¹)
Mangroves	13.8-15.2 (14.5)	0.7-3.0 (1.9)	373-1492 (933)	0.09-0.45 (0.24)
Tidal Marsh	2.2-40 (5.1)	1.0-2.0 (1.5)	237-949 (593)	0.2-0.24 (0.06)
Seagrass	17.7-60 (30)	0.4-2.6 (1.5)	131-522 (326)	0.5-0.33 (0.15)
Total	33.7-115.2 (48.9)			0.15-1.02 (0.45)

3 Given the substantial contribution of coastal blue carbon to carbon budget, loss of these coastal ecosystems 4 because of climate change (Section 5.3.3) will also directly impact such service. When coastal ecosystems 5 are lost or degraded the huge stocks of blue carbon in the soils are exposed and released as CO_2 into the 6 atmosphere contributing to global warming. Current rates of loss of blue carbon ecosystems may result in 7 0.15–1.02 billion tons of CO₂ released annually. These emissions are equivalent to 3–19% of those from 8 deforestation globally (Table 5.5), and result in economic damages of USD6–42 billion annually. For 9 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 10 11 important roles in coastal carbon budgets, forming critical linkages between rivers, estuaries, and oceans 12 (Najjar et al., 2018). As a result of tidal flooding, saltmarsh soil does not dry out and high levels of carbon 13 can accumulate under anaerobic conditions (Figure 5.16). This is coupled with generally low rates of 14 methane emission, which is strongly limited in saline marshes (*medium confidence*) (Poffenbarger et al., 15 2011; Abella Perez et al., 2016). The carbon burial rate of salt marshes is comparable to mangroves due to high belowground productivity $(218 \pm 24 \text{ gC m}^{-2} \text{ yr}^{-1})$ (McLeod et al., 2011). Despite their importance for 16 17 coastal protection and biodiversity, the global area of saltmarshes has only recently been properly quantified, 18 at around 550 ha, in 99 countries (Mcowen et al., 2017b). Climate change and the temperature-driven 19 displacement of saltmarsh plants by mangrove trees in the mangrove-saltmarsh ecosystem are predicted to 20 increase carbon sequestration in coastal wetlands (Megonigal et al., 2016) (medium confidence). 21

22 Recent expansion of mangroves into saltmarshes are *likely* to have large impacts on the structure, function 23 and service provisioning of coastal wetlands (Kelleway et al., 2017b). Although mangrove encroachment 24 may increase nutrient storage and improve storm protection (Sheng and Zou, 2017), the declines in 25 saltmarshes will reduce habitat availability for fauna requiring open vegetation structure, and consequently 26 the recreational and cultural activities associated with this fauna (Kelleway et al., 2017b). The impact on 27 provisional services such as fisheries productivity and cultural services is projected to be site-specific and 28 dependent on the species involved (Kelleway et al., 2017b).

Chapter 5



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.

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

1 recent study by Deloitte Access Economics, Australia's Great Barrier Reef contributed \$29 billion through 2 tourism, \$24 billion through non-use values, and \$3 billion to recreation activities while also supporting 3 64,000 jobs (O'Mahoney et al., 2017). Globally, around 30% of the world's reefs are of value in the tourism 4 sector, with a total value estimated at nearly US\$36 billion, or over 9% of all coastal tourism value in the 5 world's coral reef countries (Spalding et al., 2017). In the Caribbean 2.4 million divers per year contributed 6 \$2.1 billion to the region's economy (Burke et al., 2004). Further, mangroves in Southeast Asia have been 7 estimated to provide \$10,821 per hectare per year in storm protection services (Sala et al., 2013). It should 8 also be taken into consideration that in many instances the values for environmental services are estimated in 9 a simplified way, thus not reflecting their real values (Rinkevich, 2015b). Moreover, most of the services 10 provided by any specific ecosystem are usually not considered when employing conventional macro-11 economic indicators (such as the Gross Domestic Product) (TEEB, 2010), since many of the goods and 12 services are not traded in markets. 13 14 Another important class of reef services is coral reef tourism marked with a global value of US\$ 36 billion 15 per year (Spalding et al., 2017). This is a reef service highly impacted by global changes (*high confidence*). 16 A direct relationship between coral cover and recreational use (a subsector in the tourism industry) can be 17 drawn (while varied between sites) revealing that all types of tourist activities are highly vulnerable to 18 climate change impacts on the coral reefs (Biggs et al., 2015; Lane et al., 2015). All above ecosystem 19 services in the coral reefs can be significantly impacted from the increased frequency and intensity of 20 extreme climatic events that have emerged from global warming. These impacts are not easy to model, since 21 current impact assessment models are largely based on gradual warming trends (Wernberg et al., 2012). 22 23 Overall, climate change impacts on coastal ecosystems are impacting all ecosystem services offered by 24 coastal ecosystems (high confidence). Particularly, loss and degradation of coastal ecosystems under climate 25 change impacts the availability and quality of habitats of fish and invertebrates populations (very high 26 confidence), consequently reducing the diversity and productivity of fisheries resources that support 27 commercial, subsistence and recreational fishing (high confidence). However, the level of impacts varies 28 regionally, with tropical coastal ecosystems being the most at risk to such impacts (high confidence) relative 29 to those in mid-latitude regions (medium confidence). Loss coastal ecosystems, particularly coral reef, 30 mangroves and saltmarshes, as a result of sea level rises and increased extreme events increases the exposure 31 of coastal communities to storms, erosion and saltwater intrusion because of the loss of their coastal 32 protection services (high confidence) and contributes to the loss of carbon sink (medium confidence). The 33 climate change-induced degradation of these regulating services that are important for moderating climate 34 impacts represent a positive feedback that further exacerbates the risk of climate change (medium 35 *confidence*). Intrinsic values and cultural importance of coastal ecosystems, such as indigenous culture, 36 recreational fishing and tourism, that are dependent on their biodiversity and ecosystem functions as well as 37 the regulating and supportive services provided by these ecosystems are also at risk to climate change 38 impacts (high confidence). 39

40 5.4.2 Climate Risk, Vulnerability and Exposure of Human Communities and their Wellbeing 41

42 Human communities are heavily dependent on the oceans through the goods and services that marine 43 ecosystems offer (Section 5.4.1) (Hilmi et al., 2015a). A recent report estimated that the total asset from 44 these ocean-based economic sectors is at least US\$24 trillion, generating hundreds of millions of jobs (WWF 45 2015). As climate change is impacting marine biodiversity and ecosystem services (Section 5.3.1), various 46 dimensions of benefits that contribute to sustaining human communities and their wellbeing will also be 47 affected. However, detection and attribution of climate change impacts on the wellbeing of human 48 communities are confounded by wide-ranging non-climatic factors that strongly affect the exposure, 49 sensitivity and adaptive capacity of human communities to climate impacts.

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51 Therefore, the assessment in this section are based on studies that provide diverse types of information from 52 quantitative modelling to qualitative studies using expert opinion, local experience and Indigenous 53 knowledge. Projection and assessment of risk and vulnerabilities are also strongly dependent on assumptions 54 about future social-economic development. Specifically, the risk of climate impacts on human communities 55 are based on assessments on the level of dependence of ecosystem services, and the risks of climate impacts 56 of these ecosystem services (Section 5.4.1). The Shared Socio-economic Pathways (SSPs) is referred to for 57 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

wellbeing 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

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

achieving the United Nations' Sustainable Development Goals (SDGs) (Section 5.4.2.4).

development through the ocean contextualized by evaluating the climate effects on the possibility of

cultural (Section 5.4.2.1), economic (Section 5.4.2.2) and environmental (Section 5.4.2.3), encompassing the

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Table 5.6: The social, economic and environmental dimensions of sustainable development and examples of their linkages to marine account marines

Pillars of	Dimensions	Key linkages to	High-dependent	Key hazards
sustainable	assessed	ecosystem services	communities	
development				
Social and cultural	Human diseases (5.4.2.1.1), pollution (5.4.2.1.2), security and conflicts (5.4.2.1.3), Indigenous culture (5.4.2.1.4) and education (5.4.2.1.5)	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	Coastal Indigenous communities and the developing countries with lack of access to education, health care and infrastructure	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 entitities
Economic	Fisheries (5.4.2.2.1), food security (5.4.2.2.2), tourism (5.4.2.2.3)	Fisheries and aquaculture, coastal protection, climate regulation, tourism	Fishing, aquaculture and ecotourism sectors, coastal Indigenous communities and the global south, vulnerable population of mega-cities, poorest strata of society	Decrease in fisheries and aquaculture profits, decrease in the quantity and quality of seafood supply, degradation or loss of tourist attributions
Environments	Aesthetic aspects, property values, development of coastal cities (5.4.2.3.1), energy and mining (5.4.2.3.2)	Coastal protection, climate regulation, intrinsic values of biodiversity	Coastal Indigenous people coastal populations in poverty	Increased exposure to storm, flooding and coastal erosion, loss of biodiversity and habitats.

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5.4.2.1 Social and Cultural Dimensions

17 18 The sustainable development paradigm includes domains at the intersection between the societal and the 19 natural realms, focusing on human engagement with and within nature, including the built environment. 20 Further, the linkages between the health of the environment, ecosystem health and human health are 21 considered. This sub-section will focus on different dimensions of risks on human health and security as a 22 result of climate change effects on the oceans, biodiversity and ecosystem services. Specifically, the 23 assessment focuses on ecosystem-mediated impacts of climate change, impacts of pollutants other than CO₂ 24 and climate impacts on human security and conflicts. The cultural aspects concern indigenous culture and 25 education. Climate change will also impact cultural dimensions. 26

27 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*

1 bacteria, which is one of the most common water-borne human diseases and are considered as major health 2 concerns (Kirk et al., 2015). Assessment of HABs is described in Box 5.3. 3 4 Around twelve out of over 100 species of *Vibrio* have caused human infections, mainly via exposure to 5 seawater or consumption of raw or undercooked seafood with the bacteria (Austin, 2005). Common diseases 6 caused by Vibrio include necrotizing wound infections, primary septicemia, ear infections, and seafood-7 related food-poisoning. Elderly and other individuals with suppressed immune system are particularly 8 vulnerable to such infection (Daniels, 2011). Globally, Vibrio is causing substantial epidemiological risk. 9 For example, Vibrio cholera (causing Cholera) is estimated to be responsible for around 760,000 and 10 650,000 cases of human illness and death in the world in 2010 (Kirk et al., 2015). Naturally, the 11 nontoxigenic Vibrio species are prevalent worldwide in warm, nutrient-rich and low salinity coastal waters. 12 AR5 concludes that warming, excessive nutrient and seawater inundation due to sea level rise are projected 13 to exacerbate the expansion and threat of cholera (medium confidence, AR5 WGII Ch. 6). 14 15 New evidence since AR5 continues to support the linkages between climate change and increased risk of 16 diseases caused by Vibrio bacteria. Further analysis of time-series data confirms an increase in the 17 prevalence of Vibrio that is related to warming (Baker-Austin et al., 2017). For example, analyzing the 18 Continuous Plankton Recorder dataset (Section 5.2.3) with new molecular techniques detected a significant 19 increase in Vibrio abundance in the North Sea since the 1980s that is related to sea surface warming 20 (Vezzulli et al., 2016). Recent review also suggests a poleward expansion of Vibrio pathogens to areas with 21 no previous incidence, particularly in mid- to high-latitude regions; such expansion is suggested to be partly 22 driven by climate change, particularly in areas where rapid warming is taking place (Baker-Austin et al., 23 2013; Baker-Austin et al., 2017). Simultaneously, extreme weather events such as flooding and tropical 24 cyclones are linked to increased incidences of Vibrio-related diseases; Such increases in diseases are 25 suggested to be caused by the increased exposure of human populations to the pathogens during these 26 extreme events (Baker-Austin et al., 2017). Particularly, coastal floods are associated with intrusion of 27 seawater and wastewater into fresh water, leading to water shortage and risks of water-borne diarrheal 28 disease (Kim et al., 2014). Floods are also related to displacement of human populations, which is known to 29 increase vulnerability of populations and to contribute to the spread of infectious diseases. Therefore, recent 30 ocean warming has led to the increase prevalence and expansion of Vibrio pathogens (very high confidence). 31 Extreme weather events increase the exposure of human populations to Vibrio, potentially contributing to the 32 increased incidence of Vibrio diseases (medium confidence). However, the global surveillance of the disease 33 in human population is limited and the nontoxigenic Vibrio infection is rarely notifiable to the health 34 agencies, leading worldwide to limited understanding of the disease epidemiology and risks, and their links 35 to climate change (low confidence). 36 37 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

38 39 inputs would have an increase in prevalence of Vibrio pathogens. For example, in the coastal Baltic Sea, 40 Vibrio is expected to grow faster in coastal areas where, under climate change projections, waters are 41 expected to be warmer, increase in salinity due to reduced precipitation and have higher chlorophyll 42 concentrations (Escobar et al., 2015) resulting in an elevated risk of Vibrio infections (Semenza et al., 2017). 43 Areas that are projected to have elevated frequency of coastal floods from storm surges and sea level rise in 44 response to future climate change (Section 5.2.2) will also increase the risk of Vibrio disease as a result of 45 human exposure to the pathogen (medium confidence). However, uncertainty in the social and economic 46 developments shaping vulnerabilities of populations makes estimates of absolute mortality highly uncertain 47 (Lloyd et al., 2016).

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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).

Chapter 5

1 Overall, increases in the prevalence of pathogens for some water-borne human diseases that have large 2 public health impacts are related to climate change (high confidence), and their risks are projected to 3 increase in the 21st century (*medium confidence*). However, health risks of these pathogens are strongly 4 influenced by economic and social factors, and thus posing a major uncertainty for the projection of the 5 future risk of these diseases. The risk of these diseases may also interact indirectly with other climate change 6 impacts on marine biodiversity and ecosystems. For example, because of the loss of marine biodiversity due 7 to climate change (Sections 5.2.3, 5.3.3), the chances to discover new bioactive compounds will decrease 8 (Section 5.4.1). Many traditional medicines coming from the sea may also disappear. This can potentially 9 have an impact on human health in general, and more specifically, the discovery for effective prevention and 10 treatment for water-borne diseases (Malve, 2016) (low confidence). 11

13 [START BOX 5.3 HERE] 14

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15 Box 5.3: Harmful Algal Blooms and Climate Change

16 17 Harmful algal blooms (HABs) are proliferations of phytoplankton species that have negative effects on 18 marine environments and associated biota, with impacts ranging from water discoloration and foam 19 accumulation, anoxia, contamination of seafood with toxins, disruption of food webs, and massive 20 invertebrate and fish kills (Hallegraeff, 2010). Since the 1980s, the HABs have increasingly affect human 21 society (Anderson et al., 2015). For example, the economic cost of HABs in the United States alone is 22 estimated to be \$100 million per year (in 2012 dollars). These costs mainly resulted from impacts on public 23 health, commercial fisheries and recreation and tourism (Berdalet et al., 2017). In AR5 it was concluded that 24 'harmful algal outbreaks are increasing their frequency and intensity stimulated by warming, nutrient 25 fluctuations in upwelling areas, and coastal eutrophication (medium confidence); however, there is limited 26 evidence and low confidence in how global climate change will affect HABs' (AR5 WGII Ch. 5, 6). 27

28 Since AR5, more observational and experimental evidence allows for improved detection and attribution of 29 HABs to climate change. The main hypotheses explaining the increased HAB-related impacts include 30 increased human influences on the marine environment through nutrient enrichment, pollution, aquaculture, 31 shipping, and climate change (Lassus et al., 2016). A Global HAB Status Report initiative by the 32 International Oceanographic Commission and The United Nations Educational, Scientific and Cultural 33 Organization (IOC UNESCO) and hosted within the International Oceanographic Date Exchange (IODE), 34 has generated global datasets on HABs (Moestrup (2009); http://www.marinespecies.org/hab) to unify toxic 35 species nomenclature and track their geographical prevalence. Emerging climate-driven responses identified 36 after field and experimental work include range expansion of warm-water HAB-related phytoplankton (e.g., 37 Gambierdiscus, producer of ciguatoxins that causes ciguatera fish poisoning, CFP) and contraction of cold-38 water species (medium confidence) (Tester et al., 2010; Rodríguez et al., 2017), regional rises in abundance 39 and seasonal bloom window (medium confidence), and increased cellular toxins of key species 40 (Alexandrium, Pseudo-nitzschia, Karlodinium) due to combined effects of warming and acidification; low 41 confidence) (Fu et al., 2012). Recent experiments with large mesocosms enclosing natural planktonic 42 communities in subtropical waters, simulating CO₂ emission scenarios between RCP2,6 and RCP8.5, found 43 that acidification promoted the development of the toxic microalgae Vicicitus globosus, with severe negative 44 impacts over other co-existing phytoplankton species and in general over the whole planktonic food web 45 (Riebsell, in press). Furthermore, the implementation of precise techniques for the quantification and 46 identification of novel phycotoxins and toxic species in underexplored areas of the Ocean (e.g., in the South 47 Atlantic: Akselman et al., 2015; Tillmann et al., 2016; Guinder et al., 2018), and experiments to evaluate the 48 production of toxins in the presence of grazers (Tammilehto et al., 2015; Xu and Kiørboe, 2018) and 49 changing nutrient levels (Van de Waal et al., 2013; Brunson et al., 2018), provide evidence for the 50 significant role of climatic and oceanographic drivers in inducing HABs and their degree of impact (medium 51 confidence). For example, the biosynthesis of domoic acid by some *Pseudo-nitschia* species is induced by 52 combined phosphate limitation and high CO₂ conditions (Brunson et al., 2018). The growth rate of the toxic 53 dinoflagellates Alexandrium (producer of paralytic shellfish poisoning, PSP toxins) and Dinophysis 54 (producer of okadaic acid) are enhanced by warmer conditions. Therefore, the biogeographic expansion of 55 these dinoflagellate's ecological niche has been observed in the North Atlantic and North Pacific and are 56 related to increasing sea surface temperatures (Gobler et al., 2017). Similarly, in the Northern California 57 Current, environmental conditions linked with warm phases of ENSO are associated with the production of

1 the neurotoxin domoic acid (DA) by species of Pseudo-nitzschia (McKibben et al., 2017), with devastating 2 effects on coastal ecosystems (McCabe et al., 2016). Finally, in the coastline of the Canary Islands, modelled 3 distribution of the harmful benthic cyanobacteria Lyngbia majuscula predicts expansion of the population 4 linked to changes in topographical and wave exposure. The development of dense mats of Lyngbia implies 5 high risk for local rocky and sandy ecosystems as well as seagrass meadows, as they affect the sedimentation 6 processes and the structure of benthic communities (Martín-García et al., 2014). Overall, these observed 7 changes in biogeography and phenology of HAB-related species are consistent with general responses of 8 phytoplankton to climate change (Section 5.2.2) (medium confidence), while physiological studies confirm 9 elevated cellular toxins production by HAB species under high CO₂ conditions and nutrient-limited 10 conditions (medium confidence) (Fu et al., 2012; Brunson et al., 2018). 11 12 Regional variations of the trends in the occurrences of HABs could be explained by the spatial differences in 13 key climate drivers of temperature, water column stratification, ocean acidification (increased CO₂) and 14 extreme weather events (e.g., precipitation) (Hallegraeff, 2010). Particularly, these algal phenomena are 15 common in coastal areas where the exposure of organisms to climate threats is more variable due to 16 interactive effects with human-induced environmental changes. Thus, the occurrences of HABs are largely 17 dependent on local-scale emergence of changing ocean conditions (Box 5.1). 18 19 HABs occurrences, their toxicity and risk on natural and human systems are expected to continue to increase 20 under climate change in the 21st century. Based on observations and simulation models, occurrence of HABs 21 is projected to continue with warming and rising CO₂ (high confidence) (Glibert et al., 2014; Martín-García 22 et al., 2014; McCabe et al., 2016; Paerl et al., 2016; Gobler et al., 2017; McKibben et al., 2017; Rodríguez 23 et al., 2017; Riebsell, in press). The worldwide distribution of the key toxic species of Alexandrium and 24 Pseudo-nitzschia and the projected increase of anomalous warming events, suggest an intensification of the 25 HAB-related risk for coastal biodiversity and ecosystems services particularly those that are naturally 26 sensitive to the impacts of HABs. For instance, more severe risk is expected for estuarine organisms such as 27 ovsters, 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).
- 31 32 Overall, improved observational and experimental evidence and projection support increasing incidence and 33 intensity of HABs that could be attributed to climate change and its interactions with other human drivers 34 (high confidence). Global predictive capability is hindered by contradictory species and strain-specific 35 responses, lack of insights into evolutionary adaptation, long-term (>30 years) data streams and climate 36 variability at a regional scale. The greatest problems for human society will be caused by being unprepared 37 for range expansions or increase of biotoxins in poorly monitored areas. Therefore, action for adaptation and 38 mitigation relies on sustained monitoring programs at a local scale of phytoplankton at the species level, and 39 of chemical and physical conditions (Anderson et al., 2015). Early warning systems of potential harmful 40 outbreaks can alert resource managers and stakeholders and ameliorate the risk for human wellbeing. In 41 addition, the reduction of coastal pollution and eutrophication can mitigate the synergistic effects with 42 climate hazards and partially reduce the severity of the ecosystem damage produced by HABs, but whether 43 the reduction of anthropogenic stressors will be enough to adapt the risk of climate hazards on HABs 44 remains uncertain (Section 5.5.2).
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- 46 [END BOX 5.3 HERE]
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49 5.4.2.1.2 Interactions between climate change and non-CO₂ pollutants

50 Climate change–contaminant interactions may alter the bioaccumulation and biomagnification of two 51 contaminant classes: the toxic and fat-soluble persistent organic pollutants (POPs), such as polychlorinated

52 biphenyls (PCBs), as well as the neurotoxic and protein-binding organic form of mercury, methylmercury

53 (MeHg) (Alava et al., 2017) (*medium confidence*). POPs are bioaccumulated by marine organisms and

- 54 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
- 56 lead to serious health effects including certain cancers (i.e., by PCBs and dioxins), birth defects, and
- 57 impairments to the immune and reproductive systems, as well as greater susceptibility to disease and

1 damages to the central and peripheral nervous systems (Lallas, 2001). MeHg is highly neurotoxic and 2 nephrotoxic and bioaccumulates and biomagnifies throughout the food web via dietary uptake (Fort et al., 3 2015). Consumption of mercury-contaminated fish is linked to disease in humans, including the neurological 4 disorders due to MeHg poisoning (i.e., Minamata disease) (Ishikawa and Ikegaki, 1980; UNEP, 2013). Of 5 particular concern is the pollution risks influenced by climate change in the Arctic ecosystems and 6 indigenous communities because of the protracted bioamplification of POPs and MeHg with associated 7 long-term contamination of their traditional foods (Margues et al., 2010; Tirado et al., 2010; Alava et al., 8 2017).

9 10 In the face of ocean warming and acidification, the exposure, persistence, bioaccumulative capacity and 11 toxic nature of these pollutants can be further exacerbated by climate-change pollutants interactions 12 (McKinney et al., 2015; Alava et al., 2017) (medium confidence). Scientific evidences for human and 13 ecotoxicological risk assessments highlight that climate change increases the exposure and bioaccumulation 14 of contaminants and thus the risk of impacts of POPs and MeHg on marine ecosystems and their dependent 15 human communities (Marques et al., 2010; Tirado et al., 2010; Alava et al., 2017) (high agreement, medium 16 evidence). For example, mercury methylation processes in aquatic environments is expected to be 17 exacerbated by ocean warming coupled with more acidic and anoxic sediments. Mercury and some metals 18 (e.g., Al, Cu, Fe, Pb, Zn) are generally more bioavailable in acidified aquatic habitats (Scheuhammer, 1991; 19 Celo et al., 2006; López et al., 2010; Roberts et al., 2013; de Orte et al., 2014). Increased global temperatures 20 may also increase the metabolic activity of bacteria which will augment the cycling and conversion of 21 mercury into MeHg (Macdonald and Loseto, 2010; Corbitt et al., 2011; Krabbenhoft and Sunderland, 2013). 22 Furthermore, increased melting of snow and ice from alpine ecosystems and mountains (Chapter 2) can also 23 increase the release of POPs and mercury from land-based sources into coastal ecosystems (Morrissey et al., 24 2005). Ultimately, mercury concentrations in fish are inversely related to pH in acidic water, which would be 25 due to the higher solubility of mercury in acidic water and enhancement of mercury methylation rates at 26 lower pH (Lean, 2003) as well as increased mercury uptake under warming (Riget et al., 2010; McKinney et 27 al., 2015). Modelling projection suggest increased bioaccumulation of MeHg in the Faroe Islands region 28 under climate change, with an associated increase in human exposure to this metal from the consumption of 29 pilot whale (Globicephala melas) (Booth and Zeller, 2005). 30

31 Climate change alters the degree of human exposure to pollutants and the response of human populations to 32 this exposure (Balbus et al., 2013) (medium confidence). Changing human behaviour will also affect how 33 humans come into contact with contaminated air, water, and food in an era of climate change. Climate 34 change and regional climate variability may have an impact on the occurrence of food safety hazards, 35 including changes in transport pathways for contaminants, at various stages of the food chain, from primary 36 production to consumption (Marques et al., 2010; Tirado et al., 2010). While species inhabiting 37 industrialized areas may be most at risk for contaminants, and the vulnerability of the Arctic climate 38 suggests that the latter region may be particularly sensitive and susceptible to climate-contaminant 39 interactions (Chapter 3). In this context, contaminant concentrations in marine mammal species and wildlife 40 populations from the Arctic already exceed overall biological levels of concern for POP exposure, i.e., 1 mg 41 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, 42 43 ultimately affecting global fisheries. This leads to potential health impacts for indigenous communities in a 44 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

18 Climate change may generally increase conflict risks (Gilmore, 2017) and there is emerging empirical 19 evidence that relates climate change and conflicts to fisheries and oceans. Conflicts can be defined as events 20 involving individuals or groups of individuals (organized or disorganized) in which disputes fail to be 21 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.

24 Climate change effects on ocean conditions (Section 5.2.2) can create or exacerbate conflicts over fisheries 25 as a result of shifts in distribution of fish stocks and the fluctuations and decreases in their abundance and 26 potential catches (Spijkers and Boonstra, 2017), as well as the movements of fishers (Belhabib et al., 2016; 27 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 28 transboundary fish stocks under climate change alter the sharing of fisheries resources between countries 29 (Miller and Russ, 2014). Under climate change, new fishing opportunities will be created when exploited

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1 fish stocks shift their distribution into a country's waters where they did not previously occur in sufficient 2 abundance to support viable fisheries before (Pinsky et al., 2018). Such shifts in transboundary resources 3 have occurred before and led to international disputes in some cases (Spijkers and Boonstra, 2017; Pinsky et 4 al., 2018). For example, shifting transboundary stock distributions of Pacific salmons and Atlantic mackerel 5 contributed to international conflicts in exploitation and setting catch quotas for these fisheries. Moreover, 6 the number of new transboundary stocks occurring in exclusive economic zones was projected to be around 7 46 and 60 under RCP2.6 and RCP8.5, respectively, by 2060 relative to 1950-2014 (Pinsky et al., 2018). 8 However, most existing international fisheries governance legal framework does not have explicit provisions 9 to directly account for these shifting fish distributions and abundance. Without a climate-considered 10 governance framework, chances for disputes in the sharing and exploitation of these fisheries resources are 11 higher (high confidence), consequently increasing the risk of overfishing (high confidence) and fractured 12 international relationships, and political conflicts that could spill over into other, non-fishery areas of 13 international politics (low confidence).

14 15 Extreme events associated with climate change, such as hurricanes and floods, can exacerbate tensions 16 among 'trapped populations' unable to afford a move out of vulnerable areas (McGranahan et al., 2007; 17 Adger et al., 2014). Climate change is projected to increase the risk of coastal communities to extreme 18 events such as hurricanes and coastal flooding (Chapter 6). A substantial contribution of such risk is through 19 increased exposure due to human populations settling in vulnerable areas, particularly the poorer or marginal 20 groups in the society (Adger et al., 2014). These vulnerable populations have lower capacity to adapt 21 through migration and the increased impacts of extreme events can exacerbate inequality within the society, 22 and consequently, increase the risk of social unrest and conflicts (medium confidence). In addition, 23 competing use of natural resources internationally may also increase the risk of these conflicts. For example, 24 increased exploitation of marine resources such as fisheries sometimes offer a coping mechanism to 25 communities affected by extreme events; however, opportunistic expansion of foreign fishing into impacted 26 areas may impact fisheries and exacerbate social tensions (Belhabib et al., 2018) (low confidence). Overall, 27 high vulnerability and lack of adaptive capacity to climate change impacts (including fisheries-dependent 28 livelihoods, attachment to place, and pre-existing tensions) increase the risk of conflicts, including among 29 fishery area users and authorities (Ndhlovu et al., 2017; Shaffril et al., 2017a; Spijkers and Boonstra, 2017). 30

31 Population movements associated with climate change impacts can result in conflicts in areas receiving 32 displaced fishing communities (Reuveny, 2008). Negative inland climate effects can push some communities 33 to coastal areas, exacerbating pressure on local fisheries and social tensions between long-term coastal 34 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 35 36 countries (Cattaneo and Peri, 2016), and while many cities are located in coastal areas the impact of 37 outmigration to coastal fishing communities is not well documented. Also, the relationship between climate 38 change, migration and conflicts in unstable urban and rural demographics may be weak and short-term (Fort 39 et al., 2008) and affected from sampling and reporting biases (Adams et al., 2018). The impacts of these 40 conflicts on communities and their wellbeing can include loss of livelihood and access to fish, forced 41 displacement and community breakdown, as well as the physical and psychological harms of violence 42 (medium confidence).

43 44 Other pathways of increased risk of conflicts under climate change include territorial claims and mal-45 adaptation policies. Sea level rise can also lead to conflicts over the redefinition of territorial claims between 46 countries, especially in areas where large areas of the exclusive economic zones or the very existence of 47 countries are at stake (Maas and Carius, 2012; Stoutenburg, 2015), or maritime sovereignty is frequently 48 disputed (Nyman, 2013). Climate protection, exogenous adaptation policies, and marine conservation 49 measures for coastal populations can induce social disruptions, exacerbate inequalities and delegitimize 50 authorities, thereby increasing the risk of conflicts (Bennett and Dearden, 2014; Kamat, 2014; Sovacool, 51 2018). 52

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

through competing resources exploitation from international actors and mal-adapted policies (low confidence).

4 5.4.2.1.4 Indigenous cultures

5 Indigenous communities can be found in all continents except Antarctica (UN, 2009). For Indigenous 6 communities along the coast, recent estimates suggest that there are more than 1900 groups with around 27 7 million people across 87 countries (Cisneros-Montemayor et al., 2016). The cultures and languages of 8 Indigenous peoples are responsible for much of the cultural diversity that continues to exist (UN, 2009). 9 Many of these communities exist on their traditional lands. The Declaration on the Rights of Indigenous 10 Peoples outlines minimum standards 'for the survival, dignity and well-being of the Indigenous peoples of 11 the world' (UNPFII, 2007), including the right of self-determination. This enables Indigenous peoples to 12 'freely determine their political status and freely pursue their economic, social and cultural development'. 13 Particularly, a fundamental component of human well-being is culture, particularly Indigenous culture. 14 However, marginalisation makes their lives, values and cultures vulnerable to changes in natural resources 15 (Wessendorf and Garcia-Alix, 2009). At the same time, Indigenous values and culture are an important 16 enabling component for sustainable development, particularly in offering different foundations for and 17 perspectives of what sustainable development means (Watene and Yap, 2015). However, climate change is 18 affecting different aspects of Indigenous culture that would also affect sustainable development (Cross-19 Chapter Box 3 in Chapter 1). This assessment particularly focuses on the perspectives for coastal Indigenous 20 people in general. Specific aspects related to Arctic Indigenous people are discussed in SROCC Chapter 3. 21 Because Indigenous culture varies greatly between regions and the assessment requires specific context, 22 therefore, we draw experience Indigenous culture under the changing ocean and marine ecosystems from the 23 perspective of the Pacific.

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25 The case of the Pacific is a good example for illustrating the effects of climate change on Indigenous 26 knowledge, the transmission thereof, and ocean education. This is because in the Pacific the (climatic) 27 changes are different in nature and refer to different time frames (see also Cross-Chapter Box 3 in Chapter 28 1). Geological and climatic history refers to time intervals of thousands or even millions of years. Traditional 29 knowledge, on the other hand, relates to shorter periods ranging from several generations to a few centuries. 30 On the other hand, Indigenous knowledge, even if it has been transmitted for several generations, can be 31 appreciated over shorter periods ranging from a few centuries to several generations. Thus, the adjustment of 32 the transmission and of the network of Indigenous and local knowledge on the ocean, of the related 33 perceptions and practice, implies a reworking of these knowledge systems where the individuals and the 34 groups are actors in a narrative and historical construction (Roué, 2012; Alderson-Day et al., 2015). In this 35 sense, the latest surveys conducted in Kiribati and the Tuamotu archipelago in French Polynesia (Tamatoa 36 and Latouche, 2016) highlight the gap between the scientific discourse predicting unprecedented climate 37 change and the local perceptions that consider it to be an 'outside' discourse in a context where science has 38 often been considered as the ally of a historic colonial power (Tuhiwai, 2013). Nevertheless, climate change 39 is already transforming the shape of seashores in many low islands. Some sandbank and rocks representing 40 gods and mythological ancestors are disappearing (Camus, 2017). Other studies in Tuvalu show that during 41 the past four decades, sea level rise and climate change have resulted in a net decrease in land area (Kench et 42 al., 2018) affecting the mobility of people, residence patterns, overcrowding, and the structure of Indigenous 43 knowledge and local knowledge in the sense that more mythological syncretism may be produced by the 44 concentration of people in the safest places.

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46 Thus, the great myths of foundation and the ocean that play a central role in these communities are re-47 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 48 49 change, but because the local populations observe that climate change induces change in their environment. 50 For example, the scarcity of pelagic resources and lagoons, and the disappearance of a mythical sandbank at 51 Tabiteua (Kiribati) altered the historical trajectory of a population and its relationship to the ocean forever 52 (Camus, 2017). 53

54 In addition, in the Pacific as elsewhere, there is a shift in the transmission of local knowledge about the 55 ocean, from a family monopoly to a plurality of actors including compulsory schooling. The fear of sea level 56 rise and climate change encourage security measures and the grouping of local people to the safest places. 57 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 (Alì, 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.

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7 This paradoxical situation, however, has encouraged Pacific peoples to embrace climate change not in the 8 terms of science, but according to their cultural categories and patterns (low confidence). For example, many 9 Polynesian and Micronesian societies have been involved in the construction of double canoes reminiscent 10 of major exploration voyages. Today, they contribute to re-enactment of old sailing techniques while at 11 every stage of their journey they strive for a respectful relationship between society, the ocean and the 12 climate. In the same way, the last ten years have seen the introduction of ancestral techniques for the 13 conservation of lagoon resources, called rahui in eastern Polynesia (D'Arcy and Bambridge, 2014), now 14 extended to previously unknown scales. The Cook Islands protect their exclusive economic zone, with the 15 establishment of a rahui, marae o te moana-the sacred temple of the ocean-over an area of 1.9 million 16 km². French Polynesia has pledged to do the same on an area of 5.5 million km². This vast historical 17 movement goes beyond a simple goal of adaptation to climate change. It is also a question of restoring an 18 Oceanian identity to a space in which they consider themselves to be the historical guardians (Hilmi et al., 19 2016). Finally, 'environmentally displaced' Pacific Islanders are acquiring land in the Pacific regions 20 bordering their areas (New Zealand, Australia, and the USA) to secure the future of next generations. 21

22 AR5 concluded that climate change will affect the harvests of marine species with spiritual and aesthetic 23 importance to indigenous cultures. Recent research has further demonstrated that Indigenous fishing 24 communities that depend on traditional marine resources for food and economic security are particularly 25 vulnerable to climate change through a reduced capacity to conduct traditional harvests because of limited 26 access to, or availability of, resources (Larsen et al., 2014; Weatherdon et al., 2016) (medium confidence). 27 For instance, shellfish and traditional clam beds-which form an integral part of the culture, economy and 28 diet of many Indigenous communities situated along the Pacific Coast of North America- may be affected 29 detrimentally by increased SST, sea level rise, and changes in ocean chemistry and circulation patterns 30 (Lynn et al., 2013). Indigenous people can also be highly dependent on various marine mammals that are 31 vulnerable to climate change impacts. Any change in numbers, quality or access to these mammals (for 32 example through die-off of walrus from St. Lawrence Island) increases the vulnerability of local Indigenous 33 communities (Lynn et al., 2013) (medium confidence). 34

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*).

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41 *5.4.2.1.5 Other cultural dimensions*

42 Among the aspects of cultural wellbeing supported by the ocean, education based on scientific knowledge of 43 the marine environments, and the economic and technical benefits generated by exploration and discover 44 play an important role. There are also aesthetic and inspirational values of marine biodiversity and 45 ecosystems that are important to the psycological and spiritual well-being of people, including film, 46 literature and art, and recreation (Pescaroli and Magni, 2015). The appreciation of scientific, artistic, 47 spiritual, and health opportunities, as well as the appreciation of biodiversity, lifestyle and aesthetics, as well 48 as intrinsic values, are other cultural dimensions becoming more widely acknowledged as key cultural 49 dimensions potentially disturbed by climate change (Marshall et al., 2018). While climate change is 50 impacting marine ecosystems (Sections 5.2.3, 5.3.3), their support to human well-being through the cultural 51 dimensions could also be affected.

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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.

9 Climate change will affect the perception of local communities about the ocean (low confidence). For many 10 island nations, their deep ocean is hundreds to thousands of times larger than their land, and most of their 11 natural capital lies in the deep sea. Some such as Rapa Nui (Easter Island) have a strong spiritual connection 12 with this element. A reaction common to many low islands Pacific countries such as Kiribati, Cook island, 13 Pascua Island and French Polynesia, is to create huge large marine protected areas as an adaptation measure 14 to climate change (Bambridge and Le Meur, 2018). Marine Protected Areas also offer a cultural opportunity 15 in impacted areas to constructively and collaboratively discuss adaptation plans and plans to increase 16 resilience, whilst simultaneously reinforcing cultural identity and pride. This in return will affect the catch in 17 the Pacific and the economy of long distant fleets for example. 18

19 Climate change threatens many cultural dimensions of lives and livelihoods (Adger et al., 2012) (low 20 agreement, low confidence). People develop strong cultural ties and associate distinctive meanings with 21 many natural places in the form of traditions, customs and ways of life (Marshall et al., 2018). Nature 22 presents a place within which people can belong, whilst offering other places to visit and be inspired by. 23 Nature provides a day-to-day sense of meaning through providing opportunities for intellectual engagement, 24 shared recreation, economic well-being and support for livelihoods. The cultural opportunities provided by 25 ecosystems are many, and include concepts such as attachment to place, heritage, rootedness, spirituality, 26 tranquility, escape, togetherness, discovery, knowledge, health, judgement, aesthetics, recreation, education, 27 community development, lifestyle, identity, and emotional/psychological well-being (Chan et al., 2012; Fish 28 et al., 2016; Marshall et al., 2018). Ecosystems not only make human life possible, they also contribute to 29 making life worth living (Costanza et al., 1997). Cultural losses or impacts associated with climate change 30 on human wellbeing can be attributed as a factor contributing to mental health issues, such as depression 31 (Fritze et al., 2008; Cunsolo Willox et al., 2015; Helm et al., 2018). However, recent review suggests that 32 cultural aspects of climate change are largely under-represented in the literature (Fish et al., 2016). This may 33 be because cultural impacts are largely 'invisible' (Turner et al., 2008) where losses associated with culture 34 are less tangible than those associated with economic losses. Given the observed impacts and high risks of 35 impacts from climate change on most coastal ecosystems where people are living and interacting closely 36 with (Section 5.3.2), it could be postulated that the consequential climate impacts on the cultural values that 37 marine ecosystems offer would elevate the risk of impacts on mental and psychological aspects of human-38 welling (low agreement, low evidence). 39

40 The relationship between people and nature is continually changing, with corresponding and constantly 41 evolving opportunities for cultural enrichment (low evidence, medium agreement). Climate change, however, 42 is *likely* to alter the rate of environmental change to the extent that cultural adaptation may be unable to keep 43 pace. This is because the culture that forms around a natural environment can be so integral to people's lives 44 that disassociation from that environment can induce a sense of disorientation and disempowerment (Fisher 45 and Brown, 2014). For example, when a resource user such as a fisher, farmer, or forester is suddenly faced 46 with the prospect that their resource-based occupation is no longer viable, they lose not only a means of 47 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). 48 49 Climate change can quickly alter the quality of, or access to, a natural resource through degradation or 50 coastal inundation, where livelihoods and lifestyles are no longer able to be supported by the resource. When 51 people are displaced from places that they value, there is strong evidence to suggest that their cultures are 52 diminished, and in many cases endangered. There are no effective substitutions for, or adequate 53 compensation for, lost sites of significance (Adger et al., 2012). 54

55 The vulnerability of human communities to cultural impacts depends on the nature of the cultural 56 relationship with a climate sensitive resource and the extent to which culture can absorb or cope with such 57 changes and adapt (*low confidence*). For example, Marshall et al. (2018) assessed the importance of identity, Chapter 5

1 pride, place, aesthetics, biodiversity, lifestyle, scientific value and wellbeing within the Great Barrier Reef 2 region by 8,300 people across multiple cultural groups. These groups included indigenous and non-3 indigenous local residents, Australians (non-local), international and domestic tourists, tourism operators, 4 and commercial fishers. They found that whilst some groups granted higher levels of importance to some 5 values, all groups highly rated all (listed) cultural values, suggesting that at least these cultural values are 6 critically associated with iconic ecosystems. Climate change impacts upon the Great Barrier Reef through 7 increased temperatures, cyclones and sea level rise that cumulatively degrade the quality of the Reef, are 8 liable to result in severe cultural impacts within these cultural groups. 9

10 Climate change affects differently the human cultures and well-being: it depends on the importance of the 11 ocean in their life. The adaptive capacity of people to moderate or influence cultural impacts, and thereby 12 reduce vulnerability to cultural impacts is also culturally determined (Cinner et al., 2018). The capacity to 13 consider climate risks, share learnings and reorganize, as well as the capacity to mitigate the impacts of 14 climate change reflect the cultural tendencies of groups of people. Contemporary responses to climate 15 change have been labelled as barriers to adaptation given that they fail to address the cultural dimensions of 16 climate risk, suggesting that considerable investment in the adaptive capacity of people to consider cultural 17 impacts is critical (Adger et al., 2012). Vulnerability to climate change, and the adherence to the Sustainable 18 Development Goals, will be correspondingly affected once cultural dimensions are carefully considered in 19 strategic developments. 20

5.4.2.2 Economic Dimension

23 5.4.2.2.1 Monetary and material wealth generated from fisheries

24 Marine biodiversity and ecosystem services support the income and livelihood of coastal communities and 25 marine-related economic sectors such as fisheries, aquaculture, forestry and tourism (World Ocean 26 Assessment) (Section 5.4.1). Marine fisheries and forestry are directly dependent on provisioning service 27 (biomass production of fish stocks and mangroves). Marine aquaculture is partly dependent on capture 28 fisheries production (as feed to supply animal protein and nutrients for the farmed animals) and regulatory 29 service through provision of suitable farming habitat (nutrients, oxygen, current) (Oyinlola et al., 2018). 30 Global gross revenues from marine fisheries is around US\$ 100 billion annually (Swartz et al., 2013). 31 Capture fisheries sectors provide full-time and part-time jobs for an estimated 260 ± 6 million people, 22 ± 6 32 0.45 million of whom are small-scale fishers (Teh et al., 2011). Specifically, small-scale fisheries are 33 important for the livelihood and viability of coastal communities worldwide (Chuenpagdee, 2011). The 34 oyster industry in the Pacific Northwest of the United States has already lost nearly US\$110 million in 35 annual revenue and some 3,200 jobs due to ocean acidification. AR5 concludes with low confidence that 36 climate change will lead to a global decrease in revenue with regional differences that are driven by spatial 37 variations of climate impacts on and the flexibility and capacities of food production systems. It also 38 highlights the high vulnerability of mollusk aquaculture to ocean acidification. This assessment examines the 39 rapidly growing literature on the economics of climate change impacts of marine fisheries from global to 40 regional scales.

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42 Since AR5, global projections on the economics of climate change impacts on marine fisheries have 43 incorporated and related to a broader range of social-economic factors. Driven by shifts in species 44 distributions and maximum catch potential of fish stocks (Section 5.4.1) as well as projected changes in the 45 prices of fisheries landings, marine fisheries revenues are projected to be negatively impacted in 89% of the 46 world's fishing countries under the RCP8.5 scenario in the 2050s relative to the current status, with projected 47 global decreases of 10.4% ±4.2% and 7.1% ±3.5% under RCP8.5 and RCP2.6, respectively (Lam et al., 48 2016). The projected changes in the potential catch in a country's Exclusive Economic Zones (EEZs) do not 49 directly translate to the change in the fisheries revenues of a country because the catch composition change 50 and vessels of some countries do not only fish in their own EEZs, but also in the high seas and other 51 countries' EEZs. 52

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 Chapter 5

1 review). Adaptation strategies to fisheries productivity changes and effective transboundary institutions can 2 offset these losses; for the RCP6.0 scenario, adaptation is projected to lead to substantial higher global 3 profits (154%), harvest (34%), and biomass (60%) in the future when compared to projections without 4 adaptation. However, the total profit, harvest and biomass have negative impacts even with full adaptation 5 scenario under the RCP8.5 scenario (Gaines et al., 2018). Climate change will adversely impact the 6 monetary wealth generated from fisheries and such impacts will be higher under higher carbon emission 7 scenarios (high confidence). Higher carbon emissions under 'business-as-usual' may also impose substantial 8 limits to adaptation (medium confidence).

9 10 The implications of the effects of climate changes on the fisheries sector vary among different regions and 11 countries (high confidence) (Hilmi et al., 2015a), with the most profound implications in the coastal low-12 income food deficit countries including small island countries, African countries and tropical Asian countries 13 (Lam et al., 2016). For example, using geo-spatially referenced malnutrition and infant mortality data as a 14 proxy for poverty, 15 developing countries are found to contain over 90 percent of the world's low-elevation 15 coastal zone rural poor (Barbier, 2015). These least developed countries usually rely heavily on fish and 16 fisheries as a major source of animal proteins, nutritional needs, and income and job opportunities, and these 17 countries have a limited availability of alternative livelihoods and sources of food. Therefore, negative 18 impacts on the catch and total fisheries revenues obtained by these countries may have greater implications 19 for jobs, economies, food and nutritional security than the impacts on high Human Development Index 20 (HDI) countries (Srinivasan et al., 2010; Golden et al., 2016). These least developed countries are highly 21 vulnerable to the impacts of climate change but have relatively low capacity to adapt (Allison et al., 2009; 22 Blasiak et al., 2017). Climate change may also worsen the non-climate-related socio-economic shocks and 23 stresses, and hence is an obstacle to poverty reduction (Hallegatte et al., 2015). Hence, climate change 24 adaptation and mitigation strategies must be human-centered, with an emphasis on the need for poverty 25 eradication, food security, empowerment of local stakeholders, and addressing power imbalances and 26 inequity in fishing and aquaculture communities. (Dasgupta et al., 2017); Kalikoski et al. (2018a) investigate 27 the possible impacts of climate change on coastal communities in Bangladesh using the salinity tolerance 28 range of a variety of fish species they consume, and find that areas with poor populations that will lose 29 species are about six times more prevalent than areas gaining species. 30

31 The vulnerability of coastal communities is increased because of their dependence on some specific 32 ecosystems, which are very sensitive to climate change. For example, pelagic and coastal fisheries provide 33 vital services to Pacific island communities. Oceanic fisheries in the region rely on four species of tuna: 34 skipjack (Katsuwomus pelamis), yellowfin (Thunnus albacares), albacore (Thunnus alalunga) and bigeye 35 (Thunnus obesus). In 2014, total regional offshore fishery production amounted to 1,823,561mt with license 36 fees from foreign fleets generating US\$349,335,572 across all 22 PICTs (Pacific island countries and 37 communities; Gillett (2009)). In Tokelau and Kiribati for example, in 2014, the sale of foreign fishing vessel 38 access and licensing fees generated 98% and 84% of government revenue, respectively (Bell et al., 2018a); 39 they represented 52% and 40% of government revenue in Tuvalu and FSM for that same year (Johnson and 40 Williams, 2017). While accounting for much less volume than oceanic fisheries, coastal fisheries have a high 41 unit value, provide important local livelihoods, and are critical to local food security and nutrition, 42 representing between 40–90% of dietary animal protein (Bell et al., 2009; Bell et al., 2011). Overall, 43 fisheries and aquaculture production in the region amounted to about 2 million mt in 2014, worth an 44 estimated USD3.2 billion (Gillett, 2009). Fishing contributions to GDP - which do not include post-harvest 45 activities - were re-estimated in a consistent manner by Gillett (2009) and found to account for between 7% 46 and up to 30% of GDP for Solomon Islands, FSM, Kiribati and Marshall Islands. This heavy economic 47 reliance on fisheries make Pacific island communities particularly sensitive to the effects of climate change. 48

49 Amongst different ecosystems, coral reefs are an important asset from an economic point of view. Climate 50 change impacts to coral reefs and other fish habitats, as well as to targeted fish and invertebrate species 51 themselves, are *likely* to reduce harvests from small-scale, coastal fisheries by up to 20% by 2050, and by up 52 to 50% by 2100, under the RCP8.5 emissions scenario (Bell et al., 2018a). Changes to the ocean and to coral 53 reefs are *likely* to reduce catches of reef fish by 20% by 2050. However, national governments lack the 54 capacity to respond effectively to major events, as current patterns and projected changes require policy 55 solutions and practices that are trans-sectoral and span levels of governance (Cashman and Nagdee, 2017). 56 The net benefits accrued by the global economy of coral reefs reach US\$30 billion yr⁻¹ from food security,

57 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).

5 6 Within the fisheries sector, there may be gender differences in the level of vulnerability and risk of climate 7 change impacts on people's income and livelihood. Women are heavily engaged in the post-harvesting 8 sectors, mainly involving selling and processing fish in the fish landing sites and local markets. Often it is 9 mostly women working in canning, processing and other ancillary economic activities related to marine 10 resources. Women are also involved in the bookkeeping operations and taking care of the home while the 11 fishermen are away. However, women in the post-harvest sector are often in a disadvantaged position 12 because of the marginalization of the fishing communities and their lack of access to credits and capacity 13 development. Thus, women may be disproportionately affected relative to men when climate change impacts 14 fisheries. Gender equity and empowering women's participation in climate change discourse and actions 15 have been included in the Gender Action Plan of the United Nations Framework Convention on Climate 16 Change (UNFCCC) (Kalikoski et al., 2018b).

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18 Overall, climate change impacts on abundance, distribution and potential catches of fish stocks (see Section 19 5.3.1) are expected to reduce revenues and economic values of the global fisheries (medium agreement 20 medium evidence). These impacts on fisheries sectors will increase the risk of impacts on the income and 21 livelihood of people working on these economic sectors (medium confidence). However, these impacts have 22 not been well addressed in recent literature (Lacoue-Labarthe et al., 2016). More effort will be needed to 23 understand the effects of climate change on the dynamics of the fish prices and costs of fishing. Both of 24 these changes would lead to the adjustment of the fishing effort and hence intensify or lessen the 25 overcapacity issue. Studies have attempted to project how fishers may respond to changes in fish distribution 26 and abundance by incorporating different management systems (Haynie and Pfeiffer, 2012; Galbraith et al., 27 2017). However, the impacts of climate change on the management effectiveness and trade practices is still 28 inadequately understood (Galbraith et al., 2017) and needs further exploration. 29

30 5.4.2.2.2 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.

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38 Climate change impacts on marine ecosystem services will lead to a high risk of impacts on the nutritional 39 health of dependent communities with high seafood-dependence, particularly for Indigenous and other 40 coastal communities in the global south, through decreases in seafood availability (Hilmi et al., 2014) (low 41 confidence; Table 5.6, Figure 5.18). Marine ecosystems, through fisheries and aquaculture, support the 42 global population with proteins and vital micronutrients, such as iron, zinc, omega-3 fatty acids and vitamins 43 (Golden et al., 2016). In particular, fish are important sources of micronutrients, especially for some 44 populations that are largely dependent on fisheries for subsistence and livelihood. Deficiencies of some of 45 these micronutrients can increase the risk of perinatal and maternal mortality, contribute to growth 46 retardation, child mortality and cognitive impairments (Black et al., 2013). Climate change is projected to 47 reduce fisheries catch potential by the 2050s relative to the 2000s, particularly for regions where dependence 48 on micronutrients from fish is high (medium confidence, Section 5.3.1). The projected decrease in fish 49 supply is expected to exacerbate health risks over the next decades from a reduction in the intake of 50 micronutrients and fatty acids. In particular, low- and middle- income coastal communities around the 51 equator where large decrease in potential fisheries catches are projected (>20% by the 2050s relative to the 52 2000s under RCP8.5) are at a higher risk of such impacts on nutritional health (Golden et al., 2016) (medium 53 confidence; Figure 5.18). It has been estimated that 1.39 billion people are vulnerable to deficiencies of 54 micronutrients if fish catches continue to decline (Golden et al., 2016). 55

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

1 climate change impacts on fisheries production coincide with areas where land-based food production is 2 projected to decrease due to climate change (Blanchard et al., 2017), further exacerbating the risk of climate 3 change impacts on nutritional health of people living in these areas. Hilmi et al. (2017) conclude that climate 4 change and ocean acidification will impact fisheries and aquaculture in the Mediterranean countries, but the 5 impacts are difficult to evaluate because of uncertainties in the projected physical and ecological changes 6 (Hilmi et al., 2013a). The distribution of tuna in the tropical Pacific Ocean will shift and abundance decline 7 under climate change, because of projected increases in sea surface temperature, reduced oxygen, changes in 8 the strength of the region's major currents, and declines in primary production (Boyce et al., 2010; Lehodey 9 et al., 2011). For yellowfin and skipjack tuna, countries in the eastern Pacific are *likely* to gain from such 10 projected shifts, particularly if prices increase due to declines in catches elsewhere (Johnson and Williams, 11 2017). However, such shifts in tuna distribution would see countries in the west suffer from declining stock 12 biomass (Bell et al., 2018b). Based on skipjack only, preliminary model simulations show changes in 13 percentage contribution to PICTs' GDP, to range between -0.4% and +6% by 2050, and from -2% to +2% by 14 2100 (Bell et al., 2013). The same study estimated changes in average percent contribution to government 15 revenue to vary up to between -0.3% and +21% by 2050, and from -2% to +12% by 2100. The abundance of 16 bigeye and south Pacific albacore is *likely* to decline overall (Lehodey et al., 2015), exceeding 60% in 17 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.

20 21 22 23 24 25 26 27 Other non-climatic factors may interact with climate change effects on food resources, resulting in complex 28 patterns of risk for vulnerable coastal communities to climate impacts on food security. For example, the 29 shift from traditional nutritious wild caught seafood-based diets of coastal Indigenous communities, towards 30 increased consumption of processed (and other) energy-dense foods that are high in fat, refined sugar, and 31 sodium, due to social and economic changes (Kuhnlein and Receveur, 1996; Shannon, 2002; Charlton et al., 32 2016; Batal et al., 2017), may reduce communities' exposure (Quinn et al., 2012) to climate change effects 33 on fish stocks. Such changes, however, have important consequences on diet quality and nutritional status 34 (Thaman, 1982; Luick et al., 2014), and have led to an increased prevalence of obesity, diabetes, and other

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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.

4 5.4.2.2.3 Monetary and material wealth generated from coastal and marine tourism sector

5 Tourism is one of the largest sectors in the global economy, with coastal tourism comprising a significant 6 part of global revenue derived from the tourism sector. The development of the tourism sector has permitted 7 economic gains for coastal countries and islands. For example, about 30% of world tourism is concentrated 8 in the Mediterranean Basin (World Tourism Organisation data). The wealth in natural resources of the 9 Mediterranean coastline include unique landscapes, climate favourable for tourism, important biodiversity 10 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, 11 12 wind speed, etc.), underwater parameters (SST, invasive species such as jellyfishes, etc.), and coastal 13 hazards (erosion, marine flooding, etc.) (Safa and Hilmi, 2012). Such changes will affect the main 14 components of the tourism industry-such as tourists' preferences, tourism operators' strategies, and the 15 transportation market—and thus the tourism supply and demand balance at the global scale. The impacts will 16 vary between ecosystems (high confidence). 17

18 Climate change impacts on coral reefs and other coastal ecosystems (Section 5.2.1) may affect the quality of 19 the dive sites, beaches and the coast (Hilmi et al., 2012) (high confidence). Coral reefs and coastal 20 ecosystems draw significant tourism each year, supporting recreational activities, such as diving and 21 snorkeling, and supporting coastal communities and auxiliary sectors (Spalding et al., 2017). Reef tourism is 22 increasing in popularity, with some parts of the world showing growth in the number of visitor in recent 23 years, increasing the pressure to develop tourist facilities and accommodation. Beach recreation values are 24 also expected to decline in response to narrower or vanished beaches resulting from sea level rise and 25 erosion, with considerable losses for local economies (e.g., \$1 billion loss between 2006 and 2080 in North 26 Carolina (Whitehead et al., 2009; Nelson et al., 2013). As principal assets of coastal tourism, the loss of 27 beaches could significantly impact local economies in subtropical and topical regions, and could incur 28 additional costs associated with coastal protection and infrastructure repairs (DiSegni and Shechter, 2013). 29 The impacts are exacerbated in SIDS because of the increasing demography which carries an increasing 30 pressure on the reef resources mainly used for self-consumption. This trend is exacerbated by the 31 disappearance of traditional ways of management of the reef resources (such as taboos and religious beliefs 32 translating into informal temporary no-take zones) which were quite effective (Caillaud et al., 2004). 33

34 Climate change will cause global economic losses in tourism sector (medium agreement, medium evidence). 35 The climate change projections show that coral reef coverage declines between 6.65% (under the RCP2.6 36 scenario) and as much as 27.58% by 2095 under RCP8.5, in which case a meta-analysis by Chen et al. 37 (2015) estimates a coral reef recreational and tourism value loss ranging from US\$1.88 to US\$12.02 billion 38 annually. These economic damage estimates only account for recreational and tourism loss, although there 39 would be other losses in the services provided by coral reefs including coastal protection, fish habitats, and 40 marine biodiversity. Cesar et al. (2003) estimate the total costs of coral bleaching over a 50-year time 41 horizon with a 3% discount rate at over US\$ 84 billion in net present value. For moderate bleaching, this 42 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 43 44 distribution shows that the largest costs are in Australia (US\$ 28.4 billion) and Southeast Asia (US\$ 38.3 45 billion). A recent study by The Australia Institute (2016) showed that if coral bleaching persists, tourism 46 areas adjacent to the Great Barrier Reef risk declines in visitors from 2.8 million visitors to around 1.7 47 million per year. This the equivalent of more than \$1 billion in tourism expenditures, supporting around 48 10,000 tourism jobs in regional Queensland (The Australia Institute, 2016) (Hughes et al., 2017). 49

50 Tourism is an important sector in the Pacific region (www.wttc.org/). For many PICTs, given limited 51 options, tourism is one of the main industries that provides opportunities for social and economic 52 development (Jiang and DeLacy, 2014). With a projected additional 1 million visitors to the region by 2040, 53 tourism could further accelerate these opportunities for growth and employment across the region by 54 generating additional spending of more than US\$1.6 billion, and creating an additional 110,000 jobs (World 55 Bank, 2017). However, tourism destinations in the Pacific region, which tend to focus on nature-based and 56 predominantly marine activities, have also been identified as being most at risk of climate change (Klint et 57 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)

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6 The main economic activity in the Small Island Developing countries of the Caribbean region is tourism. In 7 addition, fisheries provide important livelihoods as well as food security and nutrition opportunities for local 8 communities, especially of low income. Extreme weather events, sea level rise and increasing sea surface 9 temperatures are three of the main marine related climate change impacts that appear to be of particular 10 concern to Caribbean islands. A USAID Rapid climate change vulnerability assessment states that 'The total 11 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 12 rebuild or relocate infrastructure due to the direct damage as well as lost land-value.' Primarily this comes 13 from losses in annual tourism revenues, as well as capital costs related to damage to tourism resorts. 14 Caribbean tourism is particularly vulnerable to climate change due to its concentration in low-lying coastal 15 areas. According to the AR5 report, the Caribbean region will be highly impacted by climate change effects 16 on the tourism sector. The report highlighted the extent of economic losses to coastal tourism in the 17 Caribbean due to hurricanes. For example, in the Caribbean community countries, rebuilding costs of tourist 18 resorts are estimated at US\$10 to US\$ 23.3 billion annually in 2050. A hypothetical 1-m sea level rise would 19 result in the loss or damage of 21 airports, inundation of the land surrounding 35 ports, and at least 149 20 multi-million dollar tourism resorts damaged or lost from erosion to the coastal beach areas (Simpson, 21 2010). The model developed by Cashman et al. (2012) suggests that, given the proximity of most hotels to 22 the beach, a 0.2 meter rise in sea levels could *likely* impact on about 12 hotels or about 7 percent of the total 23 number of hotels in Barbados. These hotels account for 122 rooms or about 3 percent of the room stock. 24 Assuming about a 56 percent occupancy rate, this translates into an annual reduction in revenue of about 25 US\$3.5 million. The long-term projected rise in sea levels, about 0.5 and 1 meter, is *likely* to have more 26 severe effects: both scenarios are *likely* to impact over 40 percent of hotels and lead to a reduction in revenue 27 of around US\$100 million, or about 3 percent of GDP, over the long-run (Cashman et al., 2012). 28

29 North Atlantic hurricanes and tropical storms have increased in intensity over the last 30 years, with climate 30 projections indicating the trend is for a continued increase in hurricane intensity (Chapter 6). An analysis of 31 climate change's impact on Jamaica's coastal transportation infrastructure (two airports, a port and a 32 container terminal) estimated current cumulative losses due to damage associated with natural disasters to be 33 around USD120 billion, roughly 7% of GDP, and rising to as high as 56% of GDP by 2025 if climate 34 projections are accurate (UNCTAD, 2017). In 2015, Tropical storm Erika devastated Dominica causing 35 USD483million in total damages and losses (mostly related to transportation, housing and agriculture 36 damages), equivalent to 90 percent of Dominica's GDP (World Bank, 2017). 2017 saw three major storms, 37 Harvey, Irma and Maria, destroy parts of the region. Total damages and losses caused by the latter two 38 storms to the tourism industry of Dominica, the British Virgin Islands, and Antigua and Barbuda were 39 estimated at USD2.2 billion, with recovery costs in the environment evaluated at USD6.8 million (UNDP, 40 2017). Total recovery needs for Dominica, Barbuda, and the British Virgin Islands alone were estimated at 41 over USD5 billion (UNDP, 2017). Based on information collected on damages to fishing operations and 42 fishing-related businesses for Florida, Puerto Rico and the US Virgin Islands, NOAA estimated total 43 damages of \$117 million and revenue losses of \$106 million.

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45 Recent studies provide additional evidence that species range shifts (Sections 5.2.3, 5.3.3) are *likely* to alter 46 tourism flows. For instance, poleward range shifts of marine fishes in the United States could yield new 47 opportunities for recreational fishing in temperate regions, while warmer conditions could benefit some 48 species purposively introduced for recreational fishing (DiSegni and Shechter, 2013). Conversely, increases 49 in the prevalence of tropical box jellyfish through climate induced poleward range shifts could detrimentally 50 affect coastal tourism and water-related activities in subtropical and temperate regions (Klein et al., 2014). 51 Increased prevalence and transmission of diseases are also *likely* to occur under warmer ocean temperatures. 52 with empirical evidence of poleward shifts in disease outbreaks under warmer conditions (Burge et al., 53 2014). Importantly, adverse publicity accompanying an increased risk of disease transmission in tropical and 54 subtropical regions could further have deleterious impacts on coastal tourism (DiSegni and Shechter, 2013). 55 If climate change and ocean acidification reduce the seafood supply (Wabnitz et al., 2017), the attractiveness 56 for tourists will also decrease (low confidence). This will have an economic impact on jobs, revenue and

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10 11 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

12 The aesthetic appreciation of natural places is one of the most fundamental ways in which people relate to 13 their environment. However, climate change threatens these aspects through degrading aesthetic elements of 14 climate sensitive natural resource (medium agreement, low confidence). Positive externalities generated from 15 ecosystems, or cultural ecosystem services, include a range of values such as appreciation of biodiversity 16 and scientific opportunities, as well as aesthetic stimulus or aesthetic appreciation. Aesthetically beautiful 17 places are critically important for supporting local and international economies (especially through tourism), 18 human wellbeing, and stewardship (Marshall et al., 2018). Marshall et al. (2018) assessed the importance of 19 aesthetic values within the Great Barrier Reef (GBR) by indigenous and non-indigenous local residents, 20 Australians (non-local), international and domestic tourists, tourism operators, and commercial fishers and 21 found that aesthetic values are a critically important cultural value for all cultural groups, and are important 22 for maintaining sense of place, pride, identity and opportunities for inspiration, spirituality, recreation and 23 wellbeing. However, climate change has and will destroy many of the features of regional aesthetical 24 ecosystem services and possible creating new aesthetical ecosystem services.

25 26 Along with negative changes in aesthetically pleasing ecosystems, climate change has and will affect real 27 estate values both in coastal areas as well as inland through direct physical impacts. Flooding along coasts, 28 drought and fires in inland areas have large economic impacts. Putra et al. (2015) study the vulnerable 29 coastal areas of the New Jersey real estate market by categorizing types of real estate parcels with flooding 30 risk. They simulate their results and calibrate the model with actual data of real estate parcels sold and 31 examine different insurance policies offered to the coastal flood zones. They found that property owners that 32 were well informed about climate change would invest heavily in defensives in the near term but then 33 abandon coastal real estate at some later time when the costs become too risky. McNamara and Keeler 34 (2013) study barrier island communities in the Atlantic Coast of US, where rising sea levels and coastal 35 erosion can damage real estate values. They find through their models that informed property owners invest 36 in defensive expenditures in the short run (medium confidence). Moreover, most urban area and mega-cities 37 are found along the coast. The combined effects of urbanization and climate change on near-surface 38 temperatures predict urban areas to be warmer and less comfortable than surrounding unaltered areas in the 39 future (Argüeso et al., 2014). Fischer et al. (2012) predict potential increases in morbidity and mortality with 40 increased temperatures and high humidity in urban areas. People may prefer to pay to compensate the 41 climate change impacts. Costello et al. (2010) study risk aversion and probability distribution of future 42 temperature changes by placing an upper bound in the fat upper tail of the distribution to find that 43 willingness to pay to avoid climate change is not sensitive to bound value.

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45 One cannot avoid the issue that climate change will alter or decimate coastal and inland real estate and land 46 values across many regions in a variety of ways and degrees (medium agreement, medium evidence). As 47 regional information accumulates on the occurrence and frequency of extreme events such as flood risks, 48 drought duration and intensity, fires, etc., data will be incorporated into risk assessments and asset pricing. 49 Decreases in the real estate values, and lack of rental incomes from jeopardized properties are going to 50 trigger a chain of events which will be unavoidable (McNamara and Keeler, 2013). Loss in household asset 51 values are crucial since homes or farmland tend to be the largest form of savings for most households in 52 many regions. Spending patterns and amounts will be affected where lower long-term spending will lead to 53 decreased economic business activity and may lead to closure of many small business firms, further leading 54 into increased local unemployment (Rubin and Hilton, 1996). If house or land values decrease below 55 outstanding mortgages, defaults on debt payments may lead to regional bank crisis. In 2050, in the Tampa 56 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).
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4 Climate change is expected to impact coastal real estate values and incur substantial costs on coastal 5 infrastructure due to increased frequency and intensity of floods and other disasters driven by extreme 6 weather events, particularly in tropical SIDS. For example, much of the estimated damages, losses and 7 associated costs incurred by Pacific SIDS from climate change will stem from sea level rise, an increase in 8 temperature as well as the severity of cyclones, floods, and drought. Natural disasters already cost PICTs an 9 average of between 0.5- 6.6 % of GDP per year (World Bank, 2017). Cyclone intensity may increase in the 10 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 11 12 respectively, can be seen as indications of what the future may hold. For Fiji, average asset losses due to 13 tropical cyclones and floods were estimated at more than USD226 million per year, representing more than 5 14 percent of Fiji's GDP (Esler, 2016). The same study found that the fraction of GDP lost every year due to 15 tropical cyclones and floods could increase by up to 50% by 2050 (i.e., 6.5 percent of GDP), and that almost 16 100 percent of GDP in investments are required over the next 10 years to strengthen Fiji's resilience to 17 future climate change impacts and natural hazards. The total economic value of damages and losses from 18 tropical cyclone Winston in Fiji was estimated at US\$ 0.9 billion (Esler, 2016), equivalent to 20% of the 19 country's GDP. If the value of destroyed environmental assets and losses in environmental services are 20 accounted for, the total impact of cyclone Winston amounts to US\$ 1.3 billion - with US\$ 0.7 billion in 21 damages and USD0.6 billion in losses. Projected production losses to the fishery sector were estimated at 22 about USD14 million annually until 2021; and total costs to rebuild, recover and build resilience at USD0.9 23 billion (Esler, 2016). The total economic value of damages and losses sustained by Vanuatu during tropical 24 cyclone Pam were estimated at US\$ 449.4 million, equivalent to 64.1% of GDP. Losses to the agricultural 25 and tourism sectors made up 33% and 26%, respectively, of total losses (Esler, 2015). Ecosystem service 26 losses while acknowledged to be substantial were not accounted for within the impacts to GDP. Recovery 27 and reconstruction needs were estimated at US\$ 316 million. 28

29 In developed countries' context, Ackerman and Stanton (2007) forecast that annual real estate losses due to 30 climate change will be from 0.17% of USA GDP in 2025 to 0.36% in 2100, with Atlantic and Gulf Coast 31 states being the most vulnerable. Bunten and Kahn (2014) build a model for regions in which capital and 32 labor are mobile and in which there are variations in the emerging climate change risks between different 33 areas. They state that if the population has homogenous preferences, real estate prices will drop when 34 negative news about climate change is revealed, but not if the population is heterogeneous with respect to 35 incomes, networks, tastes and the ability to pay for risks. Hallegatte et al. (2007) simulate several scenarios 36 for European cities and estimate climate change adaptation costs and investments for most of the 21st 37 century. They state that 'As a consequence (of uncertainty), no anticipated adaptation is possible and the 38 reactive adaptation leads to sub-optimal solutions, costly over the long-term, or to huge short-term costs of 39 adaptation. The static and enumerative approach to monetary damage valuation tends to underestimate their 40 actual costs for future generations because it neglects important mechanisms such as a crowding-out of 41 productive investment or a depreciation of real estate assets which can dampen economic growth.' (p. 14). 42 Masson et al. (2014) discuss how joint efforts by all planners in the urban areas are needed to plan adaptation 43 strategies to climate change. Time will show whether the potentially catastrophic loss in real estate and 44 consumer wealth will be addressed through more insurance or through costlier defensive measures, which all 45 depend on the asset holders and cities willingness to pay. How to discount for potential losses that may occur 46 in the future (with an unknown occurrence) are areas of research that need to be addressed on a regional 47 basis to forecast real estate loss estimates (low confidence).

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49 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.

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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

1 variable than resources such as wind and solar, and therefore offers the potential benefit of providing more 2 continuous supply within the energy network (Foteinis and Tsoutsos, 2017; Hemer et al., 2017). Tidal 3 energy is the most advanced technology with notable commercial installations of tidal barrage technology in 4 France (La Rance River, 240 MW), Canada (Bay of Fundy, 20 MW) (O Rourke et al., 2010), and most 5 recently South Korea (Sihwa Lake, 254 MW) (Kempener and Neumann, 2014a). Tidal stream technology 6 has seen significant advances in the development and testing of turbines, but has not yet advanced to the 7 stage of large-scale arrays. The complex nature of ocean waves has given rise to a great diversity in the 8 device designs, most of which can be broadly categorised as oscillating water columns, oscillating body 9 converters or overtopping converters (Kempener and Neumann, 2014a; Kempener and Neumann, 2014b; 10 Manasseh 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 11 12 generally at lower levels of technical development (Kempener and Neumann, 2014a; Kempener and 13 Neumann, 2014b; Borthwick, 2016).

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15 Direct effects of climate change which are relevant for maritime transport include ocean warming and 16 related melting of sea ice, sea level rise and floodings in coastal areas. Sea level rise, storms and associated 17 floodings will cause direct costs due to infrastructure damages in ports and other coastal facilities and costs 18 for measures for flood-defenses (Becker et al., 2018; Ng et al., 2018). The damage caused by a changing 19 climate may present more of a threat for developing than for developed economies. This needs to be 20 considered for adaptation measures and investments 'to build resilient infrastructure' (SDG 9) for maritime 21 industry of developing countries and Small Island Developing States whose economies depend on ports 22 (Koetse and Rietveld, 2009). Additionally, short-notice port closures, delays, detours and trip cancellations 23 could create indirect costs for maritime transportation (Koetse and Rietveld, 2009; Jaroszweski et al., 2010).

24 25 Global warming and changing weather patterns may have a substantial impact on global trade and transport 26 pathways (Koetse and Rietveld, 2009). The reduction in sea ice in the Arctic Ocean during summer opens up 27 the possibility for sea transport on the Northwest or Northeast Passage during at least several months per 28 year (Ng et al., 2018). Both routes may provide opportunities for more efficient transport between North 29 America, Europe, Russia and China for fleets with established Arctic equipment and may open up access to 30 known natural resources which were so far covered by ice (Guy and Lasserre, 2016). However, whether the 31 Arctic shipping routes may or may not be a realistic alternative will depend on political regulations, 32 economic aspects such as infrastructure along and reliability of the routes but also societal trends, 33 demographics, and tourism demand (Prowse et al., 2009; Wassmann et al., 2010; Pelletier and Guy, 2012; 34 George, 2013; Hodgson et al., 2016; Pizzolato et al., 2016; Dawson, 2017). Increased traffic in either the 35 Arctic or Antarctic would contribute to additional pollutants in the region, having a direct impact on the 36 climate and exacerbating localized warming (Wan et al., 2016).

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38 The shipping industry is also facing emission controls of greenhouse gases and pollutants. The combustion 39 of low-grade, high-sulphur content fuel in ships emits significant amounts of other greenhouse gases and 40 pollutants, such as sulphur oxides, nitrogen oxides, aerosols (containing organic carbon, black carbon, 41 polycyclic hydrocarbons (PAH) and heavy metals) to the atmosphere (Eyring et al., 2005; Corbett et al., 42 2007). These have substantial consequences for the atmosphere, the marine environment and human health 43 (Corbett et al., 2009). Some compounds are deposited relatively close to the source vessel and become 44 dissolved or suspended in the surface ocean. Several measures were implemented by the IMO through the 45 MARPOL Convention and other treaties to 'conserve and sustainably use the marine ecosystems and 46 resources' (SDG 14). This includes well-known ship-related measures to prevent atmospheric and seawater 47 pollution from ships (by oil, chemicals, harmful substances, sewage, garbage) and new technological 48 advances to reduce underwater noise, avoid transfer of harmful invasive species by ballast water, and recycle 49 ships in an environmentally sound way. Recently implemented, the IMO 'Global sulphur cap' limits the 50 release of sulphur oxides due to fuel combustion. Strict CO2 emission controls in maritime transport are 51 expected in future to be comparable to recent regulatory developments in aviation or land-based transport. In 52 order to reduce greenhouse gas emissions by ships, two mandatory energy efficiency measures, known as the 53 Energy Efficiency Design Index (EEDI) for new ships and the Ship Energy Efficiency Management Plan 54 (SEEMP) for existing ships, were introduced by the UN International Maritime Organization (IMO) in 2011 55 (Resolution MEPC 203(62).). A CO2 reduction strategy has been developed and adopted at the IMO in April 56 2018 (RESOLUTION MEPC.304(72), 2018). It aims to reducing total annual greenhouse gas emissions 57 from international shipping by 50% by 2050 compared to reference year 2008. In the meanwhile, rising oil

1 prices increase shipping costs significantly and drive economical decisions. Until technological measures are 2 in place, slow steaming may be a preferred option by ship operators in order to save costs but also to reduce 3 CO2 emissions (Bouman et al., 2017). During recent years, there have been large advances in technologies 4 and operational practices with high potential to cut CO_2 emissions and fuel consumption by as much as 84%. 5 Those measures include the use of alternative fuels, such as LNG and biofuels, developing better power and 6 propulsion systems, more efficient ship hull design, improved equipment, and advanced weather routing 7 (Bouman et al., 2017). However, no single measure is itself sufficient to achieve significant emission 8 reductions in a cost effective manner within the next decades (Corbett et al., 2007). In order to minimize ship 9 pollution of this pristine environment, special measures were implemented under the IMO Polar Code (IMO 10 1 January 2017) for ships operating in both Antarctic and Arctic waters. The Arctic Council recommended, 11 that in the future, IMO should extend the Polar Code to include a ban on heavy fuel oil (HFO) (Wan et al., 12 2016). 13

14 Overall, a growing number of shipping companies are intending to drastically reduce CO2 emissions from 15 ships over the next decade hence contributing to 'combat climate change and its impacts' (SDG 13) and 16 'ensure sustainable consumption and production' (SDG 12) (medium agreement, limited evidence). In 17 addition, the shipping industry aims to reduce shipping-related pollution in the wider oceans but also in ports 18 and coastal regions where ship-derived emissions may contribute to tens of thousands of cases of premature 19 mortality every year (Bonnet et al., 2005; Sofiev et al., 2018). The use of new emission reduction 20 technologies, such as scrubbers, to comply with a recent environmental regulation (in this case the IMO 21 'Global sulfur cap') will benefit the environment by significantly reducing the release of pollutants to the 22 atmosphere. However, there is incomplete understanding of the impact of, e.g., scrubber wash water 23 discharge on marine chemistry, biodiversity and biogeochemical processes (Stips et al., 2016; Endres et al., 24 2018). In the global shipping network, new strategies were established such as the management of ship 25 routing and establishment of Special Areas and Particularly Sensitive Sea Areas to avoid collisions between 26 ships and marine mammals and to protect vulnerable marine ecosystems. Despite existing guidelines for 27 levels of monitoring and compliance of ship pollution, there is still the risk for acidification, eutrophication 28 and accumulation of pollutants in the marine environment (CGDD and CETE, 2012).

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5.4.2.4 Impacts of Changing Ocean on Sustainable Development Goals

32 Climate change impacts will have consequences for the ability of human society to achieve sustainable 33 development. The SR 1.5 °C concludes that 'Limiting global warming to 1.5 °C rather than 2 °C would make 34 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 35 36 specifically on how climate change impacts on marine ecosystems would challenge sustainable 37 development. Specificially, this assessment uses the United Nations Sustainable Development Goas (SDGs) 38 as a framework to discuss the linkages between marine ecosystems and sustainable development, and how 39 climate change would affect such linkages

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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).

47 Impacts to other SDGs were considered more indirect in nature, where impacted ecosystems are mediated 48 through social and economic factors first before affecting an SDG targets (Singh et al., in review). For 49 example, climate impacts on marine ecosystem services related to primary industries that provide food, 50 income and livelihood to people have direct implications for a range of SDGs. These SDGs include 'no 51 poverty' (SDG 1), 'zero hunger' (SDG 2), 'decent work and economic growth' (SDG 8), 'reduced 52 inequalities' (SDG 10) and 'Responsible consumption and production' (Singh et al., in review) (Figure 53 5.19). These impacts relate to changing ocean under climate change that affect the pathways to build 54 sustainable economies and eliminate poverty (Sections 5.4.2.2.1, 5.4.2.2.3), eliminate hunger and achieve 55 food security (Section 5.4.2.1.2), reduce inequalities (Sections 5.4.2.1.3, 5.4.2.2) and achieve responsible 56 consumption and production (Sections 5.4.2.2.1, 5.4.2.1.5) (Carvalho et al., 2017; Castells-Quintana et al., 57 2017). Climate change is also creating living conditions in coastal areas that are less suitable to human

1 settlement and changing distributions of marine disease vector (5.3.2.1.1), reducing our chances of achieving 2 the goal for good health and wellbeing (SDG 3) (Pearse, 2017; Wouters et al., 2017). Also, women are often 3 engaged in jobs and livelihood sources that are more exposed to climate change impacts from the ocean such 4 as impacts on fisheries (Section 5.4.2.2.1) and impacts of sea level rise on coastal regions (Chapter 6). For 5 example, in Senegal, women disproportionately cultivate rice crops (Linares, 2009), and in coastal areas they 6 may be more effected by rising sea levels and resulting salinization (Dennis et al., 1995). Flooding in 7 Bangladesh has increased the vulnerability of women to harassment and abuse as the flooding upends 8 normal life and increases crime rates (Azad et al., 2013). As such, climate change may negatively affect our 9 ability to achieve 'gender equality' (SDG 5) (Salehyan, 2008). Impacts to living conditions as well as 10 changing recreational, aesthetic, and spiritual experiences also affects our ability to achieve 'sustainable 11 cities and communities' (SDG 11). 12

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 benefits all other SDGs (Singh et al., 2017). Achieving these targets is sometimes necessary to achieve other SDGs.

19 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)

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

23 SDGs indirectly (*low confidence*).

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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).

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5.5 Risk-reduction Responses and their Governance

5.5.1 Ocean-based Mitigation

7 IPCC AR5 and UNFCC definitions of climate mitigation cover both the reduction of greenhouse gas 8 emissions and the active removal of greenhouse gases from the atmosphere, and that usage is followed here 9 (SROCC Glossary). Article 4.1 of the Paris Agreement recognizes the need for both actions in its 10 requirement for a 'balance between anthropogenic emissions by sources and removals by sinks of 11 greenhouse gases in the second half of this century'. That balance (net zero emissions) is how dangerous 12 climate change can be avoided by 'holding the increase in the global average temperature to well below 13 2.0°C' Article 2.1 (Rogelj et al., 2015; Geden, 2016). To reduce the scale of the physical changes to the 14 ocean and their environmental impacts (Sections 5.2, 5.3), transformative and near-total emission reductions 15 are therefore needed in all sectors of human activity, not just energy, and for all greenhouse gases (Myhre et 16 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).

25 Instead the focus here is on the effectiveness, risk trade-offs and societal feasibility of two other ocean-based 26 mitigation approaches. First, the scope to decrease anthropogenically-driven releases of greenhouse gases 27 from the marine environment, primarily by maintaining the integrity of natural carbon stores in the coastal 28 zone. Second, the potential to promote additional carbon uptake by marine ecosystems, both in the coastal 29 zone and the open ocean, as a contribution to the additional gigaton-scale uptake (negative emissions) of 30 CO₂ that is needed in most emission pathways that limit global warming to 1.5°C-2.0°C (Fuss et al., 2014; 31 Anderson and Peters, 2016). Any ocean-based removals of CO₂ from the atmosphere could only be as an 32 addition to, not as a substitute for, very rapid reductions in greenhouse gas emissions (UNEP, 2017a). Under 33 most international definitions, including those of IPCC, such negative emissions (CO₂ removal) are 34 considered to be a form of climate geoengineering, as well as part of climate mitigation (Williamson and 35 Bodle, 2016). 36

37 The term 'blue carbon' was originally defined in UN reports to cover all carbon captured by marine 38 ecosystems, both coastal and oceanic (Nellemann and Corcoran, 2009). Subsequent usage has focused on 39 coastal carbon-rich habitats structured by rooted plants, such as mangroves, seagrasses and saltmarshes, 40 since those components are most amenable to management that might increase the security of carbon storage 41 (reducing emissions) and also potentially enhancing sinks (Kuwae and Hori; McLeod et al., 2011; Pendleton 42 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 43 44 et al., 2017). However, there are major uncertainties regarding these estimates, as discussed below. 45

46 Carbon uptake in the open ocean is driven by three processes: the biological pump, involving particulate 47 organic carbon export, the microbial carbon pump, based on microbial transformation to refractory 48 dissolved organic carbon (RDOC) (Jiao et al., 2010) and the physical pump (Ciais et al., 2013). Estimates of 49 the global carbon 'export' flux from the euphotic zone due to the biological pump range from 4 Gt C yr⁻¹ to 50 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 51 52 relatively stable climatic conditions of most of the Holocene, such inputs have been balanced by losses from 53 CO₂ outgassing, maintaining the long residence times (hundreds to thousands of years) of dissolved organic 54 and inorganic carbon in the ocean (Jiao et al., 2010; Jiao et al., 2014a; Legendre et al., 2015; Jiao et al., 55 2018a).

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Figure 5.20: Marine legal zoning based on United Nations Convention on Law of the Sea (UNCLOS) relevant to potential ocean-based mitigation of CO_2 (excluding renewable energy). Mitigation measures in pale green font are those involving increased primary production to enhance CO_2 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; costeffectiveness; 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

- 30 ecosystem-based adaptation (reducing impacts); two were considered at both scales. Ocean-based albedo
- 31 enhancement techniques are excluded here, being outside the scope of the current report. However,
- 32 physically-based marine renewables are included for comparison with the other approaches across all of the 33 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 ($< 100 \text{ km}^2$) 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,

8 through estuaries, shelf seas and a wide range of intertidal and shallow-water habitats. These systems play a 9 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.

5 6 The relatively recent appreciation of the scale of carbon storage (and its apparent vulnerability) in coastal 7 carbon ecosystems has focused attention on the opportunities and challenges for targeted 'blue carbon' 8 policies that are focused on intertidal and near-shore vegetation (Crooks et al., 2011; Herr and Landis, 2016; 9 Macreadie et al., 2017). For climate mitigation purposes, the need is for cost-effective management actions 10 that significantly either reduce greenhouse gas sources or enhance sinks, implemented at the local, national 11 and regional scale within an international framework. Such actions should be amenable to national carbon 12 accounting within emission inventories, with associated requirements for quantitative verification of 13 longterm carbon removal, together with new regulations and financial incentives, similar to those developed 14 for managing terrestrial carbon sinks (Crooks et al., 2011; Hejnowicz et al., 2015). Proposals to extend the 15 blue carbon concept to other components of the coastal ocean, e.g., shelf sea sediments and the water column 16 (Zhang et al., 2017) are being developed.

18 5.5.1.1.2 Coastal blue carbon

19 The global importance of three kinds of coastal vegetation has recently been recognised: mangroves, salt 20 marshes and seagrasses (Chmura et al., 2003; Duarte et al., 2005; Kennedy et al., 2010; McLeod et al., 21 2011). Although covering only $\sim 0.1\%$ of the Earth's surface, it has been estimated that these three systems 22 together support 1-10% of global marine primary production, and may make an even larger contribution to 23 total oceanic CO₂ uptake (Duarte et al., 2017). Some 151 countries around the world contain at least one blue 24 carbon ecosystem (seagrass, saltmarshes or mangroves) and 71 countries contain all three (Herr and Landis, 25 2016). Seaweeds (macroalgae) are considered separately in Section 5.5.1.1.3 below, since their longterm 26 carbon storage does not occur in situ, and is inherently more uncertain.

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28 The order-of-magnitude range of values for the role of vegetated coastal ecosystems in carbon processing is 29 primarily due to uncertainties in quantifying two key factors: the areal extent of these habitats, and their rates 30 of carbon burial. Differences in estimates for areal coverage depend on whether values are determined 31 empirically, or on theoretical grounds of environmental suitability (substrate, light availability and 32 temperature). Observation-based mapping provides global area estimates of $\sim 140,000$ km² for mangroves (Giri et al., 2011), ~55,000 km² for saltmarshes (McOwen et al., 2017a) and ~177,000 km² for seagrasses 33 34 (Green et al., 2003); however, theoretically-based estimates can be much higher, e.g., 500,000 km² for 35 seagrasses (Green et al., 2003). 36

Uncertainties in rates of carbon uptake and longterm 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.

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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).

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53 There is also *high confidence* that coastal wetlands can significantly assist with climate adaptation. For 54 example, by absorbing incoming wave energy, providing hurricane and storm surge protection, and 55 preventing erosion (Costanza et al., 2008; Spalding et al., 2014). Coastal ecosystems may keep pace with sea 56 level rise and, in some instances, are more cost-effective than hard civil infrastructure like seawalls 57 (Temmerman et al., 2013). Healthy coastal ecosystems provide other benefits, including fishery habitat, 1 2

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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*).

4 5 When degraded, these co-benefits are greatly diminished along with the ecosystems' capacity to sequester 6 carbon, and stored carbon can be released back to the atmosphere, along with other greenhouse gases (Marba 7 and Duarte, 2009; Duarte et al., 2010; Pendleton et al., 2012b; Lovelock et al., 2017). Around 25-50% of 8 mangrove forests and other vegetated coastal habitats have already been lost or degraded due to coastal 9 developments and other land-use change during the past 100 years (McLeod et al., 2011). Current losses are 10 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 11 12 13 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 14 15 change: 1.3 ± 0.7 GtC yr⁻¹ (Le Quéré et al., 2018). However, Lovelock et al. (2017) note that several 16 processes could reduce their maximum value. These factors include organic matter remaining in anoxic 17 conditions after vegetation is removed; interactions between CO_2 and carbonates in sediments; and the 18 offshore transport and reburial of organic carbon. 19

Awareness of the scale of potential CO₂ 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:

31 Improved regulatory protection. Legal structures already exist in many countries to protect mangrove 32 forests. The main need is for much increased enforcement, for both nation-wide regulation and site-33 specific marine protected areas (Miteva et al., 2015; Howard et al., 2017). The lack of such capacity to 34 date has allowed loss and degradation to continue (Herr et al., 2017). There is currently much less 35 regulatory protection for salt marshes and seagrass habitats, not yet well-mapped in many regions 36 (Duarte et al., 2017). Whilst some salt marsh protection does now exist in Europe and North America, it 37 may be 'too little, too late' for significant climate mitigation; in NE Europe, ~90% of estuarine wetlands 38 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 longterm 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.
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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 4 significantly alter the scale of climatic benefits (Adams et al., 2012; Chen and Ganapin, 2016; Chmura et al., 2016; Rosentreter et al., 2018).
- 6 Carbonate formation in some blue carbon systems may release CO_2 , offsetting the benefits of carbon 7 storage (Howard et al., 2017; Macreadie et al., 2017).
- 8 Some of the carbon stored in marine sediments may be recalcitrant carbon from terrestrial or 9 atmospheric sources, and should therefore be excluded (Chew and Gallagher, 2018).
- 10 • 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) 11 12 and sea level rise (Kirwan and Megonigal, 2013; Spencer et al., 2016). However, carbon sequestration 13 capacity may not necessarily decrease; e.g., if mangroves replace salt marshes (Kelleway et al., 2016), or 14 where totally new habitats might be created (Barnes, 2017).

15 16 The above complexities currently make it difficult to provide accurate estimates of the climate mitigation 17 achieved through blue carbon management measures. Furthermore, even if all coastal blue carbon 18 ecosystems could be restored to their 1990 extent (unlikely, because of the semi-permanent nature of most 19 coastal land-use change, such as human settlement and conversion to agriculture), their contribution to the 20 'emissions gap' (UNEP, 2017a) would be relatively modest at the global scale (high confidence). For 21 example, Griscom et al. (2017) estimated that maximum global mitigation benefits of cost-effective coastal 22 wetland restoration could be ~0.05 GtC yr⁻¹ by 2030, compared to current net emissions from all sources of 23 10.0 GtC vr^{-1} (Le Ouéré et al., 2018). Gattuso et al. (2018) similarly concluded that coastal restoration and 24 protection only offered a very limited (and relatively unreliable) 'solution' to climate change. Nevertheless, 25 measures to protect and, where practicable, re-instate such habitats provide many other societal benefits 26 through ecosystem services; in particular, benefitting local fisheries, enhancing biodiversity, giving storm 27 protection and reducing coastal erosion. Therefore, coastal blue carbon can be considered as a 'no regrets' 28 option at the national level in many countries, in addition to other, higher magnitude, mitigation measures. 29 Additional process studies are needed over the full range of environmental conditions to improve the *low* 30 confidence in current estimates of carbon loss from degraded vegetated coastal ecosystems, covering more 31 than an order of magnitude range.

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33 5.5.1.1.3 Seaweeds and shelf sea sediments

34 Seaweeds (macroalgae) do not directly transfer carbon to marine sediments, unlike rooted coastal vegetation 35 (macrophytes, as considered above) (Howard et al., 2017). Nevertheless, seaweed detritus can deliver carbon 36 to sedimentary sites (Hill et al., 2015) and may also provide a source of refractory dissolved organic carbon 37 (RDOC) (Hughes et al., 2012). Recent studies indicate that globally-important amounts of carbon may be 38 involved in these processes (Krause-Jensen and Duarte, 2016; Krause-Jensen et al., 2018; Smale et al., 39 2018). Nevertheless, there is low confidence that seaweed provides a significant natural contribution to 40 longterm permanent carbon removal, due to large uncertainties relating to transport pathways and the fate of 41 material transported to deeper water. The balance between carbon uptake and the scale of its subsequent re-42 release to the atmosphere is the key issue, rather than uptake *per se*. Large-scale and multi-year tracking 43 studies would be needed to provide reliable data on carbon sequestration over decadal to century timescales.

44 45 Seaweed aquaculture is inherently more manageable for mitigation purposes (N'Yeurt et al., 2012; Chung et 46 al., 2013; Chung et al., 2017; Duarte et al., 2017). If linked to biofuel/biogas production (N'Yeurt and Iese, 47 2014; Moreira and Pires, 2016; Sondak et al., 2017), there would be potential to reduce emissions (as an 48 alternative to fossil fuels); if also linked to carbon capture and storage (Hughes et al., 2012), there could be 49 potential to achieve negative emissions (net CO₂ removal from the atmosphere). Full life cycle analyses are 50 needed to assess the energy efficiency of such approaches, and the viability of their scaling-up to 51 climatically-significant levels. N'Yeurt et al. (2012) envisage 9% of the global ocean used for seaweed 52 aquaculture; however, they did not assess the associated environmental and socio-economic implications. A 53 different mitigation option using seaweeds relates to their use as a dietary supplement for ruminants, to 54 suppress methane production. In vitro studies have given promising results (Dubois et al., 2013; Machado et 55 al., 2016; Machado et al., 2018); the need is to quantify the potential scale of real-world benefits. Until that 56 has been done, there is only *low confidence* in this approach having significant mitigation potential.

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1 Actions to increase the security of carbon storage in shelf sea sediments primarily relate to the management 2 of human activities that cause sediment disturbance, through fishing, dredging and the installation of 3 offshore structures, hence affecting carbon storage and cycling both directly (physically) and indirectly 4 (biologically, through effects on benthic organisms) (Hale et al., 2017). Carbon retention, over relatively 5 large areas of shelf seas, might therefore be increased (Avelar et al., 2017). However, the effects involved 6 are complex (van de Velde et al., 2018), with lack of data on the natural variability of the processes 7 involved. In particular, there have been few large-scale studies on 'baseline' carbon sequestration rates in 8 shelf sea sediments, and of the factors affecting storage in the potentially-mobile fraction; exceptions are 9 provided by Hu et al. (2015) and Diesing et al. (2017). Due to these uncertainties, there is currently low 10 confidence that shelf sea sediments can be used for climate mitigation purposes. 11

12 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.

19 Such ocean alkanization methods might be used to reduce local impacts, e.g., for coral reefs (Albright et al., 20 2016; Feng et al., 2016), as well as contributing to wider counteraction of climate change. Rock-weathering 21 with associated permanent CO₂ uptake is a natural process, although occurring relatively slowly: it currently 22 sequesters ~0.25 GtC yr⁻¹, on land and at sea (Taylor et al., 2015). The main constraints on the development 23 and deployment of ocean alkalinization as a mitigation method relate to the governance, cost, and uncertain 24 environmental impacts (on land as well as at sea) of climatically-significant application (Gattuso et al., 25 2018). The combination of these factors results in *low confidence* that seawater alkalinization can provide a 26 viable and acceptable climate mitigation approach. 27

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.

33 5.5.1.1.5 *Refractory dissolved organic carbon ('land-sea integrated eco-engineering')*

34 Land-based measures have also been proposed, affecting marine carbon storage in the water column through 35 nutrient management and effects on dissolved organic carbon. Measurements of air-sea CO₂ fluxes (Laruelle 36 et al., 2015) and modelling studies (Bauer et al., 2013) indicate that most shelf seas are currently a net 37 carbon sink, with anthropogenic nutrients implicated through their role in stimulating shelf primary 38 production, hence CO₂ drawdown (Bauer et al., 2013; Regnier et al., 2013). However, eutrophic waters can 39 also be carbon sources, with rapid re-mineralisation of organic material by microbially-driven 40 decomposition, returning CO₂ to the atmosphere (Jiao et al., 2011). If land-based nutrient inputs can be 41 reduced, a higher proportion of refractory DOC may be produced (Jiao et al., 2014b). This idea is supported 42 by a statistical analysis of the relationship between organic carbon and nitrate in various natural 43 environments (Taylor and Townsend, 2010) as well as by experimental results in estuarine and offshore 44 waters (Yuan et al., 2010; Jiao et al., 2011; Jiao et al., 2014b).

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46 Improved management of chemical fertilizers may pronote DOC formation. Chemical fertilizers have been 47 excessively applied in farming for decades especially in developing countries. Excessive N and P are then 48 washed out into rivers and ultimately discharged into the coastal waters, causing eutrophication and algal 49 bloom (both red-and green tide) (Zhang et al., 2018). Although algal blooms seemingly produce more 50 organic carbon, this carbon is basically labile and can be respired rapidly. In addition, the labile DOC 51 produced by autochthonous primary producers has priming effects on the river discharged terrestrial RDOC. 52 i.e., remobilize RDOC for microbial uptake and respiration (Liu et al., 2014), which can create high CO₂ 53 concentration causing acidification in ambient water, and excess CO₂ can escape from water to atmosphere 54 as outgassing. That is why productive estuarine and coastal waters are often sources rather than sinks of 55 atmospheric CO₂ (Jiao et al., 2018a; Jiao et al., 2018b). Meanwhile this process consumes large quantity of 56 oxygen resulting in hypoxia. Anoxic conditions could cause massive death of macro- and micro- biomes, 57 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).

6 Therefore, reducing terrestrial input of inorganic nutrients could be a countermeasure for the enhancement of 7 carbon sequestration in coastal waters (Jiao et al., 2011). Land-ocean integrated management and 8 engineering could help reduce the application of chemical fertilizers in farming and eliminate sewage 9 discharge into the river so as to reduce the N, P inputs into the sea. Such eco-engineering could have the co-10 benefits of protect coastal ecosystems by reducing eutrophication and occurrence of red-tides while 11 increasing carbon sequestration through the MCP. Land-ocean eco-engineering may also be more effectively 12 implemented in association with reward-based carbon trading policy to replace the penalty-based pollution 13 policy that is currently being in practice in many countries (Jiao et al., 2018b). 14

15 5.5.1.2 Climate Mitigation in the Open Ocean 16

17 5.5.1.2.1 Open ocean carbon cycle: opportunities and challenges

18 Current net primary production by marine phytoplankton is estimated to be 58 ± 7 GtC yr⁻¹, similar to 19 terrestrial primary production (Buitenhuis et al., 2013). However, only ~1% of that marine production is 20 removed annually from circulation by the combined effects of the three biologically-driven ocean carbon 21 pumps (soft-tissue pump, carbonate pump and microbial carbon pump) (Legendre et al., 2015). It is the 22 concentration gradient between (increasing) atmospheric CO_2 and dissolved CO_2 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– 23 24 2015 (~25% of anthropogenic CO₂ emissions; Le Quere et al. (2016)). The residence time of dissolved 25 inorganic carbon (DIC, arising from atmospheric uptake and the relatively rapid remineralization of most 26 organic material) depends on ocean circulation, varying from days to centuries, that of refractory dissolved 27 organic carbon (RDOC) may be as long as 5000 years, while the organic carbon buried in marine sediments 28 could be permantly removed from circulation. 29

Two main methods have been proposed to increase CO₂ uptake by biological processes in the open ocean: chemically, by adding terrestrially-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.

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38 5.5.1.2.2 Ocean fertilization

39 The potential mitigation benefits of ocean iron fertilization have been investigated experimentally, also by 40 modelling and by observations of natural system behaviour (Keller et al., 2014; Bowie et al., 2015; Kearney 41 et al., 2015; Tagliabue et al., 2017). The 13 experimental studies to date (7 in the Southern Ocean, 5 in the 42 Pacific, and one in the sub-tropical Atlantic) have shown that primary production can be greatly enhanced by 43 the addition of iron to certain ocean regions (Yoon et al., 2016). However, most of the additional carbon 44 uptake is only temporarily removed from the atmosphere. Iron fertilisation implemented on a climate-scale, 45 if allowed, would involve many technical difficulties in verification of the enhanced carbon sequestration, 46 and in monitoring its potentially-adverse impacts (Bowie et al., 2015; Williamson and Bodle, 2016). There 47 are also many contested governance issues (Williamson et al., 2012; Boyd and Bressac, 2016; Fuentes-48 George, 2017; McGee et al., 2018), particularly for the Southern Ocean where ocean iron fertilization is 49 considered to be most effective (Robinson et al., 2014). Ocean iron fertilization is regulated by the London 50 Protocol, with amendments prohibiting such action unless constituting legitimate scientific research, 51 authorized under permit (see Section 5.5.4.1).

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53 Open ocean fertilization by macro-nutrients (e.g., nitrate) has also been proposed, with modelled potential 54 for gigaton-scale carbon removal (Harrison, 2017). Similar technical and governance considerations would, 55 however, apply with regard to the quantification of mitigation benefits, the monitoring of potential adverse 56 impacts, and the political acceptability of large-scale deployment.

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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.

14 15 For an introduction to climate change adaptation within this special report, including climate change 16 adaptation in natural and human systems, see Section 1.5.2. Section 5.5.2 begins with an assessment of 17 adaptation frameworks (Section 5.5.2.1), followed by assessments of the following three main focus areas of 18 adaptation response measures: ecosystem-based (5.5.2.2), built infrastructure (5.5.2.3) and socio-institutional 19 (5.5.2.4) adaptation (see Table 5.7 for a description of these categories). Finally, the costs of, and limits to, 20 adaptation is assessed in Section 5.5.2.5. Section 5.5.2 avoids, where possible, focused discussion of 21 response to sea level rise and extreme events (including heat waves, and compound and cascading events) to 22 avoid duplication with Chapter 4 (see Table 4.6 for examples) and Chapter 6, respectively.

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 Table 5.7: Description of the three main focus areas of Section 5.5.2.

Section	SROCC description
5.5.2.2 Nature-based/	The use of biodiversity and ecosystem services as part of an overall adaptation
Ecosystem-based 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. (Naravan et al., 2016b:
	Moosavi, 2017)
5.5.2.3 Built Environment Adaptation Solutions	Adaptation solutions pertaining to coastal built infrastructure and the systems that support such infrastructure. (Mutombo and Ölçer, 2016b; Forzieria et al., 2018)
5.5.2.4 Socio-Insitutional Adaptation Solutions	Adaptation responses within human social, governance and economic systems and sectors, including community-based adaptation. (Oswald Beiler et al., 2016; Thorne et al., 2017)
5.5.2.4.1 Community-based Adaptation	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: Shaffril et al. 2017a)

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28 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.,

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1 2017; Ramm et al., 2017a; Mavromatidi et al., 2018). Seasonal and decadal forecasting tools have improved 2 rapidly since AR5, especially in supporting management of living marine resources (Payne et al., 2017; 3 Tommasi et al., 2017) and modelling to support decision-making processes (Čerkasova et al., 2016; 4 Chapman and Darby, 2016; Jiang et al., 2016; Justic et al., 2016; Joyce et al., 2017; Mitchell et al., 2017). 5 Decision-making processes are supported by economic evaluations (Bujosa et al., 2015; Jones et al., 2015), 6 evaluations for ecosystem services (MacDonald et al., 2017; Micallef et al., 2018) and participatory 7 processes (Byrne et al., 2015). A growing body of literature is emerging on the development of adaptation 8 pathways, frameworks and decision making (Buurman and Babovic, 2016; Dittrich et al., 2016; Michailidou 9 et al., 2016; Cumiskey et al., 2018; Osorio-Cano et al., in press), as well as the development of indicators to 10 support evaluation of adaptation actions (Carapuço et al., 2016; Nguyen et al., 2016) through monitoring 11 frameworks (Huxham et al., 2015). 12 13 There is *high agreement* on the need for the integration of scientific research into management decision-14 making for climate adaptation (Endo et al., 2017; Thorne et al., 2017), particularly through better 15 understanding coastal ecosystems and human adaptation processes (Dutra et al., 2015; Cvitanovic et al., 16 2016). This includes consideration of non-climate change-related factors. Areas of scientific research that are 17 considered useful to facilitate the use of scientific knowledge for climate adaptation include multiple 18 stakeholders participatory planning (Archer et al., 2014b; Abedin and Shaw, 2015), transboundaries ocean 19 management (Gormley et al., 2015; Williams et al., 2016), scientific research to deepen understanding of 20 ecosystem- (Hobday et al., 2015b; Dalyander et al., 2016; McNeeley et al., 2017; Osorio-Cano et al., in 21 press), community-based adaptation with socio-economic outcomes (Merkens et al., 2016) and development 22 of climate resilient development pathways (high confidence). In addition, research on applying 'big data' and 23 high end computational capabilities could help develop a comprehensive understanding of climate and non-24 climate variables in planning for coastal adaptation (Rumson et al., 2017). Knowledge from these research 25 areas could improve the planning, implementation and monitoring of climate adaptation actions for marine 26 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).

36 Overall, challenges with adaptation planning include uncoordinated, top-down approaches, a lack of political 37 will and insufficient resources (Elias and Omojola, 2015). For example, in some countries, despite 38 investment in research and science having improved local government practitioners' technical and cognitive 39 ability, implementation of adaptation actions has been impeded by budget cuts and lack of national level 40 political support (Porter et al., 2015). In a survey of 150 Pacific Northwest coastal natural resource 41 managers, time, money and staff resources were their largest adaptation obstacles. Also, different levels of 42 access to information corresponded with differences in preparedness and willingness to plan for adaptation 43 (Thorne et al., 2017). 44

45 A number of key principles are important in developing adaptation frameworks. A robust but flexible 46 approach can deliver adaptation responses across a range of projected climate scenarios. This can be 47 achieved by accounting for deep uncertainty through well-coordinated participatory processes (Jiao et al., 48 2014b; Dutra et al., 2015; Dittrich et al., 2016; Buurman and Babovic, 2017). Monitoring programmes are 49 important for effective implementation of adaptation actions (Barrett et al., 2015; Bell et al., 2018b). Other 50 principles include taking a whole system approach that addresses underlying socio-ecological compounding 51 factors (Sheaves et al., 2016b), including cost-benefit analyses (André et al., 2016) and providing co-benefits 52 for human development and the environment (Wise et al., 2016). 53

54 There is a notable dearth of studies from African and Caribbean least developed countries and small island 55 developing states (Kuruppu and Willie, 2015) and few locally-focused coastal climate change adaptation 56 planning tools (Torresan et al., 2016). This is partly due to the highly complex nature of adaptation response 57 in these countries. In contrast, the coastal adaptation framework literature is dominated by Australian, North 1 American and European cities. There are large sub-national variations even in developed countries. For 2 example, in Australian coastal areas, the small number of available case studies suggests that only a few 3 coastal local governments have begun adaptation planning (Ramm et al., 2017b). Interest in adaptation effort 4 is more likely in individuals experiencing extreme weather (Ray et al., 2017). For example, nature-based 5 solutions and improved coordination effort increased in USA cities located in Atlantic coastal regions 6 following Hurricane Sandy (Oswald Beiler et al., 2016). Positive responses, reported from developing 7 countries, were expressed in terms of successes with community- and ecosystem-based adaptation 8 approaches. The difference in adaptation assessments between developed and developing economies and 9 cities suggests some divergence in approach to addressing climate change (medium confidence).

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11 To pursue a climate resilient development pathway, more extensive learning processes are necessary to build 12 decision-makers' capacity to tackle systemic drivers, and to scrutinize potentially maladaptive infrastructural 13 investments (Wise et al., 2016). This will require effective coordination across a range of stakeholders within 14 and between organizations, and substantial effort is required in developing countries to close the gap in 15 adaptation planning with developed countries (high confidence). In contrast with the many examples of 16 proposed frameworks for adaptation, there remains relatively little literature that has assessed the success of 17 implementing such frameworks, possibly due to the time-lag between implementation, monitoring and 18 evaluation and reporting. In terms of developing climate resilient development pathways (see Cross-Chapter 19 Box 1 in Chapter 1), there was substantial support for 'no regrets' approaches addressing both proximate and 20 systematic underlying drivers of vulnerability (Sánchez-Arcilla et al., 2016; Pentz and Klenk, 2017; 21 Zandvoort et al., 2017) with leadership, adaptive management, capacity and the monitoring and evaluation of 22 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 socioinstitutional 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.

Ecosystems/ Human systems	Hazards	Biophysical drivers	Adaptation response	Assessment (Confidence)
Coral reefs	Bleaching, Inundation, Dissolution, Bioerosion, Structural damage	Ocean warming, Sea level rise, Ocean acidification, Extreme storm events	Preventative or adaptive restoration, assisted evolution	Active restoration, still in its infancy, can be applied in a wide range of ecological and socio- economic applications (<i>high</i>). Coral epigenetics, 'assisted colonization', 'assisted evolution', novel coral–symbiont associations and coral microbiome manipulations as adaptive management tools for reef rehabilitation (<i>low</i>)
Mangroves, coastal wetlands, seagrass beds and saltmarshes	Storm damage, Inundation	Extreme storm events, sea level rise	Synergistic ^a adaptation responses, socio-institutional responses	Employing synergistic ecosystem- based adaptation responses are a cost-efficient and effective response, especially when employed alongside socio-institutional responses (<i>high</i>).

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Coastal dunes and sandy beaches	Erosion, Inundation, Fire damage	Extreme storm events, sea level rise, heat waves/ drought	Replenishment schemes and dune restoration, socio- institutional responses including stakeholder engagement	Rate of loss of sand and level of participation of stakeholders are important factors (<i>medium</i>)
Species	Direct physiological impacts, Habitat transformation, Foraging success decline, Reproductive failure	Various climate change driven variables (direct and indirect)	Species range shift, behavioural change, human facilitated responses including governance, conservation etc.	<i>Virtually certain</i> to be winners and losers, knowledge gaps need to be filled
Integrated coastal zone management	Loss of biodiversity and ecosystem change, Loss of economically important species, Infrastructure damage	Climate drivers of ecosystem change, sea level rise and storm surge	Stakeholder participation (bottom up-top down), Marine Protected Areas, strengthening Indigenous Knowledge and Local Knowledge, high future need for transboundary agreements	<i>High confidence</i> in efficacy of stakeholder participation and MPAs, but some resistance from conservative administrations to the former, <i>medium confidence</i> in IK/LK (Cross-Chapter Box 3 in Chapter 1), <i>high agreement</i> on need for transboundary agreements.
Tourism	Tourism industry decline (changing behaviour and choices), Infrastructure loss, Resource declines	CO ₂ emissions, changes to ecosystems, sea level rise/ storm surge, climate impacts on resources e.g., water availability	Participatory planning processes, ecosystem based adaptation, community based adaptation, awareness raising	<i>High confidence</i> that climate change will impact upon tourism, <i>high</i> <i>agreement, limited evidence</i> that a range of adaptation responses will be required locally, <i>likely</i> winners and losers
Fisheries	Reduction in landings and revenues, Changes in species composition, Ecosystem change	Climate drivers of changes in species distributions	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	<i>High agreement</i> that optimal fisheries management integrated with community responses will be critical to offsetting global fisheries landings declines, <i>high agreement</i> that knowledge gaps in fisheries management need to be filled for successful adaptation, <i>medium</i> <i>confidence</i> in ability of community- based responses to offset social impacts.

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 building initiatives, climate resilient development	<i>High confidence</i> in EAA and ICZM approaches, <i>high confidence</i> 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.

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 5.5.2.2.4. 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.

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).

24 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 26 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, 28 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 30 colonies, together with recruited associated biota, are transplanted to degraded reef sites (Shafir and Rinkevich, 2008; Mbije et al., 2010; Shaish et al., 2010b; Shaish et al., 2010a; Bongiorni et al., 2011; 32 Horoszowski-Fridman et al., 2011; Linden and Rinkevich, 2011; Mbije et al., 2013; Cruz et al., 2014; Chavanich et al., 2015; Horoszowski-Fridman et al., 2015; Lirman and Schopmeyer, 2016; Montoya Maya 34 et al., 2016; Ng Chin Soon et al., 2016; Lohr and Patterson, 2017; Rachmilovitz and Rinkevich, 2017). 35 Active restoration of coral reefs, while still in its infancy and facing a variety of challenges (Rinkevich, 36 2015b; Hein et al., 2017), is now capable to handle a wide range of ecological and socio-economic 37 applications (Rinkevich, 2014; Rinkevich, 2015b; Rinkevich, 2017) (high confidence).

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39 Recent studies focusing on ecological engineering approaches (that also include the gardening approach)

- 40 (Rinkevich, 2014; Forsman et al., 2015; Coelho et al., 2017; Horoszowski-Fridman and Rinkevich, 2017;
- 41 Linden and Rinkevich, 2017; Rachmilovitz and Rinkevich, 2017), species interactions that reef managers
- 42 may utilize to facilitate the restoration of corals and/or the augmentation of functional diversity (including
- 43 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
- 45 genetic diversity (Iwao et al., 2014; Drury et al., 2016; Horoszowski-Fridman and Rinkevich, 2017). Active 46 restoration becomes a major tool for reef rehabilitation in many countries worldwide (representing all major

reproduction (Horoszowski-Fridman et al., 2011).

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.,

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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.

14 While predictions for the future reef statuses under climate change are gloomy (*high confidence*),

15 contemporary active reef restoration (Box 5.4) has a promise to be used as a major management and adaptive 16 tool for the rehabilitation of the global change impacts on coral reefs (*high confidence*). The new suggested 17 adaptive management tools (conducted to harness basic science to develop corals with capacity to withstand 18 climate change impacts; Box 5.4) may further improve the active reef restoration outcomes (low confidence). 19 Results will be intensified with the employment of ecological engineering approaches (high confidence). It is 20 envisaged that the 'reefs of tomorrow' will not resemble the reefs of today or the pristine reefs (Box 5.4) 21 (high confidence) (Section 5.3.3.7), thus it is suggested to prepare a coral restoration toolbox (Rinkevich, 22 2014; Rinkevich, 2015a; Rinkevich, 2015b) for cases and the time it is needed. 23

25 [START BOX 5.4 HERE]26

Box 5.4: Coral Reef Restoration as Ocean-based Adaptation

28 29 Global change induced by human activities is impacting all warm-water corals (and the reef structures they 30 form) (Section 5.2.2.3.3). Impacts are rapidly increasing in scale and intensity, exposing coral reefs to 31 enhanced degradation rates and diminishing capacities to maintain ecological resilience, to absorb 32 disturbances, and to adapt in time to the expected changes (Box 5.1) (Graham et al., 2014; Rinkevich, 2015a; 33 Harborne et al., 2017). However, traditional coral reef conservation measures, aimed at protecting reefs from 34 human activities and allowing natural processes to mitigate human impacts, are increasingly being 35 acknowledged as insufficient to address climate change, Section 5.2.3.2). Instead, restoration measures are 36 being called for (Rinkevich, 1995; Rinkevich, 2000; Barton et al., 2017). However, various recently 37 suggested approaches for rebuilding coral reef resilience such as the 'assisted colonization' (actively moving 38 species that are confined to disappearing habitats) (Hoegh-Guldberg et al., 2008; Chauvenet et al., 2013), the 39 'assisted evolution' (aiming to develop corals resistant to climate change via accelerated natural evolution 40 processes), novel coral-symbiont associations (McIlroy and Coffroth, 2017) and coral microbiome 41 manipulation (Bourne et al., 2016; Sweet and Bulling, 2017; van Oppen et al., 2017) are still at the 'proof-42 of-concept' stage. Others, such as the use of artificial reefs (Ng Chin Soon et al., 2017) are limited in 43 impacts, and all are also revealing considerable challenges (Riegl et al., 2011; Coles and Riegl, 2013; 44 Ferrario et al., 2014).

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46 Today, the most tested reef restoration approach involves 'coral gardening', a two-step process, similar to 47 silviculture concepts and practices (Rinkevich, 2006). About 90 coral species and over 100,000 colonies that 48 were farmed in various archetypical nurseries across the world, followed by a broad range of transplantation 49 tactics showed high survivability, growth rates and reproduction (Rinkevich, 2014). Whereas the 'coral 50 gardening' restoration techniques are still lagging behind restoration practices developed for terrestrial 51 habitats, reef restoration has advanced greatly in the last two decades (Rinkevich, 2014; Lirman and 52 Schopmeyer, 2016). This advancement includes economic considerations (Rinkevich, 2015b; Barton et al., 53 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 Schopmeyer, 2016; Montoya Maya 54 55 et al., 2016; Jacob et al., 2017; Rachmilovitz and Rinkevich, 2017) have added ecological engineering

56 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).

3 4 However, the 'coral gardening' approach is facing two foreseen challenges. The first is the scaling up issue, 5 the viability of this approach as a large-scale restoration measure at the level of hundreds of thousands/ 6 millions of coral colonies/site (aiming at the elucidating of the sum ecological attributes at large-scale 7 approaches) (Rinkevich, 2014). The second challenge (Box 5.4, Figure 1), is the effectiveness of active reef 8 restoration to mitigate or rehabilitate global change impacts (Shaish et al., 2010a; Schopmeyer et al., 2012; 9 Coles and Riegl, 2013; Hernández-Delgado et al., 2014; Rinkevich, 2015a; Wilson and Forsyth, 2018) and 10 whether it is applicable for tracking future increases in sea level (Perry et al., 2018), primarily in low lying 11 ocean states. Also, the literature reveals a wide range of calculated restoration costs that did not consider the 12 high monetary value of the coral reefs' ecosystem services (Bayraktarov et al., 2016; Rinkevich, 2017). 13

14 Yet, it has been proposed (Rinkevich, 2015a) that 'coral reef gardening' has the capacity to enhance 15 adaptability of coral reef organisms to climate change. The implementation of reef restoration approaches 16 and the climate change induces poleward range expansion of corals' reef habitats (Precht and Aronson, 17 2004; Yara et al., 2011; Madin et al., 2016), altogether the evidence suggests that the 'reefs of tomorrow' 18 would not resemble the 'reefs of today' (Box 5.4, Figure 1) (Rinkevich, 2008; Ban et al., 2014) (high 19 confidence). The very high vulnerability of coral reefs to warming over reefs, ocean acidification, the 20 increase in storms' intensity and severity and sea level rise under climate change (AR5 WG2;), including 21 enhanced bioerosin (Schönberg et al., 2017) (high confidence) point to the importance of considering both 22 mitigation (Section 5.5.1) and adaptation for coral reefs, the latter includes active restoration measures and 23 for improved cultivation and transplantation methodologies.

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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'

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(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 antidipated 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]

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5.5.2.2.2 Mangrove and other coastal ecosystems

16 Mangrove swamps provide significant coastal protection services from extreme storm events, providing 17 supporting services through increased sedimentation rate (Hayden and Granek, 2015) and provisioning 18 services for local communities (Section 5.4.1.3). Mangroves can also provide carbon mitigation and job 19 creation co-benefits (for examples through Reducing Emissions from Deforestation and Forest Degradation 20 (REDD+) programs) when managed properly (5.5.1.1), and there is evidence of their value for aquaculture 21 initiatives (Huxham et al., 2015; Ahmed and Glaser, 2016). When used in combination with other adaptation 22 responses, their value increases. For example in Bangladesh sedimentation rates increased when mangroves 23 were planted on the lee side of an oyster farm, with the latter serving to reduce wave energy reaching the 24 mangroves (Ahmed and Glaser, 2016). On the other hand, loss of mangrove swamps and fringing coral reefs 25 would result in the loss of coastal protection services by these ecosystems against future cyclones in 26 Bangladesh (Ataur Rahman and Rahman, 2015; Ahmed and Glaser, 2016), so effective management is 27 important. 28

29 There are many examples of coastal protection services provided by mangrove forests, which have been 30 restored through community-based mangrove restoration programs, and such programs can provide 31 additional community and carbon sequestration benefits. There are also, however, examples where such 32 initiatives have resulted in maladaptation, generally as a result of poor governance processes or lack of 33 community compliance with restoration plans. This emphasizes the need for effective implementation of 34 adaptation responses with broad community participation. In a specific example of mangrove management 35 for climate adaptation that included community-focused mangrove management plans in Vietnam, the 36 importance of providing technical guidance to local communities as part of the adaptation action was noted 37 (Nguyen et al., 2017), along with the need for the incorporation of mangrove regulations into local planning 38 instruments (Sierra-Correa and Cantera Kintz, 2015). Community participation in the development climate 39 change adaptation for mangroves could improve its outcomes and co-benefits (medium evidence, high 40 agreement).

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42 Coastal developments and coastal squeeze transform wetlands and marshlands which reduces the ability of 43 these ecosystems to provide protection from wave and storm impacts, whilst releasing carbon into the 44 atmosphere (Wong et al., 2014). The level of protection seagrass beds provide depends partly upon their 45 ecological health, so their optimal management is an important adaptation response, especially when paired 46 with synergistic ecosystems like mussel beds on the seaward side of seagrass beds to reduce wave energy 47 and erosion (Ondiviela et al., 2014).

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49 Coastal dune systems are widely transformed globally; human disturbance and a low stabilizing ability of 50 vegetation are key causes of degradation (Ciccarelli et al., 2017) with restoration effort supported by dune 51 builder species and sand replenishment schemes. Section 4.4.4.1 provides an overview of sediment-based 52 adaptation response measures, including cost estimates for beach nourishment and dune maintenance, a 53 discussion of co-benefits and drawbacks of combining hard and soft infrastructure measures, and challenges 54 with sourcing sediment for beach replenishment. There are a number of examples of sand replenishment 55 projects as a soft measures approach to combatting erosion; see Vikolainen et al. (2017) for an example. 56 Such projects have, however not always been successful due to, for example fire damage (Shumack and 57 Hesse, 2017) or the rapid loss of sand within replenishment schemes due to coastal processes, and in some 58 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).

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7 5.5.2.2.3 Ecosystem-based adaptation assessment

8 This ecosytem-based adaptation (EBA) assessment was supported by a growing body of literature and 9 increased international funding for this approach, yet a synthesis of context-specific application and cost-10 effectiveness is a remaining gap (Narayan et al., 2016a). 'Soft' engineering options have a range of co-11 benefits like increasing ecological complexity with multiple services provided and a wide range of economic 12 benefits for society and resilience to climate change (Perkins et al., 2015; Perry, 2015; Moosavi, 2017; 13 Scarano, 2017). In their analysis, Narayan et al. (2016a) found that coral reefs and salt-marshes performed 14 best at reducing wave heights, whilst salt-marshes and mangroves were two to five times cheaper than 15 submerged breakwaters for wave heights less than half a meter. The application of EBA approaches can be 16 made more effective by incorporating traditional practices and knowledge (Ataur Rahman and Rahman, 17 2015).

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19 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;

21 MacDonald et al., 2017) (*medium evidence, high agreement*). As important as the outcomes of

22 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).

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27 The lack of the inclusion, and economic undervaluation, of intertidal ecosystem services undermines 28 coastline management, and this is compounded by ecological data gaps (Perkins et al., 2015). In an analysis 29 of urban climate planning, a good awareness of EBA measures was reported in Europe, but baseline 30 information was lacking, along with convincing implementation actions (Geneletti and Zardo, 2016). In 31 summary, the benefits of EBA are widely acknowledged in the literature, and the sum of these benefits 32 strongly suggests that they should be a key tool used for coastal climate change adaptation, but with an 33 acknowledgement of the limitations inherent in this adaptation response (*high confidence*). As with AR5, 34 further research evaluating natural infrastructure is required (Roberts et al., 2017).

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36 Ecosystem-based Adaptation (EBA) is considered to be a cost-effective coastal protection tool that has 37 multiple co-benefits. Management of ecosystems provides the opportunity for job creation, as communities 38 living adjacent to ecosystems, like mangrove forests, are incorporated into mangrove rehabilitation and 39 maintenance programs. Mangrove restoration and conservation also creates carbon sequestration 40 opportunities, and importantly, such ecosystems under effective management can recover from damage after 41 extreme events (whereas built infrastructure requires rebuilding). This effectively gives natural-based 42 adaptation solutions a cost-effective advantage over built infrastructure, besides providing other co-benefits, 43 including biodiversity conservation, utility and recreational value. 44

45 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).

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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

1 physiological tolerance, but loss of sea-ice habitat could reduce (or alternatively increase) foraging habitat 2 (Castellini, 2017). Human management of ecosystems for climate change will, therefore, require a focus 3 beyond traditional mandates (Hobday et al., 2015a). Species adaptation responses that address both the direct 4 and indirect impacts of climate change, as well as changes to ecosystems supporting species, for example 5 transformation through alien invasive species (Whitfield et al., 2016), will require broader consideration than 6 traditional species management practices (medium evidence, good agreement). This will include the 7 development of international collaborations and databases to improve ocean-scale understanding of climate 8 change impacts (Okey et al., 2014; Young et al., 2015). A key knowledge gap of where the critical 9 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.

20 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.

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29 Sea level rise (SLR) causes, projections, impacts and adaptation response is dealt with in detail in SROCC 30 Chapter 4. Wave regime and salinity act synergistically with SLR in coastal locations and are considered 31 next. Increases in extreme ocean wave heights will affect coastal infrastructure and impact many vulnerable 32 populations of low-lying islands (Vanem, 2015; Shope et al., 2016; Simon et al., 2016; Wadey et al., 2017). 33 In the Western Pacific Solomon Islands, shoreline recession as a result of extreme events, seawalls and 34 inappropriate development has badly damaged infrastructure in villages (Simon et al., 2016). Shoreline 35 recession is positively associated with areas of high wave energy, so understanding the relationship between 36 wave surge, wave energy and development is critical. Large wave events in the tropical Pacific are *likely* to 37 decrease from December to February, but the top 5% of significant wave heights will increase from June to 38 August (Shope et al., 2016). This increase, accompanied by changes in direction, will increase flooding and 39 shoreline change. Wave height will increase and swell direction is projected to shift in a counter-clockwise 40 direction in the North Pacific above 50° (Erikson et al., 2015). Climate variability, including the El Niño-41 Southern Oscillation (ENSO), North Atlantic Oscillation (NAO), and Pacific decadal oscillation (PDO), will 42 have further, region specific impacts upon wave height, wind speed and mean sea level pressure gradient 43 amplitude (Kumar et al., 2016).

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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).

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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

56 overtopping of waves (Kong et al., 2013). As with sea level rise, to reduce the risk of material displacement, 57 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).

3 4 Section 5.3.2 concludes that it is very likely that regional changes in coastal salinity correspond with the 5 dominance of either evaporation or precipitation, with surface freshening at high latitudes due to ice melt. 6 Increased salinity may accelerate deterioration of infrastructure erected in areas exposed to sea water, for 7 example coastal heritage structures (Kaja and Mallic, 2017) or affect operations of infrastructure. Salinity 8 affects built infrastructure mostly through inundation, which is closely related to sea level rise and/ or storm 9 surge, but infrastructure may also increase inundation through subsidence (Aslam et al., 2015). High salinity 10 levels can affect the operation of wastewater treatment plants and reduce the quality of treated water, corrode 11 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 12 13 Durban, South Africa are considered to be vulnerable to saline intrusion (Friedrich and Kretzinger, 2012). 14 Corrosion of steel infrastructure, or deterioration of concrete through chloride ingress could both be made 15 worse through the interactive effect of pollution (Hunting, 2016; Peng et al., 2017), with these authors 16 suggesting that further research on such interactive effects is required. Despite this, salinity is globally 17 identified as a low risk to the built environment (Escobar et al., 2016; Girjatowicz and Świątek, 2016). 18

- 19 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 ininfrastructure 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).
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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; Elshorbagy 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.

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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*).

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45 5.5.2.3.2 Decision making processes for built infrastructure adaptation

46 Within the built environment, adaptation seeks to moderate or avoid harm, or exploit beneficial opportunities 47 (McCurdy and Travis, 2018). There are numerous politically-oriented-response methodologies (Colin et al., 48 2016; Azhoni et al., 2018; Forzieria et al., 2018) which aim at analysing and reviewing governance policies, 49 legislative frameworks and institutional capacity. Perceptions around infrastructure management are shifting 50 from the purely physical towards socio-technical systems (Chappin and Van der Lei, 2014). Many authors 51 recommend the need for two levels of adaptation, technical and organisational (Antwi-Agyeia et al., 2018; 52 McCurdy and Travis, 2018). Further, concerns have been raised about the ineffectiveness of the 'predict-53 then-act' approach, due to the large uncertainties with climate predictions; thereby raising the need for 54 system flexibility when pursuing climate resilient development pathways (Antwi-Agyeia et al., 2018;

- 55 McCurdy and Travis, 2018), also see Cross-Chapter Box: Risk and Resilience. In pursuit of the latter, built 56 infrastructure adaptation becomes much more nuanced and steeped in human systems analysis (Bles et al.,
- 57 2016; Hauge et al., 2017; Forzieria et al., 2018).

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2 The common decision making tools for prioritisation of adaptation options can depend upon perceptions and prerogatives of those conducting the assessment, resulting in a wide range of tools selected to advance adaptation responses. These tools include (but are not limited to) cost benefit analysis, multi criteria analysis and cost efficiency analysis (Baum, 2012; Hoggart et al., 2014), Source-Pathway-Receptor (SPR) or Source-Pathway-Receptor-Consequences (SPRC) (Monbaliu et al., 2014; Villatoro et al., 2014), and outcome-based 7 decision models focusing on low regret, no regret and win-win outcomes. The use of tools for 8 implementation of adaptation actions results in a variety of approaches, which is discussed next.

9 10 Although there appears to be adequate scientific and technical data available to allow design engineers to 11 assess climate risks in the built environment, methodologies for assessing and incorporating these risks into 12 adaptation planning vary widely in the literature. Too often, critical adaptation decisions remain under the 13 individual prerogative of engineers (Mutombo and Ölcer, 2016c) who typically approach this matter from an 14 engineering silo perspective. The adaptation frameworks assessment (Section 5.5.2.1) concluded with a 15 statement on the importance of both building decision-makers' collective capacity to tackle systemic drivers and pursuing multi-stakeholder adaptation planning processes. That assessment is particularly relevant for 16 17 built infrastructure, to facilitate appropriate adaptation planning responses for these complex systems 18 (Brown et al., 2018).

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20 5.5.2.3.3 Built infrastructure adaptation: approaches and options

21 Given their location at the interface between sea and land, coastlines are vulnerable to climate change and 22 other human impacts. In many countries, Integrated Coastal Management is widely implemented to address 23 issues relating to coastal development and climate change (Rosendo et al., 2018). Strategies for coastal 24 management generally evolve around an action-reaction approach on a post-disaster basis (Gibbs, 2016; 25 Gracia et al., 2018) and it often entails the installation of hard engineering structures such as gabions, 26 revetments, seawalls and breakwaters. Meanwhile, there is increasing evidence of the negative effect that 27 hard engineering has on coastal ecosystems (Sheaves et al., 2016a) and a substantial body of literature is 28 reporting implementation of an ecosystem-based adaptation approach to supporting built infrastructure 29 (Gracia et al., 2018; Rosendo et al., 2018). For examples of the creation and restoration of wetlands, seagrass 30 beds and dune vegetation as effective long term responses in combatting erosion in built infrastructure see 31 (see Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; Rosendo et al., 2018). This approach is 32 considered more sustainable, adaptable, multi-functional, economically viable and has self-repair and 33 recovery ability (robust evidence, good agreement) (Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; 34 Rosendo et al., 2018).

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36 In addressing climate change, there is an increasing focus on the concept of interdependencies in the built 37 environment rather than quantifying specific risk factors (Chappin and Van der Lei, 2014; Antwi-Agyeia et 38 al., 2018). Evidence is emerging that suggests that interdependencies during catastrophic events are a major 39 cause of disruptions in societal processes (Higgins, 2013; Rodrigue et al., 2013). Minor disturbances 40 originating from the environment can cascade, through a series of non-climate change-related interdependent 41 impacts, to create major visible damage in an infrastructure system. As a result, the need for a systematic, 42 cross-sector approach to addressing climate risk in built infrastructure systems is acknowledged (Rahimi et 43 al., 2014; Araos et al., 2016); these may also be referred to as intra- and inter-organizational networks 44 (Azhoni et al., 2018). Recent literature trends reveal a gap in understanding of how climate variables are 45 linked to each other and to the built environment (Chappin and Van der Lei, 2014; Forzieria et al., 2018), 46 and how they can harmonized. Effective and informed decision-making takes into consideration a range of 47 climate change and other human system factors within the built environment, including the complex 48 interrelationship between the built environment and economic activities (Chappin and Van der Lei, 2014; 49 Antwi-Agyeia et al., 2018; Venghaus and Hake, 2018).

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51 Despite the risks posed by climate change in the built environment (Araos et al., 2016), the trade-off between 52 costs and benefits of adaptation solutions (Baum, 2012; Ölçer and Ballini, 2015) is recognised as a major 53 impediment. There is compelling evidence that this trade-off has led to maladaptation through pursuing 54 suboptimal solutions (Azhoni et al., 2018), especially in developing countries with limited resources and 55 competing developmental priorities (Hoggart et al., 2014). As a response, concepts of incremental and 56 transformational adaptation actions are emerging in literature (Felgenhauer, 2015; Bosomworth et al., 2017; 57 Mushtaq, 2018), and as outlined in Cross-Chapter Box 1. Incremental adaptation is defined as actions taken

1 progressively over medium term time frames with the aim of maintaining the functioning and integrity of the 2 system by addressing risks (Aparicio, 2017; Huber-Stearns and Cheng, 2017). However, at some point, a 3 limit to adaptation effectiveness may be reached resulting in system failure; thereby prompting the need for 4 transforming the system. Transformational adaptation changes the fundamental attributes of a system with 5 the aim to increase the capacity of the biophysical, social or economic systems to achieve the desired values 6 (Werbeloff et al., 2016; Bosomworth et al., 2017). The choice between implementing incremental or 7 transformative adaptation actions depends upon local context and climate change impacts, with on-8 adaptation being more ineffective and costly over a long term horizon (Azhoni et al., 2018; Mushtaq, 2018). 9

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.

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18 Understanding, at large scales, the interaction of resilience and resistance in built environment systems is 19 considered an important knowledge gap. This includes studying cyclical patterns of growth, collapse, and 20 renewal of resilience and resistance within the built environment (Pursiainen, 2018). Robust dynamic models 21 for built environment resilience and resistance systems have been strongly advocated in the literature. Such 22 models would enable the various role players to better prepare for extreme disruptions. Actions will require 23 holistic implementation under three main pillars: Technology, Management, and Policy (Mutombo and 24 Ölçer, 2016a). While it was found that technology and management actions are adequate to generate 25 individual benefits at small scale, efforts to implement policy will need to tackle issues of interdependency 26 on a national scale (Mileski et al., 2016), but with a cascade effect at local level. Cross-sectoral climate 27 regulations may be necessary to ensure long term regional economic sustainability. There is an optimum 28 level of regulation, given that excess regulation may potentially have an adverse effect (Chappin and Van 29 der Lei, 2014). Moreover, stakeholders may present discriminative risk exposure to climate events, and this 30 has led to an increasing recognition of the need to explore the cross scale dynamics for effectively 31 implementing these strategies. Coastal governance and policy are dealt with in Section 5.4.4. 32

33 Because of the physical nature of the built environment, the literature has focused more on 'hard' measures 34 (Chappin and Van der Lei, 2014; Taneja et al., 2014; Forzieria et al., 2018), as they are more visible, evident 35 and easily measurable in demonstrating performance than behavioural, institutional and policy measures 36 which are increasingly referred to as adaptive capacity (Carter, 2018; Woodruff, 2018). Neglecting 37 potentially critical 'soft' measures may lead to inappropriate and costly adaptation actions (Barbier, 2015; 38 Perkins et al., 2015). In this light, the concept of soft engineering is rapidly emerging as a valuable 39 alternative within the built environment (de Vriend et al., 2014; Dyson and Yocom, 2015; Loke et al., 2015) 40 to ensure that adaptation solutions sustain multiple services, provide greater economic benefits for society, 41 and resilience to climatic change.

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43 Paradoxically, concern is emerging about the high level of sophistication of built infrastructure. The more 44 technologically sophisticated an infrastructure system is, the more sensitive it could become to disruptions 45 (Rodrigue et al., 2013; Payne et al., 2017), thereby lowering its resilience. There is a gap in the literature on 46 how balancing sophistication with resilience may be feasible, depending on management's appetite and 47 preferences for risk. Contracting insurance against extreme events is regarded as an effective risk transfer 48 strategy for some (Fiksel et al., 2014) while others argue that insurance should never constitute a 49 comprehensive risk treatment due to its limitations in scale coverage and its slow payment process (Müller et 50 al., 2017). A need to enhance insurance solutions to deal with extreme events within the built environment 51 sector is, therefore, becoming increasingly necessary (Bles et al., 2016; Colin et al., 2016; Forzieria et al., 52 2018). 53

54 Given that people naturally act on the basis of perceptions (Hopkins et al., 2016), there is *high agreement* 55 that failure of existing tools that deal with climate uncertainty within the built environment is due to their 56 inability to represent a person's state of knowledge and preferences. This sentiment is strongly reflected in 57 the literature (Antwi-Agyeia et al., 2018; Azhoni et al., 2018), emphasising with *high confidence* the need 1 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

2 3 capacity building. Recent emerging literature strongly advocates for a shift to soft adaptation which implies 4 focussing on issues of psychology, culture, perceptions and education (Hauge et al., 2017; Azhoni et al., 5 2018).

6 7 Meanwhile, in spite of the consensus on the need for adaptation, defining successful adaptation still remains 8 a challenge given that perceptions and risk appetites may substantially differ. Also, in view of the long term 9 nature of climate processes, adaptation is continuous. As new challenges emerge, questions of what actually 10 counts as successful adaptation remain. It is worth noting that despite all the guidelines, research, 11 recommendations and available tools for adaptation, the ultimate call as to whether or not to adapt often 12 remains the prerogative of an individual or manager who will act based on his perceptions, beliefs, culture 13 and willingness. Environmental psychological perspective is currently receiving very little attention, yet it fully dictates the outcome of any climate initiatives.

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16 5.5.2.4 Socio-institutional Adaptation Responses

17 18 This section focusses on community and institutional adaptation responses to climate change, providing an 19 update on AR5. The section begins by giving an overview of types of socio-institutional responses, and 20 assessing their implementation within the tourism, fisheries and aquaculture sectors, before assessing policy, 21 agreement and cooperative adaptation responses. 22

23 5.5.2.4.1 Community-based adaptation

24 Community-based adaptation (CBA) refers here to coastal communities' and non-government organizations' 25 climate change adaptation responses. It includes participatory decision-making and bottom-up approaches to 26 implementing climate change adaptation, community organization for action, communities' engagements 27 with local management authorities, individual level responses to climate change at a homestead scale, 28 livelihood challenges and responses. This assessment is based on reports of climate change impacts, 29 community vulnerability and responses from around the globe, in both developed and developing nations. 30 The literature from the latter focuses more on serious threats to livelihoods (Islam et al., 2013; Ahmed and 31 Diana, 2015b; Kuruppu and Willie, 2015; Lohmann, 2016; Smith et al., 2016a; Esteban et al., 2017; Koya et 32 al., 2017; Shaffril et al., 2017a), whereas the focus from developed nations has a greater focus on threats to 33 infrastructure and the disturbance of community processes. (Serrao-Neumann et al., 2014; Dutra et al., 2015; 34 Colburn et al., 2016; Elrick-Barr et al., 2016; Richards et al., 2016; Jurjonas and Seekamp, 2018). When 35 comparing CBA approaches in communities from developed and developing economy nations, access to 36 resources is the clear defining difference (robust evidence, medium agreement).

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38 Climate change awareness, community mobilization through collective action and the ability to engage with 39 governance structures provided successful examples of community-based adaptation (Dutra et al., 2015; Ray 40 et al., 2017). In Sri Lanka, shrimp farmers demonstrated adaptive and transformative capacity through the 41 understanding of their stressors and their willingness to engage in collective action (Galappaththi et al., 42 2017). This example demonstrates the value of coherence in communities, awareness and collective 43 initiative (high confidence). Other characteristics associated with effective CBA include strong leadership, 44 policy integration (Serrao-Neumann et al., 2014), integrating CBA with other approaches e.g., integrated 45 coastal zone management (Ahmed and Diana, 2015b; Mitchell et al., 2017), acknowledging and addressing 46 tradeoffs (Finkbeiner et al., 2018), enhancing access to credit and switching skills sets (Shaffril et al., 47 2017a). 48

49 Community-based adaptation requires the support of stakeholders with a sufficient level of awarenessabout 50 climate change, supported by policy and government institutions, to improve decision-making processes to 51 avoid climate maladaptation (Tapsuwan and Rongrongmuang, 2015). For example, the climate change 52 adaptation response of participants in a Southeast Asia dive industry was reported to be based on 53 misconceptions about climate change and personal observations (Tapsuwan and Rongrongmuang, 2015). In 54 another case, a community's engagement with local authorities in a community mangrove forest 55 management project resulted in the deterioration of the mangrove forest due to poor management practices, 56 with a corresponding increase in community vulnerability as coastal protection services and economic

57 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 exampes 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 Artic (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).

11 12 Besides inclusion in adaptation programs, non-climate stressors will need to be addressed, including rural 13 poverty levels (Jurjonas and Seekamp, 2015), job opportunities and aging communities. In Canada, this 14 includes the promotion of 'age friendliness' programs (Krawchenko et al., 2016). In East Africa, self-15 organization through negotiated social practices were deemed to be suitable, context-specific local responses 16 to climate change (Kithiia, 2015). There is substantial evidence underlining the importance of social capital 17 for increasing resilience in communities (Petzold and Ratter, 2015; Salik et al., 2015; Chen and Ganapin; 18 Sakakibara, 2017; Triyanti et al., 2017). The substantial body of literature assessed suggests that community-19 based adaptation will continue to be a very important component of coastal adaptation planning and 20 implementation going forward (high confidence).

22 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).

27 The incorporation of Marine Protected Areas (MPAs) into coastal and oceanic zone management is 28 considered an important strategy in adapting to climate change (Hopkins et al., 2016) (medium agreement, 29 medium evidence), but management of such zones is, itself likely to be impacted by climate change (Johnson 30 et al., 2018). Management approaches need to accommodate a 'shifting baseline' as climate change becomes 31 the new normal, with novel dispersal movements of highly-mobile species (Elliott et al., 2015). Where such 32 shifts occur across national boundaries, management success will require transnational agreements (Gormley 33 et al., 2015). Transboundary ICM is *likely* to become more important with ecosystem shifts due to climate 34 change (medium evidence, high agreement).

36 An advantage of ICM is that it helps manage the interactions between multiple climate and non-climatic 37 drivers of coastal ecosystems and sectors. For example, sea level rise is interacting with non-climate change 38 impacts including illegal sand mining, inappropriate development and habitat destruction (e.g., mangroves), 39 impacting upon tourism (Rangel-Buitrago et al., 2015) (Section 5.4.2.2.3). Adaptation responses to coastal 40 erosion is thus marine spatial planning and coastal zone management informed by scientific, local 41 knowledge and indigenous knowledge, and with sustainable funding mechanisms and support networks for 42 decision-making. Overall, participatory spatial planning processes and ecosystem-based adaptation 43 responses could improve tourism adaptation efforts (medium confidence).

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45 Trade-offs associated with ICM and MPAs that may result in maladaptive outcomes; such that consideration 46 of such trade-offs is important in effective climate-adaptive ICM and MPAs. For example, in Brazil, poorly 47 managed MPAs did not provide increased benefits for small-scale fishers adjacent to the MPAs, with a high 48 reliance upon access to mangroves whereby the restriction in access without increased benefit was 49 considered maladaptive (Luiz Francisco Ditzel et al., 2016). Incorporating stakeholder participation 50 processes as a form of bottom-up co-management could bring local knowledge and indigenous knowledge 51 (Serrao-Neumann et al., 2013) to reduce the risk of mal-adaptation, and increase buy-in for implementation 52 (Section 5.5.2.4.1) (medium evidence, high agreement). Some institutions with a strong top-down 53 institutional framework may, however be resistant to changing to this approach (Abelshausen et al., 2015). 54

55 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.

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5 Management to reduce existing non-climatic stressors was suggested to be an effective climate risk reduction 6 adaptation measures for fisheries and aquaculture with large co-benefits (Gattuso et al. 2018). Specifically, 7 as overfishing is one of the most important non-climatic drivers affecting the sustainability of fisheries, 8 eliminating overfishing could help rebuild fish stocks, reduce ecosystem impacts and increase the adaptive 9 capacity of fisheries. In an analysis, 60% of assessed species are projected to be at high risk from both 10 overfishing and climate change by 2050 (RCP8.5), particularly tropical and sub-tropical species (Cheung et 11 al., 2018b). Pursuing sustainable fisheries practices under a low emissions scenario would decrease risk by 12 63%, highlighting the importance of effective fisheries management (Gaines et al., 2018). However, as 13 eliminating overfishing would require reducing current levels of fishing capacity, there are uncertainties 14 associated with whether the short-term reduction in catches may impact livelihood and food security of local 15 communities (Hobday et al., 2015b; Dey et al., 2016; Rosegrant et al., 2016; Campbell, 2017; Finkbeiner et 16 al., 2018). Despite the *high agreement* on the significant effectiveness of eliminating overfishing in climate 17 change adaptation for fisheries, available evidence remains aspirational rather than presenting evidence of successful adaptation outcomes.

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> 20 Range shift of fish stocks under ocean warming (Section 5.2.3) will alter distribution of fish stocks across 21 political boundaries, thus demand for consideration of transboundary fisheries management in adaptation for 22 fisheries will increase. Re-distribution of transboundary fish stocks between countries will destabilize 23 existing international fisheries agreements and increase the risk of international conflicts (Section 5.4.2). 24 Adaptation of international fisheries management to reduce such risk would involve improving planning for 25 cooperative management between countries informed by reliable projections of species shifts and associated 26 uncertainties (Pinsky et al., 2018). Regular monitoring and performance reviews of fisheries could also 27 facilitate the management adjustment towards considering species shifts. Other international fisheries 28 arrangements, such as flexible fishing effort allocation with opportunities for countries to increase 29 exploitation of resources by providing payment to other countries, may also improve the robustness of 30 fisheries management (Miller et al., 2013). Success from existing arrangements to adapt to climate 31 variability suggests that these measures can be effective. However, the ability to fully adapt to climate 32 change beyond historical variability is uncertain. Thus, although range shift poses a significant challenge to 33 transboundary fisheries management, proactive planning and adjustment of fisheries management 34 arrangements, informed by scientific projections, could help improve their adaptive capacity (high 35 confidence). 36

> 37 Fisheries management strategies depend heavily upon the accuracy of data collected in respect of predicting 38 environmental conditions, over time scales from months to decades (Dunstan et al., 2017), controlling for 39 aspects of fish population dynamics like recruitment success and fish movement. Seasonal to decadal climate 40 prediction systems allow for skillful predictions of climate variables relevant to fisheries management 41 strategies (Hobday et al., 2016b; Payne et al., 2017; Tommasi et al., 2017). A planning response in the U.S. 42 National Marine Fisheries Service has been to conduct vulnerability assessments, improve monitoring of 43 ecosystem indicators and evaluate management strategies (Busch et al., 2016). Himes-Cornell and Kasperski 44 (2015) provided a framework of indicators for Alaskan fishing communities' vulnerability to climate change 45 that includes exposure to climate change bio-physical effects, resource dependency and adaptive capacity. 46 Community response as a part of climate change adaptation for local fisheries is an important element in 47 assessing adaptive capacity (medium evidence, good agreement), and this response is affected by socio-48 economic factors like alternative income, gender and religion, which collectively shape a community's 49 adaptation response (Arroyo Mina et al., 2016). In West Africa, the industrial fishery response to climate 50 change-induced reduction in landings was the expansion of fishing grounds, which increased operational 51 costs (Belhabib et al., 2016), although this response was not available to artisanal and local fishing 52 communities, who are considered highly vulnerable (Kais and Islam, 2017). Other adaptation responses 53 included improved fishing gear and technology, use of Fish Aggregating Devices and uptake of insurance 54 products (Zougmoré et al., 2016).

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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

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expert knowledge of climate change threats to fish habitats, stocks and landings, the necessity of 1 2 transdisciplinary collaboration and stakeholder participation, broadening the range of scope of fisheries 3 systems and increased commitment of resources and capacity. This was considered in the context of the 4 ability of developing countries to manage ecosystems and existing over-exploitation of resources. More 5 research is required on socio-ecological responses to climate change-impacts on fishery communities. This 6 includes aspects like risk reduction, adaptive capacity through knowledge attainment and social networks, 7 developing alternative skills and participatory approaches to decision-making (Dubey et al., 2017; Shaffril et 8 al., 2017c; Shaffril et al., 2017b; Finkbeiner et al., 2018). 9

10 Less still is known about how climate change will impact upon the deep oceans and fisheries therein, 11 especially those resources that are not yet being harvested. Johnson et al. (2018) concluded that in a 20 year 12 to 50 year timeframe, virtually all North Atlantic deep-water and open ocean area-based management tools 13 will *likely* be affected. They concluded that more precise and detailed oceanographic data are needed to 14 determine possible refugia, and more research on adaptation and resilience in the deep sea is needed to 15 predict ecosystem response times. Until such analyses can be made, additional human impacts could be 16 avoided using a more precautionary approach, potentially setting aside more extensive conservation areas 17 and strictly limiting human uses and/or adopting high protection thresholds (Johnson et al., 2018). 18

19 As with fisheries, community- and ecosystem-based adaptation responses, within an integrated coastal zone 20 management framework, is considered necessary to cope with anticipated challenges for aquaculture 21 (Ahmed and Diana, 2015a). Where in-situ adaptation is not possible, translocation and polyculture (Ahmed 22 and Diana, 2015b; Bunting et al., 2017) have been suggested as appropriate responses (robust evidence, 23 good agreement), but this would suit commercial rather than subsistence interests. Policy, economic, 24 knowledge and other types of support are required to build socio-ecological resilience of vulnerable coastal 25 communities (Harkes et al., 2015; Bunting et al., 2017; Rodríguez-Rodríguez and Bande Ramudo, 2017), 26 which requires a deep understanding of the nature of stressors and a commitment for collective action 27 (Galappaththi et al., 2017). Climate resilient pathway development (CRPD, see Cross-Chapter Box 1) is 28 considered a useful framework for Sri Lankan shrimp aquaculture (Harkes et al., 2015). Another successful 29 example of aquaculture adaptation is the employment of near-real time monitoring technology to track the 30 carbonate chemistry in water to reduce bioerosion in shellfish from acidification (Barton et al., 2015b; 31 Cooley et al., 2016).

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33 An example of an engineering-based adaptation option that has the potential to increase productivity of 34 seaweed aquaculture under climate change is artificial upwelling (Jiao et al., 2014a). Nutrient availability is 35 the major limiting factor for open ocean primary productivity (see section 5.2.2.2). Upwelling can bring up 36 high-nutrient deep water to the euphotic zone and enhance primary production (Mackenzie et al., 2002; 37 Farias et al., 2015). However, upwelling can also bring high dissolved inorganic carbon (DIC) and low 38 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 39 40 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 41 42 zone. The aim of the adaptation intervention is then to control the degree of such upwelling to meet the 43 demands of nutrients and DIC by the cultured seaweed. This could theoretically reduce acidification and 44 hypoxia in the upwelling area as DIC is mostly used by the seaweed while oxygen is produced by 45 photosynthesis. Such artificial upwelling operations may also gradually release the 'bomb' of rich nutrients 46 and hypoxia in the bottom water, which could breakout when storms take place otherwise (Daneri et al., 47 2012). In situ monitoring of physical and chemical variables such as temperature, nutrients, oxygen, DIC 48 and pH etc. are used for adjustment of the upwelling extent for adaptative management. This type of 49 interventions would require high level of scientific knowledge, capital investment and potential unexpected 50 ecological consequences. Thus, even if such intervention is proven to be practical at large-scale, there may 51 be substantial limits to its suitability for different upwelling areas in the world. 52

53 5.5.2.4.4 Institutional responses

54 Coastal and oceanic adaptation responses are greatly complicated by the presence of competing interests 55 (either between user-groups, communities or nations), where considerations other than climate change need

56 to be incorporated into cooperation agreements and policy (Wong et al., 2014). Climate change adaptation

57 planning is required in local land-use planning and other processes (Sano et al., 2015; Elsharouny, 2016).

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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*).

- 9 In coastal communities, there is often consensus on the importance of cooperation in tackling climate change 10 (Elrick-Barr et al., 2016), but adaptation progress is hampered by competing economic interests and 11 worldviews (Hamilton and Safford, 2015) as well as lack of knowledge (Nanlohy et al., 2015), whilst factors 12 like home ownership and a general future planning ability are important (Elrick-Barr et al., 2016). Another 13 important factor is the level of trust communities have in their local governance processes, causing 14 (Massuanganhe et al., 2015; Wynveen and Sutton, 2015) to conclude that relationship-building actions are 15 an important part of an adaptation response. In a survey to understand community perception of financing 16 adaptation actions, British coastal communities displayed a high level of resistance to the implementation of 17 a coastal defense tax (Alderson-Day et al., 2015). Local collective action, using legal arguments around the 18 right to protection from environmental change, may be a useful mechanism of securing state support for 19 adaptation (Karlsson and Hovelsrud, 2015). 20
- 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
 Schopmeyer, 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).
- 35 There remains a paucity of evaluation literature pertaining to the effectiveness of adaptation efforts. In 36 evaluations of socio-institutional approaches, the importance of partnerships have been emphasized; for 37 example, Rutherford et al. (2016) reported that the length of history of collaboration in Europe was 38 positively associated with adaptation planning progress, In the UK, better access to information from 39 government investment has resulted in better adaptation planning and local government capacity, but not in 40 the implementation of adaptation plans (Porter et al., 2015). This was considered to be due to budget cuts 41 while best adaptation progress has been made in local governments addressing resilience to extreme weather 42 events, and this is consistent with other reports of political focus on more immediate pressures (Barbier, 43 2015; Gray et al., in press) (robust evidence, high agreement).
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45 *5.5.2.4.5* The role of education and local knowledge in adapting to climate change.

46 It is essential that people are informed that local and small scale behaviours accummulate very quickly and 47 contribute to the global scale phenomenon of climate change. People need very clear messages and 48 consistent messages that reiterate that new behaviours are very much needed at the individual, household and 49 local business scale as well as when individuals travel and participate in tourism opportunities. People 40 urgently need to be educated about cumulative impacts and feel empowered and inspired to take positive 41 actions. The growing popularity of outreach initiatives and scientific communication to the general audience 42 can be a critical instrument to increase awareness and concern (*medium confidence*).

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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

1 (Wynveen et al., 2015) Research suggests that a lack of education causes misconceptions about the scale of 2 climate change (temporally and spatially), and the causes of climate change (Tapsuwan and 3 Rongrongmuang, 2015). For example, ocean acidification may be an excellent educational tool to address 4 climate change issues (Fauville et al., 2011) because the acidification process can be clearly explained. A 5 critical element to reducing vulnerability to climate change is to educate people that they are core to the 6 Earth system and have a huge influence on the balance of the system. An important service of marine 7 biodiversity and ecosystems is to support such education (Section 5.4.1). Thus, education can play a pivotal 8 role in how climate change is perceived and experienced and marine biodiversity and ecosystems plays an 9 important role in this.

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11 A lack of education clearly influences the resilience and/or vulnerability of communities to climate change 12 (Qin et al., 2017). This occurs in two ways; through increasing sensitivity to climate change, and through 13 decreasing adaptive capacity. Sensitivity to climate change can be increased through increasing the 14 dependency that people have on a single natural resource. People in rural coastal areas have less education 15 and thus less options for the future, increasing their dependency on the natural resource (low confidence). 16 The greater the dependency, the greater the sensitivity to climate change. Unless resource-dependent people 17 have a correspondingly high level of adaptive capacity to diminish any social and economic impacts 18 associated with their dependency, they remain highly vulnerable (Marshall et al., 2017; Marshall et al., 19 2018). Adaptive capacity can be increased through education, where people are able to convert their newly 20 founded knowledge and information into successful adaptive strategies (Gladstone, 2009). Marshall et al. 21 (2013) show that climate change awareness is associated with enhanced adaptive capacity. Additionally, 22 education can increase the likelihood that existing technologies which aid the community to better cope and 23 adapt to climate change. For example, without education, social and behavioral patterns are more *likely* to 24 influence the effectiveness and use of warning systems, regardless of the technology involved (Pescaroli and 25 Magni, 2015). Education needs to be combined with other institutional support for the full benefits to emerge 26 and for education to be meaningful. For example, indigenous communities can be very aware and educated 27 about climate change, but can be unempowered to make decisions that would protect their communities 28 without the appropriate institutional support. 29

30 Local knowledge of environments that are under threat from climate change can both act to promote 31 adaptation and act as a barrier to adaptation. Local knowledge is a measure of the human capital that has 32 been developed by individuals to better understand their local environment (Andrachuk and Armitage, 33 2015). For example, some commercial fishers or marine-based tourism operators may have spent a 34 significant proportion of their career towards developing a deep familiarity with their local fishing grounds 35 or marine environment for the purposes of economic gain. Their local knowledge makes them particularly 36 dependent upon that area, and they are more likely to be successful within that local area when conditions are 37 stable (Marshall et al., 2007). However, this dependency makes them particularly sensitive to climate events 38 such as bleaching or a cyclone (Marshall et al., 2013). Fishers and tourism operators with higher local 39 knowledge will be more sensitive to change than those that have invested less in developing their local 40 knowledge. The local knowledge that has been developed may be so important to fishers or tourism 41 operators that they are *likely* to be particularly reluctant to move away from their familiar environment and 42 develop new local knowledge elsewhere. In these instances, local knowledge can act as a barrier to 43 adaptation (Metcalf et al., 2015).

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45 There is evidence, however, to suggest that fishers and tourism operators with high levels of local knowledge 46 (whilst sensitive) are not necessarily vulnerable to climate change if they have correspondingly high level of 47 adaptive capacity (Marshall et al., 2013). Resource users such as commercial fishers with high levels of local 48 knowledge are also *likely* to have already noticed subtle changes within the environment, and recognise the 49 need to change and adapt. This is because they recognise 'feedbacks' within the system (Adger et al., 2008). 50 In these instances, fishers with higher local knowledge are more *likely* to demonstrate a higher adaptive 51 capacity than fishers with lower local knowledge, and will more *likely* progress towards developing new 52 strategies to combat the impacts of climate change (Kittinger et al., 2012). In these instances, local 53 knowledge acts to promote adaptation (medium confidence). 54

5.5.2.5 Costs and Limits for Coastal Climate Change Adaptation

1 Chapter 5 of AR5 assessed costs of adaptation to sea level rise with estimates varying between tens to 2 hundreds of billions of US dollars, dependent upon the sea level rise scenario. Recent estimates suggest that 3 annual costs of coastal adaptation and adaptation of infrastructure to changes in rainfall and temperature 4 range from US\$400 million to US\$1.2 billion by 2040 (World Bank, 2017). Average annual direct loss due 5 to cyclone-associated flooding in 15 South Pacific countries was estimated at up to US\$ 80 million (2009 6 prices), with 60% of the damage resulting from loss of residential buildings, 30% from loss of cash crops 7 and 10% from damage to infrastructure (PCRAFI, 2011). A key research gap identified was comparison with 8 natural based solutions, with consideration of the co-benefits the latter option provides. In the recent 9 literature, estimating adaptation costs is still challenging because of wide ranging regional responses and 10 uncertainty (Dittrich et al., 2016). Despite these challenges, the protection from flooding and frequent storms 11 that coral reefs provide has been quantified (Beck et al., 2018). They estimated that without reefs, damage 12 from flooding and costs from frequent storms would double and triple respectively, while countries from 13 Southeast/ East Asia and Central America could each save in excess of US\$ 400 million through good reef 14 management. 15

16 In West African fisheries, loss of coastal ecosystems and productivity are estimated to require 5-10% of 17 countries' Gross Domestic Product in adaptation costs (Zougmoré et al., 2016). Similarly, for Pacific Islands 18 and Coastal Territories, fisheries adaptation will require significant investment from local governments and 19 the private sector (Rosegrant et al., 2016), with adaptation costs considered beyond the means of most of 20 these countries (Campbell, 2017). In Small Island Developing States, tourism could provide the funding for 21 climate change adaptation, but concerns with creating investment barriers, assumptions around cost-22 effectiveness and consumer driven demand remain barriers (Hess and Kelman, 2017). Marine Protected 23 Areas, with multiple co-benefits, are considered a cost-effective strategy (Byrne et al., 2015), but, as was 24 reported in AR5, further research evaluating natural infrastructure is required (Roberts et al., 2017).

25 26 There are still a lot of challenges in conducting economic assessments for built infrastructure adaptation due 27 to the associated, complicated uncertainties such as accuracy of climate projections, limited information 28 regarding paths for future economic growth and adaptation technologies. A global estimation of coastal 29 flood damage and adaptation costs due to sea level rise by the end of this century projected that 0.2–4.6% of 30 the global population would be flooded annually with expected annual losses of 0.3–9.3% of global Gross 31 Domestic Product (Hinkel et al., 2014). Annual investment and maintenance costs of protecting coasts were 32 projected to be US\$ 12 billion to US\$ 71 billion, which was considered significantly less than damage costs 33 in the absence of such action. In an analysis of twelve Pacific island countries, 57% of assessed built 34 infrastructure was located within 500m of coastlines, requiring a replacement value of US\$ 21.9 billion. 35 Substantial coastal adaptation costs (and international financing) are required in these countries.

- 36 37 An extreme event that exceeds an infrastructure's maximum tolerance level can result in failure of the 38 infrastructure. Changes in climatic patterns can slowly affect the service life and performance of 39 infrastructure. Unfortunately, there is little literature bridging climate change and infrastructure deterioration 40 (Ha et al., 2017). The need to understand the economy-wide aspects of climate change and its adaptation has 41 been widely recognised in literature (Fisher-Vanden et al., 2013; Fankhauser, 2017) and there is high 42 agreement on the need to shift towards climate-resilient economies (Robinson et al., 2012; Fisher-Vanden et 43 al., 2013). In addition, given the economy-wide effects captured implicitly in econometric models, various 44 authors have highlighted the relevance of systemic adaptation models (Conway et al., 2015; Millner and 45 Dietz, 2015; Fankhauser, 2017) despite difficulties faced in structuring all variables onto input-output tables 46 or in factoring social accounting matrices.
- 47 48 Due to interdependency of economic systems, it is very likely that indirect risks, and therefore adaptation 49 needs, are dominant contributing factors to direct effects of climate change for some sectors. Meanwhile, 50 understanding adaptation deficit is increasingly receiving attention. Factors such as literacy, income, income 51 distribution, institutional quality, health spending, and access to finance are linked to adaptive capacity, or 52 the ability to respond to climate risk (Fankhauser and McDermott, 2014; Carleton and Hsiang, 2016; 53 Fankhauser, 2017). There are strong interlinkages between adaptation and economic development, such that 54 future vulnerability to climate change will drive decisions on infrastructure adaptation investments, 55 industrial strategy and urban planning (Fankhauser, 2017). For effective adaptation, knowledge, planning, 56 coordination, and foresight are required. This necessitates multifaceted decision-making which makes 57 adaptation an interesting economic problem.

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2 Within the literature, there are a broad range of reported barriers and limits to climate change adaptation for 3 each of the three themes discussed in Section 5.5.2. For shoreline stabilisation, there is a key research need 4 to improve understanding of limitations in built-infrastructure, beach nourishment and nature-based 5 adaptation responses, especially in respect of cost effectiveness and resilience. (Mackey and Ware, 2018). 6 Whilst using ecosystems is considered to be a cost-effective and sustainable approach for coastal erosion 7 management (Adriana Gracia et al., 2018), limitations have been acknowledged with coastal ecosystem 8 based adaptation (Sussams et al., 2015). Limitations include the space that ecosystems require (which may 9 not be available in an urban environment), non-climate change co-drivers like human impacts that need to be 10 addressed as part of the adaptation response (Ahmed et al., 2017; Peña-Alonso et al., 2017; Triyanti et al., 11 2017), the severity of climate impact may exceed the adaptive capacity of the ecosystem, for example the 12 type of rainfall being experienced affects the efficacy of storm water associated green infrastructure (Joyce 13 et al., 2017) or acidification affecting coral reef viability, and, potentially slower recovery rates within 14 ecosystems than the recurrence of climate impacts, including the forcing of ecosystems into novel dynamic 15 equilibrium states that may not deliver the level of service required. Given these limitations, it is important 16 that EBA decision-makers are better informed about its context and site specific application (Sutton-Grier et 17 al., 2015) to avoid maladaptation (high confidence). A barrier to achieving this is the fragmented nature of 18 EbA implementation across disciplinary approaches (Brink et al., 2016). Other barriers include governance 19 structures, including incorporating longer-term scales, how to measure effectiveness, developing appropriate 20 financial mechanisms and dealing with uncertainty (Ojea, 2015). 21

22 Built infrastructure may negatively impact upon adjacent ecosystems as coastal processes can be disrupted, 23 and result in unintended consequences. Examples include New Orleans during Hurricane Katrina and Sendai 24 during the 2010 Tokoku earthquake, where the need for a 'design storm' in the engineering design process 25 was considered a critical adaptation limit (Mackey and Ware, 2018). This is particularly the case when 26 design practice does not consider climate change impacts in all design storm event characteristics (Watt and 27 Marsalek, 2013). Understanding when and how to implement a suite of built and natural infrastructure 28 adaptation options is considered essential to achieving an optimal adaptation response (Depietri and 29 McPhearson, 2017).

31 For artisanal fisheries, a range of physical and socio-institutional limits and barriers to adaptation have been 32 reported, including increasing occurrence and severity of storms, technologically poor boats and fishing 33 equipment and lack of access to credit and markets, among others (Islam et al., 2013). Conflicting interests 34 and values of stakeholders in great barrier reef-associated industries were deemed to limit adaptation 35 response (Evans et al., 2016), whereas the path-dependent nature of organisations and resistance to change, 36 reported in Australia, were considered socio-institutional barriers, (Evans et al., 2016). Governance barriers 37 to adaptation in Vancouver include inadequate collaboration, political leadership, public awareness, finance 38 and capacity (Oulahen et al., 2018). There is a key research need to understand how such limits and barriers 39 interact to suppress adaptation response.

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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).

48 Defining limits to social adaptation through a stakeholder-focussed, risk-based approach can help inform 49 debate about societal response to climate change (Dow et al., 2013). In some communities, climate change 50 may not be prioritised. For example, Fischer (2018) presented evidence in six Oregon, USA communities 51 where response to potential climate change impacts like disasters are not prioritised in the face of chronic, 52 daily challenges to secure livelihoods. In coastal Jakarta, Indonesia, a survey was conducted in poor 53 communities about subsidence and sea level rise risks. While local inhabitants appear to be aware of the 54 hazards they face, many seem to underestimate their severity, possibly due to a high frequency of exposure 55 in the recent past (Esteban et al., 2017). In a world with competing risks and urgent priorities, some local 56 inhabitants appear to be unable to avoid, or are willing to carry, the risk associated with a climate impact in 57 order to meet other, more pressing needs. This example reflects the reality of many poor, informal settlement dwellers in coastal areas around the world (*high confidence*). Other barriers to effective adaptation action
include limited knowledge, weak cognitive ability, inappropriate coping strategies, level of dependence upon
natural resources and exposure to hazards (sensitivity), past experience and weak community networks
(Islam et al., 2013; Nanlohy et al., 2015; Lohmann, 2016; Koya et al., 2017; Senapati and Gupta, 2017;
Cumiskey et al., 2018).

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.

13 5.5.2.6 Summary 14

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15 There has been a substantial amount of literature focused on coastal and oceanic adaptation since AR5. 16 Socio-institutional adaptation responses are the more numerous of the three types of adaptation responses 17 assessed in this chapter. There is broad agreement that hard engineering responses are optimally supported 18 by ecosystem-based adaptation approaches, and both approaches should be augmented by socio-institutional 19 approaches for adaptation (high confidence) (Nicholls et al., 2015; Peirson et al., 2015; Sánchez-Arcilla et 20 al., 2016; van der Nat et al., 2016; Francesch-Huidobro et al., 2017; Khamis et al., 2017). In planning 21 adaptation responses, awareness-raising and stakeholder engagement processes are important for buy-in and 22 ownership of responses (robust evidence, high agreement) as is institutional capacity within local 23 government organizations, whose importance in coastal adaptation initiatives has been emphasized in the 24 recent literature (robust evidence, high agreement). With all three types of adaptation, it is clear that getting 25 basic good governance and effective implementation of service delivery processes correct is a prerequisite 26 for successful adaptation planning and response. 27

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.

39 Existing ocean governance structure for the ocean is facing or will face multi-dimensional challenges 40 because of the climate change (Galaz et al., 2012). Considering climate change issues in the global and 41 regional ocean, environmental and fisheries governance structure is needed. Current international 42 governance regime yet to adequately address the issues of ocean warming, acidification and deoxygenation. 43 As mentioned by Oral, legal framework for ocean acidification 'appears to fall between the cracks of the two 44 principal regimes for climate change, the UNFCCC and ocean governance under UNCLOS' (Oral, 2018). 45 While adoption of these legal and governance regimes, climate change or carbon dioxide emissions related 46 changes of the ocean, particularly, the interaction of physical, chemical, biological properties of the 47 ecosystems with risks and vulnerabilities of the dependent communities was either not considered or did not 48 get proper attention. There is a need for a comprehensive assessment of exiting international legal regimes 49 for above-mentioned areas considering the changing ocean and its ecosystems and their impact on dependent 50 communities.

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52 5.5.3.2 Climate Change Governance Regime53

54 International climate change governance regime is naturally the most relevant global governance apparatus 55 for combating ocean acidification and ocean warming. However, how far current climate change legal and 56 policy framework is adequate for dealing with the issue of the ocean acidification is questionable (Stephens, 57 2015). Existing climate change related legal instruments do not adequately address the challenges the ocean
1 and coastal areas will face. Nevertheless, their overall goal for reduction of emissions will be positive for 2 mitigation of climate change impact on the ocean and coastal areas (Galland et al., 2012; Gallo et al., 2017). 3 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 4 5 legal instruments are mainly for combating global warming with a clear atmospheric focus and ocean did not 6 get proper attention (Galland et al., 2012). In 2015, parties to UNFCCC adopted the Paris Agreement, 7 according to which state parties will address climate change mitigation and adaptation as per their Nationally 8 Determined Contributions (NDCs). Unlike Kyoto Protocol, the ocean is explicitly mentioned in the Paris 9 Agreement's preamble and there may be implications from the Paris Agreement for ocean (Magnan et al., 10 2016). However, a study on 161 NDCs submitted where national pledges for climate change mitigation and 11 adaptation show that there are 'gaps between scientific [understanding] and government attention, including 12 on ocean deoxygenation, which is barely mentioned' (Gallo et al., 2017). Therefore, greater emphasis to the 13 ocean related issues in the climate change governance regime is needed.

15 5.5.3.2.1 Environmental governance of ocean

16 The United Nations Convention on the Law of the Sea (UNCLOS) elaborates the jurisdiction, duties and 17 obligation of coastal, flag and port states in different maritime zones including internal waters, territorial sea, 18 exclusive economic zone, continental shelf, and high seas. Two aspects of UNCLOS are specifically 19 important in the context of this assessment: the provisions relating to prevention of marine pollution and 20 provisions relating to the conservation and sustainable utilization of marine living resources. UNCLOS 21 imposes obligation for the conservation and sustainable utilization of marine living resources within national 22 jurisdiction (UNCLOS 1982, Articles 56, 6, 62, 63, 64, 65 and 67). UNCLOS imposes obligations on state 23 parties to take action for combating six main sources of ocean pollution including land-based and coastal 24 activities, continental shelf drilling, seabed mining, ocean dumping, vessel-source pollution, and pollution 25 from or through the atmosphere. It has been suggested that tools and techniques in UNCLOS may need 26 adjustment in response to the emerging challenges created by climate change for ocean (Redgwell, 2012). 27 However, success of the umbrella regulatory framework of UNCLOS is heavily depending on further 28 development, modification and implementation of detail regulations by relevant international, regional and 29 national institutions (Karim, 2015).

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31 One of most relevant legal instruments in this regard is the Convention on the Prevention of Marine 32 Pollution by Dumping of Wastes and Other Matter, 1972 (London Convention). In 1996, the London 33 Protocol was adopted to modernize and eventually replace this convention. 1996 London Protocol prohibits 34 dumping of any wastes or other matter to the sea unless it is listed in the Annex 1 of protocol. Through an 35 amendment in 2006, 'carbon dioxide streams from carbon dioxide capture processes for sequestration' has 36 been included in the Annex 1 of the Protocol. These amendments then followed by further two amendments 37 on sharing transboundary sub-seabed geological formations for sequestration projects and ocean fertilization 38 and other marine geoengineering. One of these new amendment prohibits ocean fertilization except for 39 research purposes (Dixon et al., 2014). Nevertheless, there are concern for these activities. International 40 Convention for the Prevention of Pollution from Ships, 1973 (MARPOL) is also relevant for climate change 41 and GHG emissions. In 2011, this convention was amended to include some technical and operational 42 measures for reduction of GHG emissions from ships. However, effectiveness of these provisions is 43 questionable (Karim, 2015).

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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).

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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 Krivickaite, 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

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4 5 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

6 The impact of climate change on marine fisheries will be very significant and it may have an adverse impact 7 on food security, livelihoods and national development in many coastal countries, and the least developed 8 countries will me particularly vulnerable (Blasiak et al., 2017). As outlined in this report, evidence shows 9 that warming of the oceans has emerged and can be attributed to human influence (Section 5.2.2), that the 10 oceans are acidifying, the oceans are losing oxygen particular in the equatorial and eastern boundaries 11 (Section 5.2.2) and these effects are projected to affect fish distributions and biomass with consequent flow 12 on effects to human communities (Section 5.3 and 5.4). Changing ocean may also create serious problem in 13 the governance of regional fisheries agreements (Brandt and Kronbak, 2010). Regional fisheries 14 management systems also need to address the emerging challenges (Brooks et al., 2013). A study on seven 15 international case studies shows that fisheries governance system will face further uncertainties because of 16 climate change necessitating adoption of progressive policy initiatives introducing flexible management 17 structure, capacity development, sustainable exploitation system and altretamine arrangement for livelihood 18 security of dependent communities (McIlgorm et al., 2010). The same report also indicates that the 19 adaptation to climate change will be more difficult if there is an absence of good existing fisheries 20 governance structure (McIlgorm et al., 2010). A review published in 2017, identifies nine major ecological 21 and socio-ecological criteria for climate resilient regulatory approach including 'sustainable & age-diverse target populations', 'conserving biodiversity & habitats', 'managing existing stressor', 'adaptive 22 23 management', 'diversified livelihoods', 'promoting longterm stewardship' 'multi-level governance', 'Fisher 24 mobility' and 'community based management' (Ojea et al., 2017). However, the effectiveness of these will 25 diverge depending on the characteristics of regulatory instruments and other factors (Ojea et al., 2017). A 26 climate-informed ecosystem-based fisheries governance approach has been suggested for climate change 27 resilience of marine fisheries in the developing world (Heenan et al., 2015a). In producing a strategy and 28 roadmap for coastal fisheries, within a participatory workshop in the Pacific concluded that robust and 29 effective management policy, legislation and planning will be required for coastal fisheries based on 30 flexibility and scientific understanding to support management decisions (Gourlie et al., 2017). An 31 integrative science-based approach has been suggested as 'a vehicle (1) to examine policy options with 32 respect to their robustness to uncertainty, particularly to climate-related regime shifts and (2) to allow better 33 assessments of behavioral responses of fish, humans and institutions.' (Miller et al., 2010). Badjeck et al. 34 (2010) indemnified the following is needed in the context of challenges in global fisheries governance: '(1) 35 management approaches and policies that build the livelihood asset base, reducing vulnerability to multiple 36 stressors, including climate change; (2) an understanding of current response mechanisms to climate 37 variability and other shocks in order to inform planned adaptation; (3) a recognition of the opportunities that 38 climate change could bring to the sector; (4) adaptive strategies designed with a multi-sector perspective; 39 and (5) a recognition of fisheries potential contribution to mitigation efforts.' (Badjeck et al., 2010). 40

41 The above discussion shows that greater emphasis and comprehensive reform within fisheries governance in 42 all levels is needed for overcoming climate change related challenges for fisheries-based livelihood and 43 conservation of global fisheries resources. Apart from the above sectoral areas of governance, governance of 44 areas beyond national jurisdiction in the context of climate change is another major challenge (Levin and Le 45 Bris, 2015). According to Warner 'collaborative structures and mechanisms for environmental assessment in 46 ABNJ are still fragmentary and underdeveloped, with limited sectoral involvement. Establishing these 47 governance structures in ABNJ involves multiple stakeholders, including states, global and regional 48 organizations, marine industries, and nongovernmental organizations focused on conservation of marine 49 biodiversity.' (Warner, 2018). Currently a negotiation is going on in the United Nations for a new agreement 50 under UNCLOS for marine biodiversity of areas beyond national jurisdiction. Climate change related 51 challenges need to be addressed in the future legal framework for areas beyond national jurisdiction.

5.5.3.3 Private Sector Stakeholders

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55 The success of ocean-based climate initiatives, and marine conservation more generally, is constrained by 56 disjointed and inadequate ocean governance, conflicts between stakeholders, limited enforcement capacity 57 and inadequate finance (Gill et al., 2017). Many recent studies have confirmed that funding for protected

1 areas and biodiversity conservation must increase significantly to achieve targets set at national or 2 international levels (e.g., Aichi targets). A recent global top-down assessment conducted by the Convention 3 on Biological Diversity High Level Panel estimated the global investment required is estimated to be up to 5 4 times the current budgets (CBD, 2013). 5 6 It has been estimated that market-based mechanisms could generate up to 50% of conservation finance for 7 coral reef in 2020, but long-term, reliable instruments need to be established and strengthened (The 8 Katoomba Group, 2010). Five areas of financial innovations have been set out concerning the private sector: 9 schemes for payment for ecosystem services; biodiversity offset mechanisms; markets for green products; 10 public-private partnerships and new forms of charity; development of new and innovative sources of 11 international development finance. 12 13 Public-private partnerships (PPPs) are presented as a type of arrangement that addresses the conservation 14 financing gap. The main advantages of PPPs include: their flexibility to set fees and charges, establish 15 funding mechanisms such as concessions, respond to customer needs, their ability to retain the money they 16 earn (which gives a resulting incentive to generate funds through greater entrepreneurship), the implication 17 of local communities in the activities and their freedom to implement staffing policies based on efficiency 18 and market salaries (World Bank, 2014). 19 20 PPPs have taken a wide range of forms, which vary in the degree of involvement of the private entity in a 21 traditionally public infrastructure (European Commission, 2003). Five main categories of agreements have 22 been observed for nature conservation in the region: a parastatal agency, management contracts, leases, 23 concessions and joint ventures. All these categories are described more precisely in (European Commission, 24 2003). Private investment in marine biodiversity and ecosystem services is in its early stage of development, 25 and for the majority of economic instruments practical experiences are very limited (The Katoomba Group, 26 2010; Crédit Suisse, 2016). There are few investments that have been designed to produce benefits to the 27 marine environment with financial returns, by the name of marine impact investments or otherwise. In the 28 same way, very few private equity and venture capital firms are specialized in marine projects. 29 30 PPPs may further be used for marine conservation through active restoration approaches, where ecosystem 31 services and biodiversity are to be improved through the capitalization of the marine ecosystem (Bull et al., 32 2013; Penca, 2013). This is based on the concept to use market-based incentive mechanisms to empower 33 biodiversity (Rinkevich, 2015b). In the coral reef ecosystem, for example, the structural complexities of reef 34 habitats are strongly associated with the biodiversity of scleractinian corals (De'ath et al., 2012) (high 35 *confidence*), a property that may be enhanced through the employment of active reef restoration initiatives 36 (Rinkevich, 2015b). This conclusion is further noticed in cases where traditionally used biodiversity offsets 37 do not compensate for habitat destruction (Fujita et al., 2012). Harnessing tradable rights in active restoration 38 (such as reef restoration; Rinkevich (2015b)) by non-public stakeholders may add real compensation for 39 losses and on the other hand may be used as an efficient tool for a long-term financial support of marine 40 ecosystem restoration, as it is done by the logging industry in forest habitats. 41

One of the main investment vehicles in marine conservation is the Entrepreneurial Marine Protected Area
(EMPA) (Credit Suisse AG, 2016). An EMPA is a management area that is primarily funded by a profitbearing 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.

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50 Benefits include food and incomes for local communities, opportunities for tourism businesses, protection 51 from coastal erosion and a country more resilient to climate change (Halpern, 2003). While many terrestrial 52 protected areas have private sector involvement in the region (Juffe-Bignoli et al., 2014), significantly fewer 53 MPAs include the private sector. A potential obstacle in the further development of this tool are results 54 showing that the management instruments employed in MPAs, in general, often fail to achieve their 55 conservation objectives (Miller and Russ, 2014; Gill et al., 2017).

Chapter 5

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 molluses 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 BOX5.5 HERE]

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

22 Ocean acidification is not mentioned in the Paris Agreement on climate change, and has not been given 23 detailed attention in other UNFCCC discussions. Is that because the topic is not considered to be part of 24 climate change, or because it does not warrant distinct governance arrangements? If climate is strictly 25 defined as the long-term descriptor of meteorological conditions, then changes in ocean chemistry are out of 26 scope. However, physical changes in the ocean, such as sea level rise, are uncontroversially considered part 27 of climate change, and the CO₂-driven change in ocean pH is an inevitable, although indirect, chemical 28 consequence of human perturbations of atmospheric composition - the driver of all other changes to the 29 climate system. The close association between ocean acidification and other components of climate change 30 was recognised in IPCC AR4 and AR5, with Working Groups I and II (but not III) giving consideration to 31 the ecological and economic impacts of ocean acidification that are already occurring, and that are projected 32 to greatly increase under future emission scenarios.

34 Most literature on ocean acidification is relatively recent, with a five-fold increase during the past decade 35 (Riebesell and Gattuso, 2014; Browman, 2016). Nevertheless, IPCC AR5 WG I considered that there was 36 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 37 38 considered there was *high confidence* that ocean acidification will increasingly affect marine organisms, 39 ecosystems, and the goods and services they provide. Such assessments are confirmed in this report 40 (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 41 international bodies, including the Convention on Biological Diversity (CBD, 2016), the United Nations 42 Convention on the Law of the Sea (UNCLOS), the United Nations Environment Programme (UNEP), the 43 Intergovernmental Oceanographic Commission (IOC) of the United Nations Educational, Scientific and 44 Cultural Organization (UNESCO), the World Meteorological Organization (WMO), and the International 45 Atomic Energy Agency (IAEA); the interests in ocean acidification of these and other bodies are 46 summarised in CBD (2014). Public awareness of ocean acidification is, however, low (e.g., only 14% of 47 Europeans consider that they are 'informed' on the topic; Buckley et al. (2017)) and the topic has been 48 neglected to date from a regulatory perspective (Fennel and VanderZwaag, 2016). The most substantive 49 governance outcome to date is UN Sustainable Development Goal 14.3 (Section 5.3.2), with its non-binding 50 and relatively general commitment to 'minimise and address the impacts of ocean acidification, including 51 through enhanced scientific cooperation at all levels'.

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53 Whilst a new UN mechanism specifically to address ocean acidification has been proposed (Kim, 2012), 54 there are pragmatic arguments favouring strengthened UNFCCC involvement (Harrould-Kolieb and Herr,

- 55 2012). Action to reduce CO_2 emissions will necessarily, to some degree, address ocean acidification,
- 56 although there are topic-specific considerations. Governance issues not only relate to pH-associated regime
- 57 changes, thresholds or tipping-points (Hughes et al., 2013; Good et al., 2018), but also with regard to ocean

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1 acidification impacts in scenario-modelling for emission reductions (Steinacher et al., 2013), and specific 2 consequences for ocean acidification of several climate policy responses. In particular, any leakage from 3 sub-seafloor carbon storage is very likely to produce local pH impacts, similar to natural CO₂ vents 4 (Blackford et al., 2014), whilst some CO₂ removal techniques (negative emissions) such as ocean 5 fertilization, are likely to worsen acidification for the ocean as a whole (Cao and Caldeira, 2010; Williamson 6 and Turley, 2012). If policy interventions were to involve solar radiation management, the climate might be 7 stabilized, yet acidification would essentially continue unabated (high confidence) (Williamson and Turley, 8 2012; Keller et al., 2014). 9

10 Adaptation to climate change could also include a more integrated approach to reduce ocean acidification 11 impacts (Section 5.5.2). Proposed actions for ocean acidification (Kelly et al., 2011; Billé et al., 2013; Strong 12 et al., 2014; Albright et al., 2016) include reduction of pollution and other stressors (thereby strengthening 13 resilience); water treatment, e.g., for high-value aquaculture; adapting human activities; and repairing 14 damages. Seaweed cultivation and seagrass restoration may slow longterm changes at the local level, but can 15 also exacerbate short-term variability (Sabine, 2018). These measures are generally applicable to relatively 16 limited spatial scales: whilst they may succeed in 'buying time', their future effectiveness will decrease, 17 unless underlying global drivers are also addressed (high confidence). 18

19 Well-coordinated and fit for purpose ocean acidification monitoring is being developed through the Global 20 Ocean Acidification Observing Network (Newton et al., 2015), integrated with other marine monitoring at 21 local, national and global levels. Such action can be considered an essential part of the governance response, 22 and is now used operationally by the US oyster cultivation industry (Barton et al., 2015a). Good progress has 23 been made in characterising seasonal patterns (Sutton et al., 2016) and identifying the many other factors 24 that can cause high pH variability in coastal waters (Duarte et al., 2013). In the US, the 2009 Federal Ocean 25 Acidification Research and Monitoring Act (FOARAM) provided the basis for coordinated national 26 monitoring of ocean acidification and its consequences (IWGOA, 2014). The FOARAM Act also required: 27 the development of adaptation strategies to conserve organisms and ecosystems vulnerable to ocean 28 acidification; the consideration of associated socio-economic impacts; and the development of standardised 29 protocols for natural values and their variation. In the European Union,: measurement of ocean acidification 30 conditions is encouraged, but is not statutory - since pH change is considered to be a 'prevailing condition', 31 outside national management control (European Union, 2008). 32

[END BOX 5.5 HERE]

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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 2007in these terms:

⁶81. 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

- 51 to urgently pursue f 52 and measurement.'
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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;

1 A/RES/66/231, para.134; A/RES/67/78, paras.143-145; A/RES/68/70, paras.153-156; A/RES/69/245 paras 2 165-170, A/RES/70/235 paras 172-177 ; A/RES/71/257 paras 185-190.). 3 4 With the Rio + 20 summit and the recommendations in the 'Future we want', the resolutions devoted more 5 paragraphs and requested more action, effort and initiatives to further study, to minimize impacts as well as 6 to enhance international cooperation. 7 8 It is on the basis of Rio + 20 summit and the resolutions of the General Assembly requesting initiatives that 9 the IAEA and the Centre Scientifique de Monaco organized the 3rd international workshop which took place 10 in Monaco in January 2015. It aimed to provide policy makers and marine resource managers with 11 information and recommendations to prepare for social and economic impacts of ocean acidification on 12 coastal communities. 13 14 With regard to voluntary instruments, land-based activities are responsible for most of the pollution of the 15 oceans and affect the most productive areas of the marine environment. The 1982 Convention sets out 16 specific provisions for the prevention, control and reduction of pollution from land-based sources. However, 17 it gives to States a certain degree of discretion with respect to the implementation of international rules, in 18 deference to the sovereignty that a State exercises over the territory where such land-based pollution may 19 arise. As seen above States, however, have accepted to be bound by applicable rules at the regional level. 20 21 The Governing Council of UNEP identified in 1982 the question of land-based source of pollution as one of 22 the three priorities for the development of environmental law. In 1985 the Montreal Guidelines for the 23 Protection of the Marine Environment from Land-based Sources were adopted. They were guided by the 24 regional examples as the North-East Atlantic, the Baltic and the Mediterranean Sea. The Guidelines have set 25 the stage for a more comprehensive and global approach to addressing the causes of marine and coastal 26 degradation. 27 28 In November 1995, two international documents were adopted by an intergovernmental conference: 29 The Washington Declaration on the Protection of the Marine Environment from Land-based Activities and 30 The Global Programme of Action for the Protection of the Marine Environment from Land-based Activities 31 (GPA) (See UN document A/51/116). 32 33 The GPA provides guidance to national and/or regional authorities to devise and to implement sustained 34 action in order to prevent, reduce, control and/or eliminate marine degradation from land-based activities. 35 36 The GPA addresses the impacts of land-based activities on the marine and coastal environment, including 37 contaminants, physical alteration, point and non-point sources of pollution and areas of concern such as 38 critical habitats, habitats of endangered species and protection of ecosystem components, such as breeding 39 and feeding grounds. 40 41 From a strictly legal perspective, both the Washington Declaration and the GPA have no binding force and 42 are considered to be 'soft law'. They have, however, both obtained an enormous amount of support from 43 States and intergovernmental and non-governmental organizations that view the GPA as an essential 44 instrument for combating marine environmental degradation from land-based sources of pollution. 45 46 Since an estimated 3 billion people depend on marine and coastal biodiversity for their livelihoods, the 47 disruptions to the delivery of marine ecosystem services caused by climate change and ocean acidification 48 will seriously affect the economy of coastal communities and could also impact food security and could in 49 turn result in increased poverty (see Report of UNGA on Oceans and the law of the sea A/70/74, paras 67-50 70) and as discussed in Section 5.3. To consider, however, fishing activities, particularly Illegal, Unreported 51 and Unregulated fishing, and associate it with acidification is not entirely correct. If there is at all a link it is 52 that it adds to the problems of overfishing which affect sustainable fisheries 53 54 To combat overfishing and particularly IUU fishing, several instruments were adopted under the auspices of

55 FAO. They deal either with the respect of international conservation measures by fishing vessels like the 56 *1993 Agreement on compliance with conservation and management measures*, or the increase control by

57 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

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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.

18 The level of knowledge and confidence of the changes in the marine environment is uneven. Many of the 19 observed changes can be directly attributed to human influence from rising greenhouse gases and other 20 anthropogenic forcings (e.g., aerosols and ozone depletion). On the other variables and systems the evidence 21 is less clear, but the relationship between rising greenhouse gases, changing ocean, marine ecosystems and 22 depedent communities is still surprisingly robust when considered together. In many cases, the assessments 23 are undertaken using a risk assessment framework. By combining ecological and physiological knowledge 24 (from experiment or direct observations) with the climate drivers, the change in risk level for the recent past, 25 and long term future are assessed (e.g., Sections 5.2.5 and 5.3.4). Taken globally, all the assessed marine 26 ecosystems have elevated risk of impacts on biodiversity, ecosystem structure and services with increasing 27 greenhouse gas emission. Specifically, all the coastal ecosystems that were assessed, where linkages between 28 natural systems and human communities are the strongest, all had increased risk, and none saw a risk 29 reduction (or beneficial change) from a warming climate.

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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).

39 Risk-reduction responses and their governance through adaptation at the local scale is the most common 40 approach to responding to climate change from ocean systems. It is clear that there are many choices for 41 reducing risk of climate change. Many of the actions have benefits and relatively few dis-benefits, while 42 others have large dis-benefits and marginal effectiveness (Figure 5.21). Many of the risk reduction 43 approaches are limited in their capacity to reduce the risks of climate change, or are at best temporary 44 solutions, and this is significant challenge to adapting to climate change. The assessment points to the 45 increased effectiveness and importance of a portfolio of different types of mitigation and adaptation options. 46 Governance is also critical element in the portfolio of options and occurs at a variety of scales, local, national 47 and international. Such responses can be more effective with the support of scientific information, Local 48 knowledge and Indigenous knowledge, and the consideration of local context and inclusion of stakeholders.

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51 5.7 Key Uncertainties and Gaps52

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

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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.

- 4 5 Variance in human systems and effectiveness of responses: the wide range of contributing factors 6 (physical, social and economic) that interact with localised climate projections (which typically have 7 high levels of uncertainty) make projecting site-specific costs of impacts and benefits of adaptation 8 difficult. There were few examples in the literature of assessments of implemented adaptation actions, 9 and there was low confidence in their reliability and provenance, thus largely precluding any assessments 10 of their cost effectiveness. This lack of evidence on cost/benefit particularly affected assessments in 11 Section 5.5. Adaptation responses to climate change have been undertaken by communities and 12 governments. However, their effectiveness for mitigating the risks of climate change (e.g., different 13 types of adaptation response on the coasts, Section 5.5.2) is largely unassessed here, and consequently 14 precludes a global understanding of the capacity in the world to address the risks of climate change on 15 our coastal seas and open ocean. A partial solution would be establishing an appropriate ocean and 16 coasts database for these types of studies.
- 17 Biological processes and monitoring: while some biological processes in the ocean are well understood 18 there are a surprising number of marine environments where the lack of scientific understanding 19 frequently precludes assessments of risks and the capacity to project their response to climate change: 20 these include incomplete 'models' for quantitative projections of the future of key organisms and 21 ecosystems, gaps in our knowledge of feedbacks in biological systems particularly in coastal ecosystems 22 (Section 5.3.4), the capacity and limits of biological adaptation for many organisms and ecosystems, and 23 the very sparse knowledge of the sensitive deep ecosystems that may be uniquely sensitive to climate 24 change (Section 5.2.2 and 5.2.4).
- 25 **Physical and biogeochemical processes:** while the Earth system is better monitored and the relevant 26 data are more accessible than the first two bullets described above, significant gaps remain. We wish to 27 highlight deep ocean temperature, salinity measurements for sea-level and closure of the energy budget, 28 oxygen and carbon measurements dense enough to measure de-oxygenation of the world ocean and track 29 the mechanisms driving the ocean carbon cycle; and the rates of carbon burial in coastal sediments as 30 significant omissions. Projections of future changes in the Earth system depend on the use of coupled 31 models. While large-scale climate changes are believed to be well described by these models, there are 32 still uncertainties in the timing, magnitude and patterns of the projected changes arising from physical 33 and ecological processes that are omitted or incompletely understood, from intrinsic biases arising from 34 the models' formulation, and from resolutions that are still too coarse to capture important regional 35 processes and their interactions with the global ocean. Other examples of under-assessed biogeochemical 36 process in the ocean that may have implications for the Earth system under climate change include 37 methane in deep ocean sediments (Section 5.2.4). Open ocean primary productivity and its projections 38 still needs critical corroborating measurements to track the potential changes in ocean productivity with 39 higher confidence (Sections 5.2.2 and 5.2.3). 40

See Cross-Chapter Box 5 'Southern Ocean Circulation: Drivers Changes and Implications' in Chapter 3.
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44 [START FAQ5.1 HERE] 45

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46 FAQ5.1: How is life in the sea affected by climate change?47

48 Life in most of the global ocean, from pole to pole and from sea surface to the abyssal depths, is now 49 experiencing higher temperatures due to human-driven climate change. In many places, that increase may be 50 barely measurable; in others, particularly in near-surface waters, warming has already had dramatic impacts 51 on marine animals, plants and microbes. Due to closely-linked changes in sea water chemistry, less oxygen 52 remains available (the process is called ocean deoxygenation) and more carbon dioxide is dissolved, causing 53 ocean acidification. Non-climatic effects of human activities are also ubiquitous, including over-fishing and 54 pollution. Whilst the combination of these stressors is *likely* to be harmful to almost all marine organisms, 55 food-webs and ecosystems, some are at greater risk (FAQ5.1, Figure 1). Consequences for human society 56 can be serious if insufficient action is taken to constrain future climate change. 57

1 Warm-water coral reefs are particularly vulnerable, since they can suffer high mortalities when water 2 temperatures persist at between 1°C–2°C above the normal range. Such conditions occurred in many tropical 3 seas between 2015–2017 and resulted in extensive coral bleaching, when the coral animal hosts ejected the 4 algal partners upon which they depend. After mass coral mortalities due to bleaching, reef recovery typically 5 takes at least 10–15 years. Accelerating sea level rise, reef erosion and acidification are slowing down reef 6 recovery rates, especially when bleaching reoccurs. Whilst some coral species are more resilient than others, 7 further reef degradation due to future climate change now seems inevitable. This can have serious 8 implications, like loss of coastal protection for many islands and low-lying areas and loss of the high 9 biodiversity these reefs host. Coral habitats can also occur in deeper waters and cooler seas. Although these 10 are not at risk from bleaching, they may weaken or dissolve under ocean acidification, and other ocean 11 changes.

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13 More mobile species, such as fish, may respond to climate change by moving to more favourable 14 regions/areas, with populations moving poleward, or to deeper water, to find their preferred range of water 15 temperatures or oxygen levels. As a result, projections of total future fishery yields under different climate 16 change scenarios only show a moderate decrease of around 4% (~3.4 million tonnes) per degree Celsius 17 warming. However, there are dramatic regional variations. Under scenarios of high greenhouse gas 18 emissions, fish biomass and maximum catch potentials could be up to 50% lower in tropical regions by the 19 end of the century. Losses may be partly offset by increases in some fish stocks in polar regions. The extent 20 of such gains is uncertain, as fish populations depleted by over-fishing, and subject to other stressors, may 21 not be capable of making the migrations that are assumed in model simulations. 22

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).

Region	pH change	uncertainty	Study	Study period	Study type
	(pH units decade ⁻¹)			
			Bates et al.		
Irminger Sea	-0.026	0.006	(2014)	1983-2012	Time series
N. Atlantic			Bates et al.		
(BATS)	-0.017	0.001	(2014)	1983-2012	Time series
N. Atlantic			Takahashi et al.		
(BATS)	-0.018	0.002	(2014)	1983-2010	Time series
			Olafsson et al.		
Iceland Sea	-0.023	0.003	(2009)	1985-2008	Time series
N. Pacific			Dore et al.		
(ALOHA)	-0.018	0.001	(2009)	1988-2007	Time series
N. Pacific			Bates et al.		
(HOT)	-0.016	0.001	(2014)	1988-2012	Time series
N. Pacific			Takahashi et al.		
(HOT)	-0.018	0.001	(2014)	1988-2009	Time series
			Ishii et al.		
N.W. Pacific	-0.02		(2011)	1994-2008	Time series
N. Atlantic			Bates et al.		
(ESTOC)	-0.018	0.002	(2014)	1995-2012	Time series
			González-		
N. Atlantic			Dávila et al.		
(ESTOC)	-0.017	0.001	(2010)	1995-2004	Time series
Caribbean			Bates et al.		
(CARIACO)	-0.025	0.004	(2014)	1995-2012	Time series
N. Atlantic			Takahashi et al.		
(ESTOC)	-0.02	0.004	(2014)	1996-2010	Time series
S.W. Pacific			Bates et al.		
(Munida)	-0.013	0.003	(2014)	1998-2012	Time series
E. Equatorial			Xue et al.		Merged ship
Indian	-0.016	0.001	(2014)	1962-2012	occupations
Polar Zone			Midorikawa et		Merged ship
Southern Ocean	-0.02	0.003	al. (2012)	1963-2003	occupations
			Midorikawa et		Merged ship
N.W. Pacific	-0.015	0.005	al. (2010)	1983-2007	occupations
			Byrne et al.		Merged ship
N. Pacific	-0.017		(2010)	1991-2006	occupations
Pacific Southern			Williams et al.,		Merged ship
Ocean (S4P)	-0.022	0.004	2015	1992-2011	occupations
			Waters et al.		Merged ship
S. Pacific	-0.016		(2011)	1994–2008	occupations
Pacific Southern			Williams et al.,		Merged ship
Ocean (P16S)	-0.024	-0.009	2015	1995-2011	occupations
			Takahashi et al.		Merged ship
Drake (PZ)	-0.015	0.008	(2014)	2002-2012	occupations
			Takahashi et al.		Merged ship
Drake (SAZ)	-0.023	0.007	(2014)	2002-2012	occupations

Appendix 5.A, Table 2: Tidal exposure: IT: intertidal; S: submerged. Ecosystem assessed component: GM: geomorphology. BGQ: biogeochemistry. BI: Benthic invertebrates. P: Plankton. V: Vegetation. F: Fishes. MM: Marine							
Coastal	SB: Sea t Tidal exposure	DIRDS. SE: So Ecosystem component	Sensitivity	Ies. MA: Maci Adaptive capacity	roalgae. Vulnerability (Risk)	Other hazards	Impacts
Estuaries	← → IT	GM, BGQ, BI, P, V, F, MM	SST->BGQ, BI, P, V, F, MM- medium (obs+proj) pH/CO2->BI, F- low (proj) SLR->BGQ, GM, V- high (obs+proj) DO->BGQ, P, F- medium (obs+proj) (especially in high latitud estuaries) Precipitation/river runoff (droughts/floods) > GM, BGQ, BI, V.	Moderate/High (High resilience. Shallow, microtidal estuaries are more vulnerable, especially in flooding or arid areas).	Moderate (medium confidence)	Habitat degradation due to human activities: aquaculture, agriculture, urbanization. Eutrophication. Pollution. Overfishing. Shipping/Dredging. Sedimentation/mouth closure.	Changes in salinity and turbidity gradients depending on the interaction between river runnof, sea intrusion and circulation patterns. Redistribution of benthic communities dependening on their tolerance range to salinity, sediment type, inundation. Hypoxia and anoxia. Fish kills. Increased bacterial respiration. OM accumulation. Occurrence of harmful algal blooms (HABs).
Salt marshes	₹₹	V, SE, F, SB	SST- >V, F- medium (obs+exp+proj) but positive/negative effects. pH/CO2->V- low (obs+proj+exp) but positive/negative effects SLR- >V, SE, SB- high (obs+proj). Increased storm disturbance (but positive/negative effects depending on sediment supply) > V, SE.	Moderate (Saltmarshes are initially resilient to SLR due to soil accretion (biomass accumulation and sediment deposition) but system wont be able to withstand SLR after 60 years under RCP 8.5.	Substantial (medium confidence)	Coastal development causing squeeze, altered flushing regimes. Species invasions. Mangrove encroachment. Eutrophication. Land use change for agriculture. Dredging. Overfishing.	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 restructuration from saltmarshes to mudflats, biodiversity loss. The projected loss in global coastal wetlands (20-60 %) is in part counterbalanced by some increase in arid and sub-tropical marshes (3-6%) under low SLR scenario and lateral reacomodation of sediments (if not constricted by human constructions) (some projections up to 60 % eain)
Mangroves	↓ ↓	V, SE, F, SB	SST- >V, F- low/medium (obs+proj) but positive/negative effects pH/CO2- >V- low (obs+exp+proj) but positive/negative effects SLR- >V, SE, SB- high (obs+proj) Increased storm disturbance (but positive/negative effects depending on sediment supply) >V, SE.	Low/Moderate (Long-living, large- size plants. Initially resilient to SLR due to soil accretion. Ecosystem under intense human impacts. Rehabilitation practices can stimulate soil elevation).	Low (medium confidence)	Coastal development causing squeezing. Habitat degradation due to human activities: Deforestation, aquaculture, agriculture, urbanization.	Hydro-geomorphological settings play important role in mangrove responses to SLR. Soil accretion can cope with low SLR scenario (RCP 2.6) throughout the 100 years projection period, but only up to mid-century under RCP 8.5. Fringe mangroves are more vulnerable to SLR than basin magroves. Mangroves under microtidal regime with low soil accretion capacity are also more vulnerable.
Seagrass meadows	S	V, F, MM, SB, T	SST- >V, F, MM, T- high (obs+exp+proj)pH/CO2- >V- high but positive/negative effects (obs+exp+proj)SLR- >V, SB- low (obs+proj)Increased storm disturbance >V.	Low/Moderate	Very high (very high confidence)	Eutrophication, habitat degradation, biological invasions.	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, 70% by 2050 and potential extinction by 2100 under RCP 8.5. Warming will lead to significant reduction of <i>Cymodosea nodosa</i> meadows (46 %) in the Mediterranean, and expansion into the Atlantic. Increased herbivory by tropical consumers on temperate seagrasses, ecosystem biodiversity loss
Sandy beaches	→ IT	GM, BI, V, P, T	SST- >BI, P, T- high (obs+proj) pH/CO2 >BI, P-low (obs+proj) SLR- > GM, BI, V, T- high (obs+proj) Increased storm disturbance, runoff >GM, BI, V.	Uncertain	Moderate (low confidence)	Coastal development causing squeezing.	Increasing erosion/sediment loss related to storm frequency. Poleward shifts in macrobenthic communities, reduction in body size with warming, mass mortality of clams. Dune scarping, vegetation loss, reduced turtle nesting sites. Occurrence of HABs.
Coral reefs	FIT S	GM, BGQ, BI, P, F	SST->BGQ, BI, P, F-high (obs+exp+proj) pH/CO2>BGQ, BI, P-high (obs+exp+proj) SLR->GM, BI- high (obs+exp+proj) Increased storm disturbance, rain (turbidity/nutrients) > GM, BI, P.	Moderate (some populations seem adapted to climate changes)	Very high (very high confidence)	Habitat degradation, eutrophication, overfishing, pollution.	Species-specific responses to multiple- climate drivers. Coral bleaching, mass mortality. Drastic reduction in coral fitness (growth, reproduction) due to combined effects of warming, acidification and SLR. Spread of invasive species. Reef dissolution and bioerosion. Shift in habitat structure from calcified corals towards algal or soft coral dominated reefs. Ecosystem biodiversity loss.
Rocky reefs	IT S	BI, MA, F, SB	SST->BI, MA, F- high (obs+exp) pH/CO ₂ ->BI, MA- high (obs+exp) SLR->BI, MA, SB- medium (proj)	Uncertain	High (medium confidence)	Eutrophication. Coastal development causing squeezing.	Poleward shifts of benthic fauna and algal species. Heat exposure during low tide and SLR constrict the area for relocation of the intertidal benthic communities. Simplification of the food web structure at low trophic levels due to warming, acidification and increased grazing on calcareous

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							species. Reduction in habitat complexity (shift from calcareous species to weedy algae). Macroalgae responses to acidification and warming depend on light and nutrient levels. Ecosystem biodiversity loss.
Kelp forests	S.	MA, MM, F	SST- >MA, MM, F- high (obs+exp+proj)pH/CO2- > MA low (exp+proj)SLR (nd)Increased heatwaves and storm disturbances. > MA. Reduced nutrient input in upwelling areas associated with warm waters of El Niño event >MA.	Low (kelps are highly sensible to warming and have low dispersal capacity)	High (high confidence)	Habitat degradation by human activities. Overgrazing by sea urchins.	Mass mortality of kelps due to heatwaves combined with high irradiance; eutophication delays the re- establishment. Global range contractions of kelps at the warm end of distributional margins and expansions at the poleward end, spread of invasive species. Reduction in habitat complexity (from kelps to turfs). Macroalgae responses to acidification and warming depend on light and nutrient levels. Ecosystem biodiversity loss. In polar fjords, kelp fitness is reduced by warming and increased turbidity due to ice-melting.

Appendix 5.A, Table 3: Review of literature on linkages between ecosystem and ecosystem services in eastern boundary upwelling systems.

System components	Observed	Projected	Confidence and Uncertainty	References & Lines of evidence
Upwelling-favourable winds and upwelling intensification	Intensification of upwelling- favourable winds (except in the Canary Current). No evidence of multi-decadal trends in strength or position of pressure systems.	Increasing (decreasing) trends in coastal upwelling in poleward (equatorward) regions of the EBUS. Poleward migration of ocean high-pressure systems, but little to no change in intensity of the thermal continental low- pressure cells.	Medium confidence Uncertainty due to local and regional factors of the EBUS.	Sydeman et al. (2014) O Rykaczewski et al. (2015) M Wang et al. (2015) M Brady et al (2017) M
Coastal temperature and stratification	Complex integration of global (climate change) and local (coastal upwelling) processes. Differences in global and EBUS SST trends are observed.	Ocean warming and stratification might be ameliorated by increased upwelling. At local scales however coastal upwelling may be reduced by ocean stratification.	Low confidence Trends are sensitive to location (nearshore vs. offshore), resolution, period and dataset considered, as well as to decadal variability.	Brochier et al. (2013) M Schroeder et al. (2013) O Oyarzún & Brierley (2018) M Xiu et al. (2018) M
Biogeochemistry	Decreased pH and oxygen concentration	Decreased pH and oxygen concentration, and increased nutrient concentration. Increased frequency of hypoxic events	Low confidence Uncertainty due to short period of data availability, as well as to decadal variability (e.g., ENSO). Lack of comparison between all EBUS	Lachkar (2014) M Chavez et al. (2017) O Graco et al. (2017) O Levin (2017) O, M Breitburg et al. (2018) O,M Turi et al. (2018) M
Primary production	Significant NPP increase over 1989-2011 in the California Current; flat trend over 1998- 2015 in the Canary current; flat trend in surface Chl over 1965-2008 in the Humboldt Current.	Variable trends and complex non-linear response of primary productivity to increased nutrient input in relation to upwelling intensity.	Low confidence Deep uncertainty, given to range of relevant processes operating at different regional, temporal and spatial scales. Mismatch between upwelling intensity and productivity. NPP derived from remote sensing models lacks validation with decadal	Messie & Chavez (2015) Renault et al. (2016) M Espinoza-Morriberón et al. (2017) O,M Gómez-Letona et al. (2017) O Pennington & Chavez (2017) O Xiu et al. (2018) M

Fisheries	Decrease in important fisheries such as Peruvian anchoveta (<i>Engralis ringens</i>) during El Nino in Humboldt current ecosystems, and California off Oregon – hypoxia has supressed fishery catch Expansion of hyoxia-tolerant resources (Jumbo Squid)	If upwelling intensify, nutrient supply in subtropical euphotic zones may be enhanced, resulting in a potential increase in fisheries production; However, an increase in acidic and hypoxic conditions in shelf habitats (Sydeman et al., 2014), which could detrimentally impact species sensitive to these parameters. Consequences on human dependent communities of changing oceanographic conditions in the upwelling systems could be observed from historical inter-annual variability such as during ENSO events.	Sydeman et al. (2014) Allison and Bassett (2015) Gattuso et al. (2015) Levin and Le Bris (2015) Kämpf and Chapman (2016) Blasiak et al. (2017)
Other Services	Decrease open ocean carbon uptake, coastal protection and recreational services. Scientific research could intensify;	Habitat and biodiversity loss projected from expanding hypoxia	McClatchie et al. (2010) Sperling et al. (2016)
Social-economic-political characteristics	Large regional differences in direct dependence on living marine resources, current and future population growth, political stability and human security	The expected increase in population growth and climate change on coastal regions exacerbate vulnerabilities to the biophysical impacts of climate change	Allison et al. (2009) Blasiak et al. (2017)

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

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



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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

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