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

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Executive Summary

The oceans have continued to warm unabated since 2004 and last AR5 assessment report. Warming still dominates the increase in heat energy stored in the climate system. It is *virtually certain*[1] that the upper ocean (0−700 m) warmed from 2004 to 2015. The layer 700–2000 m is steadily warming and the trend signal is very clear for the 2005–2017 period (*virtually certain*). It is very likely that anthropogenic forcings have made a substantial contribution to increases in global upper ocean heat content (0–700 m) observed since the 1970s. There is evidence for warming resulted from human-induced greenhouse gas emission in some individual ocean basins (*high confidence*[2]). The abyssal ocean continues to warm in the Southern Hemisphere (*high confidence*). This overall warming of the oceans will continue to increase this century even if radiative forcing has been stabilized (e.g., RCP2.6, *high confidence).* Under the RCP2.6 and 8.5 scenarios, by 2100 the ocean is *likely* to take up about 3 or 6 times the roughly 500 ZJ that the oceans have already taken up since the 20th century. {5.2.1.2.1}

It is very likely that regions of high salinity where evaporation dominates have become more saline, while regions of low salinity where precipitation dominates have become fresher since the 1950s. These regional trends in ocean salinity provide indirect evidence that evaporation and precipitation over the oceans have changed (*medium confidence*). It is likely that anthropogenic influences have affected the global water cycle since 1960 as evidenced by changes in surface and sub-surface ocean salinity (*very likely*). There is evidence of surface freshening at high latitudes from ice sheet melt. {5.2.1.2.2}

Ocean 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*). This trend reduces surface exchange with deep ocean affecting heat and carbon uptake, re-oxygenation of ocean and nutrient cycles. Significant increases in basin-averaged stratification has been observed (*high confidence*). It is *very likely* that the average stratification of the top most 500 m (outside of well-mixed boundary layers) will increase significantly throughout the 21st century in most major ocean basins. {5.2.1.2.2}

The quasi-steady ocean circulation, ocean mesoscale eddy field, tides and ocean turbulence are projected to change over the next century. The Atlantic Meridional Overturning Circulation is projected to decrease by 2100 (*very likely*), which would reduce the northward transport of the Gulf Stream (*high confidence).* In the deep ocean there will *likely* be circulation changes caused by reduced rates of formation of North Atlantic Deep Water and Antarctic Bottom Water. Stronger Southern Ocean winds will *very likely* drive a more intense Southern Ocean mesoscale eddy field. Increased stratification and rising sea levels are *very likely* to alter spatial tidal patterns and change the tidal amplitudes in a majority of coastal areas by the end of the 21st century. Increased stratification in the upper ocean in the tropics and subtropics is *likely* lead to a reduction in the turbulent vertical diffusivities and thus affecting ecosystems. {5.2.1.2.2, 5.2.1.2.3, 5.2.1.2.3, 5.2.1.2.4}

The oceans are already acidifying. Anthropogenic trends at local scales in carbon speciation variables, including acidity (pH), calcium carbonate and aragonite saturation states (Ω_{CAL} and Ω_{ARAG}) are emerging rapidly from the background variability (*high confidence*) {5.2.1}. The anthropogenic pH signal has already emerged over the entire ocean {Box 5.1}. There is *high confidence* that the surface ocean has already transitioned to a new state, where pH levels are consistently lower than the standard deviation of the previous state. For Ω_{ARAG}, the anthropogenic signal has already emerged from the background over much of the ocean (*virtually certain*). Emission scenarios are the most important control of ocean pH relative to

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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 1.8.3 and Table 1.2 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 1.8.3 and Table 1.2 for more details).

internal variability and inter-earth system model uncertainty for most of the 21st century at both global and local scale. {5.2.1.3} (*virtually certain*).

Observed global oxygen concentrations have decreased by 1–2% since 1970's and oxygen minimum zones have expanded. These changes are attributed to human influence. Over the next century oxygen declines are predicted by all CMIP5 models, projecting around a further 3.5% loss of total ocean oxygen by 2100 *(medium confidence)* due to the combination of warming and increase stratification under RCP8.5 emission scenario. At the seafloor, 200–3000 m depths, regions of the North Pacific, North Atlantic, Arctic and Southern Oceans may see oxygen declines by an average of 0.3% to 3.7% by 2100, with some areas having a decrease of 20–40%. {5.2.1.4}

Ocean primary productivity has declined, with converging evidence concluding overall declines in open ocean chlorophyll (as a proxy for phytoplankton biomass) since 1950s. These declines in productivity are attributed to enhanced stratification and the consequent depletion in surface nutrient concentrations (*low confidence*) in the tropics. In general, CMIP5 models project a small drop in global organic matter production (*medium confidence*) with increases in high latitude (*low confidence*) and decreases in low latitude (*medium confidence*) organic matter production. Across different biomes in CMIP5 models the projected declines in organic matter production are generally attributed to increased stratification, but in some models the temperature enhancement of phytoplankton growth is the dominant driver. {5.2.1.5, 5.2.2.2.2}

All coastal ecosystems are under stress from the combination of climate change impacts in the ocean and on land as well as non-climatic human activities, with observable and projected climate impacts through modification of geomorphology, reduction in habitat area, and shifting distributions and community structure (*high confidence*) {5.2.2}. Such impacts are evidenced, for example, in sandy beaches and saltmarshes where rising sea level and more severe storms are leading to habitat erosion and retreat that result in losses of vegetation and benthic fauna (*high confidence*) {5.2.2.3.4, 5.2.2.3.5}. Kelp forest and seagrass meadow ecosystems will continue to shift poleward as increased temperature and extreme events such as heat waves and storms (*high confidence*), and enhanced grazing by warming will result in physical and physiological stress, reducing their reproduction and productivity (*medium confidence*) {5.2.2.3.4, 5.2.2.3.8}.

There are almost no major coral reef systems (shallow and deep) that are not vulnerable to climate change with clear regional differences in their sensitivities. Shallow coral reefs that are not degraded by non-climatic impacts such as overfishing and nutrient enrichment are generally less exposed to disturbances and could constitute an important refuge to reefs degraded by climate change. However, loss of deep-water coral reef habitat is *virtually certain* under projected ocean acidification (aragonite undersaturation) through dissolution and intensified bioerosion of the non-living matrix. Naturally more climate change resilient coral reefs, where the reef systems were never bleached or support corals that are more resistant to stress or with the capacity to recover rapidly could offer hope for enhancing resilience of coral reefs to climate change. {5.2.2.3.6}

The strong positive relationship between annual Particulate Organic Carbon (POC) flux and abyssal sediment community oxygen consumption combined with projected changes in biomass suggests that benthic communities in abyssal and deep-sea habitats, which cover 56% of the ocean floor, will experience structural and functional changes that affect the carbon cycle in this century under all emission scenarios (*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*).{5.2.2.2.2, 5.2.2.4.7, 5.2.2.6.10, 5.2.1.5}

There is unequivocal evidence that distribution and phenology is responding to warming and changes in net primary production across all ecosystems and taxonomic groups from microbes to marine mammals (*very high confidence*) {5.2.2, 5.2.3}, leading to local population decrease in the equator-ward range boundary (*medium confidence*), expansion in the poleward boundary (*high confidence*), and overall shift in biomass and species composition (*very high confidence*) {5.2.2, 5.2.3}. The rate of range shifts is observed to be fastest for marine microbes and zooplanktons (hundreds of km per decade on average) {5.2.3.1, 5.2.3.2}, followed by othe vertebrates (tens of km per decade on average) {5.2.3.4} (*high* *confidence*). If evolutionary adaptation is slower that the rate of climate change, changes in biogeography and community structure are projected to continue in the 21st century (*high confidence*), with potential animal biomass projected to decrease by 4.8% and 17.2% under RCP2.6 and 8.5, respectively, by 2090–2099 relative to 1990–1999 (*high confidence*). The shifts in distribution and abundance of key prey species impacts predators, including marine mammals, seabirds and reptiles in which species with specialized ecology such as diet requirement, small range size and specific habitat requirement being most vulnerable (*high confidence*). The risk of population declines is compounded by interactions between climate change and non-CO_2 pollutants, particularly persistent organic pollutants (POPs) and mercury (*medium confidence*) { 5.3.2.1.3}.

Climate change is already affecting the flow in important ecosystem services in sensitive ecosystems, and is projected to affect all ecosystem services (*high confidence*) {5.3.1}. Fisheries catches and their composition are already affected by warming and decrease in net primary production through reduction in fish stock's reproductive successes and increase of warm water species (*high confidence*). In the 21st century, global fisheries catches are projected to decrease in the future at a rate of around 3 million tonnes per degree Celsius atmospheric warming relative to 1951–1960 level under projected decrease in net primary productivity and warming (*medium confidence*). Decrease in catch in the tropical ocean is projected to amongst the highest (*very likely*). Climate change impacts on marine ecosystems and biodiversity will also degrade regulatory, cultural and support services. This is evidenced from reduction in nutrient cycling in the deep sea floor ecosystems through warming-induced reduction in biodiversity (*medium confidence*) {5.3.1.3}, reduction in the quality and quantity of tourists attractions as coral reefs are degraded from warmings and increased severity of storm events (*high confidence*), and reduction in carbon stock and sequestration in salt marshes through reduced habitat availability for fauna requiring open vegetation structure as a result of sea level rise (*high confidence*).

Climate change is expected to substantially impact the security of coastal human communities, particularly Indigenous people and the Global South, through elevated risks on health, and loss of income and livelihood in the 21st century (*medium confidence*) {5.3.2}. Risk of water-borne disease such as *Vibrio cholera* is directly related to ocean warming (*high confidence* for the increased prevalence of the bateria, but *low confidence* for increased impacts on human health) {5.2.2, 5.3.2.1.1}. If decreases in fish stocks and potential catches are as projected by models (*medium confidence*) {5.3.1}, people who depend on fishing and related-sectors, particularly coastal Indigenous communities and the Global South, will experience substantial decline in their income, livelihood and availability of animal-sourced nutrients (*medium confidence*){5.3.2.2.2}. Global degradation and loss of corals, erosion of beaches will further impact these coastal communities that depend on marine-related tourism {5.3.2.2.2}. Their health risk also increases substantially because of the projected decline in fish supply and key fish micronutrients. These fish related declines may compound the shift from traditional seafood diets to processed food that could elevate the risk of diet-related chronic diseases {5.3.2.1.2}.

Coastal blue carbon ecosystems, including mangroves, salt marshes and seagrasses, can be a nature-based solution with multiple co-benefits. Some 151 countries around the world contain at least one blue carbon ecosystem and 71 countries contain all three. In addition to a range of provision (e.g., direct harvesting), supporting (e.g., nursery habitats for fishes) and cultural (e.g., tourism) services {5.3.1.1}, blue carbon ecosystems contribute 3–30% of oceanic CO_2 uptake and provide adaptation benefits such as coastal protection {5.4.1.1.2}. Successful implementation of measures currently to maintain and promote carbon storage in coastal ecosystems that are not specified in nationally determined contributions (NDC) would assist in reaching the net zero emissions target of the Paris Agreement (*high confidence*). These measures include improved regulatory protection, restoration and rehabilitation, and other measures such as reducing anthropogenic nutrient inputs and restoring hydrology. The carbon mitigation effectiveness from other coastal blue carbons such as seaweed ecosystems {5.4.1.1.3}, and other coastal non-biological CO_2 removal methods are smaller or have low feasibility currently {5.4.1.1.4}.

Even if blue carbon ecosystems could be restored to their 1990 extent (*unlikely*, because of the nature of much coastal development), globally, it cannot replace the need for the very rapid phase-out of fossil fuels, the control of all other emissions, and the development of environmentally-friendly negative emission technologies. However, it has greater potential importance at a national level for some countries. Given the co-benefits on enhancing ecosystem services and climate adaptation, coastal blue carbon can be considered as a 'no regrets' option, in addition to other, higher magnitude, mitigation measures. The potential for carbon mitigation by biological processes in the open ocean is limited {5.4.1.2.1}. Only ~1% of that production is removed annually from circulation by the combined effects of the three biologically-driven ocean carbon pumps. Human interventions to enhance CO_2 sequestration by these methods may have limited effectiveness, with risk of other undesirable climatic and non-climatic consequences, and contested governance issues {5.4.1.2}.

Existing socio-institutional adaptation is more common than ecosystem-based adaptation and built environment/engineered adaptation. Ecosystem-based adaptation (EBA) is considered to be a cost-effective coastal protection tool that has multiple co-benefits, with demonstrated applications in coastal ecosystems. There is *high agreement* that hard engineering responses are optimally supported by ecosystem-based adaptation approaches. Both approaches should be augmented by socio-institutional approaches for adaptation (*robust evidence, high confidence*). In planning adaptation responses, awareness-raising and stakeholder engagement are important for buy-in and ownership of responses (*robust evidence, high confidence*). Of particular importance in coastal adaptation is institutional capacity within local government organisations as emphasised in the recent literature (*robust evidence, high confidence*) {5.4.2.1, 5.4.2.2, 5.4.2.3}.

5.1 Introduction

The ocean is a key element of the earth. It stores the heat trapped in the atmosphere by rising greenhouse gases, it stores carbon dioxide in its ocean depths and it masks and slows surface warming of the earth. It is the home to the largest continuous ecosystem, providing habitats for 2,260,000 described eukaryotic species, including 35 animal phyla, 14 of which are exclusively marine, and is an essential source of food and livelihood that contributes to the world's food and human security. The ocean also provides other essential services to human through tourism, renewable energy, aesthetic appeal and sustaining local and traditional knowledge. Governance in the ocean also has a different set of approaches, challenges and opportunities from land that requires special treatment particularly under a changing climate.

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

The Working Group II of IPCC AR5 shows that marine ecosystems are changing with implications for human dependent society through changes in ecosystem services, with Chapter 5 covering coastal ecosystems (Wong et al. 2014), Chapter 6 covering ocean processes (Pörtner et al. 2014), Chapter 30 (Hoegh-Guldberg et al. 2014) covering the regional differences between ocean basins, and Chapters 14 – 20 covering general aspects related to adaptation and multi-sector impacts, risks, vulnerabilities, and opportunities that are also applicable to the ocean. These assessments concluded that ocean warming, acidification and deoxygenation are affecting marine life from molecular processes to organisms and ecosystems, with major impacts on the use of marine systems by human societies. Specifically, IPCC AR5 WGII highlights that "marine species have shifted their ranges, seasonal activities, migration patterns, abundance, and demonstrated altering of species interactions in response to ongoing climate change" (*high confidence*, Chapter 6 and 10). AR5 WGII also concludes that "future climate change will cause species richness to increase at mid and high latitudes (*high confidence*) and to decrease at tropical latitudes (*medium confidence*), resulting in global redistribution of catch potential for fishes and invertebrates, and implciations for food security (*medium confidence*)". Also, "ocean acidification poses risks to ecosystems, especially sensitive ones such as coral reefs (*medium* to *high confidence*)" (Adger et al., 2014).

This report updates these earlier assessments with new evidence on the changing ocean and marine environment. Unlike previous assessment reports, this Chapter undertakes an integrated assessment of the coupled human-natural marine system, with the current research and knowledge of how changes in physical, chemical, biological properties of the ecosystems interact with risks and vulnerabilities of the dependent communities under climate change. We also specifically assess methods and approaches that have been taken to reduce risk and the governance options that might mitigate the risks of anthropogenic climate change. Radiation management techniques (also known as sunlight reflection methods) are excluded here, since such approaches are direct geo-engineering and are outside the scope of the current report. However, marine renewables are included, for comparative purposes with the efficacy of natural marine systems.

The chapter design is structured around a suite of guiding questions. What are the key changes in the physical and biogeochemical properties of the ocean and major habitats and their taxonomic groups (Section 5.2). How have these changes impacted key ecosystems, risks to ecosystems services and human wellbeing (Section 5.3), and finally the effectiveness of coastal blue carbon and open/deep ocean initiatives for mitigating carbon emissions and pathways for adaptation for marine dependent communities (Section 5.4). These guiding questions mean that the report covers both regional and global geographic scales and across all aspects of human 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. This chapter subdivides the ocean into coastal, pelagic and deep-sea floor ecosystems. 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 and summarised in the footnote below[3].

5.2 Changing Oceans and Biodiversity

5.2.1 Changes in Physical and Biogeochemical Properties

5.2.1.1 Introduction

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 gasses and aerosols (Bindoff et al., 2013). These trends in the global average ocean temperature will continue for centuries after the anthropogenic forcing is stabilized (Collins et al., 2013).

The impacts on ocean ecosystems and human societies is primarily driven by regional trends and the local manifestation of the global-mean changes. At these smaller scales, the temperature, salinity, nutrient and oxygen fields of the ocean are also expected to exhibit well-understood basin and local-scale changes. However, the ocean also has significant natural variability at local and basin scales with timescales from days to decades and longer. This variability can mask the underlying trend and projected trends. The impact of multiple stressors on marine ecosystems, which can compound risks, is also less well understood. There is new evidence and understanding about multiple stressors which has accumulated since the last assessment report. The most severe impacts of a changing climate will typically be experienced when conditions are driven outside of the range of previous experience (see Box 5.1).

This section summarizes our emerging understanding of the primary changes to the ocean, along with a survey of several key areas of scientific uncertainty about the changes in the oceans. Because many of these long-term trends already have been extensively discussed in previous assessments (IPCC, 2013), much of this summary of the physical changes will be brief except where there are significant new findings. Our emerging understanding of the processes driving ocean mixing and its rich geography and temporal variability suggests the potential for there to be climate-related changes that are not yet accounted for in our projections.

5.2.1.2 Changing Temperature, Salinity, Circulation

5.2.1.2.1 Temperature changes- observed and projected

The oceans as a whole are warming (*high confidence*) as a direct result of anthropogenic changes to the radiative properties of the atmosphere and the heat budget of the Earth (*high confidence*). Over the past few decades, our ocean observing system has documented an increase in ocean temperature (Figure. 5.1) that corresponds with an uptake of about 93% of the excess heat accumulated in the Earth system over multiple decades (Bindoff et al., 2013; Rhein et al., 2013). The ocean heat uptake of 274 ZJ[4] during 1970-2010 is equivalent to a planetary energy imbalance of 0.42 W m^{-2} (with respect to the Earth sufarce area $5.10x10^{14}$

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[3] FOOTNOTE: In this chapter, the terminology for uncertainty as proposed by Hegerl et al. (2010) is used. Where there is only a confidence assessment, the terminology used to describe the available evidence is: 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. In this Chapter we adopt the terminology for detection and attribution as proposed by the IPCC Good Practice Guidance Paper (Hegerl et al., 2010). Thus, where we have an assessed likelihood the following terms have been used: 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.

[4] FOOTNOTE: ZJ is Zettajoule and is equal to 10^{21} joules.

m^2,(Rhein et al., 2013)) and has significantly contributed to global mean sea level rise (see Chapter 4). It is *virtually certain* that the oceans will continue to take up heat throughout the 21st century, at a rate that depends upon which radiative forcing scenario we collectively choose to follow. The ensemble of CMIP5 climate models project that under the RCP2.6 scenario (see Section 1.8.2.3) for description of RCP scenarios), the oceans will *likely* take up an additional 1500 ZJ of heat (Figure 5.1), or about 3 times the roughly 500 ZJ that the oceans have already taken up since the 19th century Roemmich et al. (2012). With the RCP8.5 scenario, the oceans will *very likely* take up about twice as much more heat than RCP2.6 (Figure. 5.1). Even a low carbon emissions pathway of RCP2.6 there is only a 66% chance of avoiding 2°C of surface temperature warming relative to preindustrial times and the oceans will continue to warm for several centuries to come (Collins et al 2013).

Historically, scientific research expeditions starting in the 19th century have provided occasional sections measuring deep ocean properties (Roemmich et al., 2012). Greater spatial and temporal coverage of temperatures down to about 700 m from expendable bathythermographs along commercial shipping tracks starting in the 1970s (Abraham et al., 2013). Since the early 2000s, thousands of autonomous profiling floats (Argo floats) have provided high-quality temperature and salinity profiles of the upper 2000 m of ice-free regions of the ocean (Abraham et al., 2013; Riser et al., 2016). Further advances in autonomous floats have been developed that now allow these floats to operate in seasonally ice-covered oceans (Wong and Riser, 2011; Wong and Riser, 2013), more recently to profile the entire depth of the water column down to 4000 or 6000 m (Johnson et al., 2015; Zilberman, 2017) and to include biogeochemical tracers (Johnson et al., 2017). Autonomous floats have revolutioned our sampling and accuracy of the global ocean temperature record and increased certainty and confidence in global estimates of the earth heat (temperature) budget, particularly since 2004 (Von Schuckmann et al., 2014; Roemmich et al., 2015; Riser et al., 2016). Findings using data collected from such observatory systems mark significant progress since AR5 assessment report.

Detailed regional patterns of temperature and heat content at depths of 0–2000 m during the early 21st century are consistently shown owing to the improved observing network (Roemmich et al., 2015; Desbruyères et al., 2016a) (Figure 5.1A and B). All ocean basins have experienced significant warming since the last decades (Figure 5.1A and B), with the greatest warming in the southern oceans (Roemmich et al., 2015; Trenberth et al., 2016), the tropical/subtropical Pacific Ocean (Roemmich et al., 2015), and the tropical/subtropical Atlantic Ocean (Cheng and Chen, 2017). Large scale patterns of natural variability at interannual to decadal time scales can mask the long-term warming trend (Figure 5.1A), particularly in the tropical Pacific and Indian Ocean (England et al., 2014; Liu et al., 2016). The significant upper 700 m warming occurred in a large extratropical band of the Southern Hemisphere between 30º S and 60º S mostly centred around 40º S, in the tropical North Indian Ocean north, and in the Northern Pacific. Warming of the extra-tropical Southern Ocean is mainly concentrated over southern Indian and Pacific subtropical gyres.

The upper 700 m cooling dominates in the North Atlantic around 40º S-60º S. In the North Atlantic, extreme cold surface and subsurface temperature conditions persist since the year 2014 (Josey et al., 2015) linked both to circulation changes (Robson et al., 2014; Smeed et al., 2014; Yeager et al., 2015) and anomalous atmospheric conditions (McCarthy et al., 2015; Josey et al., 2018) and together cause a cooling pattern in regional ocean heat content trends (Figure 5.1A). This robust cooling of North Atlantic is consistent with a reduction in the strength of the ocean circulation and heat transport that linked to record low densities in the deep Labrador Sea due to deep ocean warming since 1995 and role played by long-term freshening (Robson et al., 2014).

Cooling trends are also found in the south-western Indian Ocean and within most of the northern subtropics (10º N and 30º N) and eastern basins of the Pacific Ocean. The observations show that most of the global ocean heat is stored into the southern hemisphere subtropical gyres. These subtropical gyres are driven, in part, by the surface winds, which has been intensified in recent decades, facilitating the penetration of heat to deeper depths (Gao et al., 2018).

At depths of 700–2000 m, significant warming patterns in the Southern Hemisphere extratropics around 40º S and the subpolar North Atlantic are observed. Apart from this, all the ocean basins are showing uniform warming patterns. This suggests that deep ocean heat uptake is quite robust and continuously increasing as mentioned by previous literatures indicating the emergence of climate change signature is quicker than in upper ocean where the winds have a larger influence on the circulation (<700 m).

Inspite of these patches of cooler water every one of the world's ocean basins have experienced significant warming since the last decade (Figure 5.1 E and F and also (Desbruyères et al., 2016a)). The greatest warming is observed in the southern oceans Roemmich et al. (2015); (Trenberth et al., 2016), the tropical/subtropical Pacific Ocean, and the tropical/subtropical Atlantic Ocean (Cheng and Chen, 2017). Indeed the Southern Hemisphere is 67-98% of the total heat increase and for this period of 2005 to 2015 the warming of the ocean in the instrumental record since 1970 has continued unabated ((Roemmich et al., 2015); Figure 5.1 E and F). For the recent period (2004 to 2014) recent analyses are all in agreement and show that the oceans have continued to warm with a rate of 0.4–0.6 W m^{-2} (ie *very likely* range), and thus in the same range as the IPCC AR5 WGI report Rhein et al. (2013).

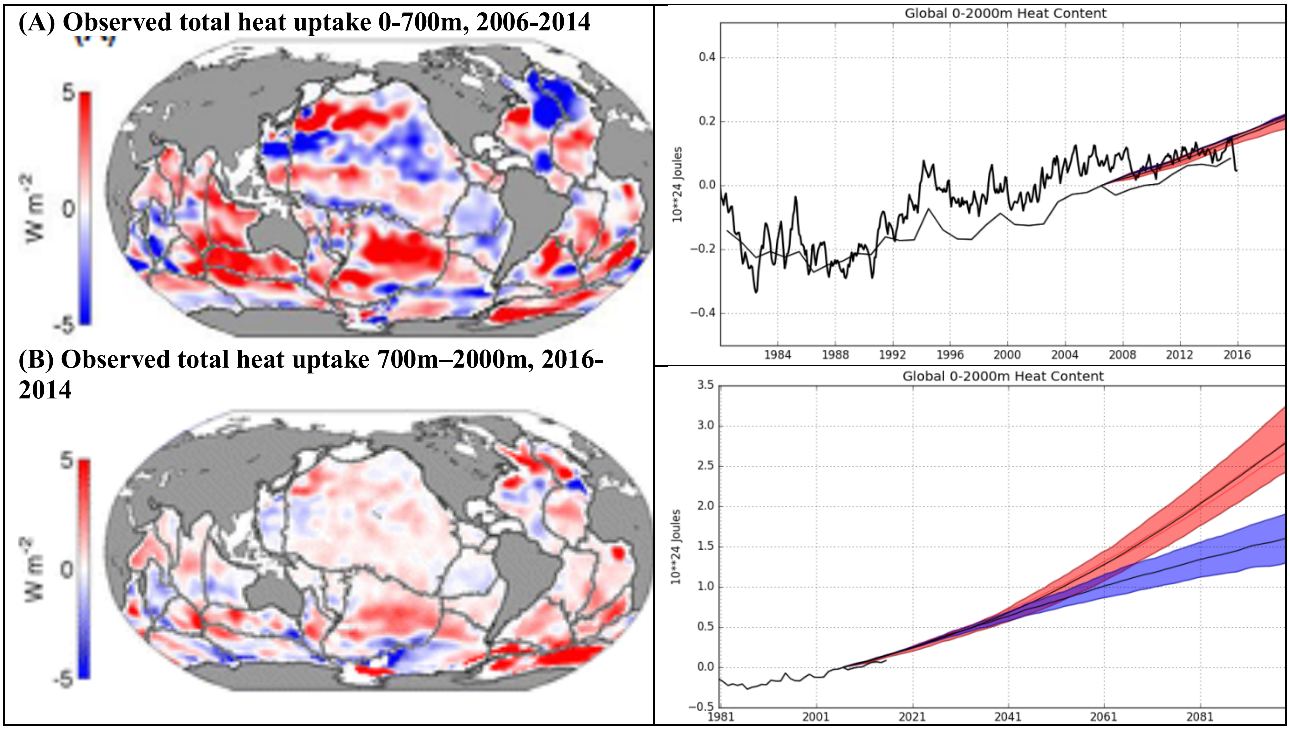
The ensemble average of the CMIP5 climate models project widespread ocean warming over the coming century, concentrated in the upper ocean (Figure 5.1). The anthropogenic heat will penetrate into the ocean following well-established watermass advection pathways. The greatest vertically integrated heat uptake occurs where there is already the formation of interior waters, such as Antarctic Intermediate Water along the Antarctic Circumpolar Current or North Atlantic Deep Water in the Nordic Seas, but all watermasses that are subducted with timescales of order decades are expected to experience significant warming (see Figure 5.5). The warming in the subtropical gyres penetrates further into the interior than other gyres (roughly 15 to 45°N and 15 to 45°S in Figure 5.5), following the wind-driven bowing down of the density surfaces (the solid lines in Figures 5.4 and 5.5) in these gyres. The greater warming at 700-2000 m in the Atlantic than the Pacific or Indian Oceans reflects the strong southward transport of recently formed North Atlantic Deep Water at these depths by the Atlantic Meridional Overturning Circulation (AMOC). Two areas that commonly exhibit substantially reduced near-surface warming over the course of the 21st century are the northern North Atlantic, where a slowing AMOC (see chapter 6) reduces the northward heat transport and brings the surface temperatures closer to what is found in other ocean basins at these latitudes, and the southern side of the Southern Ocean, where water upwells that has been submerged for so long that it has not yet experienced significant climate change. Most of these projected warming patterns are broadly consistent across the current and previous generations of climate models, are consistent with observations and understood theoretically, giving high confidence that they describe the changes the real world will experience.

The global ocean below 2000 m has warmed significantly between the 1990s and 2010s (Figure 5.2), contributing to ocean heat uptake and through thermal expansion to sea level rise (Purkey and Johnson, 2010; Desbruyères et al., 2016b). The observed deep warming rate varies regionally and by depth reflecting differences in the waters influencing particular regions. The strongest warming is observed in regions of the deep ocean Antarctic bottom water (AABW) (Purkey Sarah et al., 2014). The Southern Ocean south of the Sub-Antarctic Front, directly down stream from AABW formation sites, has experienced the largest warming rate of 0.02°C per decade below 1000 m (Figure 5.2a). To the north along AABW flow pathways, the warming signal attenuates with decreasing influence of AABW, but is still statistically different from zero as far north as the North Pacific (Figure 5.2b). The total deep ocean warming below 2000 m is equivalent to a heat flux of +22 TW, resulting in a mean sea level rise of 0.07 mm yr^{-1} from thermal expansion.

Since the 1990s, some regional variability has been observed in deep warming rates, although mostly within estimated errors. The deep and abyssal North Atlantic, fed by North Atlantic Deep Water (NADW), has reversed from warming to cooling possibly associated with the North Atlantic Oscillation (e.g., Yashayaev, 2007; Desbruyères et al., 2014) or longer-term weakening in North Atlantic overturning circulation (Caesar et al., 2018; Thornalley et al., 2018). Elsewhere, regions of the ocean fed by AABW from the Weddell has shown some variability with a possible slowdown in local AABW warming rates (Johnson et al., 2014), while the Pacific, fed by AABW from the Ross and Adelie Coast, has continued to warm at an accelerating rate between 1990 and 2018 (Desbruyères et al., 2016b).

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

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



(A) Observed total heat uptake 0-700m, 2006-2014

(B) Observed total heat uptake 700m–2000m, 2016-2014

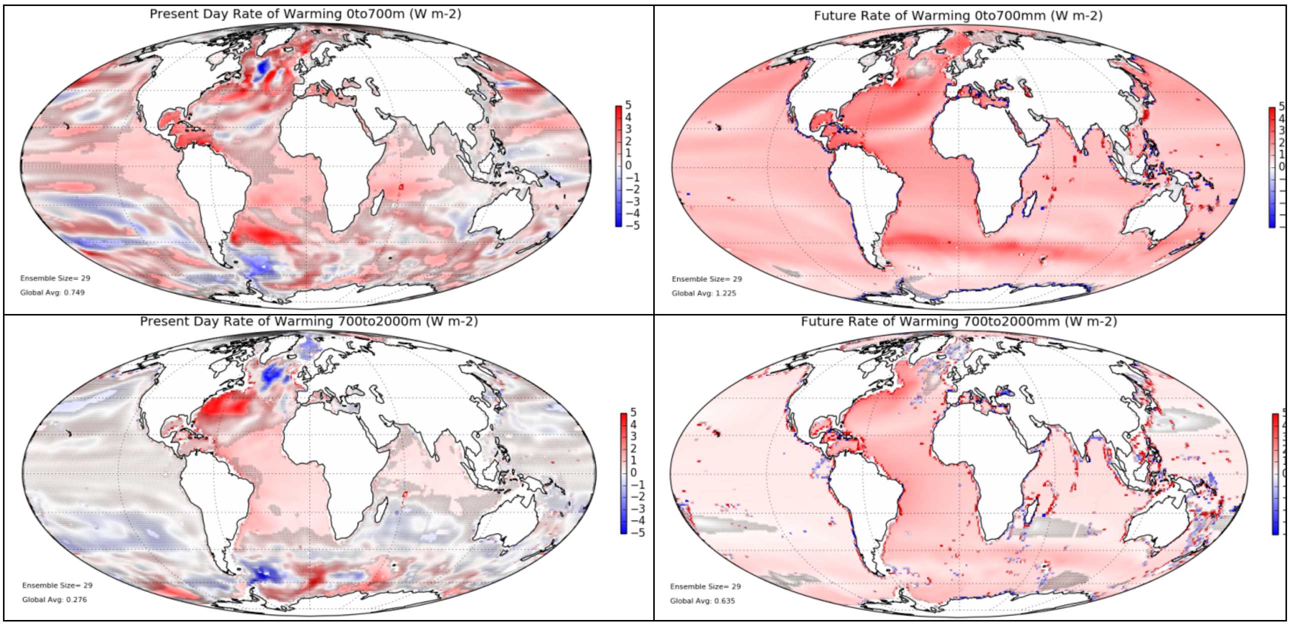


Figure 5.1: Ocean heat content changes: (A) Spatial distribution of the heat content changes of the uppermost 700 m converted into a heat flux in W m^{-2} as observed 2006–2014 (Desbruyères et al., 2016a); (B) Observed heat uptake from 2007-2014 but for the region from 700 m to 2000 m; (C) CMIP5 CM2G historical ensemble mean heat content changes integrated over the top 700 m RCP4.5(2013 to 2017) – Historical (2000 to 2004) with shading to indicate where the changes from the control runs are not significant at the 95% level as estimated based on the internal variability of the control run; (D) like (C), but heat content changes from 700 to 2000 m; (E) Globally integrated 0 to 2000 m ocean heat content changes as observed from two observational datasets (black lines) and CMIP5 projected (1980-2020) ensemble mean and spread for RCP2.6 (blue) and RCP8.5(red) forcing; (F) Like (E) but with the time axis expanded to show projections through the 21st century; (F) Projected CMIP5 ensemble mean RCP8.5 heat content uptake for the top 700 m from (2081 to 2100) minus (1986 to 2005); (G) like (F) but for the 700 m to 2000 m depth range.

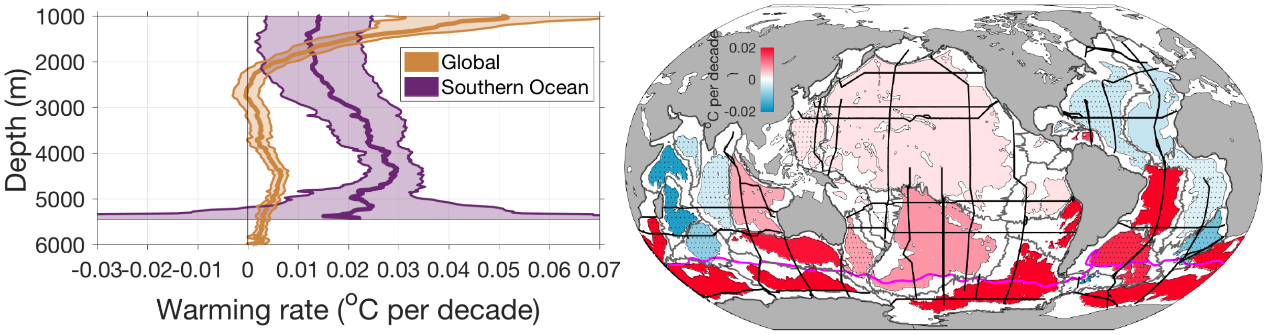


Figure. 5.2: Observed rates of warming from 1981 to 2018 (left panel) as a function of depth globally (orange) and south of the Sub-Antarctic Front or about 55S (purple) with 90% confidence intervals and (right panel) average warming rate (colors) in the abyss (below 4000 m) over various ocean basins (whose boundaries are shown in grey lines), with stippling indicating basins with no significant changes.

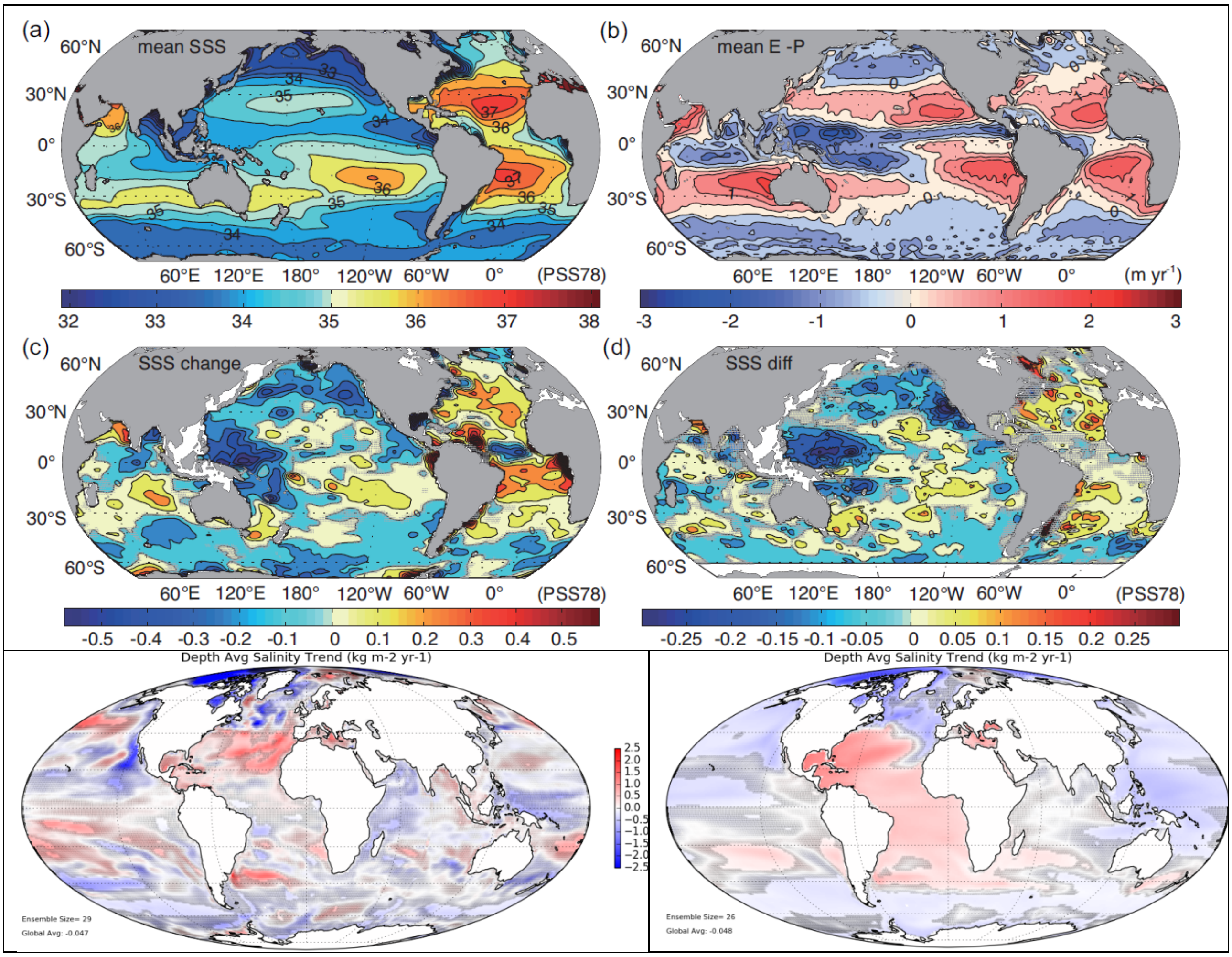


Figure 5.3: Salinity and water cycle (6 panels): (A) Mean Salinity (0–100 m); (B) Mean P-E; (C) Observed salinity changes (0–100 m; 1965–2015); (D) historical simulations (0–100 m CMIP5 mean; 1965–2015); (E) CMIP5 ensemble mean rates of salinity change integrated over the topmost 100 m for the present (2013 to 2017) – Historical (2000 to 2004); (F) CMIP5 ensemble mean projected 21st century rates of salinity change (2081 to 2100) minus (1986 to 2005) with the RCP8.5 forcing scenario integrated over the topmost 100 m. Panels (a,b,c and d) are from (Rhein et al., 2013), Figure 3.4 and 3.21]

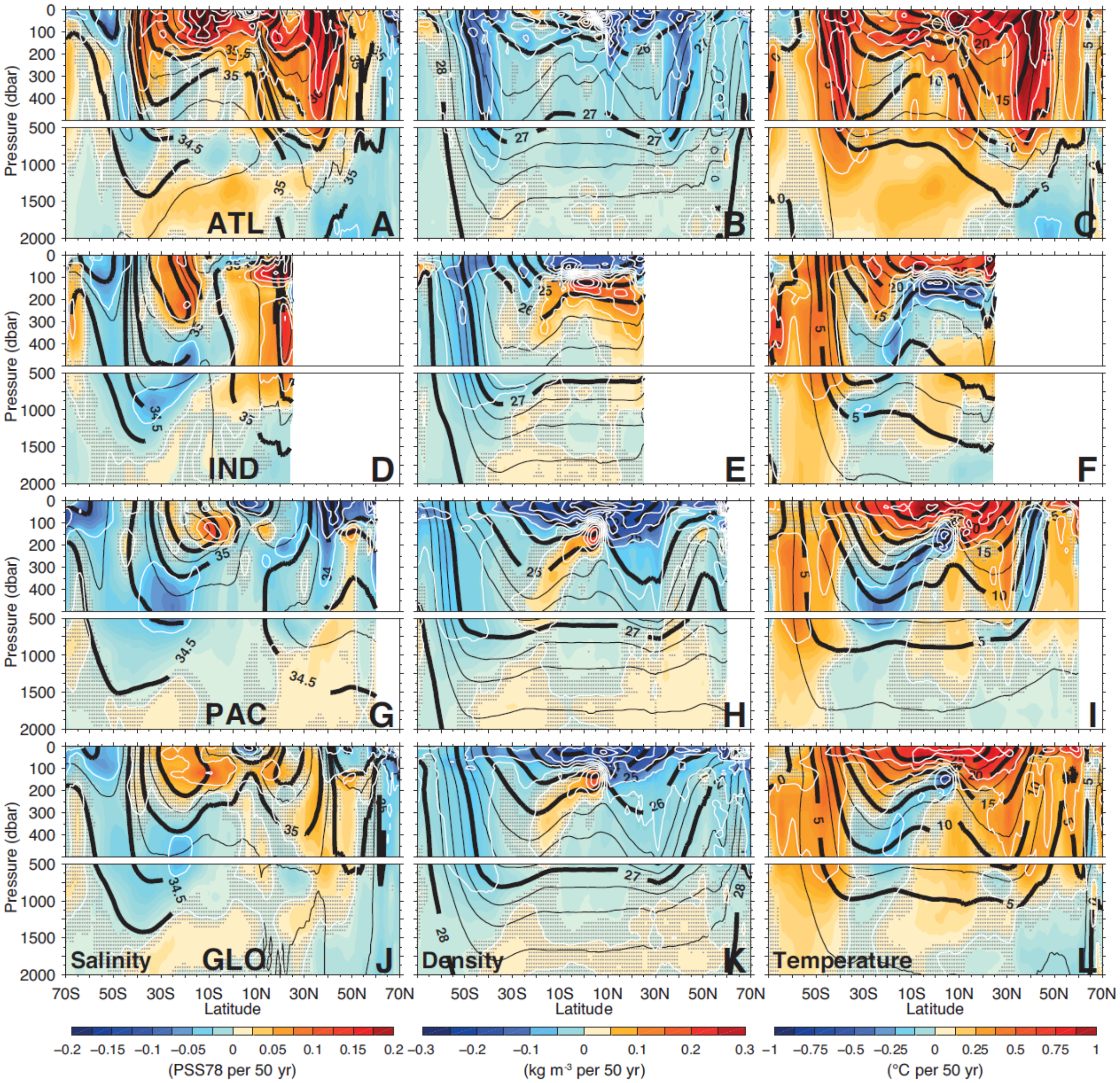
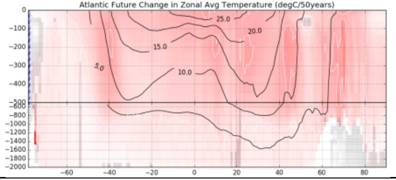
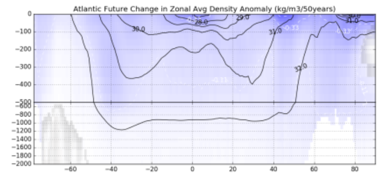
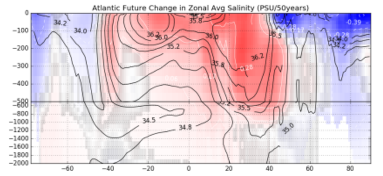
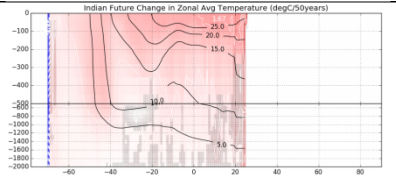
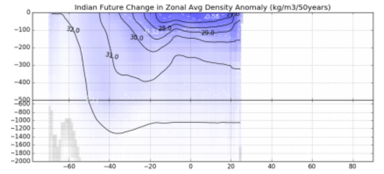
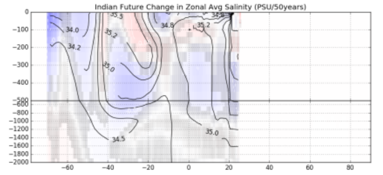


Figure 5.4: Trends from 1950 to 2000 of basin-averaged zonal-mean observed (1965–2015) watermass changes [change/50 years]: top-to-bottom (Atlantic, Indian, Pacific, Global); left-to-right (Temperature, Salinity, In-situ Density)]

	Temperature Change (°C/50 years)	Density Change (kg m^{-3}/ 50 years)	Salinity Change (PSU / 50 years)
Atlantic			
Indian			

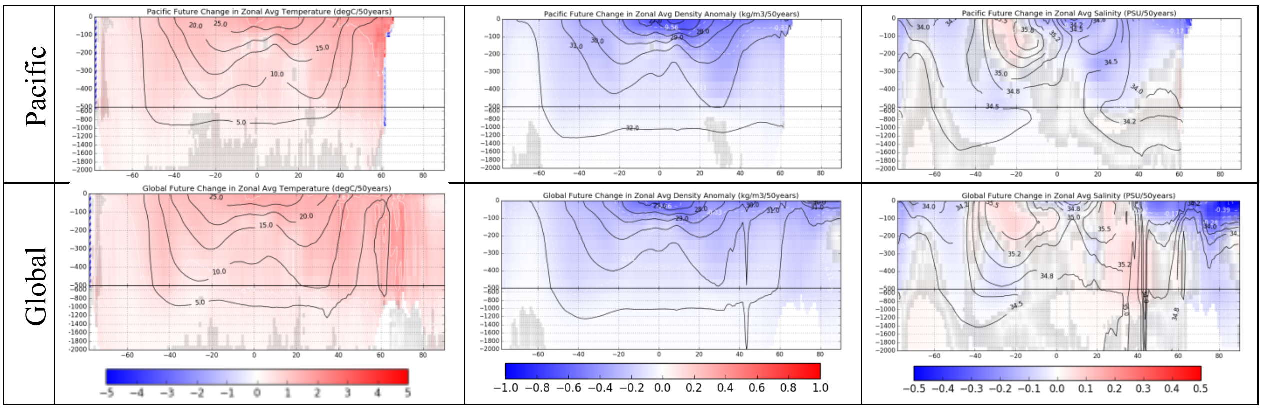


Figure 5.5: Side-view basin-averaged zonal-mean CMIP5 projected (to 2100) trends water-mass changes [change per 50 years] with RCP8.5 forcing: top-to-bottom (Atlantic, Indian, Pacific, Global); left-to-right (Temperature, In-situ Density, Salinity). Shaded areas show where the projected changes are not 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.

5.2.1.2.2 Salinity changes

Ocean salinity is a very important Essential Ocean Variable Bojinski et al. (2014) as key element of the Earth's water cycle affecting weather, climate and environmental systems (Stocker et al., 2013). Approximately 80% of the global water cycle occurs at the ocean-atmosphere interface (Trenberth et al., 2016). The spatial structure of the global ocean surface and subsurface salinity field is maintained by ocean circulation and mixing, which are driven by ocean density gradients and air-sea fluxes. At this interface, Sea Surface Salinity (SSS) responds to changing evaporation, precipitation and river runoffs patterns. It has long been noted that the climatological mean SSS and the surface Evaporation–Precipitation-River runoffs (E-P-R) flux field are highly correlated (Josey et al., 2013), which reflects the long-term balance between ocean advection and mixing processes and E–P-R fluxes at the ocean surface that maintain local salinity gradients (Durack, 2015).

The near surface salinity of the ocean is both observed and projected to evolve in ways that reflect the increased intensity of the Earth's hydrologic cycle (Figure 5.3) (Durack, 2015). The Clausius-Clapeyron relationship between air temperatures and saturation humidity exhibits a pronounced increase in the atmosphere's ability to hold water with warmer temperatures, roughly 7% more moisture for every degree of warming. In both observations and models, the relative humidity of the atmospheric boundary layer does not vary much to changes in temperature, so rates of evaporation, which generally scale with the under-saturation of the atmosphere (along with wind speeds) also increase with increasing temperature. Although there are atmospheric circulation changes with a changing climate, the overall circulation patterns, which carry moisture into areas of low-level convergence remain qualitatively similar; as a result areas that currently have net evaporation have more intense net evaporation in a warming climate while areas with net precipitation have more intense net precipitation (Held and Soden, 2006). Because the ocean's near-surface salinity patterns reflect a time-average of the net precipitation, those areas that are currently relatively salty tend to become more salty in a warmer climate, and conversely for relatively fresh areas (compare Figure 5.3 (a) and (c)). This pattern of surface salinity and its enhancement is clearly reflected in the observed basin-scale trends in salinity; oceanic salinity observations are the clearest observational demonstration of the intensification of the hydrologic cycle (Figure 5.3) (Rhein et al., 2013; Durack et al., 2016).

At longer timescales, the larger scale changes in the oean circulation and basin-integrated fresh water imbalances emerge in the near-surface salinity changes, with an increasingly salty tropical and subtropical Atlantic and Mediterranean contrasting with a freshening Pacific and polar Arctic emerging as robust signals across the suite of climate models (Fig. 5.2f). The freshening of the high latitudes in the North Atlantic and Arctic basin is consistent with the widely expected weakening of the Atlantic Meridional Overturning Circulation (discussed in Chapter 6) and a decline in the volume of sea ice (discussed in Chapter 3).

Deeper in the ocean, salinity changes reflect changes in the rates of formation of watermasses or their newly formed properties. Due to their various origins, ocean watermasses tend to be relatively fresh (like Antarctic

Bottom Water, Antarctic Intermediate Water or North Atlantic Intermediate water) or salty (like Mediterranean Water, Red Sea Overflow Water, or North Atlantic Deep Water) (Figure 5.4 and in projections Figure 5.5). Over time, as these watermasses are transported away from their sources in marginal seas or at the ocean's surface, these distinctive properties are diluted by mixing with ambient interior ocean waters. Changes in the interior ocean salinities can reflect either changes in the rates of formation of these watermasses or changes in the properties of their source (Figure 5.4). Thus, projected freshening of the Southern Ocean (Figure 5.3) leads to a freshening of the relatively fresh Antarctic Intermediate Water that is formed and subducted in the Southern Ocean, flowing northward at depths of 500-1500 m (Figure 5.5). Increased surface salinity in the Atlantic subtropical gyres are pumped into the interior by the winds, leading to an increased salinity of the interior subtropical gyres, along with contributions from increasingly salty Mediterranean water. Conversely, fresh-water capping of the northwestern North Atlantic (Figure 5.3) inhibits deep convection in the Labrador Sea and the consequent production of Labrador Sea Water in many models, and contributes to the increased salinity of the North Atlantic between 1000m and 2000m depths by the reduction of the signal from a relatively fresh watermass (Figure 5.5).

The ocean's properties are changing most rapidly in the near surface waters that are more immediately exposed to atmospheric forcing. As a result of the surface-intensified warming, the upper few hundred meters of the ocean are becoming more stably stratified (Helm et al., 2011; Talley et al., 2016). The combination of surface intensified warming and near-surface freshening at high latitudes leading to a projection of more intense near-surface stratification across all ocean basins (Figure 5.5) is a robust result with a *high* level of agreement across successive generations of coupled climate models (Capotondi et al., 2012; Bopp et al., 2013). By contrast, the bottom intensified warming in the abyss (see Figure 5.2) which is consistent with a slowing in the rate of bottom water formation, is also associated with a reduction in the abyssal stratification of the ocean (Johnson et al., 2014; Desbruyères et al., 2016b). Both of these changes have consequences for the evolving turbulence and ocean watermass structure. Based on observational evidence, theoretical understanding and robust climate model projections, it is *very likely* that stratification in the upper few hundred meters of the ocean but below the mixed layer will increase significantly in the 21st century over most ocean basins as a result of climate change, and abyssal stratification will *likely* decrease.

The dynamical consequences of increased stratification are understood with *very high confidence* (see, for instance, the textbooks by (Gill, 1982) or Vallis (2017)). The projected amount of increase (or decrease) in stratification will vary by region, with the time-horizon in question, between forcing scenarios, and to some extent across models, but for the purpose of illustration, consider the impacts of increasing stratification by 21% (chosen because the square root of 1.21 is 1.1). A 21% increase in stratification will increase the lateral propagation of internal Rossby waves (which set up the basin-scale ocean density structure) by 21% and increase the lateral propagation speed of internal gravity waves and boundary waves by 10%. A 21% increase in 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 by 10% (see section 5.2.1.2.4). For the same forcing, a 21% increase in stratification reduced the geostrophically balanced slope of density surfaces by 17.5% and the vertical excursions of internal gravity waves by 9.1%. For the same turbulent kinetic energy dissipation, a 21% increase in stratification reduces the turbulent vertical diffusivity of heat, salinity, oxygen and nutrients by 17.5% (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 increase stratification inhibits advective exchange between the surface and interior ocean, 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).

5.2.1.2.3 Changing eddies, ocean circulation and modelling uncertainties

The ocean circulation plays a major role in redistributing heat laterally and in regulating the exchange between the surface and interior ocean. The heat transported by the Atlantic Meridional Overturning Circulation (AMOC), in particular, has long been identified as source of the clement climate of northern Europe, although the wind-driven ocean gyres also contribute to the meridional ocean heat transport. As a result, there is a concern that significant changes in ocean circulation could lead to localized climate changes that are much larger than the global mean. Projected and observed changes in the AMOC and the rates of formation of deep water-masses in the North Atlantic are discussed in Chapter 6, along with the possibility of abrupt or enduring changes as a result of forcing from Greenland. Climate-change induced changes of the circulation in other mid-latitude basins may be difficult to detect or reliably project because of significant natural variability at inter-annual (e.g., El Niño) to decadal (e.g., the Pacific Decadal Oscillation) timescales.

The Southern Ocean is believed to be relatively important for the uptake of heat and carbon by the ocean, even beyond what would be expected given its vast size (Rhein et al., 2013); Frölicher et al. (2015). The circulation in the Southern Ocean is comprised of an eastward flowing mean current characterized by strong small scale transient features, known as jets and eddies. The mean flow circumnavigates Antarctica as a series of sinuous, braided jets that, taken together, form the world's largest ocean current, the Antarctic Circumpolar Current (or ACC). The ACC transports approximately 173.3 ± 8.7 Sv (Donohue et al., 2016), driven by a combination of strong westerly winds and production of dense water near Antarctica.

Trends in the Southern Ocean are dominated by an increase in westerly winds in recent decades. However, there is no evidence that this enhanced wind stress has altered the Antarctic Circumpolar Current (ACC) transport, which appears to be remarkably stable (Koenig et al., 2014). Theoretical predictions and high-resolution ocean modelling suggest that the insensitivity of the ACC to changes in wind stress is a consequence of eddy saturation (Munday et al., 2013), whereby additional energy input from stronger winds cascades rapidly into the fine-scale eddy field. Thus, rather than enhancing the mean flow, wind stress increases are expected to alter the strength of the eddy field. Satellite measurements of Southern Ocean eddy kinetic energy over the last two decades are consistent with this expectation, showing a statistically significant upward trend (Hogg et al., 2015).

While mesoscale eddies (geostrophic rotating vortices with spatial scales of 10-100 km that penetrate deeply into the water column, and are often described as the ocean's weather, analogous to the midlatitude atmosphere's high and low pressure systems) play an important role in regulating the larger scale ocean circulation, sub-mesoscale eddies (surface intensified, rotationally influenced motions with smaller horizontal scales of 100s of m to about 10 km and intrinsic timescales of a few days that especially arise in association with fronts in the ocean's surface properties) are known to be particularly important in the dynamics of the near-surface ocean boundary layer (see the review by (Mahadevan, 2016)). Submesoscale instabilities are associated with restratifying overturning circulations that can limit the thickness of the well-mixed ocean surface boundary layer near fronts Bachman et al. (2017). Moreover, submesoscale motions generate strong vertical velocities that drive fluxes of nutrients from the interior ocean into the euphotic zone or create pockets of reduced mixing with increased phytoplankton residency time within the euphotic zone (Lévy et al., 2012), so that the submesoscale structures are often readily visible in images of ocean colour. Intense mesoscale eddies are known to create favourable conditions for submesoscale instabilities both observationally (Bachman et al., 2017) and in numerical studies (Brannigan et al., 2017). Intensifying Southern Ocean eddy fields will *likely* have a significant impact on biological productivity, ecosystem structure, and carbon uptake, both directly and via submesoscale processes.

Future projections for the Southern Ocean are dominated by the potential for a continued increase in the westerly winds (Bracegirdle et al., 2013), as well as a combination of warming and increased freshwater input (Downes and Hogg, 2013). If further increases in the westerly winds are sustained, as is widely expected from climate model projections, then it is *very likely* that the eddy field will continue to grow, with potential consequences for the overturning circulation and transport of tracers. However, the magnitude of the mean ACC flow will *likely* remain insensitive to winds; model projections suggest that freshwater input from the melting of Antarctica may instead reduce the strength of the ACC in the coming decades, by reducing the rate of formation of the densest waters around Antarctica (Hogg et al., 2015).

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

5.2.1.2.4 Tides in a changing climate

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

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

5.2.1.2.5 Ocean turbulence in a changing climate

Ocean turbulent mixing is a key process regulating the ocean circulation and climate. Turbulent mixing is important for the uptake and redistribution of heat, carbon, nutrients, oxygen and other tracers in the ocean. Due to the importance of turbulent mixing on the lateral and vertical distribution of ocean properties, it is anticipated that ocean mixing changes can have large impacts on marine ecosystems and on the rates of heat and carbon uptake by the ocean (Schmittner et al., 2009); MacKinnon et al. (2017). It has become increasingly clear that turbulent mixing in the ocean is not constant, either in space or time. Global estimates of both the turbulent kinetic energy dissipation rate and the diapycnal diffusivity, two measures of ocean turbulence, vary over several orders of magnitude throughout the ocean (Figure 5.6), (Polzin et al., 1997; Waterman et al., 2012; Whalen et al., 2012; Alford et al., 2013; Hummels et al., 2013; Sheen et al., 2013; Waterhouse et al., 2014; Kunze, 2017). Turbulent mixing both dissipates energy and diffuses tracers across isopycnals (Fischer et al., 2013). For a given energy dissipation rate, the turbulent diffusivities of heat, salinity, nutrients and other tracers tend to be smaller with stronger stratification. This dependency on stratification helps explain why the observationally inferred diffusivity in the heavily stratified main thermocline (250-1000 m depth; Figure 5.6a) is of similar magnitude to those deeper in the water colmn

(Figure 5.6c), while the turbulent energy density and dissipation rate are much strong at the shallower depths (Figures. 5.6b and 5.6d). Consistent global patterns in the variability of turbulent mixing are found using a range of techniques, including microstructure measurements, tracer release experiments, and indirect inference techniques, reflecting global patterns of the underlying physical processes that produce turbulence (Ledwell et al., 2010; Waterhouse et al., 2014). Turbulence also fluctuates in time, modulated by tidal cycles (Klymak et al., 2008) and seasonal changes (Sloyan et al., 2010; Whalen et al., 2012). In the mixed layer and directly below, turbulence changes according local conditions such as the winds, heating rates and local stratification (Sloyan et al., 2010; Moum et al., 2013; D'Asaro, 2014; Tanaka et al., 2015) at diurnal to seasonal and longer timescales; these variations in near-surface turbulence must be taken into account for climate models to reproduce the observed seasonal cycle of surface properties and spatial structure of the depth of the thermally well-mixed near surface layer of the ocean. The dramatic spatial and temporal patterns of ocean turbulence help shape ocean tracer distributions (heat, dissolved greenhouse gasses, nutrients) and will regulate the oceanic manifestations of a changing climate.

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; Alford et al., 2016; Melet et al., 2016; Meyer et al., 2016; Zhao et al., 2016b).

Many of the 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 Annual Mode is expected to intensify as a result of climate change (Young et al., 2011; Jones and Cheung, 2015), bringing with it stronger winds, and more wind-energy input over most of the Southern Ocean and a more intense mesoscale eddy field (Hogg et al., 2015). These intensified eddies will energize the internal wave field where they flow over rough bathymetry, in turn increasing topographically generated mixing in the Southern Ocean (Melet et al., 2015). Changes in the near-bottom stratification will alter the rate that the barotropic tides generate internal waves, thereby altering the strength and distribution of the tidally generated mixing. Given the sparsity of the ocean observing system and the large spatial and temporal variability of ocean mixing, climate-change related changes in the overall levels of and patterns ocean turbulence are only just now beginning to be documented observationally (Dosser and Rainville, 2016).

Despite the difficulties with observationally documenting ongoing changes in turbulent mixing, several of the robustly projected changes in the state of the large scale ocean will directly impact our expected changes in ocean turbulence over the coming century. The projected more intense Southern Ocean winds forcing and mesoscale eddies (discussed previously) are *very likely* to lead to more intense turbulent vertical mixing (by all metrics) in the Southern Ocean. The increased stratification in the tropics and subtropical gyres that have been noted in section 5.2.1.2 as a *very likely* consequence of surface-intensified climate change are *likely* to lead to a net reduction in the vertical diffusivities of nutrients and other gases within the main thermocline, reducing the flux of nutrients into the euphotic zone and increasing the contrast in oxygen concentrations between the near surface ocean and the interior. Dramatic seasonal reductions in Arctic sea ice are *virtually certain* to increase wave-driven mixing in the upper Arctic ocean and at rough topography surrounding the Arctic ocean basin.

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

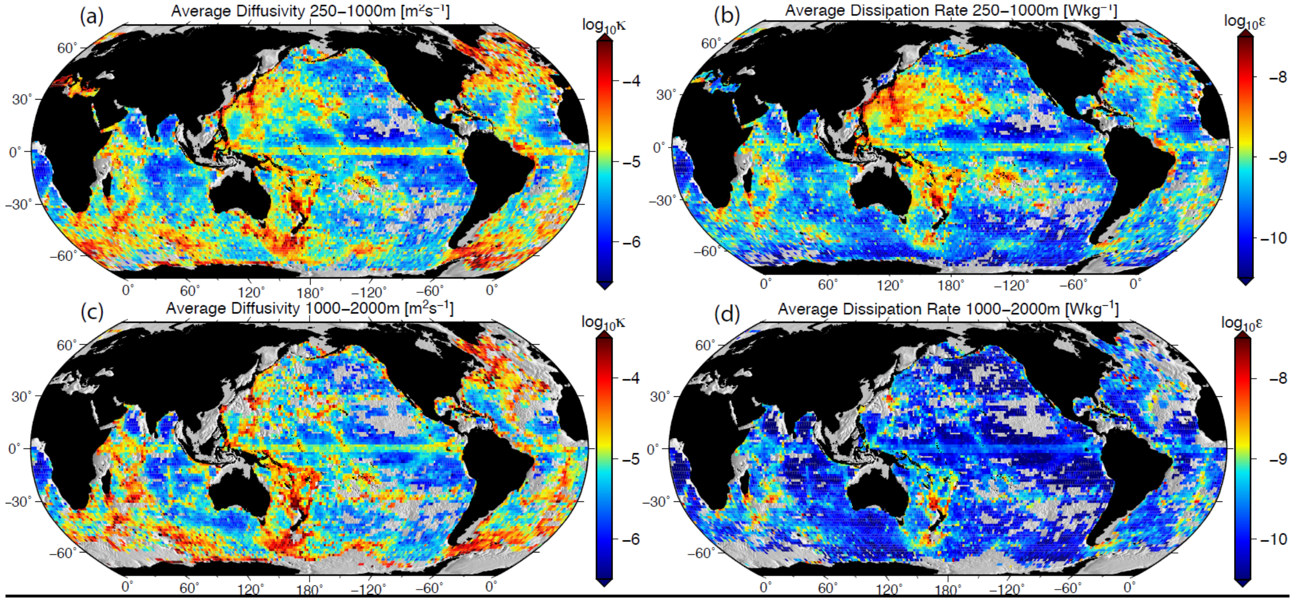


Figure 5.6: Estimates of the average (b, c) diapycnal diffusivity, and (b, d) turbulent energy dissipation rate between (a, b) 250–1000 m, and (c, d) 1000–2000 m calculated by applying fine structure techniques to Argo float data. Only bins with at least three estimates are plotted and regions with insufficient data are coloured grey. All panels are using data updated from (Whalen et al., 2012).

5.2.1.3 Changes in Ocean Carbon

The latest analyses of community datasets and ocean models demonstrate that ocean carbon uptake has increased since 2000, in broad agreement with the atmospheric carbon dioxide growth rate (*high confidence*) (Figure 5.8a) Rödenbeck et al. (2015); (Landschützer et al., 2016). However, observations show a substantial degree of variability at inter annual and decadal scales that was not understood during AR5, which is not captured by models– most notably variations in the tropics linked to ENSO variations in ocean circulation (Landschützer et al., 2016) and consensus around the weakening and strengthening of the Southern Ocean sink that has been attributed to changes to wind patterns (Landschutzer et al., 2015; Landschützer et al., 2016) (Figure 5.8a). The variability in model simulations of global ocean carbon uptake is discussed by (Lovenduski et al., 2016).

Previous efforts based on data synthesis have estimated that the ocean took up around 106±17 Pg C between 1850 and 1994 Sabine et al. (2004) and steady state models suggest the ocean had taken up between 160-166 PgC by 2012 (across different iterations of ocean circulation assumptions, (DeVries, 2014)), similar to other model estimates benchmarked to earlier eras Khatiwala et al. (2009); (Khatiwala et al., 2013) and the AR5 estimate of 155 Pg. Overall, this provides high confidence that the ocean has continued to act as an important sink for anthropogenic carbon over the past few decades, notwithstanding variability at regional and global scales that are linked to the underlying changes in ocean physics (Landschutzer et al., 2015; Landschützer et al., 2016).

Carbon speciation is key to understanding how ongoing ocean carbon uptake drives acidification and impacts on ecosystems. The speciation of carbon is defined as the differentiation of the bulk dissolved inorganic carbon (DIC) pool into its constituent parts; the underlying chemistry is extensively reviewed elsewhere (Zeebe and Wolf-Gladrow, 2001). The three forms of DIC are bicarbonate (HCO_3^-), carbonate (CO_3^{2-}) and aqueous carbon dioxide ($CO_{2(aq)}$), which are typically found at a ratio of 100:10:1. These three inorganic carbon species are coupled via a set of reversible reactions known as the ‘buffer system’ and their relative abundance is strongly controlled by ocean pH. For example, if the ocean pH lowers (or acidity increases) by one pH unit from its typical value in seawater of 8.1 (to 7.1), then $CO_{2(aq)}$ becomes more abundant, while CO_3^{2-} concentrations are reduced (HCO_3^- is not very sensitive at these pH levels). A further implication of the coupled carbon speciation reactions is that as the ocean absorbs more CO_2 from the atmosphere, the pH drops, driving a concomitant change in carbon speciation.

Ecosystems are sensitive to specific aspects associated with carbon speciation in response to ocean acidification. As reviewed in IPCC AR5 and elsewhere (Aze et al., 2014; Howes et al., 2015), photosynthesising organisms may respond favourably to the increasing availability of dissolved $CO_{2(aq)}$. Calcifying organisms are generally negatively affected by changes in pH and carbonate concentrations, although increasing bicarbonate could, in theory, promote calcification (Roleda et al., 2012). Reduced carbonate levels affect the saturation state of calcite and aragonite minerals (Ω_{CAL} and Ω_{ARAG}), causing dissolution of unprotected calcium carbonate structures, e.g., coral exoskeletons, in undersaturated water (Orr et al., 2005; Takahashi et al., 2014). Habitat and taxon-specific impacts of ocean acidification are considered in greater detail in 5.2.2 and 5.2.3 below.

Analyses of direct pH trends from ocean time-series and derived pH changes from shipboard studies show consistent decreases in pH and the saturation state of aragonite and calcite have occurred over the past few decades (*high confidence*). The largest changes are observed from time-series in the Irminger Sea (-0.0026 yr^{-1}) and the Cariaco Basin (-0.0025 yr^{-1}), with the longest available time series in Bermuda and Hawaii (spanning >30 and >25 years, respectively) showing similar trends of $-0.0016/7$ yr^{-1} (Bates et al., 2014). At larger spatial scales (e.g., for ocean basins), assessing temporal trends in pH require combining shipboard observations of the fugacity of CO_2 with estimates of ocean alkalinity (Takahashi et al., 2014; Lauvset et al., 2015). Over the period 1991–2011, mean ocean pH (seawater scale) has declined by 0.0018 ± 0.0004 yr^{-1} in 70% of ocean biomes, with the largest declines in the Indian Ocean (-0.0027 yr^{-1}), eastern Equatorial Pacific (-0.0026 yr^{-1}) and the South Pacific subtropical (-0.0022 yr^{-1}) biomes, with lower rates of change in the Atlantic and Southern Oceans (Lauvset et al., 2015). In contrast, a local analysis finds the northern Drake Passage region displays large trends in pH over the 2002–2012 period of 0.0023 ± 0.0007 yr^{-1} (Takahashi et al., 2014). In general, trends in the saturation state of aragonite and calcite follow those of pH, with high latitude regions most vulnerable to under-saturation as they naturally have lower mean values (Takahashi et al., 2014). There is observational and theoretical evidence (*medium confidence*) that seasonal variability in pH has increased, and will continue to do so with increasing atmospheric CO_2 (Landschützer et al., 2018).

In the observations compiled by (Lauvset et al., 2015), pH trends are primarily attributed to the rising DIC levels due to the increased flux of anthropogenic carbon into surface waters from the atmosphere. These lowered pH waters are being transferred into the ocean interior by overturning circulation in the north and south Atlantic Rios et al. (2015). At both the surface and in the ocean interior, natural variability is noted as an important local player alongside secular changes (Lauvset et al., 2015; Rios et al., 2015). For example, the seasonal cycle in carbon speciation variables (pH, Ω_{CAL} and Ω_{ARAG}) is equivalent to around 10 years of the long-term trend at some sites (Takahashi et al., 2014; Sutton et al., 2016). Changes in alkalinity and in particular the ratio of alkalinity to DIC may be particularly important in driving regional trends in Ω_{CAL} and Ω_{ARAG} (Lenton et al., 2012; Takahashi et al., 2014).

Over the next century, pH is projected to decline by 0.07–0.31 pH units for RCP2.6 and 8.5 scenarios, respectively (Bopp et al., 2013; Gattuso et al., 2015), with good agreement across the range of CMIP5 models (*high confidence*) (Figure 5.7a). Projected changes in pH show relatively weak spatial variations, but they are greatest in the Arctic Ocean and the high latitudes of the Atlantic and Pacific Oceans due to their lower buffering capacity and are lowest in contemporary upwelling systems (Figure 5.7b). Other ocean regions show changes similar to the global average. Similar declines in the concentration of CO_3^{2-} ions is predicted by the CMIP5 models (Bopp et al., 2013; Gattuso et al., 2015), with high latitude and Arctic Ocean regions first to be undersaturated due to their systemic vulnerability (Orr et al., 2005). Seafloor changes in pH over the next century are highly localised, with over 20% of the North Atlantic sea floor deeper than 500m experiencing pH reductions greater than 0.2 units by 2100 under the RCP8.5 scenario, including canyons and seamounts designated as marine protected areas (Gehlen et al., 2014a). Changes in pH in the abyssal ocean (>3 km) deep, are greatest in the Atlantic and Arctic Oceans, with lesser impact in the Southern and Pacific Oceans mainly due to circulation timescales (*high confidence*) (Sweetman et al., 2017). Model skill is generally good for carbon speciation variables, such as pH, thus increasing confidence in their projections (Bopp et al., 2013; Frölicher et al., 2016) (Figure 5.7a).

Anthropogenic trends in carbon speciation variables (pH, Ω_{CAL} and Ω_{ARAG}) emerge from the background variability by 2010 or 2020 for the tropical ocean or regions of low and high latitude upwelling, respectively (Keller et al., 2014; Rodgers et al., 2015). As the trends in pH can be observed to emerge with a median of 15 years of data is needed for the emergence of a pH trend (Henson et al., 2016), there is high confidence that the surface ocean has already transitioned to a new state, where pH levels are consistently lower than the standard deviation of the previous state. For Ω_{ARAG}, the anthropogenic signal has already emerged from the background over much of the ocean (Rodgers et al., 2015). By the end of century, the anthropogenic pH signal is projected to have emerged over the entire ocean (Frolicher et al., 2016). While internal variability and model uncertainty are most important for the next few decades, the largest source of uncertainty in the projection for end-of-century pH change (Figure 5.7c) at both global and local scale is the emissions scenario (Frolicher et al., 2016), with large changes in the trajectory and magnitude of pH changes between the RCP8.5 and RCP2.6 scenarios (Figure 5.7a and c).

5.2.1.4 Changing Ocean Oxygen

Ocean oxygen (O_2) levels result from the balance between oxygen production during photosynthesis, consumption during respiration and redistribution by the ocean circulation and mixing, with temperature controlled equilibrium with the atmosphere important for surface waters. Due to the localisation of photosynthetic activity and relatively short timescales of atmospheric equilibrium, surface waters are close to atmospheric equilibrium and relatively high in oxygen. Organic material aggregates into particles and sinks out of the euphotic zone, so the sink of O_2 from respiration dominates in the interior ocean. Oxygen minimum zones (OMZs) arise in the interior ocean where the circulation cannot carry in oxygen enriched waters to meet the respiratory demand, especially in the shadow zones in the eastern sides of tropical oceans where the time-mean advective pathways of the circulation are blocked by topography rather than originating in the directly ventilated surface ocean.

Using ocean oxygen datasets going back the 1960s, it is estimated that the oxygen content of the global ocean has declined (*high confidence*). The most recent study estimates oxygen levels have dropped by 961 ± 429 Tmol per decade, or by 2% in total (Schmidtko et al., 2017). By region, the Equatorial Oceans that contain the lowest oxygen waters, North Pacific, Southern Ocean and South Atlantic show the strongest trends (from 210 to 119 Tmol decade–1), while the Arctic Ocean emerges as a regional hotspot (Schmidtko et al., 2017). Syntheses of datasets from specific time series document stronger trends, with oxygen declines of over 20% at sites in the Northeastern Pacific and the California Current (Levin, 2017). Despite holding the highest inventory of oxygen in the ocean, oxygen levels in Southern Ocean contributed 25% to the global decline between 1970–1992 (Helm et al., 2011) and have fallen by over 150 Tmol per decade since the 1960s (Schmidtko et al., 2017). Repeat occupations of various ocean sections as part of the CLIVAR programme have documented broad thermocline oxygen declines in the northern hemisphere oceans, accompanied by oxygen increases in subtropical and southern hemispheres (Talley et al., 2016). These large scale changes have been attributed to human induced climate change (Andrews et al., 2013).

Around 15% of the global ocean loss of oxygen, 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 attribute due to the lack of clear proxies and their often overlapping nature. For example, while it is likely that increasing ocean stratification is driving an important part of the decline in oxygen (Talley et al., 2016), 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 (Bograd 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).

Over the next century oxygen declines are predicted by all CMIP5 models, projecting around 3.5% loss of total ocean oxygen by 2100 (*medium confidence*) due to the combination of warming and increased stratification (Bopp et al., 2013). Increased tropical ocean stratification reduces interior ocean oxygen by flattening the advective pathways of ventilation in the subtropical gyres (Luyten et al., 1983) and by inhibiting turbulent mixing with the oxygen-rich surface ocean (see section 5.2.1.2.5).This relatively robust global modelled trend (Figure 5d) however masks important uncertainty in the projection of regional trends (Figure 5.7e), particularly in the tropical ocean OMZs (Bopp et al., 2013; Cocco et al., 2013; Cabré et al., 2015). In these regions, an overall projection of oxygen increases (*low confidence*) is more uncertain as the inter-model standard deviation is over four times greater than the projected trend (Bopp et al., 2017). The uncertainty in the trends in tropical ocean OMZs arises due to the fact that oxygen depletion due to warming induced reductions in oxygen saturation are opposed by oxygen enrichment due to reduced oxygen consumption during respiration in response to predicted declines in marine export production (Bopp et al., 2017). Since dissolved oxygen emerges as a residual of these two opposing mechanisms (warming and export production), it can show substantial variability on decadal timescales (Ito and Deutsch, 2013). At the seafloor, bathyal (200–3000 m depth) regions of the North Pacific, North Atlantic, Arctic and Southern Oceans may see oxygen declines by 0.3 to 3.7% by 2100, while abyssal ocean changes are lower and localised around regions in the North Atlantic and Southern Ocean affected by deep water formation (Sweetman et al., 2017). Simulations extended to 2300 suggest that by 2150 the trend of declining tropical ocean oxygen (both in terms of concentrations and volume of low oxygen waters) may reverse itself, mainly due to the effect of strong declines in primary production and organic matter fluxes to the ocean interior (Fu et al., 2018). Model skill assessment demonstrates a significant degree of inter-model variability in the reproduction of global oxygen fields (Bopp et al., 2013).

A median of just over 25 years of data is required for an interior ocean oxygen trend (average concentration over 200-600 m depth) to emerge from the natural background, but more than 40 years data is needed in certain more localised highly dynamic regions (Henson et al., 2016). By 2100, the anthropogenic signal in interior ocean oxygen is projected to have emerged above uncertainties associated internal and inter-model variability for around 25% of the ocean area in this depth range (Frolicher et al., 2016)(Box 5.1). As seen for pH, the scenario uncertainty is an important component of the total uncertainty in projection of interior ocean oxygen by 2100 (Figure 5.7f), but the contributions of internal variability and model uncertainty dissipate more slowly with time and remain important, especially at more regional scales, by 2100 (Frolicher et al., 2016). Differences between scenarios remain important for oxygen, relative to the model uncertainty (Figure 5.7d and f)

5.2.1.5 Changing Ocean Nutrients and Primary Production

Nutrients support the base of marine food webs and their levels in different parts of the ocean result from the interplay of external supply (e.g., from rivers or dust) and internal cycling (consumption by primary producers and replenishment during degradation of organic matter). Nutrients that are remineralized in the interior ocean need to be advected or mixed back into the surface ocean before they can again support phytoplankton growth. As such, their projected trends and changes integrate modifications to ocean biology, chemistry and physics and place ultimate constraints on trends on the production of particulate organic matter by phytoplankton primary production. This particulate organic carbon production is an important component of ocean biogeochemical cycling and atmospheric CO_2 uptake as well as being a source of food for both pelagic and benthic ecosystems.

Biological activity by phytoplankton and particulate organic matter production in the surface ocean is known to be affected by both the magnitude and ratio of nutrient supply, particularly in terms of the different components of the phytoplankton community (*high confidence*). On the one hand, the largest rates of phytoplankton activity occur in regions where nutrient supply is maximised by upwards mixing of deep nutrient reserves (Falkowski, 1998) (Figure 5.7). On the other hand, the distribution of specific forms of plankton is closely linked to the match between the relative supply of different resources and the their requirements (Hagstrom et al., 2017), which themselves can be strongly sensitive to warming (Yvon-Durocher et al., 2015; Yvon-Durocher et al., 2017). For example, the competitive success of nitrogen fixing diazotrophic plankton in the future ocean will be controlled by warming (Fu et al., 2016), the relative supply of iron, nitrogen and phosphorus (Ward et al., 2013).

Nutrient changes in the surface ocean reflect modifications to physical supply by mixing and upwelling, as well as external input, typically from dust and internal cycling. In the coastal ocean, the largest influence comes from ongoing increases nutrient loading due to riverine inputs (Seitzinger et al., 2010; Beusen et al., 2016; Van Cappellen and Maavara, 2016; Jickells et al., 2017) (*medium confidence*). In the open ocean, increasing stratification is occurring in the major tropical regions (Dave and Lozier, 2013; Talley et al., 2016; Kwiatkowski et al., 2017), which is likely (*medium confidence*) to retard nutrient supply to the surface by suppressing upwelling and turbulent mixing (see section 5.2.1.2.5). Increasing inputs of anthropogenic nitrogen from the atmosphere are perturbing ocean nutrient levels (Jickells et al., 2017). In the western Pacific in particular, additional atmospheric nitrogen input has enriched the system in nitrogen (*medium confidence*), which raises the nitrogen to phosphorus ratio and induces a progressive shift towards phosphorus limitation in this region (Kim et al., 2014). Enhanced microbial turnover of dissolved organic nitrogen and phosphorus (DON and DOP, respectively) may act to counter balance reduced vertical supply (*low confidence*), but this requires the bulk of microbial turnover to occur in the upper ocean (Letscher et al., 2013; Letscher and Moore, 2015). Microbial turnover of DON and DOP in the upper ocean depends critically on their lability (Banse, 1968; Letscher et al., 2013) and the presence of the necessary metal cofactors for the hydrolysing enzymes (Mahaffey et al., 2014; Browning et al., 2017a).

Previously, our conceptual understanding of how changing resource availability would affect primary production and the generation of organic matter emphasised the so-called major nutrients (nitrogen and phosphorus, as well as silica for diatoms specifically) and ignored the role of trace micronutrients (Tagliabue et al., 2017). However, in recent years a new understanding of resource limitation in the ocean has emerged from diverse studies (including shipboard experiments and use of protein biomarkers) that highlights nitrogen and phosphorus limitation in the stratified tropical ocean regions accompanied by widespread iron limitation at high latitudes and in upwelling regions that typically have high rates of productivity (*high confidence*, Figure 5.8) (Moore et al., 2013; Saito et al., 2014; Browning et al., 2017b; Tagliabue et al., 2017). Moreover, more extensive experimental work has demonstrated nitrogen-iron colimitation in at the boundary of the south Atlantic gyre (Figure 5.8) that is linked to the common occurrence of simultaneous drawdown of nitrogen (as nitrate) and iron levels, driving greater microbial diversity (Browning et al., 2017b). These findings echo the substantial overlap in nutrient stress biomarkers seen in transition between upwelling and gyre systems in the Pacific (Saito et al., 2014).

Much effort has been made to extract information on the trends in organic matter production over the recent past using satellite datasets and time series sites. Globally, a range of studies, mostly based on satellite datasets have all reported overall declines in open ocean chlorophyll (as a proxy for phytoplankton biomass) of < 1% yr^{-1} that are attributed to enhanced stratification (*low confidence*), although many estimates report global trends closer to zero (Boyce et al., 2014; Gregg and Rousseaux, 2014; Boyce and Worm, 2015; Hammond et al., 2017) and there is no information on how biomass consumption (e.g., by zooplankton) may have changed. For regional scales, the overall low decline estimated globally may mask larger changes at regional scales (*low confidence*). For instance, regional analyses of satellite datasets suggest stronger (e.g., up to ±2% yr^{-1}) or non-significant trends in chlorophyll *a* (Hammond et al., 2017). In general, the length of the time series plays an important role in the magnitude of the phytoplankton trend derived, with trends shorter than the typical natural variability timescale of ~30 years (Beaulieu et al., 2013) producing much larger estimates of change (Boyce and Worm, 2015). A synthesis of trends from a variety of methods found nearshore waters mostly displayed increases in phytoplankton abundance, attributed to enhanced nutrients input from run off (Boyce et al., 2014). At low latitudes, as the maxima in organic matter production occurs away from the surface layer observed by satellite records, with observed organic matter production trends not reproduced by satellite records (Saba et al., 2010) and satellite records are not yet long enough to unambiguously isolate long term trends from natural variability (Beaulieu et al., 2013).

Upper trophic levels also respond to changes in the timing of phytoplankton bloom, known as the phenology, with historical datasets demonstrating trends of around 5-10 days per decade (Poloczanska et al., 2016)(Section 5.3.1). Attributing a climate signal to changes in modelled changes in phenology at locations where observing systems exists suggests time series of between 20-90 years would be required (*low confidence*), although if data can be averaged over larger spatial scales then trends are easier to detect (Henson et al., 2017).

In general, CMIP5 models project a small drop in global productivity and organic matter production (*medium confidence*) by 2100 from the RCP8.5 scenario that arises from a balance between increases in high latitude (*low confidence*) and decreases in low latitude (*medium confidence*) (Bopp et al., 2013). Globally, CMIP5 models project overall declines in upper ocean nitrate levels (Figure 5.7g, *high confidence*) that are localised in tropical upwellings, subpolar gyres and other frontal systems (Figure 5.7h). Globally, net primary productivity and organic matter production trends mirror the decline in nutrients (Figure 5.7j). The apparent virtual compensation in the effect of primary production and organic matter production is underpinned by substantial spatial and inter-model heterogeneity in the projected changes (Figure 5.7k). For example, in the tropical ocean a decline of 11 ± 24% in organic matter production is projected across the suite of CMIP5 models by 2100 under RCP8.5 (Laufkötter et al., 2015). However, this large inter-model range can be restricted to a projected decline of 11 ± 6% if emergent constraints from the historical record that link variability productivity to temperature anomalies are used, a four-fold decline in inter-model uncertainty (Kwiatkowski et al., 2017). Via this emergent constraint approach, there is a projected decline in global marine particulate organic matter production (*medium confidence)* of 6 ± 3% (Kwiatkowski et al., 2017). Projected declines in organic matter production in different biomes across CMIP5 models are generally attributed to increased stratification that reduces in surface nutrient concentrations in low latitude biomes (Cabré et al., 2014; Laufkötter et al., 2015; Fu et al., 2016) (Figure 5.7e), but in some models the temperature enhancement of growth is the dominant driver (Laufkötter et al., 2015), whereas bottom up controls by iron and light drive the response in the Southern Ocean (Laufkötter et al., 2015; Leung et al., 2015). On multi-century timescales, it is projected that enhanced physical-biogeochemical feedbacks act to supress biological activity for a millennium (Moore et al., 2018). This arises due to the trapping of essential nutrients in the ocean interior in response to changes to winds, sea ice and ongoing warming, which exacerbates the declining trend found to 2100 (Figure 5.10).

In the surface ocean, the progressive decline in upper ocean major nutrients (nitrogen and phosphorus) with increasing stratification will affect the food quality available upper trophic levels via its impact on the C:N:P ratio of organic matter (*high confidence*). Projections with variable stoichiometry climate models suggest that in oligotrophic regions and the Arctic Ocean food quality will decline over 20% due to changing phytoplankton nutrient ratios (Kwiatkowski et al., 2017) affecting food supply to upper trophic levels. Moving to the ocean interior, changes in organic matter flux from surface waters or export production are generally closely linked to changes in organic matter production in the surface layer that are exported deeper. In general, CMIP5 models predict declines in export production (*medium confidence*) of around 1–20% from 2000–2100 for the RCP8.5 scenario (Bopp et al., 2013; Fu et al., 2016; Laufkötter et al., 2016). Models with a more detailed representation of the marine food web tend to predict a stronger decline in export production than total organic matter production, highlighting the role of food web changes in driving changes to organic matter supply to the ocean interior (Fu et al., 2016). Parallel to these declines in export production, the flux of organic material to sea floor is also predicted to decline globally (*medium confidence*) with increases projected only in the polar regions (Sweetman et al., 2017). The projected changes in export production are attributed to both the magnitude of organic matter production in the surface ocean, but also by how the food web structure modulates the 'transfer efficiency' of particulate organic material, which affects the sinking speed and lability of exported particles (Fu et al., 2016).

Confidence in model projections can be appraised using the model skill in reproducing distributions of the limiting nutrient in specific ocean regions (Figure 5.8). Thus, relatively high model skill in reproducing surface distributions of nitrate and phosphate (Laufkötter et al., 2015), raises confidence in projections in tropical gyre systems. In contrast, poor model skill in reproducing iron dynamics (Tagliabue et al., 2016) lowers confidence in projections in the higher productivity regions where it is the limiting nutrient (Figure 5.8). Despite apparently high skill in the reproduction of bulk nitrate and phosphate fields, skill is reduced when underlying mechanisms (Duteil et al., 2012) or the ratio of resources is assessed. Indeed, properly representing the ratio of different resources correctly has been shown to have significant impacts on the response of organic matter cycling to environmental change (Moreno et al., 2017). Model skill in reproducing patterns of organic matter production and export range from poor to average (correlation coefficients of 0.1-0.6 across different models (Laufkötter et al., 2016; Moreno et al., 2017), but it should be noted that complete observational datasets do not exist for these metrics and model evaluations rely on empirical models based on satellite and other datasets. Improving model skill is important as trends in the future magnitude of changes in nutrient supply and associated impacts on organic matter production are related to the magnitude of model biases in the contemporary ocean nutrients and stratification (Fu et al., 2016).

Trends in surface nitrate, organic matter production and export of organic matter into the ocean interior emerge from background variability after a median of around 30 years, but can take >50 years in some regions (Henson et al., 2016). Using CMIP5 models, it is suggested that attributing trends using ocean observing systems is challenging because they often need a long time series (>~35 years) and are only statistically representative of a small area (Henson et al., 2016). The anthropogenic trend in organic matter production does not emerge from background uncertainty due to internal and model dynamics by 2100 (Frolicher et al., 2016). Unlike pH and oxygen, scenario uncertainty is not a dominant component of the uncertainty in future projections of nutrient levels and organic matter production, which is instead dominated by model uncertainty, especially at regional scales (Figure 5.7i and l) (Frolicher et al., 2016). Indeed, globally the inter-model uncertainty in the trends in nitrate levels and productivity overlap between the RCP2.6 and RCP8.5 (Figure 5.7g and j) scenarios highlighting the important role for model uncertainty.

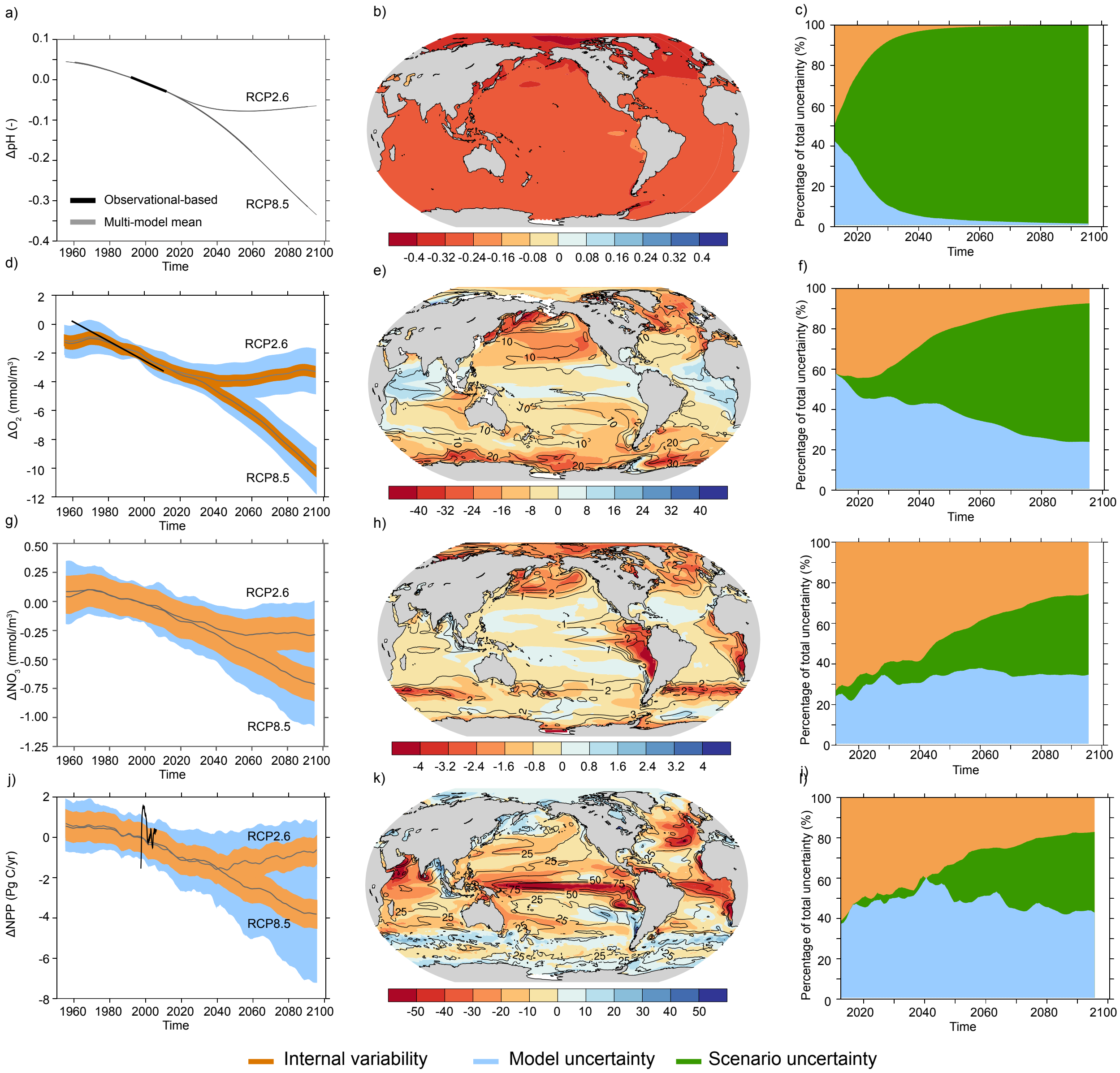


Figure 5.7: Panels a, d, g and j display Observation-based estimates (black lines) and simulated global changes over the period of 1950 to 2100. For surface pH, upper 100m nitrate concentrations, O_2 concentration averaged over 100 to 600m depth, and NPP integrated over the top 100 m. Panels b, e, h and k show spatial patterns of simulated change in surface pH, upper 100m nitrate concentrations, O_2 concentration averaged over 100 to 600m depth, and NPP integrated over the top 100 m. Panels c, f, i and l display time series of the percentage of total uncertainty ascribed to internal variability uncertainty, model uncertainty, and scenario uncertainty in projections of global annual mean changes in surface pH, upper 100m nitrate, O_2 averaged over 100–600 m, and NPP integrated over top 100 m. Figure adapted after (Frolicher et al., 2016).

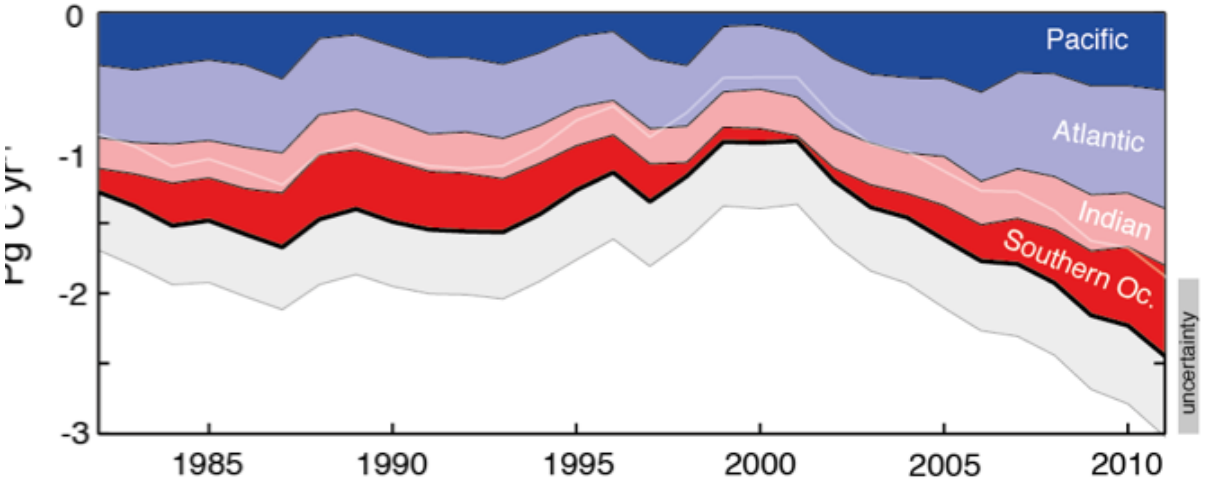


Figure 5.8: The evolution of the total ocean carbon sink and the contribution of the Pacific, Atlantic, Indian and Southern Oceans, based on observations Landschützer et al. (2016).

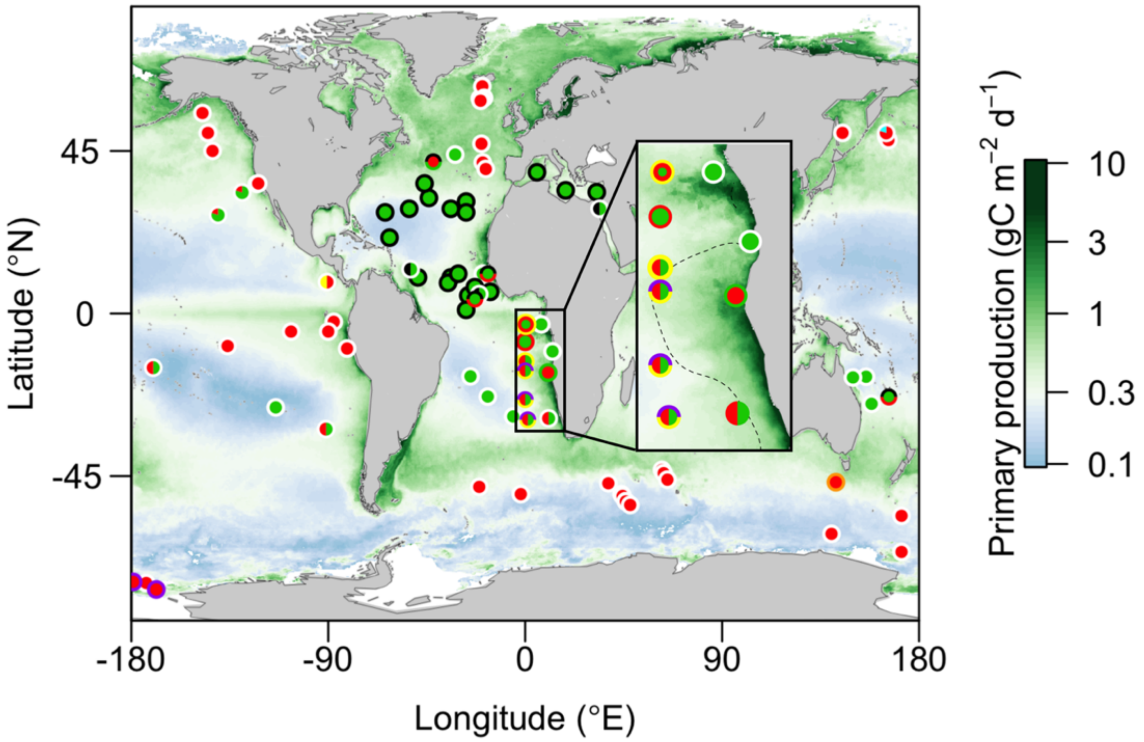


Figure 5.9: Map of limiting resource (Moore et al., 2013), updated to include new experiments from the north Pacific, tropical Atlantic and south east Atlantic (Browning et al., 2017a; Browning et al., 2017b; Shilova et al., 2017). The background is depth integrated primary productivity. Patterns of nutrient limitation in the context of annual average surface nitrate concentrations. Symbols indicate the primary (central circles) and serial (outer circles) limiting nutrients inferred from chlorophyll and/or primary productivity increases following artificial amendment of: N (green), P (black), Fe (red), Si (orange), Co (yellow), Zn (cyan) and vitamin B12 (purple). Divided circles indicate potentially co-limiting nutrients. White outer circles indicate that no secondary limiting nutrient was identified (note in many cases there was no testing for more than one nutrient). The boxed region in the south east Atlantic has been expanded to highlight the prevalence of colimitation (red/green divided circles) at the gyre boundary where the dashed line indicates the 1 mmol m–3 nitrate contour (concentrations increase landward). Note that there have been no experiments conducted in the Indian Ocean.

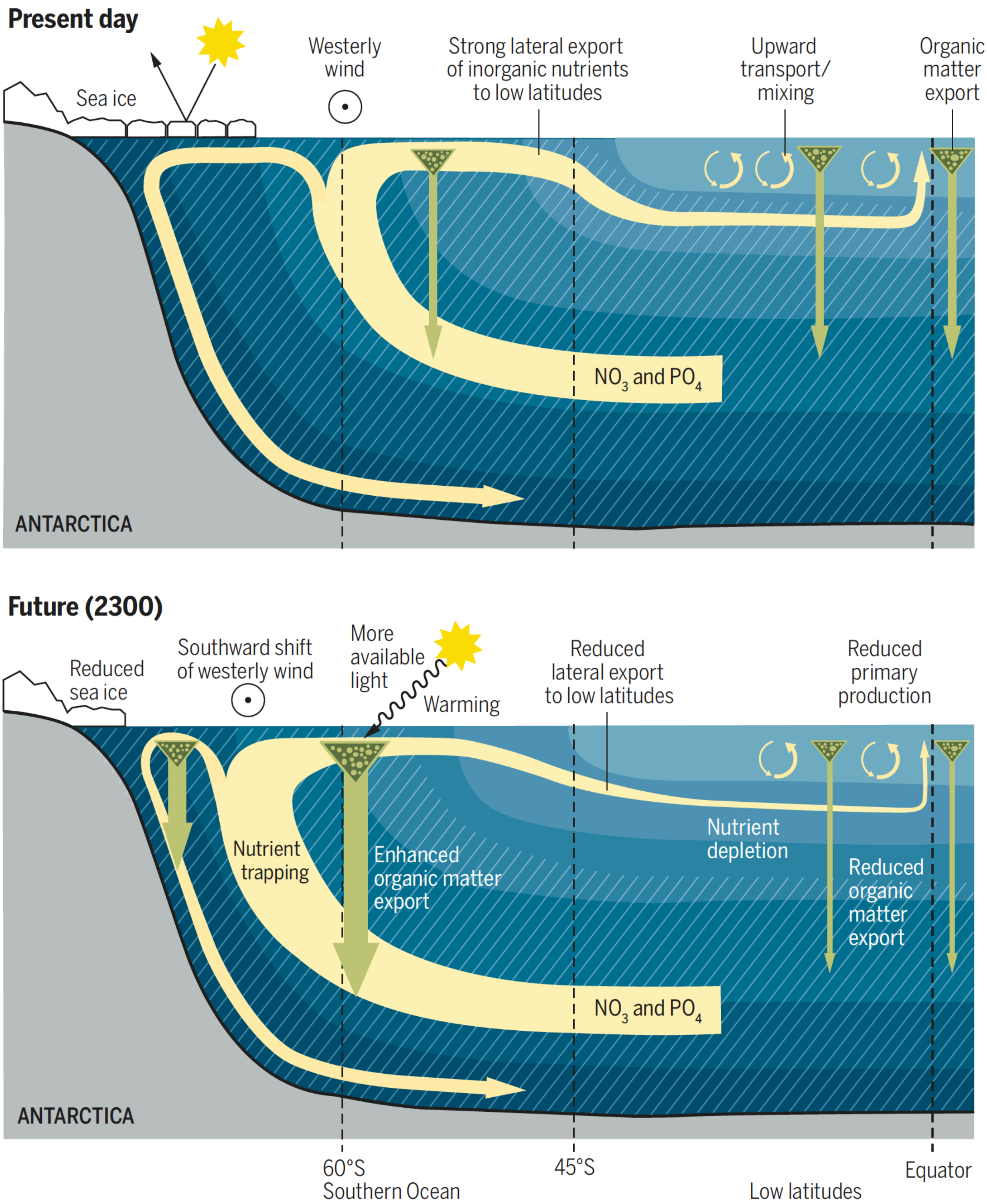


Figure 5.10: Today, the Southern Ocean plays a key role in the transport of nutrients to lower latitudes. A model study shows that this may change as a result of climate change. Under a high emission scenario, nutrients will be trapped in the Southern Ocean, reducing nutrient export to low latitudes. Hatching indicates elevated nutrient concentrations (taken from (Laufkötter and Gruber, 2018)).

[START BOX5.1 HERE]

Box 5.1: Time of Emergence and Exposure to Climate Hazards

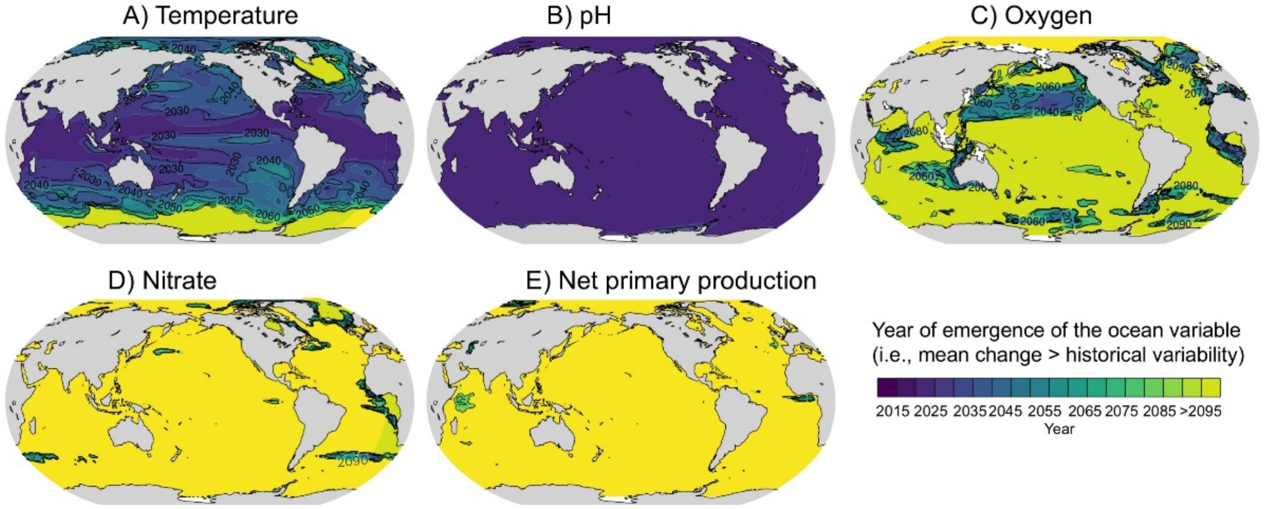
IPCC AR5 WG1 concluded that “It is *extremely likely* that human influence has been the dominant cause of the observed warming since the mid-20th century”. This conclusion was established based on observations of warming of the atmosphere and the ocean, in changes in the global water cycle, in reductions in snow and ice, in global mean sea level rise, and in changes in some climate extremes. Detection and attribution approaches demonstrate that signals of anthropogenic influences have already emerged from internal variability in the late 20th century for global and basin-scale averaged ocean heat content, surface and sub-surface temperature (*very likely*) and ocean salinity (*likely*), and Arctic sea ice (*very likely*) (AR5 WGI SPM). Such assessments have provided the critical evidence to attribute the significant contribution of human activities as the dominant cause of changes in climate and ocean systems.

The concept of ToE is defined as the time at which the *signal* of climate change in a given variable emerges from the *noise* of background variability (AR5 WGI 11.3.2.1, SROCC Glossary). In associating a calendar date with the detection and attribution of climate trends, the concept of a ToE has proved useful for policy and planning Hawkins and Sutton (2012). However, there is not a single agreed metric of ToE and the ToE for a given variable thus depends on choices regarding the space and time scale, the threshold at which emergence is defined and the reference period (AR5 WG1 11.3.2.1). Beginning with trends in atmospheric variables (Santer et al., 2011; Hawkins and Sutton, 2012) and ocean temperature (Mora et al., 2013), the ToE concept has been expanded to include ocean ecosystems. Recent studies consider essential variables such as pH, carbonate ion concentrations, aragonite and calcite saturation states, nutrient levels and marine primary productivity (Box 5.1, Figure 1) (Ilyina et al., 2009; Friedrich et al., 2012; Keller et al., 2014; Lovenduski et al., 2015; Rodgers et al., 2015).

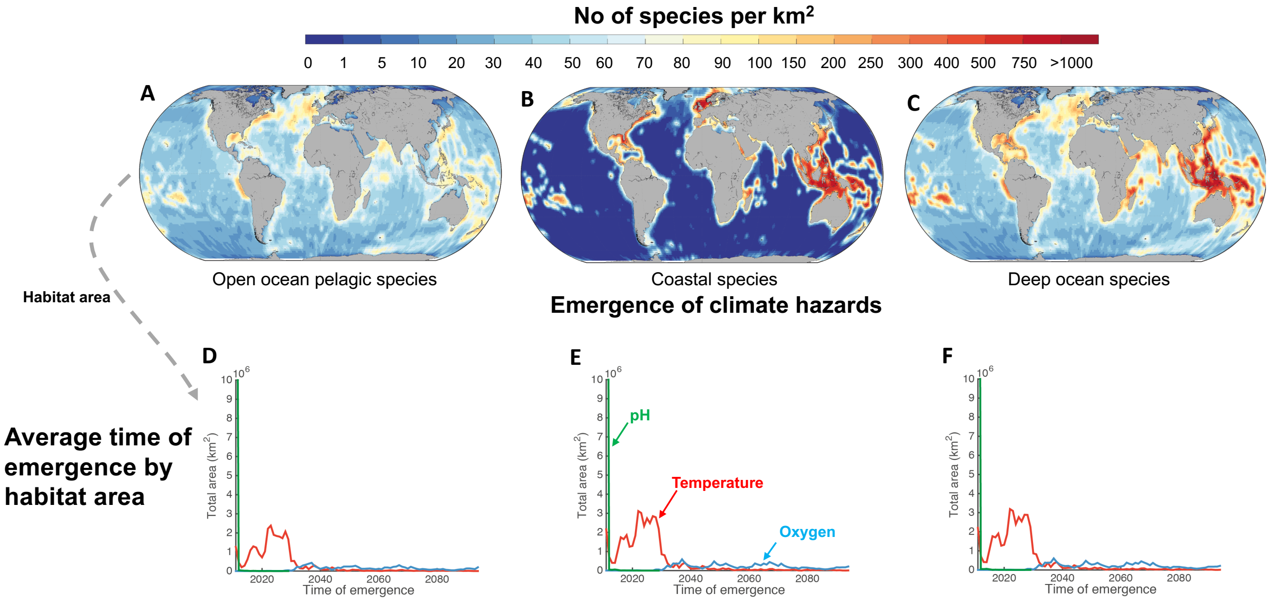
An important distinction to prior work reported in IPCC AR5 WG1 (Stocker et al, 2013) when applied to organisms and policy response relates to the spatial scales of interest. For instance, organisms respond to climate hazards that emerge locally, rather than to the global and basin-scale averages reported in IPCC AR5 WGI. This scale distinction means that ToE metric results in emergence occurring much later because of the greater noise due to internal variability at local scales. This Box assesses ToE at local scales for key marine environmental variables and their impact on key organisms and policy.

ToE of key marine environmental variables

Previous efforts to determine the local ToE for key ecosystem variables, here taken to be temperature, pH, interior oxygen, nitrate and net primary productivity, have used different time periods, noise assumptions and reference periods to calculate to ToE. Most assessments quantify the noise using the standard deviation of the detrended data over a given time period, known as the 'internal variability' (Keller et al., 2014; Rodgers et al., 2015; Henson et al., 2016; Henson et al., 2017). Other studies include the uncertainty associated with future projections (Frölicher et al., 2016), which increases the noise and thus lengthens the local ToE. The autocorrelation of noise is often neglected, which can be important in accurate ToE detection since even a relatively small amount of autocorrelation noise can extend the ToE by a decade (Weatherhead et al., 1998). Here we show the local ToE for five key ocean condition variables (Box 5.1, Figure 1). Calculated using the preindustrial period and accounting for internal and model variability in the noise (Frölicher et al., 2016), we find that the pH signal has already emerged everywhere, even at local scales, whereas sea surface temperature has emerged more rapidly in the tropics. Interior ocean oxygen depletion emerges from local background variability only in the north Pacific subtropical gyre, Arabian Sea, southeastern Indian Ocean, north and central eastern Atlantic Ocean, whereas nitrate and net primary productivity will not generally emerge, except at some local regions, until the next century. From an observational standpoint, similar analyses that account for autocorrelation of noise suggest time series of around a decade are sufficient to detect a trend in pH or SST, whereas datasets spanning 30 years or longer are typically needed for detection of emergence at local scales for oxygen, nitrate and primary productivity (Henson et al., 2016).



Box 5.1, Figure 1: Time of emergence of key ocean condition variables: (A) sea surface temperature (SST), (B) acidity (pH), (C) interior oxygen (O_2), (D) nitrate (NO_3), and (E) net primary production (NPP). The year of emergence represents the year when the mean change relative to pre-industrial period is above the standard deviation of each variable over the historical period ((Frölicher et al., 2016), Section 5.2.2).



Box 5.1, Figure 2: Projected emergence/exposure to climate hazards in ocean areas where occurrence of each taxonomic group has been recorded. The maps show distribution of recorded species richness for different ocean systems: (A, D) open ocean pelagic (left panel), (B, E) coastal (middle panel), (C, F) deep ocean (right panel). Occurrence records are based on OBIS, GBIF, and IUCN databases. The lower plots show the total habitat area with different time of emergence (warming, deoxygenation and acidification). Definition of time of emergence is based on Frölicher et al. (2016), (Figure 5.11).

The rapidity of change, encompassed in the ToE can be linked to concepts of exposure to hazard and vulnerability of biota by combining the ToE with life history traits to yield a time of organismal response (ToOR) (Box 5.1, Table 1; Box 5.1, Figure 2). As organisms have evolved to be adaptable to natural variations in the environmental conditions of their habitats. Thus, changes to their habitat conditions larger than that typically experienced may become hazardous Mora et al. (2013). Equally, species with fast generation times relative to the ToE of key habitat conditions (e.g., phytoplankton) may evolve more quickly to environmental change and be less vulnerable to climate change than longer-lived, slower generation time species (e.g., large sharks) (Jones and Cheung, 2018). However, evidence on evolutionary adaptation to expected climate change is limited, thus while shorter generation time may facilitate adaptation to environmental change, it does not necessarily result in successful adaptation of organisms (5.2.4, 5.4.3, *low confidence*). Pelagic microbes, such as phytoplankton or calcifying plankton like pteropods, reproduce rapidly and have the potential to respond to changes beyond background variability rapidly with a short ToOR. Other organisms like fish and sharks have multi-decadal generational life histories that will likely result in a longer ToOR. Sessile organisms like corals, will likely have the longest ToOR since their success relies on their calcite platform, which requires decades to rebuild if they are damaged (5.2.3).

Box 5.1, Table 1: Time of emergence of ocean variables and their challenges to marine biodiversity and ocean governance. The time scale for local emergence for ocean conditions are based on Frölicher et al. (2016) (see Box 5.1, Figure 1), while those for ToOR is based on the average ToE of the habitat range of the species. The scope for adaptation for biological system is discussed in 5.2.2 and 5.2.3, while those for human systems is discussed in 5.4.2 and 5.4.3.

Dimension	**Time scale for local emergence/generation time/Responses timeframe**	**Spatial pattern/Scope for adaptation**	**Confidence**	**References**
Ocean conditions (>20% the ocean with signal-to-noise ratio bigger than 1)				
SST	Present-day	Earliest emergence in the tropical regions	*High*	1, 2, 3
pH	Present-day	Across the ocean	*High*	1, 2, 3
Oxygen	Not in 21st century (RCP2.6) End of century (RCP8.5)	Earliest emergence in north Pacific subtropical gyre, Arabian Sea, southeastern Indian Ocean, north and central eastern Atlantic Ocean	*Low*	1, 2, 3
Nitrate concentrations	Not in 21st century (RCP2.6 & 8.5)	Very little emergence by end of century.	*Low*	1, 2, 3
Net primary production	Not in 21st century (RCP2.6 & 8.5)	Very little emergence by end of century.	*Low*	1, 2, 3
Selected biological groups (Generation time)				
Vibrio bacteria (*Vibrio vulnificus*)	Generation time: < an hour Key climate hazards: temperature Average time of emergence of ocean condition variables within its range: 2030 years	Tropical to temperate across different oceans High tolerance to environmental stress Capable of rapid adaptation to environmental changes.	*High*	4
Common jellyfish (*Aurelia aurita*)	Generation time: A year Key climate hazards: Temperature - average time of emergence within its range: 28 ± 10.9 years from 2000; Oxygen - average time of emergence within its range: 55 ± 16.0 years from 2000;	Circum-global, from equator to the poles High tolerance to wide range of temperature, pH and oxygen level Potentially high adaptive capacity, but lack published evidence of rapid adaptation to environmental changes	*Moderate*	5

Pacific oyster (*Ostrea gigas*)	Generation time: A few years Key climate hazards: Temperature - average time of emergence within its range: 27 ± 9.0 years from 2000; Oxygen - average time of emergence within its range: 57 ± 16.0 years from 2000; pH - average time of emergence within its range: 11 ± 0.0 years from 2000.	Native to northwest Pacific; introduced in the Pacific, Atlantic and the Mediterranean, subtropical Moderate temperature tolerance, larvae highly sensitivity to ocean acidification. Potentially moderate adaptive capacity, lack published evidence of adaptation to environmental changes	*Moderate*	6
Nassau grouper (*Epinephelus striatus*)	Generation time: A decade Key climate hazards: Temperature - average time of emergence within its range: 12 ± 2.0 years from 2000; Oxygen - average time of emergence within its range: 77 ± 11.0 years from 2000;	Western tropical Atlantic Narrow temperature tolerance Potentially low adaptive capacity, particularly as populations are highly dependent on spawning aggregation and the species is classified as endangered (IUCN Red List); lack published evidence of adaptation to environmental changes	*Low*	7
Patagonian toothfish (*Dissostichus eleginoides*)	Generation time: A few decades Key climate hazards: Temperature - average time of emergence within its range: 34 ± 12.0 years from 2000; Oxygen - average time of emergence within its range: 57 ± 12.0 years from 2000;	Southeast Pacific, southwest Atlantic and sub-antarctic Narrow temperature tolerance Potentially low adaptive capacity; lack published evidence of adaptation to environmental changes	*Low*	7
Greenland shark (*Somniosus microcephalus*)	Generation time: > a century Key climate hazards: Temperature - average time of emergence within its range: 47 ± 20.0 years from 2000; Oxygen - average time of emergence within its range: 45 ± 11.0 years from 2000;	Arctic and north Atlantic Narrow temperature tolerance Potentially very low adaptive capacity; lack published evidence of adaptation to environmental changes	*Low*	7
Human sectors (Timeframe of investment/operational decision)				
Small-scale coral reef fisheries	Seasonal, annual to several years Key climate hazards:	Common in tropical regions where coral reef occur, substantial contribution to livelihood and food security	*Moderate*	8

	Temperature - average time of emergence where the fisheries operate: 14 ± 3.0 years from 2000; Oxygen - average time of emergence where the fisheries operate: 53 ± 21.0 years from 2000; pH - average time of emergence where the fisheries operate: 11 ± 0.1 years from 2000.	Generally classified as low adaptive capacity because of spatial dependence, insufficient assets and access to resources particularly in developing countries.		
Oyster farming (Mariculture)	Annual to a decade Temperature - average time of emergence where the farms operate: [xx ± x.0] years from 2000; Oxygen - average time of emergence where the farms operate: xx ± x.0 years from 2000; pH - average time of emergence where the farms operate: xx ± x.0 years from 2000.	Temperate region of Pacific North and South America, Northeast Atlantic and Mediterranean, East Asia, Eastern Australia and New Zealand Generally classified as moderate adaptive capacity. The potential rapid responses through the use and adoption of technological interventions. However, the strong interactions with environmental change from other non-climatic human activities such as nutrient enrichment from run-off that results in eutrophication and hypoxia may exacerbate climate change effects and increase the rate of ToR of key environmental variables.	*Moderate*	9
Governance (Timeframe of policy-making)				
International	One to a few decades	Low adaptive capacity as they generally require consensus from all countries, thus take long time to adjust (5.4.2).	*High*	10.
National	< a decade	Moderate adaptive capacity with large variability between countries. Self-determination and sovereignty allow for setting of policy (5.4.2).	*Moderate*	10.
Local	A few years	High adaptive capacity because specific interventions to target cause of impact at site specific locations are possible See Section 5.4 for adaptation options at a local level, including ecosystem-based adaptation (5.4.2.1), built environment adaptation (5.4.2.2] and socio-institutional responses (5.4.2.3).	*High*	10.

Notes:
1. (Frölicher et al., 2016); 2. Henson et al. (2016); 3. Henson et al. (2017); 4. Turner et al. (2016); 5. Onstein Renske et al. (2016); 6. Wendling and Wegner (2015); 7. Crozier and Hutchings (2014); 8. Cinner et al. (2018); 9. Allison et al. (2017); 10. Dunn et al. (2016)

ToE and policy response

The challenge for marine-related policies to respond to climate change is dependent on the ToE of the changing ocean conditions (5.2.1), their impacts on marine ecosystems (5.2.2) and their services (5.3.1), as well as the time of policy response (ToPR) (Box 5.1, Table 1). Particularly, the rapid emergence of stressors at the local scale in the near-term (next few decades) such as warming and ocean acidification and the resulting impacts on some of the less adaptive biodiversity and ecosystem services (Box 5.5). These factors pose substantial challenges to international ocean governance such as through the United Nations Conventions of the Law of the Seas (UNCLOS) (5.4.2, 5.4.3). International ocean law and policies such as UNCLOS take decades to negotiate and amend. In contrast, scope for adaptation for national and local ocean governance can be more responsive to rapid changes. This highlights the opportunities for multi-level adaptation that allow for reducing climate risks that are expected to emerge at different time frame Mackenzie et al. (2014).

[END BOX 5.1 HERE]

5.2.2 Key Ecosystem Vulnerabilities to Multiple Climate Drivers

5.2.2.1 Introduction

Marine ecosystems face increasing pressures due to multiple environmental drivers from the changing climate and increasing intensity of human activities. Past assessments by the IPCC focused strongly on impacts and vulnerabilities of specific individuals or species assemblages. However, science-based conservation and restoration policies require a mechanistic understanding of ecosystem-level responses to multiple climatic and human drivers. The interaction of environmental drivers can produce additive, synergistic (i.e., amplifying) or antagonistic (i.e., dampening) effects and may operate distinctly at individual, population and ecosystem scales (Piggott et al., 2015; Rocha et al., 2015; Gunderson et al., 2016; Nõges et al., 2016). Since AR5, there is substantially more field and experimental evidence, as well as, model simulations that have allowed progress in understanding the effects of multiple climate drivers, such as warming, acidification and deoxygenation, on marine ecosystems. 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 the short-term natural fluctuations, ranging from the seasons to climate oscillations like El Niño (Boyd et al., 2018). The so-called 'time of emergence' for specific drivers beyond background variability is distinct in different ecosystems and is strongly sensitive to projected mitigation scenarios (Henson et al., 2017) (Box 5.1).

Ecosystems are affected by different combinations of climate drivers. These drivers alter the interconnected biogeochemical and ecological dynamics at different rates and magnitudes (Section 5.2.1, Box 5.1, Table 1). For example, pelagic systems face intensified thermal stratification and nutrient depletion in the upper layers, which, in turn, is thought to reduce plankton productivity and carbon transport to the deep ocean (Section 5.2.1.5); while coastal ecosystems are threatened by sea level rise and erosion, where vegetated coasts lose carbon from sediment sinks, with potential adverse feedback to the climate system. In addition, coastal ecosystems are further exposed to a growing number of human-related threats such as eutrophication, coastline modifications and disposal of pollutants. Therefore, policy makers must face a dilemma of reducing the effect of dominant drivers, while acknowledging that the underlying dominance may change in the future, and that their management actions may produce unexpected biological responses (Breitburg et al., 2015).

Multi-scale environmental signals act simultaneously at different biological levels producing physiological, ecological and evolutionary outcomes (see also Section 5.2.3). Common ecological responses are modifications in the temporal and spatial distribution of species, altered gene expression, physiology, reproduction, or behaviour (Gobler and Baumann, 2016). Whereas some organisms may adapt, others may not be able to tolerate the present and future environmental rate of change and could be excluded from many localities. At the ecosystem level there can be alterations in assemblage composition, production, diversity, trophic structure, nutrient cycling and more, ultimately affecting ecosystem services (Riebesell and Gattuso, 2014). The ability of ecosystems to compensate global change and to provide meaningful ecological services, is partly dependent on their structural and functional diversity (see Section 5.3) (Soliveres et al., 2016). For instance, combined elevated CO_2 and temperature improves the performance of consumers such as fishes and crustaceans in different food webs from open sand, rocky reef and seagrass habitats.

The complexity of natural environments and the emergence of positive ecological interactions and traits in the consumers community, e.g., habitat selection, foraging and predator avoidance, compensate the negative effect on single species reported in simplified empirical studies (Goldenberg et al., 2018). This emphasizes the need for sustained field observations and long-term experiments that allow the emergence of different-level of biological responses and their feedbacks, to design mitigation actions (Gieswein et al., 2017). This section summarizes the responses to climate change at the ecosystem level and asses the vulnerability to the dominant combination of climate drivers occurring at specific habitats.

5.2.2.2 Pelagic Systems

5.2.2.2.1 The epipelagic ocean

Since the AR5 WGII, there have been a wide range of advances in understanding the effect of climate change on the surface epipelagic ocean (0-200 m) (see Glossary) across observation, experimental and modelling studies. Studies have continued to explore time-series datasets of planktonic observations from both ocean surveys (such as the Continuous Plankton Recorder, often five to six decades long) and time-series sites (typically, shorter, at one to three decades long). As was discussed in AR5 WGII (Pörtner et al., 2014), short time series of two decades or less in length are too short for determining trends (Henson et al., 2010; Chavez et al., 2011), and in longer time-series it is important to establish the relative influences of climate change and variability (Edwards et al., 2013).

Observational studies have focussed on investigating the cause(s) of alteration in epipelagic community structure from time-series plankton tows in the North Atlantic (and other oceans), and have attempted to tease apart the relative roles of regional versus global drivers (Rivero-Calle et al., 2015). They report, using Continuous Plankton Recorder (CPR) archives, that stocks of coccolithophores over the last five decades have increased by 20%, and that this increase is linked (by statistical analysis) to higher CO_2 concentrations (*medium confidence*). However, the outcome of their analysis depends on whether global (Mona Loa archive) or regional (Takahashi archive) CO_2 time-series are employed in the statistical analysis; with global CO_2 and the Atlantic Multidecadal Oscillation (AMO), respectively, being the main driver attributed to this floristic shift. Other studies have examined whether phytoplankton can respond to the observed changes in biologically-influential properties over a decade or more. (Irwin et al., 2015) analysed shifts in phytoplankton community (across ~60 species) at the CARIACO (Carbon Retention in a Colored Ocean) time series in Central America using 15 years of records. They reported that the resident population adapted in concert with trends in warming and higher underwater light levels (*low confidence*). However, it should be noted that the approach used to make these conclusions has been disputed by Brun et al. (2015). There has also been further use of the nano-fossil records from the geological past to explore the role of multiple drivers in altering pelagic community structure Gibbs et al. (2015). They reported that during the PETM (Paleocene-Eocene Thermal Maximum, 56 MA) warming played an equally important role than ocean acidification in altering the distribution of coccolithophores from the low latitude to the high latitude ocean. Hence the biogeography was set by both warming and acidification–with range shift of coccolithophores towards high latitudes with relatively cooler waters.

Other studies in the open ocean have investigated the rate and direction at which climate is changing, which dictates the speed at which plankton communities must move in order to maintain their present day (or previous) environmental regime. In the North Atlantic, (Barton et al., 2016) revealed - from multi-decadal CPR records combined with future modelling projections - northwards (12 km a^{-1}) and eastward (~40 km a^{-1}) shifts in diatoms and dinoflagellate stocks that were driven by the cumulative changes from a range of environmental drivers (sea surface temperature, sea surface salinity, mixed layer depth, photosynthetically active radiation, and the surface concentrations of nitrate, phosphate, and silicate). Chivers et al. (2017) analysed six decades of CPR data from the North Atlantic and pointed out that while the range of dinoflagellates and copepods tended to closely track the velocity of climate change (the rate of isotherm movement), the range of the diatoms moved much more slowly. These differential responses between species in CPR data to warming are indicative of the reorganisation of planktonic community structure in the coming decades (*medium confidence*).

Climate change experiments, in particular those in large volume mesocosms containing a wide range of pelagics from microbes to zooplankton, in the coastal (Norwegian fjord) and open (subtropical Atlantic) ocean have shed light on the effects of ocean acidification on pelagic community structure. In the coastal ocean, *Emiliania huxleyi*, a widely distributed bloom-forming coccolithophore decreased in abundance, in response to acidification. The decline in abundance was such that it could not form a bloom under high CO_2 conditions and lead to a major decrease in the associated biogeochemical signatures such as export flux Riebesell et al. (2016).

Table 5.1: Projected future changes to the ocean biological pump (adapted from (Henson et al., 2017)).

Pump component	Oceanic driver	Projected change	Confidence	References
Epipelagic Ocean				
Phytoplankton growth	Temperature (warming)	~10% Faster (nutrient-replete) no change (nutrient-deplete)	*High*	Boyd et al. (2013) **E**; Maranon et al. (2014) **O**;
Net Primary production	Climate change (temperature, nutrients, CO_2)	10-20% decrease (low latitudes); 10-20% increase (high latitudes)	*High*	Bopp et al. (2013) **M**
Partitioning of NPP (POC, TEP, DOC)	OA	~20% increase in TEP production (Engel, 2002)	*Medium*	Engel et al. (2002); Riebesell et al. (2007)**E**;
Floristic shifts	Climate change (Warming/OA/Iron/ Salinity	Shift to smaller cells (*less export*)	*Medium*	Moràn et al. (2010) **O**
Differential susceptibility	Temperature (warming)	Growth-rate of grazers more temperature dependent than prey (*less export*)	*Low*	Rose and Caron (2007) **O**;
Grazer physiological responses	Warming	Copepods had faster respiration and ingestion rates, but higher mortality (*inconclusive*)	*Low*	Isla et al. (2008)
Faunistic shifts	Temperate and subpolar zooplankton species shifts	Temperature (*inconclusive*)	*Low*	Edwards et al. (2013)
Deep Ocean				
Bacterial hydrolytic enzyme activity	Temperature	20% increase (resource-replete) to no change (resource-deplete)	*Low*	Wohlers-Zöllner et al. (2011)**E**; Endres et al. (2014) **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*	Burd et al. (2002) **M**; Ikeda et al. (2001) **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*	Hofmann and Schellnhuber (2009) **M**

Notes:

Environmental controls on individual factors that influence downward POC flux are based on published reports from experiments (denoted by E), modelling simulations (M) and observations (O). In some cases, due to the paucity, and regional specificity, of published reports it has been indicated the sign of the projected change on export (in italics), as opposed to the magnitude. DOC is Dissolved Organic Carbon. Climate change denotes multiple controls such as nutrients, temperature and irradiance, as parameterised in coupled ocean atmosphere models. &denotes observed for low latitudes only. ? represents major uncertainty over environmental modulation of this component of the biological pump. *denotes joint influence of temperature and acidification.

A wide range of studies, from laboratory experiments Taucher et al. (2015), mesocosm enclosures (Isla et al., 2008), synthesis of observations (Rose and Caron, 2007) to modeling experiments (Bopp et al., 2013) provide insights into how the multi-faceted components of the "biological pump" (the biologically mediated processes responsible for transporting carbon from the upper ocean to depth) are projected to be altered in the coming decades. A synthesis of the individual components reported to both influence the performance of the biological pump, and which are sensitive to changing ocean conditions, is presented in Table 5.1. They include those catalogued by (Pörtner et al., 2014) along with additional factors - such as seawater viscosity and its influence on particle sinking rates Taucher et al. (2014) and the joint influence of warming/acidification on bacterial solubilization of particles (Piontek et al., 2015) - from recent research. Table 5.1 lists the putative controlling environmental factor, such as warming, on each individual component that influences the biological pump, and the reported modification (where available) of each individual factor by changing ocean conditions for both surface and subsurface waters.'

5.2.2.2.2 The deep ocean

The pelagic realm of the deep ocean, containing the mesopelagic (200–1000 m depth) and bathypelagic (>1000 m depth) zones, represents a key site for remineralisation of organic matter and long-term 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). Global ocean warming between 0.5°C (RCP2.6) and 1.5°C (RCP8.5) is expected to reach the bathypelagic zone by the end of the century (Figure 5.5), with the largest warming of deep waters occurring first in the Southern Ocean (Ciais et al., 2013; Patara and Böning Claus, 2014); Roemmich et al. (2015) (*high confidence*). There is *medium confidence* that warming and changes to primary productivity in the upper ocean will reduce the export of organic matter to the deep sea (Section 5.2.1.5). Analyses of long-term trends in primary production and particle export production, as well as model simulations, reveal that increasing temperatures, leading to enhanced stratification, nutrient limitation and shifts towards small phytoplankton, will have the greatest influence on decreasing the flux of particulate organic carbon (POC) to the deep ocean (Bopp et al., 2013; Boyd et al., 2015; Fu et al., 2016; Laufkötter et al., 2016). Observational and model-based methods provide evidence that the transfer efficiency of organic carbon through the mesopelagic zone is controlled by temperature and oxygen (*medium confidence*), affecting microbial metabolism and zooplankton community structure, with highest efficiencies for high-latitude and oxygen minimum zones (OMZs) (see Section 5.2.1.4) for more detail on OMZs), while below 1000 m the highest transfer efficiencies occur in subtropical regions, and are controlled by particle sinking speed (Boyd et al., 2015; Marsay et al., 2015; DeVries and Weber, 2017). There is however *low confidence* on the effects of increasing temperatures on particulate organic carbon (POC) remineralisation to CO_2 versus POC solubilisation to dissolved organic carbon (DOC) by microbial communities and its storage as refractory DOC (i.e., with life times of >16,000 years) (Legendre et al., 2015). The global magnitude of the 'active flux' of organic carbon from the surface to the mesopelagic due to vertical migration of zooplankton and fishes is uncertain, but has been reported to account from 10 to 100% of the gravitational sinking flux (Davison et al., 2013; Hudson et al., 2014; Jónasdóttir et al., 2015) (*low confidence*).

Predictions based on model studies suggest that mesopelagic zooplankton and fish communities living at deep scattering layers (DSLs) will increase their biomass by 2100, enhancing their trophic efficiency, because of ocean warming and shallowing of DSL (Proud et al., 2017) (*low confidence*). Recent model projections agree with the earlier results from IPCC AR5 WGI Chapter 3, predicting an average 3.5% loss of total ocean oxygen by 2100 and expansion of OMZs (see Section 5.2.1.4). There is *high confidence* that OMZ expansion and shoaling would favour anaerobic microbial processes leading to denitrification. It will also widen the DSL and increase the exposure of mesopelagic organisms to shallower depths (Gilly et al., 2013; Netburn and Anthony Koslow, 2015). In the California Current, the abundance of mesopelagic fishes is closely tied to variations in the OMZ, whose dynamic is linked to the Pacific Decadal Oscillation (PDO) and El Niño-Southern Oscillation (ENSO) cycles (Koslow et al., 2015). Hypoxia-adapted animals, like the Humboldt squid, will benefit from expanding OMZs (Stewart et al., 2014), but many non-adapted fish and invertebrates (like diurnal vertical migrators) will have their depth distributions squeezed, affecting the carbon transport and trophic efficiency of food webs in the mesopelagic (Stramma et al., 2011; Brown and Thatje, 2014; Rogers, 2015) (*high confidence*). In OMZ waters, where zooplankton is almost absent, like in the Eastern Tropical North Pacific, the microbial remineralisation efficiency of sinking particles would be reduced, eventually increasing the transfer efficiency of organic matter to the deep ocean and thus biological carbon storage (Cavan et al., 2017) (*medium confidence*). However, increases in ocean temperature may also lead to shallower remineralisation of POC in warm tropical regions, counteracting the storage of carbon in the dark ocean (Marsay et al., 2015). Overall, there is *low confidence* in predicting how the different components of the biological pump will respond to climate change variability in the deep ocean (Table 5.1.).

5.2.2.3 Coastal Systems

Coastal ecosystems are amongst the most diverse due to their complex geomorphology and steep physicochemical gradients, and they are highly sensitive to changes in atmospheric conditions. They are also exposed to other non-climatic human drivers and their potential interactions with climate change because of their close proximity to human populations. All coastal ecosystems either classified by their geomorphological structure (i.e., estuaries, sandy beaches, rocky reefs, fjords) or foundation species (i.e., saltmarshes, mangroves, seagrass meadows, kelp forest, coral reefs) are threatened by sea level rise, warming, acidification, deoxygenation and extreme weather events with substantial differences in the type and level of vulnerabilities. While some ecosystems such as estuaries, sandy beaches, mangroves and saltmarshes are more vulnerable to sea level rise and erosive processes from extreme weather events such as storm surges and flooding, others like rocky reefs, kelp forests and coral reefs, are mostly threatened by warming and acidification. In particular, saltmarshes, mangroves and coral reefs have a disproportional ecological value due to their role in the protection against erosion and the reduction of exposure and sensitivity of biological communities to climatic stressors such as warming.

At the species level, the main biological responses to the interactive effects of climate drivers are shifts in species distribution towards the poles, deeper water or specific physicochemical gradients, as well as inland migration of vegetation and benthic fauna in the coastal zone, resulting in species range expansion and local extinction. Consequently, the restructuring of coastal communities affects ecosystem functioning and services such as carbon storage, productivity and storm protection (Section 5.3.1.1). Common examples of these synergies are the expansion of hypoxic zones enhanced by eutrophication in estuaries and low adaptability to sea level rise driven by coastal squeezing due to human barriers in sandy beaches and saltmarshes.

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

5.2.2.3.1 Estuaries

The land-sea interface provides high habitat heterogeneity to estuarine ecosystems, surrounded by marshes and mudflats that have a diverse and abundant community of algal and vascular plant producers (see Sections 5.2.2.3.2 and 5.2.2.3.4). Estuaries are sources of CO_2 to the atmosphere and emit an average of 0.25 ± 0.25 PgC yr^{-1} (Regnier et al., 2013). The coastal vegetated wetlands, however, are important areas of carbon burial that can sequester between 0.067 and 0.215 PgC yr^{-1} (Hopkinson et al., 2012). Modifications on the drainage area due to e.g., urban settlements, land reclamation and dredging have resulted on a significant reduction of estuarine area that is estimated to ranges between 1% and 7% per year, and translates to a decrease of 0.05 PgC yr^{-1} on carbon burial (Regnier et al., 2013) (*high confidence*). Accordingly, sea level rise represents one of the most important threats to estuaries. Rising sea level affects submergence of wetlands, and in the case of Australia is estimated to cause a 30% decline in carbon burial within these ecosystems by 2100 under the A1FI emissions scenario (Rogers et al., 2013).

More observational evidence since AR5 WGII (Wong et al., 2014a) supports that climate change modifies the estuarine gradients of salinity, turbidity and nutrients through interactive effects of sea level, precipitation, temperature and estuarine circulation patterns (Robins et al., 2016; Raimonet and Cloern, 2017). The interaction of sea level rise and changes in rainfall, and thereby river flows, will have a more severe impact on shallow estuaries (<10 m) than on deep estuaries (>10 m). In a projected sea level rise of 1 m, indices of vulnerability indicate that shallow estuaries will increase the tidal current amplitude by 5%, and the energy dissipation, the vertical mixing, and the salinity intrusion by 25% (Prandle and Lane, 2015). Meso- and macrotidal estuaries are more resilient to global climate changes than microtidal estuaries, which are more vulnerable to sea level rise and changes in river flow (*medium confidence*). 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). The response of estuaries to river floods will vary regionally depending on the infiltration capacity of soils and the mechanisms that generate the floods (rainfall, snow melting and soil saturation), as well as on the balance between rainfall and evaporation rate (Arnell and Gosling, 2016).

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

Increasing flooding during rainy periods over fertilized agriculture land will enhance estuarine eutrophication and stratification, and consequently will increase the likelihood of phytoplankton blooms (Li et al., 2015) (*high confidence*). Such bloom events are difficult to synchronize with local grazers (Thackeray et al., 2016) leading to short-term biomass accumulation that ultimately will reduce the water quality by increasing the risk of hypoxia and production of microalgae toxins (Anderson et al., 2015). The evidence of harmful algal blooms (HABs) has increased in coastal areas worldwide over the last 20 years (Anderson et al., 2015) (see Sections 5.2.2.3.2 and 5.2.2.3.4) in relation to interactive effects of climate warming and nutrient inputs such as nitrogen and phosphate from land and wind surge (Paerl et al., 2016). Additionally, interactive effects between human pollution, warming and extreme precipitation events on estuarine turbidity, dissolved organic matter and oxygen concentrations, will increase the occurrence of pathogenic bacteria, e.g., *Vibrio* species (Baker-Austin et al., 2017; Kopprio et al., 2017) (Section 5.3.2, *low confidence*). Likewise, extreme river runoff increases the inflow of detritus and suspended sediments and reduces primary productivity by changing 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 conditions in bottom layers (Breitberg et al., 2015; Gobler and Baumann, 2016).

In particular, expansion of oxygen-depleted dead zones in coastal areas is projected under intensification of co-occurrence of climate threats and eutrophication, with warming as the main climate-related driver that enhances hypoxia through a variety of physical and biological mechanisms. 94% of these coastal zones are predicted to experience a 2°C temperature increase in the A1B emissions scenario by the end of the century (Altieri and Gedan, 2015). As the sensitivity of oxygen solubility is greater at lower temperatures, the effect of warming will be more pronounced on high latitude, temperate and arctic estuaries, where the predicted rates of warming are higher and the initial water temperatures are lower. Additionally, hypoxia will be more pronounced in shallow temperate estuaries because they are more responsive to warming due to their limited exchange with the open ocean (e.g., Río de La Plata Estuary, Baltic Sea and Chesapeake Bay), and seasonality leads to dead zone development when temperature reaches critical values in summer (e.g., Black Sea) (Altieri and Gedan, 2015). Eutrophic and shallow estuaries face magnification on hypoxic conditions trough algae blooms and microbes respiration, which generate CO_2 and reduce the pH (Breitberg et al., 2015). The coastal acidification related to the expansion of hypoxic dead zones is of main concern for conservation of sensitive organisms (Beck et al., 2011; Duarte et al., 2013). The widespread observational evidence from shallow estuaries has already linked the increased levels of eutrophication and hypoxic conditions to warming (*high confidence*), and projections show that this processess will be more pronounced in estuaries from high latitudes (*high confidence*).

Large-scale climate signals that further influence estuarine gradients, such as North Atlantic Oscillation (NAO), ENSO and Southwest Atlantic Meridional Overturning Circulation, have evidenced persistent anomalies since the 1970s associated with climate change (Wang and Cai, 2013; Delworth and Zeng, 2016). In turn, these changes in estuarine gradients affect the phenology and composition of coastal plankton, as displayed by sustained observations (*high confidence*). For instance, at the temperate Mondego estuary (northeast Atlantic Ocean), higher temperatures driven by NAO fosters the intrusion of marine, cosmopolitan zooplankton species at the expense of native, estuarine species (Marques et al., 2017). Heavy rainfall events driven by ENSO, causes surges in river runoff and nutrient supply in temperate estuaries from the southern hemisphere, which in turn modifies the structure of phytoplankton communities (Sathicq et al., 2015; Thompson et al., 2015; Andrade et al., 2016) (*medium confidence*). In the temperate Bahía Blanca Estuary (SW Atlantic Ocean), warmer phases of ENSO mediate the emergence of water turbidity (enhanced by dredging) as a dominant driver of phytoplankton phenology, and promoted the decline of chlorophyll concentration at a yearly rate of 1% (López Abbate et al., 2017). Negative resonance effects from ENSO-driven droughts (i.e., increase soil salinity) also occur within salt marshes, where a reduction in plant production affects direct consumers (Pascual et al., 2015). In addition, ENSO exacerbates the effect of estuarine habitat modification by humans (e.g., coastal infrastructure, dredging) on the mobilization of blue carbon (Macreadie et al., 2015). Heatwaves associated with La Niña in Shark Bay, Australia, produced nutrient pulses from degraded submerged coastal vegetation and triggered phytoplankton and bacterial blooms (Thompson et al., 2015; Arias-Ortiz et al., 2018) (Box 5.2). Although these changes may be attributed to climate variability, they provide tentative evidence of the sensitivity of different ecosystem components to aspects of climate change, which is relevant given the *likely* intensification of ENSO events projected for 2100 (Stocker et al., 2013).

Overall, global observational evidence supports that the land-sea interface confers estuaries high habitat heterogeneity, and these ecosystems are sensitive to climate change and human impacts (*high confidence*) such as eutrophication. Future scenarios of sea level rise, warming and extreme precipitation events will affect the balance between river runoff and marine water intrusions, thus modifying the physical and biogeochemical properties of the estuarine gradient. Shallow and microtidal estuaries in those areas exposed to higher probability of flooding will be the most vulnerable (*high confidence*) to experience transformation in the ecosystem functioning and services.

5.2.2.3.2 Salt Marshes

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

Nearly half of the pre-industrial, natural extent of salt marshes has been lost, due to drainage, agriculture, coastal settlement, hydrological alterations and reductions in sediment supply (Adam, 2002; Wang et al., 2014; Kroeger et al., 2017; Li et al., 2018). Sea level rise is a major threat to salt marshes; between 20-60 % of coastal wetlands are expected to be lost globally by 2100, depending on emission scenarios (*medium confidence*) (Watson and Hinojosa Corona, 2017). Salt intrusion results in community restructuring (Janousek et al., 2017) and changing above-and below-ground productivity (McLeod et al., 2011). Plants with low tolerance to flooding are especially vulnerable. For example in New England salt marshes, *Spartina* *patens* is being replaced with flood-tolerant *S. alterniflora* or converted to pools of unvegetated standing water (Raposa et al., 2017), while *S. patens* specialized foragers are rapidly declining (Johnson and Williams, 2017). A similar expansion of the flood-tolerant *S. alterniflora* in response to rising sea level occurred in a temperate estuary in the southwestern Atlantic, Argentina. Over 33 % of saltmarsh area covered by *Sarcocornia perennis* and 6 % of halophytic shrub-like steppes dominated by *Cyclolepis genistoides, Allenrolfea patagonica and Atriplex undulata,* have been lost in the last 40 years and replaced by either mudflats or *S. alterniflora* marshes (Pratolongo et al., 2013).

Arid, sub-tropical marshes may show a contrasting response to raising sea level (*medium confidence*). For instance, the area of five of the largest coastal wetlands in 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).

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

Warming will increase metabolic rates of salt marsh organisms but effects will vary between species (*medium confidence*). For example, warming experiments in New England salt marshes showed greater increases in shoot productivity of the low marsh dominant grass, *Spartina alterniflora,* than that of the high marsh grasses, *Spartina patens* and *Distichlis spicata* (Charles and Dukes, 2009). The combination of higher CO_2 and warming could increase marsh elevation sufficiently to withstand moderate increases in sea level (Langley et al., 2009; Kirwan and Mudd, 2012). However, in models of rising temperatures and higher sea levels, *Spartina alterniflora* marsh responds initially with increasing accretion rates and carbon burial, therefore having negative feedbacks on climate, while by 2075, larger carbon pools and dominant root zone processes slow down rates of accretion and increase decay rates, switching to a positive climatic feedback (Kirwan and Mudd, 2012). Increased decay rates in response to warming are attributed to increased microbial and fungal activities (Kirwan and Blum, 2011).

To promote salt marsh ecosystem resistance to long-term climate trends, local stressors such as eutrophiction, overfishing, and species invasions, must be understood and addressed. In New England salt marshes, for example, overfishing has led to uncontrolled grazing by a herbivorous marsh crab, *Sesarma reticulatum*, that is currently degrading *Spartina alterniflora* zones at rates much faster than sea level rise. This trophic cascade synergistically interacts with rising sea level because water-induced softening of inundated marsh peats permits new expansion of the crab into high marsh zones (*medium confidence*) (Crotty et al., 2017).

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

Invasive plant species are likely to be favored by multiple factors associated with climate change. In North America, one of the most aggressive invasive species is the cosmopolitan *Phragmites australis* which is facilitated by excess nitrogen. As a C3 species, *P. australis* displays strong potential to expand under rising CO_2 levels (Mozdzer and Megonigal, 2012) and is evading many management strategies (Martin and Blossey, 2013). However, recent mesocosm experiments revealed that in conditions of warming without excess nutrients, *S. alterniflora* resists *P. australis* invasion, thus minimizing nutrient pollution makes native species more competitive as temperatures increase (Legault II et al., 2018). In east China, experimental warming with the invasive *S. alterniflora* increases net nitrification and mineralization rates compared to soils with native plant species. Thus, invasive species may disproportionately benefit from warming-associated gains in nutrient availability (*medium confidence*) (Zhang et al., 2016).

Hydrological alteration is a key stressor to salt marshes worldwide; however, this factor is frequently reversible. Tidally-restricted wetlands comprise roughly 20% of coastal wetlands on the Atlantic Coast of the US. In this region, it is likely that significant permanent reductions in methane emissions, and enhanced carbon storage can be achieved by restoring tidal flows (see Section 5.4.1) (Kroeger et al., 2017). Such actions also increase the capacity of the ecosystems to maintain elevation relative to sea level. The scale of benefits depends on how completely tidal restrictions can be reversed as well the duration of prior alteration (Emery and Fulweiler, 2017). Areas for future research include the potential for marsh plants to locally remediate ocean acidification (via CO_2 uptake) or deoxygenation (through O_2 production/transport), as is currently considered for seagrasses (Garrard et al., 2014). Seasonal extremes may, however, be increased, reducing potential benefits (Sabine, 2018).

Substantial evidence supports with *high confidence* that salt marshes will continue to reduce their area in response to sea level rise in the coming century, especially if landward migration is further constrained by human modification of shorelines. Plants with low tolerance to flooding are particularly vulnerable and they can be locally extinguished. The loose of vegetated coastal ecosystems causes a reduction in carbon storage with positive feedbacks to the climate system (*high confidence*). Furthermore, the flooded area can become a mudflat or be colonized by more tolerant, invasive species (e.g., *Spartina alterniflora*, C3 plants), whose expansion is favored by combined effects of warming, rising CO_2 and nutrient enrichment (*medium confidence*). These shifts in community structure denote severe consequences in the ecosystem functioning and services.

5.2.2.3.3 Mangroves

Mangroves are vital but fragile coastal ecosystems found within sheltered areas in tropical and sub-tropical coasts around the world (Spalding, 2010). 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) (Figure 5.11). 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).

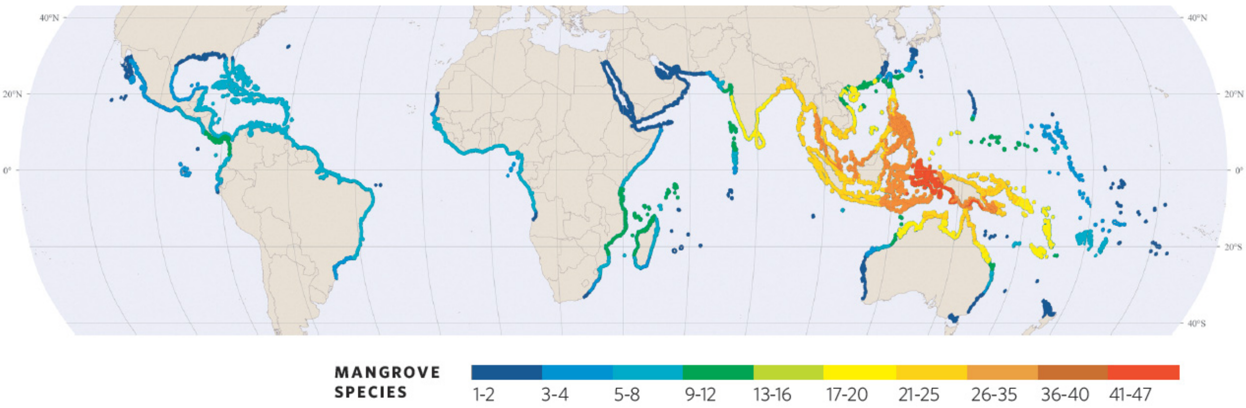


Figure 5.11: Global distribution of mangroves showing species diversity.

While over-exploitation of mangrove wood products, and conversion of mangrove areas into other lands uses are the major mangrove threats in the world, climate change will likely exacerbate mangrove loss. Mangrove ecosystems are particularly vulnerable to rising sea levels, increasing temperatures, and anoxic conditions brought about by sporadic sedimentation due to extreme rainfall (Alongi, 2008; Bosire et al., 2008; Gilman et al., 2008). A rise in sea level from the greenhouse is predicted to increase flooding of the low-lying coastal areas and drawn mangroves. Where accretion of the sediments is sufficient and topography ideal, mangroves will migrate towards higher elevations (Cahoon et al., 2006; Lovelock et al., 2010). Increased salinity caused by sea level rises may also result in decreased productivity and stunted growth in certain species. The expected increase in mangrove growth under high CO_2 conditions has been experimentally observed, also an unexpected change in salinity optima, favouring more saline conditions (Reef et al., 2015)

Mangroves operate in the C3 pathway of carbon fixation for photosynthesis. Research indicates that increases in atmospheric CO_2 increase the productivity and efficiency of water use by C3 plants. Thus it is anticipated that projected increases in CO_2 will enhance mangrove tree growth and litter production.

Overall, effects of climate change on mangroves will negatively impact on coastal protection, agriculture, aquaculture and forestry activities (Table 5.2). Mangrove forestry activities will for instance be affected by changes in phenology, aridity, salinity and direct disruption of specific sites by sea- level rise and storms. Mangrove scientists and managers should closely monitor the status and responses of mangrove forests to climate change for early indication of change and thus employ appropriate management interventions.

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 productivity Recruitment Inundation period Sedimentation rates	Forest mortality, dieback from the seaward edge Migration landward, but dependent on sediment inputs, topography and human modifications	(Ellison, 1993; Semeniuk, 1994; Cahoon et al., 2006; Gilman et al., 2007; Gilman et al., 2008; Soares, 2009; Ellison and Zouh, 2012)
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 CO2	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)

5.2.2.3.4 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 disturbances, for instance, seagrass meadows form highly refractory carbon deposits that can be store over thousands of years within sediments (Pergent et al., 2014; Hyndes et al., 2016; York et al., 2017), while they also offer protection to neighbouring coral reefs by their buffering ability against ocean acidification (*low confidence*) (Camp 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 antrophogenic drivers is limiting the tolerance of plants to rising temperature and biological invasions, and is causing a worldwide decline of seagrass area (*medium confidence*) (Tomas et al., 2015; Pagès et al., 2017).

Responses of seagrasses to climatic stressors change among species/genera, according to their life-history strategies (colonizing, opportunistic and perennial species) and climatic affinities (Rasheed et al., 2014; Kilminster et al., 2015; O'Brien et al., 2017; York et al., 2017). Certain seagrass species are expected to colonize new areas, i.e., 'winners', while others are expected to retract, i.e., 'losers'. Overall, warm-temperate regions, such as Western Australia, are the most vulnerable to loose seagrass area and the associated habitat complexity (Hyndes et al., 2016). Synergistic interactions of climatic stressors, e.g., heat waves and increased turbidity associated with floods, have been shown to cause drastic changes in the composition and biomass of co-occurring seagrass species, potentially eroding provision of ecosystem services (*high confidence*) (Fraser et al., 2014; Nowicki et al., 2017). An example of a seriously damaged seagrass ecosystem following a marine heatwave (see Box 5.2) is in Shark Bay, Australia, which has lost 36 % of the meadows dominated by *A. Antarctica* and to a minor extent by *Posidonia australis* (Arias-Ortiz et al., 2018). While Shark Bay seagrasses occupy an area of ~4% of the Australia´s vegetated coasts, they account for a 9% of the C sequestration by seagrasses, mangroves and tidal marshes. The losses of seagrasses due to thermal stress impacts reduce the sediment C stocks and enhance CO_2 emission due to oxygenation of the eroded sediments (Section 5.3.2).

In the iconic Mediterranean seagrass meadow formed by *Posidonia oceanica,* warming causes a number of negative indirect effects (*medium confidence*) such as increased toxicity of chemical hazards such as sulphide (García et al., 2013), and intense competition and predation by the arrival of tropical species (Pergent et al., 2014). It also reduced the fitness of seedlings by increasing necrosis and susceptibility to consumers and pathogenic pressure, while reducing establishment potential and nutritional reserves (Olsen et al., 2016; Hernán et al., 2017). However, temperature-driven flowering of *P. oceanica*, has been advocated as way of increasing resilience to warming (Ruiz-Frau et al., 2017).

Under increased CO_2 levels in the future, seagrasses are predicted to physiologically perform better, as photosynthetic rates increase (Koch et al., 2013; Borum et al., 2015). However, there is no consensus on long-term consequences of increased CO_2 on seagrass carbon budgets and the stoichiometric composition of seagrasses (Russell et al., 2013). Acidification in the future, moreover, will not counterbalance lethal effects of warming on the physiological fitness of certain temperate species, such as *Zostera noltii* (Repolho et al., 2017).

Responses of seagrasses to anthropogenic-mediated perturbations (e.g., CO_2 increments and fertilization) further change the feeding behaviour of herbivores, and can result in their community (*medium confidence*) (Tomas et al., 2015). Importantly, the intensity of herbivory over seagrasses is globally expected to increase with global warming, particularly in temperate areas, because of the migration of tropical herbivores into temperate seagrass-dominated environments (Hyndes et al., 2016). In turn, intense activity of tropical herbivores reduces the production of refractory organic matter and thus the bulk carbon sequestration (Heck et al., 2015; Hyndes et al., 2016).

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

5.2.2.3.5 Sandy beaches

Sandy beaches comprise the world's longest land-sea interface and play an important role in society, including recreation, tourism and commercial extraction of living or non-living resources. As assessed in AR5 WGII (Wong et al., 2014b), the worldwide pervasive coastal squeezing, which leaves sandy beaches trapped between rising sea levels on the wet side and urbanization from expanding human populations on landside, will continue to limit the ability of these ecosystems to compensate climate threats (*high confidence*). One of the most important compensatory responses of sandy beaches to erosive processes is the relocation of sediments that facilitates profile migration, but this response is constrained by the reduction of effective beach area. Along with sea level rise, severe erosive events such as storms surges, flooding and onshore winds, challenge the conservation of sandy beaches, accelerating dune scarping with vegetation loss (Castelle et al., 2015) Also, excessive precipitation driven by climate change (Westra et al., 2013) will also accentuate erosion, since the escape of this water to sea carries sand with it and causes a rise in groundwater levels.

The Southwestern Atlantic Ocean (SAO) has been identified as a major global-warming hotspot, where SST is increasing at several times the average global rate (Hobday et al., 2016). The position of the warm water front (represented by the 20ºC isotherm, a proxy for the front of tropical waters) showed a consistent long term poleward shift at a rate of ca. 9 km·y^{-1} (Ortega et al., 2016). This was accompanied by an increase in speed and frequency of onshore southern winds (Escobar et al., 2004; Bischoff, 2005; Ortega et al., 2013), which produced high energy waves that induced morphological changes in sandy beaches. An increase in the frequency and duration of southeast (onshore) storm surges (Escobar et al., 2004; D'onofrio et al., 2008) determined an increase in frequency and height of the waves (Codignotto et al., 2012) that strengthened the swash width and augmented erosion rates (Ortega et al., 2013; Gutierrez et al., 2016). There is *high confidence* that morphological changes resulting from this erosive process persist, suggesting a reduced capacity for recovery of the subaerial profile of sandy beaches. For the Pacific Ocean basin, a 33-year analysis for 48 beaches showed that observed erosion across Pacific coasts varies most closely with El Niño-Southern Oscillation (ENSO) (Barnard et al., 2015). Considering an increasing projected frequency of extreme ENSO events, opposite coastal sides of the Pacific Ocean basin could accelerate erosion rates and flooding, independent of sea level rise (Barnard et al., 2015). Moreover, predictions of shoreline change indicate that 31%–67% of southern California beaches may become completely eroded by 2100 under SLR scenarios of 0.93–2.0 m (Vitousek et al., 2017).

Evidence from long-term observations and laboratory experiments of South American sandy beaches suggest that macrofauna is reorganizing under the influence of climate change **(**Table 5.3**)**. Coherence in responses across taxa and geographical regions, via global meta-analyses, supports with *high confidence* that climate change is a potential primary causal agent of these changes (McLachlan and Defeo, 2017). Mass-mortality events of cool-water clam species are being documented as a response to warming waters, and the low recovery of the populations denotes a poor adaptive capacity to respond to the changing climate. Mass mortalities have negative socio-economic implications, particularly in those cases where recreational and local fisheries are developed (Defeo et al., 2013; Aburto et al., 2014; Gianelli et al., 2015; Turra et al., 2016). Opposite responses in macrofaunal warm water species with different biogeographic origins were observed, resulting in a tropicalization of filter feeding guilds on both the Pacific and the Atlantic coasts of South America (*high confidence*). The ecological responses in demographic and health features presented in Table 5.3 suggest that sandy shore assemblages have exceeded critical biophysical thresholds (*tipping points*), triggering regime shifts in the sandy beach ecosystem (*high confidence*; (McLachlan and Defeo, 2017)).

Additional evidences from other geographical regions support these findings.The poleward-range edge of adults of the ghost crab *Ocypode cordimanus* in the east coast of Australia has extended its leading edge 270 km south of the previous southernmost record, and recruitment was observed in beaches 248 km (along the coast; 0.9 degrees of latitude) beyond the adult-range edge previously documented Schoeman et al. (2015). This range edge does fall within a "hotspot" of ocean warming, where surface isotherms are moving southward along the coast at 20–50 km/decade and coastal air temperatures in the region are also warming. The northern range limit of the isopod *Tylos punctatus* in sandy beaches of California shifted southward as a result of habitat fragmentation mainly due to intense urbanization and habitat loss (Hubbard et al., 2014). This upper beach species will have little possibility for landward retreat if California beaches (narrow, bluff-backed beaches) are further affected by sea level rise associated with climate change: <10% of the 450 km of southern California coast will have the potential to provide suitable upper beach habitat under a 1.40 m sea level rise scenario by 2100.

Table 5.3: Lines of evidence summarizing the potential effects of climate change in South American sandy beach macrobenthic communities. Based on the theoretical framework detailed in (Parmesan et al., 2013). Extracted from McLachlan and Defeo (2017).

Lines of evidence	Attribution process	References
Long-term observations and resurveys	Mass mortality events of *Mesodesma donacium* clam across the northern portion of its range (Perú and Northern Chile) consistently follow warm sea surface temperature events in the Pacific (e.g., El Niño)[1] and occurred sequentially in a north-south direction in the Atlantic (*M. mactroides*), following oceanographic shifts, increasing SST and isotherms moving poleward[2]. 30 years of systematic sampling in Uruguay[2] and resurveys in Brazil and Argentina[3] across the distribution range of the clam *M. mactroides* document declines and local extirpations related to increasing SST and associated factors.	1. Riascos et al. (2009) 2. Ortega et al. (2013); Ortega et al. (2016) 3. Herrmann et al. (2009); Adams et al. (2011); Herrmann et al. (2011)
Fingerprints	Population extirpations along the northern (trailing) range edge and lowest levels of impact along southern (leading) range boundaries[1,2,4], uniquely consistent with regional warming in the Pacific and Atlantic oceans and not with local human-caused stresses. Mass mortalities of yellow clams occurred during warmer seasons throughout its entire distribution range [5], concurrently with accelerating SST warming.	4. Riascos et al. (2011) 5. Fiori et al. (2004)
Meta-analyses: global coherence of responses across taxa and regions	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. *Mesodesma* clams were virtually extirpated at their northern range edges, being partially replaced by *Donax* clams and *Emerita* mole crabs (tropical affinities) in both Pacific[6,7] and Atlantic[8,9,10] sandy beaches. Impoverished macrofaunal communities and *Mesodesma* species never reached biomasses at pre-mortality levels[7, 9,11] Parasites were found in clams, and also necrosis in gills and stomachs, during mortality events in the Atlantic (Argentina[4] and Uruguay[2]) and in the Pacific during strong El Niño events[6] Body downsizing was consistently observed for the guilds of intertidal suspension feeders in Atlantic[2,8] and Pacific[1,6] sandy beaches of South America.	6. Riascos et al. (2011) 7. Arntz et al. (1987); Arntz et al. (2006) 8. Celentano and Defeo (2016) 9. Dadon (2005) 10. Herrmann et al. (2009) 11. Lercari et al. (2018)
Experiments	Controlled in-vitro experiments showed that *D. obesulus* (tropical affinities) was tolerant to El Niño temperatures, whereas *M. donacium* (temperate) was sensitive to both El Niño and La Niña extreme conditions, with sublethal and lethal effects[12]	12. Carstensen et al. (2010)

Long-term changes in phytoplankton composition and abundance in the surf zone of sandy shores have been related to extreme pulse perturbations like El Niño 1997/1998 Odebrecht et al. (2010); (Odebrecht et al., 2014). Additionally, increasing occurrence of harmful algal blooms (HABs) has been related to interactive effects of climate warming and eutrophication, with tropical species being increasingly represented in coastal subtropical ecosystems (*medium confidence*: (Martínez et al., 2017)). Exploitation of beach clams is often constrained by HABs, which can cause mass mortalities or render clams unsafe for human consumption Gianelli et al. (2015). Eutrophication has also led to the mass development of mats of drifting algae along sandy beaches that deteriorated the environment and the resident macrofauna (Quillien et al., 2015; Quillien et al., 2016). Detached algae invade the water column and are able to increase their nutrient supply, building up large biomass and forming massive green (*Ulva*) and golden (*Sargassum*) tides (Smetacek and Zingone, 2013). The frequency, intensity and periodicity of these mass stranding events have been increasing worldwide (Ye et al., 2011) and have triggered high economic losses associated with beach deterioration and the consequent impacts on tourism and fisheries.

The concurrent effect of multiple stressors (sea level rise, coastal squeeze and the consequent erosion of the narrow physical habitat) are affecting turtle nesting sites in sandy beaches. Warming could also affect the reproductive output, because sea turtles have temperature-dependent sex determination (Hays et al., 2003). As incubation temperatures near lethal levels, growth and survival rates of turtle populations will tend to decrease in the long-term (*medium confidence*) (Laloë et al., 2016; Laloë et al., 2017). Shorebird richness and abundance may decrease in the long-term due to a combination of habitat loss, decreased accessibility because of SLR and reduced prey availability, particularly when the upper shore and sand dunes are modified by engineering interventions that place armoring structures on beaches (Dugan et al., 2008). SLR is expected to affect suitable breeding sites for birds, particularly in the case of long coastlines with extensive low-lying coastal areas. Similarly, tropical cyclones (Convertino et al., 2011) and ENSO events (Hubbard and Dugan, 2003), which may become more extreme due to climate change (Cai et al., 2014), will tend to decrease nesting and shorebird abundance (*medium confidence*).

The above events have modified beach food webs, as well as ecosystem structure and functioning. For example, the prevalence of warm waters and increasing onshore winds in a sandy beach in Uruguay caused a decrease in primary productivity that affected food availability for macrofauna, and therefore its biomass (Lercari et al., 2018). Similarly, drastic ecosystem responses to the 1997/98 El Niño event have been recorded along Californian beaches (Revell et al., 2011), which include (1) short-term reductions in beach width (60%), loss of sand volumes in dune-backed beaches, (2) reduction of beach width in bluff-backed shorelines, which in turn caused a reduction in macrophyte wrack abundance; (3) a reduction in biomass and in individual size of macroinvertebrates; and (4) lower abundance and species richness of shorebirds.

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

5.2.2.3.6 Coral reefs

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

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

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

In contrast, the literature (DeBiasse and Kelly, 2016; Gibbin et al., 2017; Wall et al., 2017; Camp et al., 2018; Donelson et al., 2018; Drake et al., 2018; Rose et al., 2018; Veilleux and Donelson, 2018) attests that corals as other marine organisms possess the cellular, physiological and molecular machineries that help compensating for the effects of global change (*medium confidence*). These outcomes and the existing critical gaps in our understanding of the cumulative global change impacts on coral reefs point to the conclusion for increased uncertainty in predicting the spectrum of future responses by reef corals to climate change (Bell et al., 2013; McClanahan et al., 2014; Mumby and van Woesik, 2014; Pandolfi, 2015; Folkersen, 2018) (*high confidence*).

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

There are almost no major reef systems that are not vulnerable to climate change (*high confidence*). An example of reef vulnerability is the Coral Triangle, the global pinnacle of tropical coral biodiversity, which has experienced thermal stress followed by extensive coral bleaching and coral mortality events are directly associated with extremes in the El Niño-Southern Oscillation (ENSO) (i.e., 1998 and 2010). Above all, the coral systems in tropical small islands are at major risk of being already severely affected by current climate change impacts, including both extreme events and gradual environmental changes, such as sea level rise (Duvat et al., 2017; Harborne et al., 2017) (*high confidence*).

5.2.2.3.7 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 algal-dominated shallow, subtidal ecosystems.

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

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

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

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

5.2.2.3.8 Kelp forest

Kelp forests are three-dimensional highly productive coastal ecosystems with a global net primary production (NPP) between 1,020 and 1,960 TgC yr–1 (Krause-Jensen and Duarte, 2016), and an extension of approx. 25% of the worlds coastlines (Filbee-Dexter et al., 2016), (Krause-Jensen and Duarte, 2016), (Filbee-Dexter et al., 2016), from temperate to polar (see Chapter 3) rocky reefs (Steneck et al., 2003). Around 43% of the NPP is exported either as DOC (52%) or POC (48%) to neighbouring soft sediment habitats, deep canyons and the deep sea where it provides additional food source for invertebrates and fishes, or it is buried in the sediments and thereby contributes to global CO_2 sequestration (Krause-Jensen and Duarte, 2016).

Two global datasets and one dataset spanning European coastlines (Araujo et al., 2016; Krumhans et al., 2016; Poloczanska et al., 2016) identify large local and regional variability of kelp abundance, with a small instantaneous rate of change of -0.018 yr^{-1} displaying 38% of decline, 27% of increase and 35% of no change in the ecoregions (Krumhans et al., 2016). The average decline is much smaller than any observed regional loss which is a reflection of the naturally high spatio-temporal variability and resilience of kelp forests (Cavanaugh et al., 2011; Reed et al., 2016). A consistent negative response to regional warming between 1950 and 2009 in the abundance, phenology, demography or calcification of macroalgae on a global scale became evident for the northeast Atlantic and southeast Indian Ocean (Poloczanska et al., 2016). Declines in kelp forest abundance attributed to climate change and not related to sea urchin overgrazing which is a major driver of decline (Ling et al., 2015) have been documented since the late 1980's and evidence has increased within the last 2 decades (Box 5.2). Conversely, climate related cooling events, e.g., along the southern African coastline (Bolton et al., 2012) or the warming of Polar sites led to a spread of kelp beds or algal biomass increase. There is a general lack of data from some regions such as southern America and also a lack of historical baseline information (Bolton et al., 2012; Poloczanska et al., 2016) but since AR5, substantially more observational evidence supports that loss of kelp forest systems is associated with a phase shift from a 3-dimensional highly diverse kelp habitat to a less diverse and less complex turf forming habitat (*high confidence*) (Filbee-Dexter et al., 2016). Further evidence supports model predictions (Raybaud et al., 2013; Assis et al., 2016) showing that range contractions of kelps at the warm end of distributional margins and expansions at the poleward end are ongoing at diverse sites globally (*high confidence*).

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

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

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

In the northwest Pacific, decline of kelp bed ecosystems have been observed since the 1980s, especially along the coasts of Japan which span from sub-tropical to temperate areas and are considered a global biodiversity hotspot (Tittensor et al., 2011). Along the coast of Japan seawater temperatures have risen by 1.08°C per century (1891–2012) which is more than the double of the mean global ocean temperatures. In 1989–1991 the total area of seaweed beds covered 2012 km² while it decreased by nearly 30% in 1998 ((Takao et al., 2015) (*medium confidence*). The southern kelp beds of Japan are inhabited by one of the most warm adapted kelp species, *Ecklonia radiata*, which can still survive up to 27°C Takao et al. (2015) although temperatures above 20°C will negatively impact the productivity (Serisawa et al., 2004). *Ecklonia* beds have rapidly declined since the 1990's, and partially disappeared by 2000 in southern Japan mostly attributed to the overall increase of seawater temperatures, considerably affecting abalone fisheries which depend on *Ecklonia* as food source (Serisawa et al., 2004). Along a coastline of approx. 700 km length in southwest Japan, a poleward expansion of tropical species and a retraction of temperate species follow an increase of mean annual SST of 0.3°C per decade between 1970–2009 ((Tanaka et al., 2012) (*high confidence*). A significant reduction in size of potential habitats is projected under all four RCPs using sea surface temperature projected from CMIP5 models, in combination with a species distribution model of kelp (*Ecklonia*) and its main herbivorous fish species Takao et al. (2015). The reduction for 2090 is projected to be 15% in the RCP2.6 scenario, but up to 55%, 75% and 85% habitat loss compared to the present-day extent under RCP 4.5, 6.0 and 8.5 scenarios, respectively. The loss process is expected to accelerate from 2020 onwards ((Takao et al., 2015); *medium confidence*). Even the kelp species that are most adapted to warmer waters, known in the order Laminariales, *Ecklonia radicosa*, apparently disappeared from its former southern distribution limit in Kagoshima Bay concomitant with the 1°C increase in mean seawater temperatures in the past four decades Komazawa et al. (2014); (Terada et al., 2016).

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

In summary, kelp forest ecosystems will continue to decline in temperate regions and to expand poleward driven by warming (*high confidence*). Subtropical invasive kelp species will continue spreading facilitated by ocean warming and rising CO_2 (*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 enhanced grazing by warming cause physical and physiological stress to kelps, reducing their population fitness, e.g., reproduction and productivity (*high confidence*).

[START BOX 5.2 HERE]

Box 5.2: Retraction of Kelp Forests in Australian Coastlines Driven by Heatwaves

In the last decade, steadily increasing seawater temperatures have leaded to a retraction of kelps (*Ecklonia radiata*) at their warmer (northern) limits in the western and eastern Australian coastlines (Wernberg et al., 2011; Wernberg et al., 2012). Originally the western Australian coastline was dominated by approx. 2300 km² of kelp forests (27.7°S–30.3°S) covering approx. 70% of the shallow reefs (Wernberg et al., 2016). Since the 1970s these kelp forests has experienced rising seawater temperatures (Wernberg et al., 2012) culminating in a series of heatwaves. In early 2011, the most severe warming in the past 140 years impacted this long stretch of coastline. Seawater temperatures reached unprecedented levels and 2°C–4°C warming anomalies were present for more than 10 weeks (Wernberg et al., 2012). Immediate community level responses were not the same along the whole coastline, probably due to different absolute temperature thresholds that were surpassed. In the warmer northern part mean summer temperatures between 2006-2010 were 22°C–23°C while they increased to 24°C–27°C during the 2011 heatwave. In the cooler southern part the temperature increase during the heatwave only reached levels comparable to the 2006–2010 situation in the North (22°C–24°C). In the more northerly ‘warm’ region with species living at their physiological limit (Wernberg et al., 2016) the changes were considerable: the cover of habitat engineering macroalgae (kelp and fucoids) declined from approx. 90% to about 55%, while the abundance of tropical fish species considerably increased. Encrusting coralline algae also decreased from approx. 35% to about 15% cover while the cover of turf forming algae more than doubled (Wernberg et al., 2012). Thus, in the ‘warm region’ the heat-wave completely altered the community structure. In the more southerly ‘cool region’ the typical temperate communities probably embraced a few marginal populations and the heat wave did not surpass their temperature threshold limits (Wernberg et al., 2012; Wernberg et al., 2016; Phelps et al., 2017).Two years after the heatwave, in 2013, the previously dense kelp forests disappeared in the ‘warm region’ north of 29°S and even five years later in 2015 no recovery occurred (Wernberg et al., 2016). The inability of recovery is attributed to a combination of lethal temperature thresholds that were surpassed during the heatwave (Wernberg et al., 2016), altered competition and a constant supply of tropical propagules and immigrants which are delivered by the expansion of the poleward tropical current to this system (Leeuwin Current) (Wernberg et al., 2016). The tropicalization of the reef above 29°S continues as expected even under the moderate SRES A1B emission scenario (Bartsch et al., 2012) and kelp forests are retracting towards the southern Australia coastline (Burrows et al., 2014). Thus, the east-west orientation of the southern Australian coastline makes it one of the most vulnerable coastal areas worldwide where catastrophic extinction of kelp forest communities over wide areas is projected if climate warming continues (Wernberg et al., 2016).

Also further south, the eastern coastline of Tasmania is affected by intrusion of warm, nutrient poor water masses by the intensifying East Australian Current (EAC) which is now stretching approx. 350 km further south than in the 1950s. The associated coastal warming led to a regional decline of the dense beds of giant kelp (*Macrocystis pyrifera*) which became obvious after 1980 (Johnson et al., 2011). Despite large interannual fluctuations, the reduction of kelp cover decreases along the north-south axis of eastern

Tasmania and thereby follows the diminution of the warm current. Although warming seems to be the major driver, the effects may be confounded by low nutrients which are associated to the EAC (Johnson et al., 2011) as *Macrocystis* is not readily able to store nitrogen (Gerard, 1982). Furthermore the general change in kelp and other macroalgal habitats off Tasmania has to be seen in interaction with intensified grazing by a sea urchin species that only recently established (Johnson et al., 2011).

[END OF BOX 5.2 HERE]

5.2.2.3.9 Fjords

Fjords account for <0.1% of the marine surface area (Nuwer and Keil, 2005; Keil, 2015), but represent a significant interface and a buffer area between freshwater systems (i.e., lakes, rivers), glaciated continents, and the adjacent coastal ocean (Syvitski et al., 2012). Consequently, fjords constitute a typical case requiring a landscape-wide approach to ensure a thorough understanding of ecosystem functioning and vulnerability under climate change scenarios (Vargas et al., 2011). Fjords areas are located in polar (see Chapter 3) and subpolar regions, many of them receive small anthropogenic influence because of their remoteness, whereas those that are located near population and industrial centres are impacted by human activities (Skei et al., 2000). Climate change will most likely influence fjord ecosystems both in the ends opening out into the sea, and the inner ends abutting the glaciers or river runoff, rendering these ecosystems a sensitive indicator of climate change (Svendsen et al., 2002). Recent evidence suggests freshening and warming of polar fjords (Holland et al., 2008; Holland, 2010) (*medium confidence*), causing increases in boreal-subarctic species and decreases in polar species and altering community structure and functions (see Chapter 3).

For temperate regions, decrease in temperature and salinity, and increase in suspended sediments from enhanced freshwater inputs may affect primary production and community structure in fjords; however local adaptation to naturally variable environment may increase their adaptive capacity to climate change impacts (*low confidence*). For example, modelling projections suggest increased sediment loads from larger glacial-lake outburst sediment floods due to glacier melting in Chilean Patagonia, which reduces light penetration within the inner fjord's area (Baker Fjord) and eventually primary productivity in this region (Marín et al., 2013) *(low confidence*). Also, the combined effect of cold and low alkalinity waters due to ice melting of calving glaciers in Chilean fjord environments may create conditions of corrosive waters for $CaCO_3$, with important consequences for marine calcifying organisms in Southern Patagonia (Vargas et al., 2018). Similar evidence resulted from mesocosm studies in a Norwegian fjord, where expected ocean pCO$_2$ levels can reduce the seawater concentrations of the climate-active compound dimethylsulfide lowering calcifying phytoplankton productivity, which in turns may create a positive feedback to the climate system (Riebesell et al., 2016)*(medium confidence*). In the Baltic Sea, warming and increased freshening may cause a reduction in scyphozoan jellyfish (*e.g., Cyanea capillata)* that is physiologically less tolerant to waters lower in salinity and pH (Goldstein et al., 2017).

Fjord environments are in occasional naturally acidified habitats, and some marine organisms, such as bivalves, seem to create local adaptation capacity to future high pCO$_2$/low pH conditions expected for year 2100 in the Baltic Sea and southern Chile (Thomsen et al., 2017; Vargas et al., 2017). For example, some populations of marine invertebrates (e.g., mussels, gastropods, planktonic copepods) are adapted to local conditions of low pH/high pCO$_2$ in tidal inlets and fjords of North Patagonia. Similarly, toxic dinoflagellates, such as *Alexandrium catenella* are highly adapted to spatio-temporal pCO$_2$/pH fluctuations in Chilean fjords, and could become a resilient winner from expected climate change effects (Mardones et al., 2016).

Fjords from higher latitudes are the most vulnerable to warming, where glacier retraction implies, with *high confidence*, that most fjord ecosystems will continue to freshen in the 21st century. Further, warming will continue causing shifts in community composition, from dominance of polar to sub-polar species (*high confidence*). The enhanced inflow of terrestrial material due to snowmelt and glacier ice melt reduces the underwater light availability. Hence, changes in water turbidity and salinity will reduce phytoplankton productivity (*medium confidence*), whereas responses of particular planktonic organisms (e.g., toxic dinoflagellates and jellyfish) will vary depending on their physiological tolerance (*low confidence*). Calcifying organisms, both from pelagic and benthic habitats, will be most vulnerable to the lower alkaline waters provided by ice melt with negative consequences for their biomass and productivity (*medium confidence*).

5.2.2.3.10 Summary for coastal ecosystems

Overall, all coastal ecosystems are under stress from the combination of climate change impacts in the ocean and on land as well as non-climatic human activities, with observable and projected climate impacts through modification of geomorphology, reduction in habitat area, and shifting distributions and community structure (*high confidence*) (Section 5.2.2). Coastal ecosystems reflect a heterogeneous network of habitats that provide critical ecosystem services ranging from shoreline protection and fisheries support, to carbon sequestration and tourism (Sections 5.2.2, 5.3.1). However, sandy beach and saltmarshes morphology is changing as a result of rising sea level and more severe storms that leads to their erosion and retreat (Sections 5.2.2.3.4, 5.2.2.3.5). These consequently lead to losses of vegetation, benthic fauna (*high confidence*), with detrimental consequences for saltmarshes carbon storage (*high confidence*) (Section 5.3.2.3). The increased levels, frequency and duration of eutrophication and hypoxic conditions in estuaries and coastal zones worldwide in the last few decades are attributable to warming (*high confidence*), with projected impacts being most pronounced in the high latitudes (*high confidence*) (Section 5.2.2.3.1**)**. Synergism between acidification and warming exacerbates the negative effects on rocky reef communities, causing a shift towards a less diverse ecosystem in terms of species richness and spatial heterogeneity (i.e., rocky reefs dominated by macroalgae) (*medium confidence*) (Section 5.2.2.3.7). Kelp forest and seagrass meadow ecosystems will continue to shift poleward as increased temperature and extreme events such as heat waves and storms (*high confidence*), and enhanced grazing by warming will result in physical and physiological stress, reducing their reproduction and productivity (*medium confidence*) (Sections 5.2.2.3.4, 5.2.2.3.8). Ice and snow melting will continue freshening fjord ecosystems and increasing water turbidity, causing shifts in pelagic and benthic communities (*high confidence*), where calcifying organisms will be more vulnerable due to lower alkaline melting water intrusions (*medium confidence*) (Section 5.2.2.3.9).

5.2.2.4 Deep Sea Floor Systems

5.2.2.4.1 Defining the deep

The deep sea is assessed here as the vast area of the ocean >200 m, deeper than the continental shelves (Figure 5.12). Below 200 m changes in light, food supply, and the physical environment lead to altered animal taxonomic composition, morphologies, lifestyles, and body sizes collectively understood to represent the deep sea (2003). Although (2014) defined the deep sea as below 1000 m, the absence of photosyntheticaly useful light and ensuing critical ecological changes, biogeochemical transformations, and altered human interactions that occur on the sea floor below 200 m have led both pelagic and benthic biologists to include the 200-1000 m realm within the deep sea (Herring and Dixon, 1998); Gage (2003).

5.2.2.4.2 Detection and attribution of oceanographic and biological changes in the deep sea

The deep sea (below 200 m) is vast and remote, difficult to access and expensive to study and as a result hosts relatively few long biological time series or experiments suitable for assessment of climate impacts (Smith et al., 2013). However, there are long-term records of physical change derived from deep observing programs (see Section 5.2.1.2.1) from repeat hydrography (Talley et al., 2016) to > 6000 m) and autonomous float instruments to 2000 m (Roemmich et al., 2015). The paleo record holds information about hydrological, biogeochemical and biological change on historical or geological time scales and contributes understanding valuable for interpreting modern responses to climate change. To approach the ecosystem changes for the deep sea, this section (Section 5.2.2.4) relies on (a) observed and projected changes in environmental exposures to climate drivers relative to natural variation (climate hazard), (b) definition of habitat requirements and tolerance thresholds from current species distributions and habitat suitability modeling, (c) mechanistic understanding derived from experiments with shallow water relatives or with eurybathic deep-sea species reared in the laboratory, (d) observed ecological, physiological or trophic responses to drivers based on long-term, time series data or spatial variation across natural gradients, (e) ecological insights from fossils in the paleo record, and (f) meta analyses of driver impacts and their interactions. Taken together they provide substantial evidence for future changes to most deep-sea ecosystems.

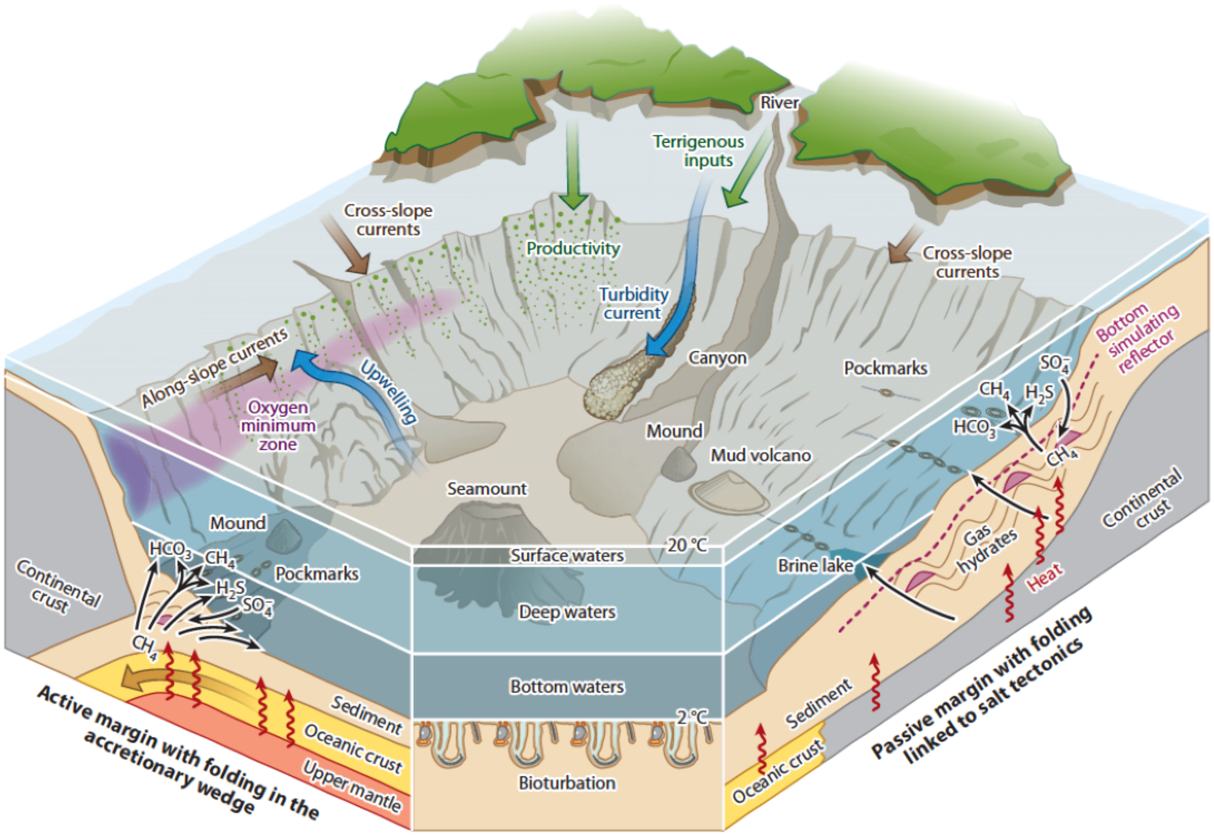


Figure 5.12. A conceptual diagram illustrating how climate drivers operating as inputs from land, atmosphere, and the crust alter upwelling, surface production, circulation, and fluid flows with the potential to modify deep-sea ecosystems.

5.2.2.4.3 Changes on the deep-sea floor

Large areas of the deep-sea floor will be exposed to rising temperature (see Section 5.2.1.2) and declining oxygen, pH and POC flux over the next 50 to 100 years (see Sections 5.2.1.3 to 5.2.1.5) (Sweetman et al., 2017; FAO, 2018) (*high confidence*) with most values exceeding natural variation by 2060 (FAO, 2018) (*medium confidence*). The greatest cumulative negative exposures will be in the Arctic and Atlantic Oceans over the next 100 years.

Projected changes in different ecosystems

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, O_2, 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 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 [Convention on Conservation of Antarctic Marine Living Resources (CCAMLR), General Fisheries Commission for the Mediterranean (GFCM), North Atlantic Fisheries Organization (NAFO), Northeast Atlantic Fisheries Commission (NEAFC), North Pacific Fisheries Commission (NPFC), South-East Atlantic Fisheries Organization (SEAFO), and South Pacific Regional Fisheries Management Organization (SPRFMO)], these metrics were evaluated 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.

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

Biological responses to stressors

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

Deoxygenation in the present and past derives largely from warming in the open ocean (Keeling et al., 2010; Levin, 2018). For example, in the Santa Barbara Basin sediments, O_2 changes are associated with Dansgaard Oeschger events (rapid climate fluctuations) and abrupt de glacial warming (Cannariato and KENNETT, 1999; Moffitt et al., 2015). The warming and oxygen loss coincide with shifts from diverse invertebrates and multiple feeding guilds to extremophile invertebrates and eventually to extremophile foraminifera (*medium confidence*), accompanied by increased reliance on chemosynthesis (Moffitt et al., 2015) (*medium confidence*). Oxygen loss affects deep seafloor taxonomic composition, taxon-specific abundances, biomass, body size, diversity, physiology, reproduction and behavior (Levin, 2003; Breitberg et al., 2015).

Ocean acidification, the availability of carbonate ions, and the aragonite and calcite saturation horizons (boundaries above which aragonite and calcite carbonate ions are saturated and below which they are undersaturated, making calcification more energetically costly) exert strong control on the calcification rates of organisms, the energy required and the dissolution rates of living and dead carbonate skeletons, tests and sediments which form habitat in the deep sea. While ocean acidification is recognized to influence a suite of behavioural and physiological responses in shallow-water taxa, including growth rate, metabolic rates, mode of catabolism, development, gene expression, energetics and more, most of these have not been studied in deep-sea species. A meta analysis reported in AR5 Chapter 6 ((Pörtner et al., 2014), Table 6-3) identifies 10 studies involving 6 species of cold-water corals that suggest low vulnerability to CO_2 changes at RCP6.0 and medium vulnerability at RCP8.5, with negative effects starting at pCO_2 of 445 uatm. Nevertheless, oxygen and capacity dependent thermal tolerance is likely to integrate a broad range physiological responses to multiple climate drivers: warming-induced CO_2 accumulation in body fluids links to the effects of ocean acidification mediated by the weak acid distribution of CO_2 while temperature-induced hypoxemia links to the hypoxia sensitivity of thermal tolerance Portner (2012).

Decline in food supply (POC flux) to the deep sea cused by warming and subsequent stratification of the ocean, combined with slowing of thermohaline circulation will reduce the nutrient supply to the surface ocean, causing declines in export of primary production to the deep sea (Section 5.2.1.5) *(high agreement, limited evidence).* Reductions in POC flux are predicted at low and mid latitudes but increases are possible at high latitudes, associated with reduction in sea ice cover (Yool et al., 2013; Rogers, 2015; Sweetman et al., 2017; Yool et al., 2017; FAO, 2018).

The Coupled Model Intercomparison Project (CMIP5) used in the IPCC Fifth Assessment Report (AR5) found that, under a high greenhouse gas emissions scenario (RCP8.5), declines in primary production will lead to a global fall in export production from –7% to –16% (Bopp et al., 2013) (Section 5.2.1.5). When linked to ocean acidification, which shoals the calcite compensation depth and decreases production of biogenic calcium carbonate, the export production declines can be up to –40.7% at 1000 m (Yool et al.,

2013) or up to 38% at the northeast Atlantic Seafloor (Jones et al., 2014). Those environments already subject to food stress (e.g., abyssal plains beneath oligotrophic gyres) are likely to be most affected (Smith et al., 2008; Jones et al., 2014) (*high confidence*). Arctic and Southern Ocean POC flux 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).

Changes in POC flux are likely to alter the relative importance of the microbial loop and benthic invertebrates in carbon transfer (Dunlop et al., 2016) (*medium confidence*). Deep-sea biota demonstrate increased efficiency at low food inputs (due to small size and dominance by prokaryotic taxa), thus they should be better able to adapt than shallow biota to reduced POC flux with climate change (Gambi et al., 2017) (*low confidence*, based on E Mediterranean Sea ultra-oligotrophic sediments). Changes in the overlying mesopelagic and bathypelagic communities (see Section 5.2.2.2) will affect food flux to the deep sea, as nekton and zooplankton transfer energy through diel vertical migrations, ontogenetic migrations and falls of dead carcasses (Gage, 2003) (*medium confidence*). Abyssal responses to changes in POC flux are discussed in Section 5.2.2.4.4.

5.2.2.4.4 Abyssal plains

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

5.2.2.4.5 Continental slope sediments.

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

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). 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) (*high confidence*) and a simplification of trophic structure reducing energy flows to upper trophic levels (Sperling et al., 2013) (*medium confidence*) and shifting in carbon processing pathways from metazoans to protozoans (Woulds et al., 2009) (*medium confidence*). A loss of deep bioturbating faunas at lowest oxygen levels is *virtually certain*, and would reduce carbon burial rates (Smith et al., 2000; Levin and Dayton, 2009) (*low confidence*). Hypoxia reduces colonization potential and community resilience of slope biota (*low confidence*, one study) (Levin and Sibuet, 2012). Carbon sequestration and nitrogen recycling are highly sensitive to small changes in oxygenation (Deutsch et al., 2011) (*high confidence*).

Protozoans (foraminifera) are more tolerant to changing climate drivers on margins than larger invertebrates. Short-term experiments have showed no significant effect of OA on survival for *Ammonia* spp. (Dissard et al., 2010; Haynert et al., 2011; Keul et al., 2013), *Bulimina marginata* and *Bolivina argentea* (McIntyre-Wressnig et al., 2014) and *Globobulimina turgida* (Wit et al., 2016). However, lower pH exacerbates shallow foraminiferal sensitivity to warming (Webster et al., 2016) and shallow water (reef) foraminifera microbiomes are sensitive to rising temperature (Webster et al., 2016). In contrast, Foraminifera exhibit well-recognized changes in abundance and diversity in response to oxygen loss (Bernhard and Reimers, 1991; Gooday et al., 2000; Moffitt et al., 2014). Hypoxia in combination with elevated pCO_2 favors survival of some foraminifera (*Globobulimina*) (Wit et al., 2016), but combined declines in pH and oxygen may lead to increase in some agglutinating taxa and a decrease in the carbonate producing forams, including those using carbonate cement (van Dijk et al., 2017) (*low confidence*).

Among the metazoan meiofauna, nematodes can respond rapidly to temperature change; an anomalous temperature drop of only 0.4◦C in the Mediterranean deep sea caused a significant decrease in nematode abundance and functional diversity, concomitant with increased species richness and evenness (Danovaro et al., 2001; Danovaro et al., 2004), whereas nematode abundance and biomass can decrease with increasing sediment temperatures (Yodnarasri et al., 2008). Ice shelf collapse transforms nematode assemblages from depauperate, low-diversity communities, to richer and denser communities dominated by opportunistic species (Ingels et al., 2012). Elevated CO_2 affects nematode survival (Barry et al., 2004; Fleeger et al., 2006; Fleeger et al., 2010) and although tolerant to hypoxia as indicated by high abundances, their diversity declines significantly within low-oxygen sediments (Neira et al., 2001). Meiofaunal biomass is sensitive to POC flux (Smith et al., 2008). Reduced POC flux with depth causes Mediterranean nematodes to decline in body size, conferring efficiency (biomass:respiration) in exploitation of food resources under very low food conditions (Gambi et al., 2017).

Warming may dissociate gas hydrates buried beneath shelf and slope sediments (*low confidence*), fueling anaerobic methane oxidation and yielding more sulfidic sediments (Boetius and Wenzhoefer, 2013) (*medium confidence*) which promotes chemosynthesis-based ecosystems (methane seeps) (*high confidence*). Warming-induced changes in circulation are linked to abrupt changes in benthic ostracod diversity in the historical record (Yasuhara et al., 2008) (*medium confidence*). Gamete synthesis, skeletal structure and biochemical composition of deep-sea echinoderms are sensitive to ocean acidification (deep-sea holothurians - (Verkaik et al., 2016)) and to ocean deoxygenation (echinoids - Sato et al. (2017)). Shoaling oxyclines can cause changes in echinoid vertical distributions Sato et al. (2017) (one study, *low confidence*). However species in upwelling regions, such as the echinoid *S. fragilis*, that have adapted to oxygen minimum zones with characteristically high CO_2 levels are less vulnerable to the negative impacts of ocean acidification (Taylor et al., 2014). Where both oxygen and CO_2 stress occur together, oxygen can to be the primary driver of change (Taylor et al., 2014; Sato et al., 2017), but this is not always true (Gobler and Baumann, 2016).

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

5.2.2.4.6 Chemosynthetic ecosystems

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

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

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

At some vents, high CO_2 conditions limit mussel shell calcification, reduce shell weight and slow growth (in *Bathymodiolus septemdierum*) at low saturation states (Tunnicliffe et al., 2009; Rossi and Tunnicliffe, 2017) (*medium confidence*). However, the absence of predators reduces the need for calcification and CO_2 stress can be relieved by high food supply (through symbiont autotrophy including hydrogen-based metabolism) (*low confidence*).

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

5.2.2.4.7 Canyons

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

5.2.2.4.8 Seamounts

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

5.2.2.4.9 Biogenic structures: cold-water coral ecosystems.

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

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

Lophelia pertusa is a widely distributed, habitat-forming, foundation species, and is probably the best studied with respect to climate tolerances. In many instances *L. pertusa* can continue to calcify at aragonite undersaturation (Georgian et al., 2016; Kurman et al., 2017) (*medium confidence*) and can acclimate to rising CO_2 levels (Hennige et al., 2015) (*low confidence*). It can also calcify when exposed to multiple environmental stresses in the laboratory (Hennige et al., 2015; Büscher et al., 2017), however acidification exposure over 12 months will cause the biomineralized aragonite skeleton to exhibit less organization, a longer and thinner growth form, and reduced structural integrity of exposed skeletal framework, that forms the base of cold-water coral reefs and a significant proportion of the habitat, (*low confidence*) (Hennige et al., 2015). Weakening of skeletons by acidification enhances bioerosion of carbonates (e.g., by bacteria, fungi, annelids and sponges) (Schönberg et al., 2017), and bioerosion may exacerbate dissolution; dissolution and bioerosion will cause loss of cold-water coral habitat (*virtually certain*).

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

Experiments on cold-water coral growth over a range of time scales have now demonstrated positive (but variable) calcification rates under future scenarios of ocean acidification and warming (Maier et al., 2013a; Maier et al., 2013b; Hennige et al., 2015; Rodolfo-Metalpa et al., 2015; Gori et al., 2016; Maier et al., 2016; Büscher et al., 2017) (*medium confidence*). However, net calcification rates (calcification+dissolution) of *L. pertusa* exposed to aragonite-undersaturated conditions or close to saturation ($\Omega < 1$) often decrease to close to zero or even became negative (Lunden et al., 2014; Hennige et al., 2015; Büscher et al., 2017). There is genetic variability in *L. pertusa's* ability to calcify at low aragonite saturation, and some species (e.g., in the Gulf of Mexico) may more readily adapt to ocean acidification (Kurman et al., 2017) (single study *low confidence*). However, hypoxia (1.5 ml L^{-1}) can cause mortality in Gulf of Mexico *L. pertusa* populations, which already live under low-oxygen, high CO_2 conditions (Lunden et al., 2014) (single study*, low confidence*).

In the solitary coral *Desmophyllum dianthus*, a species closely related to *L. pertusa*, exposure to elevated pCO_2 when combined with elevated temperatures reduces calcification and respiration rates and shifts metabolic substrates from a mixture of protein and carbohydrate/lipids to a less efficient protein-dominated catabolism (*limited evidence,* single study). Populations of this species waxed and waned over the last 40,000 years as the global ocean was influenced by glacial advances and retreats and changing concentrations of CO_3^{2-}, O_2, and surface productivity (Thiagarajan et al., 2013). Coral populations on seamounts in the N. Atlantic and SW Pacific Oceans altered their depth distribution in response to changes in the aragonite saturation depth, which has shoaled since the last glacial maximum (Thiagarajan et al., 2013); future increases in atmospheric CO_2 are expected to further decrease suitable habitat for deep-dwelling hard coral species and their associated benthic organisms (Thiagarajan et al., 2013) (*medium confidence*).

Scleractinian corals have the capacity to acclimate to high CO_2 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 750ppm CO_2. 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 in the Gulf of Mexico oxygen minimum zone (3.3 ml/l for North Atlantic specimens), highlighting critical thresholds for coral populations living at the edge of their tolerance (e.g., periphery of hypoxic zones, warm Mediterranean deep-sea). The role of temporal dynamics in the combination of stressors is recognized but is still poorly studied under in situ conditions (one study supports species-specific response to seasonal influence in the growth of *L. pertusa* and *M. occulata*) (Lartaud et al., 2014).

There is a decrease in the northeast Atlantic suitable habitat for key coral species (*Desmophyllum* spp. and *Lophelia pertusa*) predicted by habitat suitability modelling under projected changes in T, O_2, pH and POC flux for 2081-2100, but an increase for 2041-2060. Habitat suitability analyses suggest that food supply (POC flux) is a critical variable for the corals *Desmophyllum* spp., *Lophelia pertusa*, *Madrepora oculata*, *Acanella arbuscula*, and *Paragorgia arborea*, while dissolved oxygen is an important variable in the northwest Atlantic area for *Desmophyllum* spp., *Lophelia pertusa*, *Madrepora oculata* (Morato et al., 2006; Levin, 2018). In the Southern hemisphere, future projections to 2099 of warming and acidification predict extreme loss of habitat for cold water coral (*Solenosmilia variabilis*) on seamounts off Australia and New Zealand under RCP 4.5 and nearly complete loss under RCP8.5 both warming above a tolerance threshold of 7°C and decline of aragonite saturation ($\Omega_{\text{arag}} < 0.84$) below that required for survival are responsible (Thresher et al., 2015).

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

5.2.2.4.10 Biogenic structures: sponges and other taxa.

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

5.2.2.4.11 Gaps

Understanding of climate change impacts on deep-sea ecosystems is challenged by the absence of long-term observations in most systems, and limited numbers of laboratory or *in situ* studies, most with an emphasis on single species and on single climate drivers. Studies on the effects of ocean acidification in the deep ocean are restricted primarily to calcifying organisms (i.e., cold-water corals) while similar work on non-calcifying ecosystem engineers (i.e., sponges, xenophyophores, sea pens) and the ecosystems that they form (e.g., sponge grounds), is almost absent. Multiple driver studies are based largely on climate projections combined with habitat suitability assessment for foundation species and key commercial species, but additional groundtruthing is required, ideally through sustained deep-ocean observing programs that incorporate biology across a broader diversity of ecosystems. Baseline assessments are needed to reveal the environmental status of deep-sea ecosystems, with development of spatial and temporal scales to be assessed and abiotic and biotic indicators to be used for measuring response to changing climate drivers (Danovaro et al., 2017; Sweetman et al., 2017).

Some components of the ecosystem are fundamental to deep-sea ecosystem functioning but are severely understudied in the deep sea including viral and protist communities, and their links to higher forms. Viruses are key drivers of microbial metabolism (Dell'Anno et al., 2015; Danovaro et al., 2016). Labile C released by viral cell lysis and virus decomposition stimulates microbial heterotrophs which are coupled to ammonium-dependent archaeal chemosynthesis and nitrogen regeneration (Danovaro et al., 2016). Viral dynamics are key to understanding microbial mediated CO_2 production and consumption and carbon storage capacity of the deep ocean, and cumulative impacts could cause regime shifts that alter life support services provided largely by microbes (Carbon sequestration and Nitrogen regeneration).

5.2.2.4.12 Overview

Despite its seeming remoteness, the most of the deep-sea floor and its varied ecosystems already have or are projected to experience rising temperatures and declining oxygen, pH and POC flux beyond natural variability within the next half century (*medium confidence*). Vast areas of abyssal seafloor are expected to experience declines in food supply that diminish benthic biomass, change community structure and rates of carbon burial (*medium confidence*). Expansion of oxygen-limited and oxygen-minimum zones on margins will reduce fish and invertebrate biodiversity, simplify food webs, reduce ecosystem resilience and alter elemental cycling (*high confidence*). Even chemosynthetic ecosystems like hydrothermal vents and seeps, once thought to be isolated from surface phenomena, are recognized to be vulnerable to changing ocean conditions through their pelagic larvae (e.g., shrimp and mussels), partial reliance on surface derived particulate organic matter (e.g., mussels), microbial symbionts that require oxygen to fix carbon (e.g., tubeworms, mussels), and reliance on mesoscale circulation features for connectivity (*low confidence*). Biogenic habitats that form biodiversity hotspots (e.g., corals, sponges, xenophyophores), and the topographic features that support them (seamounts, canyons, slopes) will be exposed to a combination of stressors (warming, reduced food supply, increasing acidity and oxygen loss) (*high confidence*). Some of the greatest cumulative negative exposures will be in the Arctic and Atlantic Oceans over the next 100 years (*medium confidence*). Ecological responses are expected to vary regionally, but as critical thresholds of temperature, oxygen and CO_2 are exceeded, changing species depth distributions, dissolution, and bioerosion, exacerbated by lower food supply, will cause loss of biogenic habitat critical to biodiversity maintenance and ecosystem health (*medium confidence*).

5.2.3 Risk for Organisms

Different taxa of marine organisms vary in their vulnerability and risk of impacts to climate change due to their differences in exposure levels to climate hazards (see Box 5.1), differences in inherent biological sensitivities and the species capacity to cope with the exposed hazards through acclimation, evolutionary or transgenerational adaptations (Palumbi et al., 2014; Gattuso et al., 2015; Kavousi et al., 2015; Gunderson et al., 2016; Putnam et al., 2016; Krueger et al., 2017). Findings from AR5 highlight shared biological principles that drive responses of marine microbes, plants, invertebrates and fishes to single and multiple climate change drivers (WGII AR5 6.3, (Pörtner et al., 2014)). Particularly, temperature predictably affect 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 and the organisms cannot survive with temperature beyond their tolerance limits (Pörtner et al., 2014); Pörtner et al. (2017). Field and laboratory experiments and observations further support the assessment in WGII AR5 that temperature-dependent biological sensitivity vary between species, populations and life history stages, and contribute to determining the organism biogeography and phenology (Poloczanska et al., 2016; Pörtner et al., 2017)) (*high confidence*) (Figure 5.13).

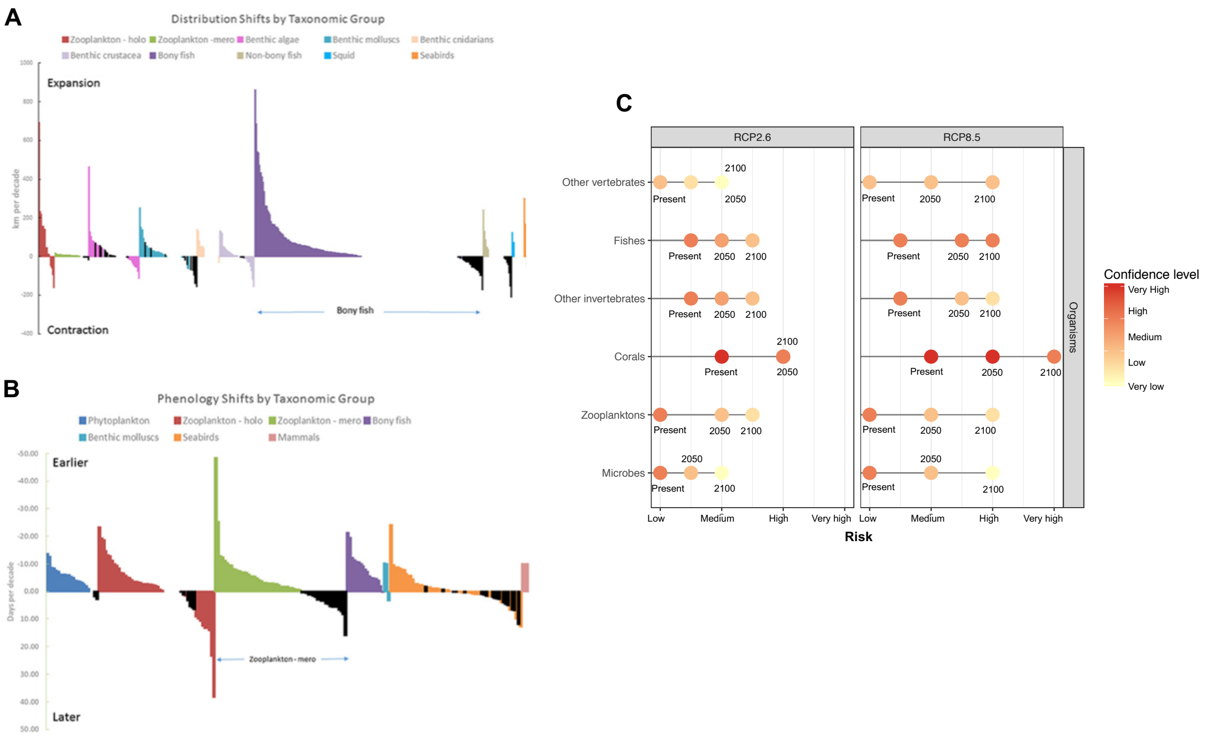


Figure 5.13: Contrasting risks of impact for different groups of marine organisms in the present-day (present), mid-21st century (2050) and end of 21st century (2100) under RCP2.6 and RCP8.5. (A) Observed rate of range shifts and (B) observed rate of shifts in phenology (redrawn from (Baudron et al., 2014)), and (C) assessment of the risk and confidence level (see 5.3.2 for evidence) of different organismal groups.

Historical changes in fish and invertebrates abundance since the 1950s are partly attributed to changes in temperature primary production in relation to changes in recruitment, growth and survivorship Howes et al. (2015); (Britten et al., 2016; Poloczanska et al., 2016). Based on outputs from an ensemble of marine ecosystem and fisheries models, potential total consumer (mainly invertebrate and fish) biomass without fishing is projected to decrease by 4.8% and 17.2% under RCP2.6 and 8.5, respectively, by 2090-2099 relative to 1990-1999, while the decrease is around 4.5% by the mid-21st century across all RCPs (*likely*) (Figure 5.14). Accounting for the removal of biomass by fishing exacerbates the decrease in biomass to -27.5% for all and to -52.4% for large-bodied animals which are particularly sensitive to fishing (*likely* for the direction of changes). Regionally, total consumer biomass decreases largely in tropical and mid-latitude oceans (*likely*) and increases in the Arctic ocean (Figure 5.14, *as likely as not*). The high uncertainty in the Arctic is because of model uncertainties as a result of the ways the model represent the effects of ocean variables (temperature and net primary production) (Lotze et al., in review). In the Southern Ocean, the decrease in consumer biomass is mainly in the southern Indian Ocean while other parts of the Southern Ocean are projected to have an increase in consumer biomass by 2100 under RCP8.5, reflecting mainly the projected pattern of changes in net primary production from the earth system models (see Section 5.2.1.3)

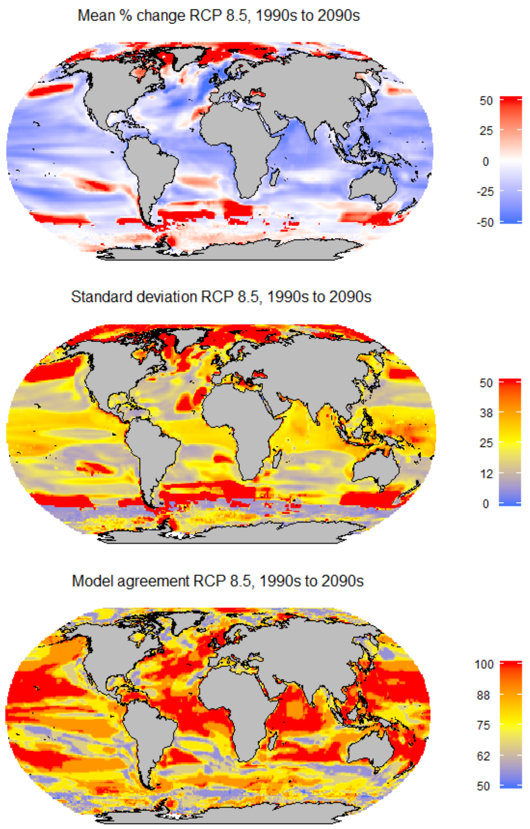


Figure 5.14: Projected changes in biomass of fishes and invertebrates (except zooplanktons) based on outputs from the Fisheries and Marine Ecosystem Impact Models Intercomparison Project (FISH-MIP).

The effects of ocean deoxygenation, acidification and changes in nutrient availability add to, and potentially interact with, temperature effects on marine organisms and ecosystem functions (Sections 5.2.1.3, 5.2.1.4, 5.2.1.5) (Pörtner et al., 2014). Anoxic condition makes it difficult to sustain marine life that obtain oxygen from waters while hypoxic condition reduces their metabolic scope for growth, and consequently limit the ability of animals to forage, avoid predation, or fend off diseases and exacerbate impacts of warming that raises oxygen demand (Breitburg et al., 2018). At the same time, increasing CO_2 and decreasing pH in the ocean disrupts organism's vital biological processes such as growth, reproduction and calcification through increased corrosion of calcium carbonate exoskeleton, elevated metabolic demand from disruption of intra-cellular and inter-cellular acid-base balance, and neuro-sensory and behaviour such as foraging and predatory avoidance, with the effects vary between species and ecosystems (Kroeker et al., 2013; Cattano et al., 2018)(*high confidence*). Warming may exacerbate the effects of ocean acidification of selected biological functions such as increasing rate of photosynthesis in phytoplanktons (Sett et al., 2014), reducing development of larvae of some sea urchins (echinoderms) (Lefevre, 2016), or increasing predation rate of reef fishes (Ferrari Maud et al., 2014) and behaviour in general (Nagelkerken and Munday, 2016), although interactions vary strongly between species and biological processes (Lefevre, 2016; Nagelkerken and Munday, 2016) (*medium confidence*). Hypoxia and acidification can also limit the thermal reaction norm of organisms and exacerbate their sensitivity to warming (Gobler and Baumann, 2016; Pörtner et al., 2017). Limitation of nutrient and food availability and predation pressures can further increase the sensitivity of organismal groups to climate change in specific ecosystems (Mackenzie et al., 2014). Climate change also affect organisms indirectly through the impacts on competitiveness between organisms that favour those that are more adaptive to the changing environmental conditons (Riebesell et al., 2017) and trophic interactions (Alguero-Muniz et al., 2017).

At individual and population levels, marine organisms may respond to warming, ocean acidification, deoxygenation and changing nutrient supplies by physiological acclimatization (Seebacher et al., 2014) transgenerational and evolutionary adaptation (Munday, 2014; Niinemets et al., 2017), behavioural changes (Clements and Hunt, 2015), shifts in phenology and biogeography (Poloczanska et al., 2016). In general, marine ectotherms that obtain oxygen from water are more sensitive to climate change than endothermal organisms. Species that evolved in fluctuating and variable environments (e.g., seasonally variable temperate environment) often become eurythermal (i.e., the ability to tolerate or adjust to a wide range of temperature) while those that evolved in more stable environments become stenothermal (e.g., tropical coral reef) (Cheung and Pauly, 2016; Pörtner and Gutt, 2016). Also, experimental evidence demonstrate that less complex organisms are better in acclimatizing and adapting to warming and deoxygenation because of their wide aerobic scope and short generation time (Pörtner et al., 2017). In contrast, acclimatization to warming and deoxygenation is often limited by one or more physiological processes in higher organisms with complex organization (Seebacher et al., 2014). However, more complex organisms are able to adjust for the changes in their internal acid-base balance under high CO_2 in waters. Although scope for acclimatizing or adapting to climate change impacts exist in marine organisms, climate change impacts will not be fully eliminated given that species have been responding to changing ocean environment through shifting biogeography and phenology in recent decades (Poloczanska et al., 2016).

Section 5.2.3 assesses the key vulnerability and risk of impacts of organismal groups as a result of the direct and indirect effects of ocean warming, deoxygnation, ocean acidification and changes in nutrients supply under climate change. Indirect effects of multiple climate and non-climatic drivers on marine organisms and ecosystems at specific ecosystems are assessed in (5.2.2). This sub-section focuses on impacts that have *moderate* to *high conference* i.e., evidence and agreement on detection, attribution and projection.

5.2.3.1 Microbes

The projected rate of change in the ocean raises questions about how readily individuals can respond to multiple drivers (including oxygen, CO_2, nutrients and temperature) and shifting patterns in the future ocean (Boyd et al., 2016; O'Brien et al., 2016; Poloczanska et al., 2016). Planktonic microbes (bacteria, archaea, unicellular algae, and protozoans), with short turnover times (hours to days), would adapt faster than larger animals, due to their phenotypic plasticity (Schaum et al., 2013) (Box 5.1, *high confidence*). Microbes will quickly adapt to rapidly rising of CO_2 or warming, partly compensating for the negative effects within the limits of their thermodynamic constraints (Schaum and Collins, 2014) (*medium confidence*).

The response of microbes to multiple drivers is however complex since it may involve interactive effects (i.e., synergisms and antagonisms) between changing biologically influential properties (Chust et al., 2014; Maranon et al., 2014; Breitberg et al., 2015; Hutchins and Boyd, 2016) . The majority of studies suggest that reduced pH will result in a shift in the microbial communities associated with invertebrate hosts, with direct impacts on host health and disease susceptibility, although the magnitude of effects will vary among species (O'Brien et al., 2016) (*medium confidence*). An increase in ocean temperature has been shown to cause a shift in the microbial community associated with corals and sponges (Lee et al., 2015; Tracy et al., 2015), as well as in the functional roles these communities perform (like N_2 fixation by coral-associated bacteria; (Raedecker et al., 2014)), and new evidence suggests the effects of ocean pH and temperature are interactive Webster et al. (2016) (*medium confidence*). Under both RCP4.5 and 8.5 there will be significant poleward migrations and phenological shifts in many groups of microorganisms (Pörtner et al., 2014), increasing infectious disease outbreaks with pathogens moving polewards (Baker-Austin et al., 2013; Burge et al., 2014) (*medium confidence*).

A wide range of N_2 fixation responses of diazotrophic cyanobacteria, from no change, increases to 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 (Böttjer et al., 2014; Eichner et al., 2014), possibly due to differences in the mechanisms of N_2 fixation between strains and species (Gradoville et al., 2014) (*medium confidence*). Moreover, nitrogen fixation by *Trichodesmium* appears 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 (Hutchins and Boyd, 2016; Walworth et al., 2016; Walworth et al., 2018) (*medium confidence*).

In situ experiments with natural plankton communities revealed some consistent patterns in phytoplankton responses to ocean acidification (summarized in (Schulz et al., 2017)). Across a wide range of climate zones and ecosystem types, autotrophic standing stocks (chlorophyll a) were consistently higher at high pCO$_2$, more pronounced in smaller-sized taxa, with variable responses depending on nutrient availability. Haptophytes (coccolithophores and *Phaeocystis* sp.) were the only phytoplankton class known to respond negatively to ocean acidification (Bermudez et al. 2016; Riebesell et al. 2017; Hoogstraten et al. 2012). Coccolithophores, a group of phytoplankton that forms calcium carbonate plateles, are vulnerable to projected end of century pH Pörtner et al. (2014) (*medium confidence*). However, new evidence, based on long-term experiments of acclimation or adaptation to increasing temperatures in combination with elevated CO_2, show that growth, calcification and carbon fixation rates of coccolithophores at high CO_2 are modulated by temperature, and could change or even reverse negative calcification (Sett et al., 2014). Calcification of planktonic foraminifera will be negatively affected by acidification (Khanna et al., 2013) (*medium to high confidence*), and their populations are predicted to experience the greatest decrease in diversity and abundance in sub-polar and tropical areas, under RCP8.5 (Roy et al., 2015).

The CO_2 effect on primary producers is channelled differently into the food web depending on the inorganic nutrient availability *(medium confidence).* Under nutrient-limiting conditions, bacteria and micro-heterotrophs consistently benefit. This is probably related to an increased production of polysaccharides by phytoplankton under high CO_2, leading to higher bacterial biomass and in turn to an increase in small-sized, bacterivore ciliates. When inorganic nutrients are replete, phytoplankton produce less exudates, resulting in lower bacterial production and standing stocks and lower abundances of bacterial predators.

Another indirect CO_2 effect on the bacterial and phytoplankton community could be through viral lysis, which is reported to respond positively to pCO_2 (Brussaard et al., 2013; Crawfurd et al., 2017). At the level of mesozooplankton, consistently higher abundances at elevated CO_2 are found for adult copepods under nutrient-limiting conditions and for juvenile copepods under nutrient-replete conditions. With early life stages feeding relatively more on the pico-size range (Brucet et al., 2008), they can be expected to benefit from the higher abundances of picoautotrophs under OA. Transfer of CO_2-stimulated primary production up the foodweb is also indicated by enhanced survival of herring larvae under elevated CO_2 (Sswat et al., 2018) (*low to medium confidence*).

Taken together, these results point towards a comparatively high resilience of pelagic ecosystem to ocean acidification due to high functional redundancy and biological buffering, despite significant CO_2 sensitivities of several key plankton groups (*medium confidence*).

5.2.3.2 Invertebrates

5.2.3.2.1 Zooplankton

There is unequivocal evidence that zooplankton distribution and phenology is responding to warming. The mean rate of poleward shift of zooplankton is 142 ± 28 km per decade, faster than most marine groups (Richardson, 2008; Poloczanska et al., 2013) (Figure 5.13). North Atlantic copepods (Bonnet et al., 2005; Beaugrand, 2009; Helaouet et al., 2011; Chust et al., 2014; Beaugrand et al., 2015; Harris et al., 2015) and cladocerans (Johns et al., 2005; Harris et al., 2015) are moving poleward. Most fish larvae in the California Current are slowly moving poleward, especially those in surface layers with greatest warming (Hsieh et al., 2008; Hsieh et al., 2009). Distribution shifts of copepods are accompanied by a decrease in their mean size, leading to a system with higher metabolism, more recycling and less export (Beaugrand, 2009).

In spring, zooplankton are advancing by 11.6 ± 2.9 days per decade faster than any other marine group (Richardson, 2008; Poloczanska et al., 2013). Copepods, cladocerans, ctenophores, and larvae of decapods, echinoderms, molluscs and fish are all peaking earlier (Mackas et al., 1998; Bertram et al., 2001; Edwards and Richardson, 2004; Greve et al., 2004; Costello et al., 2006; Schlueter et al., 2010), although not all species are advancing (Philippart et al., 2003; Edwards and Richardson, 2004; Burthe et al., 2012), 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 (Asch, 2015).

The warming-stratification hypothesis suggests warming decreases phytoplankton and zooplankton biomass in the tropical ocean because of nutrient limitation associated with enhanced stratification and a shallower mixed layer, and increases biomass in polar regions through greater water column stability and faster growth (Richardson and Schoeman, 2004; Behrenfeld et al., 2006; Boyce et al., 2010). There is some support for this hypothesis. Although there were methodological challenges, a data synthesis from the tropical Atlantic showed a 10-fold drop in biomass from 1950 to 2000, related to reduced phosphate concentration and thinning of the thermocline. Similarly, a recent modelling study showed a 5% decline in zooplankton biomass in the past 50 years (Laufkötter et al., 2013). However, the best available tropical data (from the Bermuda Atlantic Time Series) shows a 61% increase in zooplankton biomass associated with sea surface warming and increased stratification (Steinberg et al., 2012), which is likely to be driven by the increase in smaller phytoplankton translating up the microbial food web to zooplankton.

Evidence is mixed on whether impacts of ocean acidification on calcifying zooplankton are already observable (Havenhand, 2012). Evidence from multi-decadal studies has generally shown no detectable decline in abundance of pteropods and forams, including studies in the northwest Mediterranean (Howes et al., 2015), the California Current (Öhman, 2007), and the northwest Atlantic (Head and Pepin, 2010), although one study found a decline in the pteropod *Limacina* in the northeast Pacific (Mackas and Galbraith, 2012). Two centennial-long datasets from cores have shown a decline in calcification in terms of shell mass of forams in the Arabian Sea (de Moel et al., 2009) and the Southern Ocean (Moy et al., 2009), but a longer study has shown no decline in foram abundance in the California Current (Field et al., 2006). Overall, it is likely that increased ocean acidification has not yet caused sufficient reduction in fitness to decrease abundances of calcifying zooplankton. Similarly, reanalysis of data that suggested jellyfish in the North Sea were increasing in response to ocean acidification (Attrill et al., 2007) showed there was no relationship (Richardson, 2008).

The poleward shift of zooplankton species is likely to continue in the future (Reygondeau and Beaugrand, 2011). Most modelling and experimental work has focused on elucidating future zooplankton biomass and carbon export. 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 (Woodworth-Jefcoats et al., 2017), 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 (Stock et al., 2014). These same models show that zooplankton amplify the climate change signal that propagates up from phytoplankton, by 10-30% (Woodworth-Jefcoats et al., 2017) to 100% (Stock et al., 2014). Recent mesocosm experiments also support the warming-stratification hypothesis. Warming leads to reduced nutrient flux and lower phytoplankton and zooplankton biomass, and increased zooplankton grazing relative to phytoplankton growth (O'Connor et al., 2009; Lewandowska et al., 2014).

Experimental studies on the impact of realistic ocean acidification scenarios for 2100 show zooplankton responses are species-specific and can be both positive and negative. Many experimental studies have shown no ocean acidification impact on the diversity of mesozooplankton (Alguero-Muniz et al., 2017) early life stages of copepods (Weydmann et al., 2012; McConville et al., 2013; Cripps et al., 2014; Li et al., 2015; Bailey et al., 2016) or schyphomedusa ephyrae (Winans and Purcell, 2010; Alguero-Muniz et al., 2016). However, some studies have shown a negative impact on zooplankton, including shell degradation in pteropods (Schulz et al., 2013), lower abundance of an anthomedusa jellyfish (Alguero-Muniz et al., 2017), increased mortality of copepod nauplii (Cripps et al., 2014), and a reduction in demersal zooplankton (Smith et al., 2016a). By contrast, some experimental studies have shown a benefit of ocean acidification on zooplankton, including a positive effect on overall abundance but with differences in direction of changes between species (Alguero-Muniz et al., 2017; Taucher et al., 2017). For example, in a mesocosm experiment with high pCO_2 treatment, *Hybocodon prolifer* (Anthomedusa) reacted negatively by lower abundances, while *Aglantha digitale* (Trachymedusa) became more abundant (Alguero-Muniz et al., 2017).

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

Despite zooplankton having high evolutionary potential and presumably being well suited to adapt to climate change (Peijnenburg and Goetze, 2013), no thermal adaptation in a 50-year time series of two key copepod species in the North Atlantic was found (Hinder et al., 2014). Some work suggests that sensitivity to ocean acidification is dependent upon the degree of natural exposure, with species that vertically migrate and commonly encounter low pH conditions more able to adapt to lower pH conditions (Lewis et al., 2013).

5.2.3.2.2 Warm-water corals

A major difficulty to understand the risks of climate change on marine organisms is the multifarious nature of environmental changes (Gunderson et al., 2016) (*high confidence*). To assess organismal performances and risk, past studies have relied on experiments performed on a single environmental variable. However, increased understanding that single-stressor experiments may not be appropriate in assessing the climate change risks on marine organisms, including corals (Wernberg et al., 2012), recent studies incorporate two or more environmental factors into their experimental designs (Holcomb et al., 2010; Anlauf et al., 2011; Edmunds, 2011; Albright and Mason, 2013; Rivest and Hofmann, 2014; Przeslawski et al., 2015; Ramajo et al., 2016). Such approach, for example, has revealed that additive effect occurs when the combined effect of multiple stressors equals the sum of the effects of each stressor in isolation: increased temperature and reduced pH in *Porites panamensis* colonies in isolation, decreased coral polyp mass by 21% and 24% respectively; combination of increased temperature and reduced pH caused a reduced polyp mass by 45% (Anlauf et al., 2011). Further, studies on *Acropora tenuis* revealed that synergistic effect occurs when the combined effect of multiple stressors is greater than the expected additive effect of the stressors in isolation (Albright and Mason, 2013) (*medium confidence*).

Various global change drivers carry wide range of risks to individual corals (*high confidence*) (Figure 5.13). Many coral colonies lack the adaptive capacity to cope with the unprecedented disturbances they currently face, while in parallel, anthropogenic impacts directly affect vulnerable life history stages of corals, reducing reproductive output, further affecting the supply of larvae to denuded reefs (Mora et al., 2017) (*high confidence*). Physiological and/or evolutionary mechanisms of environmental accommodation are recorded for various coral species (Fabricius et al., 2011; Shamberger et al., 2014). Not only different coral individuals in various coral species/coral populations differ in their risk responses to global impacts (Rodolfo-Metalpa et al., 2011; Edmunds et al., 2013; Munday et al., 2013; Takahashi and Kurihara, 2013; Wittmann and Poertner, 2013; Palumbi et al., 2014; Kavousi et al., 2015; Putnam et al., 2016; Krueger et al., 2017), but the risk to early life history stages is more distinctive than adults (Chua et al., 2013) (*high confidence*). Furthermore, the metabolic phenotypes of planulae among cohorts released from the same coral colony varied among day of release, affecting metabolic responses to warming and acidification (Rivest and Hofmann, 2014). Even the risk for the most known response of coral to climate change (coral bleaching) is highly variable as bleaching is also sparked by pulses of warm-water exposure, and is usually interacting with high levels of irradiation (Hoegh-Guldberg, 1999) (*high confidence*). All these parameters are variable and are affected by other risks and environmental conditions (*high confidence*). For example, increased nutrient supply to corals can offset the risk of the reductions in calcification and growth associated with seawater acidification (Cohen and Holcomb, 2009; Holcomb et al., 2010; Edmunds, 2011; Andersson and Gledhill, 2013; Ramajo et al., 2016) and many other physiological parameters, such as the risk for reduction in feeding rates (Houlbrèque et al., 2015). Indirectly, ocean acidification affects coral settlement by changing available substrate compositions, including crustose coralline algae and/or microbial biofilms and by influencing settlement cues (Albright et al., 2010; Albright and Langdon, 2011) (*medium confidence*).

Related to the above is the Coral Trait Database (Madin et al., 2016) that may serve as a commonly curated database for risk of impacts on coral traits (56 traits for 1547 species) including physiological, morphological, ecological, phylogenetic and biogeographic traits. A meta-analysis on the sensitivity of coral calcification to ocean acidification (Chan and Connolly, 2013) revealed that under business as usual conditions, risk decline in coral calcification by end-of-century will be ~22%, on average, or ~15% if only studies integrating light and dark calcification are considered (*medium confidence*). In the same way, stress emerged from global changes can alter the coral microbiomes, that are central to reef resilience (reviewed in

(Bourne et al., 2016)) (*high confidence*), also increasing the risks for the prevalence of marine infectious diseases Altizer et al. (2013); (Burge et al., 2014; Maynard et al., 2015) (*low confidence*). Impacts may disrupt the holobiont equilibrium, shifting defensive mechanisms and nutrient cycling pathways that contribute to high risks of bleaching and disease susceptibility (McDole et al., 2012; Maynard et al., 2015; Miller and Richardson, 2015; Bourne et al., 2016; Haas et al., 2016; Webster et al., 2016).

5.2.3.3 Vertebrates

5.2.3.3.1 Fishes

Changes in abundance causes geographic range shifts, with warming being related to expansion of leading geographic range edge (*high confidence*) and contraction of trailing edge in the past (*medium confidence*) (Poloczanska et al., 2016). The success of warming-driven range expansion is related to life history and ecological characteristics of fishes such as larger body size, higher dispersal ability, availability of suitable habitats and food contribute to the successfulness of range expansion (Feary et al., 2014). Thus, the rate of historical geographic range shift horizontally and in depth towards cooler water during the 20th century is partly correlated with the rate of ocean warming, the slope of their habitat's temperature gradient and species' biological characteristics (Sunday et al., 2011; Pinsky et al., 2013; Poloczanska et al., 2016) (*high confidence*). Projections of range shifts of fishes under changes in temperature and net primary production agree with expectations from theory and observations (Figure 5.13), 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*) (Jones and Cheung, 2015; 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 are projected to limit range shifts, resulting in larger relative decrease in species richness (*medium confidence*) (Ben Rais Lasram et al., 2010; Burrows et al., 2014; Jones and Cheung, 2015).

Changes in oxygen content and ocean acidification exacerbate temperature effects on fishes and invertebrates. Decrease in oxygen content under warming projected to reduce maximum body size (Cheung et al., 2013; Pauly and Cheung, 2017) and contraction of suitable environmental conditions for fishes (Deutsch et al., 2015) (*medium confidence*). Fishes exposed to ocean acidification level expected under RCP8.5 showed impairments of sensory ability and alteration of behavior including olfaction, hearing, vision, homing and predator avoidance (Heuer and Grosell, 2014; Munday, 2014; Nagelkerken et al., 2015) (*high confidence*). A wide variety of biological processes, ranging from growth, survivorship, reproduction, and calcification are impacted by ocean acidification expected under RCP8.5. with crustaceans and calcifying molluscs being most sensitive to such effects (Kroeker et al., 2013). Early life stages are particularly sensitivity to ocean acidification impacts. Also, ocean acidification and warming have shown to have synergistic effects with warming that exacerbate the risk of population decline. The risks of population decrease of marine fishes under ocean warming and deoxygenation is further exacerbated by overfishing and pollution (Alava et al., 2017) (*high confidence*).

Fishes have some capacity for physiological adjustment and evolutionary adaptation that lower their sensitivity to warming and decrease in oxygen (Munday et al., 2013) (*low confidence*) (Figure 5.13). 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 sufficient to avoid observed impacts and that the risk of impacts of climate change on fishes shall remain high under high emission scenario (*medium confidence*).

5.2.3.3.2 Marine mammals, birds and reptiles

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

Hatfield et al., 2012; Funayama et al., 2013; Reece et al., 2013; Katselidis et al., 2014; Patino-Martinez et al., 2014; Pike et al., 2015; Reynolds et al., 2015).

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

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

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

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

The future for species that have limited ranges, limited habitat availability or small populations is quite precarious (very high confidence). Some species like Galapagos penguins (*Spheniscus mendiculus*), sea lions (*Zalophus wollebaeki*), fur seals (*Arctocephalus galapagoensis*), and flightless cormorants (*Phalacrocorax harrisi*), have very small populations with restricted ranges and do not have alternative habitats to utilize (Vargas et al., 2005; MacLeod and Steinfartz, 2016; Riofrio-Lazo et al., 2017). In general, specialist species are more vulnerable than generalists (see Figure 5.13). This can be in terms of diet or habitat requirements, or in terms of existing life history traits (Gallagher et al., 2015).

5.3 Changing Marine Ecosystem Services and Human Wellbeing

The Fifth Assessment Report concluded that climate change increases the risk of impacts on the goods and services from marine biodiversity and ecosystems. These goods and services include fisheries and other provisional services, climate regulation by marine microbes and coastal protection by vegetation and other biogenic habitats, cultural value such as tourisms, heritage and ceremonial importance (Pörtner et al., 2014). This section further assesses the impacts of climate change on ecosystem goods and services in specific coastal, pelagic and deep-sea floor ecosystems. Under the conceptual framework developed and used by the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Hughes et al., 2017) (Figure 5.15), climate change is considered to be a direct driver that affects ecosystem goods and services (provisioning and others including regulation, supporting and cultural, see Section 5.3.1). This section assesses how climate change impacts on marine biodiversity and ecosystems (Section 5.2) affect changes in ecosystem services ad their consequences on human wellbeing (social, cultural, economic and environmental, see Section 5.3.2). Climate change can also impact ecosystem services through the amplification of the effects of non-climatic drivers on marine ecosystems and biodiversity, such as increasing the variation and magnitude of runoff and its associated impacts of erosion and coastal sedimentation (Singh et al., 2017). Furthermore, the section assesses the implications of changing ecosystem goods and services for human wellbeing and their challenges to the United Nations' Sustainable Development Goals (Section 5.3.3).

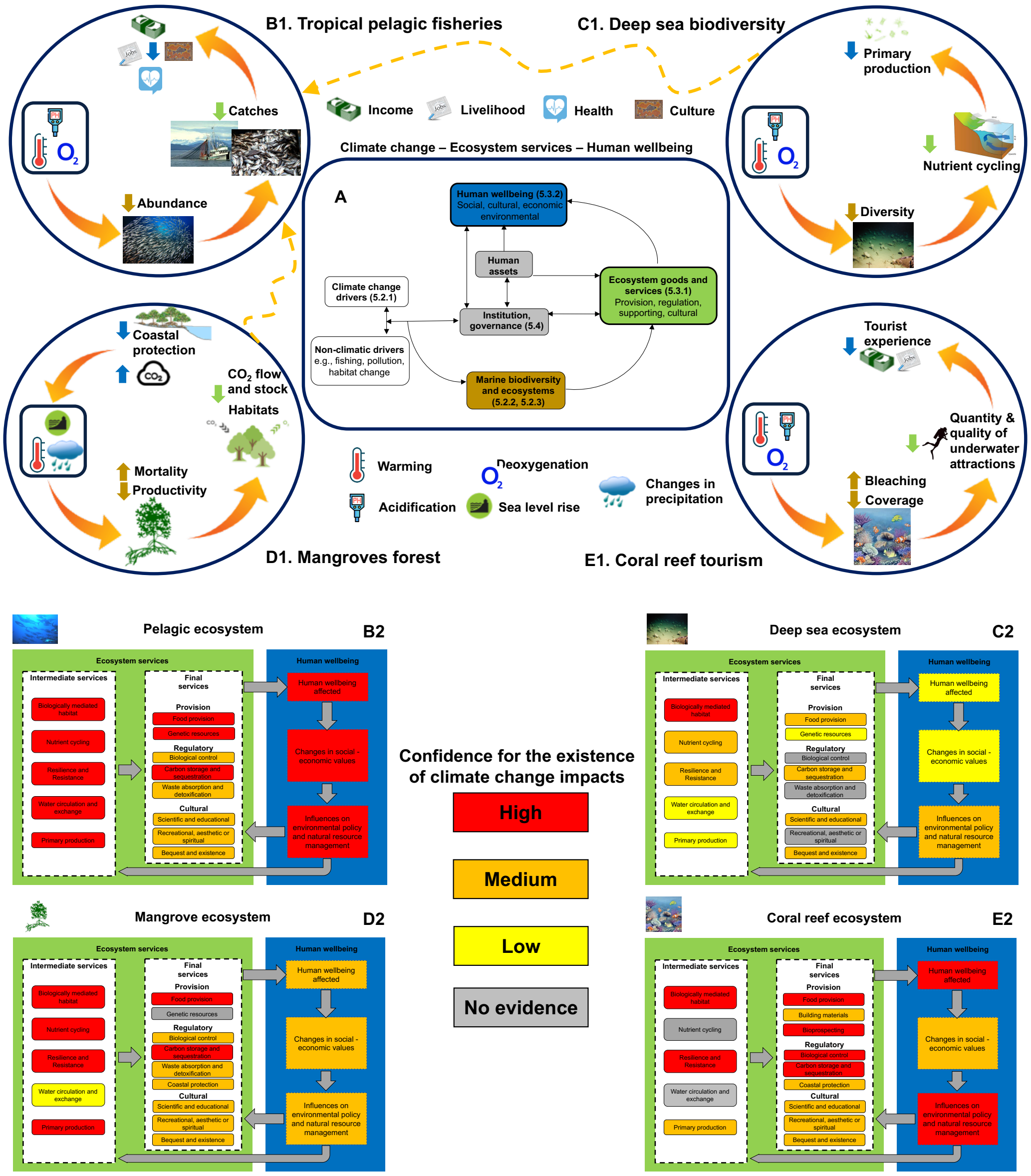


Figure 5.15: Conceptual framework of the linkages between climate change, ecosystems services and human wellbeing and the corresponding sections in this chapter (adapted from (Hughes et al., 2017)) (A), with examples from different types of ecosystems and ecosystem services: (B1) tropical fisheries, (C1) deep sea biodiversity, (D1) mangrove forest, and (E1) coral reef tourism. The boxes highlighted in green and blue colour represent the components (ecosystem services and human wellbeing, respectively) that are assessed in 5.3.1 and 5.3.2. The ecosystems and their services are interconnected (highlighted by the orange dash lines). The risk of climate change on (B2) pelagic ecosystem, (C2) deep sea floor ecosystem, (D2) mangrove and (E2) coral reef ecosystems are highlighted (see 5.3.1, 5.3.2).

5.3.1 Changes in Key Ecosystem Services

Ecosystem services are the environmental processes that render benefits to people Tallis et al. (2010) (recently also called nature' contribution to people) (Díaz et al., 2018) (Figure 5.15). Ecosystem services are broadly divided into provisional services, regulating services, cultural services and supporting services although they are interconnected with one another (Figure 5.15) (Leadley et al., 2014a). Provisioning services include material or energy outputs from ecosystems. 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 while the potential reduction in genetic diversity will lead to loss of potential genetic resources. Regulating services are those processes of the ecosystem to ensure that 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 reduction in climate regulation on increased greenhouse gas emission further exacerbate climate impacts on other regulating or types of ecosystem services. For example, projected increases in the intensity of storms in the 21st century under high CO_2 emission scenario can overwhelm local wetlands from regulating and treating waste-water (Erwin, 2009). Climate change also impacts ecosystem functions that support or maintain other ecosystem services, known as *supporting ecosystem services* (Costanza et al., 2017). Climate change impacts biogenic habitats such as coral reefs (Section 5.2.2.3) and mangrove forest (Section 5.2.2.6), consequently reducing their suitability for organisms that are associated with those habitats (Pörtner et al., 2014; Wong et al., 2014a) (Sections 5.2.2, 5.2.3). The final major category of ecosystem services – *cultural ecosystem services* – that include recreation, tourism, aesthetic and spiritual experiences. These services are a product of both human experiencing nature and the availability of nature to provide the experiences (Chan et al., 2012), the quality and quantity of the latter are impacted by climate change. Also, in some cases, climate change can change where these ecosystem services are enjoyed although such adjustment may not be possible for place-based services, particularly these service that are rely on indigenous and local knowledge. Following the overall structure of this assessment, ecosystem services from three ocean biomes (coastal, pelagic and deep sea floor) are assessed.

5.3.1.1 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 (Figure 5.16) and contribute large benefits to human society. For example, a study estimated based on total economic valuation that the values of coastal ecosystems ae \$352,249/ha for coral reefs, and \$193,843/ha for tidal marshes and mangroves (Pendleton et al., 2016), although the values are not static and have wide error margin (Hilmi et al., 2013b). Coastal ecosystems such as coral reef are also providing ecosystem services to half a billion of people, worldwide ((Hughes et al., 2012). However, the deterioration of coastal ecosystems due to human activities is intense and increasing; 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and 29% of seagrasses worldwide have been lost or degraded over the last 50–100 years Valiela et al. (2001); (Duarte et al., 2005; Spalding, 2010; Giri et al., 2011). The main activities that contribute to such deterioration are coastal development, over-exploitation, pollution and climate change (Section 5.2.2.2). Over the 20th century, losses of coastal ecosystems such as mangroves, seagrass and saltmarshes were estimated to be 25–50% of their area (McLeod et al., 2011), with a recent loss rate of approximately 0.5–3.0% per year (Pendleton et al., 2012b). At such loss rates, 30–40% of tidal marshes and seagrasses (IPCC, 2007) and nearly 100% of mangroves (Duke et al., 2007) could be lost in the next 100 years. The complex interaction between these human stressors on coastal ecosystems and the lack of long-term dataset render quantitative attribution of relative contribution of the observed habitat loss due to climate change difficult.

Global loss and degradation of coastal ecosystems is known to affect at least three critical ecosystem services (Worm et al., 2006; Inniss et al., 2017). A meta-analysis of global datasets suggest that the loss of coastal biodiversity impacts the number of viable fisheries (33% decline); the provision of nursery functions (69% decline); and filtering services provided by coastal wetlands (63% decline) (Worm et al., 2006). The loss of biodiversity, ecosystem functions, and coastal vegetation in coastal ecosystems have also contributed to impacts on their recreational and cultural values, biological invasions, carbon capture and sequestration, declining water quality, and decreased shoreline protection (Braatz et al., 2007; Cochard et al., 2008; Koch et al., 2009). More recent study provides further evidence to the significant relationship between coastal biodiversity and ecosystem functions that support these services (Gamfeldt et al., 2015). Thus, human activities and their consequences, including climate change, are substantially degrading all coastal ecosystem services that are important to human wellbeing (*very high confidence*). This sub-section focuses on assessing observed and projected impacts of climate change on coastal ecosystem functions and services.

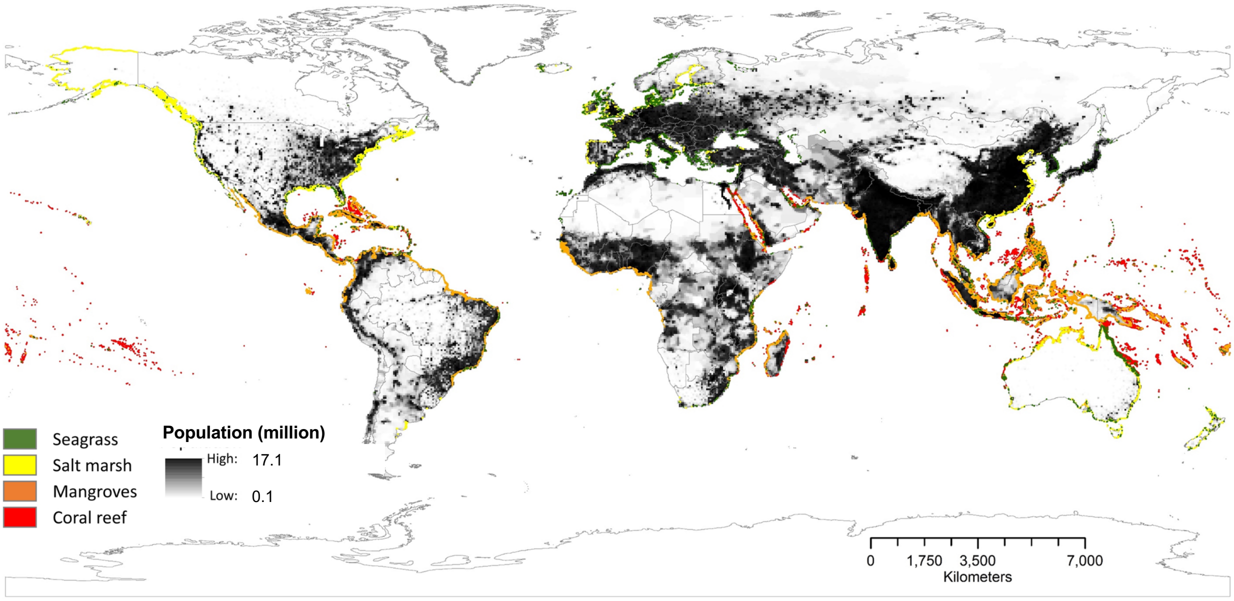


Figure 5.16: Global distribution of seagrasses (UNEP-WCMC and FT, 2017) saltmarshes (Mcowen et al., 2017), mangroves (Spalding, 2010), coral reef (UNEP-WCMC and WRI, 2010) and human population count in 2015 (University, 2017).

5.3.1.1.1 Provisioning services

Coastal and shelf seas ecosystems provide the largest contribution to global seafood production from capture fisheries and aquaculture (FAO, 2016). These ecosystems provide nurseries, breeding, spawning and hatching sites and through transportation of organic matter to the marine environment, thus providing nutrients for marine and estuarine fauna (Lee et al., 2015). Based on global fisheries catch data from the Sea Around Us project (Pauly and Zeller, 2016), in the 2000s, 67 - 74 million MT of fishes and invertebrates are caught in waters with depth less than 200 m. This amounts to >60% of total fisheries catches while marine aquaculture is almost exclusively produced from coastal and shelf waters. Most of these catches are supported by coastal habitat such as coral reefs (Pratchett et al., 2014), seagrass bed (de la Torre-Castro et al., 2014) and mangroves (Carrasquilla-Henao and Juanes, 2017). For example, coral reef fishes are importance nutrient source to many coastal communities worldwide (Cinner, 2014; Micheli et al., 2014; Hoegh-Guldberg et al., 2017).

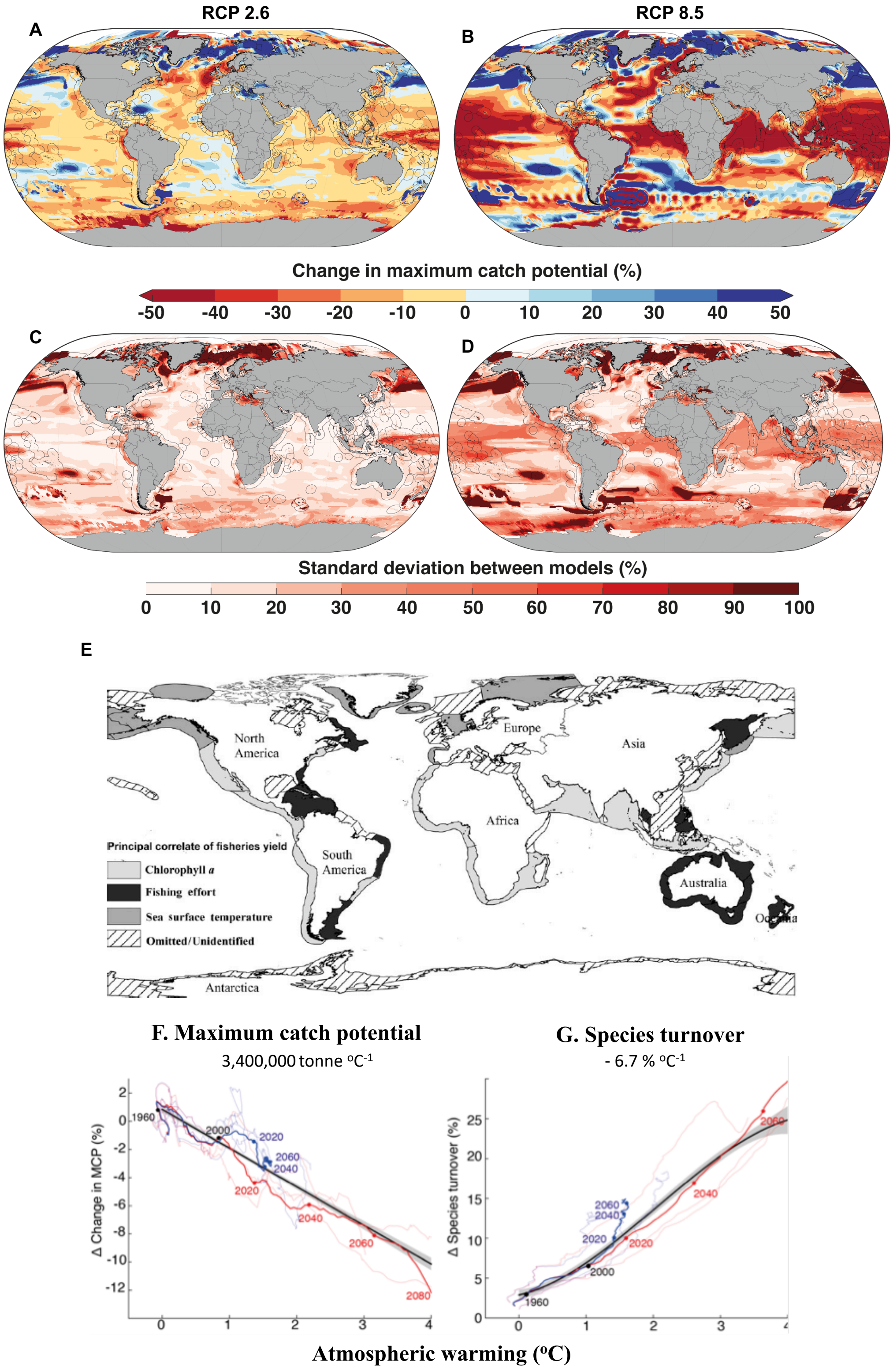


Figure 5.17: Projections of changes fisheries production by shelf seas and open ocean in the 21st century under climate change (Cheung and Pauly, 2016; Cheung et al., 2016; Stock et al., 2017). Projected mean (A, B) and standard deviation (C, D) of changes in maximum catch potential 2081–2100 relative to 1987-2006 under RCP2.6 (A, C) and RCP8.5 (B, D) based on outputs from four models participated in the Fisheries and Marine Ecosystem Models Impact Assessment Project (FISH-MIP) using outputs from two CMIP5 earth system models (GFDL ESM 2G and IPSL CM2-LR). (E) Ocean regions where historical fisheries catches are significantly related to environmental variables: sea surface temperature and chlorophyll *a* concentration; (E, F) scaling between projected global atmospheric warming (relative to 1950-1961) and changes in maximum catch potential (E) and species turnover (F) of >800 species of exploited fishes and invertebrates in the global ocean. Warming and changes in ocean conditions are projected by three CMIP5 Earth System Models while maximum catch potential and species turnover are projected by the Dynamic Bioclimate Envelope model (Cheung et al., 2016).

The impacts of ocean warming and changes in net primary production on invertebrates and fishes are affecting biomass production level and composition of available species in coastal and shelf seas ecosystems with regional differences in the magnitude and direction of impacts (*likely,* see Sections 5.2.2, 5.2.3 Figure 5.17). Temperature and net primary production have shown to explain regional (Halpern et al., 2008; McOwen et al., 2015) and temporal (Britten et al., 2016) differences in fisheries catches in contemporary shelf seas. Trends of recent (during the 2000s period) reported fisheries catches in majority of the coastal and shelf seas are significantly related to temperature (indicated by SST) and primary production (indicated by Chlorophyll *a* concentration) (Figure 5.17). Recent catch trends in some areas such as shelf seas around Australia and New Zealand are mainly related to changes in fishing effort (Figure 5.17). Analysis of historical catch records since AR5 continue to support a change in species composition of fisheries catches that is related to warming, with increased dominance of warm-water associated species in the coastal and shelf seas regions (Keskin and Pauly, 2014; Tsikliras et al., 2014; Maharaj et al., 2018) (*high confidence*).

Loss of critical coastal habitats is a major driver of loss of coastal fisheries productivity. Loss of coral reef habitats driven by climate change (see Section 5.2.2) and the subsequent declines in the physical reef structures have marked effects on the abundance of coral reef fishes and invertebrates, thus impacting not ecosystem functions such as resilience and stability, as well as fisheries productivity (Prather et al., 2013; Pratchett et al., 2014) (*high confidence*). For mangrove forest, spatial comparison between Indo-West Pacific and Atlantic eastern Pacific suggest that higher floral and faunal diversity in the earlier is correlated with a greater range of species exploited for fuel, timber and crustaceans (Huxham et al., 2017). Global meta-analysis of mangrove-fishery linkages show a significant relationship between mangrove area and fishery catches (Carrasquilla-Henao and Juanes, 2017). Thus, if climate change reduces area and/or of mangrove forest (see Section 5.2.2), potential fisheries catches will also decrease (*medium confidence*).

Ensemble of simulations from multiple models driven by projected changes in temperature, net primary production (and in some models, oxygen, salinity and sea ice extent) from CMIP5 Earth System Models project a decrease in maximum catch potential by 2.8–8.7% and 2.8–9.1% under RCP2.6 by 2050 and 2095 relative to 2000 respectively (FAO, 2018). In contrast, the decrease was larger, 7.0–11.8% and 16.2–25.5% by 2050 and 2095 relative to 2000, respectively, under RCP8.5. These impacts will result in a loss of potential catches of 3.4 million tonnes and changes of 6.4% of the exploited species per degree Celsius atmospheric warming relative to 1951–1960 level (Cheung et al., 2016) (*medium confidence*) (Figure 5.17). The interactions between temperature, net primary production and efficiency of transfer of energy across the foodweb are projected to amplify these trends, producing changes that may exceed 50% in some regions by the end of the 21st century under high-emissions scenarios (Stock et al., 2017). Climate change impacts on biological productivity of fish stocks have shown to interact with fishing (Pershing et al., 2015; Britten et al., 2016) resulting in decline in fish stocks that further reduces fisheries productivity (*medium confidence*).

Regionally, tropical and shallower waters are projected to have the largest decrease in maximum catch potential by 2100 relative to the 2000s levels, while catch potential in deeper water of the shelf seas is projected to increase because of shift in distribution towards deeper, cooler waters (Section 5.2.3) (Figure 5.17). As a result of changes in species distribution range, coastal fisheries resources are also projected to shift geographically towards higher latitude or deeper waters (Figure 5.17). For example, around the Pacific Islands, with projected changes in SST, decreases in surface dissolved oxygen and NPP by ≥3°C, ≥0.01 ml L^{-1}, 0.5 g m^{-2} d^{-1} by 2100 under RCP8.5, more than 50% of exploited fishes and invertebrates are projected to become locally-extinct in many regions; these factors cause 74% of the area to project a loss in catch potential by >50%. The area of large projected catch loss is projected to be halved under RCP2.6 (Asch et al., 2017).

Marine aquaculture (i..e, mariculture; marine and brackish) exerts a profound influence as part of the global food systems (*high confidence*). Adding about 27.6 million tonnes to total aquaculture production in 2016 (FAO, 2016), with potentially large contribution of this sector to food and nutrition security (Béné et al., 2015). Climate change and ocean acidification will impact the sustainability of mariculture production (*high confidence*). Most mariculture activities take place mainly in sheltered areas of the coastal zone and includes habitats such as estuaries, salt marshes, mud flats and more recently the open ocean. A study on growth of farmed Atlantic salmon, Cobia and seabream evaluated the effect of future climate trends on optimal growth and concluded that climate change will alter the species growth potential in many ocean areas (Klinger et al., 2017), thereby affecting the general productivity of the sector. Disease is another major treats that may be exacerbated by extreme weather events and other ocean changes associated with climate change (*low confidence*). (Karvonen et al., 2010) shows that increasing water temperature is associated with higher prevalence of the parasite *Ichthyophthirius multifiliis* and bacteria *Flavobacterium columnare* that can cause disease and mortality of farmed fishes such as salmon. Changes in global fisheries abundance and catch of forage fish such as anchovy and sardine under climate change (Merino et al., 2012); Barange et al. (2014) (Sections 5.2.2, 5.2.3) will further impact mariculture productivity as fishmeal and fish oil production is reduced with consequences for the supply of aqua feeds (*medium confidence*).

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

5.3.1.1.2 Regulation, supportive and cultural services

Climate change also affects coastal ecosystem services in additional to fisheries that connect directly to human well-being, health, livelihoods and survival. 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 6% diminution in coral cover projected under a scenario of 10% increase in carbon emission (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). It should also be taken into consideration that in many instances the major price tags for environmental services are weighed in a simplified way, thus not reflecting their real values (Rinkevich, 2015b) and that most of the services provided by any specific ecosystem are usually not deliberated when employing conventional macro-economic indicators (such as the Gross Domestic Product) (TEEB, 2010), since many of the goods and services are not traded in markets.

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*). 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, highlighted temperature, ocean acidification, sea level rise and ENSO variability) are also constructing marine regime shifts (Chen et al., 2011; Leadley et al., 2014b; Rocha et al., 2015), with different biodiversity outlines that downstream reduction in coral reef services (*high confidence*). Thus, global change impacts that will inevitably alter reefs biodiversity, resulting in new reef assemblages that consequently degrade critical ecosystem functions and services. It is predicted that these novel reefs assemblages 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*).

There are various ecosystem services provided by reefs, with different sensitivities to a range of future climate change scenarios (Rogers, 2015) as to the combinations of local stressors and climate change (Elliff and Silva, 2017). The above reflects different views and approaches for ecosystem service priorities (Hicks et al., 2013). However, it is still unclear 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). Another important class of reef services is coral reef tourism marked with a global value of US$ 36 billion per year (Spalding et al., 2017). This is a reef service highly impacted by global changes (*high confidence*). A direct relationship between coral cover and recreational use (a subsector in the tourism industry) can be drawn (while varied between sites) revealing that all types of tourist activities are highly vulnerable to climate change impacts on the coral reefs (Biggs et al., 2015; Lane et al., 2015). All above ecosystem services in the coral reefs can be significantly impacted from the increased frequency and intensity of extreme climatic events that have emerged from global warming. These impacts are not easy to model, since current models are largely based on gradual warming trends (Wernberg et al., 2012).

Carbon sequestration by coastal ecosystems is a key regulatory service that is vulnerable to climate change. Carbon is biologically sequestered in coastal sediments, commonly known as 'blue carbon' (see Section 5.4.1). It is the carbon captured by marine plants and stored, primarily in the soil and sediment of coastal ecosystems such as mangroves, salt marshes, seagrasses and potentially algae. Carbon storage capacity in the top meter of soil is estimated at 280 MgC ha^{-1} for mangroves, 250 MgC ha^{-1} for salt marshes, and 140 MgC ha^{-1} for seagrass meadows, equivalent to 1,030 MgCO$_2$eq ha^{-1} for mangroves, 920 MgCO$_2$-eq ha^{-1} for salt marshes, and 520 MgCO$_2$-eq ha^{-1} for seagrass bed. In addition to carbon captured by plants, the mean carbon storage is 1,494, 951 and 607 MgCO$_2$-eq ha^{-1} for mangroves, salt marshes and seagrass beds, respectively (Figure 5.20). Carbon burial rates are also affected by variability in hydrological regimes, salinity, nutrient status, and management status. Long-term rates of carbon accumulation in sediments of salt marshes, mangroves, and seagrasses range from 18–1713 gC m^{-2} yr^{-1}(Figure 5.18). Overall, (McLeod et al., 2011) estimated blue carbon uptake at about 0.2 Giga (billion) tonnes of carbon per year (GtC yr^{-1}). By contrast, at 3.1 GtC yr^{-1} during 2006–2015 Le Quere et al. (2016) the terrestrial biosphere stores 10 times more carbon than blue carbon sinks. Even though the uptake rate per unit of surface area is much higher in coastal vegetated areas than in forests, it is the total area and thus amount that is significant in the global carbon budget.

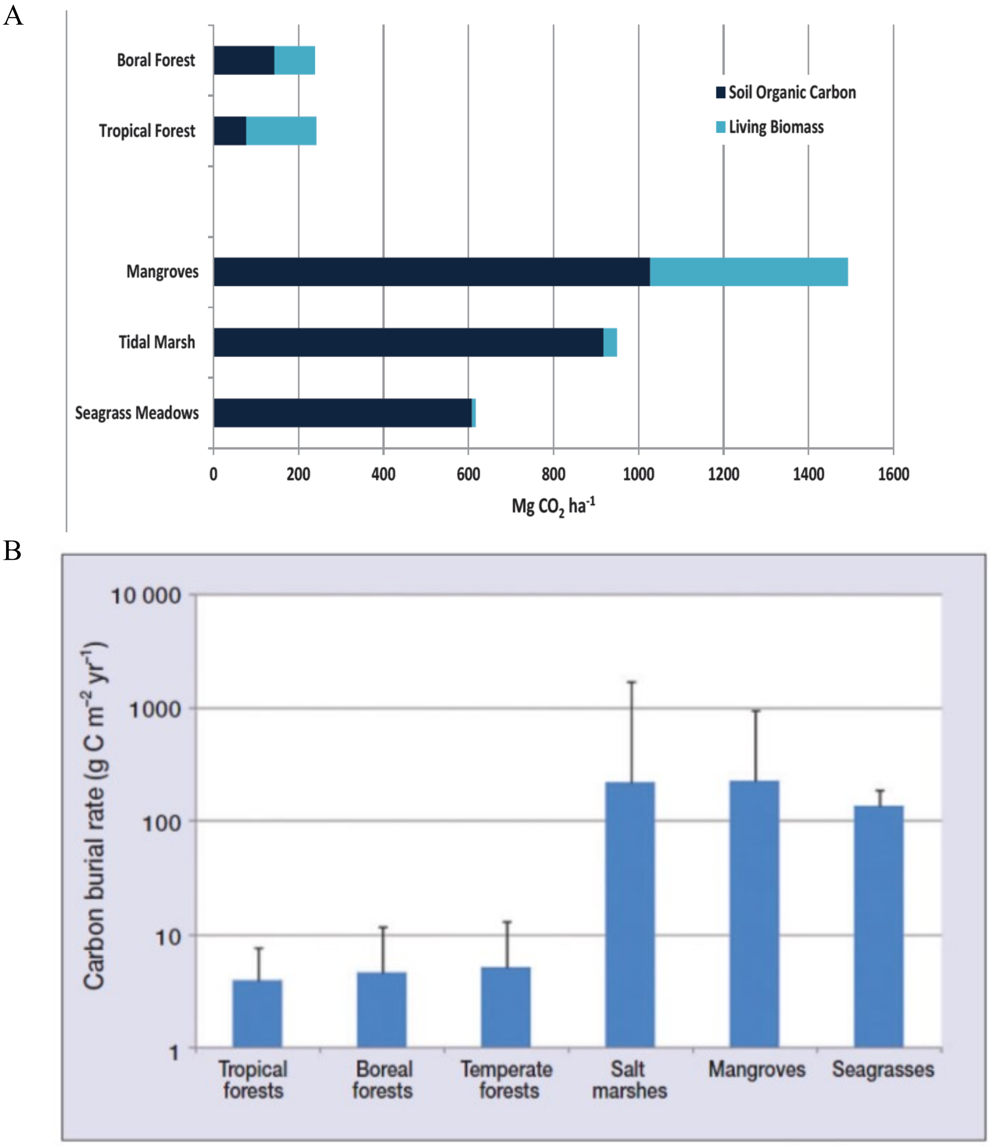
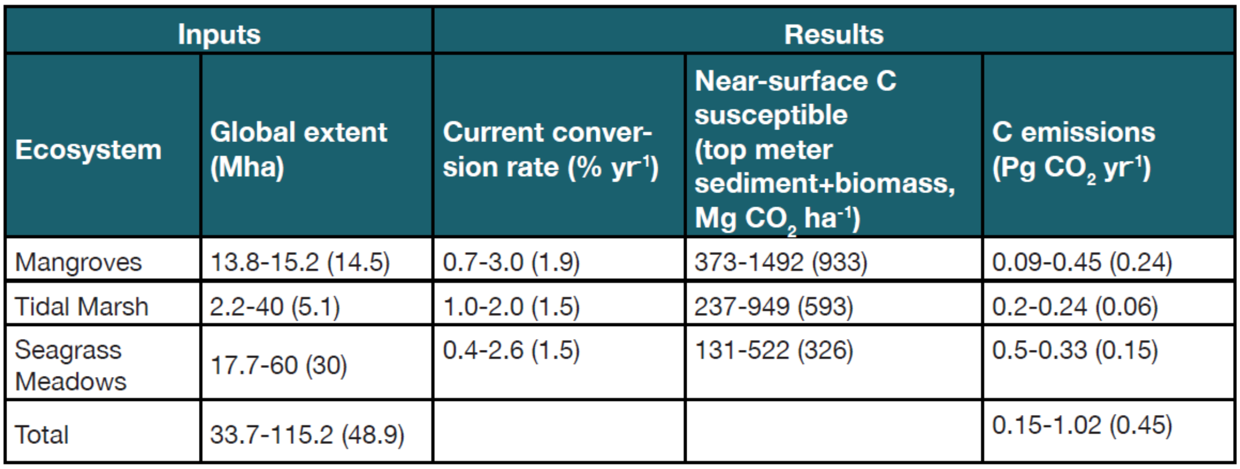


Figure 5.18: Mean carbon storage (A. above and belowground) (Pan et al., 2011; Fourqurean et al., 2012; Pendleton et al., 2012a) and mean long-term carbon burial rate (B) in coastal ecosystems versus terrestrial forests ((McLeod et al., 2011).

Given the substantial contribution of coastal blue carbon to carbon budget, loss of these coastal ecosystems because of climate change (see Section 5.2.2) will also directly impact such service. When coastal ecosystems are lost or degraded the huge stocks of blue carbon in the soils are exposed and released as CO_2 into the atmosphere contributing to global warming. Current rates of loss of blue carbon ecosystems may result in 0.15–1.02 billion tons of CO_2 released annually. These emissions are equivalent to 3–19% of those from deforestation globally, and result in economic damages of USD6–42 billion annually (Table 5.4). For example, current releases from mangrove deforestation are estimated at 0.02–0.12 GtC yr^{-1}, accounting for up to 10% of global deforestation emissions Donato et al. (2011). Tidal wetlands play disproportionately important roles in coastal carbon budgets, forming critical linkages between rivers, estuaries, and oceans (Najjar et al., 2018). As a result of tidal flooding, saltmarsh soil does not dry out and high levels of carbon can accumulate under anaerobic conditions (Figure 5.19). This is coupled with generally low rates of methane emission, which is strongly limited in saline marshes (*medium confidence*) (Poffenbarger et al., 2011; Abella Perez et al., 2016). The carbon burial rate of salt marshes is comparable to mangroves due to high belowground productivity (218 ± 24 gC m^{-2} yr^{-1}) (McLeod et al., 2011). Despite their importance for coastal protection and biodiversity, the global area of saltmarshes has only recently been properly quantified, at around 55,000 km^2, in 99 countries (Mcowen et al., 2017). Climate change and the temperature-driven displacement of saltmarsh plants by mangrove trees in the mangrove-saltmarsh ecosystem are predicted to increase carbon sequestration in coastal wetlands (Megonigal et al., 2016) (*medium confidence*).

Table 5.4: 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 km^2, 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 CO_2 ha^{-1})	C emissions (Pg CO_2 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 Meadows	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)

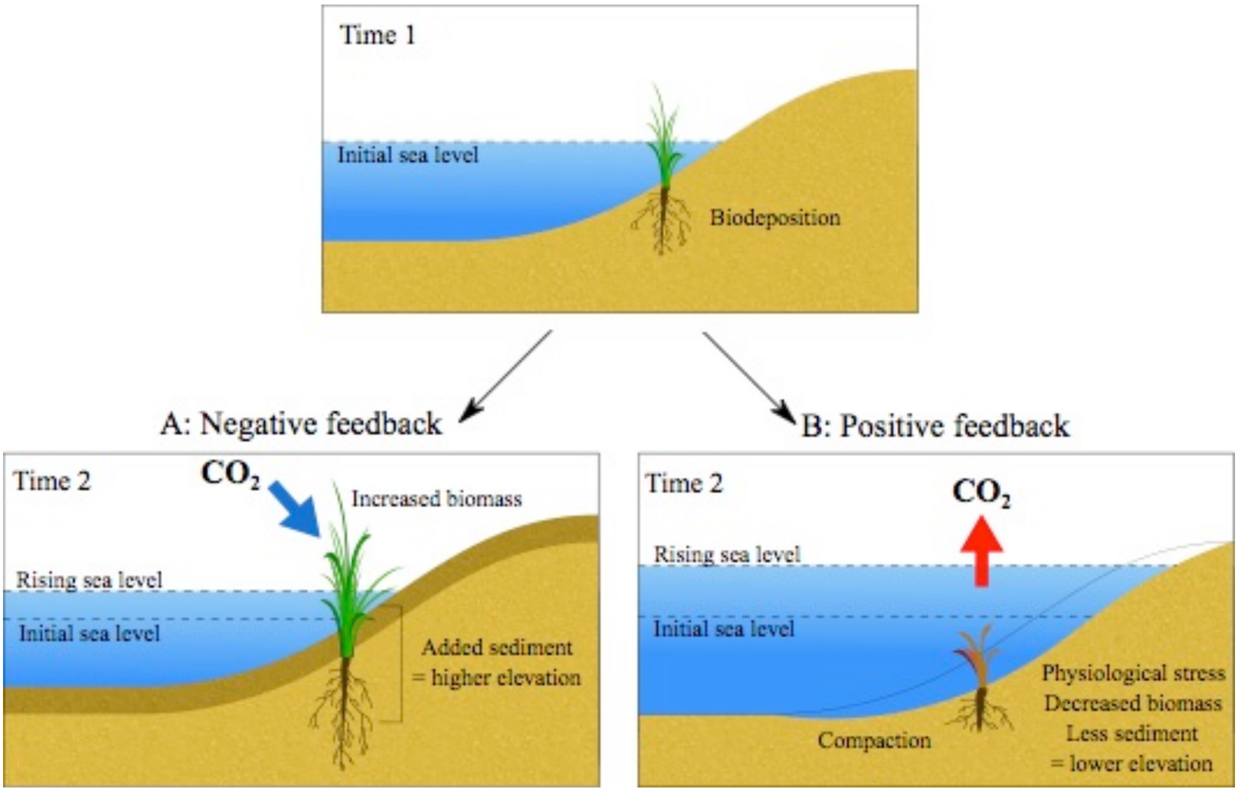


Figure 5.19: Biogeomorphic feedbacks involving plant biomass, sediment accretion and inundation that control the response of vegetated coastal ecosystems to rising sea levels. (A) Under high rate of soil formation, plants are able to offset sea level rise and may produce a negative feedback by increasing the uptake of atmospheric CO_2. 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 CO_2 outgassing. This figure does not consider landward movements, controlled by topography and human land-use.

Losses and degradation of coastal ecosystems continue to impact other regulatory, supportive and cultural services including shoreline stabilization, improved water quality, and key habitats for marine species (Assessment, 2005), impacting a wide range of ecosystem services. Recent expansion of mangroves into saltmarshes is likely to have large impacts on the structure, function and service provisioning of coastal wetlands (Kelleway et al., 2017b). Although mangrove encroachment may increase nutrient storage and improve storm protection (Sheng and Zou, 2017), the declines in saltmarshes will reduce habitat availability for fauna requiring open vegetation structure, and consequently the recreational and cultural activities associated with this fauna (Kelleway et al., 2017b). The impact on provisional services such as fisheries productivity and cultural services is projected to be site-specific and dependent on the species involved (Kelleway et al., 2017b). Overall, climate change-induced reduction in the area and productivity of coastal ecosystems are expected to degrade ecosystem regulatory, supportive and cultural services (*medium confidence*). However, such assessment is based largely on influences from principles of flows of coastal ecosystem services. Having in mind the above literature, we still possess a limited understanding on the contribution of coastal ecosystem services to the multidimensional human well-being, and there is even little understanding of how human well-being is affected by changes in ecosystem services as a result of global changes (Micheli et al., 2014; Daw et al., 2016; Hoegh-Guldberg et al., 2017).

5.3.1.2 Pelagic Ecosystems

5.3.1.2.1 Provisioning service

Fisheries from the open ocean pelagic ecosystems remain important sources of food, nutrition, income and livelihoods for many millions of people around the world (FAO, 2016). Globally, total fisheries catches of pelagic species amount to 44–53 million MT annually in the 2000s (based on the data from (Pauly and Zeller, 2016)). Based on fisheries catch records and model projections, ocean warming, decrease in primary production and expansion of oxygen minimum zones in some ocean regions are shifting patterns of fisheries productivity in these pelagic ecosystems towards higher latitude as well as longitudinally following environmental gradients (*high confidence*) while the distribution of commercially important fish stocks and their fisheries are shifting their distribution towards higher latitude (*high confidence*). Distribution of fisheries on large pelagic species such as tunas correlate strongly with historical climate variability (*high confidence*). For example, in intermediate latitude across the Atlantic, Indian and Pacific oceans, catches of tropical tunas, including skipjack and yellowfin tuna, is significantly positively related to increases in sea surface temperature, although the overall catches across latitude zones did not show a significant change Monllor-Hurtado et al. (2017). Such poleward shifts in pelagic fisheries agrees 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). Globally, potential fisheries catch in the open pelagic oceans are projected to decrease by 2100 relative to 2000s under RCP8.5 and RCP2.6 scenarios with regional differences (Figure 5.17). Based on ensemble of three fisheries models from the Fisheries and Marine Ecosystem Models Intercompariosn Project (FISH-MIP) driven by outputs from two CMIP5 earth system models suggest high, large (more than -30% by 2100 relative to present day) decrease in catch potential is projected in the tropical open oceans, particularly in the western central Pacific Ocean, eastern central Atlantic Ocean and the western Indian Ocean under RCP8.5 (*likely*, Figure 5.17). In contrast, catch potential in the Arctic is projected to increase although there is high intermodal variability in the projection. In addition to poleward shifts in fisheries catches, selected open ocean pelagic resources are also projected to show a longitudinal shift, for example, eastward shifts of skipjack tuna under ocean warming (Lehodey et al., 2011; Bell et al., 2013). However, the actual catches and profitability of the fisheries interact strongly with the level of exploitation of the resources (Cheung and Pauly, 2016; Dueri et al., 2016). Currently, knowledge is limited regarding how climate change would affect the large but unexploited biomass resources in the mesopelagic ecosystems (St. John et al., 2016).

[START OF BOX 5.3 HERE]

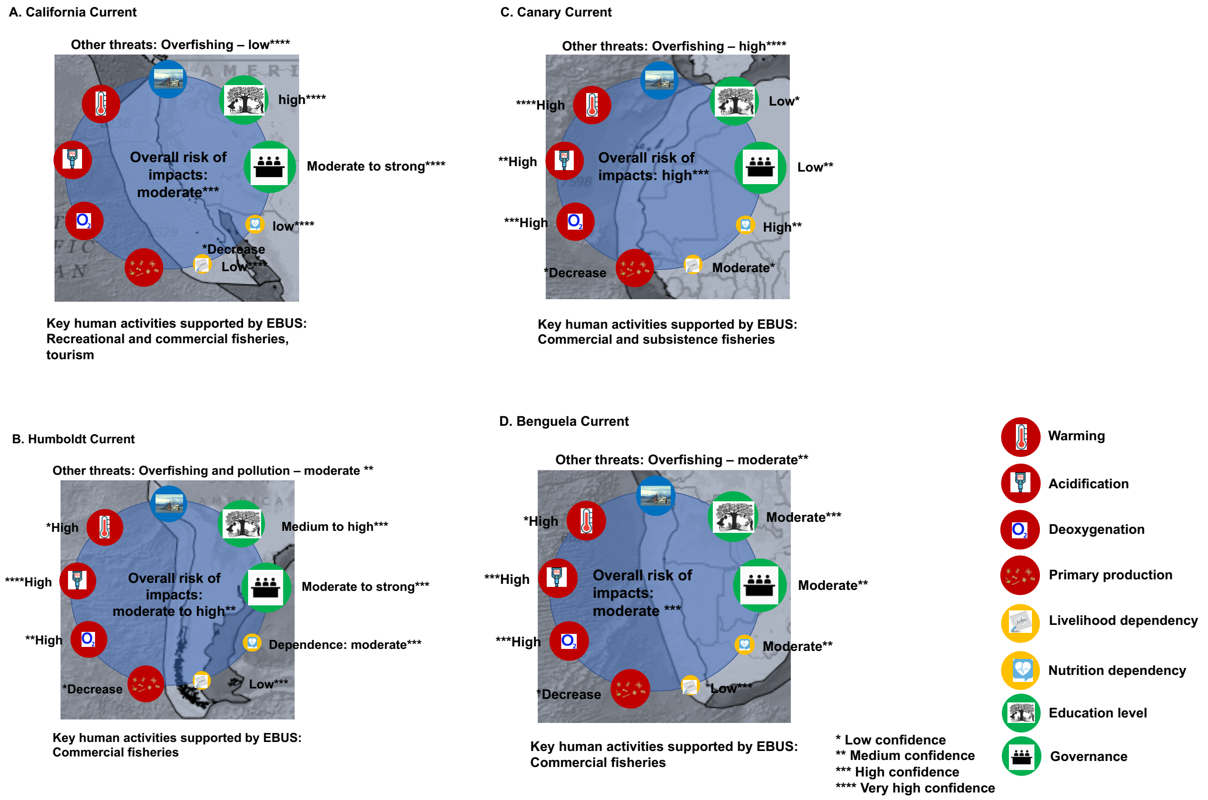
Box 5.3: Responses of Coupled Human-natural Eastern Boundary Upwelling Systems to Climate Change

Coupled with the vast coastal human populations, Eastern Boundary Upwelling Systems (EBUS) host among the most productive ocean ecosystems in the world (Kämpf and Chapman, 2016). These regions are important to livelihood of coastal and other dependent communities, being at the same time threatening by changes in climate and increasing human pressure (García-Reyes et al.; Levin and Le Bris, 2015). The main services the human benefits from EBUS include fisheries and aquaculture (provisioning), generation of moisture and precipitation (regulation), whale-watching (cultural), and providing nutrients to support coastal foodweb (supporting) (Bakun et al., 2015; Gattuso et al., 2015). For example, total annual fisheries catch from the four main EBUS highlighted in this box (California Current, Humboldt Current, Canary Current and Benguela Current) amount to 16-24 MT in the 2000s, approximately 17% of the global catches (based on data from (Pauly and Zeller, 2016)). Upwelling of cold deeper water increases condensation of wet air in coastal areas that brings in moisture and precipitation that are important for maintenance of coastal vegetation, agriculture, as well as prevention of forest fire Black et al. (2014). The high concentration of marine mammals supported by the productive upwelling ecosystem support lucrative eco-tourism such as whale-watch in California Current (Kämpf and Chapman, 2016). A total economic value calculation of the goods and services provided by the Humboldt Current alone is estimated to be US$19.45 billion per annum (Gutiérrez et al.). Thus, climate change impacts on EBUS would have disproportionately large ramification on human societies relative to its smaller area compared to other pelagic ecosystems (*very high confidence*) EBUS are vulnerable to the multiple effects of climate change (Blasiak et al., 2017) in spite of the high resilience of these ecosystems adapted to the high natural intrinsic variability (see review in (Bakun et al., 2015), also see Box 5.3, Table 1) (*medium confidence*); however, there is large regional variation in how the natural-human systems are changing under anthropogenic stress and how they might change in the future Allison and Bassett (2015); (Gattuso et al., 2015). Climate induced changes in the biophysical conditions of the major EBUS in the world ocean to increasing CO_2 emission may differ. In general, EBUS show the same general century scale increases in sea surface temperature (Demarcq, 2009; Pennington et al., 2017)and decreases in pH (Turi et al., 2016; Chavez et al., 2017) and dissolved oxygen (Levin, 2018) seen in the global estimates, although these changes are not steady and vary between and along the different EBUS

(Table B5.2) (*low to medium confidence*). Predicted increases in upwelling-favourable winds, caused by enhanced land-ocean differential heating due to global warming (Bakun, 1990), are also uncertain because of large regional differences in observed and predicted changes (*low confidence*). Some recent coupled climate model studies and meta-analysis of observational data suggest that upwelling-favourable winds have intensified in the poleward areas of EBUS and will continue to increase by the end of the 21st century (Sydeman et al., 2014; García-Reyes et al., 2015; Rykaczewski et al., 2015). However, these patterns are not consistent across all the EBUS (Sydeman et al., 2014; Wang et al., 2015). Moreover, coastal warming and wind intensification may lead to variable countervailing responses to upwelling intensification at local scales (García-Reyes et al., 2015; Wang et al., 2015; Oyarzún and Brierley, 2018; Xiu et al., 2018). Local winds and mesoscale oceanographic features (not resolved in most coupled climate models), rather than large-scale wind patterns, are thought to have a greater impact on regional productivity (Renault et al., 2016; Xiu et al., 2018) (*low confidence*). In the California Current EBUS, increases in primary productivity of 3% over the 1990 to 2011 period have been reported (Pennington et al., 2017). However, recent analysis of net primary production derived from remote sensing models for other EBUS, like the Canary Current, do not show significant trends over the past two decades, in spite of the large sub-regional variability in absolute rates (Gómez-Letona et al., 2017).

The climate change impacts on ecosystem services from EBUS vary regionally depending on both the biophysical and the social and economic characteristics of the upwelling systems ((García-Reyes et al., 2015), Box 5.3, Table 1; Box 5.3, Figure 1). 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. The main industrial-scale fisheries in the upwelling systems are highly sensitivity to upwelling conditions, for example, with large decreases in anchoveta population in the Humboldt Current during El Niño that impacts the fishing industries Gutiérrez et al. (2017). Since the small pelagic fisheries from upwelling regions are the main source of the global fishmeal market, the decrease of its production leads to increase in international fishmeal price that consequently affects the price of other food commodities that rely on fishmeal for their production e.g., aquaculture (Merino et al., 2010; Carlson et al., 2017). Moreover, the decrease in catches would also affect regional food security as catches from the upwelling region is an important source of nutrients. For example, in the Canary Current, coastal fisheries are important source of micronutrients to the nearby West African countries (Golden et al., 2016), where their risk to climate change impacts are particularly high because of their strong dependence on the fisheries resources, a rapidly growing population and regional conflicts render these communities having high sensitivity and low adaptive capacity to climate impacts (Box 5.3, Figure 1) Decrease small pelagic fish stocks also increase the mortality and reduce reproduction of large vertebrates such as hake (Guevara-Carrasco and Lleonart, 2008), whales and seabirds (Essington et al., 2015) while reduction in upwelling strength may improve oxygenation of the sea floor that could be beneficial to demersal fisheries (Arntz et al., 2006). Ecosystem services that appear to be most at risk to climate change in the 21st century include fisheries (*high confidence*), aquaculture (*medium confidence*), coastal tourism (*low confidence*) and climate regulation (*low confidence*), particularly in Canary Current EBUS that are expected to have high risk of impacts from climate change (*high confidence*, Box 5.3, Figure 1).

Overall, climate change effects on upwelling system can have far reaching impacts to human society, including both coastal dependent communities as well as those that are far from the coast (*high confidence*).



Box 5.3, 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).

Box 5.3, Table 1: Observed and projected climate-change impacts on Eastern Boundary Upwelling System (EBUS). Based on García-Reyes et al. (2015).

EBUS characteristic	**Evidence for current changes**	**Evidence for projected scenarios (Expected)**	**Confidence** *High**** *Medium*** *Low**	**Uncertainty**	**References**
Local and Large-scale drivers of upwelling-favourable winds	Intensification of upwelling-favourable winds (except in the CanC) during warm months. Increased variability and trends in phenology. No evidence of multi-decadal trends in strength or position of pressure systems.	Increasing (decreasing) trends in upwelling winds in poleward (equatorward) regions of the EBUS. Models and observational data suggest poleward migration of ocean high-pressure systems, but little to no change in intensity of the thermal continental low-pressure cells.	California** Humboldt** Canary** Benguela**	Uncertainty due to local and regional factors of the EBUS.	Sydeman et al. (2014) García-Reyes et al. (2015) Wang et al. (2015) Sulca et al. (2018) Oyarzún and Brierley (2018)
Coastal temperature and stratification	Complex integration of global (climate change) and local (coastal upwelling) processes.	Ocean warming and stratification might be ameliorated by increased upwelling. Coastal upwelling reduced by surface	California * Humboldt* Canary* Benguela*	Trends are sensitive to location (nearshore vs. offshore), resolution,	García-Reyes et al. (2015) Wang et al. (2015) Oyarzún and Brierley (2018)

	Differences in global and EBUS SST trends are observed.	warming.		period and dataset considered, as well as to decadal variability.	
Acidification	A time series from the CalC shows that the 1993 to 2016 trend in sea surface *p*CO_2 is not significantly different from that in the atmosphere. In CalC pH and $\square_{arag}$ in the top 60 m of the water column decreased from 1979–2012 by about−0.02 per decade and−0.12 per decade, respectively with shoaling of the aragonite saturation depth of on average−33 m per decade and up to−50 m per decade HCast-CCMP simulation indicate however a marked decreasing trend in pH and Ω_{arag} up to 50% larger in nearshore regions than in the open ocean, from 1979 to 2012 in the CalC	Decreased in pH, Ω_{arag}, and in the aragonite saturation depth. In CalC< 100m ph controlled by DIC/upwelling. Upwelling of low pH waters onto the shelf. Ω_{arag} drops rapidly, with much of the nearshore region developing summer-long undersaturation in the top 60 meters by 2040. By 2050, waters with Ω_{arag} above 1.5 will have largely disappeared, and more than half of the waters will be undersaturated year-round.	California** Humboldt* Canary* Benguela*	Uncertainty due to short period of data availability, as well as to decadal variability (e.g., ENSO). Lack of comparison between all EBUS	Gruber et al. (2012) Bakun et al. (2015) Turi et al. (2016) Chavez et al. (2017) Turi et al. (2018)
Deoxygenation	Oxygen declines of 20-40% are documented over the past half century within the CalC, but not off Peru. In CalC -0.023 ml y^{-1} O2 loss on σ = 26.5 kg m"3 isopycnal surface from 1984-2012. The OMZ and nutrients impacted by El Niño Southern	Decreasing oxygen at intermediate water depths. Increasing frequency of hypoxic events. In CalC <100 m temperature controls oxygen via solubility. >100 m upwelling controls oxygen, especially during La Nina.	California*** Humboldt** Canary** Benguela***	Uncertainty due to short period of data availability.	Bakun et al. (2015) Bograd et al. (2015) Graco et al. (2017) Breitburg et al. (2018) Levin (2018) Turi et al. (2018)

	Oscillation (ENSO) at interannual scales of variability. Seasonal variation in O_2 and pH.				
Nutrients, Plankton and Primary Production	Variable trends in nutrients, chlorophyll and plankton. Increased nitrate and phosphate and elevated primary production (3%) over 1990-2011 in the CalC; flat trend over 1998-2015 in the CanC	In CalC elevated nutrient flux and variable trends and complex non-linear response in primary production. Mismatch of variables that fishery scientists identified as important in determining species' response to climate and physical forcing and the variables that current GCMs can now resolve at the regional level.	California** Humboldt* Canary* Benguela*	Deep uncertainty, given to range of relevant processes operating at different temporal and spatial scales. NPP derived from remote sensing models lacks validation with observational data in most EBUS	Chavez and Messié (2009) King et al. (2011) Bograd et al. (2015) Espinoza-Morriberón et al. (2017) Gómez-Letona et al. (2017) Pennington et al. (2017) Xiu et al. (2018)
Provisioning Services	EBUS contribute disproportionately to ocean biological productivity with nearly 25% of the reported global fish catch. Decrease finfish fisheries. CalC off Oregon – hypoxia has supressed fishery catch Expansion of hypoxia-tolerant resources (Jumbo Squid)	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. In CalC changes in fish distribution (northward, shoreward movement of sardine, hake, shark, tuna), decline in Chinook Salmon, increase in Dover Sole, sablefish, rockfish. Seabird decline Threats to offshore aquaculture	California*** Humboldt** Canary*** Benguela***		Arntz et al. (2006) Keller et al. (2010) King et al. (2011) Doney et al. (2012) 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	California*		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.	Canary * Benguela**	Allison et al. (2009) Blasiak et al. (2017)

[END OF BOX 5.3 HERE]

5.3.1.2.2 *Regulation, supportive and cultural services*

In addition to fisheries as provisioning services, other key ecosystem services from the pelagic open ocean includes carbon storage (regulating service), tourism and recreational fishing as cultural service, recreational, and biodiversity as a supporting service Martin et al. (2016), which will be impacted by climate change (*medium confidence*). Specifically, the ocean is estimated to have taken up 32 ± 5 PgC over the period 1994–2007 (see Section 5.2.1.3) (IPCC, 2007; IPCC, 2014) thus making the biggest contribution in carbon sequestration relative to coastal and deep sea ecosystems. Assuming a carbon price of USD7 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 2000–2012 (see Figure 5.8). Because of saturation effects, ocean carbon update is projected to decrease even under increased atmospheric CO_2 under RCP2.6.

Recreation and tourism in the open ocean are popular in many regions. For example, total expenditure and employment generated by whale watching operations in the world are estimated to be USD413 million and employs 5762 people, respectively (Cisneros-Montemayor et al., 2010). Sport fishing on large pelagic 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*).

5.3.1.3 *Deep Sea Floor Ecosystems*

Deep-ocean processes and organisms (Table 5.5) generate many types of provisioning regulating, supporting and cultural ecosystem services (Figure 5.20) (Armstrong et al., 2012; Thurber et al., 2014), most of which are poorly studied and rarely quantified. The stakeholders for the deep ocean, which covers 54% of the surface area of the planet and most of the habitable volume, are all the people on the earth. This realm holds a wealth of living and non-living resources and provides services that are likely to sustain humans for centuries to come. Understanding climate change impacts on deep-sea ecosystem services and valuating their changes is particularly challenging as the realm is vast, remote and only partially explored; many deep-sea habitats are unfamiliar and the functions of organisms are poorly known.

Increasing demands on deep-sea resources such as fish/shellfish, oil and gas and seabed minerals are creating the need to identify, understand and valuate the services provided by vulnerable deep-sea ecosystems and raising interest in ecosystem services in the context of environmental and financial management (Le et al., 2017). In shallow water there is an appreciation that ecosystem-based approaches can contribute to both climate solutions as well as biodiversity and conservation goals (Wüstemann et al., 2017), but similar efforts for the deep ocean are just beginning, despite its major role in climate mitigation through heat and CO_2 uptake (Gattuso et al., 2015). New approaches such as the Ecosystem Principles Approach (EPA) are being developed or applied to draw the link between the ecological and socio-economic dimensions of the deep sea (Table 5.5) (Jobstvogt et al., 2014). When Willingness to Pay (WTP) methods were applied to cold-water corals (CWCs) the possibility that CWCs play an important role as habitat for fish was the single most important variable to explain respondents' WTP for CWC protection (Aanesen et al., 2015). There is a need for advanced thinking about ecosystem services in the deep ocean, beyond intrinsic and instrumental values to incorporate relational values (Eudaimonia) that collectively acknowledge the human associations with nature in multiple forms across different cultural practices (Chan et al., 2016).

Table 5.5: Examples of detected or projected climate change impacts on key deep sea floor ecosystems.

Mode of attribution: * theoretical inference; + model projection; # field observation ^ laboratory experiment $ paleo record

Confidence: yellow - low; orange - medium, red - high, white - no assessment

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 organic matter export and benthic production (+)	Reduced food provision due to loss of organic matter export and benthic production (+)	Unknown effect: Species reliant on methane seeps could benefit if warming induces additional methane release ($); loss of chemosynthetic foundation species (mussels, tubeworms) expected on upwelling margins as OMZs expand (#)	Loss/dissolution of nursery habitat could translate into loss of fish production and fisheries (+)	Limited food provision from this setting	Transient increased catch may occur as a result of 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 biotechnological, pharmaceutical 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
Disease and invasive species	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
Supporting (intermediate)						
Habitat and trophic support	Habitat loss, reduced biodiversity, loss of spawning and nursery grounds (*, +)	Habitat loss, reduced biodiversity, loss of spawning and nursery grounds (*, +)	Some commercial species affiliate with methane seeps; elasmobranchs use vents and seeps as nursery grounds. Could expand services. (#)	Habitat loss inducing loss of nursery grounds, trophic support (*, +)	Unknown. Potential loss of VMEs such xenophyophores; could interact with physical disturbance from mining (*, +)	Warming, OMZ expansion and acidification causing habitat loss and reduced trophic support for hypoxia/acidification/warming-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 (+)
Chemosynthetic primary production	No assessment	No assessment	Produduction 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, deoxygenation, acidification (*)	Reduced via decline in food supply, warming, deoxygenation, acidification (*)	Reduced via deoxygenation, 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, deoxygenation, 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

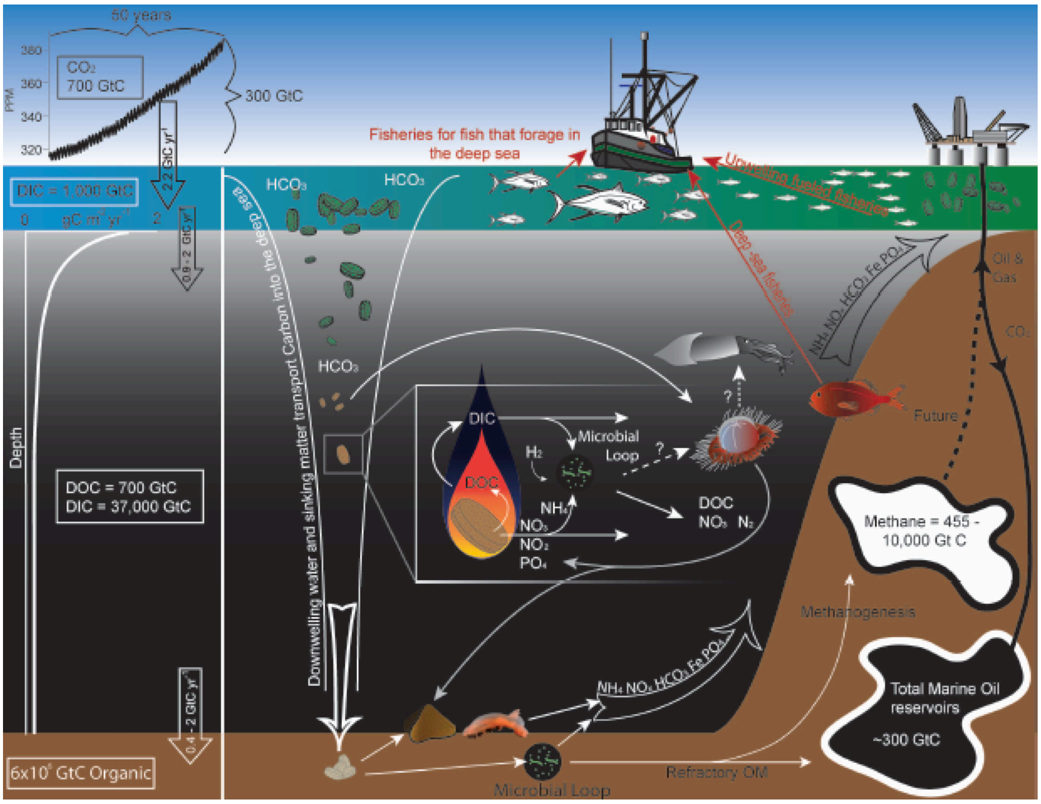


Figure. 5.20: Deep-sea processes underlying carbon and nutrient cycling (regulating) and market services (From (Thurber et al., 2014)).

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

The effects of warming and ocean deoxygenation on deep sea organisms and ecosystems (Section 5.2.3) affect the quality and quantity of deepwater fisheries catches (*moderate confidence*). Warming may reduce body size, cause oxygen limitation, and raise energy demands (*moderate confidence*). It will also redistribute both vertebrate and invertebrate fishery species, generally moving them to shallower depths and higher latitude (habitat compression) (*high confidence*), with effects on availability, accessibility, costs of fishing and transport (*medium confidence*) (Cheung and Pauly, 2016; Lam et al., 2016). Habitat compression above oxyclines can increase catchability but may lead to overfishing or increased vulnerability to natural predators (Prince et al., 2010; Stramma et al., 2011). Ocean deoxygenation is leading to expansion of oxygen minimum zones in the tropics and subtropics (Stramma et al., 2008; Stramma et al., 2010). On the US West coast, near-bottom oxygen concentration is positively associated with both the biomass of commercially harvested fish species (Keller et al., 2010) and catch per unit effort (Keller et al., 2015). Deoxygenation is predicted to reduce available rockfish habitat (McClatchie et al., 2010) and rockfish (*Sebastes spp*.), and crabs (*Cancer magister*) are also likely to be negatively impacted by expanded hypoxia (Gallo and Levin, 2016). In Peru, India, and Pakistan, the catch per unit of effort (CPUE) of demersal fisheries decrease when bottom oxygen levels decline (Banse, 1968; Rosenberg et al., 1983), and some species such as hake, octopus and scallops only appear on the Peru margin during oxygenation events (associated with El Niño) (Arntz et al., 2006). Beyond direct effects, deoxygenation can reduce the diversity (Sperling et al., 2016), size and abundance (Levin, 2003) of infaunal prey, as well as supress benthic-pelagic trophic interactions. However, hypoxia-tolerant species such as tuna crabs (*Pleuroncodes planipes*) (Pineda et al., 2016), Humboldt squid (*Dosidicus gigas*) (Stewart et al., 2014) and the pink urchin *Strongylocentrotus fragilis* (Sato et al., 2017) appear to be expanding their ranges (*medium confidence*) and could offer new fishery opportunities (Sato et al. 2018b). As OMZs expand on margins waters also become more acidic (Paulmier et al., 2011), with potential effects on calcifying taxa (see Section 5.2.2.4) (*medium confidence*).

5.3.1.3.2 Other services

The large areas and long residence times over which regulating services are provided in the deep sea mean that they are massive (*high confidence*), although the processes are often far removed from the humans that receive them (Thurber et al., 2014). The deep ocean ecosystem service that is most directly linked to climate is the sequestration of carbon, both the storage in the deep-sea pool and long-term burial in sediments (*high confidence*). A net flow of approximately 1.6 GtC yr^{-1} from the atmosphere into the deep sea, of which 0.2 GtC is buried, represents a value of USD108 billion yr^{-1} if valued at a conservative USD19 per tCO$_2$-eq (which is USD62 per tC assuming all the carbon is in the form of CO$_2$) (Armstrong et al., 2010) (*medium confidence*). The value of carbon export to the deep ocean was estimated at USD12.9 billion yr^{-1} (using average European C market prices) for the Eastern Tropical Pacific covering 21 million km^2 with waters of 12 nations + international commons) (Martin et al., 2016) (*medium confidence*). Only one study has examined the economic cost of climate driven changes in carbon fluxes. Declines in C flux at 1000 m in the North Atlantic, estimated between 27–43% by 2100 depending on RCP scenario, were valued at USD170–300 billion in mitigation costs and 23–401 billion in adaptation costs over the 21st century (Barange et al., 2017) (*low confidence*). Carbon sequestration by the biological pump over the Mediterranean Basin was estimated at 281 million Euros per year (=USD29 billion over the same period) (Melaku Canu et al., 2015) (*low confidence*). Uncertainties in these values derive from uncertainties in the ecosystem models (mechanisms, scenario pathways) and in carbon price estimates and discounting rates (Treasury, 2011; van den Bergh and Botzen, 2015). Other forms of carbon sequestration in the deep-sea merit valuation, such as the removal of methane released from the beneath the seabed through microbial anaerobic oxidation and the sequestration in carbonate as well as aerobic methane oxidation by free-living microbes and symbionts (in mussels, sponges, ciliates) (Marlow et al., 2014; Thurber et al., 2014). These services could become increasingly important as warming accelerates methane release (*low confidence*).

Cycling of nitrogen, silica, phosphorus, hydrogen and sulfur also occur in the deep sea. mediated by nitrogen fixation, denitrification, nitrification and anaerobic ammonium oxidation in deep waters and sediments (Marlow et al., 2014) (Figure 5.20) (*high confidence*). Warming, acidification and deoxygenation will influence nitrogen sources, sinks and transformations (*low confidence*), but the extent of change and the value of these services have yet to be quantified.

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

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

Biodiversity, which is understood to be high in the deep sea but remains mostly undescribed (Ramirez-Llodra et al., 2010), has existence value itself for genetic resources (pharmaceuticals industrial agents, cosmetics, biofuels or biomaterials (Skropeta and Wei, 2014; Harden-Davies, 2017), but also contributes significantly to overall ecosystem functioning that supports other services as well. Ocean deoxygenation will reduce biodiversity of fauna at intermediate depths (see Section 5.2.3.3) but valuation of deep-sea biodiversity is in its infancy and translation of climate-induced loss into services has yet to occur. The greatest effect of biodiversity loss is likely to be loss of support functions. Nearly all deep-sea supporting services (e.g., secondary production, nutrient remineralization, trophic support, habitat provision, nursery and spawning grounds) that support fisheries production and climate regulation (Thurber et al., 2014; Bopp et al., 2017) will be affected by changes in temperature, oxygen, pH and POC flux, as discussed in (Section 5.2.3) on impacts on ecosystems.

5.3.2 Climate Risk, Vulnerability and Exposure of Human Communities and their Wellbeing

Human communities are heavily dependent on the oceans through the goods and services that marine ecosystems offer (Hilmi et al., 2015a). As climate change is impacting marine biodiversity and ecosystem services (see Section 5.3.1), various dimensions of benefits that contribute to sustaining human communities and their wellbeing will also be affected. However, detection and attribution of climate change impacts on the wellbeing of human communities are confounded by wide-ranging non-climatic factors that largely affect the exposure, sensitivity and adaptive capacity of human communities to climate impacts. For example, the shift from traditional nutritious wild caught seafood-based diets of coastal Indigenous communities, towards increased consumption of processed (and other) energy-dense foods that are high in fat, refined sugar, and sodium, due to social and economic changes (Kuhnlein and Receveur, 1996; Shannon, 2002; Charlton et al., 2016; Batal et al., 2017), may reduce communities' exposure (Quinn et al., 2012) to climate change effects on fish stocks . Such changes however, have important consequences on diet quality and nutritional status (Thaman, 1982; Luick et al., 2014), and have been and parallel by an increased prevalence of obesity, diabetes, and other diet-related chronic diseases (Gracey, 2007; Sheikh et al., 2011).

Therefore, in this section, risk of climate impacts on human communities are based on assessments on the level of dependence of ecosystem services, and the risks of climate impacts of these ecosystem services (Section 5.3.1). The current and future sensitivity and adaptive capacity of these communities are based on qualitative evidence combined with published indicators under different Shared Socio-economic Pathways (SSPs) (O'Neill et al., 2014). The linkages between ecosystem services and human communities and their wellbeing are sub-divided 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 (WBCSD). These three pillars are social and cultural (Section 5.3.2.1), economic (Section 5.3.2.2) and environmental (Section 5.3.2.3) (Table 5.6). The Sustainable Development Goals (SDGs) published by the United Nations provide a context to synthesize climate risks on these different pillars on the future sustainability of human societies (Section 5.3.2.4).

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

Pillars of sustainable development	**Dimensions assessed**	**Linkages to biodiversity and ecosystem services**	**High-dependent communities**	**Key risks**
Social and cultural	Human health including diseases, migration and displacement, education indigenous and local culture	Distribution and prevalence of water-borne pathogens Fisheries and aquaculture Traditional and local ecological knowledge	Coastal Indigenous communities and the global south with lack of access to education, health care and infrastructure	Decrease in potential seafood production (5.3.1)
Economic	Wealth and poverty, livelihood, food and nutrition.	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	Loss of jobs and incomes Loss of property Decrease in food supply Decrease in diverse nutrition sources
Environments	Human relationship with the environment, natural capital,	Aesthetic sense of place, Traditional and local culture	Coastal Indigenous people Coastal populations in poverty	Impacts on mental wellbeing, ability to adapt to societal and environmental changes, including climate change

5.3.2.1 Social and Cultural Dimensions

The sustainable development paradigm includes domains that concern the intersection between the social and the natural realms, focusing on human engagement with and within nature, but also including the built-environment. This includes the linkages between the environment, ecosystem and human health. This sub-section will focus on different dimensions of risks on human health and security as a result on climate change effects on the oceans, biodiversity and ecosystem services. Specifically, the assessment focuses on ecosystem-mediated impacts of climate change, impacts of pollutants other than CO_2 and climate impacts on human security and conflicts. The cultural aspects concern the indigenous culture and the education. Climate change will also impact cultural dimensions.

5.3.2.1.1 Ecosystem-mediated impacts of climate change on disease

Ocean warming is expected to increase the risk of water-borne diseases (*low confidence*). Nontoxigenic *Vibrio cholerae* is a bacterial pathogen causing either gastrointestinal disease through food poisoning after consumption of contaminated seafood, or wound infections and associated blood poisoning, after exposure to contaminated seawater. Health risk from exposing to *Vibrio cholerae* is particularly high through severe wound infections in elderly and other individuals with suppressed immune system (Daniels, 2011). Wound infection of *Vibrio cholerae* is associated with necrotizing ulcers, septicaemia, and even death. Particular risk has been associated with bathing with open wounds (Daniels, 2011). The nontoxogenic *Vibrio* species are prevalent worldwide in warm, nutrient-rich and low salinity coastal waters. The bacteria are, for example, proliferating in the Baltic Sea, or around coastal estuaries. In coastal area where waters are expected to be warmer, have lower salinity from decreased precipitation and higher chlorophyll concentration under climate change (see Section 5.2.2.3), *Vibrio* is expected to grow faster (Escobar et al., 2015). For example, sea surface temperatures above 16°C was associated with rapid increases in the relative risk of *Vibrio cholerae* infection along the Swedish coast of the Baltic Sea (Semenza et al., 2017). A

considerable risk of infections was projected for the Baltic region, associated with climate change scenarios of warming sea temperature (Semenza et al., 2017). The prevalence of *Vibrio* increases under warming (*high confidence*, see Section 5.2.2), however, the global surveillance of the disease in human population is limited and the nontoxigenic *Vibrio* infection is rarely notifiable to the health agencies, leading to limited understanding of the disease epidemiology and risks, and their attribution to climate change, worldwide (*low confidence*).
Increase frequency of coastal floods from storm surges and sea level rise in response to future climate change (Section 5.2.2) will also increase the risk of water-born diarrheal disease (*low confidence*). Coastal floods are associated with intrusion of seawater and wastewater into fresh water leading to water shortage and risks of water-borne diarrheal disease (Kim et al., 2014). Floods are also related to displacement, which is known to increase vulnerability of populations, as well as contribute to spreading of infectious diseases. However, uncertainty in the social and economic developments shaping vulnerabilities of populations makes estimates of absolute mortality highly uncertain (Lloyd et al., 2016).

The impacts on health is not only a social, but also an economic problem. It is a health care issue and it may reduce the productivity of workers if they are undernourished. Moreover, because of the loss of biodiversity in the seas due to climate change (Sections 5.2.3, 5.2.4), the chance to find pharmaceutical products from the sea will decrease (Section 5.3.1). Many traditional medicines coming from the sea may also disappear. This will have a huge impact on human health and on the benefits of the pharmaceutical industry (Malve, 2016) (*low confidence*).

5.3.2.1.2 Interactions between climate change and non-CO_2 pollutants

Climate change–contaminant interactions may alter the bioaccumulation and biomagnification of two contaminant classes: the toxic and fat-soluble persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs), as well as the neurotoxic and protein-binding organic form of mercury, methylmercury (MeHg) ((Alava et al., 2017), *medium confidence*). POPs are bioaccumulated by marine organisms and biomagnified in food webs, reaching exposure concentrations that become harmful and toxic to populations of apex predators such as marine mammals Desforges et al. (2016) (Figure 5.21). Exposure to POPs can lead to serious health effects including certain cancers (i.e., by PCBs and dioxins), birth defects, and impairments to the immune and reproductive systems, as well as greater susceptibility to disease and damages to the central and peripheral nervous systems (Lallas, 2001). MeHg is highly neurotoxic and nephrotoxic and bioaccumulates and biomagnifies throughout the food web via dietary uptake (Fort et al., 2015). Consumption of mercury-contaminated fish is linked to disease in humans, including the neurological disorders due to MeHg poisoning (i.e., Minamata disease) (Ishikawa and Ikegaki, 1980; UNEP, 2013). Of particular concern is the pollution risks influenced by climate change in the Arctic ecosystems and indigenous communities because of the protracted bioamplification of POPs and MeHg with associated long-term contamination of their traditional foods (Marques et al., 2010; Tirado et al., 2010; Alava et al., 2017).

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

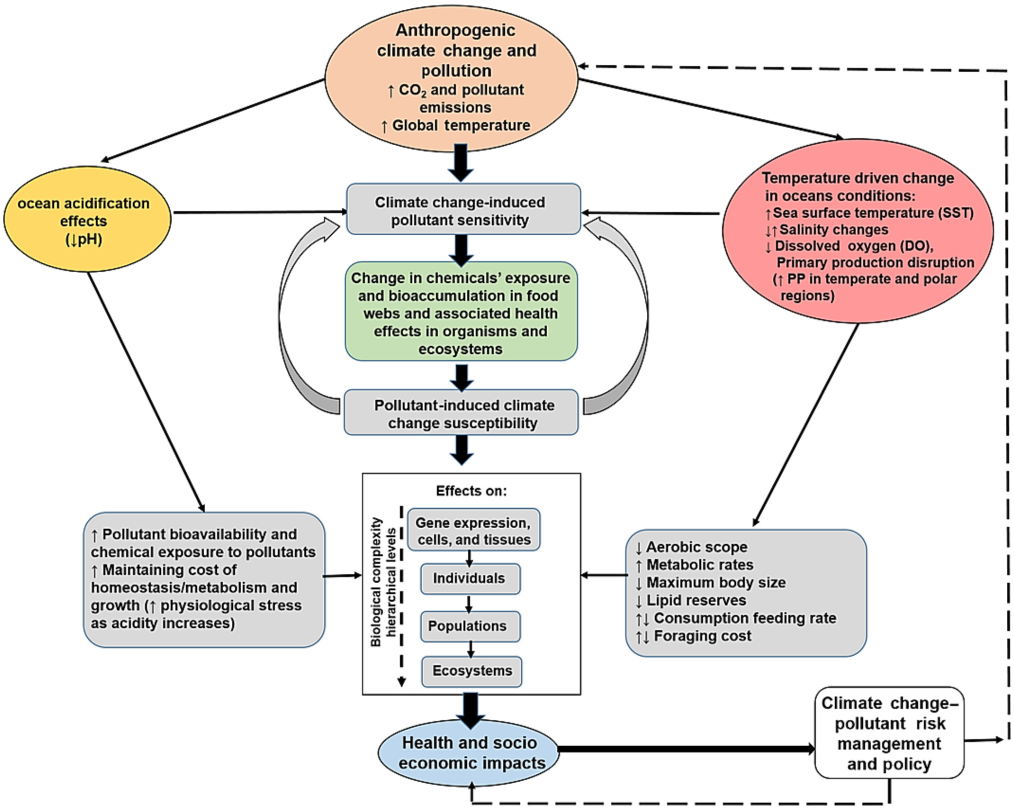


Figure 5.21: The biophysical, eco-physiological and eco-toxicological pathways of impacts of climate change through climate change-induced pollutant sensitivity on contaminant bioaccumulation in marine food webs and effects on hierarchical levels of biological organization, with implications for public 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).

Climate change alters the degree of human exposure to pollutants and the response of human populations to these exposures Balbus et al. (2013) (*medium confidence*). Changing human behaviour will also affect how humans come into contact with contaminated air, water, and food in an era of climate change. Climate change and regional climate variability may have an impact on the occurrence of food safety hazards, including changes in transport pathways for contaminants, at various stages of the food chain, from primary production through to consumption (Marques et al., 2010; Tirado et al., 2010). While species inhabiting industrialized areas may be most at risk for contaminants, the particular vulnerability of the Arctic climate suggests that the latter region may be particularly sensitive and susceptible to climate–contaminant interactions (Chapter 3). In this context, contaminant concentrations in marine mammal species and wildlife populations from the Arctic already exceed overall biological effects of concern in relation to POP exposure, i.e., 1 mg kg^{-1} or ppm (Letcher et al., 2010). This has obviously potential health impacts for Inuit and indigenous communities, who rely strongly on traditional seafood and marine mammals in a warming Arctic (Alava et al., 2017).

The high exposure to climate-pollution risks is not limited to Arctic communities, but indigenous communities and First Nations peoples from other ocean and coastal regions of the world (i.e., temperate, sub-tropical and tropical regions) in general (*medium confidence*). 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 oceans and rely strongly on the harvest and consumption of traditional seafood and fish products. As a result, these communities are also exposure to higher pollution risks and their potential exacerbation from climate change effects.

5.3.2.1.3 Human security and conflicts

There is the general narrative that climate change may increase conflict risks (Gilmore, 2017), but there is limited empirical evidence in relation to fisheries and oceans (*low confidence*). Yet three main categories of risks on human communities and their wellbeing from climate change effects on the oceans through conflicts can be identified. First, the effects on ocean temperatures and currents can create or exacerbate conflicts over fisheries as a result of fish stock fluctuations and displacements (Spijkers and Boonstra, 2017), as well as the movements of fishers (Belhabib et al., 2016; Pomeroy et al., 2016; Blasiak et al., 2017). Second, coastal communities displaced by extreme weather events, including flooding and hurricanes, can result in conflicts in receiving areas (Reuveny, 2008) or exacerbate tensions among 'trapped populations' unable to afford a move out of vulnerable areas (McGranahan et al., 2007; Adger et al., 2014). Sea level rise can also lead to conflicts over the redefinition of territorial claims between countries, especially in areas where large area of the exclusive economic zones or the very existence of countries are at stake (Maas and Carius, 2012; Stoutenburg, 2015), or maritime sovereignty is frequently disputed (Nyman, 2013). Third, climate protection and marine conservation measures for coastal populations can lead to conflicts (Bennett and Dearden, 2014; Kamat, 2014; Sovacool, 2018). More indirectly, negative inland climate effects can push some communities to coastal areas, exacerbating pressure on local fisheries and social tensions between long-term coastal residents and in-coming populations (Bohra-Mishra et al., 2014). The impacts of these conflicts on communities and their wellbeing can include loss of livelihood and access to fish, forced displacement and community breakdown, as well as the physical and psychological harms of violence (*low confidence*). Climate change impacts on rural communities may displace towards cities, although such migration may not occur for low-income countries (Cattaneo and Peri, 2016). Although many cities are located in coastal areas, the impact of outmigration to coastal fishing communities is not well documented. Also, the relationship between climate change, migration and conflicts in unstable urban and rural demographics may be weak and short-term (Fort et al., 2008) and affected from sampling and reporting biases (Adams et al., 2018), rendering the assessment of climate change effects on human conflicts having *low confidence*.

5.3.2.1.4 Indigenous culture

How climate change can affect indigenous knowledge and ocean education can be illustrated in the case of the Pacific where this is a delicate issue as the changes are different in nature and refer to different time frames (see also Cross Chapter Box 3). Geological and climatic temporality refers to a long time that is evaluated over millions or thousands of years. On the other hand, traditional knowledge, even if it has been transmitted for several generations, can be appreciated over shorter periods ranging from a few centuries to several generations. Thus, the way in which climate change modifies perceptions, practice, transmission and the network of local knowledge on the ocean implies a reworking of this knowledge where the person and the groups are part of a narrative and historical construction (Roué, 2012; Alderson-Day et al., 2015). In this sense, the latest surveys conducted in Kiribati and the Tuamotu archipelago in French Polynesia (2016) highlight the gap between the scientific discourse predicting unprecedented climate change and the local perceptions that consider it to be is an ‘outside’ discourse in a context where science has often been considered as the ally of a historic colonial power (Tuhiwai, 2013). Nevertheless, climate change is already transforming the shape of seashore in many low islands. Some sandbank and rocks representing gods and mythological ancestors are desappearing (Camus, 2017). Other studies in Tuvalu shows that shows that sea level rise and climate change during the past four decades had resulted in a net increase in land area (Kench et al., 2018) but in return it may affect mobility of people, residence pattern, overcrowded effects and the structure of traditional knowledge in a way that more mythological syncretism may be produced by the concentration of people in the safest places.

Thus, the great myths of foundation or the ocean plays a central role are re-elaborated, the knowledge and the practices are modified, not because science prophesies a major climate change, but because the local populations observe it: scarcity of the resources pelagic and lagoon, the disappearance of a mythical sandbank at Tabiteua (Kiribati) that forever alter the historical trajectory of a population and its relationship to the ocean (Camus, 2017).

In addition, in the Pacific as elsewhere, there is a shift in the transmission of local knowledge about the ocean, from a family monopoly to a plurality of actors including compulsory schooling. The fear of sea level rise and climate change encourage security measures and the grouping of local people to the safest places. Therefore, not only do this plurality of modes of transmission and that concentration of people contribute to the erosion of local knowledge about the ocean (Bambridge and Le Meur, 2018), but it has resulted in almost no 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 used, are still in effect (Bambridge and Le Meur, 2018) and the validity of their prediction is often discussed, compared to the supposed effects of change climate.

This paradoxical situation, however, has encouraged Pacific peoples to embrace climate change not in the terms of science, but according to their cultural categories and patterns. For example, many Polynesian and Micronesian societies have been involved in the construction of double canoes reminiscent of major exploration voyages. Today, they contribute to re-enactment of old sailing techniques while at every stage of their journey they strive for a respectful relationship between society, the ocean and the climate. In the same way, the last ten years have seen the introduction of ancestral techniques for the conservation of lagoon resources, called rahui in eastern Polynesia (D'Arcy and Bambridge, 2014), now extended to previously unknown scales. The Cook Islands protect their exclusive economic zone, with the establishment of a rahui, marae o te moana—the sacred temple of the ocean—over an area of 1.9 million km^2. French polynesia has pledged to do the same on an area of 5.5 million km^2. This vast historical movement goes beyond a simple goal of adaptation to climate change. It is also a question of restoring an Oceanian identity to a space in which they consider themselves to be the historical guardians (Hilmi et al., 2016). Finally, if United Nations circles are considering granting status to 'environmental displaced', Pacific Islanders from many islands are acquiring land in the Pacific regions bordering their areas (New Zealand, Australia, and the USA) to secure the future of next generations.

Recent research as demonstrated that Indigenous fishing communities that depend on traditional marine resources for food and economic security are particularly vulnerable to climate change through a reduced capacity to conduct traditional harvests because of limited access to, or availability of, resources (Larsen et al., 2014; Weatherdon et al., 2016) (*high confidence*). For instance, shellfish and traditional clam beds—which form an integral part of the culture, economy and diet of many Indigenous communities situated along the Pacific Coast of North America— may be affected detrimentally by increased SST, sea level rise, and changes in ocean chemistry and circulation patterns (Lynn et al., 2013). This has critical implications for the food and economic security of coastal Indigenous communities, the preservation and transfer of their traditional knowledge, and the legal upholding of their rights to access traditional resources (Lynn et al., 2013) (*medium confidence*).

5.3.2.1.5 Education

Education can play a pivotal role in how climate change is perceived and experienced (*high agreement*). 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). Ocean acidification has been described as an excellent educational tool to address climate change issues (Fauville et al., 2011).

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

5.3.2.1.6 Other cultural dimensions

While climate change is impacting marine ecosystems (Section 5.2), it is unclear how they will affect the cultural wellbeing of human communities (*low confidence*). Among the cultural wellbeing supported by the oceans, education from scientific knowledge obtained from marine environments, the educational value and the economic and technical benefits generated by exploration and discovery feature prominently. There are also aesthetic and inspirational services, including film, literature and art, as well as tourism and spiritual well-being (Pescaroli and Magni, 2015). For many island nations, their deep ocean is hundreds to thousands of times larger than their land, and most of their natural capital lies in the deep sea. Some such as Rapa Nui (Easter Island) have a strong spiritual connection with this element. A reaction common to many low islands Pacific countries such as Kiribati, Cook island, Pascua Island, French Polynesia, is to create huge large marine protected areas as an adaptation measure to climate change (Bambridge and Le Meur, 2018). This in return will affect the catch capture in the Pacific and the economy of long distant fleet for example.

Climate change threatens many cultural dimensions of lives and livelihoods (Adger et al., 2012) (*medium confidence*). People develop strong cultural ties and associate distinctive meanings with many natural places in the form of traditions, customs and ways of life (Marshall et al., 2018). Nature offers a place within which people can belong, whilst offering other places to visit and be inspired by. Nature provides a day-to-day sense of meaning through providing opportunities for intellectual engagement, shared recreation, economic well-being and support for livelihoods. The cultural opportunities provided by ecosystems are many, and include concepts such as attachment to place, heritage, rootedness, spirituality, tranquility, escape, togetherness, discovery, knowledge, health, judgement, aesthetics, recreation, education, community development, lifestyle, identity, and emotional/psychological well-being (Chan et al., 2012; Fish et al., 2016; Marshall et al., 2018). Ecosystems not only make human life possible, they also contribute to making life worth living (Costanza et al., 1997). However, in a recent review, (Fish et al., 2016), suggesting that cultural aspects of climate change are very much under-represented in the literature. This may be because cultural impacts are largely ‘invisible’ Turner et al. (2008) where losses associated with culture are less tangible than those associated with economic losses.

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

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

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

5.3.2.2 Economic Dimension

5.3.2.2.1 Monetary and material wealth generated from fisheries

Marine biodiversity and ecosystem services are supporting the income and livelihood of coastal communities and marine-related economic sectors such as fisheries, aquaculture, forestry and tourism (World Ocean Assessment) (Table 5.6). Marine fisheries and forestry are directly dependent on provisioning service (biomass production of fish stocks and mangroves). Marine aquaculture is partly dependent on capture fisheries production (as feed to supply animal protein and nutrients for the farmed animals) and regulatory service through provision of suitable farming habitat (nutrients, oxygen, current) (Oyinlola et al., (in review)). Marine biodiversity and geomorphology of the coastal areas contribute to marine-related tourisms, such as recreational diving, fishing and whale watching. Thus, climate impacts on biodiversity (Sections 5.2.2, 5.2.3) and ecosystem services (Section 5.3.1) will increase the risk of impacts on the income and livelihood of people working on these economic sectors (*medium confidence*).

Climate change impacts on abundance, distribution and potential catches of fish stocks (see Section 5.3.1) are expected to reduce revenues and economic values of the global fisheries (*medium confidence)* (Lam et al., 2016; Sumaila et al., (in prep)). Currently, global marine fisheries landings are estimated officially at between 80 and 85 million tonnes a year since 1990 (FAO, 2016), with corresponding mean annual gross revenues fluctuating around USD 100 billion annually (Swartz et al., 2013). Accounting for unreported catches, a recent study estimated the likely 'true' annual global catch to be about 130 million tonne (Pauly and Zeller, 2016). Globally, a study revealed that the marine fisheries revenues is projected to be negatively impacted in 89% of the world's fishing countries under RCP8.5 scenario in the 2050s relative to the current status (Lam et al., 2016). The projected changes in the MCP in a country's Exclusive Economic Zones (EEZs) do not directly translate to the change in the fisheries revenues of a country as the catch composition change and vessels of some countries do not only fish in their own EEZs, but also in the high seas and other countries' EEZs. The implications of the impact on fisheries sector under climate changes vary among different regions and countries (*high confidence*) (Hilmi et al., 2015a) with the extent being the greatest in the coastal low-income food deficit countries including small island countries, African countries and tropical Asian countries (Lam et al., 2016). Using geo-spatially referenced malnutrition and infant mortality data, as a proxy for poverty, (Barbier, 2015) finds that just 15 developing countries contain over 90 percent of the world's low-elevation coastal zone rural poor. These least developed countries usually heavily rely on fish and fisheries as a major source of animal proteins, nutritional needs, and income and job opportunities. Also, the availability of the alternative sources of food and livelihoods is limited in these countries. Therefore, negative impacts on the catch and total fisheries revenues obtained by these countries may have greater implications on the jobs, economies, food and nutritional security than the impacts on high Human Development Index (HDI) countries Srinivasan et al. (2010); (Golden et al., 2016). These least developed countries are also highly vulnerable to the impact of climate change but have relatively low capacity to adapt to the change (Allison et al., 2009; Blasiak et al., 2017). (Dasgupta et al., 2017) investigate possible impacts of climate change on coastal communities in Bangladesh using the salinity tolerance range of variety of fish species they consume. The results indicate that areas with poor populations that lose species are about six times more prevalent than areas gaining species.

Climate change may affect the subsequent benefits of the indirect sectors (secondary and ancillary) that related to fisheries, for example, fish canning, processing industries and boat repair. However, these impacts have not been well addressed in recent literature Lacoue-Labarthe et al. (2016). Also, more effort will be needed in understanding the effect of climate change on the dynamics of the fish prices and costs of fishing. Both of these changes would lead to the adjustment of the fishing effort and hence intensify or lessen the overcapacity issue. Studies have attempted to project how fishers may respond to change in fish distribution and abundance by incorporating different management systems (Haynie and Pfeiffer, 2012; Galbraith et al., 2017). But the understandings on the impacts of climate change on the management effectiveness and trade practices (Galbraith et al., 2017) are still not adequate and will need to be further explored.

Amongst different ecosystems, coral reefs are an important asset from an economic point of view. The net benefits accrued by the global economy of coral reefs reach US$30 billion yr^{-1} from food security, coastal protection against natural hazards, tourism, financial incomes, sources for bio-prospecting, the development of novel pharmaceuticals and other types of goods and services (Cesar et al., 2003), 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 the poor rural communities' income, worldwide (Ring et al., 2010).

5.3.2.1.2 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. In developing countries fish is often the only affordable and relatively easily available source of animal protein (FAO, 2017).

Climate change impacts on marine ecosystem services will lead to high risk of impacts on nutritional health of dependent communities with high seafood-dependence, particularly for Indigenous and other coastal communities in the global south, through decreases in seafood availability (Hilmi et al., 2014) (*low confidence*, Table 5.6, Figure 5.22). Marine ecosystems, through fisheries and aquaculture, supports the global population with proteins and vital micronutrients, such as iron, zinc, omega-3 fatty acids and vitamins (Golden et al., 2016). Particularly, fish are important sources of micronutrients, especially for some populations that are largely dependent on fishery for subsistence and livelihood. Deficiencies of some of the micronutrients can increase the risk of perinatal and maternal mortality, contribute to growth retardation, child mortality and cognitive impairments (Black et al., 2013). Climate change is projected to reduce fisheries catch potential by 2050s relative to 2000s particularly for regions where dependence on micronutrients from fish is high (*medium confidence,* Section 5.3.1). The projected decrease in fish supply is expected to exacerbate health risks over the next decades from reduction in intake of micronutrients and fatty acids. Particularly, low- and middle- income coastal communities around the equator where large decrease in potential fisheries catches are projected (>20% by 2050s relative to the 2000s under RCP8.5) are at higher risk of such impacts on nutritional health (Golden et al., 2016) (*medium confidence*, Figure 5.22). It has been estimated that 1.39 billion people are vulnerable to deficiencies of micronutrients if fish catches continue to decline (Golden et al., 2016).

Decreases in fish supply under climate change will also contribute to decrease in protein supply to these coastal communities in general (Lam et al., 2012). Area with strong fish dependence and high risk of impacts on fisheries production from climate change coincide with areas where land-based food production is projected to decrease under climate change (Blanchard et al., 2017), further exacerbating the risk of climate change impacts on nutritional health of people living in these areas. (Hilmi et al., 2017) conclude that climate change and ocean acidification will impact fisheries and aquaculture in the Mediterranean countries, but the impacts are difficult to be evaluated because of lack of data from natural sciences Hilmi et al. (2013a).

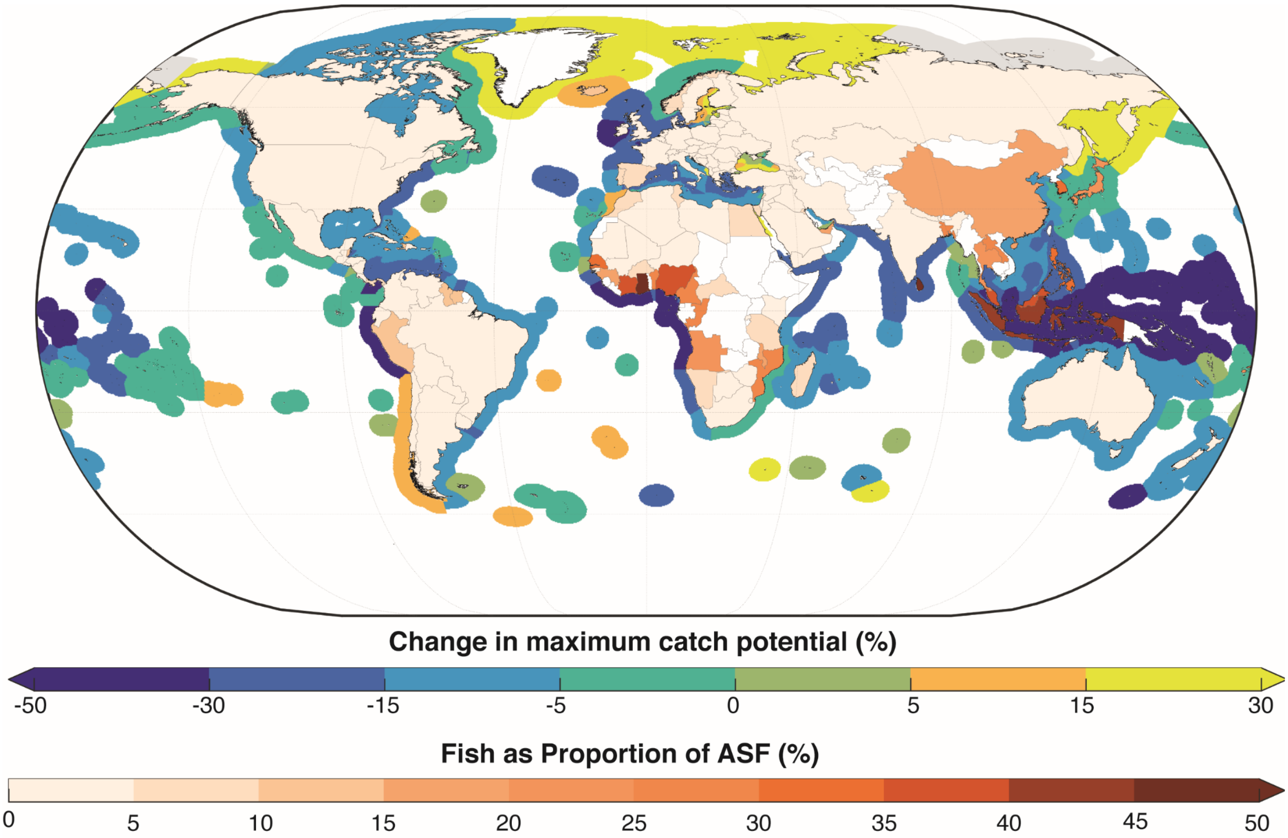


Figure 5.22: Projected changes in catch potential and the reliant of the human population on fish and their vulnerability of micronutrient malnutrition (Golden et al., 2016). The colour scale on land represent the proportion of fish micronutrient intake relative to the total animal sourced food (ASF) for the country's population, the scale on sea represent projected changes in maximum catch potential under RCP8.5 by 2100 relative to the 2000s period.

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

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

Climate change impacts on coral reefs and other coastal ecosystems (Section 5.2.1) may affect the quality of the dive sites, beaches and the coast (Hilmi et al., 2012) (*high confidence*). Coral reefs and coastal ecosystems draw significant tourism each year, supporting recreational activities, such as diving and snorkeling, and supporting coastal communities and auxiliary sectors (Spalding et al., 2017). Reef tourism's popularity is increasing, with some parts of the world showing increases in visitors' numbers in recent years growing the pressure to develop tourist facilities and accommodation. Beach recreation values are also expected to decline in response to narrower beaches resulting from SLR and erosion, with considerable losses for local economies (e.g., $1 billion loss between 2006 and 2080 in North Carolina (Whitehead et al., 2009; Nelson et al., 2013)). As principal assets of coastal tourism, the loss of beaches could significantly impact local economies in subtropical and topical regions, and could incur additional costs associated with coastal protection and infrastructure repairs (DiSegni and Shechter, 2013). The impacts are exacerbated in

SIDS because of the increasing demography which carries an increasing pressure on the reef resources mainly used for self-consumption. This trend is exacerbated by the disappearance of traditional ways of management of the reef resources (such as taboos and religious beliefs demonstrated as informal no-take zones even temporary) which were quite effective.

Recent studies provide additional evidence that species range shifts (Sections 5.2, 5.3) are likely to alter tourism flows. For instance, poleward range shifts in the United States could yield new opportunities for recreational fishing in temperate regions, while warmer conditions could benefit some species purposively introduced for recreational fishing (DiSegni and Shechter, 2013). Conversely, increases in the prevalence of tropical box jellyfish through climate induced poleward range shifts could detrimentally affect coastal tourism and water-related activities in subtropical and temperate regions (Klein et al., 2014). Increased prevalence and transmission of diseases are also likely to occur under warmer ocean temperatures, with empirical evidence of poleward shifts in disease outbreaks under warmer conditions (Burge et al., 2014). Importantly, adverse publicity accompanying an increased risk of disease transmission in tropical and subtropical regions could further have deleterious impacts on coastal tourism (DiSegni and Shechter, 2013). If climate change and ocean acidification reduce the seafood supply (Wabnitz et al., 2017), the attractiveness for tourists will also decrease (*low confidence*). This will have an economic impact on jobs, revenue and income, and thus on the GDP (growth domestic product) of the countries. This is more deleterious for small countries relying on tourism to have foreign currency and be able to pay their imports of subsistence products. Visitor activity includes swimming, boating, snorkeling and scuba diving (Newsome et al., 2002).

5.3.2.3 Environmental Dimension

5.3.2.3.1 Aesthetic aspects/property values/ development of coastal cities

The aesthetic appreciation of natural places is one of the most fundamental ways in which people relate to their environment, however, climate change threatens these aspects through degrading aesthetic elements of a climate sensitive natural resource (*low confidence*). Positive externalities generated from ecosystems, or cultural ecosystem services, are many and they encompass aesthetic stimulus or aesthetic appreciation. Aesthetically beautiful places are critically important for supporting local and international economies (especially through tourism), human wellbeing, and stewardship (Marshall et al., 2018). (Marshall et al., 2018) assessed the importance of aesthetic values within the Great Barrier Reef (GBR) by indigenous and non-indigenous local residents, Australians (non-local), international and domestic tourists, tourism operators, and commercial fishers and found that aesthetic values are a critically important cultural value for all cultural groups, and important for maintaining sense of place, pride, identity and opportunities for inspiration, spirituality, recreation and wellbeing. However, climate change has and will destroy many of the features of regional aesthetical ecosystem services.

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

One cannot avoid the issue that climate change will alter or decimate coastal and inland real estate and land values across many regions in a variety of ways and degrees (*high confidence*). As regional information accumulates on the occurrence and frequency of extreme events such as flood risks, drought duration and intensity, fires, etc., data will be incorporated into risk assessments and asset pricing. Decreases in the real estate values, lack of rental incomes from jeopardized properties are going to trigger a chain of events which will be unavoidable. Loss in household asset values are crucial since homes or farmland tend to be the largest form of savings for most households in many regions. Spending patterns and amounts will be affected where lower long-term spending will lead to decreased economic business activity and may lead to closure of many small business firms, further leading into increased local unemployment. If house or land values decrease below outstanding mortgages, defaults on debt payments may lead to regional bank crisis.

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

5.3.2.3.2 Energy and industrial activities on the ocean

Energy activities in the oceans give rise to significant changes in the marine environment as well as to climate change at a global level if the energy is derived from fossil fuels (*high confidence*). Primarily, petroleum, gas, and coal have significant external costs in the form of emissions of greenhouse gases, marine pollution, ocean acidification, and associated impacts on the marine environment and marine species' health.

Cordes et al. (2016) synthesize current information on typical effects of offshore oil and gas operations ranging from impacts on species, populations, assemblages, to ecosystems by modifying a variety of ecological parameters (e.g., biodiversity, biomass, productivity, etc.). They note that drilling for oil and gas has become routine in all offshore environments, with major deep-water (>200 m) production in regions including the Arctic, northern North Atlantic Ocean (UK and Norwegian waters), East and West Africa, Gulf of Mexico, South America, India, Southeast Asia, and Australia. The authors point to the potential problems that emerge at different phases of oil and gas activities. For instance, "during the exploration phase, impacts can result from indirect (sound and traffic) and direct physical (anchor chains, drill cuttings, and drilling fluids) disturbance". In the production phase, additional direct physical impacts occur "as pipelines are laid and the volume of discharged produced water increases". In the decommissioning phase, a series of direct impacts are caused on the sea floor and can re-introduce contaminants to the environment Cordes et al. (2016).

In regards to global climate change, climate change has a global impact but the Arctic and Antarctic regions are particularly sensitive. This region will continue to generate monetary incomes even if ice melts.

The Arctic region, which represents only 6% of the world's surface, would contain many resources (*medium confidence*). According to some studies that have been done on fossil fuels [reference needed], there would be 29% of gas reserves in the basements and 10% of oil reserves, as well as coal. Oil and gas in the Arctic region is estimated to account for 1/10 of world oil and ¼ of global gas. Russia would have about 80% of these reserves and the remaining percentage would be divided between Arctic Canada, Alaska and Norway. The development of the offshore exploitation is a real challenge in this region because of the climate and the cost of these operations, but it will also depend on the state of advancement of the technologies (*low* *confidence*). Different studies have different results regarding the future of the Arctic as a producer of gas and oil. In addition, there are many mineral resources in this region, whether metallic or non-metallic. There are also biological and fishery resources. Indeed, a large part of the marine subsoil is not exploited for fishing and would contain many resources (fish stock), especially the central part which is not under any jurisdiction.

In addition, the extraction of these resources will also lead to a sharp increase in maritime traffic (*low confidence*). Melting ice has opened up new waterways such as the Northwest Passage and the Northeast Passage. The opening of these passages can result in considerable time savings on journeys, but their crossing remains 'extreme' and poses problems with regard to environmental safety. Indeed, these new shipping routes could also allow the convoy of goods by large cargo ships. As long as these passages remain few used, the risks of pollution are rather low. But the potential presence of drifting icebergs could increase the risk of accidents. In the same way, discharges at sea, even if they are practiced miles of coast, will still damage the surrounding biodiversity (*low confidence*).

In the case of Antarctica, resources are certainly available. First of all, with regard to mineral resources, ore deposits have been found. Secondly, in the case of fossil fuels, coal has been found in two areas of Antarctica, but in too small a quantity, making it economically unsustainable for the moment (*low confidence*). For oil and gas, some rocks would contain some, but no drilling was done to prove it. Even if it were proven that these resources are there, the exploitation of oil, for example, would be too expensive. Indeed, the barrel would be 100 US dollars, while it is actually 58 US dollars. In the same way, its exploitation would be too expensive because of the extreme conditions and the current state of knowledge. In any case, the issue of resource exploitation in Antarctica is not relevant. Indeed, the Antarctic Treaty prohibits the exploitation of natural resources. Thus, scientific exclusivity has the corollary prohibition of all military activities, as well as the prohibition of prospecting and exploitation of natural resources, fossil or mineral. This prohibition is the result of subsequent negotiations of the Treaty, which led to the Madrid Protocol of 1991. Thus, Article 7 of this Protocol therefore prohibits all activities, relating to mineral resources, which are not scientific.

As long as we do not switch to renewable energy, the risks associated with hydropower will persist. Marine energy is still relatively low compared to other renewable energy and the technology is risky and expensive.

5.3.2.4 Climate Change Stressors, Ecosystem Services, and Sustainable Development Goals

Climate change impacts to ecosystem services will have consequences for our ability to achieve the internationally agreed upon the 17 Sustainable Development Goals (SDGs) (*medium confidence*). Failing to avoid or mitigate climate change impacts will already fail to achieve the sustainable development goal related to climate action (SDG 12). Similarly, failing to avoid or eliminate climate change impacts likely means that not enough has been done to achieve clean energy (SDG 7). One way to mitigate climate change is energy transition. Climate change will have direct consequences for the goals that consider sustainable oceans (SDGs 14) because of impacts to raw materials, species, habitats, and regulating processes (Singh et al., 2017). Indirectly, these climate change impacts are likely to have consequences for economic and social goals that depend on natural resources across the globe such as building sustainable economies, eliminating poverty, eliminating hunger, achieving food security (SDG2) reducing inequalities (some countries' economies are more tied to natural resources and climate change impacts are projected to be more pronounced in the poles and the global south than the global north), and achieving responsible consumption and production (SDGs 8, 1, 2, 10, and 12, respectively) (Castells-Quintana et al., 2017; Tol, 2018). Climate change is also creating living conditions that are less suitable to humans through heat stress and changing distributions of disease vector (5.3.2.1.1), reducing our chances of achieving the goal for good health and wellbeing (SDG 3) (Carvalho et al., 2017; Wouters et al., 2017). Impacts to living conditions as well as changing recreational, aesthetic, and spiritual experiences also affects our ability to achieve sustainable cities and communities (SDG 11).

Less certain is the consequence to the remaining SDGs. However, globally the role of women in primary sector, such as fisheries, is different than men and are their work is more often unrecognized, so climate impacts may disproportionately affect women in these roles if governments do not protect their jobs as they might for men (whose work is more likely officially recognized through established markets) – so in this way climate change may negatively affect our ability to achieve gender equality (SDG 5) (Pearse, 2017). The negative effects on raw materials, food, water, and other primary resources may make it harder for nations to peacefully cooperate or establish strong multilateral institutions, reducing the likelihood of achieving peaceful societies and fostering international cooperation (SDGs 16 and 17, respectively) (Salehyan, 2008). A reduction in cooperation may also negatively affect our abilities for innovation and education (SDGs 9 and 4), however, climate change impacts may also mean that innovation is necessary, leading to leaps in invention and adaptation.

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 (SDG2) (*medium confidence*), achieving good health and wellbeing (SDG 2) (*low confidence*) as well as all other SDGs indirectly (*low confidence*).

5.4 Risk-reduction Responses and their Governance

5.4.1 Ocean-based Mitigation

The IPCC definition of climate mitigation covers both the reduction of greenhouse gas emissions and the active removal of greenhouse gases from the atmosphere. Article 4.1 of the Paris Agreement recognizes the need for both actions in its requirement for net zero emissions of greenhouse gases in the second half of this century. Indeed, that balance between emissions and removals is how the climate system can be stabilized, limiting warming to ‘well under 2.0˚C’ (Rogelj et al., 2015; Geden, 2016). To reduce the scale of the physical changes to the ocean and their environmental impacts (Sections 5.2, 5.3), transformative and near-total emission reductions are therefore needed in all sectors, not just energy, and for all greenhouse gases, not just CO_2 (UNEP, 2017).

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. Therefore, ocean-based mitigation through renewable energy is not considered here, nor are ocean-based albedo enhancement methods, also known as solar radiation management or sunlight reflection methods (SRM).

Instead the focus is on the effectiveness, risk trade-offs and societal feasibility of two other ocean-based mitigation approaches. First, the scope to decrease anthropogenically-driven releases of greenhouse gases from the marine environment, primarily by maintaining the integrity of natural carbon stores in the coastal zone. Second, the potential to promote additional carbon uptake by marine ecosystems, both in the coastal zone and the open ocean, as a contribution to the additional gigaton-scale uptake (negative emissions) of CO_2 that is needed in most emission pathways that limit global warming to 1.5-2.0˚C (Fuss et al., 2014; Anderson and Peters, 2016). These ocean based removals of CO_2 from the atmosphere will be needed as an addition to, not as a substitute for, very rapid reductions in greenhouse gas emissions (UNEP, 2017). Under most international definitions, including those of IPCC, such negative emissions are considered to be a form of climate geoengineering, as well as part of climate mitigation (Williamson and Bodle, 2016).

The term ‘blue carbon’ refers to both these two marine mitigation approaches: reducing emissions and enhancing sinks. This term is mostly used in a coastal context, to cover carbon-rich habitats structured by rooted plants, such as mangroves, sea grasses and saltmarsh vegetation (McLeod et al., 2011; Pendleton et al., 2012a; Thomas, 2014; Macreadie et al., 2017). However, the term has also been used more generally, covering all biogenic marine carbon (Nellemann and Corcoran, 2009), not only in both coastal and deep ocean sediments, but also in all marine biota, and in the form of refractory dissolved organic carbon (RDOC) (Jiao et al., 2010). Figure 5.23 shows the spatial distribution of the coastal and open ocean mitigation methods considered here, in the context of marine legal zoning relevant to carbon accounting and governance.

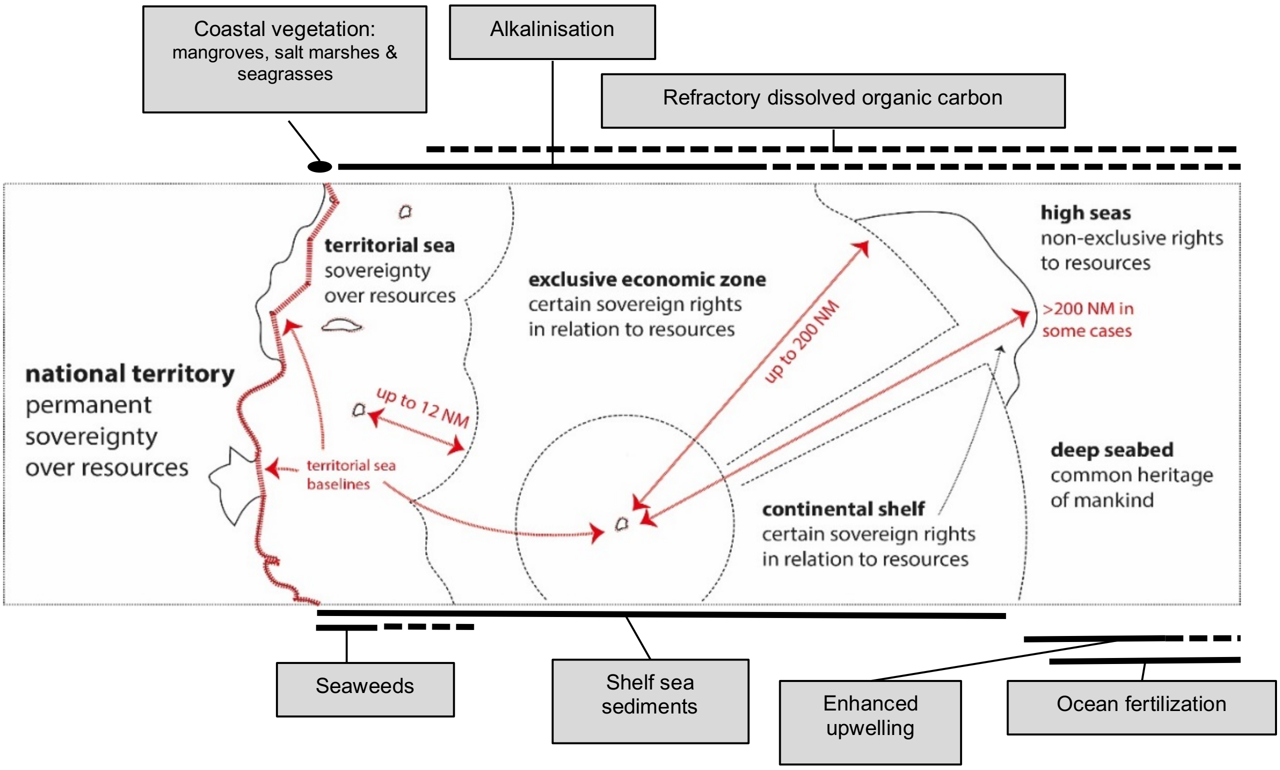


Figure 5.23: Marine legal zoning relevant to ocean-based mitigation of CO_2 (excluding renewable energy).

The scope for climatically-significant, effective and societally-acceptable blue carbon 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 will be considered 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. Eight such criteria were assessed by (Gattuso et al., 2018). These criteria are based on quantitative and semi-quantitative expert scoring of the following factors: maximum potential effectiveness in reducing climatic drivers (ocean warming, ocean acidification and sea level rise) by 2100, assuming full theoretical implementation; technological readiness and lead time to full potential effectiveness (subsequently combined as technical feasibility); duration of benefits; co-benefits; dis-benefits; cost-effectiveness; and governability. These criteria were considered at either global or local scales, or both.

Effectiveness scores by Gattuso et al. (2018) for 11 ocean-based responses that involve ocean processes and ecosystems; i.e., not considered societal adaptation, are summarized in Table 5.7. Those grouped as 'global' can be regarded primarily as mitigation (reducing drivers), those as 'local', as ecosystem-based adaptation (reducing impacts); two were considered at both scales. Two ocean-based albedo enhancement techniques are excluded here, since such approaches are outside the scope of the current report. However, physically-based marine renewables are included, for comparative purposes with the other approaches across all of the criteria.

Table 5.7: Summary of ocean-based risk-reduction options, based on Gattuso et al. (2018), excluding solar radiation management and societal adaptation responses. Maximum mitigation effectiveness is a combined score for moderating ocean warming, ocean acidification and sea level rise. Major advantages and disadvantages correspond to scores of 4-5 (high/very high) or 1-2 (very low/low) respectively according to the semi-quantitative assessment criteria used.

	Option	Key features of approach	Maximum mitigation effectiveness	Major advantages (based on scoring for other criteria)	Major disadvantages (based on scoring for other criteria)
Global	Marine renewable energy (physical processes)	Multiple energy sources: wind, tides, currents, waves, thermal and salinity gradients	High/very high	High/very high governability; high cost-effectiveness (for more mature technologies: wind, tidal and wave)	???
	Ocean fertilization (open ocean)	Iron additions to increase phyto-plankton production and C sequestration	Very low	Not assessed	Very low governability
	Land-ocean hybrid methods	Land to marine transfers of biomass, or vice versa	Moderate	Not assessed	Very low technical feasibility
Global & local	Alkalinization (enhanced weathering)	Addition of substances (e.g., carbonates) to consume CO2 and/or neutralize acidity	Very high (global); low (local)	Not assessed	Very low governability; low/very low technical feasibility
	Restoration and conservation of coastal vegetation	Halting/reversing loss of mangroves, salt marshes and seagrass habitats; restoring coral reefs	Very low (global) – moderate (local)	High ratio of co-benefits to dis-benefits; conservation has high cost-effectiveness	Impacts on biodiversity; changing the nature of the 'reef of tomorrow'
Local	Marine Protected Areas	Conservation of habitats and ecosystems	Low	High ratio of co-benefits to dis-benefits	Limited successful cases, very low effectiveness
	Reducing pollution (incl. nutrients)	Decreased release of anthropogenic, harmful substances	Low	Moderate to very high ratio of co-benefits to dis-benefits	Not assessed
	Restoring hydrological regimes	Maintaining/restoring delivery of water and sediment to coastal regions	Low	High/very high technical feasibility	Not assessed
	Eliminating over-exploitation	Sustainable harvesting of marine bioresources; avoidance of irreversible ecological impacts	Low/very low	High/very high ratio of co-benefits to dis-benefits	Not assessed
	Assisted evolution	Genetic modification, captive breeding and release of tolerant organisms	Very low	Not assessed	Low/very low technical feasibility
	Relocation and reef restoration, ecological engineering approaches	Relates to coral and oyster reefs, with creation of potential new habitats	High/Low/very low (depending on approaches used)	Very high governability; high ratio of co-benefits to dis-benefits; conservation has high cost-effectiveness	Impacts on local biodiversity

5.4.1.1 Climate Mitigation in the Coastal Ocean

5.4.1.1.1 Coastal carbon cycle: opportunities and challenges

The coastal ocean connects the mountain cryosphere (and other terrestrial systems) with the open ocean, through estuaries, shelf seas and a wide range of intertidal and shallow-water habitats. These systems play a highly dynamic and important role in the global carbon cycle, not only through in situ primary production, by macrophytes, macroalgae and phytoplankton, but also by processing riverine organic carbon Dai et al. (2012) (Section 5.3.1.1). Coastal carbon flows and storage can be strongly affected by human activities, indirectly in river catchments (Jiao et al., 2011; Regnier et al., 2013) and directly by coastal land-use changes and marine resource exploitation in shelf seas. Despite intensive study, significant uncertainties remain (Cai, 2010; Bauer et al., 2013), with many knowledge gaps arising from the complexity of bio-physical interactions and their local and regional spatial heterogeneity.

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

5.4.1.1.2 Blue carbon

The global importance of three kinds of coastal vegetation has recently been recognised: mangroves, salt marshes and seagrasses (Chmura et al., 2003; Duarte et al., 2005; Kennedy et al., 2010; McLeod et al., 2011). Although covering only ~0.1% of the Earth's surface, these three systems together support 1–10% of global marine primary production, and contribute 3–30% of oceanic CO_2 uptake (Duarte et al., 2017). Some 151 countries around the world contain at least one blue carbon ecosystem (seagrass, saltmarshes or mangroves) and 71 countries contain all three (Herr and Landis, 2016).

The wide (order of magnitude) range of such estimates primarily relates to uncertainties in the poorly-mapped areal coverage of salt marshes and seagrass beds, determined either empirically, or on theoretical grounds of substrate and light availability. The global area of mangrove habitats is better known, estimated at 0.14×10^6 km^2 from satellite imagery (Giri et al., 2011). Mangroves are particularly effective in retaining carbon: their associated sediments and above- and below-ground woody structures 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). However, there can be large differences in the amount of carbon stored by mangroves (Adame et al., 2017; Schile et al., 2017), also by seagrasses (Lavery et al., 2013) and in salt marshes (Kelleway et al., 2017b). The sequestered carbon may have accumulated over several thousand years, providing a natural carbon sink (McKee et al., 2007; Lo Iacono et al., 2008).

In addition, coastal wetlands provide adaptation benefits by absorbing incoming wave energy, providing hurricane and storm surge protection, and preventing erosion (Costanza et al., 2008; Spalding et al., 2014). Coastal ecosystems may keep pace with sea level rise and, in some instances, are more cost-effective than hard civil infrastructure like seawalls (*medium confidence*) (Temmerman et al., 2013). Healthy coastal ecosystems provide other benefits, including fishery habitat, control of water quality, and local livelihoods (Section 5.3.1.1). Thus, blue carbon protection has multiple co-benefits through supporting biodiversity and a wide range of ecosystem services (*high confidence*).

When degraded, these co-benefits are greatly diminished along with the ecosystems' capacity to sequester carbon, and stored carbon can be released back to the atmosphere, along with other greenhouse gases (Marba and Duarte, 2009; Duarte et al., 2010; Pendleton et al., 2012b; Lovelock et al., 2017). Around 25–50% of mangrove forests and other vegetated coastal habitats have already been lost or degraded due to coastal developments and other land-use change during the past 100 years (McLeod et al., 2011). Current losses are estimated at 0.2–3.0% yr^{-1}, depending on vegetation type and location (Hiraishi et al., 2014; Alongi and Mukhopadhyay, 2015; Atwood et al., 2017). Associated global carbon emissions due to these habitat losses have been estimated at 0.04–0.28 GtC yr^{-1} (Pendleton et al. 2012); 0.06–0.61 GtC yr^{-1} (Howard et al., 2017) and 0.1–1.46 GtC yr^{-1} (Lovelock et al., 2017). The highest of these estimates exceeds the most recent global estimate of carbon emissions from all land-use change: 1.3 ± 0.7 GtC yr^{-1} (Le Quéré et al., 2018). However, (Lovelock et al., 2017) note that several processes could reduce their maximum value. These factors include organic matter remaining in anoxic conditions after vegetation is removed; interactions between CO_2 and carbonates in sediments; and the offshore transport and reburial of organic carbon.

Awareness of the scale of potential CO_2 emissions from the continued loss of coastal vegetated habitats (together with increased appreciation of other ecosystem services that they provide) has resulted in the protection and restoration of coastal blue carbon ecosystems being recognized as a national priority for both climate change mitigation and adaptation Martin et al. (2016); (Gallo et al., 2017). In their National Determined Contribution (NDC) to 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 5.24).

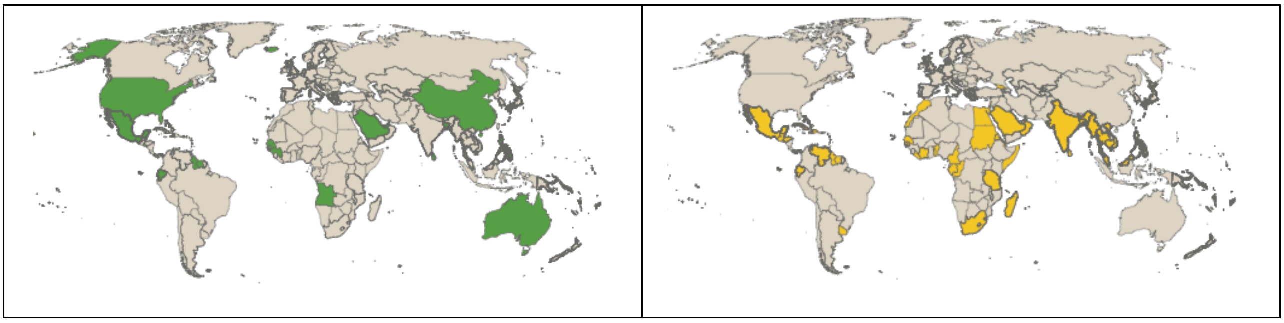


Figure 5.24: Left panel: Map of 28 countries that include a reference to coastal wetlands in terms of mitigation in their NDCs. Right panel: Map of 59 countries that include coastal ecosystems and the coastal zone in adaptation strategies in their NDCs (Martin et al., 2016).

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

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

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

- *Specific measures to increase carbon sequestration.* (Zhao et al., 2016a) identifies three catchment-level management measures to increase carbon uptake and storage for coastal ecosystems: reducing anthropogenic nutrient inputs; controlling bioturbators to prevent carbon loss (e.g., by reinstating predators; Atwood et al. (2015)); and restoring hydrology, by removing barriers to tidal flow.

Successful large-scale implementation of the above measures would assist in global climate mitigation, and hence in reaching the net zero emissions target of the Paris Agreement (*high confidence*). For some countries, e.g., Indonesia and Malaysia, 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 CO_2 release from carbon-rich sediments may continue for 10–40 years after vegetation loss, with the rate of that release depending on local environmental factors (Lovelock et al., 2017).

- Emissions of methane and nitrous oxide, from both intact and degraded blue carbon ecosystems, can significantly alter the scale of climatic benefits (Adams et al., 2012; Chen and Ganapin, 2016; Chmura et al., 2016).

- Carbonate formation in some blue carbon systems may release CO_2, offsetting the benefits of carbon storage (Howard et al., 2017; Macreadie et al., 2017).

- The permanence of vegetated coastal systems, even if well-protected, cannot be assumed under future temperature regimes (Ward et al., 2015; Duke et al., 2017; Jennerjahn et al., 2017; Nowicki et al., 2017) and sea level rise (Kirwan and Megonigal, 2013; Spencer et al., 2016). However, carbon sequestration capacity may not necessarily decrease; e.g., if mangroves replace salt marshes (Kelleway et al., 2016), or where totally new habitats might be created (Barnes, 2017).

There are undoubtedly opportunities to include and expand blue carbon ecosystems in the revised NDCs of most, if not all, coastal countries. However, the above complexities currently make it difficult to provide accurate estimates of the climate mitigation achieved through blue carbon management measures (*high confidence*). Furthermore, even if all blue carbon ecosystems could be restored to their 1990 extent (*unlikely*, because of the semi-permanent nature of most coastal land-use change, such as human settlement and conversion to agriculture). However, even if all blue carbon ecosystems could be restored to their 1990 extent (*unlikely*, because of the nature of much coastal development) their contribution to the 'emissions gap' (UNEP, 2017) would be relatively modest at the global scale (*high confidence*). (Griscom et al., 2017) estimated that global mitigation benefits of cost-effective coastal wetland restoration could be ~0.05 GtC yr^{-1} by 2030, compared to current net emissions from all sources of 10.0 GtC yr^{-1} Le Quéré et al. (2018). (Gattuso et al., 2018) similarly concluded that coastal restoration and protection only offered a very limited (and relatively unreliable) 'solution' to climate change. Nevertheless, measures to protect and, where practicable, re-instate such habitats provide many other societal benefits through ecosystem services; in particular, benefitting local fisheries, enhancing biodiversity, giving storm protection and reducing coastal erosion. Therefore, coastal blue carbon can be considered as a 'no regrets' option, in addition to other, higher magnitude, mitigation measures. Additional process studies are needed over the full range of environmental conditions to improve the *low confidence* in current estimates of carbon loss from degraded vegetated coastal ecosystems, covering more than an order of magnitude range.

5.4.1.1.3 Seaweeds, shelf sea sediments and dissolved organic carbon

Seaweeds (macroalgae) are not as effective in carbon storage as rooted coastal vegetation (macrophytes, as considered above) Howard et al. (2017). Whilst seaweed detritus can deliver carbon to sedimentary sites (Hill et al., 2015) and provide a source of refractory dissolved organic carbon (RDOC) (Hughes et al., 2012), the global importance of these processes seems unlikely to be high (*medium confidence*). Nevertheless, seaweed aquaculture may offer significant mitigation opportunities (N'Yeurt et al., 2012; Chung et al., 2017; Duarte et al., 2017). In particular, for biofuel/biogas production (N'Yeurt and Iese, 2014; Moreira and Pires, 2016; Sondak et al., 2017) that could be linked to carbon capture and storage (Hughes et al., 2012).

The viability of scaling-up these approaches to climatically-significant levels has yet to be established: (N'Yeurt et al., 2012) envisage 9% of the global ocean used for seaweed aquaculture, but they did not assess the associated environmental and socio-economic implications. A different mitigation option proposes using seaweeds as a dietary supplement for ruminants, to suppress their methane production. In vitro studies have given promising results Dubois et al. (2013); (Machado et al., 2016); the need is to quantify the potential scale of real-world benefits. Until that has been done, there is only *low confidence* in this approach having significant mitigation potential.

Regional carbon sequestration rates have been estimated (Hu et al., 2015), with quantification of the factors affecting organic carbon storage in the potentially-mobile fraction (top 10 cm) (Diesing et al., 2017). Such information is relevant for mitigation since there is scope for managing sediment disturbance by human activities, such as fishing, dredging and the installation of offshore structures that affect carbon storage and cycling, both directly (physically) and indirectly (biologically, through effects on benthic organisms) (Hale et al., 2017). Carbon retention, over relatively large areas of shelf seas, might therefore be maximised (Luisetti et al., 2018).

Land-based measures to nutrient fluxes to shelf seas might also assist with mitigation (*low confidence*). Measurements of air-sea CO_2 fluxes (Laruelle et al., 2015) and modelling studies (Bauer et al., 2013) both indicate that most shelf seas are currently a net carbon sink, with anthropogenic nutrients implicated through their role in stimulating shelf primary production, hence CO_2 drawdown (Bauer et al., 2013; Regnier et al., 2013). However, in eutrophic waters, the fixed carbon is more rapidly re-mineralised by microbially-driven decomposition, returning it to the atmosphere (Jiao et al., 2011). If land-based nutrient inputs can be reduced, a higher proportion of refractory DOC may be produced, potentially removing carbon from circulation for decadal to millennial timescales (Jiao et al., 2014b) (*medium confidence*). The global scale of achievable climatic benefits via such approaches has yet to be determined, and its national accounting would not be straightforward. In practice, such approaches can be applied to coastal waters within national jurisdictions.

5.4.1.1.4 Alkalinization of coastal seas and other non-biological CO_2 removal methods

There is theoretical potential for substantive enhancement of coastal (and open ocean) geochemical processes that remove carbon from the atmosphere (*high confidence*) (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 CO_2.

Such ocean alkanization methods might be used to reduce local impacts, e.g., for coral reefs (Feng et al., 2016), as well as contributing to wider counteraction of climate change. Rock-weathering with associated permanent CO_2 uptake is a natural process, although occurring relatively slowly: it currently sequesters ~0.25 GtC yr^{-1}, on land and at sea (Taylor et al., 2015). The main constraints on the development and deployment of ocean alkalinization as a mitigation method relate to the governance, cost, and uncertain environmental impacts (on land as well as at sea) of climatically-significant application (Gattuso et al., 2018).

CO_2 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 CO_2 would need to be safely stored through carbon capture and storage (CCS) technologies. Detailed assessment of the feasibility of electrochemical and geochemical methods for ocean-based mitigation is not in scope for this Special Report.

5.4.1.1.5 Land-sea integrated eco-engineering

Chemical fertilizers have been excessively applied in farming for decades especially in developing countries. Excessive N and P are then washed out into rivers and ultimately discharged into the coastal waters, causing eutrophication and algal bloom (both red-and green tide) (Zhang et al., 2018) (Figure 5.25). Although algal blooms seemingly produce more organic carbon, this carbon is basically labile and can be respired rapidly. In addition, the labile DOC produced by autochthonous primary producers has priming effects on the river discharged terrestrial RDOC, i.e., remobilize RDOC for microbial uptake and respiration (Liu et al., 2014), which can create high CO_2 concentration causing acidification in ambient water, and excess CO_2 can escape from water to atmosphere as outgassing. That is why productive estuarine and coastal waters are often sources rather than sinks of atmospheric CO_2 (Jiao et al., 2018a; Jiao et al., 2018b). Meanwhile this process consumes large quantity of oxygen resulting in hypoxia. Anoxic conditions could cause massive death of macro- and micro- biomes, resulting in the breeding of anaerobic bacteria that transform organics into CH_4, H_2S, NH_3 and other toxic substances, which in turn are destructive for the ecosystem (Jiao et al., 2018a; Jiao et al., 2018b). On top of that, excess discharge of nutrients (nitrogen, N and phosphorus, P) shapes the C/N and C/P elemental ratios in favour of remobilization of RDOC for respiration, lowering the microbial carbon pump (MCP) efficiency and carbon sequestration (Jiao et al., 2010).

Therefore, reducing terrestrial input of inorganic nutrients could be a countermeasure for the enhancement of carbon sequestration in coastal waters (Jiao et al., 2011). This idea is verified by a statistical data analysis of relationship between organic carbon and nitrate in various natural environments (Taylor and Townsend, 2010) as well as by experimental results in estuarine and offshore waters (Yuan et al., 2010; Jiao et al., 2011; Liu et al., 2014). Therefore, land-ocean integrated management and engineering is necessary, and it is feasible to reduce the application of chemical fertilizers in farming and eliminate sewage discharge into the river so as to reduce the N, P inputs into the sea. Such eco-engineering is not aimed at changing the natural ecosystems, but rather, protecting them by reducing eutrophication and occurrence of red-tides while increasing carbon sequestration through the MCP. This idea also brings new policy such as reward-based carbon trading policy to replace the penalty-based pollution policy that is currently being in practice in many countries (Jiao et al., 2018b).

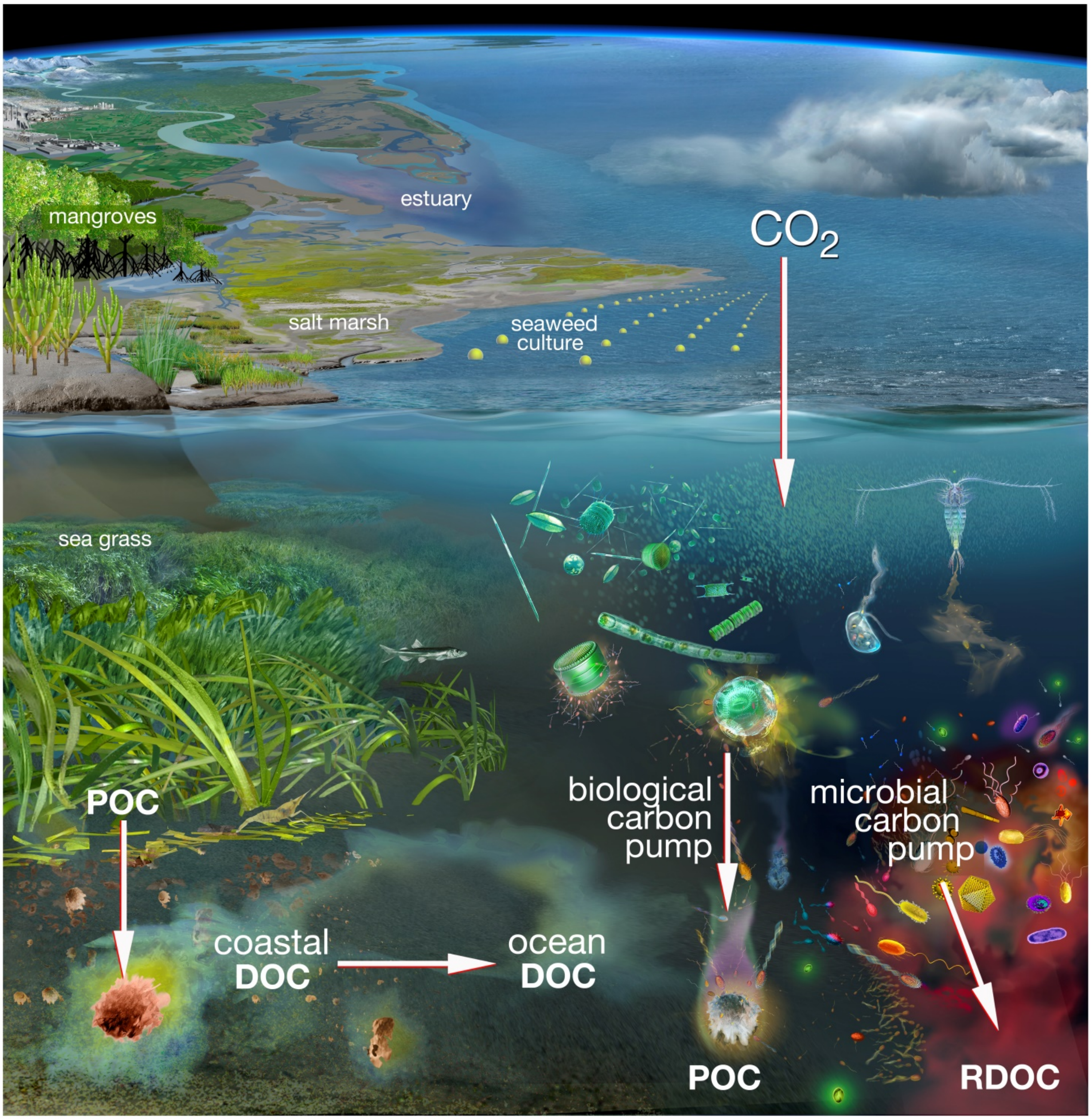


Figure 5.25: Illustration of blue carbon for climate mitigation, including coastal blue carbon (mangroves, saltmarshes, seagrasses) and ocean blue carbon mechanisms (POC based biological carbon pump, and RDOC based microbial carbon pump) (Jiao et al., 2018b).

5.4.1.2 Climate Mitigation in the Open Ocean

5.4.1.2.1 Open ocean carbon cycle: opportunities and challenges

Current net primary production by marine phytoplankton is estimated to be 58 ± 7 GtC yr^{-1}, around half of global primary production (Buitenhuis et al., 2013). However, only ~1% of that production is removed annually from circulation by the combined effects of the three biologically-driven ocean carbon pumps (soft-tissue pump, carbonate pump and microbial pump; (Legendre et al., 2015)). It is the physically-driven solubility pump that is primarily responsible for the global net removal of 2.6 GtC yr^{-1} from the atmosphere in the decade 2006–2015 (~25% of anthropogenic CO_2 emissions; Le Quere et al. (2016)). The residence time of the dissolved inorganic carbon (DIC) by the solubility pump depends on ocean circulation, that of the refractory dissolved organic carbon (RDOC) is about 5000 years, while the buried organic carbon could be gone forever.

Two main methods have been proposed to increase CO_2 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 CO_2, 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 except for seaweeds culture systems (see Section 5.4.2.2.2).

5.4.1.2.2 Ocean fertilization

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

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

5.4.2 Ocean-based Adaptation

The IPCC Fifth Assessment Report (AR5) concluded, with *high agreement* but *limited evidence*, that the climate change impacts of on human settlements and communities could be reduced through coastal protection activities (Wong et al., 2014a). Although there was very high confidence in the adaptation practices being adopted, there was limited evidence of the context-specific application of principles or approaches, and this was recognised as a key research gap. Furthermore, it was stated that besides the evidence that was presented, adaptation activities remained aspirational, and inappropriate coastal development and ecosystem degradation continued. Section (5.4.2) reviews the scientific literature that has been published since AR5 to determine whether coastal and marine adaptation solutions have advanced to address key knowledge gaps identified in AR5, and whether the implementation of adaptation activities remains aspirational.

Section 5.4.2 begins by assessing the adaptation frameworks that have developed for addressing coastal and marine climate change adaptation (excluding sea level rise, considered in Chapter 4). Discussion then focusses on soft solutions, like nature- and community-based adaptation responses (Section 5.4.2.1), before considering adaptation responses from the built environment (Section 5.4.2.2). Section 5.4.2.3 assesses socio-institutional and planning-based adaptation responses, including incentives and partnerships, before assessing the effectives of adaptation actions and scaling up of responses. Finally, consideration is given to gender, health and socio-economic factors in adaptation responses. A description of these three sections is given in Table 5.8

Table 5.8 Description of the three main focus areas of Section 5.4.2.

Section:	SROCC description
5.4.2.1 Nature-based/ Ecosystem-based Adaptation	The use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change.

	Ecosystem-based adaptation uses the range of opportunities for the sustainable management, conservation, and restoration of ecosystems to provide services that enable people to adapt to the impacts of climate change. (Narayan et al., 2016b; Moosavi, 2017)
5.4.2.2 Built Environment Adaptation Solutions	Adaptation solutions pertaining to coastal built infrastructure and the systems that support such infrastructure. (Mutombo and Ölçer, 2016; Forzieria et al., 2018)
5.4.2.3 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)
5.4.2.3.1 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)

5.4.2.1 Ocean-based Climate Change Adaptation Frameworks

In AR5, there was limited evidence for the effectiveness of integrated coastal management and adaptive management as effective tools for climate change adaptation planning (Wong et al., 2014a). In this section, an assessment is made of the science that has supported the development of adaptation frameworks and guided climate change adaptation actions. This is followed with an assessment of the development of assessments of adaptation frameworks in the literature.

5.4.2.1.1 Adaptation planning, modelling and knowledge support:

There is a substantial body of literature that reports on scientific endeavor that supports adaptation planning, These include vulnerability assessments on ecoystems (Kuhfuss et al., 2016), species (Cheung et al., 2015; Cushing et al., 2018), communities (Islam et al., 2013; Himes-Cornell and Kasperski, 2015), integrated assessments of all of the above (Peirson et al., 2015; Kaplan-Hallam et al., 2017; McNeeley et al., 2017; Ramm et al., 2017a; Mavromatidi et al., 2018) and modelling to support decision making processes (Čerkasova et al., 2016; Chapman and Darby, 2016; Jiang et al., 2016; Justic et al., 2016; Joyce et al., 2017; Mitchell et al., 2017). The decision making process is aided by economic evaluations (Bujosa et al., 2015; Jones et al., 2015), evaluations for ecosystem services (Micallef et al., 2018; MacDonald et al., in press) and participatory processes: (Byrne et al., 2015).

A growing body of literature is emerging on the development of adaptation pathways, frameworks and decision making: (Buurman and Babovic, 2016; Dittrich et al., 2016; Michailidou et al., 2016; Cumiskey et al., 2018; Osorio-Cano et al., in press), as well as the development of indicators to support evaluation of adaptation actions (Carapuço et al., 2016; Nguyen et al., 2016) through monitoring frameworks (Huxham et al., 2015).

There is near universal agreement on the need for the integration of scientific research into management decision-making (Endo et al., 2017; Thorne et al., 2017), deepening understanding of climate change adaptation in coastal ecosystems and human adaptation processes(Dutra et al., 2015; Cvitanovic et al., 2016). This includes consideration of non-climate change-related factors; the inclusion of multiple stakeholders in participatory planning processes (Archer et al., 2014b; Abedin and Shaw, 2015) (*high confidence*), the need to integrate research across national boundaries (Gormley et al., 2015; Williams et al., 2016), scientific research to deepen understanding of how ecosystems can support adaptation (Hobday et al., 2015b; Dalyander et al., 2016; McNeeley et al., 2017; Osorio-Cano et al., in press), socio-economic (Merkens et al., 2016) and development of climate compatible development pathways (see BoG) (*high confidence*)

Despite the increased development of the literature since AR5, there remains a need for monitoring programmes that assess adaptation actions including regional-scale prediction sand climate drivers with local monitoring actions (Barrett et al., 2015; Bell et al., 2018). There is a notable lack of studies from African and Caribbean least developed countries and small island developing states. (Kuruppu and Willie, 2015) and few locally-focussed coastal climate change adaptation planning tools (Torresan et al., 2016). To develop a comprehensive understanding climate and non-climate variables in planning for coastal adaptation, big data, using high end computational capabilities are required (Rumson et al., 2017).

5.4.2.1.2 Adaptation frameworks assessment

This section assesses progress that has been made with implementing coastal adaptation measures, mostly from coastal communities, and continues with as assessment of what is required for successful adaptation, and concludes with limits to adaptation.

In a global analysis of 401 local governments, (Araos et al., 2016) found that just 15% reported on adaptation actions, and 18% reported on planning towards adaptation policy. The 15% was dominated by large cities in high income countries Araos et al. (2016).

In an analysis of the state of adaptation planning in Australian coastal local governments, the small number of case studies suggests that only a few coastal local governments have begun adaptation planning and that there is opportunity to adopt principles of best practice (Ramm et al., 2017b). Five South East Queensland local governments that were analysed had capacity to address coastal hazards, but some suburbs retained high levels of vulnerability due to their inherently risky location (Sano et al., 2015). This residual risk indicates the limits to which adaptation is possible in this example. The approach by Australian households to climate change is dominated by greenhouse gas emissions mitigation activities and coping strategies. This approach by Australian households to climate change is associated with household risk perception and climate change belief (Elrick-Barr et al., 2016). In stakeholder led adaptation strategies on the Gold Coast, a poor understanding of climate change issues could lead to maladaptation (Torabi et al., 2018). Similarly, the incorrect treatment of uncertainty, and therefore its potential for maladaptation responses, led Ramm (Ramm et al., 2017b) to conclude flexible adaptation pathways were better suited for managing uncertainty. Australian cities are engaging with climate change adaptation, the range of strategies and level of household awareness, as outlined in the examples given in this paragraph, implies some cities/ suburbs/ housholds may fare better than others as climate impacts deepen (*medium confidence*).

In the northern hemsiphere, investment in research and science has improved British local government practitioners technical and cognitive ability and local adaptive capacity has been developed. Implementation of adaptation actions has, however, been impeded by budget cuts and lack of national level political support. (Porter et al., 2015). In a survey of 150 Pacific North West coastal natural resource managers time, money and staff resources were their largest adaptation obstacles. Differing levels of access to information corresponded with differences in preparedness and willingness to plan for adaptation (Thorne et al., 2017). Interest in adaptation effort is more likely in individuals experiencing extreme weather (Ray et al., 2017). For example, nature-based solutions and improved coordination effort increased in USA cities located on the Atlantic coastal regions following Hurrican Sandy (Oswald Beiler et al., 2016). The assessment for northern hemisphere cities is similar to that of Australian cities.

There is a paucity of papers reporting adaptation assessments from developing countries, with the discourse focussing more on challenges with adaptation plannning, including uncoordinated, top-down approaches, lack of political will and resources (Elias and Omojola, 2015). Positive responses, reported from developing countries, were expressed in terms of successes with community (see Section 5.4.2.3.1) and ecosystem (Section 5.4.2.1.5) based adaptation approaches. The difference in adaptation assessments between developed and developing economy cities suggests some divergence in approach to addressing climate change (*medium confidence*).

In contrast to the many examples of proposed frameworks for adaptation, there remains relatively little literature that has assessed the success of such frameworks, possibly due to the time-lag between implementation, monitoring and evaluation and reporting. Integrated adaptation planning with non-climate change related co-impacts remains an under-achieved ambition, especially in developing countries (Finkbeiner et al., 2018). The dearth of literature from developing countries suggests that those cities from developing countries lag behind cities from developed countries.

In developing adaptation frameworks, a number of key principles are repeated, including the requirement for a robust but flexible approach that delivers adaptation responses across a range of projected climate scenarios and which accounts for deep uncertainty through well-coordinated participatory processes (Jiao et al., 2014b; Dutra et al., 2015; Dittrich et al., 2016; Buurman and Babovic, 2017).

Examples of these principles that aim to increase efficiency and robustness include taking a whole system approach that addresses underlying socio-ecological compounding factors (Sheaves et al., 2016b) , including within cost-benefit analyses (André et al., 2016) and providing co-benefits for human development and the environment (Wise et al., 2016). In terms of developing adaptation pathways, there was substantial support for 'no regrets' approaches addressing both proximate and systematic underlying drivers of vulnerability (Sánchez-Arcilla et al., 2016; Pentz and Klenk, 2017; Zandvoort et al., 2017) with leadership, adaptive management, capacity and the monitoring and evaluation of actions deemed necessary governance responses (Dutra et al., 2015; Doherty et al., 2016). Despite the progress that has been made, the residual risk faced after adaptation, for example the suburbs in the South East Queensland local governments above, points to a limitation in adaptation as a response. For climate compatiable development, more extensive learning processes are necessary to build decision-makers' capacity to tackle systemic drivers, and to scrutinise potentially mal-adaptive infrastructural investments (Wise et al., 2016)

The next three sections of this chapter assesses three categories of adaptation: nature-based (Section 5.4.2.1), built environment and engineered (Section 5.4.2.2) and socio-institutional solutions (including community-based adaptation; Section 5.4.2.3). Community-based adaptation typically refers to bottom-up community-driven adaptation responses in which public participation processes are central to their implementation. A summary of the discussion of the adaptation solutions assessment can be found in Table 5.9. The categorisation used here is consistent with recent treatment of coastal adaptation issues in the IPCC Fourth and Fifth Assessment Reports (Wong et al., 2014a)

Table 5.9: Summary table of climate change impacts upon eco- and human-systems, and an assessment of adaptation responses, reported in Secton 5.4.2. See Section 5.2.2 for a discussion of key ecosystem vulnerabilities to multiple climate hazards for an in-depth assessment of the Hazards column, and Section 5.3 for a discussion on the impact of climate hazards on ecosystems and human wellbeing.

Ecosystems/ Human systems	Hazards	Impacts & Drivers	Adaptation response	Assessment (*Confidence*)
Coral reefs	Bleaching Inundation Dissolution Structural damage	Warming Sea level rise 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 (*high*) Coral epigenetics, 'assisted colonization', 'assisted evolution', novel coral–symbiont associations and coral microbiome manipulations as adaptive management tools for reef rehabilitation (*low*)
Mangroves, coastal wetlands, seagrass beds and saltmarshes	Storm damage Inundation	Extreme storm events Sea level rise	Synergistic[1] 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 (*high*).
Coastal dunes and sandy beaches	Erosion Inundation Fire	Extreme storm events Sea level rise	Replenishment schemes and dune restoration Socio-institutional response include	Rate of loss of sand and level of participation of stakeholders of important factors (*medium*)

		Heat waves/ drought	stakeholder engagement	
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.	*Virtually certain* 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 and Local Knowledge (ILK) High future need for transboundary agreements	*High confidence* in efficacy of stakeholder participation and MPAs, but some resistance from conservative administrations to the former. *Medium confidence* in ILK (Cross-Chapter Box 3) *High agreement* on need for transboundary agreements.
Tourism	Tourism industry decline (changing behaviour and choices) Infrastructure loss Resource declines	CO_2 emissions Changes to ecosystems Sea lever rise/ storm surge Climate impacts on resources e.g., water availability	Participatory planning processes Ecosystem Based Adaptation Community Based Adaptation Awareness raising	*High confidence* that climate change will impact upon tourism *High agreement, limited evidence* that a range of adaptation responses will be required locally *Likely* winners and losers
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, capacity development Target fishery switching Fish Aggregating Devices Transnational agreements Knowledge development Community responses	*High agreement* that optimal fisheries management integrated with community responses will be critical to offsetting global fisheries landings declines *High agreement* that knowledge gaps in fisheries management need to be filled for successful adaptation *Medium confidence* in ability of community based responses to offset social impacts.
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	*High confidence* in EAA and ICZM approaches *High confidence* for translocation and polyculture Resilience and climate compatible development very important

			Socio-ecological resilience building initiatives Climate compatible development	

Notes:
[1] synergistic adaptation response: multiple adaptation actions that complement and act synergistically with each other.

5.4.2.2 Nature-based / Ecosystem-based Adaptation

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

5.4.2.2.1 Coral reefs

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

Driven by the qualities of the silviculture approaches, (Rinkevich, 1995; Rinkevich, 2005; Rinkevich, 2006; Rinkevich, 2008; Bongiorni et al., 2011) has proposed a two-step restoration strategy termed gardening of denuded coral reefs. The central concept of the first step is the farming of large pools of corals in situ, in mid-water floating nurseries that are installed in sheltered zones, in which coral material can be cultured for up to several years. In the second step, nursery-grown coral colonies, together with recruited associated biota, are transplanted to degraded reef sites (Shafir and Rinkevich, 2008; Mbije et al., 2010; Shaish et al., 2010b; Shaish et al., 2010a; Bongiorni et al., 2011; 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 et al., 2016; Ng et al., 2016; Lohr and Patterson, 2017; Rachmilovitz and Rinkevich, 2017). Active restoration of coral reefs, while still in its infancy and facing a variety of challenges (Rinkevich, 2015b; Hein et al., 2017), is now capable to handle a wide range of ecological and socio economic applications (Rinkevich, 2014; Rinkevich, 2015b; Rinkevich, 2017) (*high confidence*).

Recent studies focus also on ecological engineering approaches (Rinkevich, 2014; Forsman et al., 2015; Coelho et al., 2017; Horoszowski-Fridman and Rinkevich, 2017; Linden and Rinkevich, 2017; Rachmilovitz and Rinkevich, 2017), on species interactions that reef managers may utilize to facilitate the restoration of corals and/or the augmentation of functional diversity, including microbiome (Casey et al., 2015; Horoszowski-Fridman and Rinkevich, 2017; Shaver and Silliman, 2017), the transplantation of whole niches (Shaish et al., 2010b; Gómez et al., 2014) and on keeping/enhancing the genetic diversity (Iwao et al., 2014; Drury et al., 2016; Horoszowski-Fridman and Rinkevich, 2017). Active restoration becomes a major tool for reef rehabilitation in many countries worldide (representing all major reef regions), particularly in Southeast Asian region (Hilmi et al., 2018).

While the coral gardening approach is continually 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 rehabilitating (*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, recent studies (Drake et al., 2018) reveal that coral cells possess the molecular machinery that helps compensating for the effects of global change, such as ocean acidification.

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

[START BOX 5.4 HERE]

Box 5.4: Coral Reef Restoration as Ocean-based Adaptation

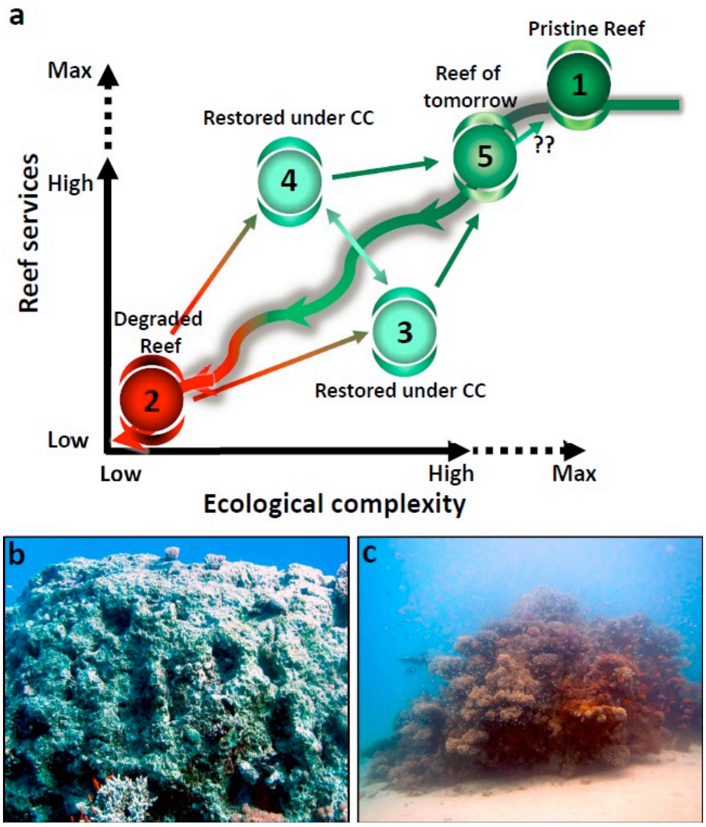
Global change induced by human activities is impacting all warm-water corals (and the reef structures they form) (Section 5.2.2.3.3). Impacts are rapidly increasing in scale and intensity, exposing coral reefs to enhanced degradation rates and diminishing capacities to maintain ecological resilience,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

Today, the most tested reef restoration approach involves 'coral gardening', a two-step process, similar to silviculture concepts and practices (Rinkevich, 2006). In the first step, a large pool of coral colonies (derived from coral nubbins and fragments, and from sexually derived spat) (Shafir et al., 2006; Barton et al., 2017; Linden and Rinkevich, 2017; Calle-Triviño et al., 2018) are farmed in underwater nurseries, preferably on mid-water floating devices. In the second step, upon reaching adequate size and maturity, nursery-farmed corals are transplanted in degraded reef sites. About 90 coral species and over 100,000 colonies that were farmed in various archetypical nurseries across the world, followed by a broad range of transplantation tactics showed high survivability, growth rates and reproduction (Rinkevich, 2014). Whereas the 'coral gardening' restoration techniques are still lagging behind restoration practices developed for terrestrial habitats, reef restoration has advanced greatly in the last two decades (Rinkevich, 2014; Lirman and Schopmeyer, 2016). This advance includes economic considerations (Rinkevich, 2015b; Barton et al., 2017; Flores et al., 2017; Hein et al., 2017) and citizen science involvement (Cruz et al., 2014; Hesley et al., 2017). Recent studies (Frias-Torres and van de Geer, 2015; Lirman and Schopmeyer, 2016; Montoya Maya et al., 2016; Jacob et al., 2017; Rachmilovitz and Rinkevich, 2017) have added ecological engineering perspectives, while also targeting the enhanced conscription of reef dwelling invertebrates and fish and augmented recruitment of coral spats. The addition of the assisted evolution tools, when developed, may improve coral restoration outcomes (van Oppen et al., 2015; van Oppen et al., 2017).

However, the 'coral gardening' approach is facing two foreseen challenges. The first is the scaling up, its viability as a large-scale restoration measure at the level of hundreds of thousands/millions of coral colonies/site (aiming at the elucidating of the sum ecological attributes at large-scale approaches)

(Rinkevich, 2014) (*high confidence*). The second (Box 5.4, Figure 1), is the effectiveness of active reef restoration to mitigate or rehabilitate global change impacts and whether it is applicable also for low lying ocean states scenarios (Shaish et al., 2010a; Schopmeyer et al., 2012; Coles and Riegl, 2013; Hernàndez-Delgado et al., 2014; Rinkevich, 2015a; Wilson and Forsyth, 2018) (*high confidence*). Also, the literature reveals a wide range of calculated restoration costs that did not consider the high monetary value of the coral reefs' ecosystem services (Bayraktarov et al., 2016; Rinkevich, 2017) (*high confidence*).

Yet, it has been proposed (Rinkevich, 2015a) that 'coral reef gardening' has the capacity to enhance adaptability of coral reef organisms to climate change. If reef restoration approaches are implemented and that climate change induces poleward range expansion of corals' reef habitats (Precht and Aronson, 2004; Yara et al., 2011; Madin et al., 2016), it is envisaged that the "reefs of tomorrow" would not resemble the "reefs of today" (Box 5.4, Figure 1) (Rinkevich, 2008; Graham et al., 2014) (*high confidence*). The very high vulnerability of coral reefs to warming over reefs, ocean acidification, the increase in storms' intensity and severity and sea level rise under climate change (AR5 WG2) points to the needs for active restoration measures (*high confidence*) and for improved cultivation and transplantation methodologies.



Box 5.4, Figure 1: Coral reef restoration as an ocean-based adaptation tool to climate change. (A) 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), the 'reef of tomorrow' state (circle 5) and two intermediate 'restored reef-state' scenarios (circles 3,4). 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; Hernàndez-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. The path from the 'reef of tomorrow' towards the pristine state is still at a theoretical level. Based on (Rinkevich, 2014). (B) A denuded knoll at the Dekel Beach, Eilat, Israel before reef transplantation was taken (November 2005; Photo: Y. Horoszowski-Fridman). (C) The same knoll, restored, which 300 nursery-grown colonies of 7 coral species were transplanted during three successive transplantation (2005, 2007, 2009), 11 years later (June 2016; photo by Shai Shafir). The knoll is surrounded by reef inhabiting schools of fish.

[END BOX 5.4 HERE]

5.4.2.2.2 Protecting and restoring mangrove swamps

Mangrove swamps provide significant coastal protection services from extreme storm events, supporting services through increased sedimentation rate Hayden and Granek (2015) and provisioning services for local communities (Section 5.3.1.1). They can also provide carbon mitigation and job creation co-benefits (through Reducing Emissions from Deforestation and Forest Degradation (REDD+) programmes) when managed properly, and there is evidence of their value for aquaculture initiatives (Huxham et al., 2015; Ahmed and Glaser, 2016). When used in combination with other adaptation responses their value increases; for example in Bangladesh sedimentation rates increased when mangroves were planted on the lee side of an oyster farm, with the latter serving to reduce wave energy reaching the mangroves (Ahmed and Glaser, 2016). Using a combination of synergistic adaptation responses improves climate change adaptation outcomes and co-benefits (*medium evidence, high agreement*). Loss of mangrove swamps and fringing coral reefs has the potential to increase impacts of future cyclones in Bangladesh (Ataur Rahman and Rahman, 2015; Ahmed and Glaser, 2016). In establishing community-focussed mangrove management plans in Vietnam, the importance of providing technical guidance as part of the adaptation action was noted (Nguyen et al., 2017), along with the need for the incorporation of mangrove regulations into local planning instrumentst (Sierra-Correa and Cantera Kintz, 2015), who found that mangrove swamps are the ecosystem most threatened by sea level rise.

There are many examples of coastal protection services provided by mangrove forests have been restored through community-based mangrove restoration programmes (*high confidence*), and such programmes can provide additional community and carbon sequestration benefits (*medium evidence*). There are also, however, examples where such initiatives have resulted in maladaptation, generally through poor governance processes or lack of community compliance with restoration plans. This emphasises the need for effective implementation of adaptation responses.

5.4.2.2.3 Coastal ecosystem-based adaptation

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

Coastal dune systems are widely transformed globally, partly because of their desirability, being adjacent to the shoreline and this has caused many dunes to be severely degraded. Human disturbance and a low stabilising ability of vegetation are key causes of degradation (Ciccarelli et al., 2017) with restoration effort supported by dune builder species and sand replenishment schemes. There are a number of examples of sand replenishment projects as a soft measures approach to combat erosion; see (Vikolainen et al., 2017) for an example. Such projects have, however not always been successful due to, for example fire damage (Shumack and Hesse, 2017) or the rapid loss of sand within replenishment schemes due to coastal processes, and in some cases due to stakeholder rejection of adaptation activities (Pranzini, 2017), with the latter suggesting insufficient stakeholder engagement prior to project operationalisation. An example of an intermediate response between natural and built infrastructure for adaptation was reported in a Mauritian demonstration project, where a gravel beach was constructed to protect low-lying developments (Onaka et al., 2015). The combination of engineered and sand replenishment responses are considered complementary approaches (Martínez et al., 2017).

5.4.2.2.4 Summary of ecosystem-based adaptation assessment

In the following section, the natural and ecosystem-based adaptation (EBA) responses described above are assessed for their efficacy, learning outcomes and gaps in the knowledge set.

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

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

There are, however, limitations acknowledged with coastal ecosystem based adaptation (Sussams et al., 2015; Adriana Gracia et al., 2018). These include the space that ecosystems require may not be available in an urban environment, non-climate change co-drivers like human impacts need to be addressed as part of the adaptation response (Ahmed et al., 2017; Peña-Alonso et al., 2017; Triyanti et al., 2017), the severity of climate impact may exceed the adaptive capacity of the ecosystem, for exampl the type of rainfall being experienced affects the efficacy of storm water associated green infrastructure (Joyce et al., 2017) or acidification affecting coral reef viability, and, potentially slower recovery rates within ecosystems than the recurrence of climate impacts, including the forcing of ecosystems into novel dyamic equilibrium states that may not deliver the level of service required. Given these limitations, it is important that EBA decision-makers are better informed about its context and site specific application (Sutton-Grier et al., 2015) to avoid maladaptation (*high confidence*).

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

5.4.2.2.5 Species adaptation

The previous sections dealt with adaptation solutions for ecosystems through EBA. This section assesses the scientific literature for adaptation solutions from a species perspective. An accurate understanding of climate change impacts upon species, their sensitivity and adaptive capacity and consequent ecological effects is required to estimate extinction risk, so that an appropriate management response can be developed (Butt et al., 2016). It is particularly important to consider both indirect as well as direct physiological impacts. For example, warming of the Arctic might be beneficial in terms of physiological tolerance, but loss of sea-ice habitat could reduce (or alternatively increase) foraging habitat (Castellini, 2017). Management of ecosystems for climate change will, therefore, require a focus beyond traditional mandates (Hobday et al., 2015a). Species adaptation responses that address both the direct and indirect impacts of climate change, as well as changes to ecosystems supporting species, for examples transformation through alien invasive species (Whitfield et al., 2016) will require broader consideration than traditional species management practices (*medium evidence, good agreement*). This will include the development of international collaborations and databases to improve ocean-scale understanding of climate change impacts (Okey et al., 2014; Young et al., 2015). A key knowledge gap of where the critical thresholds are for irreversible change for species remains, and this should be prioritised (Powell et al., 2017). Examples of individual species' responses to climate change can be found in Sections 5.2.3: Risks for Organisms.

5.4.2.3 Built Environment and Engineered Climate Change Adaptation Solutions

This section assesses the impact of climate change upon the built environment and adaptation measures to address these impacts. 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. An assessment of the efficacy of adaptation measures for climate impact variables is provided. 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). The interaction between complex human systems and the variable nature of climate risks presents 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, in press) (*high confidence*).

Increasingly, there is a shift in literature towards viewing infrastructure as a complex set of interactive human-based systems (Chappin and Van der Lei, 2014; Monbaliu et al., 2014; Rahimi et al., 2014). This means that the traditional adaptive approach based on linear climate risk assessment (where risk from singular impacts is associated with an adaptation response for that impact) is likely to be inappropriate for these complex systems (Bles et al., 2016; Mutombo and Ölçer, 2016; McCurdy and Travis, 2018) (*robust evidence, good agreement*). Although there appears to be adequate scientific and technical data available to allow design engineers to assess climate risks in the built environment, methodologies for assessing and incorporating these risks into adaptation vary widely in the literature. Too often, critical adaptation decisions remain under individual prerogatives of engineers (Mutombo and Ölçer, in press) who often approach this matter from an engineering silo perspective. A proposed solution to this could be collective consideration of adaptation response with multiple stakeholder input to provide a more inclusive approach to built infrastructure adaptation.

5.4.2.3.1 Climate impacts upon built infrastructure:

The climate variables, discussed next, are most likely to affect the built environment: sea level, wave, salinity,wind, rainfall and humidity (Colin et al., 2016; Mutombo and Ölçer, 2016; Antwi-Agyeia et al., 2018; Forzieria et al., 2018). Depending on type of infrastructure, geographical location and level of economic and technological development, some climate variables may have a greater influence than others (Carter, in press). 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 each climate variable on the built environment.

Sea level rise:

Sea level rise causes, projections, impacts and adaptation response is dealt with in Chapter 4. Approximately 70% of the coastlines worldwide will experience sea level change within 20% of the global mean sea level change (Hartmann et al., 2013). The impact of sea level rise on coastal infrastructure include inundation due to shifting tidal and splash zone levels, degradation due to increased wave overtopping, failure of foundation due to barriers and material displacement (Mutombo and Ölçer, 2016).

Despite the persistence of uncertainty in predicting sea level rise, substantial progress has been made in respect of the ability to model, understand and predict sea level rise processes. This has allowed the development of improved adaptive responses, which are underpinned by three main strategies in literature: retreat, accommodate or protect. (Demirbilek, 2013; Hunter et al., 2013; Gibbs, 2016; Ruleau and Rey-Valette, 2017).

Wave:

Most literature on wave studies are associated with ship movements and manoeuvrability (Park et al., 2014; Hizir et al., in press) rather than for the built infrastructure. Coastal infrastructure on shorelines is designed to withstand specific wave-induced load based on historical wave data. These structures may therefore not be able to perform optimally under changing wave conditions. Furthermore, excessive deterioration of infrastructure may be experienced due to the overflow of sea water onto infrastructure as a result of overtopping of waves (Kong et al., 2013).

As with sea level rise, to reduce the risk of material displacement, inundation and overtopping, there is high agreement in the literature that adaptation to wave climate should be underpinned by retreat, accommodate or protect (Demirbilek, 2013; Hunter et al., 2013; Gibbs, 2016; Ruleau and Rey-Valette, 2017).

Salinity:

Increased salinity often accelerates deterioration of infrastructure erected in areas exposed to sea water. Despite this, given the minor predicted change in salinity (Girjatowicz and Świątek, 2016) salinity is globally identified as a low risk to the built environment (Escobar et al., 2016; Girjatowicz and Świątek, 2016).

Wind:

Changing wind patterns and speeds is one of the consequences of global warming, which impacts mainly on operations and manoeuvrability of vehicles. Prevailing wind patterns based on available recorded data are traditionally factored into infrastructure design during development stage. Unfortunately, as a result of global warming, factoring accurately wind factor during design stage has now become very challenging and the built environment is therefore increasingly becoming vulnerable to the effects of wind (Hdidouan and Staffell, 2017). Despite the negative impacts on operations and manoeuvrability, change in wind patterns offer tremendous opportunities for renewable energy.

Rainfall/ run-off:

It is very unlikely that drainage infrastructure designed on the basis of current rainfall records will be suitable for future rainfall patterns (Villatoro et al., 2014; Elshorbagy et al., in press). Storm water infrastructure is traditionally designed under the understanding that the probability of extreme rainfall is statistically stationary. However, climate change has now brought new challenges. Precipitation patterns have changed dramatically, causing major impacts such as inundation, flooding, erosion, and deterioration of structures. This places additional pressure on storm water design engineers in their quests to develop cost effective solutions for the worst case precipitation scenario.

Despite the high consensus on the importance of adjusting storm water infrastructure (Kong et al., 2013) as part of adaptation within the built environment, there is still a very low level of confidence in respect of understanding and predicting precipitation variability. This remains a major impediment for storm water infrastructure adaptation response (*medium evidence, high agreement*).

Humidity:

Humidity is the measure of the amount of water vapour in the air. An increased concentration of CO_2 in the atmosphere affects temperature and humidity. There is 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. Despite this, humidity is globally identified as low risk to the built environment (*high confidence*).

5.4.2.3.2 Climate change adaptation solutions for built infrastructure:

Within the built environment, adaptation seeks to moderate or avoid harm, or exploit beneficial opportunities (McCurdy and Travis, 2018). There are numerous politically-oriented-response methodologies (Colin et al., 2016; Azhoni et al., 2018; Forzieria et al., 2018) which aim at analysing and reviewing governance policies, legislative frameworks and institutional capacity. In addition, adaptation guidelines within the built environment tend to follow a general flow: 1, Study impact of climate change, 2, identify areas of vulnerability and, 3, propose adaptive responses to address vulnerability (Bles et al., 2016; Colin et al., 2016; Hauge et al., 2017; Forzieria et al., 2018).

Given the shift in the definition of the word infrastructure, from a purely physical to a socio-technical system which embodies soft notions (Chappin and Van der Lei, 2014), many authors recommend the need for two levels of adaptation, technical and organisational (Antwi-Agyeia et al., 2018; McCurdy and Travis, 2018); and this is also often referred to in literature as hard and soft adaptation. Further, concerns have been raised about the ineffectiveness of the "predict-then-act" approach, due to the large uncertainties with climate predictions; thereby raising the need for creating system flexibility through dynamic adaptive pathways (Antwi-Agyeia et al., 2018; McCurdy and Travis, 2018). There is evidence of a shift in how adaptation to the built environment is viewed, from a purely structural issue, to one much more nuanced and steeped in human systems analysis (Bles et al., 2016; Hauge et al., 2017; Forzieria et al., 2018) (*high confidence*).

The common management tools for prioritisation of adaptation options rely more on perception and prerogative of those conducting the assessment. This explains the low agreement in respect to tool selection. These tools include (but are not limited to) the following: 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 decision models focussing on low regret, no regret, win-win.

Given their vulnerable locations on the interface between sea and land, coastlines are mostly affected by climate change. In many countries, Integrated Coastal Management is widely implemented to address issues relating to coastal development and climate change (Rosendo et al., 2018). Strategies for coastal management generally evolve around an action-reaction approach or/and post-disaster basis (Gibbs, 2016; Gracia et al., 2018) and it often entails the installation of hard engineering structure such as gabions, revetments, seawalls and breakwaters. Meanwhile, there is increasing evidence of the negative effect that hard engineering has on the coastal ecosystems (Sheaves et al., 2016a) and this has led to a new drive in favour of an ecosystem-based approach to adaptation (Gracia et al., 2018; Rosendo et al., 2018). This approach is more sustainable, adaptable, multi-functional, economically viable and has self-repair and recovery ability (Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; Rosendo et al., 2018) (*high confidence*). For instance, there is high agreement that the creation and restoration of wetlands, seagrass beds and dune vegetation are effective long term responses in combatting erosion to support built infrastructure (Gibbs, 2016; Sheaves et al., 2016a; Gracia et al., 2018; Rosendo et al., 2018) (*high confidence*).

Despite the high agreement on the risks posed by climate change in the built environment (Araos et al., 2016), in the pursuit of adaptation, the persistent trade-off between cost and solutions (Baum, 2012; Ölçer and Ballini, 2015) has been recognised as a major impediment (*high confidence*). There is compelling evidence that this trade-off has led to maladaptation (Azhoni et al., 2018), especially in developing countries with competing developmental priorities (Hoggart et al., 2014). However, to curb this challenge, recently, concepts of incremental and transformational adaptations are emerging in literature (Felgenhauer, 2015; Bosomworth et al., 2017; Mushtaq, 2018).

Incremental adaptation is defined as actions taken progressively over medium term with the aim to maintain the essence and integrity of the system while addressing threats (Aparicio, 2017; Huber-Stearns and Cheng, 2017). However, at some point, the level of adaptation effort to maintain the effectiveness of the system may become unfeasible; thereby prompting the need for transforming the system. Transformational adaptation is defined as adaptation that changes the fundamental attributes, or irreversible regime change, of a system with the aim to increase the capacity of the biophysical, social or economic systems to achieve the desired values (Werbeloff et al., 2016; Bosomworth et al., 2017).

Paradoxically, in developed countries where there is a rising tide of cultural anxiety towards risk, (Mythen, 2004) warned society of the danger of becoming less concerned with things that can go right and more concerned with things that can go wrong. This constant anxiety in society could lead to adaptation in excess of what may actually be required, resulting in wasted expenditure Azhoni et al. (2018). Nevertheless, an important element on the necessity of adapting infrastructure to climate change is the fact that, due to the gradual increase in maintenance regime, there is greater consensus that non-adaptation is more ineffective and costly over a long term horizon (Azhoni et al., 2018; Mushtaq, 2018).

In addressing climate change, there is an increasing focusing on the concept of interdependency in the built environment. There is clear evidence that a shift from quantifying specific risk factors to portraying the interdependencies between these factors (Chappin and Van der Lei, 2014; Antwi-Agyeia et al., 2018) is largely advocated as the most favourable approach to adaptation (*medium evidence, good agreement*). Evidence is progressively emerging to showcase that, due to interdependency, known catastrophic events directly from climate variables are not always the cause of major disruptions (Higgins, 2013; Rodrigue et al., 2013) (*medium confidence*). Minor disturbances originating from the environment can cascade, through a series of non-climate change related impacts, to create major visible damage in an infrastructure system. As a result, there is increasing agreement within the built environment sector on the need for a systematic cross-sector approach to addressing climate risk (Rahimi et al., 2014; Araos et al., 2016), also referred to as inter-organizational networks (Azhoni et al., 2018) (*high confidence*). Recent literature trends reveal a gap in understanding of how climate variables are linked to each other and to the built environment (Chappin and Van der Lei, 2014; Forzieria et al., 2018), and how they can harmonized. Effective and informed decision-making takes into consideration a range of climate change and other human system factors within the built environment, including the complex interrelationship between the built environment and economic activities (Chappin and Van der Lei, 2014; Antwi-Agyeia et al., 2018; Venghaus and Hake, in press)(*high confidence*).

Resistance and resilience are considered to be the two major effective adaptation response strategies for the built environment (Lebaka et al., 2016; Mutombo and Ölçer, 2016; Pursiainen, 2018). Resistance, which is often referred as 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 would be resilience (Lebaka et al., 2016; Pursiainen, 2018), which (Azhoni et al., 2018) describes as transformational adaptation.

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

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

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

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

Guidelines for planning for adaptation tend to follow a general path from assessing impact, to identifying vulnerability and finally developing responses. 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.

Meanwhile, in spite of the high agreement on the need for adaptation, defining successful adaptation still remains a challenge given that perceptions and risk appetites may substantially differ. Also, in view of the long term nature of climate processes, adaptation is continuous. As new challenges emerge, questions of what actually counts as successful adaptation remain.

It is worth noting that despite all the guidelines, research, recommendations and available tools for adaptation, the ultimate call to whether or not to adapt remains the prerogative of an individual or manager who will act based on his perceptions, beliefs, culture and willingness. Environmental psychological perspective is currently receiving very little attention, yet it fully dictates the outcome of any climate initiatives, and probably dictating our future survival on the planet.

5.4.2.3.3 Adaptation solutions: technology, artificial upwelling system (for aquaculture)

A paradigm is the artificial upwelling enigineeringfor aquaculture. In the open oceans, nutrients availability is the major limiting factor for primary productivity (Arrigo et al., 1999; Hlaili et al., 2006; Leinen, 2008). Upwelling can bring up high-nutrient deep water to the euphotic zone and enhance primary production (Mackenzie et al., 2002; Farias et al., 2015). However, upwelling can also bring high DIC and low oxygen water to the surface, which can cause acidification and hypoxia (Demarcq, 2009; Narayan et al., 2010; Bauer et al., 2013; Miranda et al., 2013). It is thus of critical importance to take the advantages but avoid the disadvantages of upwelling effects. Currently there are two types of experimental practice in artificial upwelling. One is to mimic natural upwelling processes to contribute to the development of fisheries at GeoMar (http://www.geomar.de/en), the other to improve the seaweeds culture system at COCA (pan China Ocean Carbon Alliance).

The key to this goal is the extent of the upwelling (Jiao et al., 2014a). The application of artificial upwelling powered by green energy (such as solar energy, wind/wave/tidal energy) (Zhang et al., 2016) to seaweeds culture is a paradigm (Jiao et al., 2014b; Zhang et al., 2015; Pan and Schimel, 2016). By controlling the extent of the artificial upwelling flux, moderate amount of deep water can be gradually brought up to the euphotic zone just to meet the demands of nutrients and DIC by the seaweeds, and thus neither acidification nor hypoxia will happen, as DIC is mostly used, and oxygen is efficiently produced by photosynthesis. Under such scenarios, the microbial carbon pump (MCP) could work efficiently and the sum of the MCP and the biological pump could reach its maximum (Jiao et al., 2010; Jiao et al., 2014b). Such artificial upwelling operations also gradually release the ‘bomb’ of rich nutrients and hypoxia, which could breakout when storms take place otherwise (Daneri et al., 2012). The artificial upwelling units can be deployed in massive seaweeds caluture areas after assessment of the environmental conditions. *In situ* monitoring of physical and chemical varialbles such as temperature, nutrients, oxygen, DIC and pH etc. are used for adjustment of the upwelling extent for adaptative management.

5.4.2.4 Socio-institutional Adaptation Solutions

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

5.4.2.4.1 Community-based adaptation

Community-based adaptation (CBA) refers here to coastal communities’ and non-government organisations’ climate change adaptation responses. It includes participatory decision-making and bottom-up approaches to implementing climate change adaptation.

Community based adapation actions include community organisation for action, communities’ engagements with local management authorities, individual level responses to climate change at a homestead scale, livelihood challenges and responses. This assessment is based on reports of climate change impacts, community vulnerability and responses from around the globe, in both developed and developing nations.

The literature from the latter focusses more on serious threats to livelihoods (Islam et al., 2013; Ahmed and Diana, 2015b; Kuruppu and Willie, 2015; Lohmann, 2016; Smith et al., 2016a; Esteban et al., 2017; Shaffril et al., 2017a; Koya et al., in press), whereas the focus from developed nations has a greater focus on threats to infrastructure and the disturbance of community processes. (Serrao-Neumann et al., 2014; Dutra et al., 2015; Colburn et al., 2016; Elrick-Barr et al., 2016; Richards et al., 2016; Jurjonas and Seekamp, in press) Among the threats that contribute to vulnerability in developing economy communities, ext, access to resources is the clear defining difference between communities from the Global North and South (*robust evidence, medium agreement*).

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

In coastal Jakarta, Indonesia, a survey was conducted in poor communities about subsidence and SLR risks. While local inhabitants appear to be aware about the hazards they face, many seem to underestimate their severity, possibly due to a high frequency of exposure in the recent past. (Esteban et al., 2017). In a world with competing risks and urgent priorities, the local inhabitants appear to be unable to avoid, or are willing to carry the risk associated with a climate impact in order to meet other, more pressing needs. This example is a reflection of 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; Senapati and Gupta, 2017; Cumiskey et al., 2018; Koya et al., in press).

In an example of maladaptation, a community's engagement with local authorities in a community mangrove forest management project, resulted in the deterioration of the mangrove forest, with a corresponding increase in community vulnerability as the coastal protection and economic services provided by the mangrove forest eroded (Nguyen et al., 2017). Despite good intentions with this project in Kien Giang, Vietnam, a lack of enforcement, weak governance processes and insufficient technical guidance drove the project towards a maladapive outcome. This example serves to illustrate a point that is made regularly in community-based adaptation projects, around the need for communities to be served by clear policy and government institutions with accountable and firm governance processes and enforcement (*high confidence*).

5.4.2.4.2 Planning tools, marine protected areas and integrated coastal zone management

The Fifth Assessment Report concluded that Integrated Coastal Zone Management (ICM) was the best likely framework for addressing coastal climate change adaptation needs (Wong et al., 2014a). Incorporating stakeholder participation processes as a form of bottom-up co-management is desirable from the point of view of strengthening local and traditional knowledge bases and buy-in for implementation (Serrao-Neumann et al., 2013), *medium evidence, high agreement*). Some institutions with a strong top-down institutional framework, for example Thua Thien in Vietnam, may, however be resistant to changing to this approach (Abelshausen et al., 2015).

The incorporation of marine protected areas into coastal and oceanic zone management is considered an important strategy in adapting to climate change (Hopkins et al., 2016) (*medium agreement, medium evidence*), but management of such zones is, itself likely to be impacted by climate change (Johnson et al., 2018). Management approaches need to accommodate 'shifting baselines' due to climate change and novel dispersal movements of highly-mobile species (Elliott et al., 2015). Where such shifts occur across national boundaries, management success will require transnational agreements (Gormley et al., 2015). Transboundary ICM are likely to become more important with ecosystem shifts due to climate change (*medium evidence, high agreement*).

5.4.2.4.3 Tourism

The coastal tourism economic sector is highly sensitive to climate change, and will need to respond both in terms of mitigating carbon emissions and adapting to climate change impacts. An analysis, ranking mitigation and adaptation responses in Greece, prioritised rational energy use, energy efficiency and water management/ saving measures (Michailidou et al., 2016). It is clear that climate change impacts upon tourism are nuanced and not restricted just to physical impacts on tourism establishments (Biggs et al., 2015), but also tourist behaviour and choice changes, economic flows and market share adjustments. (Bujosa et al., 2015; De Urioste-Stone et al., 2016). In a coastal erosion example, sea level rise is interacting with non-climate change impacts including sand mining, inappropriate development and habitat destruction (e.g., mangroves), impacting upon tourism (Rangel-Buitrago et al., 2015). The management recommednation was appropriate legislation with a marine spatial planning emphasis, enforcement, sustainable funding mechanisms and support networks for decision-making.
Considering the above, participatory spatial planning processes and ecosystem based adaptation responses could improve tourism adaptation efforts (*medium confidence*). The climate change adaptation response of participants in the Thailand dive industry was reported to be based on misconceptions about climate change and personal observations (Tapsuwan and Rongrongmuang, 2015). In respect of community-based adaptation, effort needs to be invested in broadening the level of awareness about climate change to improve decision-making processes (Tapsuwan and Rongrongmuang, 2015). It is very likely that climate change will have direct and nuanced impacts upon coastal tourism (*high confidence*) and that a range of adaptation responses will be required to address this sector within specific geographic locations (*low evidence, good agreement*).

5.4.2.4.4 Fisheries and aquaculture

There is a substantial body of literature around the impacts of climate change on fisheries in the South Pacific and aquaculture in South and South East Asia. In the former, there are examples of best practice for optimal fisheries management, including employing ecosystem-based adaptation management, fish aggregating devices, switching to pelagic fisheries, trans-national agreements and engaging with aquaculture, amongst other actions for adaptation to climate change impacts like ocean warming, acidification and sea level rise, and consequential impacts upon coral reefs and fishing communities (Weng et al., 2015; Dey et al.; Valmonte-Santos et al.; Bell et al., 2017; Le Cornu et al., 2017; Senapati and Gupta, 2017; Wabnitz et al., 2017). It has been questioned, with conflicting conclusions, whether these adaptation measures will offset deficits anticipated through reductions in landings from existing efforts and growing demand (Hobday et al., 2015b; Dey et al., 2016a; Finkbeiner et al., 2016; Rosegrant et al., 2016; Campbell, 2017). In a systematic review of fisheries management frameworks, (Pentz and Klenk) found that adaptive management, institutional capacity development and input/output controls, respectively, were the most frequently used options of best practice for optimal fisheries management, including employing ecosystem-based adaptation management, fish aggregating devices, switching to pelagic fisheries, trans-national agreements and engaging with aquaculture, amongst other actions for adaptation to climate change impacts like ocean warming, acidification and sea level rise, and consequential impacts upon coral reefs and fishing communities (Dey et al.; Valmonte-Santos et al.; Bell et al., 2017); Wabnitz et al. (2017). Despite the good agreement on how fisheries can be managed for climate change adaptation, there are few studies where this has been shown to be successful (most papers remain aspirational) (*robust evidence, high agreement*).

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

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

In South Asia, where there is a heavy dependence upon aquaculture for both subsistence and commercial purposes, climate change is likely to substantially impact aquaculture activities through a number of mechanisms including biophysical impacts on farmed species, disruption from extreme weather events and salinity intrusion, amongst others (*robust evidence, good agreement*). These impacts will compound existing problems like disease and poor management practices.

As with fisheries, community- and ecosystem-based adaptation responses, within an integrated coastal zone management framework, is considered necessary to cope with anticipated challenges (Ahmed and Diana, 2015a). Where in-situ adaptation is not possible, translocation and polyculture (Ahmed and Diana, 2015b; Bunting et al., 2017) have been suggested as appropriate responses (*robust evidence, good agreement*), but as with the West African fisheries response, this would suit commercial rather than subsistence interests. Policy, economic, knowledge and other types of support are required to build socio-ecological resilience of vulnerable coastal communities (Harkes et al., 2015; Bunting et al., 2017; Rodríguez-Rodríguez and Bande Ramudo, 2017), which requires a deep understanding of the nature of stressors and a commitment for collective action (Galappaththi et al., 2017). Climate compatible development (CCD), incorporating mitigation and adaptation activities within a development framework, is considered a useful framework for Sri Lankan shrimp aquaculture (Harkes et al., 2015).

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

5.4.2.4.5 Partnerships, agreements, incentives, behavioural changes, transboundary considerations, policy and legal interventions, incentives, subsidies and offsetting

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

In coastal communities, there is often consensus of the importance of cooperation in tackling climate change (Elrick-Barr et al., 2016), but adaptation progress is hampered by lack of knowledge (Nanlohy et al., 2015), whilst factors like home ownership and a general future planning ability are important (Elrick-Barr et al., 2016). Another important factor is the level of trust communities have in their local government governance actions, causing (Massuanganhe et al., 2015; Wynveen and Sutton, 2015) to conclude that relationship-building actions are an important part of an adaptation response.

In a survey to understand community perception of financing adaptation actions, British coastal communities displayed a high level of resistance to the implementation of a coastal defence tax (Alderson-Day et al., 2015). Local collective action, using legal arguments around the right to protection from environmental change, may be a useful mechanism of securing state support for adaptation (Karlsson and Hovelsrud, 2015).

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

5.4.2.4.6 Effectiveness, scalability, enabling and disabling factors, cost/ benefits, monitoring, reporting, awareness and education, compatibility, conflict, trade-offs and co-benefits

AR5 provided an overview of good adaptation practice that included raising awareness and early warning, using the best available science, early and decisive action, developing partnerships and guidelines and the removal of barriers (Wong et al., 2014a). In a survey of 401 local governments with more than one million inhabitants globally, however (Araos et al., 2016) found that only 15% of cities reported adaptation initiatives and 18% planning towards adaptation policy. These authors concluded that leadership within government was more important for driving adaptation than wealth levels and institutional barriers. In an analysis of Australian coastal local government adaptation programmes in 2015, Bradley et al. (2015) found that few councils had progressed beyond seeking to understand the problem, and those that had, focussed almost exclusively on sea level rise (omitting warming and acidification). In their study, capacity was not found to be a predictor of adaptation action.

In general, governmental organisations experiencing an extreme event are likely to engage with adaptation more readily and in a more collaborative manner where dependencies upon external agencies are high Oswald Beiler et al. (2016). The same applies for individuals and communities, but modestly so, and likeliness to engage with adaptation declines with time following an event (Feltman et al., 2017; Ray et al., 2017). Barriers to adaptation include cultural differences (and distrust) between local knowledge and science, inappropriate governance structures and a lack of political will and resources (Kuruppu and Willie, 2015; Wynveen and Sutton, 2015; Cvitanovic et al., 2016; Doherty et al., 2016; Esteban et al., 2017). In a review of African and Caribbean least developed countries and small island developing states, (Kuruppu and Willie, 2015) found the most common barriers to engaging with adaptation to be governance, technical, cognitive and cultural Elias and Omojola (2015).

Technology for environmental monitoring, for examples using drones (Clark, 2017) and web-based coastal information systems (Mayerle et al., 2016; Newell and Canessa, 2017) promise to improve the local 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 should proceed on the basis of a set of general principals of best practice (Sheaves et al., 2016b; Thorne et al., 2017).

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

5.4.2.4.7 Consideration of gender, health and other socio-economic factors in society

In the sections above, it is clear that climate change will impact the most vulnerable communities, and members therein, including those drawing their livelihoods from subsistence activities, women, children, the aged and those in poor health (Barbier; Dunlop et al., 2016; Lohmann, 2016; McNeeley et al., 2017; Scandurra et al., 2018). 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) (Cross Chapter Box 3).

Besides inclusion in adaptation programmes, non-climate stressors will need to be addressed, including rural poverty levels (Jurjonas and Seekamp, 2015), job opportunities and aging communities. In Canada, this includes the promotion of 'age friendliness' programmes (Krawchenko et al.). In East Africa, self-organisation through negotiated social practices were deemed to be suitable context-specific local responses to climate change(Kithiia, 2015). There is substantial evidence underlining the importance of social capital for increasing resilience in communities (Petzold and Ratter, 2015; Salik et al., 2015; Chen and Ganapin; Sakakibara, 2017; Triyanti et al., 2017) (*robust evidence, high agreement*).

5.4.2.5 Summary

There has been a substantial amount of literature focussed on coastal and oceanic adaptation since the Fifth Assessment Report. Socio-institutional adaptation responses are the more numerous of the three types of adaptation responses assessed in this chapter. There is broad agreement that hard engineering responses are optimally supported by EBA approaches, and both approaches should be augmented by socio-institutional approaches for adaptation (*robust evidence, high confidence*) (Nicholls et al., 2015; Peirson et al., 2015; Sánchez-Arcilla et al., 2016; van der Nat et al., 2016; Francesch-Huidobro et al., 2017; Khamis et al., 2017). In planning adaptation responses, awareness-raising and stakeholder engagement processes are important for buy-in and ownership of responses (*robust evidence, high confidence*) as is institutional capacity within local government organisations, whose importance in coastal adaptation initiatives has been emphasised in the recent literature (*robust evidence, high confidence*).

5.4.3 Trade-offs in Ocean Economies

5.4.3.1 Economic Sectors

While many economic sectors are impacted by climate change (Section 5.3.2), environmental policies that aim to mitigate and adapt these impacts will also have effects, positive or negative, on these sectors.

5.4.3.1.1 Tourism

Tropical tourism

Tourism is a major economic activity for many countries, especially the SIDS. It as a source or revenue for many people and an entry of foreign currency (*high agreement*). Coral reefs are amongst the most attractive ecosystems in the world. Climate change and ocean acidification will affect negatively coral reefs causing economic difficulties in the impacted countries (Hilmi et al., 2018) (*high confidence*).

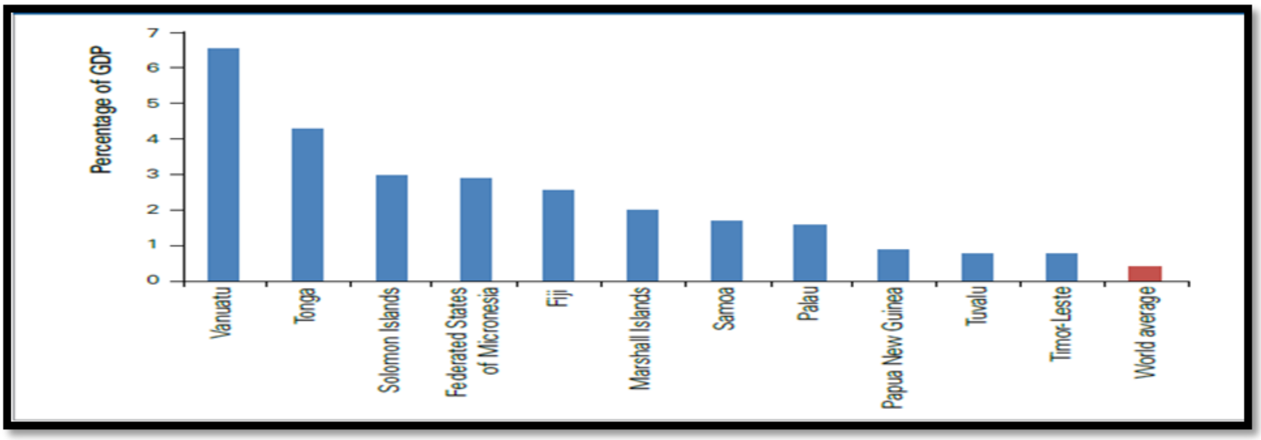


Figure 5.26: Worst affected SIDS by the consequences of climate change by share GDP: Pacific Region (Betzold, 2015).

The average Annual Losses as a percentage of GDP is much higher in SIDS compared with the global average as shown by Chart above. The total value of infrastructure, buildings, and cash crops considered at some level of risk in the Pacific is estimated at over US$111 billion. In the Pacific Region, the asset replacement cost is on average 4 times greater than the GDP (in 2013), for some countries such as Timor-Leste the ratio of asset replacement cost is 14 times their GDP (Betzold, 2015).

Another aspect should be stressed: coral reefs are certainly very attractive to tourists but at the same time, an overuse of the reef could lead to its degradation (Schmutter et al., 2017) through: the damage due to direct contact such as walking, touching, kicking, standing, or gear contact, the breakage from boat anchors, humans' behaviour towards reef creatures, water land-based pollution and trash, invasive species (*high confidence*). Those impacts are exacerbated in SIDS because of the increasing demography which carries an increasing pressure on the reef resources mainly used for self-consumption and because of the disappearance of traditional ways of management of the reef resources (such as taboos and religious beliefs demonstrated as informal no-take zones even temporary) which were quite effective (Weir and Pittock, 2017) (*low confidence*). As they are considered as commons, coral reefs necessitate specific management tools (Weir and Pittock, 2017).

That is why if some adaptation policies such as reefs restoration are implemented, the touristic activities may increase (*medium confidence*) and the positive and negative economic impacts as well (*high confidence*). The Marine Protected Areas and Integrated Coastal Zone Management will have positive economic impacts in terms of biodiversity and touristic attractiveness (*low confidence*), but they will decrease the fish catches in the short term (*low confidence*).

Polar tourism

Because of climate change, polar regions are evolving rapidly (*high agreement*). Tourism in this area increased because tourists want to see the ice before it melts and the species before they disappear: 'last chance tourism'. Polar tourism was reserved for wealthy elite and is now opening for mass tourism (Dehoorne et al., 2012) (*high confidence*). The increase of tourism activity can have positive and negative socio-economic impacts.

The increasing of tourism has a positive impact on the economy. It is an opportunity to create significant source of revenue, jobs, personal income and public finance (*medium confidence*). The employment generates by this tourism, in the Arctic, can be of three types: direct, in which jobs are created as a result of tourist expenditure and directly support tourist activity; indirect, wherein jobs are created within the tourism supply sector but not as a direct result of tourism activity; and induced, whereby jobs are created as a result of tourism expenditure as locals/residents spend money earned from tourism (Enzenbacher, 2011). Tourism increases accommodation and food services demand, thus the GDP (growth domestic product), the growth of tourism will also increase the tax collection. Unfortunately, the economic impacts of tourism can also be negative (*medium agreement*). Indeed, some cities, ports or communities can become economically or socio-economically depend of tourism (Etienne, 2005). This dependence might rapidly become vulnerability when there is a global economic crisis and the number of arrivals decreases. Furthermore, collective voyage experiences can maintain a stereotyped comportment. But the stereotype has not just a negative impact, although it is potentially discriminatory, he has a regulatory function for tourism in is relation on otherness (Kohler, 2012). Communities may experience several negative socio-cultural impacts from tourism (*medium confidence*). Local communities may engage in so called 'staged authenticity' to adapt to tourists' demands. Fast food and hotel chains enter into local communities, as tourists often demand standardized services for accommodation, food, and beverage. Economic inequality between locals and tourists becomes more apparent and may also increase within the local community. They can be victim of invasion and have the impression of loosening privacy. Their culture is merchandised and give an inauthentic image. Indeed, some assert themselves culturally just to respond at the touristic demands and other present an image which correspond to the touristic imaginary hiding their true culture (Hébert, 2008). Traditional cultures may also change as a result of tourism (*medium confidence*).

The increase of tourism can also have an environmental impact (*high confidence*). In Antarctica for example, tourists can introduce non-native species, invasive species taken there on ships, attached as seeds to boots and clothing. The effects of non-native species on the ecosystems are largely unknown. Then, tourists can have an impact on breeding birds, just because of their presence. Finally, tourists can cause erosion and disturbance on the fragile Antarctic environment because visits are located on the same sites. So, greater is the numbers of tourists, greatest are the impacts, because overcapacity can make substantial and irreversible consequence on this fragile environment (Kariminia et al., 2013). Visitors come to see the particular environment of polar regions threatened by climate change, but by doing so, they increase anthropogenic threats on this environment increasing its disappearance (*high confidence*).

Mitigation policies will have positive effects on the ecosystems of polar oceans because of the reduction of CO_2 emissions (*medium confidence*). But if they improve the biodiversity situation in the Arctic and Antarctica (*low confidence*), it may impact negatively their economies because they will be more attractive for tourists (*low confidence*). Adaptation policies, like community-based management, will be necessary to decrease the negative impacts of climate change on the economies and ecosystem services of those regions.

5.4.3.1.2 Sustainable maritime transport/shipping industry

Shipping is not directly related to the health of ocean, meaning that even if the ocean is polluted or warmed, boats can navigate. But by their passage, they increase the environmental disturbances on the ocean (*high agreement*).

About cruises transporting persons, the environmental impact of shipping includes greenhouse release of oil and chemicals through accidental spills and operational discharges, transfer of invasive alien species through ballast water and on ship hulls, release of biocides, chemicals used in anti-fouling paints, dumping of wastes (garbage and sewage), air pollution (emissions of sulphur dioxide, nitrogen oxides, and carbon dioxide), physical and other damage caused by dropping anchors, noise and wave disturbances, and the striking of whales and other marine mammals (*high confidence*).

Merchandises transportation increases also the threats on the ocean because commercial shipping mainly uses HFO, emitting significant amounts of sulphur, nitrogen, metals, hydrocarbons, organic compounds and black carbon and fly ash to the atmosphere during combustion (Eyring et al., 2005; Turner et al., 2017). The use of new technology, like scrubbers, might reduce the impact of the shipping industry and the increase of maritime traffic (*low confidence*).

The International Maritime Organization (IMO) estimates that carbon dioxide emissions from shipping were equal to 2.7% of the global human-made emissions in 2007 and expects these emissions to rise by as much as 2 to 3 times by 2050 if no action is taken.

The impacts of climate change on the marine life and maritime industries are complex. Climate change could affect the shipping industry directly, through (a) increased risk from weather conditions and changing global trade patterns, or (b) indirectly through strict emission controls of greenhouse gases and pollutants. Besides coping with these impacts, in recent years, more shipping companies have shown increased commitment to the reduction of their environmental impact and a sustainable development of the shipping industry (*medium agreement*).

Increased risk from weather conditions and changing global trade patterns

Direct effects of climate change which are relevant for maritime transport include ocean warming and related melting of sea ice, sea level rise and floodings in coastal areas. Simultaneously, the frequency and intensity of hurricanes and cold-season storms are expected to increase (Humphrey, 2008). Naturally, these effects of climatic changes are different for different regions. This applies also for the impact of climate change effects on transportation infrastructure and on global transport patterns. Sea level rise, storms and associated floodings will cause direct costs due to infrastructure damages in ports and other coastal facilities and costs for measures for flood-defenses (Koetse and Rietveld, 2009; Ng et al., 2018). The damage caused by a changing climate may present more of a threat for developing than for developed economies. This needs to be considered for investments "to build resilient infrastructure" (SDG 9) for maritime industry of developing countries and Small Island Developing States whose economies depend on ports (Becker et al., 2018). Additionally, short-notice port closures, delays, detours and trip cancellations could create indirect costs for maritime transportation (Koetse and Rietveld, 2009; Jaroszweski et al., 2010). Global warming and changing weather patterns may have a substantial impact on global trade and transport pathways. Changes in passenger transport are already observed due to shifting tourism patterns temporally (shift in holiday periods) and spatially (shift in holiday destinations) (Koetse and Rietveld, 2009). Agricultural production and the associated transportation is projected to relocate as countries at higher longitudes may become more suited for food production. The reduction in sea ice in the Arctic Ocean during summer opens up the possibility for sea transport on the Northwest or Northeast Passage during at least several months per year (Koetse and Rietveld, 2009). Both routes may provide opportunities for more efficient transport between North America, Europe, Russia and China for fleets with established Arctic equipment and may open up access to known natural resources which were so far covered by ice (Ng et al., 2018). However, whether the Arctic shipping routes may or may not be a realistic alternative will depend on political regulations and economic aspects such as infrastructure along and reliability of the routes (Guy and Lasserre, 2016). Polar regions are especially exposed to climate change as the rates of change (e.g., ocean warming and acidification) are exceeding the global average putting pressure on polar marine communities and biogeochemical cycles (Wassmann et al., 2010). In order to minimize ship pollution of this pristine environment, special measures were implemented under the IMO Polar Code (IMO 1 January 2017) for ships operating in both Antarctic and Arctic waters.

In summary, climate change-related weather effects and changing transport patterns are likely to change global shipping routes as shipping industry will try to adapt to increasing maritime transportation costs. However, future socio-economic developments need to be taken into account in order to fully assess how the shipping industry will be affected by climate change (Jaroszweski et al., 2010).

Strict emission controls of greenhouse gases and pollutants

Besides CO_2, the combustion of low-grade, high-sulphur content fuel in ships emits significant amounts of other greenhouse gases and pollutants, such as sulphur oxides, nitrogen oxides, aerosols (containing organic carbon, black carbon, polycyclic hydrocarbons (PAH) and heavy metals) to the atmosphere (Cooper and Gustafsson, 2004; Eyring et al., 2005). These have wide-ranging consequences for the atmosphere, the marine environment and human health (Corbett et al., 2007). Some compounds are deposited relatively close to the source vessel and become dissolved or suspended in the surface ocean.

Strict CO_2 emission controls in maritime transport are expected in future to be comparable to recent regulatory developments in aviation or land-based transport. In order to reduce greenhouse gas emissions by ships, two mandatory energy efficiency measures, known as the Energy Efficiency Design Index (EEDI) for new ships and the Ship Energy Efficiency Management Plan (SEEMP) for existing ships, were introduced by the UN International Maritime Organization (IMO) in 2011 (Resolution MEPC 203(62).). Currently, a CO_2 reduction strategy is being developed at the IMO which will adopted initially in 2018 and revised by 2023 (MEPC 70, 2016). In the meanwhile, rising oil prices increase shipping costs significantly and drive economical decisions. Until technological measures are in place, slow steaming may be a preferred option by ship operators in order to save costs but also to reduce CO_2 emissions (Corbett et al., 2009). During recent years, there have been large advances in technologies and operational practices with high potential to cut CO_2 emissions and fuel consumption by as much as 84%. Those measures include the use of alternative fuels, such as LNG and biofuels, developing better power and propulsion systems, more efficient ship hull design, improved equipment, and advanced weather routing ((Bouman et al., 2017) and references therein).

However, no single measure is itself sufficient to achieve significant emission reductions in a cost effective manner within the next decades Bouman et al. (2017).

Overall, a growing number of shipping companies are intending to drastically reduce CO_2 emissions from ships over the next decade hence contributing to “combat climate change and its impacts” (SDG 13) and “ensure sustainable consumption and production” (SDG 12). In addition, the shipping industry aims to reduce shipping-related pollution in the wider oceans but also in ports and coastal regions where ship-derived emissions may contribute to tens of thousands of cases of premature mortality every year (Corbett et al., 2007; Winebrake et al., 2009; Sofiev et al., 2018). Additionally, several measures were implemented by the IMO through the MARPOL Convention and other treaties to “conserve and sustainably use the marine ecosystems and resources” (SDG 14). This includes well-known ship-related measures to prevent atmospheric and seawater pollution from ships (by oil, chemicals, harmful substances, sewage, garbage) and new technological advances to reduce underwater noise, avoid transfer of harmful invasive species by ballast water, and recycle ships in an environmentally sound way. Ship-derived high levels of nutrients and pollutants directly entering the upper mixed layer may affect biological carbon and nitrogen cycling in the oligotrophic open ocean (Bonnet et al., 2005; Pulido-Villena et al., 2008) and coastal areas and semi-enclosed basins which are already overloaded by emissions and human activities (Svendsen et al., 2015). In order to “prevent and significantly reduce marine pollution of all kinds, […] including marine debris and nutrient pollution” (SDG 14), it is necessary to further limit greenhouse gas and particulate matter emissions from ships’ exhaust gases and oblige ships to discharge sewage and other waste into adequate reception facilities or alternatively at sea only after treatment with advanced onboard treatment facilities (HELCOM, 2017).

By 2020, IMO regulations will reduce the permitted sulphur content in fuel globally to 0.5% - and less in designated Emissions Control Areas (Resolution MEPC 280). This is expected to have a major beneficial impact on the environment and on human health supporting SDG 3 “to ensure healthy lives and promote well-being for all”. The comparison of two possible abatement options to comply with stricter fuel emission regulations indicate that switching to low-sulphur fuel seems to be an attractive option for smaller vessels, while the installation of exhaust gas scrubbers may be the favored option for larger vessels (Lindstad et al., 2017). The use of new emission reduction technologies, such as scrubbers, will benefit the environment by significantly reducing the release of pollutants to the atmosphere. However, there is incomplete understanding of the impact of e.g., scrubber wash water discharge on marine chemistry, biodiversity and biogeochemical processes (Endres et al., 2018). It has been estimated that emissions of SO_x and NO_x in the heavily trafficked waters of the open ocean could lead to significant regional pH reductions of the same order of magnitude as anthropogenic CO_2-driven acidification, in particular if the effects of acidic scrubber wash water discharge are also included (Stips et al., 2016). In the global shipping network, new strategies were established such as the management of ship routing and establishment of Special Areas and Particularly Sensitive Sea Areas to avoid collisions between ships and marine mammals and to protect vulnerable marine ecosystems. Despite existing guidelines for levels of monitoring and compliance of ship pollution, there is still the risk for acidification, eutrophication and accumulation of pollutants in the marine environment (Endres et al., 2018).

Because of the impacts of sea level rise (SLR) on the coastline and the built infrastructures like harbours, the shipping activity will be heavily impacted by climate change (*low confidence*). Concerning adaptation policies, the coastal protection is essential to face SLR (*high confidence*). This protection can be natural (coral reefs) or human (walls built in the sea). The investment may be very expensive if corals are threatened or if higher walls are needed because of SLR (*high agreement*).

5.4.3.2 Implications for Sustainable Development and Community Livelihood

5.4.3.2.1 Irreversible changes and compensation

The marine/coastal interface is a site of great environmental contention, as humans are drawn to these areas by the ecosystem services provided (such as aesthetic appeal) and at the same time, engage in activities, like construction, which can impact negatively on such ecosystems. There is a need to control development and its activities in order to reduce environmental degradation (*high agreement*).

At the global level, there is a recognition of the issues associated with coastal development, and this has led to a series of international conferences, conventions, treaties etc. aimed at managing the world's resources in order to ensure continued economic development, starting with the Stockholm Conference in 1987 (CGDD and CETE, 2012).

Environmental Impact Assessment (EIA) is a tool used globally to identify the environmental and social impacts of projects, prior to final decision-making. Many national governments have enacted legislation making it mandatory for EIAs to be carried out prior to activities that may affect the natural environment (UNEP, 2002). The EIA process thus underpins decisions on whether the environmental damage arising from proposed activities are acceptable or unacceptable, and supports identification of mitigation measures. However, EIAs rarely address biodiversity compensation, a tool that may be used to balance negative impacts on biodiversity through positive compensatory measures. When the goal is to achieve no net loss of biodiversity, this is called a biodiversity offset (Levrel et al., 2012). In neglecting this aspect, many governments are unable to ensure that the impact of developments approved in EIAs is adequately compensated for, and thus the primary objective for the EIA is not being realized (Pioch, 2015).

The mitigation hierarchy, Prevent Reduce Offset (PRO) or Avoid, Minimise, Restore/Compensate and Offset is recommended in all Environmental Impact Assessment (BBOP and UNEP, 2010). Compensation measures are to be considered only after all possibilities for impact avoidance and minimisation have been explored.

The primary aim of compensation measures is to offset compositional (number of species), structural (community structure) and functional (ecological relationships and ecosystem services) loss in the impacted ecosystem. Habitat restoration, areas for reproduction, growth and feeding, as well as corridors that enable species to complete their biological cycles must be ensured (Levrel et al., 2012).

Under most types of legislation or policy, this compensation must, as far as possible, be carried out near the impacted site, be scaled according to the project's residual effects and ensure compensation for losses, so that the results of the environmental operation lead to a zero (equivalent) or a positive balance (Dunford et al., 2004; Pinault et al., 2017).

“The restoration of coral reef ecosystems is still in the experimental phase, but today it is possible, under certain conditions, to restore on average 65% of degraded coral reef habitats and salt marshes, and approximately 38% of seagrass beds in tropical areas.” (Bayraktarov et al., 2016). With regards to mitigation hierarchy in coral reefs, it has been claimed that “trustworthy assessments should target the expansion of reef restoration activities and the search for novel methodologies for reef restoration” (Rinkevich, 2017), a statement based on current studies on active reef restoration approaches (including ecological engineering tactics), primarily those based on the reef gardening approach (Rinkevich, 2015a).

5.4.3.2.2 Marine natural capital implications

Empirical studies of marine natural capital and human impacts emphasize that there are significant interactions between human activities and marine environment (*high agreement*). For instance, mean temperature increases due to climate change have led to tremendous decline in glaciers, ice sheets, and Arctic sea ice (IPCC, 2014). Most seas including those which surround Europe suffer from ocean acidification and increasing ocean temperatures. (Wu and Tsai, 2014) argue that human capital is utilized in order to adapt to the changes in marine natural capital due to the dependence of societies on marine social-ecological systems for their livelihoods. Their case study of Penghu Archipelago in Taiwan proves that human capital may not be sufficient to sustain marine resources. Instead, they put forward the need to devote the society’s resources to the formation of social capital as to develop the institutions that sustainably utilize physical and human capital. Another study by the Marine Ecosystems Research Program (MERP) reports a higher impact of rising temperature for cold water species such as grey seals, cod, haddock, whiting and herring in the West Coast of Scotland. These resources are all expected to decline by 2100 under the worst case climate change scenario of the MERP modelling. Similarly, Mueter and Litzow (2008) and Spencer (2008) find that marine fishes and invertebrates have a tendency to relocate at higher latitudes and/or in deeper seas (*medium confidence*). These findings are in accordance with the patterns observed in the West Coast of Scotland and the North Sea. Reviewing the impacts of climate change on marine organisms and ecosystems, Brierley and Kingsford (2009) confirm that ice loss leads to habitat loss for many species including polar bears, which are suffering as early ice melt reduces the time they can hunt at sea (*high confidence*). They further suggest that several north Atlantic fish stocks declined due to climate change.

Besides, overexploitation of fish stocks and the gradual decline in the amount of fish communities (e.g., in the Black Sea) lead to changes in the structure and functioning of marine ecosystems Brierley and Kingsford (2009).

Rising CO_2 concentration in the oceans and ocean warming have other direct consequences as well. (Doney et al., 2012) argue that these include, e.g., coral bleaching and loss of coral reefs, and consequences for society's dependence on the ocean for wild-caught and farmed food, recreation, nutrient cycling, waste processing, protection from natural hazards, climate regulation, and other services (*high confidence*). The authors exemplify that "climate-induced sea level rise could put added pressure on coastal infrastructure and thus the health and safety of human communities" (pg. 11). Eissa and Zaki (2011) put forward that, as a result of climate change, coastal regions might face "multiple burdens associated with globalization of aquatic animal production / trade" (pg. 256) (*medium confidence*).

Recently, Oceania Ecosystem Services Forum (2017) held in Brisbane, Australia focused on the need to integrate policies for climate change mitigation and adaptation in the natural capital accounting framework provided by the SEEA. The Wealth Accounting and the Valuation of Ecosystem Services (WAVES) Partnership identifies this need and tries to address climate change in natural capital accounting. Examples may include measuring the benefits of retaining mangroves for storm surge protection and estimating the impact of climate change on cultural and recreational services (and hence tourism revenues) Vardon (2017). Attempts to sustain the value of marine natural capital continue. Technological advancements, innovations as well as establishment of marine protected areas offer considerable potentials in order to maintain existing resources (*medium confidence*).

5.4.3.2.3 The role of local environmental knowledge in adapting to climate change.

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

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

5.4.4 Governance Across All Scales

5.4.4.1 Global and Regional Responses

This part critically examines and evaluate the existing international and regional governance system for climate change with a particular focus to ocean governance. It will examine the international regulatory and institutional framework. Global and regional governance structure to response to ocean acidification and ocean warming and their impact on marine ecosystem and dependent community is highly fragmented.

Legal, policy and institutional response is shared by the global and regional institutions developed for a number distinct but inter-related fields including inter alia global and regional governance regimes for climate change, ocean and marine environment, fisheries and the environment generally (see Table 5.10).

Current international legal and governance regime does not adequately address the issues of ocean warming, acidification and deoxygenation (*high confidence).* Significant reform is needed in many areas of international law including climate change, ocean and the marine environment, environment generally, and fisheries (*high confidence*). It must be acknowledged that while adoption of these legal and governance regimes, climate change or carbon dioxide emissions related changes of the ocean, particularly, the interaction of physical, chemical, biological properties of the ecosystems with risks and vulnerabilities of the dependent communities was either not considered or did not get proper attention (*high confidence*). There is need for a comprehensive review of exiting international legal regimes for above mentioned areas considering the changing ocean and its ecosystems and their impact of dependent communities (*high confidence*).

International climate change regime is naturally the most relevant global governance apparatus for combating ocean acidification and ocean warming. However, how far current climate change legal framework is adequate for dealing with the issue of the ocean acidification is questionable (Stephens, 2015). Ocean acidification and ocean warming did not get full attention during the negotiation of the United Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol in the 1990s. These legal instruments are mainly for combating global warming with a clear atmospheric focus and ocean did not get proper attention (Galland et al., 2012). Unlike Kyoto Protocol, the ocean is explicitly mentioned in the Paris Agreement's preamble and there may be implications from the Paris Agreement for ocean (Magnan et al., 2016). However, a study on 161 nationally determined contribution (NDCs) submitted where national pledges for climate change mitigation and adaptation show that there are 'gaps between scientific [understanding] and government attention, including on ocean deoxygenation, which is barely mentioned' (Gallo et al., 2017).

Widely recognized as the constitution of ocean, the United Nations Convention on the Law of the Sea (UNCLOS), elaborates the jurisdiction, duties and obligation of coastal, flag and port states in different maritime zones including internal waters, territorial sea, exclusive economic zone, continental shelf, and high seas. Two aspects of UNCLOS are specifically important in the context of a changing ocean: UNCLOS's provisions relating to prevention of marine pollution and provisions relating to the conservation and sustainable utilization of marine living resources. UNCLOS imposes a number of obligation for the conservation and sustainable utilization of marine living resources within national jurisdiction (UNCLOS 1982, Articles 56, 6, 62, 63, 64, 65 and 67).

Pollution of the marine environment has been defined by UNCLOS as: 'the introduction by man, directly or indirectly, of substances or energy into the marine environment, including estuaries, which or is likely to result in such deleterious effects as harm to living resources and marine life, hazards to human health, hindrance to marine activities, including fishing and other legitimate uses of the sea, impairment of quality for use of sea water and reduction of amenities' (UNCLOS 1982, Article 1). UNCLOS imposes obligations on state parties to take action for combating six main sources of ocean pollution including land-based and coastal activities, continental shelf drilling, seabed mining, ocean dumping, vessel-source pollution, and pollution from or through the atmosphere. It has been suggested that tools and techniques in UNCLOS may need adjustment in response to the emerging challenges created by climate change for ocean (Redgwell, 2012). However, success of the umbrella regulatory framework of UNCLOS is heavily depending on further development, modification and implementation of detail regulations by relevant international, regional and national institutions (Karim, 2015).

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

For example, the issue of ocean acidification has been considered within the framework of the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) and the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) (Herr et al., 2014). A number of sectoral international environmental conventions are relevant directly or indirectly in respect of changing ocean. For example, Convention Concerning the Protection of the World Cultural and Natural Heritage may play a role in respect of world heritage coral reefs. Most of the 29 world heritage listed coral reefs are facing severe heat stress (Heron, 2017) (*high confidence*). Changing ocean may also create serious problem in the governance of regional fisheries agreements (Brandt and Kronbak, 2010).

Existing ocean governance structure for the ocean is facing or will face multi-dimensional challenges because of the climate change (Galaz et al., 2012). Mainstreaming climate change issues in the global and regional ocean, environmental and fisheries governance structure is needed to address climate impacts.

Table 5.10: Existing international institutions and legal instruments of ocean governance.

Area of Governance	Major Institutions	Major Legal Instruments	Comments
Climate Change	United Nations UN Environment World Meteorological Organization (WMO) UNFCCC Secretariat Intergovernmental Panel on Climate Change (IPCC) Regional Organizations (EU, ASEAN, SAARC, OAS, AU, PIF etc.)	United Nations Framework Convention on Climate Change Kyoto Protocol Paris Agreement	Existing climate change related legal instruments do not adequately address the challenges the ocean and coastal areas will face. Nevertheless, their overall goal for reduction of emissions will be positive for mitigation of climate change impact on the ocean and coastal areas (Galland et al., 2012; Gallo et al., 2017). Greater emphasis to the ocean related issues in the mainstream climate change legal and governance regime is needed.
Ocean and Marine Environment	United Nations Division for Ocean Affairs and the Law of the Sea, OLA, United Nations. International Maritime Organization (IMO) UN Environment Regional Seas Programmes /Organizations (including Antarctic, Arctic, Baltic, Black Sea, Caspian, Eastern Africa, East Asian Seas, Mediterranean, North-East Atlantic, North-East Pacific, Northwest Pacific, Pacific, Red Sea and Gulf of Aden, ROPME Sea Area, South Asian Seas, South-East Pacific, Western Africa and Wider Caribbean regional seas programmes) Regional Organizations (EU, ASEAN, SAARC, OAS, AU, PIF etc.)	United Nations Convention on the Law of the Sea (UNCLOS) International Convention for the Prevention of Pollution from Ships, (MARPOL) and other IMO Conventions Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention) Regional Seas Conventions: Convention for the Protection and Development of the Marine Environment in the Wider Caribbean Region (Cartagena Convention) and its associated protocols. Convention for the protection of the Mediterranean Sea against Pollution (Barcelona Convention) and its associated protocols. Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region (Nairobi Convention) and its associated protocols. Convention for Co-operation in the Protection and Development of the Marine and Coastal environment of the West and Central African Region (Abidjan Convention) and its associated protocols. Framework Convention for the Protection of the Marine Environment of the Caspian Sea (Tehran Convention) and its associated protocols The Convention on the Protection of the Black Sea against Pollution (Bucharest Convention) and its protocols Convention for Cooperation in the Protection and Sustainable	Marine environment related international and regional legal governance structure is slowly considering the climate change or carbon emissions related issues. For example, the MARPOL convention was amended in 2011 to incorporate some new provision for reduction of emissions from the international shipping industry. However, these new provisions are grossly inadequate (Karim, 2015). The London Protocol was amended to regulate carbon capture and storage in sub-seabed geological formations for permanent isolation and marine geoengineering engineering activities in 2006 and 2013 respectively. Regional marine environmental legal governance systems yet to incorporate adequate provisions for emerging challenges of climate change for ocean and coastal areas. Just a few of these instruments now deal with the emerging issues of climate change and ocean governance.

		Development of the Marine and Coastal Environment of the North-East Pacific (Antigua Convention) Regional Convention for the Conservation of the Red Sea and Gulf of Aden Environment (Jeddah Convention) and its associated protocols. Regional Convention for Cooperation on the Protection of the Marine Environment from Pollution (Kuwait Convention) and its associated protocols. Convention for the Protection of the Marine Environment and Coastal Zones of the South-East Pacific (Lima Convention) and its associated protocols. Convention for the Protection of the Natural Resources and Environment of the South Pacific region (the Noumea Convention and its associated protocols. Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR). Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki Convention) Convention for the Protection of the Marine Environment of the North-East Atlantic of 1992 (OSPAR Convention)	
Environment generally	United Nations UN Environment Different Convention Secretariats and other treaty bodies World Health Organization (WHO) United Nations Educational, Scientific and Cultural Organization (UNESCO) Intergovernmental science-policy Platform on Biodiversity and	Convention on Biological Diversity (CBD) and its associated protocols. Convention for the Protection of the Ozone Layer (Vienna Convention) and its associated protocols. Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention) Convention on Wetlands of International Importance especially as Waterfowl Habitat (Ramsar Convention)	Major international environmental convention systems are increasingly 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 synchronisation (Proelss and Krivickaite, 2009).

	Ecosystem Services (IPBES) Regional Organizations (EU, ASEAN, SAARC, OAS, AU, PIF etc.)	Convention on the Conservation of Migratory Species of Wild Animals (CMS or the Bonn Convention)	
Fisheries	United Nations Food and Agriculture Organization Committee on Fisheries (COFI) International Commission for the Conservation of Atlantic Tunas (ICCAT) Indian Ocean Tuna Commission (IOTC) Western and Central Pacific Fisheries Commission (WCPFC) Inter-American Tropical Tuna Commission (IATTC) Agreement on the International Dolphin Conservation Programme (AIDCP) Commission for the Conservation of Southern Bluefin Tuna (CCSBT) Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) North-East Atlantic Fisheries Commission (NEAFC) Northwest Atlantic Fisheries Organization (NAFO) North Atlantic Salmon Conservation Organisation (NASCO) South-East Atlantic Fisheries Organisation (SEAFO) South Indian Ocean Fisheries Agreement (SIOFA) South Pacific Regional Fisheries Management Organisation (SPRFMO) General Fisheries Commission for the Mediterranean (GFCM) Convention on the Conservation and Management of Pollock Resources in the Central Bering Sea (CCBSP) Western Central Atlantic Fisheries Commission (WECAFC) Fisheries Committee for the Eastern Central Atlantic (CECAF)	The United Nations Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks Regional fisheries Agreements: Agreement for the establishment of the General Fisheries Commission for the Mediterranean Agreement for the Establishment of the Indian Ocean Tuna Commission Agreement for the Establishment of the International Organisation for the Development of Fisheries in Eastern and Central Europe Agreement on straddling stocks and highly migratory fish stocks Agreement on the International Dolphin Conservation Program Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas Convention for the Conservation of Salmon in the North Atlantic Ocean Convention for the Conservation of Southern Bluefin Tuna Convention for the Establishment of an Inter-American Tropical Tuna Commission (IATTC) Convention on future multilateral cooperation in North-East Atlantic fisheries (NEAFC) Convention on Future Multilateral Cooperation in the Northwest Atlantic Fisheries (NAFO) Convention on the Conservation and also Management of High Seas Fishery Resources in the South Pacific Sea Convention on the Conservation and Management of Fishery Resources in the South East Atlantic Ocean Convention on the Conservation and Management of High	Regional fisheries management systems will face many tithing challenges because of the changing ocean (Brandt and Kronbak, 2010). Mjaor refrom in the regional fisheries management systems may be needed to address the emerging challenges (Brooks et al., 2013).

		Migratory Fish Stocks in the Western and Central Pacific Ocean Convention on the Conservation and Management of High Seas Fishery Resources in the South Pacific Sea Convention on the Conservation and Management of Pollock Resources in the Central Bering Sea Convention on the Conservation of the Living Resources of the South-East Atlantic International Convention for the Conservation of Atlantic Tunas Regional Convention on Fisheries Cooperation among African States Bordering the Atlantic Ocean Southern Indian Ocean Fisheries Agreement	

5.4.4.2 National Policy and Administration

[PLACEHOLDER FOR SECOND ORDER DRAFT]

In the 2015 United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement, national and sub-national actors are required to address climate change adaptation as per Parties' Nationally Determined Contributions. A key component of adaptation effort is good governance, at both the national and local government levels. This section assesses the performance of governance as a tool to combat climate change.

In producing a strategy and roadmap for coastal fisheries, within a participatory workshop in the Pacific, (Gourlie et al., 2017) concluded that robust and effective management policy, legislation and planning will be required for coastal fisheries. This required flexibility and scientific understanding to support management decisions. Similarly, key characteristics from South Asia identified by Vij et al. (2017) included institutional flexibility, adaptive nature, scalability and reflexivity.

Capacity, institutional structure and political will be ctrical factors. In a survey of 350 municipalities across five continents, analysing institutional governance structures in relation to climate change, over 70% of respondents indicated integration of adaptation and mitigation planning into long-range and sectoral plans Aylett (2015). The authors found that many organisations are divided along sector-based response pathways, which was considered to lack the internal institutional networks of governance required for effective adaptation. In a study of North Carolina coastal communities, under moderate levels of risk and uncertainty, (Bulla et al., 2017) concluded that moderate political ideology proponents were more likely to take climate adaptive actions than conservative. It is clear that, as with socio-insitutional adaptation, governance in local organisations is subject to a range of non-climate change based factors (*medium evidence, good agreement*).

5.4.4.3 Non-governmental Stakeholders

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

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

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

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

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

One of the main investment vehicles in marine conservation is the Entrepreneurial Marine Protected Area (EMPA) (Credit Suisse AG 2016). An EMPA is a management area that is primarily funded by a profit-bearing business model, typically associated with nature tourism. EMPAs belong to marine impact investments as they are designed to produce environmental and social impacts, and they primarily employ business models instead of grants to achieve those outcomes. While many terrestrial protected areas have private sector involvement in the region (IUCN & UNEP 2014), significantly fewer MPAs include the private sector. A potential obstacle in the further development of this tool are results showing that the management instruments employed in MPAs, in general, often fail to achieve their conservation objectives Miller and Russ (2014); (Gill et al., 2017).

5.4.4.4 Conflicting and Ineffective Governance

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

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

[START BOX5.5 HERE]

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

Ocean acidification is not mentioned in the Paris Agreement on climate change, and has not featured in other UNFCCC discussions. Is that because the topic is not considered to be part of climate change, or because it does not warrant distinct governance arrangements? If climate is strictly defined as the long-term descriptor of meteorological conditions, then changes in ocean chemistry are out of scope. However, physical changes in the ocean, such as sea level rise, are uncontroversially considered part of climate change, and the CO_2-driven change in ocean pH is an inevitable, although indirect, chemical consequence of human perturbations of atmospheric composition – the driver of all other changes to the climate system. The close association between ocean acidification and other components of climate change was recognised in IPCC AR4 and AR5, with Working Groups I and II (but not III) giving consideration to the ecological and economic impacts of ocean acidification that are already occurring, and that are projected to greatly increase under future emission scenarios.

Most literature on ocean acidification is relatively recent (Riebesell and Gattuso, 2014; Browman, 2016). Nevertheless, IPCC AR5 WG I considered that there was *high confidence* that anthropogenic CO_2 had already caused global surface ocean pH to emerge from natural variability, and that pH will continue to decrease with increasing CO_2 emissions. AR5 WG II considered there was *high confidence* that ocean acidification will increasingly affect marine organisms, ecosystems, and the goods and services they provide. Such assessments are confirmed in this report (Sections 5.2.1, 5.2.2, 5.2.3, 5.3). Concerns regarding ocean acidification have been raised by many international organisations, including the Convention on Biological Diversity (CBD) (CBD, 2016), the United Nations Convention on the Law of the Sea (UNCLOS), the United Nations Environment Programme (UNEP), the Intergovernmental Oceanographic Commission (IOC) of the United Nations Educational, Scientific and Cultural Organization (UNESCO), the World Meteorological Organization (WMO), and the International Atomic Energy Agency (IAEA); the interests of these and other bodies in ocean acidifcation are summarised in CBD (2014). Public awareness of ocean acidification is, however, low (e.g., only 14% of Europeans consider that they are ‘informed’ on the topic; (Buckley et al., 2017)) and the topic has been neglected to date from a regulatory perspective Fennel and VanderZwaag (2016). The most substantive governance outcome to date is UN Sustainable Development Goal 14.3 (Section 5.3.2), with its non-binding and relatively general commitment to “minimise and address the impacts of ocean acidification, including through enhanced scientific cooperation at all levels”.

Whilst a new UN mechanism specifically to address ocean acidification has been proposed, (Kim, 2012), there are strong pragmatic arguments favouring greater UNFCCC involvement (Harrould-Kolieb and Herr, 2012). Action to reduce CO_2 emissions will necessarily, to some degree, address ocean acidification, although there are topic-specific considerations. Additional governance attention would not only seem necessary with regard to pH-associated regime changes, thresholds or tipping-points (Hughes et al., 2013; Good et al., 2018), but also with regard to ocean acidification impacts in scenario-modelling for emission reductions (Steinacher et al., 2013), and the consequences of climate policy responses for ocean acidification. In particular, any leakage from sub-seafloor carbon storage is *very likely* to produce local pH impacts, similar to natural CO_2 vents (Blackford et al., 2014), whilst some CO_2 removal techniques (negative emissions) such as ocean fertilization, are *likely* to worsen acidification for the ocean as a whole (Cao and Caldeira, 2010; Williamson and Turley, 2012). Policy interventions involving solar radiation management might stabilise the climate, yet allow acidification to essentially continue unabated (*high confidence*) (Williamson and Turley, 2012; Keller et al., 2014).

Adaptation to climate change could also include a more integrated approach to reduce ocean acidification impacts (Sections 5.4.1, 5.4.2). Proposed actions for ocean acidification (Kelly et al., 2011; Billé et al., 2013; Strong et al., 2014; Albright et al., 2016) include reduction of pollution and other stressors (thereby strengthening resilience); water treatment, e.g., for high-value aquaculture; adapting human activities; and repairing damages. Seaweed cultivation and seagrass restoration may slow longterm changes at the local level, but can also exacerbate short-term variability (Sabine, 2018). These measures are generally applicable to relatively limited spatial scales: whilst they may succeed in 'buying time', their future effectiveness will decrease, unless underlying global drivers are also addressed (*high confidence*).

Well-coordinated and fit for purpose ocean acidification monitoring (Newton et al., 2015) is as an essential part of the governance response (*high confidence*), and is beginning to be used operationally by the oyster cultivation industry (Barton et al., 2015). Good progress has been made in characterising seasonal patterns (Sutton et al., 2016) and identifying the many other factors that can cause high pH variability in coastal waters (Duarte et al., 2013). In the US, the 2009 Federal Ocean Acidification Research and Monitoring Act (FOARAM) provided the basis for coordinated national monitoring of ocean acidification and its consequences (Interagency Working Group on Ocean Acidification, 2014). The FOARAM Act also required the development of adaptation strategies to conserve organisms and ecosystems vulnerable to ocean acidification, and the consideration of associated socio-economic impacts. However, there are no statutory criteria for permissible pH levels in the FOARAM Act, nor in the US Clean Water Act (regulated by the US Environmental Protection Agency), since standardised protocols for natural values and their variation have not been established. A similar situation occurs in Europe: measurement of ocean acidification conditions is encouraged by the European Union, but is not statutory – since pH change is considered to be a 'prevailing condition', outside national management control (EU, 2008).

[END BOX 5.5 HERE]

5.4.4.5 Policy framework and Voluntary Instruments

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

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

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

Hence, each year, the General Assembly continued in the omnibus resolutions to deal with this troubling phenomenon in order to make States more and more aware of it and of its dangerous consequences to the marine ecosystems (A/RES/63/111 para.99; A/RES/64/71, para.113; A/RES/65/37A, para.129; 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 165-170, A/RES/70/235 paras 172-177 ; A/RES/71/257 paras 185-190.).

With the Rio + 20 summit and the recommendations in the 'Future we want', the resolutions devoted more paragraphs and requested more action, effort and initiatives to further study, to minimize impacts as well as to enhance international cooperation.

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

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

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

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

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

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

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

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

To combat overfishing and particularly IUU fishing, several instruments were adopted under the auspices of FAO. They deal either with the respect of international conservation measures by fishing vessels like the *1993 Agreement on compliance with conservation and management measures*, or the increase control by port States to ensure that fishing vessels entering in a port did not violate the measures taken at the regional or sub regional level. In this case two major instruments can be mentioned: i) the 1995 *Agreement on straddling fish stocks and highly migratory fish stocks*, and ii) the most recent one, far reaching in the fight of IUU fishing that is the 2009 *Agreement on port state measures to prevent, deter and eliminate IUU fishing,* in force in 2016 (FAO, C2009/REP et Corr.1 et 3, Annexe E). It will enhance regional and international cooperation and block the flow of IUU caught fish into national and international markets.

5.5 Synthesis

[PLACEHOLDER FOR SECOND ORDER DRAFT: Text and figure to be included]

5.6 Key Uncertainties

[PLACEHOLDER FOR SECOND ORDER DRAFT: Text and figure(s) to be included]

[START FAQ5.1 HERE]

FAQ5.1: What are the key ocean ecosystem vulnerabilities to climate change?

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[END FAQ5.1 HERE]

[START FAQ5.2 HERE]

FAQ5.2: How does climate change impact the deep ocean?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ5.2 HERE]

[START FAQ5.3 HERE]

FAQ5.3: What is the role of the ocean and coastal ecosystems in capturing and storing anthropogenic carbon?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ5.3 HERE]

[START FAQ5.4 HERE]

FAQ5.4: How are fisheries affected by climate-related and other human-induced stressors?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ5.4 HERE]

[START FAQ5.5 HERE]

FAQ5.5: How do multiple climate-related and other human-induced stressors interact to affect ecosystems?

[PLACEHOLDER FOR SECOND ORDER DRAFT]

[END FAQ5.5 HERE]

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