Chapter 4.

PLAUSIBLE FUTURES OF NATURE, ITS CONTRIBUTIONS TO PEOPLE AND THEIR GOOD QUALITY OF LIFE

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CHAPTER 4

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EXECUTIVE SUMMARY

Chapter 4 focuses on scenarios and models that explore the impacts of a wide range of plausible future changes in social, economic and institutional drivers on nature, nature's contributions to people (NCP) and good quality of life. The chapter's assessment concentrates on studies published since 2008 that cover large regional to global spatial scales and time periods from the present to 2050, and up to 2100. This framing of the assessment means that this chapter is best suited to help setting the agendas for decision-making at national to international levels by identifying future challenges and providing a compelling case for action. Chapter 4 provides new insights compared to previous assessments by including the most recent scenarios and models, by examining a broad range of global change drivers and their interactions, and by highlighting the impacts on a wide range of indicators of nature, nature's contributions to people and good quality of life. Where possible, results are also interpreted in view of their implications for achieving the Aichi Biodiversity Targets and the Sustainable Development Goals.

This chapter endeavours to provide a balanced perspective on drivers of change and their impacts, but the strong bias in the scenario literature towards climate change impacts on nature limits the scope to which the chapter can provide a comprehensive vision of plausible futures to decision makers. Climate change has been studied far more extensively than other drivers (such as land use change, pollution, use and extraction of natural resources, and invasive alien species), and studies of interactions between drivers, especially more than two drivers, are relatively rare *(well established)* {4.2.1, 4.2.2, 4.2.3, 4.2.4}. Terrestrial systems are studied more extensively than marine systems, with a paucity of studies of freshwater systems (*well established*) {4.2.1.1}. Impacts on biodiversity and ecosystem function have been the focus of much more attention than nature's contributions or good quality of life (78%, 16% and 5% of literature reviewed, respectively; (*well established*) {A1.1}. Among nature's contributions to people, material (such as food production) and regulating contributions (such as carbon dioxide removal from the atmosphere into ecosystems) are more studied than non-material contributions in relation to scenarios (*well established*) {4.3.1}.

The large majority of the studies covered in this chapter is based on scenarios developed in support of climate change assessments (93% of literature reviewed; {4.1.3}, the most recent of which are the Representative greenhouse gas Concentration Pathways (RCPs) and their associated Shared Socio-economic Pathways (SSPs). This has the benefit of providing strong coherence with climate assessments but results in biases in terms of drivers of change and socio-economic processes included in the scenarios. For example, only few of the scenarios assessed in this chapter explore mechanisms leading to social or ecological regime shifts {4.5}. In addition, most scenarios do not explicitly take into account different worldviews and values associated with many non-material nature's contributions to people and, in general, were not designed to address a wide range of Sustainable Development Goals {4.5, Chapter 5}. Nonetheless, this chapter recognizes that the different scenario archetypes hold inherently different worldviews and values that ultimately drive the scenario outcomes {4.1}. Participatory scenarios are one means of including a richer range of processes and values explored, but it is difficult to extrapolate from the local scale of most participatory scenarios to the large regional and global spatial scales that are the focus of this chapter {4.4.2, 4.7}.

① **Significant changes at all biodiversity levels – from genetic diversity to biomes – are expected to continue under future global changes. Despite projections of some local increases in species richness and ecosystem productivity, the overall effect of global changes on biodiversity is projected to be negative (*well established*). Interactions within and between biodiversity levels can significantly influence future biodiversity responses to global changes (*established but incomplete*).** A substantial fraction of wild species is simulated to be at risk of extinction during the 21st century due to climate change, land use, natural resource extraction and impact of other direct drivers (*well established*) {4.2.1, 4.2.2, 4.2.3, 4.2.4}. Loss in intraspecific genetic diversity is expected due to the projected decrease in species population sizes and spatial range shifts. Genetic loss should be recognized as a serious threat to future potential for adapting to global change (*established but incomplete*) {4.2.1.2, 4.2.1.3}. Expected species range shifts, local species extinctions, changes in species abundances will lead to disruptions of species relations including disturbance of trophic webs, plant-pollinator and other mutualistic relations (*well established*) {4.2.2, 4.2.3, 4.2.4}, that can cascade through the entire ecosystem. Novel (no-analogue) communities, where species will co-occur in historically unknown combinations, are expected to emerge (*established but incomplete)* {4.2.1.2, 4.2.4.1}. As a consequence, new approaches to conservation are warranted that are designed to adapt to rapid changes in species composition and ensuing conservation challenges. Intraspecific diversity and interactions between different biodiversity levels need to be represented in global models and scenarios to improve future projections of nature {4.2.1.2, 4.2.1.3}.

2 In marine ecosystems, most scenarios and models point towards a global decrease in ocean production and biodiversity, but the level of impact can vary widely, depending on the drivers, scenarios, and regions considered (*well established*). All anthropogenic greenhouse gas emission scenarios result in a global increase in sea temperature, ocean acidification, deoxygenation and sea level rise (*well established*) {4.2.2.1}. By the end of the century, these environmental changes are projected to decrease net primary production (by ca. -3.5% under the low greenhouse gas emissions scenario, RCP2.6 and up to -9% in the very high emissions scenario, RCP8.5), and secondary production up to fish (by -3% to -23% under RCP2.6 and RCP8.5, respectively), as well as top predator biomass (*established but incomplete*) {4.2.2.2.1}. Fish populations and catch potential are projected to move poleward due to ocean warming (*well established*) with a mean latitudinal range shift of 15.5 km to 25.6 km per decade to 2050 (under RCP2.6 and RCP8.5, respectively) (*inconclusive*), leading to high extirpation rates of biomass and local species extinctions in the tropics (*well established*) {4.2.2.2.1}. The rapid rate at which sea ice is projected to retreat in polar seas, and the enhanced ocean acidification, imply major changes to be expected in the future for biodiversity and ecosystem function in the Arctic and Southern oceans (*well established*) {4.2.2.2.4}. All components of the food webs will potentially be impacted, from phytoplankton to top predators, and from pelagic to benthic species (*established but incomplete*).

3 Relative to climate change impacts, published scenarios project that the choice of fisheries management and market regulation measures can have the strongest impacts on the future status of marine fish populations *(well established)* {4.2.2.3}. In the face of continuous growth of human population that is projected to reach 9.8 billion (± ca. 0.4 billion) people in 2050 combined with rising incomes, the demand for food fish will likely increase (*well established*). Business-as-usual fisheries exploitation is foreseen to increase the proportion of overexploited and collapsed species (*well established*), as well as species impacted by bycatch {4.2.2.3}. Adaptive fisheries management that responds to climate induced changes of fish biomass and spatial distribution could offset the detrimental impacts of climate change on fish biomass and catch in most RCPs (but RCP8.5) (*inconclusive*) {4.2.2.3}.

4 For marine shelf ecosystems, additional future threats include extreme climatic events, sea level rise and coastal development which are foreseen to cause increased pollution and species overexploitation but also fragmentation and loss of habitats that directly impact the dynamics of marine biodiversity (*well established*) {4.2.2.2.2, 4.2.2.3}. These impacts could potentially feedback to the climate as coastal wetlands play a major role in carbon burial and sequestration globally (*well established)* {4.2.2.2.2}. In coastal waters, increasing nutrient loads and pollution in combination with sea warming are expected to stimulate eutrophication and increase the extent of oxygen minimum zones with potential detrimental effects on living organisms (*well established*) {4.2.2.3}. Coral reefs are projected to undergo more frequent extreme warming events, with less recovery time in between, declining by a further 70-90% at global warming of 1.5°C, and by more than 99% at 2°C causing massive bleaching episodes with high mortality rates (*well established*) {4.2.2.2.2}.

5 Concerns about rapidly increasing plastic pollution now match or exceed those for other persistent organic pollutants. If current production and waste management trends continue, about 12,000 Mt (million tons) of plastic waste will accumulate in the environment by 2050, especially in the ocean which acts as a sink (*established but incomplete*). The harmful effects of plastics have been evidenced at all levels of marine food webs from plankton to top predators but are not yet projected into the future {4.2.2.4.1}.

6 In freshwater ecosystems, all scenarios and models point towards a decrease in freshwater biodiversity and substantial changes in ecosystems state and functioning, especially in tropical regions (*well established*). Freshwater ecosystems cover only 0.8% of the world surface area but host almost 8% of the world's species described, making a high contribution to global biodiversity. Given that all scenarios are based on continued growth of human population density until 2050, impacts due to combined anthropogenic drivers on freshwater biodiversity and ecosystems are projected to increase worldwide, and to be strongest in tropical regions where human population growth and biodiversity are concentrated (*well established*) {4.2.3}. Increases in land area used for urbanization, mining, cropland and intensification of agriculture are projected to boost the risk of pollution and eutrophication of waters, leading to extirpation of local populations, changes in community structure and stability (e.g. algal blooms) (*well established*) {4.2.3.3}, and establishment and spread of pathogens (*established but incomplete*) {4.2.3.3}. Under all scenarios, habitat fragmentation (e.g., damming of rivers) and exploitation are projected to increase the risk of species extinction with potential effects on food web dynamics, especially in tropical regions (*well established*) {4.2.3.4, 4.2.3.6}. These impacts on freshwater flows, biodiversity and ecosystems will likely be exacerbated by climate change, especially under moderate (RCP4.5) and high emissions (RCP6.0, RCP8.5): higher temperatures are projected to generate local population extinctions especially for cold-water adapted species, and species extinctions in semi-arid and Mediterranean regions, since the area extent of these climatic regions will shrink due to projected decrease in precipitation (increase of estimated extinction rates by ca. 18 times in 2090 under the SRES A2 scenario, compared with natural extinction rates without human influence) (*inconclusive*) {4.2.3.2}.

7 In terrestrial ecosystems, scenarios and models point towards a continued decline in global terrestrial biodiversity and regionally highly variable changes in ecosystem state and functioning (*well established*). Land-use change, and invasive alien species will continue to cause biodiversity loss across the globe in the future, with climate change rapidly emerging as an additional driver of loss that is increasing over the coming decades in relative importance across all scenarios (*well established*) {4.2.4}. Although large uncertainties exist regarding the exact magnitude of loss, it is well established that increasing global warming will accelerate species loss {4.2.4}. Already for relatively minor global warming, biodiversity indices are projected to decline (*established but incomplete*) {4.2.4}. Extinction risks are projected to vary between regions from 5% to nearly 25%, depending on whether a region harbours endemic species with small ranges or is projected to experience climate very different from today (*inconclusive*). Substantial climate change driven shifts of biome boundaries, in particular in boreal and sub-arctic regions, and (semi)arid environments are projected for the next decades; warmer and drier climate will reduce productivity (*well established*) {4.2.4.1}. In contrast, rising atmospheric CO_2 concentrations can be beneficial for net primary productivity of ecosystems, and is expected to enhance woody vegetation cover especially in semi-arid regions (*established but incomplete*) {4.2.4.1}. The combined impacts of CO_2 and climate change on biodiversity and ecosystems remain (*unresolved*) {4.2.4.1}.

8 The relative impacts of climate change versus land-use change on biodiversity and ecosystems are context-specific and vary between scenarios, regions, and indicators of biodiversity and ecosystem functioning (*well established*) {4.2.4.2, 4.2.4.3}. Land-use change pressures differ between scenarios, but managed land area continues to increase, with exception of some scenarios exploring sustainability trajectories. Scenarios of large-scale, land-based climate change mitigation rely on large increases of bioenergy crop area or large reforestation or afforestation with potentially detrimental consequences for biodiversity and some ecosystem functioning *(well established)* {4.2.4.2, 4.2.4.3, 4.5.2}. Interactions of land-cover change and future climate change enhance the negative impacts on biodiversity and affect multiple ecosystem functions (*established but incomplete*) {4.2.4.2, 4.2.4.3}. Pressure on biodiversity and ecosystem function from other drivers such as biological invasions will likely be accentuated at global scale, as trade between climatically and environmentally similar regions are projected to increase, and habitats continue to be disturbed (*established but incomplete)*. Overall, the small number of regional to global scale scenario studies that assess pollution or invasive alien species' impacts on nature precludes a robust assessment {4.2.4.4, 4.2.4.5}.

9 Many scenarios project increases in material nature's contributions to people, which are generally accompanied by decreases in regulating and non-material contributions (*established but incomplete*) {3.1, 3.2}. The simulated trade-offs between material vs. regulating and non-material ecosystem services are especially pronounced in scenarios with strong human population growth and per capita consumption (*established but incomplete*) {4.3.4, 4.2.2.3.1, 4.2.4}. Assumptions about population growth and increase in per capita consumption are projected to lead to rising demand for material services, especially food, materials and bioenergy, and are projected to reduce regulating contributions such as provision of clean water, pollination, or ecosystem carbon storage (*well established*) {4.3.2, 4.3.3, 4.5.3, 4.2.2.4, 4.2.2.5, 4.2.3, 4.2.4}. In the long term, substantial decreases in regulating contributions may have detrimental effects on material contributions, for example climate change impacts on all systems will be increased if climate regulation by forests or oceans is weakened (*well established*). The future magnitude of these cascading effects has yet to be determined *(inconclusive). This is because* most scenarios and models do not consider fully the interactions between multiple drivers and multiple ecosystem impacts, and as a consequence cannot quantify important feedbacks {4.3.3, 4.3.4, 4.5.1, 4.5.4}.

10 Scenarios examining trends in nature and nature's contributions to people show significant regional variation (*well established*). The

interconnectedness of the world regions emphasizes the need for decision-making on ocean, freshwater and land management to be informed by considerations of regional trade-offs among nature's contributions to people (*well established*). Future scenarios show that many regions will experience a general decrease of biodiversity and many regulating and non-material ecosystem services, but others will see increases (*well established*) {4.2.2, 4.2.4, 4.3.3}. The degree to which regions differ regarding impacts of global environmental changes depends on the underlying socio-economic scenarios, with climate change being an additional driver (*established but incomplete*) {4.1, 4.2, 4.3}. Scenarios of a world with regional political- and trade-barriers (Regional Competition Scenario) tend to result in the greatest divergence across regions, scenarios that emphasize liberal financial markets (economic optimism and reformed market scenarios) in intermediate levels of disparity, while scenarios that encapsulate aspects of sustainable development (Regional Sustainability and Global Sustainability scenarios) result in more modest differences between regions (*established but incomplete*) {4.3.3, 4.2.4}. For example, an analysis of the impacts of the shared socio-economic pathway (SSP) scenarios indicates that terrestrial biodiversity and regulating contributions will be more heavily impacted in Africa and South America than in other regions of the world, especially in a regional competition scenario and in an economic optimism scenario compared to a global sustainability scenario {4.2.1, 4.2.4.2}.

Irrespective of the underlying socio-economic assumptions, spatial telecoupling (socioeconomic and environmental interactions over distances) implies that increasing future demand for ecosystem services in certain regions will affect supply of services in others. Material contributions, especially food and energy production, play a dominant role in these telecouplings (*well established*) {4.2.4, 4.3.3, 4.5.2}. Material contributions tend to be traded between regions {4.1, 4.2.4.4., 4.2.4.5, 4.5.2, 4.6}, but locally declining biodiversity cannot be replaced by increased biodiversity in a different location {4.2.2-4.2.4}. If tele-couplings are not accounted for in future scenarios, unrealistically overoptimistic responses to a regional political intervention (e.g., land-based climate mitigation, negative emission policies, sustainable fisheries management for local resources and not for imported ones) are assumed, and measures to reduce detrimental side effects not taken (*established but incomplete*) {4.3.3}.

⑪ **Limiting mean global warming to well below 2°C will have large co-benefits for nature and nature's contributions to people in marine, freshwater and terrestrial ecosystems. Land-based climate change mitigation efforts offer opportunities for co-benefits, but if large land areas are required, trade-offs with biodiversity conservation and food and water security goals will need to be addressed in terrestrial and freshwater ecosystems (*well established*).** Climate warming and ocean acidification associated with increasing atmospheric CO_2 are already causing damage to marine, freshwater and terrestrial biodiversity (*well established*) {4.2.2, 4.2.3, 4.2.4} which confirms the urgency of meeting the goals of the Paris Climate Agreement. The degree to which marine and land ecosystems will continue to remove CO_2 from the atmosphere, which at present amounts to nearly 50% of anthropogenic CO_2 emissions, is highly uncertain {4.2.2.1, 4.2.4.1}. On land, reduction of deforestation combined with management practices in cropland, pastures and forests can contribute notably to greenhouse gas emissions reductions (*well established*). Recent cost-effective estimates are between ca. 1.5 and 11 Gt CO_2eq a^{-1} over the coming few decades, the undetermined range depending, amongst others, on which types of measures are included {4.5.3}. Along coastlines, a combination of reduced nutrient discharge (mitigating pollution) and space to allow inland wetland migration (adapting to sea level rise), is essential to preserve the capacity of coastal wetlands to sequester carbon (*established but incomplete*) {4.2.2.2.2, 4.2.2.5}.

Regionally, land conversion pressure is large both in scenarios of high population growth and lack of sustainability considerations, and in scenarios requiring land for bioenergy or afforestation and reforestation to mitigate climate change (*established but incomplete*) {4.1, 4.2.4.3}. Recent projections of an annual carbon uptake in 2050 projected for bioenergy pathways (with carbon capture and storage about 0.9-2.2 GtC a^{-1}) and afforestation/reforestation (0.1-1 GtC a^{-1}) are equivalent to an additional one third to three quarters of today's land carbon sink {4.2.4.3}. It remains uncertain whether the required land area would be available for large bioenergy plantations or afforestation/reforestation efforts, where these areas would be located and whether such net carbon uptake rates can be achieved and maintained {4.2.4.3, 4.5.2}. Likewise, detrimental environmental and societal side effects have been projected to arise from strong mitigation scenarios that rely on large area expansion of managed crop and forested land associated with intensification of production (*established but incomplete*) {4.2.4.3, 4.3.2.1, 4.5.2}.

⑫ **Scenarios repeatedly show that changing food consumption patterns and reducing waste and losses in the food system can contribute significantly to mitigating loss of biodiversity and ecosystem services.** Human population growth over the coming decades is projected to increase to nearly 9.8 billion (± 0.4 billion) by 2050 and to 11.4 billion (± 1.8 billion) by 2100. As a consequence of the projected population growth, continued urbanisation, and changes in many countries' diets towards increasing per capita animal protein share and processed food, most scenarios foresee increasing crop area, and in some cases pasture area as well. These projected changes in agricultural land area are combined with intensification of land management and continued increases in crop yields, that are projected to have detrimental environmental and biodiversity side effects associated with agricultural intensification (*well established*) {4.2.2.4.2, 4.3.2.1, 4.3.2.2, 4.5.2}. An increasing number of scenarios emphasizes the potential role of consumption as part of the solutions to overcome these challenges, such as shifting diets towards a globally equitable supply of nutritious calories or reducing wastes and losses along the entire chain from crop production to consumers (*well established*) {4.5.4}. Enhancing efficiencies in the food system has large potential to free up land for other uses such as for biodiversity conservation. Studies that explore dietary scenarios of reduced consumption of animal protein estimate that between ca. 10% and 30% of today's area under agriculture may be freed for other purposes, with possible co-benefits in the form of a globally more equitable distribution of animal protein intake by humans and improved health. Reduced greenhouse gas emissions from the land sector, and reduced irrigation water needs are an additional benefit, which will also release pressure on freshwater pollution and biodiversity (*established but incomplete*). Nearly one-quarter of total freshwater used today in food crop production are estimated to be spared if wastes and losses in the food system were minimized (*inconclusive*) {4.3.1, 4.3.2, 4.5.2, 4.5.3}.

13 Societies and individuals within societies value differently the regulating, material, and non-material contributions from nature that underpin their quality of life (*well established*). In future scenarios governed by market forces, multiple dimensions of good quality of life are expected to decline. The decline is particularly pronounced for indicators related to livelihood and income security (*established but incomplete*) {4.4.1, 4.4.2}. Market-based and regionally-fragmented scenarios, associated with growth in population and consumption, indicate continuous deterioration of nature to support economic growth, with some regions affected more than others. Without decoupling economic growth from unsustainable extraction and uses, scenarios show continuous decline in nature's contributions to people. Scenarios exploring sustainability or reformed financial market pathways are projected to result in improved good quality of life (*established but incomplete*) {4.4.1, 4.4.2}. In general, the lack of explicit consideration in global scenarios of good quality of life explicitly, and its regionally and socially differentiated nature, impedes robust projections into the future, in particular for non-material aspects. Interactions of future changes in nature, its contributions to people and good quality of life can be better understood and, therefore, potentially better anticipated and managed, when they are evaluated at regional scales as well as the global scale.

Small-scale farming, fishing and other communities, and Indigenous Peoples around the world that depend directly on local environments for food production, especially in low-income countries, are particularly vulnerable to climate-related food insecurity, which raises important equity and fairness issues. Similarly, in coastal regions, decreases in precipitation and fresh water supplies, along with projected increases in sea level, sea surface temperatures and air temperatures, and ocean acidification are projected to have major negative effects on water security for societies. Nature-based livelihoods may become precarious with intensifying future trends in environmental change (*established but incomplete*) {4.4.1, 4.4.2}. Future threats to biodiversity and ecosystem services also constitute imminent challenges to the cultural identity of communities, particularly when faced with environmental degradation (*unresolved*) {4.4.2}.

14 The role of people's knowledge, values and traditions, and their potential future changes have been barely explored in global scenarios of future socio-economic and environmental change. A challenge to the assessment of nature's contribution to people and good quality of life under different future scenarios is their socially differentiated nature. People's values and traditions are crucial in shaping the future, yet they are rarely central to scenario exercises (*established but incomplete*) {4.4.1}. Novel methods are beginning to be developed to fully integrate people's worldviews into scenario planning, however transcendental values held by the social groups have so far not been well incorporated. The process of elaborating scenarios with participatory approaches is increasingly taking into account value negotiations around the meaning of good quality of life (*established but incomplete*) {4.4.2}. Consequently, ethical questions emerge regarding how to build scenarios so that local knowledge, particularly that of Indigenous Peoples and Local Communities (IPLCs), are not coopted in ways that may exacerbate processes of their social marginalization.

15 Different social groups experience change in ecosystem function and services differently so that a given change scenario usually implies winners and losers in terms of the projected impacts on good quality of life (*established but incomplete*) {4.4.1, 4.4.2, 4.4.3}. People vary in their access to ecosystem services, exposure to disservices, dependence on ecosystems, needs and aspirations. These are further mediated by societal structures and norms as individual characteristics and power relations {4.4.2, 4.4.3}. Many IPLCs are found in protected areas, where dimensions of good quality of life such as food and energy security may trade off with other dimensions of ecosystem functioning. Indirect drivers of change such as climate mitigation policy (e.g., REDD+) may disproportionately impact the possible trajectories towards achieving good quality of life by IPLCs (*unresolved*) {4.4.1}.

Thus, decision-making about environmental management with implications for different bundles of ecosystem services is an intently political process, with often divergent stakeholder interests and power dynamics. Evaluating the implications for the good quality of life of IPLCs under different scenarios of change can benefit from deliberative and participatory approaches that consider a wide range of stakeholder views, and disciplinary perspectives. Such a diversity of perspectives needs to draw on indigenous and local knowledge, to take account of the multiple interacting factors and socially differentiated experiences, vulnerabilities and preferences (*established but incomplete*) {4.4.2, 4.4.3}. A limitation with participatory approaches is the difficulty of imagining future scenarios of changes in the 'demand side' of nature's contributions. So, a group may discuss how changes in a resource might be affected by climate change, but it is often framed in terms of current social conditions. Likewise, participatory approaches are likely to be more successful if the scale of scenarios (e.g., local, regional, global) and stakeholder group perspective can be matched.

16 Most internationally agreed policy goals and targets for biodiversity are missed by most countries under business-as-usual scenarios because the current patterns and future trends of production and consumption are not environmentally sustainable. Indeed, trajectories of most biodiversity indicators under business-as-usual increasingly deviate from targets over time (*well established*) {sections 2 and 6}. The achievement of most biodiversity targets therefore requires a steer away from the current socio-economic trajectory and the worldviews and values that underpin it (*well established*). Scenarios that assume increased sustainability show that achieving most SDGs is possible at some point in the future, but this requires substantive and immediate action (*established but incomplete*) {4.6.1}, and the time horizon of the possible achievement of the SDGs is undetermined.

Scenarios and models can support the formulation of future biodiversity targets in terms of concept, phrasing, quantitative elements, and selection of indicators to monitor progress (*established but incomplete*). Scenario and models are also amenable to exploring interactions among targets (*well established*). For example, scenarios have shown that ambitious protected area expansion plans would conflict with agricultural production under business-as-usual assumptions, and that achieving SDGs for both biodiversity and hunger would require a 50-70% increase in land productivity (*inconclusive*) {4.6.1}.

Focusing future quantitative targets for biodiversity on management outcome rather than effort may improve policy implementation and related management decisions. For example, the numeric component of Aichi BiodiversityTarget 11 relates to the global proportion of protected areas. But the aim of protected areas is to achieve the long-term conservation of nature, which suggests to move the focus to the amount of nature that is protected and the effectiveness of protection rather than proportion of area under protection. Scenarios and models have shown that the outcome of a protected area network is determined by its location, connectivity and management, other than its size.

17 There is a lack of global-scale impact analyses that integrate across natures, nature's contributions to people and good quality of life. Most scenarios developed for global environmental assessments have explored impacts of humans on ecosystems, such as biodiversity or productivity loss {4.1, 4.2}. The effects of alternative trajectories of socioeconomic development on ecosystems and ecosystem services have been assessed as one-way outcomes, ignoring the possible interactions between natural and socioeconomic systems. A better understanding of feedback mechanisms is needed on many fronts, for instance: in what ways pollution arising from agricultural intensification does impact pollinators and/or water quality, which in turn impact land use and intensification? How do changes in food prices arising from different land uses feed back to land-use decision-making? How is overfishing leading to the depletion of large predatoryfish and development of global markets for alternative species, often their own prey, leading to further collapse of marine resources? To what extent climate change induced sea level rise is decreasing wetland area and is affecting carbon sequestration? (*established but incomplete*) {4.1, 4.3.2.1, 4.5.1-4.5.3, 4.6.1, 4.7.3}. In addition, storylines of socio-economic development that underlie global scenarios consider mostly material aspects of GQL and do not consider other indicators of GQL {4.4.1-4.4.3}. There is a knowledge gap in scenario studies about non-material contributions to people compared to material contributions and regulating contributions, which limits our capacity to understand quantitatively how nature, its contributions to people and good quality of life interact and change in time.

In particular, human decision-making at multiple levels is not well integrated in global scenario modelling tools such as Integrated Assessment Models that focus on economic objectives (*well established*) {4.1, 4.2, 4.5.1, 4.5.2, 4.4.1-4.4.3}. A paradigm shift in scenario design could be achieved by considering, alongside of economic principles, provisioning of multiple ecosystem services and GQL as part of the storyline and human decisions (and subsequent scenario realisation), rather than as an outcome of socio-economic drivers {4.6.1}. For a more robust scientific underpinning of biodiversity and multiple sustainability targets, these non-material aspects need to be explicitly addressed in the scenarios (*unresolved*) {4.6.1}. Such scenarios would facilitate policy-relevant scientific evidence through exploration of trade-offs and co-benefits between targets related to biodiversity and ecosystem services, including the interconnected nature of drivers across regions {4.3.4, 4.5.1}. Participatory Scenario Planning, with stakeholders aligned to the scale of the scenario (e.g., the CBD for global scenarios) would allow for a differentiated assessment of good quality of life across stakeholder groups and highlighting winners and losers across environmental or policy scenarios (*established but incomplete*) {4.4.2}.

18 **Large uncertainties remain in future scenarios and related impact studies at the global scale.** Careful analysis and communication of sources of uncertainty in scenarios and models are vital when using them in support of decision-making (*well established*). Global modelling tools to explore futures of biodiversity and futures of ecosystem state and function are still mostly disconnected and do not consider diversity-function links {4.2, 4.7}. Projected future changes in species ranges, community diversity or ecosystems may be under- or overestimated by most studies because they do not explicitly account for impacts of multiple drivers, adaptive capacity of species and for feedbacks arising from species interactions {*established but incomplete*) {4.2.5, 4.5}. Effectively linking scenarios and models across spatial and temporal scales is methodologically difficult and in early stages of development and use but can make important contributions to decision-making when achieved (*established but incomplete*). However, linking must be done with considerable caution because it creates additional complexity that can make the behaviour of scenarios and models difficult to understand and may introduce important sources of uncertainty {4.5, 4.7}. Substantial efforts are needed to identify uncertainty related to models and scenarios and improve the treatment of uncertainty between and within models {4.2, 4.6, 4.7}. Strong, sustained dialogue between modellers, stakeholders and policymakers are one of the most important keys to overcoming many of the significant challenges to dealing with uncertainty and scales issues when mobilizing scenarios and models for decision-making.

4.1 INTRODUCTION

4.1.1 Context and objectives of the chapter

Rapid biodiversity loss and its adverse consequences for nature, nature's contributions to people and Good quality of life clearly remain as key challenges for the coming decades. Economic inequality, societal polarization and intensifying environmental threats have been identified by the World Economic Forum's *Global Risks Report (GRR) 2017* (WEF, 2017) as the top three challenges for global developments over the next decade or more. For the first time, all five environmental risks in the report (extreme weather; failure of climate change mitigation and adaptation; major biodiversity loss; natural disasters; human-made environmental disasters) were ranked both high-risk and high-likelihood (WEF, 2017). These challenges emphasize the importance of the UN 2030 Agenda and the Sustainable Development Goals (SDGs) and the 2050 Global Vision for Biodiversity to facilitate a sustainable future state for the planet, with a recognition of the connections between humans and ecosystem well-being at their core (Costanza *et al.*, 2016).

This chapter focuses on the assessment of scenarios and models that have been used to explore a wide range of plausible futures of nature, nature's contributions to people (NCP) and good quality of life (GQL), focusing on the current-to-2050 time frame and on continental to global spatial scales. One objective is to alert decision makers to potential undesirable impacts of a broad range of plausible

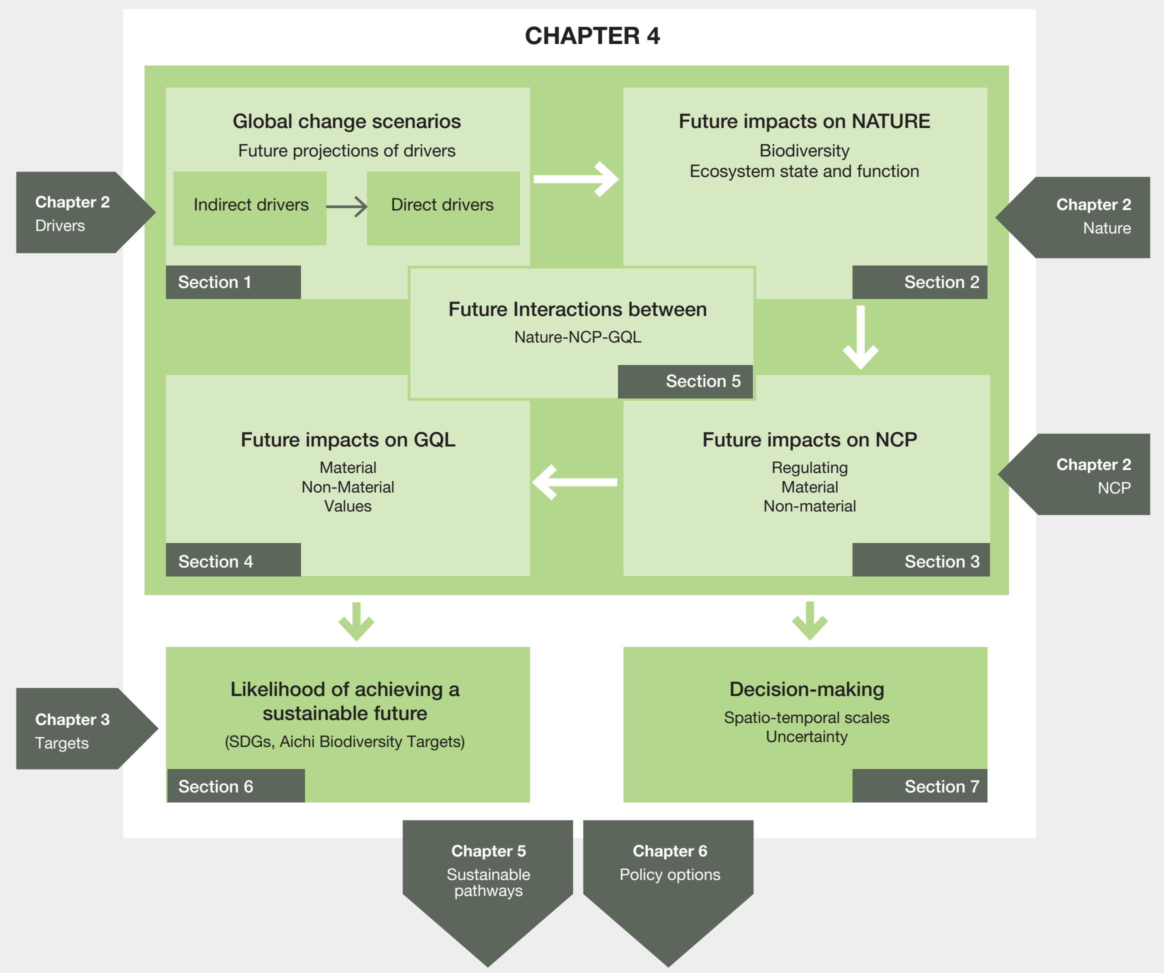


Figure 4 1 1 Scope of Chapter 4 "Plausible futures of nature, its contributions to people and their good quality of life" of the IPBES Global Assessment, content of sections and their relationships, and linkages with the other chapters of the Global Assessment. NCP: nature's contributions to people; GQL: good quality of life.

socio-economic development pathways. A second objective is to highlight development pathways and actions that can be taken to minimize impacts, as well as restore nature and enhance its contributions to people. As is clearly highlighted in Chapters 2 and 3 of this assessment, the context is that pressures, such as resource exploitation and climate change, continue to increase, and most measures of the state of nature and nature's contributions to people continue to decline. This chapter is designed to help understand the conditions under which these trends might accelerate vs. stabilize or even improve over the coming decades.

Scenarios are a means of exploring plausible future trajectories of direct and indirect drivers of environmental change (IPBES, 2016b). Models provide a means to estimate qualitatively or quantitatively the impacts of indirect and direct drivers on nature and nature's contributions to people (IPBES, 2016b). Building upon an analysis of drivers of change presented in chapter 2.1, this chapter starts with an assessment of the key underlying assumptions about drivers in scenarios and a synthesis of the projected trajectories of key direct drivers, such as climate change and land-use change, and indirect drivers, such as human population and economic growth, over the next several decades and places these in the context of current trends (section 4.1; **Figure 4.1.1**, see Chapter 2.1).

Sections 4.2 and 4.3 of this chapter focus on the assessment of a wide range of quantitative models that have been used to project future dynamics of nature and its contributions, and these sections also place these projections in the context of observed trends as well as the current understanding of the mechanisms underlying these trends (see Chapter 2). Models can also be used to evaluate the impacts of changes in nature and its contributions on quality of life, but this has rarely been done (IPBES, 2016b). As such, section 4.4 focuses on the underlying assumptions about quality of life embedded explicitly or implicitly in models and scenarios, as well as making qualitative connections with modeled impacts on nature and its contributions. Projected synergies and trade-offs between nature, NCP and GQL are explored in section 4.5.

Finally, comparisons of scenarios and model outcomes are then made with internationally agreed objectives, such as the Sustainable Development Goals for 2030 and the Convention on Biological Diversity's 2050 Vision, in order to better understand the types of socio-economic development pathways that lead to outcomes that are closest to or furthest from these objectives (section 4.6). This is then put in the broader context of the use of scenarios and models in decision-making (section 4.7), with a focus on the importance of scales and uncertainty in the use of models and scenarios to inform decisions. Chapter 5 follows by providing a more in-depth analysis of "target-seeking" scenarios designed to evaluate sustainable futures, including evidence regarding sustainable transition pathways, for which specific policy options are discussed in Chapter 6.

4.1.2 Exploratory scenarios

Scenarios can be defined as plausible representations of possible futures for one or more components of a system, or as alternative policy or management options intended to alter the future state of these components (IPBES, 2016b). They provide a useful means of dealing with many distinct possible futures (Cook *et al.,* 2014; Pereira *et al.*, 2010). Policy and decision-making processes rely on estimates of anticipated future socio-economic pathways, and knowledge of the potential outcomes of actions across distinct geographic regions, sectors and social groups. The process of scenario development itself can help to build consensus by integrating the objectives of different stakeholder groups (Priess & Hauck, 2014). This is particularly germane in efforts that seek to integrate the knowledge, perspectives and goals of local stakeholders, particularly Indigenous Peoples and Local Communities (IPLCs), who are frequently marginalized from policy and decision-making processes (IPBES, 2016b; Petheram *et al.*, 2013).

When assessing future impacts on nature, its contribution to people and related good quality of life, there is a need to link the trajectory of direct and indirect drivers to different future scenarios. Exploratory scenarios can be either qualitative, in the form of storylines, or quantitative, in the form of model outputs (van Vliet & Kok, 2015). The main objective of exploratory scenarios is informing stakeholders of the potential impacts of different driver combinations, e.g., a proactive set of actions that may increase the likelihood of social, economic or political targets versus a "business-as usual" scenario that involves no major interventions or paradigm shifts in the organization of functioning of a system. Exploratory scenarios may provide a plurality of plausible alternative and contrasting futures.

Exploratory scenarios for global scale environmental studies and assessments have been developed for a range of UN related assessments, including scenarios developed under the IPCC process, such as the so-called SRES scenarios (Nakicenovic *et al.*, 2000) in the late 1990s, the Representative Concentration Pathways (RCPs) and the recent Shared Socio-economic Pathways (SSPs), as well as scenarios considered for the UNEP Global Environmental Outlook (GEO) (UNEP, 2012) process, Global Biodiversity Outlook (GBO) and the Millennium Ecosystem Assessment (MA, 2005). The Global Scenario Group has also developed a range of contrasting global scenarios (Raskin *et al.*, 2002).

In addition, organizations such as FAO, OECD, IEA and UNESCO have developed several scenarios for specific purposes, such as the OECD Environmental Outlook to 2050 where a trend-based scenario was developed and a large number of policy alternatives were evaluated (OECD, 2012). Several of these scenarios have been evaluated by Integrated Assessment Models (IAMs) to specify and quantify ecological and environmental changes, including climate change, land-use change, vegetation dynamics and water (Kok *et al.*, 2018).

An important advance in the last few years has been to link representative concentration pathways (RCPs) with shared socio-economic pathways (SSPs) (O'Neill *et al.*, 2014) in support of the IPCC process, to inform deliberations under the UN Framework Convention on Climate Change (UNFCCC). Some of these scenarios imply significant mitigation efforts in the land-use sector, including large-scale reforestation and afforestation, or bioenergy crops with implications for both biodiversity and ecosystem services (Riahi *et al.*, 2017).

Existing environmentally relevant scenarios include scenarios that are most often either exploratory (this chapter focus) or target-seeking (Chapter 5) (IPBES, 2016b). In many cases, these scenarios may be appropriate for specific temporal or spatial scales or limited in scope (e.g. relevant to one or a few sectors). They can also be incomplete with regard to quantitative information about nature, NCP and GQL, and thus less useful for the purposes of this IPBES assessment. This is because integrated assessment models that often underpin scenarios of future greenhouse gas emissions, land-use change, or demand for food have a strong economic perspective and do not consider e.g., monetary or non-monetary values of ecosystem services. Issues related to conservation or biodiversity, or feedbacks from changes in ecosystem services to socio-economic decision-making, have typically not been well considered in the wide range of global scenarios that are well established in the climate change scientific communities. Likewise, scenarios of the future of biodiversity typically do not seek to quantify the possible co-benefits related to ecosystem services (Kok *et al.*, 2017; Pereira *et al.*, 2010; Powell & Lenton, 2013). Important gaps remain in scenario development, such as the development of integrated scenarios for areas projected to experience significant impacts and possible regime shifts (e.g. Arctic, semi-arid regions and small islands), and socioeconomic scenarios developed for and in collaboration with Indigenous Peoples and Local Communities (IPLCs) and their associated institutions, values and worldviews (Furgal & Seguin, 2006).

4.1.3 Archetype scenarios

From the many scenarios developed in the last few decades, it is apparent that groups of scenarios have many aspects of their underlying storylines in common and may be considered as "archetype scenarios". Archetypes represent synthetic overviews of a set of assumptions about the configuration and influence of direct and indirect drivers used in scenarios. They vary mainly in the degree of dominance of markets, dominance of globalization, and dominance of policies toward sustainability. Hunt *et al.* (2012) and van Vuuren *et al.* (2012) analysed a large number of local and global scenarios and came to the similar

Box 4 1 1 Scenario archetypes.

(from Hunt *et al.,* 2012; IPBES, 2016b; van Vuuren *et al.*, 2012; see also section 5.2.2 in IPBES, 2018i): description of underlying storylines, and links with indirect and direct drivers.

Economic Optimism. Global developments steered by economic growth result in a strong dominance of international markets with a low degree of regulation. Economic growth is assumed to coincide with low population growth due to a strong drop in fertility levels. Technology development is rapid and there is a partial convergence of income levels across the world. Environmental problems are only dealt with when solutions are of economic interest. The combination of a high economic growth with low population growth leads to high demands of commodities and luxury goods. These demands will however be unequally distributed among regions and within regions. Consequently, energy use and consumption are high. In addition, high technological development in combination with increased global market leads to high yields in agricultural and wood production on the most productive lands. Therefore, pollution and climate change will be relatively high, but land use relatively low. Direct exploitation will continue but also replaced by cultivation of for example fish and livestock. Global trade will increase the risks of invasive species.

Reformed Markets. Similar to the economic optimism scenario family but includes regulation and other policy assumptions to correct market failures with respect to social development, poverty alleviation or the environment. Thereby, relative to the economic optimism archetype, high demands for goods are expected to be more equally distributed and pollution will be lower.

Global Sustainable Development. A globalized world with an increasingly proactive attitude of policymakers and the public at large towards environmental issues and a high level of regulation. Important aspects on the road to sustainability are technological change, strong multi-level governance, behavioural change through education, and a relatively healthy economy. All variations of this archetype are beneficial for biodiversity. This scenario combines a low population growth with moderate economic development, and sustainable production and consumption. Low demands of especially luxury goods are expected, and a shift in diet towards less meat can be expected. Energy use will be low to moderate and fossil fuel use will be reduced, leading to low climate change and low land-use change. Due to environmental policies and sustainable production, pollution will be lower and direct harvesting will partly be replaced by cultivation. The global focus will increase the risk of invasive species

Regional Sustainability. A regionalized world based on an increased concern for environmental and social sustainability. International institutions decline in importance, with a shift toward local and regional decision-making, increasingly influenced by environmentally aware citizens, with a trend toward local self-reliance and stronger communities that focus on welfare, equality, and environmental protection through local solutions. The scenario combines a low economic growth with moderate population growth rates. The demands for goods are low and production focusses on sustainability with low levels of energy use or environmental degradation associated with higher importance for intrinsic and relational values of nature. Low rates of climate change are expected. Supply of agricultural products will be organised with regions with low levels of global trade. A slow technological development and a sub-optimal land use lead to relatively high rates of land-use change. Direct exploitation of natural systems will be within the carrying capacity of natural systems, and risks for invasive species will be relatively low.

Regional Competition. A regionalized world based on economic developments. The market mechanism fails, leading to a growing gap between rich and poor. In turn, this results in increasing problems with crime, violence and terrorism, which eventuates in strong trade and other barriers. The effects on the environment and biodiversity are mixed. Overall, there is a tendency towards increased security, which can either be positive (protect biodiversity) or negative (intensify agricultural production). Particularly in low-income countries, deforestation and loss of natural areas are a risk. In this scenario, due to a lack of global co-operation and trade, a high population growth is expected combined with low economic growth. Thereby, the demand for goods including agricultural products increases, but the demand for luxury, energy intensive goods is relatively low, and thus relatively low climate change is expected. Agricultural supply will be mainly within regions, which, combined with slow technological development, will result in lower productivity and high rate of land-use change. Direct exploitation will continue, low rates of replacement by cultivation are expected. The risk of invasive species will be lower than in the archetypes that focus on globalization.

Business-As-Usual. Assumes that the future can be characterised by a continuation of historical trends, including the implementation of international agreements. Sometimes referred to as a reference scenario, or as a middle-of-the-road scenario. It can also be considered as a less extreme variant of the economic optimism archetype. Business-as-usual is characterized by moderate economic growth, moderate population growth and moderate globalization. Demands are not high nor low, and in combination with moderate technological development, environmental changes will also be moderate.

conclusion that four to six scenario archetypes cover the large range of possible futures **(Box 4.1.1)**.

This chapter makes frequent reference to archetype scenarios because the use of scenario archetypes was also adopted in the IPBES regional assessments. This approach helped to synthesize results across a very broad range of scenario types. Synthesis across regional assessments is hampered by the use of different archetype classifications for each of the regions, which was done in order to match archetypes to regional contexts.

The IPBES methodological assessment on scenarios and models (IPBES, 2016b) adopted the "scenario families", as described in van Vuuren *et al.* (2012), which include the scenario archetypes **(Box 4.1.1)** distinguished by Hunt *et al.* (2012).

The different scenario archetypes describe different visions of the future (de Vries & Petersen, 2009), reflecting different values, guiding principles of society, understanding of good quality of life, approaches to decision-making and distribution of power (among other aspects). These aspects are often included in scenarios as implicit assumptions and have a large impact on the outcomes of the scenarios. For example, some scenario archetypes may prioritize intrinsic values of nature, while others may emphasize instrumental or relational values (Pascual *et al.*, 2017). These differences ultimately affect the different archetypes in various ways. **Table 4.1.1**. shows all these aspects synthesized across the six scenario archetypes. The most common global scale scenarios encountered in the literature can be assigned to these archetypes **(Table 4.1.2)**, with the caveat that individual scenarios do not match all of the characteristics of the archetype defined in **Table 4.1.1** and **Box 4.1.1**.

Analysis of the data sourced from the systematic literature review (Appendix A4.1.1) carried out as part of the background work for this chapter indicates a skewed representation of scenarios between and across the three components nature, NCP and GQL **(Table 4.1.3)**. This skew reflects to some extent the length of time scenarios have been available, but also reflects a bias towards climate change related scenarios. The analysis shows

Table ④ ① ① **Different guiding principles, values, approaches to good quality of life (GQL), distribution of power and decision-making approach across scenario archetypes.**

	Economic optimism	Reformed Markets	Global Sustainable Development	Regional Sustainability	Regional Competition	Business-As-Usual
Guiding Principles	Prosperity based on economic growth	Economic efficiency & sustainability	Global Sustainability	Equity & local sustainability	Individualism and safety concerns	No change
Main value in human-nature relationships	Instrumental / Utility value	Instrumental / Utility value	Intrinsic / Relational	Relational	Instrumental / Utility value	Instrumental / Utility value
Environmental principles	More "efficient" use of nature with new technologies, but protection is not prioritised	Use of nature is regulated with reformed polices	Protecting nature and environmental sustainability	Local sustainable use of nature	Lack of concern/ low priority for nature	Overexploitation of nature with elements of regulation and protection
Social principles	Individualism	Individualism with elements of cooperation	Global cooperation	Cooperation within the community	Individualism in a fragmented world	Individualism with elements of cooperation
Economic principles	Market oriented based on profit maximization	Market regulation based on efficiency & sustainability targets	Market regulation and non-market mechanisms based on global environmental sustainability and equity	Markets oriented to local environmental and quality of life priorities.	Market oriented with trade barriers and growing economic asymmetries / polarisation.	Market oriented with some barriers and some regulation
Approach to good quality of life	Material aspects	Material aspects, health and other GQL components included in international goals (e.g. SDG)	Respect for nature at the global scale is important for GQL	Livelihoods, Social relationships and health	Public security	Material aspects, and other components such as health, public security
Power relations among countries	Large countries powerful	Power imbalance moderated by negotiation	Power balanced by global institutions and collaboration	Decentralized among and within countries	High differences in power among regions	Large countries are powerful, power partially balanced by negotiation, high differences in power among regions
Decision-making processes	Top-down	Top-down	Horizontal / Participatory	Bottom-up / Participatory	Top-down with growing exclusion (marginalisation) of the poorest (most vulnerable) regions & social groups	Top-down
Powerful stakeholders	Private sector	Alliance of governments and private sector	Balance of power among the various stakeholders, global institutions	Communities	National Governments and private sector	Private sector & governments, with participation of NGOs

the available literature is strongly dominated by studies of future trajectories of nature, with considerably fewer studies on NCP and very few studies providing information on GQL. This may reflect the lack of integrated assessment tools available to conduct this type of work quantitatively. This inconsistency of coverage constrained the work in this chapter, and explains the emphasis put on nature (section 4.2).

Table ④ ① ② **Scenarios from earlier global assessments attributed to archetypes or families.**

Source: IPBES, 2016b; van Vuuren *et al.*, 2012.

Source	Economic Optimism	Reformed Markets	Global sustainable development	Regional Sustainability	Regional Competition	Business-As-Usual
SRES	A1F1		B1 (A1T)	B2	A2	B2
GEO3/GEO4	Market first	Policy first	Sustainability first		Security first	
Global scenario group	Conventional world	Policy reform	New sustainability paradigm	Eco-communalism	barbarization	
Millennium Ecosystem Assessment		Global Orchestration	Technogarden	Adapting mosaic	Order from strength	
OECD Environmental Outlook						Trend
Shared Socio-economic Pathways	SSP5		SSP1		SSP3/SSP4	SSP2
Representative Concentration Pathways (RCP)	RCP8.5		RCP 2.6		RCP 6.0	RCP 4.5
Roads from Rio/ fourth Global Biodiversity Outlook		Consumption Change	Global technology	Decentralized Solutions		Trend

Table ④ ① ③ **Classification of studies according to scenario represented along a continuum from nature via NCP (nature's contributions to people) to GQL (good quality of life) focused studies.**

The number of papers reported comes from the systematic literature review conducted for this chapter (Appendix A4.1.1).

Scenario	All	Nature	NCP	GQL
RCP8.5	237	198	39	0
RCP6.0	9	9	0	0
RCP4.5	50	41	9	0
RCP2.6	150	144	6	0
A1	6	4	1	1
A1b	119	108	8	3
A1B	4	0	4	0
A1F1	76	76	0	0
A1T	1	0	1	0
A2	200	191	7	2
B1	113	106	6	1
B2	123	117	5	1
SSP1	1	0	1	0
SSP2	13	1	12	0
SSP3	2	1	1	0

Scenario	All	Nature	NCP	GQL
SSP5	1	1	0	0
BAU	23	20	3	0
Global orchestration	13	11	2	0
Order from strength	12	9	3	0
Technogarden	11	10	1	0
Adapting mosaic	8	7	1	0
Consumption change	6	6	0	0
Global Technology	3	0	3	0
Decentralized solutions	1	1	0	0

4.1.4 Projected indirect and direct drivers of change in scenarios

The main indirect drivers of change of nature and its contributions to people, and consequently the quality of life include economic development, demographic trends and factors, technological development, governance and institutions, and various socio-cultural aspects such as worldviews and values. These indirect drivers have multiple impacts on direct drivers of change, which include climate change, land-use change, pollution, direct harvesting, invasive species and disturbance. In each scenario archetype, assumptions on the indirect drivers lead to different combinations of direct drivers **(Box 4.1.1)**.

Drivers are always multiple and interactive, so that one-to-one linkage between particular drivers and specific changes in ecosystems rarely exists. The causal linkage between drivers is often mediated by other factors or a complex combination of multiple factors, thereby complicating the understanding of causality or attempts to establish the contributions by the various drivers to changes in nature, NCP and GQL (see also Bustamante *et al.*, 2018; Elbakidze *et al.*, 2018; Nyingi *et al.*, 2018; Wu *et al.*, 2018). The cumulative effects of multiple stressors may not be additive but may be magnified by their interactions (synergies) and can lead to critical thresholds and transitions of ecological systems (Côté *et al.*, 2016). Cascading impacts of co-occurring stressors are expected to degrade ecosystems faster and more severely (section 4.7 in Bustamante *et al.*, 2018).

4.1.4.1 Indirect Drivers (including consideration of diverse values) in scenarios

Indirect drivers (also referred to as 'underlying causes') operate diffusely by altering and influencing direct drivers as well as other indirect drivers (also see chapter 1 in this report and IPBES, 2016b). They influence human production and consumption patterns with subsequent environmental implications. Economic drivers, including trade and finances, and demographic drivers interact with other indirect drivers such as technology, governance/institutions and social development including equity. Archetype environmental scenarios for this century consider explicit reference to relevant indirect anthropogenic drivers in different combinations, as indicated in **Table 4.1.4**.

Economic development has historically been the key indirect anthropogenic driver of changes in nature, NCP and GQL, across all scales (global, regional, national and local). World GDP (at constant 2010 USD) increased by 6.9 times between 1960 to 2016 (based on Worldbank, 2017). Taking a historical perspective, past and prevailing patterns of production and consumption embodied in global economic trends have generated growing pressures on natural resources, the environment, and ecosystem functions. In all scenarios, world GDP will continue to grow **(Table 4.1.5)**. However, some studies also refer to the plausibility of sustainable de-growth, as a transformative pathway leading to a steady-state at a reduced level of economic output (Schneider *et al.*, 2011).

Economic activities, international trade and financial flows are closely related, particularly in recent decades due to increasing economic globalization. These considerably influence changes in nature, NCP and GQL through various direct and indirect pathways. In turn, these pathways are influenced by a number of policy channels and mechanisms, like trade policies, including incentives (tax exemptions, subsidies) and trade barriers, the dynamics of foreign debt and foreign debt service, flows of foreign direct investments, and monetary policies (dynamic of exchange rates, interest rates).

Demographic trends are a major indirect anthropogenic driver of changes in nature, NCP and GQL, across

Table ④ ① ④ **Selected indirect drivers in archetype scenarios.**

Source: Based on Cheung *et al.* (2016: table 6.3); van Vuuren *et al.* (2012).

Selected indirect drivers	Archetype / scenario family					
	Economic Optimism	Reformed Markets	Global sustainable development	Regional Sustainability	Regional Competition	Business-As-Usual
Economic development	Very rapid	Rapid	Ranging from slow to rapid	Medium	Slow	Medium
Trade	Globalisation	Globalisation	Globalisation	Trade barriers	Trade barriers	Weak globalisation
Technological development	Rapid	Rapid	Ranging from medium to rapid	Ranging from slow to rapid	Slow	Medium
Population growth	Low	Low	Low	Medium	High	Medium
Policies & institutions (Governance)	Policies create open markets	Policies reduce market failures	Strong global governance	Local steering	Strong national governments	Mixed

Table ④ ① ⑤ **Economic development (in GDP PPP) for the scenario archetypes.**

Source: MA, 2005; Nakicenovic *et al.*, 2000; OECD, 2012; Raskin *et al.*, 2002; Riahi *et al.*, 2017; UNEP, 2007). Global GDP was approximately 50 trillion $ at purchasing power parity in 2000. GDP PPP: Global Domestic Product based on purchasing power parity.

	GDP PPP in trillion 2000 US$					
	Economic Optimism	Reformed Markets	Global sustainable development	Regional Sustainability	Regional Competition	Business-As-Usual
2050	182-323	181-229	168-251	139-145	106-198	145-241
2100	458-895	427	213-498	310	177-321	310-473

all scales (global, regional, national and local). World population increased by 2.5 times, respectively between 1960 and 2016 (based on the World Bank Database, 2017). Population / demographic drivers consider changes in population size, migration flows, urbanization as well as demographic variables such as population distribution and age structure. Urbanisation driven by growing populations and internal migration acts as an indirect driver of land-use change through various ways, including through linear infrastructures such as transportation networks as well as synergies with other forms of infrastructure development (IPBES, 2016b). By 2050, all archetype scenarios project great increase in human population size, while towards the end of the century, downward trends are projected for the “economic optimism” (SSP5), “global sustainable development” (SSP1), “reformed markets” scenarios **(Table 4.1.2, Figure 4.1.2)**.

Per capita GDP trends combine the impacts of GDP and population growth on environment. Growing per capita GDP has historically implied increasing demand of key natural resources such as food, water and energy with adverse impacts on ecosystems and biodiversity, due to the persistence of unsustainable patterns of production and consumption. Humanity’s demand has exceeded the planet’s biocapacity for more than 40 years, and the Ecological Footprint shows that 1.6 Earths would be required to meet the demands humanity makes on nature each year, with consumption patterns in high-income countries resulting in disproportional demands on renewable resources, often at the expense of people and nature elsewhere in the world (WWF, 2016).

Technology development can significantly increase the availability of some ecosystem services, and improve the efficiency of provision, management, and allocation

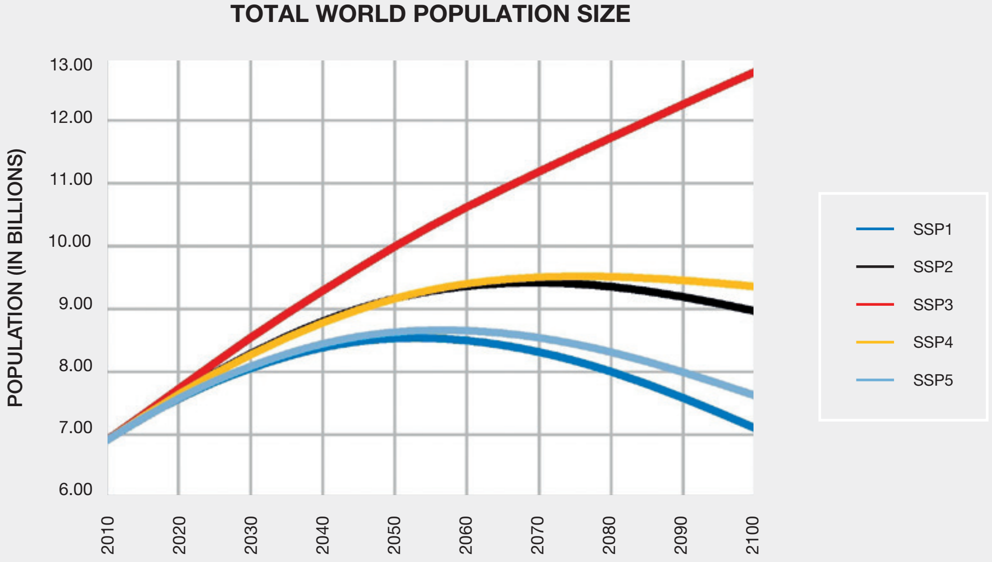


Figure ④ ① ② **Projected changes in world population according to the five Shared Socio-economic Pathways (KC & Lutz, 2017).**

Note: For the narratives of the SSPs, see O'Neill *et al.* (2017).

of different ecosystem services, but it cannot serve as a substitute for all ecosystem services. Technologies associated with agriculture and other land uses have a large impact as drivers of biodiversity and ecosystem change (IPBES, 2016a).

As part of the problem, some technologies can result in increased pressure on ecosystem services through increased natural resource demand as well as lead to unforeseen ecological risks, particularly natural resource intensive technologies, as those associated to agricultural land expansion (e.g., first generation of biofuels when produced unsustainably). In addition, climate change is directly related to the use of fossil-fuel-intensive technologies. As part of the solution, sustainability-oriented technological innovation may contribute to decouple economic growth and the consumption of natural resources through increasing efficiency, resilience and equity (e.g. agroecological food production systems) (IPBES, 2016a; Trace, 2016; Vos & Cruz, 2015).

Governance and institutions play an important role in the management of biodiversity, ecosystem services and ecosystem functions. Weak governance, including corruption, frequently leads to environmental mismanagement as well as the adoption of environmentally unsustainable policies, and growing conflicts (Pichs-Madruga *et al.*, 2016). The lack of recognition of indigenous and local knowledge (ILK) and institutions may also generate adverse consequences for nature, NCP and GQL as well as for Indigenous Peoples and Local Communities (IPLCs). In addition to governments, new actors and coalitions (e.g. NGOs, researchers, indigenous groups) with different – and sometimes divergent and conflicting – perceptions and values are performing critical roles in environmental decision-making processes.

Social development and culture are critical ingredients of future scenarios on biodiversity, yet there is a lack of attention towards understanding how values, norms, and beliefs affect attitudes and behaviours towards the environment, and their roles in shaping the future and in driving transformation pathways. While there has been advances in methodologies supporting social-ecological analyses, emphasis has been on measurable indicators with less attention to the role of sociocultural values and practices in shaping other indirect drivers of change, and thus future pathways (Pichs-Madruga *et al.*, 2016).

Social inequity is a key concern in many regions, sub-regions, countries and territories. In many cases, poverty conditions correlate with increasing pressures on nature, but globally per capita consumption of natural resources is strongly correlated with affluence. World per capita private consumption, in dollars at constant 2010 prices, rose by 44.5% between 1990 and 2016 (Worldbank, 2017). The emergence of new waves of affluent consumers is projected to significantly increase the demand for already limited natural resources (Myers & Kent, 2003). For this reason, the impact of consumers' purchasing power on the demand of natural resources is receiving growing attention in scenarios. This discussion is very relevant in the context of the global debate on the Sustainable Development Goals (SDGs), multidimensional progress in human development (UNDP, 2016) and their interlinkages with nature and NCP.

4.1.4.2 Direct Drivers

Climate change

By the end of the 21st century, three of four explored Representative Concentration Pathways (RCP; van Vuuren *et al.*, 2011) result in an increase in global average surface temperatures above 1.5°C compared to the present-day reference period 1986-2005 (Stocker *et al.*, 2013). Averaged over years 2046-2065, temperature increases range from (model median) 1.4°C (RCP4.5) to 2.0°C (RCP8.5) above the reference period (1986-2005). Only the RCP2.6 scenario could possibly lead to a below 2°C world, with projected warming above the reference period from 0.3 to 1.7°C averaged over the last two decades of the 21st century, and from 0.4-1.6°C for years 2046-2065. Warming will be larger over land and by far highest in the Arctic. The frequency of extreme hot weather events will increase (Stocker *et al.*, 2013). Precipitation patterns will change in a complex, spatially non-uniform way.

Based on climate modelling done for the IPCC 5th assessment report, and recent work presented in the IPCC special report on 1.5 degrees (IPCC, 2018), limiting warming to 1.5°C above preindustrial levels will require rapid, historically unprecedented mitigation efforts (Millar *et al.*, 2017). Applying a different, statistical modelling approach found below 2°C warming at the end of the 21st century unlikely, and requiring a much accelerated decline in carbon intensity compared to the past decades (Raftery *et al.*, 2017). By 2050, in the RCP2.6 pathway, CO_2 emissions are projected to be lower than they were in 1990. Projected atmospheric concentrations range from ca. 440 ppm (RCP2.6) to ca. 540 ppm (RCP8.5) by 2050 to ca. 420-935 ppm by 2100, but uncertainties are of several tens/hundreds of ppm.

Land-use change

Land-use and land-cover changes have direct and large impacts on the physical environment. They include expansion of crops and pastures, as well as intensification and management changes, mineral and biomass extraction, urbanization and infrastructure expansion (Geist & Lambin, 2002). Eitelberg *et al.* (2015) estimated the global potential for crop area to range from ca. present-day expanse (1500 Mha) to nearly a tripling (5100 Mha), depending on different future socio-economic and governance assumptions. Synthesising projected future crop, pasture and forest areas, Alexander *et al.* (2017c) showed a huge spread in projected future land-use change, and found that this spread depended on the type of scenario, as could be expected, but also was heavily dependent on the type of model used to quantify land use for a given scenario (i.e. the same scenario archetype results in very different land-use change patterns depending on the underlying model's assumptions and structure). Overall, these studies suggest that there remains a high level of uncertainty in future land-use change potential and in scenarios of land-use change.

The five main SSP storylines that have been developed in support of the IPCC can be classified by archetypes **(Table 4.1.2)**, but considerable caution should be exercised when interpreting land-use projections from the SSP storylines as being representative of a particular archetype. For example, the largest declines in global area of forest and other natural land occur in the reference scenarios (also referred to as "marker scenarios") for SSP3, SSP4 and SSP5 (Popp *et al.*, 2017), i.e. scenarios that emphasise competition or free markets. However, the range of variation of the projected change in managed land area by 2100 is nearly as large within SSPs (i.e. variation due to application of different IAMs to the same SSP storyline) as it is between marker scenarios across SSPs (Popp *et al.*, 2017). Given this large variation within SSPs and high uncertainty in land-use projections identified by Alexander *et al.* (2017c), considerable caution must be exercised when making the connection between the underlying assumptions of scenario archetypes **(Tables 4.1.1 and 4.1.4)** and an individual projection of land use by a single Integrated Assessment Model (e.g., **Figure 4.1.3**).

In the wake of the Paris COP21 agreement, terrestrial ecosystems will make crucial contributions to meeting agreed climate mitigation objectives. Achieving the RCP2.6 pathway (or the most recent RCP1.9 pathway, see IPCC, 2018) requires, in nearly all scenarios developed with IAMs, negative emissions through carbon-dioxide removal. The majority of this is generally achieved through reforestation, afforestation and avoided deforestation, as well as bioenergy plantations coupled with carbon capture and storage (Anderson & Peters, 2016; Smith *et al.*, 2016). Depending on how fast fossil fuel emissions decline, substantial negative emissions to balance continued fossil emissions need to be achieved by 2050, or even earlier (Anderson & Peters, 2016) which, if implemented, will have large consequences for terrestrial ecosystems. Recent results indicated that SSPs 1, 2, 4 and 5 might be consistent with low greenhouse gas emissions (i.e., RCP2.6; Kriegler *et al.*, 2014; Popp *et al.*, 2017) (see also examples in **Figure 4.1.3**). Despite the very different assumptions contained in the SSPs (and in the IAMs simulating these) there is consistent projected decline in food crop and pasture area at the end of the 21st century, even though demand for crop and livestock products tend to be larger than today. At the same time, area under bioenergy plantation increases by between ca. 200 Mha (SSP1/AIM) and 1500 Mha (SSP4/GCAM4).

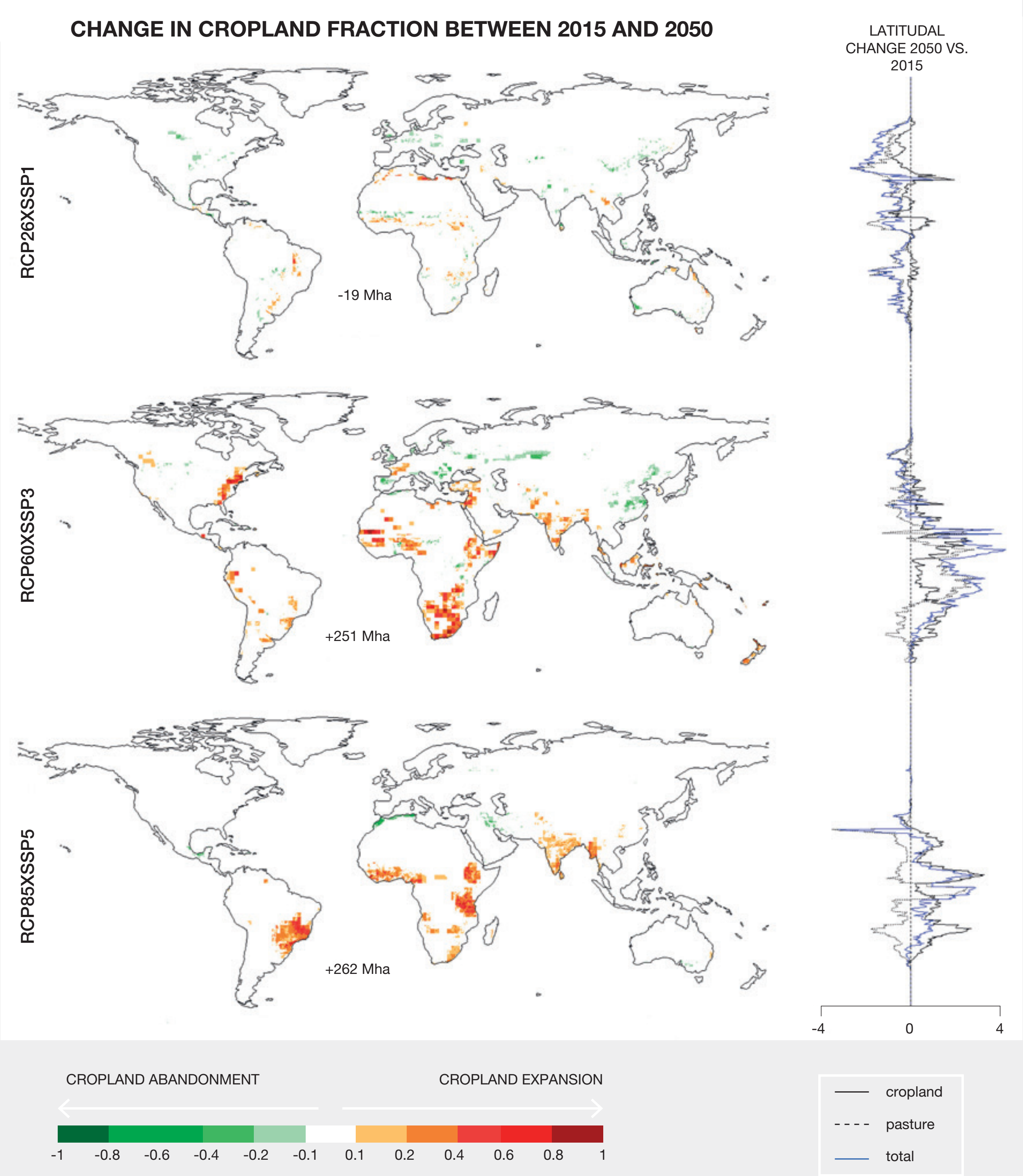


Figure ④ ① ③ **Projected changes in cropland area used in the Biodiversity and Ecosystem Services Model Intercomparison Project (BES-SIM, Kim *et al.*, 2018; see box 4.2.5).**

Note that the depicted projections are from a single Integrated Assessment Model per SSP and may not be representative of the range of land-use projections for archetypes to which the SSP is assigned **(Table 4.1.2)**. For instance, trends in agricultural area used by Krause *et al.* (2017, 2018) for their RCP2.6 baseline case were opposite (increasing agricultural area) to the trends seen in the scenario RCP2.6/SSP1 used in BES-SIM. A map with cropland as well as pasture area changes is provided in Appendix 4.1 (Figure A4.1.12).

The intensity of land-use change can be as important as the change in area. In particular, the productivity of croplands is assumed to increase in the future as a result of increased application of technology, including the use of fertilizers, high producing varieties, machinery and pesticides. Intensification has huge impacts on biodiversity in agricultural landscapes, where for example species richness reduces by more than 50% in intensively used croplands, compared to low input systems (e.g., Newbold *et al.*, 2015). Intensification will continue in the coming decades and a recent analysis for the SSP scenarios showed trade-offs between land-use change and intensification **(Table 4.1.6)**.

To meet the demand of a growing and wealthier population, increased agricultural production results from land conversion to cropland in the SSP3/RCP6.0 and SSP5/RCP8.5 scenarios and from intensification in all scenarios, where in SSP3/RCP6.0 scenario a relatively low increase of the yield is assumed.

Pollution

Pollution here refers to solid and chemical waste of various kinds, excluding the gases referenced in the Kyoto and Montreal Protocols. Large increases in waste generation have occurred in the past decades, with a particular challenge for persistent organic pollutants (POPs) and synthetic organic polymers (plastics) which are physically harmful, chemically toxic, and slow to metabolize (see 4.2.2.4.1). Solid waste generation rates depend strongly on urban population growth trends, together with changing standard of living and societal efforts towards waste reduction. On current trends, waste production will attain 11 Mt day^{-1} by 2100, and will continue to rise into the latter half of this century particularly in sub-Saharan Africa (Hoornweg *et al.*, 2013). However, socio-economic pathways could strongly affect waste production trends, with SSP1 stabilising global waste production by about 2070 at roughly 8.5 Mt day^{-1} relative to values of 12 Mt day^{-1} in SSP2 and SSP3 (Hoornweg *et al.*, 2013).

Direct harvesting of natural resources

Scenarios relating to direct harvesting will have complex relationships with distinct socio-economic futures. In terrestrial ecosystems, while an increase in human wealth may reduce direct harvesting of provisioning resources (such as bushmeat), increasing wealth may increase demands for some traditional (e.g. medicinal) and “luxury” (e.g. Rhino horn) resources. On the other hand, marine and freshwater natural resources might undergo increased fishing pressure in the face of rising affluence and continuous growth of human population that is projected to reach 9.8 billion people by 2050 (UNDESA, 2017). Scenarios of governance in fisheries management, human consumption of seafood, improvement of fishing technology (Squires & Vestergaard, 2013) are starting to be integrated into future global scale projections (section 4.2.2.3).

Invasive Alien Species

Invasive alien species (IAS) are those that have been moved by direct human actions beyond their native geographic range, and have established and actively expand geographic range after introduction (Blackburn *et al.*, 2014). The main impacts of socio-economic scenarios on IAS are likely to be through vectors for dispersal (with international trade and long-distance transport being the most important), and economic resources to combat IAS. Higher impacts are thus to be expected under future scenarios of greater global trade with weaker local governance.

Quantification of the impacts of IAS tends to focus on adverse ecological effects (Simberloff *et al.*, 2013), including adverse impacts on ecosystem services. It is thus difficult to develop a fully integrated understanding of positive, neutral and negative impacts, though current consensus strongly suggests overall adverse impacts (Pyšek & Richardson, 2010). For example, invasive plants can cause catastrophic regime shifts and indigenous diversity reduction (Gaertner *et al.*, 2014), such as through N-fixing species increasing N concentrations in nutrient-poor soil (Blackburn *et al.*, 2014), and by increasing fire frequencies and intensities, or even introducing novel fire regimes (Pausas & Keeley, 2014).

Table 4 1 6 **Changes in global cropland area and productivity increase for three SSP scenarios, as analysed in a model comparison study by BES-SIM.**

	SSP1/RCP2.6	SSP3/RCP6.0	SSP5/RCP8.5
Cropland in 2015 in km^2	15885409	15885409	15885409
Cropland in 2050 in km^2	15696191	18399153	18507559
Cropland area increase 2015-2050 %	-1.2	15.8	16.55
Crop production increase 2015-2050 %	31.7	40.5	58.4
Yield increase 2015-2050 %	33	21	36
Yield increase per year %	0.95	0.61	1.03

Invasive animals may cause extreme indigenous diversity loss particularly if they are predators and invade in islands (Medina *et al.*, 2011).

The number of documented IAS is most probably a significant underestimate of the true number, partly because of inadequate research effort particularly in some developing countries with potentially high IAS densities (McGeoch *et al.*, 2010). The IUCN Red List Index indicates that the adverse impacts of IAS include increased rates of decline in species diversity (McGeoch *et al.*, 2010).

Disturbance

Disturbance is a fundamental driver of biodiversity, and ecosystem structure and function, and may strongly control ecosystem services delivered. Almost all ecosystems experience episodic events like floods, droughts and wildfire. Where disturbance is frequent enough, natural selection both permits nature to adapt, and some species may even become dependent on disturbance, and enhance its frequency (Parr *et al.*, 2014). A prime example is wildfire, which is of global significance in that it is an important factor in determining local to landscape scale ecosystem structure over vast areas of the subtropics and tropics. Without fire, ecosystem structure and function in fire-prone regions may alter their biodiversity, structure and function entirely (Bond *et al.*, 2005). Many plant species are designed to accelerate fire frequency and intensity (Keeley *et al.*, 2011). Disturbance is thus an important tool available in the management of biodiversity, ecosystem structure and function, and the ecosystem services that result (Folke *et al.*, 2004). Disturbance is likely to be most strongly affected by climate (especially in case of fire) as well as socio-economic scenarios. Fire, droughts and flooding would be expected with higher frequency under low future climate change mitigation scenarios. However, for fire it has been argued that changes in human population density, and shifts in urban to rural lifestyles affect future burnt area to the same degree as climate change, through reducing fire spread (Knorr *et al.*, 2016). However, as more people are projected to live in fire-prone areas, potentially detrimental impacts on societies may nonetheless increase (Knorr *et al.*, 2016).

4.1.5 Considering Indigenous Peoples and Local Communities (IPLCs) and indigenous and local knowledge (ILK) in scenarios

The integration of indigenous and local knowledge (ILK) into scenarios developed at the regional and global scales, as well as the assessment of the impacts of scenarios on Indigenous Peoples and Local Communities (IPLCs), have been limited and remain a key challenge in scenario development (Hill *et al.*, 2012; Wohling, 2009). Varying combinations of indirect drivers, and especially government policy, can disproportionately impact IPLCs and their livelihoods. This is particularly significant when considering scenarios as alternative policy or management options intended to alter the future state of these (system) components (IPBES, 2016b). The following examples provide evidence for the potential benefits that could be gained from a better recognition of and respect for ILK and IPLCs in conservation of nature, as well as adaptation to and mitigation of climate change.

Government policies that (i) define agro-industrial plantations as forests, (ii) change property systems, including privatization and land titling over areas of customary tenure, and (iii) incentivize migration to historically low population density areas, undermine ILK that promote biodiversity and human well-being, and traditional land-use practices (Dressler *et al.*, 2017).

Some cases where governments have recognized IPLC land rights and pursued climate mitigation policies, such as through REDD+ projects (Reducing Emissions from Deforestation and Forest Degradation), have led to thus-far successful collaborations and demonstrated that ILK could make significant contributions to future forest and biodiversity conservation (see also review in chapter 6). For instance, the case of GuateCarbon, which incorporates the Association of Forest Producers of Petén (ACOFOP, in northern Guatemala) as full partners alongside government entities and international NGOs, has proved a potentially important model for negotiation, benefit sharing, and monitoring, reporting, and verification that respects local land-use practices and values (Hodgdon *et al.*, 2013). Positive livelihood outcomes have accompanied a pattern of strong forest protection in areas with community-led management here.

Studies suggest that policy scenarios such as protected area designation – including territorial recognition for IPLCs – could play a significant role in avoiding future deforestation, such as in the Amazon, despite continued pressures to downgrade, downsize, and degazette protected areas (PADDD) for infrastructure development and more intensive land uses (Forrest *et al.*, 2015; Soares-Filho *et al.*, 2010). For example, a recent Brazilian moratorium on mega-dams – long demanded by indigenous groups on ecological and spiritual grounds – could enhance ecosystem protection, especially if accompanied by increased support for forest groups (Branford, 2018), despite continuing plans for inter-modal transport projects essentially promoting agro-industry and colonization (Molina *et al.*, 2015). While the Brazilian Amazon has served as an important testing ground for recognizing the importance of ILK in forest management and for REDD+, the continued discounting of ILK systems in broader land-use policy throws doubt on the long-term viability of such participative initiatives (Cromberg *et al.*, 2014; Vitel *et al.*, 2013). Specific major drivers vary by country and by region, but global demand for basic commodities and national enabling environments for investment in forest-rich countries will likely continue to contribute to terrestrial emissions and biodiversity loss – including through incursions on IPLCs' traditional lands and the attendant loss of ILK. Thus, even where REDD+ and conservation initiatives have tried to ensure community participation, they achieve variable success, in part because they often fail to address the strongest indirect drivers of losses of forests, biodiversity and ecosystem services (Angelsen *et al.*, 2017).

Notwithstanding these limits, the long period of negotiation over the program internationally and nationally, in addition to a pivot away from market-based approaches implementation, has provided IPLCs with opportunities to insert their priorities (tenure security, Free, Prior and Informed Consent, social services) into the debate (Angelsen *et al.*, 2017; Van Dam, 2011). Increasing rates of recognition of IPLCs' rights to inhabit and manage their lands alongside new sources of dedicated funding (such as the UNFCCC's Green Climate Fund) could suggest stronger outcomes for avoided deforestation and ecosystem health.

4.2 PLAUSIBLE FUTURES FOR NATURE

4.2.1 Impacts of future global changes on biodiversity: feedbacks and adaptation capacity

4.2.1.1 Projected negative changes at all levels of biodiversity

The scientific community has focused on climate change as a major driver of concern in exploring possible futures for nature **(Table 4.2.1)**. Based on our systematic literature review (Appendix A4.1.1), 88% of the global scenario literature addressed climate change impacts on nature, followed by 8% and 2% of the papers addressing land-use change and natural resource extraction, respectively. A vast majority of the papers addressed single drivers, as few integrated models are able to represent combination of drivers and interactions are more complex to implement (IPBES, 2016b). Of all the scenarios exploring climate change impacts, only 18% were combined with other direct drivers of change such as land use or natural resource extraction.

Table 4 2 1 **Major drivers represented in global change scenarios addressing impacts on nature at global scale, across terrestrial, freshwater and marine ecosystems.**

The number of scenarios published is reported, and in parentheses, the number of scientific papers from the Chapter 4 literature database (Appendix A4.1.1). Scenarios addressed single drivers (purple cells) or combination of drivers.

	Climate change	**Invasive alien species**	**Land-use change**	**Natural resource extraction**	**Pollution**	**Others**
Climate change	569(270)	4(3)	104(36)	12(6)	8(4)	11(8)
Invasive alien species		10(2)				
Land-use change			45(19)	7(4)	4(2)	1(1)
Natural resource extraction				16(7)	1(1)	
Pollution					1(1)	1(1)
Others						27(8)

Most scenarios of biodiversity change are terrestrial or marine, while far fewer exist for freshwater (**Figure 4.2.1**; IPBES, 2016b). Therefore, most evidence provided in section 4.2.3 for freshwater biomes is based on local and regional studies. Overall, relatively few metrics of biodiversity and ecosystem function have been explored deeply enough to draw strong conclusions about their interactions in a globally changing environment.

The systematic literature review indicates that the effects of global environmental changes on biodiversity are mostly projected to be negative **(Figure 4.2.1)** and embrace all biodiversity levels – from genetic diversity to biomes (Bellard *et al.*, 2012; **Box 4.2.1**). Marine systems are projected to be generally more negatively impacted by global change drivers than terrestrial systems **(Figure 4.2.1)**. For example, projected changes in species biomass or abundance cover the spectrum of negative to positive trends in terrestrial systems (see evidence provided in sections 4.2.4.1 to 4.2.4.4), but negative trends stand out in marine systems (see section 4.2.2). There are a few metrics, such as terrestrial C pools or organisms' growth, where positive trends are the most common response in the literature (see 4.2.4.1). In case of C-pools this reflects chiefly the impact of CO_2 on photosynthesis and growth, which in some models outpace the impacts of warming. In boreal and temperate regions, climate change was also shown to possibly have positive effects on organisms' growth, e.g., plant growth (Pretzsch *et al.*, 2014). All other metrics of biodiversity and ecosystem function are dominated by projected neutral or negative trends in response to projected global change drivers. Negative trends are particularly dominant for indicators of production, reproduction success, terrestrial species richness and extinction, marine species biomass and abundance, and the area and quality of marine habitats.

Figure ④ ② ① **Future trends of selected indicators in marine Ⓐ, terrestrial Ⓑ and freshwater ecosystems Ⓒ, based on global scale scenarios referenced in the literature database (Appendix A4.1.1), all drivers combined.**

The results are extracted from scenarios with increasing pressures from direct drivers (all climate change scenarios and business-as-usual scenarios for resource exploitation, land-use change and pollution). The selected scenarios were at global scale. Regional/local scale scenarios were not referenced in the literature database. Colours code the projected trends in the indicators. N=the number of trends reported and in parentheses the number of papers.

A substantial fraction of wild species is predicted to be at risk of extinction during the 21st century due to climate change, land use and impact of other direct drivers (Bellard *et al.*, 2012; Pimm *et al.*, 2014; Settele *et al.*, 2014; see sections 4.2.2-4.2.4). In a recent review of published future global extinction risk, Urban (2015) found that extinction risk is projected to increase from 2.8% at present to 5.2% at the international policy target of a 2°C post-industrial rise, to 8.5% if the Earth warms to 3°C, and to 16% in a high greenhouse gas emissions scenario (RCP 8.5; 4.3°C rise). Extinctions might not occur immediately but after substantial delay called because when a population has been reduced to very small numbers, it has a high risk to go extinct at some point in the future (referred to as «extinction debt»). This means that long-term effects of global change can be much more severe than short term impacts (Cronk *et al.*, 2016; Dullinger *et al.*, 2012; Fordham *et al.*, 2016; Hylander & Ehrlén, 2013).

Notwithstanding a majority of expected negative impacts of future climate change on biodiversity, **Figure 4.2.1** suggests the potential for some positive effects in species distributions areas and species richness. General poleward movement of marine and terrestrial species and upward movement of terrestrial mountain species may lead to increase in local species richness in high latitudes and in mountainous regions, while the opposite is projected in the tropics and flat landscapes (Gilg *et al.*, 2012; Jones & Cheung, 2015; Settele *et al.*, 2014; Thuiller *et al.*, 2014).

Global scale scenarios can mask the spatial heterogeneity of projected biodiversity response at finer scales (Urban, 2015; Vellend *et al.*, 2017). For example, the highest species extinction risk due to climate and land-use changes is projected in the tropics and polar regions as well as in top mountain habitats because of projected "novel" climates in tropics that these regions have never experienced in the past (Mora *et al.*, 2013a), narrow physiological tolerances of tropical and polar species, expected disappearance of polar and top-mountain habitats (Deutsch *et al.*, 2008; Gilg *et al.,* 2012; Mora *et al.*, 2013a; Pörtner *et al.*, 2014; Settele *et al.*, 2014) and the highest risk of conversion of ecosystems to crops and biofuel in the tropics (Kehoe *et al.*, 2017; Newbold *et al.*, 2015). Biodiversity hotspots are also projected as subject to high species extinction (Bellard *et al.*, 2014; see 4.2.2, 4.2.3, 4.2.4).

To account for the spatial differentiation of global changes impacts on nature, the following sections 4.2.2, 4.2.3, and 4.2.4 cover the outcomes of the literature database analysis (Appendix A4.1.1), but also include detailed examination of key studies and specific biomes (IPBES units of analysis). The major drivers of change and the primary impacts differ depending on the biome considered **(Figure 4.2.2)**, and therefore need to be addressed by specific, and sometimes local, adaptation and mitigation policies.

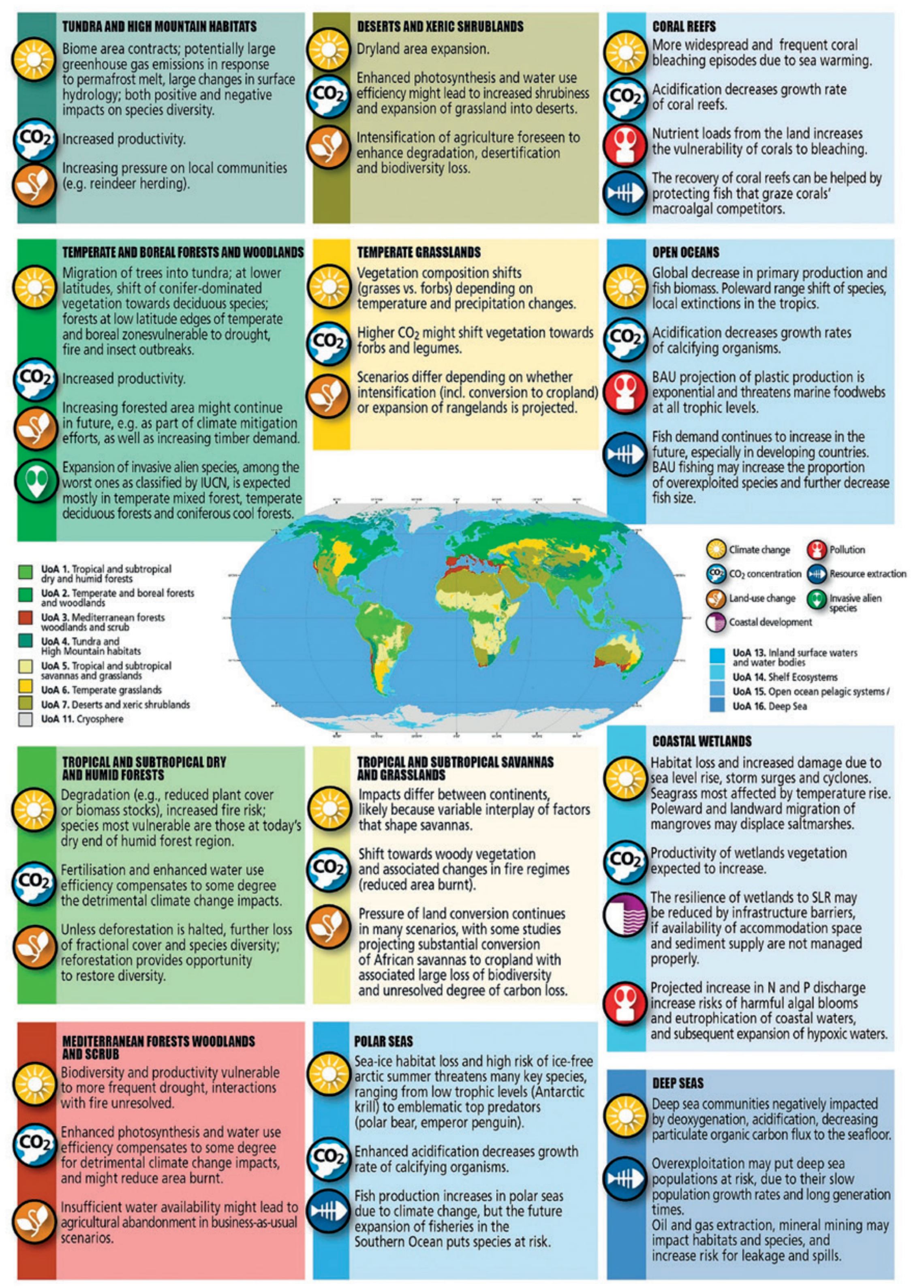
4.2.1.2 Future biodiversity adaptation and reorganisation

Species can respond to environmental changes in many different ways that are not mutually exclusive. In response to changes in climate, species can adapt to new conditions, they can shift their geographical distribution following optimal environmental gradients or can go locally extinct.

A large number of scenarios explore ***species distribution shifts***. Terrestrial species may respond to climate changes by shifting their latitudinal and elevation ranges. Marine species may respond by shifting their latitudinal and depth ranges. Models predict latitudinal range shifts for plant and animal species of hundreds of km over the next century as well as significant range contraction and fragmentation (Leadley *et al.*, 2010; Markovic *et al.*, 2014; Meller *et al.*, 2015; Rondinini & Visconti, 2015; Warren *et al.*, 2013). Comparisons of projected climate velocity (the rate of movement of the climate across a landscape) and species displacement rates across landscapes showed that many terrestrial species (e.g., plants, amphibians, and some small mammals) will be unable to move fast enough to track suitable climates under medium and high rates of climate change (i.e. RCP4.5, RCP6.0, and RCP8.5 scenarios). Most species will be able to track climate only under the lowest rates of climate change (RCP2.6) (Settele *et al.*, 2014). Natural geographical barriers (Burrows *et al.*, 2014) and human-made habitat disruptions are predicted as important factors limiting movement of species ranges (Meier *et al.*, 2012; Schloss *et al.*, 2012).

Species adaptation to novel conditions is likely to mitigate the predicted impacts of global changes (Hoffmann & Sgrò, 2011; Lavergne *et al.*, 2010; Neaves *et al.*, 2015; Pauls *et al.*, 2013; Skelly *et al.*, 2007). Models that ignore adaptation may overestimate extinction probabilities. For example, the inclusion of local adaptations due to phenotypic plasticity and microevolution in models of terrestrial carnivore and ungulate species decreases the expected decline in population abundance by 2050, from 31–34% to 18% (Visconti *et al.*, 2016; see **Box 4.2.1**)

Intraspecific diversity of behavioral, phenological, physiological and morphological traits allows populations and species to survive under rapid climate change through standing genetic variation (GD1 in **Box 4.2.1**), and provides material for selection in new conditions (Alfaro *et al.*, 2014; Hof *et al.*, 2011; Jump *et al.*, 2009). On the one hand, incorporating intraspecific variation in species models increases the likelihood of their survival as shown for several tree species (Benito Garzón *et al.*, 2011; Morin & Thuiller, 2009; Oney *et al.*, 2013). On the other hand, projections that do not consider probable loss of intraspecific diversity can underestimate future negative effects on biodiversity. The loss of genetic diversity is projected for a number of



TUNDRA AND HIGH MOUNTAIN HABITATS

Biome area contracts; potentially large greenhouse gas emissions in response to permafrost melt, large changes in surface hydrology; both positive and negative impacts on species diversity.

Increased productivity.

Increasing pressure on local communities (e.g. reindeer herding).

DESERTS AND XERIC SHRUBLANDS

Dryland area expansion.

Enhanced photosynthesis and water use efficiency might lead to increased shrubiness and expansion of grassland into deserts.

Intensification of agriculture foreseen to enhance degradation, desertification and biodiversity loss.

CORAL REEFS

More widespread and frequent coral bleaching episodes due to sea warming.

Acidification decreases growth rate of coral reefs.

Nutrient loads from the land increases the vulnerability of corals to bleaching.

The recovery of coral reefs can be helped by protecting fish that graze corals' macroalgal competitors.

TEMPERATE AND BOREAL FORESTS AND WOODLANDS

Migration of trees into tundra; at lower latitudes, shift of conifer-dominated vegetation towards deciduous species; forests at low latitude edges of temperate and boreal zonesvulnerable to drought, fire and insect outbreaks.

Increased productivity.

Increasing forested area might continue in future, e.g. as part of climate mitigation efforts, as well as increasing timber demand.

Expansion of invasive alien species, among the worst ones as classified by IUCN, is expected mostly in temperate mixed forest, temperate deciduous forests and coniferous cool forests.

TEMPERATE GRASSLANDS

Vegetation composition shifts (grasses vs. forbs) depending on temperature and precipitation changes.

Higher CO_2 might shift vegetation towards forbs and legumes.

Scenarios differ depending on whether intensification (incl. conversion to cropland) or expansion of rangelands is projected.

OPEN OCEANS

Global decrease in primary production and fish biomass. Poleward range shift of species, local extinctions in the tropics.

Acidification decreases growth rates of calcifying organisms.

BAU projection of plastic production is exponential and threatens marine foodwebs at all trophic levels.

Fish demand continues to increase in the future, especially in developing countries. BAU fishing may increase the proportion of overexploited species and further decrease fish size.

- **UoA 1.** Tropical and subtropical dry and humid forests
- **UoA 2.** Temperate and boreal forests and woodlands
- **UoA 3.** Mediterranean forests woodlands and scrub
- **UoA 4.** Tundra and High Mountain habitats
- **UoA 5.** Tropical and subtropical savannas and grasslands
- **UoA 6.** Temperate grasslands
- **UoA 7.** Deserts and xeric shrublands
- **UoA 11.** Cryosphere

- Climate change
- Pollution
- CO_2 concentration
- Resource extraction
- Land-use change
- Invasive alien species
- Coastal development

- **UoA 13.** Inland surface waters and water bodies
- **UoA 14.** Shelf Ecosystems
- **UoA 15.** Open ocean pelagic systems /
- **UoA 16.** Deep Sea

TROPICAL AND SUBTROPICAL DRY AND HUMID FORESTS

Degradation (e.g., reduced plant cover or biomass stocks), increased fire risk; species most vulnerable are those at today's dry end of humid forest region.

Fertilisation and enhanced water use efficiency compensates to some degree the detrimental climate change impacts.

Unless deforestation is halted, further loss of fractional cover and species diversity; reforestation provides opportunity to restore diversity.

TROPICAL AND SUBTROPICAL SAVANNAS AND GRASSLANDS

Impacts differ between continents, likely because variable interplay of factors that shape savannas.

Shift towards woody vegetation and associated changes in fire regimes (reduced area burnt).

Pressure of land conversion continues in many scenarios, with some studies projecting substantial conversion of African savannas to cropland with associated large loss of biodiversity and unresolved degree of carbon loss.

COASTAL WETLANDS

Habitat loss and increased damage due to sea level rise, storm surges and cyclones. Seagrass most affected by temperature rise. Poleward and landward migration of mangroves may displace saltmarshes.

Productivity of wetlands vegetation expected to increase.

The resilience of wetlands to SLR may be reduced by infrastructure barriers, if availability of accommodation space and sediment supply are not managed properly.

Projected increase in N and P discharge increase risks of harmful algal blooms and eutrophication of coastal waters, and subsequent expansion of hypoxic waters.

MEDITERRANEAN FORESTS WOODLANDS AND SCRUB

Biodiversity and productivity vulnerable to more frequent drought, interactions with fire unresolved.

Enhanced photosynthesis and water use efficiency compensates to some degree for detrimental climate change impacts, and might reduce area burnt.

Insufficient water availability might lead to agricultural abandonment in business-as-usual scenarios.

POLAR SEAS

Sea-ice habitat loss and high risk of ice-free arctic summer threatens many key species, ranging from low trophic levels (Antarctic krill) to emblematic top predators (polar bear, emperor penguin).

Enhanced acidification decreases growth rate of calcifying organisms.

Fish production increases in polar seas due to climate change, but the future expansion of fisheries in the Southern Ocean puts species at risk.

DEEP SEAS

Deep sea communities negatively impacted by deoxygenation, acidification, decreasing particulate organic carbon flux to the seafloor.

Overexploitation may put deep sea populations at risk, due to their slow population growth rates and long generation times.
Oil and gas extraction, mineral mining may impact habitats and species, and increase risk for leakage and spills.

Figure ④ ② ② **Examples of future projected impacts of major drivers of change on nature (supporting evidence in sections 2.2 and 2.4 of the chapter, and Table A4.2.1 in Appendix 4.2).**

Examples are given for IPBES terrestrial and marine units of analysis (UoA).

species belonging to very different terrestrial and aquatic taxa and thus, should be recognized as a serious threat to future biodiversity rescue (Bálint *et al.*, 2011; Jump *et al.*, 2009; Neaves *et al.*, 2015; Pauls *et al.*, 2013).

Phenotypic plasticity helps to reduce the risk of species extinction (GD2 in **Box 4.2.1**) allowing a rapid (within individual's lifetime) adjustment of populations to novel conditions whereas evolutionary responses require several generations (Chevin *et al.*, 2010). Incorporating phenotypic plasticity in models predicting future species' distributions reduced the extinction risk in southern populations of several species (Benito Garzón *et al.*, 2011; Morin & Thuiller, 2009).

Rapid adaptive evolution (GD3 in **Box 4.2.1**) occurring at similar time scale as global environmental change has the potential for "evolutionary rescue", i.e. population survival *in situ* due to ongoing selection of standing genetic variations as well as relatively slower selection of new mutations (Gonzalez *et al.*, 2013; Hendry *et al.*, 2011; Hoffmann & Sgrò, 2011; Settele *et al.*, 2014). However, evolutionary responses may be too slow for species with low capacity for adaptive evolution, especially under large-scale and rapid environmental changes (Gienapp *et al.*, 2012; Jump *et al.*, 2006).

Adaptation can cascade to entire communities or ecosystems, thus maintaining community properties beyond the level of change in the driver. However, adaptive capacity is not unlimited and so even evolving systems can eventually switch to a new state if a change in a driver is too severe or too rapid. Return to the original system state when change pressure is removed to the original state can be harder than would have been the case without evolution, due to the depletion of the genetic variation **(Figure 4.2.3)**.

Along with the vital importance of preserving the short-term adaptive capacity of biodiversity, the necessity of *long-term maintenance of further evolutionary* processes generating biodiversity and potential future ecosystem services was recognized as a key goal that requires preservation of evolutionary heritage and phylogenetic diversity of the Tree of Life (Faith, 2015; Faith *et al.*, 2010; Forest *et al.*, 2007; Mace & Purvis, 2008).

Reorganization of ecological communities and novel communities: Substantial changes in *species composition and biotic interactions* are expected due to shifts in species distribution (S1 in **Box 4.2.1**), local species extinctions, alterations of species abundance, functioning and phenology (S2 in **Box 4.2.1**). Projected changes in species composition can lead to disruptions of food webs and mutualistic relationships, increased prevalence of pests and pathogens, introductions of alien species, biotic homogenization and loss of biological uniqueness of communities (Blois *et al.*, 2013; Buisson *et al.*, 2013; Thuiller *et al.*, 2014).

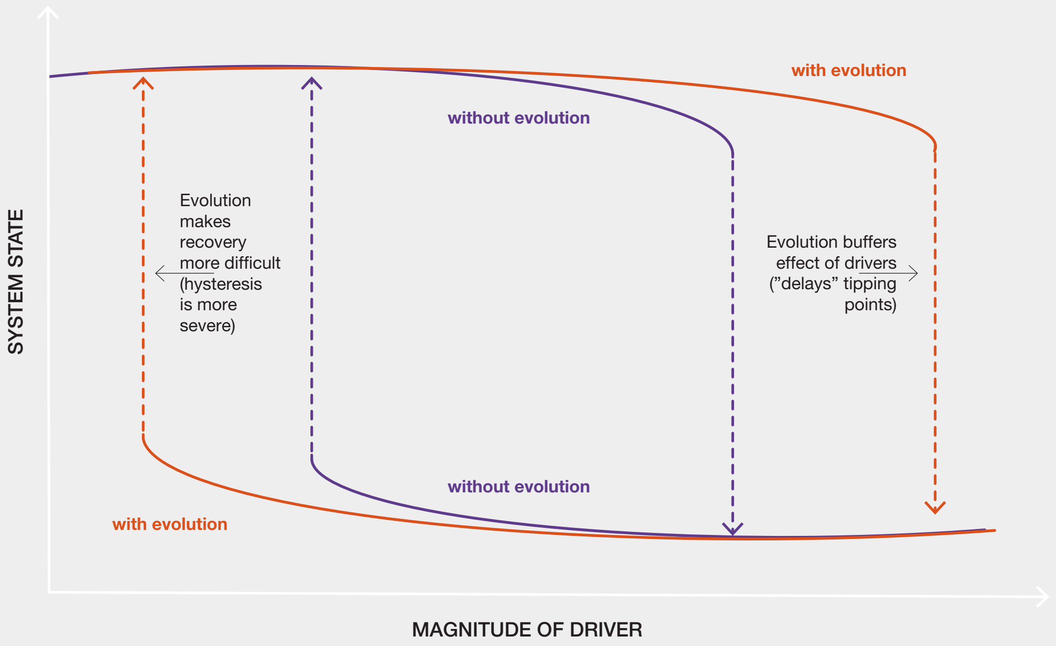


Figure ④ ② ③ **Potential role of evolution (more generally, "adaptive capacity") in mediating tipping points, alternative stable states, and hysteresis.**

Novel (no-analog) communities, in which species will co-occur in historically unknown combinations, are expected to emerge (Ordonez *et al.*, 2016; Radeloff *et al.*, 2015; Williams & Jackson, 2007). Novel communities are expected to become increasingly homogeneous and shifted towards smaller size species and generalists with broader ecological niches (Blois *et al.*, 2013; Lurgi *et al.*, 2012). Novel interactions can strongly affect species fitness because species will lack a long coevolutionary history in new conditions (Gilman *et al.*, 2010; see also Appendix 4.2).

4.2.1.3 The importance of feedbacks between hierarchical levels of biodiversity

Some well described feedbacks between different hierarchical levels and facets of biodiversity are self-reinforcing and could likely amplify negative effects of global changes on biodiversity (Brook *et al.*, 2008). Integration of processes acting at different organizational biodiversity levels is essential for future predictions of global change impacts on nature (Mouquet *et al.*, 2015; Thuiller *et al.*, 2013).

The feedback between population size and genetic diversity (S4 in **Box 4.2.1**) is known as an extinction vortex (Frankham *et al.*, 2014) because the reduction in population size leads to the loss of genetic diversity which in turn, leads to decrease in population fitness and adaptability and further reduction in population size. The *feedback between species' range and genetic diversity* (S5 in **Box 4.2.1**) means that the contraction and fragmentation of species ranges are expected to cause genetic loss through decrease in effective population size and extinction of genetic lineages as well as extinction of local populations with unique genetic characteristics (Bálint *et al.*, 2011; Pauls *et al.*, 2013). Genetic loss, in turn, may decrease species adaptability and migration capacity. The *feedback between species composition and genetic diversity* (SD3 in **Box 4.2.1**) means that changes in species composition alter the selection pressure affecting genetic diversity. For example, reduction in pollinator abundance could lead to selection favoring self-fertilization in plant populations, leading to a decrease in genetic diversity (Neaves *et al.*, 2015). Introductions of alien species may result in hybridization, out-breeding depression and decrease in genetic diversity of native species. However, hybridization may also facilitate adaptation to novel environments (Hoffmann & Sgrò, 2011). Changes in genetic diversity, in turn, contribute to further disturbance of species relationships.

The *feedback between species composition and single species extinctions* (SD4 in **Box 4.2.1**) make changes in species composition and single-species extinctions modify the web of interactions at the community level and lead to cascading and catastrophic co-extinctions called "chains of extinction" (Bellard *et al.*, 2012; Brook *et al.*, 2008). The loss of key species as well as invasions and proliferation of pests and pathogens can have the most drastic effects. Failing to account for changes in biotic interactions could cause models to under- or overestimate extinction risks (Gilman *et al.*, 2010). The *feedback between species composition and species' capacity to track climate change* (SD5 in **Box 4.2.1**) implies that interspecific interactions can modulate the outcome of species range shifts. Mutualistic interactions, such as plant-pollinator relations, may fail in tracking fast environmental change (Lavergne *et al.*, 2010). Competition and predation can both hamper and facilitate range shifting (Holt & Barfield, 2009; Svenning *et al.*, 2014). Interactions can slow climate tracking and produce more extinctions than predicted by models assuming no interactions (Urban *et al.*, 2013). Moreover, interspecific interactions can modulate the direction of species range shifts, for example, species may shift downslope due to competitive release at the lower margin of species distribution (Lenoir *et al.*, 2010). Changes in species distribution, in turn, contribute to further changes of species composition. The *feedback between landscape homogenization and species extinctions* (ED2 in **Box 4.2.1**) involves that predicted biotic homogenization and loss of biological uniqueness of communities within a region (Blois *et al.*, 2013; Buisson *et al.*, 2013; Thuiller *et al.*, 2014) can synchronize local biological responses to disturbance across individual communities and thus, compromise the potential for landscape- and regional-level disturbance buffering (Olden, 2006). Taxonomic homogenization of communities can reduce resistance of a landscape to future invasions (Olden, 2006). As a result, local extinctions of native species and invasions of alien species should be expected that, in turn, will contribute to further biotic homogenization (for details, see Appendix 4.2).

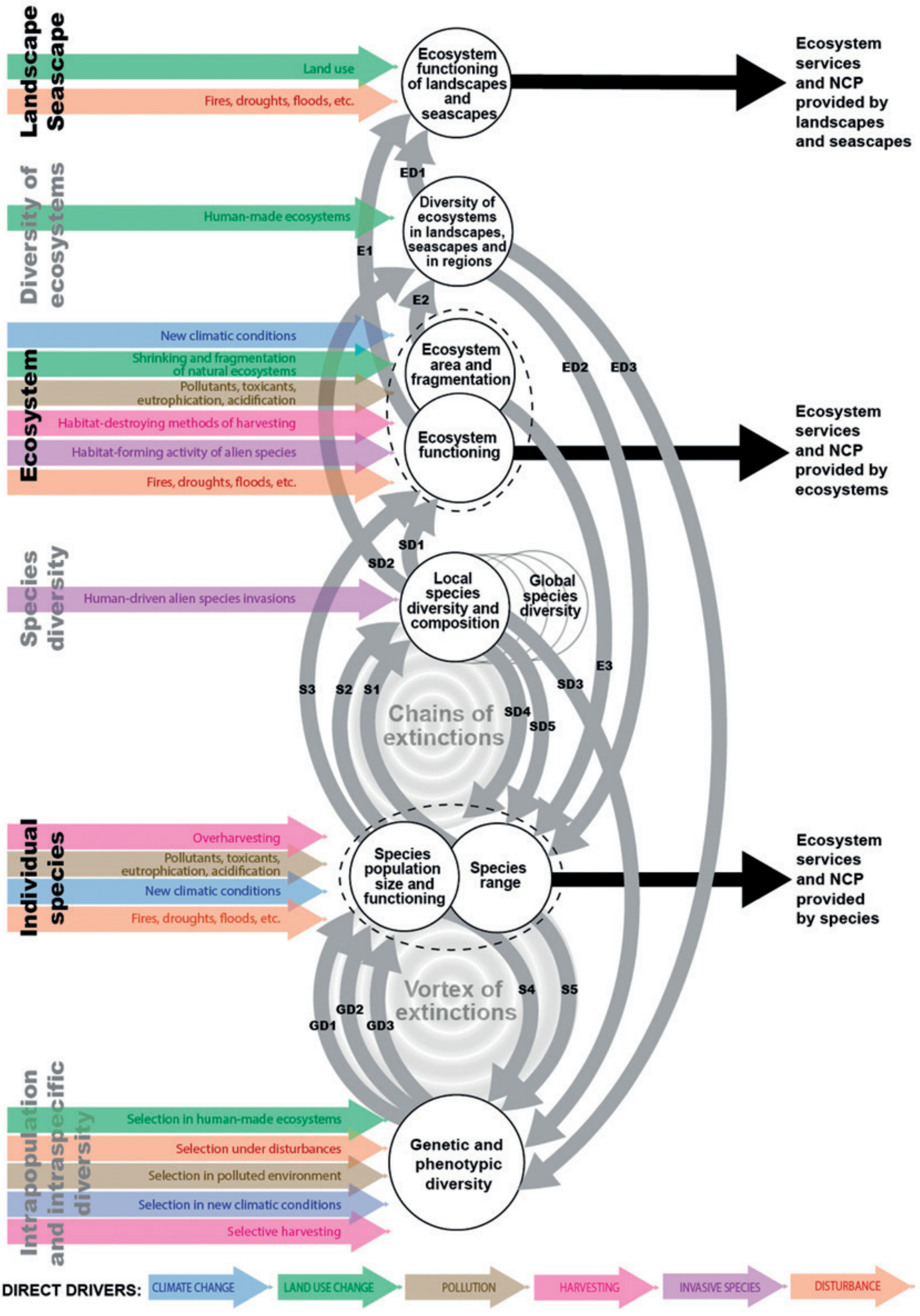
4.2.2 Marine ecosystems

4.2.2.1 Global state and function of marine ecosystems and future drivers of change

The ocean is central to regulating the Earth's climate. The ocean absorbs around 25% of the anthropogenic emissions of CO_2 (Le Quéré *et al.*, 2016), leading to ocean acidification with a decrease in surface seawater pH of 0.1 units since the beginning of the industrial era (Orr *et al.*, 2005). The ocean absorbs 93% of the Earth's excess heat energy, resulting in warming of 0.11°C per decade in the upper 75m of the ocean between 1971 and 2010 (Rhein *et al.*, 2013). ***Oceans are essential to life and provide major services to human societies.*** Marine phytoplankton produce about half of the global O2 (Pörtner *et al.*, 2014). The ocean supports fisheries and aquaculture activities and produced on average 104.3 million tons

Box ④ ② ① The main interrelations and feedbacks between hierarchical levels that are important for the future of biodiversity.

Direct drivers of global change affect all levels of biodiversity, either directly (coloured arrows) or indirectly through feedbacks (grey arrows). Even one-way interactions are important for biodiversity response, while self-reinforcing feedbacks can potentially significantly increase expected negative effects of global change drivers (for details, see Appendix 4.2).



Effects of changes in genetic and phenotypic diversity
GD1 – adaptation of populations to new conditions through standing genetic and phenotypic variations
GD2 – adaptation of populations due to phenotypic plasticity
GD3 – adaptive evolution, “evolutionary rescue” of populations and species

Effects of changes in functioning, population size and range of individual species
S1 – changes in local species composition due to alteration of species range (shift, change in area, fragmentation)
S2 – changes in local species composition due to local species extinctions and alteration of species abundance and functioning (including changes in phenology)
S3 – changes in ecosystem structure and functioning due to changes in key species abundance and functioning
S4 – changes in genetic diversity due to changes in population size
S5 – changes in genetic diversity due of alteration in species range (shift, change in area, fragmentation) and dispersal ability

Effects of changes in local species diversity, species composition and interspecific relations
SD1 – weakening and destabilization of ecosystem functioning due to loss of local species diversity
SD2 – biotic homogenization as a result of species shift, local species extinctions and invasions
SD3 – changes in selection pressure because of alteration of species composition and interspecific relations (including effects of alien species invasions)
SD4 – species extinctions as a result of cascading effects of alteration of species composition
SD5 – impact of alteration of species composition on species capacity to track climate change

Effects of changes in structure and functioning of ecosystems
E1 – the contribution of individual ecosystems to the total landscape/seascape ecosystem functioning
E2 – disappearance of the most vulnerable ecosystems in landscapes/seascapes and regions
E3 – reduction of species population size, reduction and fragmentation of species’ ranges and disruption of population structure because of habitat loss and fragmentation

Effects of changes in diversity of ecosystems, heterogeneity of landscapes and seascapes
ED1 – weakening and destabilization of the total landscape/seascape functioning because of loss of ecosystem/habitat diversity
ED2 – influence of landscape heterogeneity on local species persistence
ED3 – influence of landscape heterogeneity on genetic diversity and evolution

per year of fish and invertebrates from 2009-2014, which represented approximately 17% of the animal protein consumed by humans (FAO, 2016). Oceans supports rapid socioeconomic development and growth of human population on coastlines, with increasingly intensive, multiple uses leading to heavily degraded habitats (Spalding *et al.*, 2014; Wong *et al.*, 2014). ***Marine populations and communities have been impacted at unprecedented rates*** by climate change (mainly in the form of ocean warming, ocean acidification, deoxygenation, and sea level rise) and direct anthropogenic activities (mainly in the form of fishing, pollution, and habitat degradation) (Chapter 2; Hoegh-Guldberg *et al.*, 2014; Poloczanska *et al.*, 2016; Pörtner *et al.*, 2014).

Globally, none of these pressures are projected to decrease in the future. Earth System Models have been used to project future environmental conditions (IPCC, 2013), showing that the state of the future ocean will strongly depend on the amount of carbon emitted in the coming decades (Gattuso *et al.*, 2015; IPCC, 2018). Climate change is, among other drivers, the main driver considered in global scale scenarios **(Table 4.2.2)**.

Mean sea surface temperature is projected to increase by +2.7°C in 2090-2099 as compared to 1990-1999 for the high emission scenario (RCP8.5), whereas the warming is limited to +0.71°C for the more stringent RCP2.6 emission scenario (Bopp *et al.*, 2013); model-mean values from the Coupled Model Intercomparison Project 5). At the regional scale, stronger warming occurs in the tropics, in the North Pacific and in the Arctic Ocean, with the sea surface warming more than +4°C at the end of the 21st century under RCP8.5 (Bopp *et al.*, 2013; Collins *et al.*, 2013).

As global temperatures rise, so does the **mean sea level** due primarily to the thermal expansion of ocean water and by melting of glaciers, ice caps and ice sheets. A sea level model calibrated with empirical data and forced by the IPCC high emission scenario (RCP8.5) projects a sea level rise (SLR) of 52-131 cm by 2100 relative to year 2000 (Kopp *et al.*, 2016).

A broadly uniform decrease of the **mean sea surface pH** of -0.33 pH units (model-mean) by the 2090s relative to the 1990s is predicted under RCP8.5 (Bopp *et al.*, 2013), which is accompanied by a decrease in carbonate ion concentration and in the saturation states of calcium carbonates (e.g., calcite, aragonite), essential components of shells or skeletons of many marine organisms. The volume of undersaturated waters with respect to aragonite is projected to increase between 1990 and 2100 from 76% to 91% of the global ocean under RCP8.5 (Gattuso *et al.*, 2015).

Earth system models also project **decreasing global ocean oxygen** due to climate change. The mechanisms at play are a reduction of oxygen solubility due to ocean warming and the combination of increased stratification and reduced ventilation that prevents the penetration of oxygen into the deep ocean (Breitburg *et al.*, 2018). Deoxygenation will continue over the 21st century irrespective of the future scenario, with decreases of global O_2 of -1.8% and -3.45% (model-mean) under RCP2.6 and RCP8.5, respectively (Ciais *et al.*, 2013), with a stronger drop for the North Pacific, the North Atlantic, and the Southern Ocean (Bopp *et al.*, 2013). Despite a consistent global deoxygenation trend across models, there is as yet no consensus on the evolution of hypoxic and suboxic

Table ④ ❷ ② **Major climate-related and direct human-mediated drivers of change impacting marine ecosystems (by IPBES subunits) as highlight in this chapter's sections 4.2.2.2 to 4.2.2.5.**

Cells are colored when there is substantial evidence from the reviewed scenarios and models that drivers have a major impact on one of the marine ecosystems. Where the information exists, the second column of the table reports the percentage of marine global scale scenarios implementing changes in the drivers and quantifying impacts on nature, based on our literature database (Appendix A4.1.1).

Direct drivers of change	**Global scale**	**Open ocean pelagic**	**Polar seas**	**Shelf ecosystems**					**Deep sea ecosystems**
				Tropical coral reefs	Rocky and sandy shores	Mangrove forests	Seagrass meadows	Kelp forests	
Climate-related drivers of change									
Ocean warming	45%	■	■	■	■	■	■	■	■
Ocean acidification	8%	■	■	■					■
Deoxygenation	4%	■		■			■		■
Sea ice melt	2%		■						
Sea level rise (SLR)	16%		■		■	■	■		
Extreme events	3%			■	■	■	■	■	
Direct human-mediated drivers of change									
Fishing	16%	■	■	■	■	■	■	■	■
Pollution	5%	■	■	■	■	■	■	■	■
Maritime transport		■	■						
Species introduction			■	■	■	■	■	■	
Land-use change	1%			■	■	■	■	■	
Coastal development	1%		■	■	■	■	■	■	
Aquaculture				■	■	■	■	■	
Oil and gas extraction, mineral mining		■	■	■	■				■
Main direct impacts on nature									
Habitat degradation			■	■	■	■	■	■	■
Biodiversity decline		■	■	■	■	■	■	■	■
Species invasion / range shift		■	■	■	■	■	■	■	
Shifts in food webs and biogeochemical cycles		■	■	■	■	■	■	■	■
Eutrophication				■	■		■		
Hypoxia		■		■	■		■		■

waters due to uncertainties in potential biogeochemical effects and in the evolution of tropical ocean dynamics (Cabré *et al.*, 2015). Along coastlines, deoxygenation and the increase of hypoxic "dead zones" are largely driven by direct human activities (which combine with sea warming), with rivers draining large nitrogen and phosphorus loads from fertilized agricultural watersheds, and from sewage, aquaculture and atmospheric nitrogen deposition, causing eutrophication and subsequent aerobic microbial decomposition (Glibert *et al.*, 2018; Levin *et al.*, 2009; Rabalais *et al.*, 2009).

Future climate change will hence alter marine habitats and modify biogeochemical cycles. Recent modelling work has shown that climate change may continue to produce more hostile conditions and threaten vulnerable ecosystems and species with low adaptive capacity (Gattuso *et al.*, 2015; Hoegh-Guldberg *et al.*, 2014; Mora *et al.*, 2013a; Pörtner *et al.*, 2014; Wong *et al.*, 2014).

Adding to future climate change and potentially amplifying impacts on marine ecosystems, direct human-mediated pressures will likely intensify in future. An **increase in fisheries and aquaculture production** is plausible as a response to increasing demand for fish and seafood (Chapter 11 of the World Ocean Assessment, UN, 2017) which is expected to arise as a result of population growth and increasing average income that allows for augmenting the proportion of fish in the diet (World Bank, 2013). Under assumptions of increasing technological efficiencies and increasing demand for fish, the FAO and OECD project that total world marine seafood production (fishery plus aquaculture) would exceed 120 million tons in 2025, or plus 17% relative to 2013-2015. Diverse forms of **pollution** (excessive nutrient loads, toxic contaminants, persistent organic pollutants, plastics, solid waste) will likely continue to pervade marine ecosystems in the future, constituting additional threats to living organisms (Bergman *et al.*, 2012; Geyer *et al.*, 2017; Lamb *et al.*, 2018; Sutton *et al.*, 2013; Worm *et al.*, 2017). The oceans are sinks for landborne and airborne inputs of persistent pollutants which can both travel great distances in the near-surface water masses (Eriksen *et al.*, 2014) of the open ocean, and sink into the deeper ocean (Chapter 20 of the World Ocean Assessment, UN, 2017). In coastal oceanic waters, increasing nutrient loads and pollution in combination with warming will likely stimulate eutrophication and increase the extent of oxygen minimum zones (Breitburg *et al.*, 2018; Rabalais *et al.*, 2009).

The impacts of global change on marine biodiversity will vary geographically, with latitudinal gradients of expected in many global scale scenarios (Gattuso *et al.*, 2015), and depending on the type of ecosystems **(Table 4.2.2)**. Major drivers of change in the open ocean pelagic ecosystems that are included in global scale models and scenarios are climate-related drivers (sea warming, acidification, deoxygenation), and fisheries exploitation. Additional future threats included in scenarios for shelf ecosystems are sea level rise, extreme events, nutrient pollution and coastal development which may cause degradation, fragmentation and loss of habitats **(Table 4.2.2)**.

Future scenarios of climate change impacts on marine biodiversity at global scales are the most documented in the literature (78% of the scenarios in our literature database – **Table 4.2.2**). They will therefore form the main content of this section (section 4.2.2.2), with evidence provided by type of ecosystems (IPBES units of analysis). The rest of the drivers are much less, or not at all, represented in scenarios projecting impacts on marine biodiversity at global scale, even though their historical and current impacts on biodiversity have been shown to be significant. Moreover, there are relatively few global scale scenarios involving multiple pressures on marine ecosystems and biodiversity (23% of the marine scenarios involve a combination of multiple drivers in our global scale literature database), so in addition to updating recent global assessments with the latest modelling and scenarios work, sections 4.2.2.2 to 4.2.2.5 report evidence from more local studies of how direct anthropogenic drivers may combine with climate change in impacting future marine biodiversity.

4.2.2.2 Future climate change impacts on marine biodiversity and ecosystem functioning

4.2.2.2.1 Climate change impacts in open ocean ecosystems

Low trophic levels

Net Primary Production (NPP) by marine phytoplankton is responsible for 50% of global carbon fixation through photosynthesis, but is also the basis of marine food webs, controlling the energy and food available to upper trophic levels. Earth System Models project a mean decrease of NPP in 2100 under all RCP greenhouse gas emissions scenarios, ranging from -3.5% to -9% under RCP2.6 (low emissions) and RCP8.5 (very high emissions), respectively (Bopp *et al.*, 2013), though there is significant variation between individual model projections. The global decrease of NPP is accompanied by a change in the seasonal timing of peak NPP, with an advance by ~0.5–1 months by 2100 globally, particularly pronounced in the Arctic (Henson *et al.*, 2013).

The projections are heterogeneous over space with general agreement that NPP is expected to decrease in the tropics and in the North Atlantic, and increase at high latitudes (Bopp *et al.*, 2013; Boyd *et al.*, 2014; Steinacher *et al.*, 2010). Some regional discrepancies between models exist, with nonlinear dynamics making some projections uncertain. In the tropics, the mechanisms at play are largely model-dependent, with both stratification–driven reduction in nutrient availability and increases in grazing and other phytoplankton loss processes (Laufkötter *et al.*, 2015). This results in large inter-model differences, with the decline in tropical NPP being projected between -1 and -30% by 2100 under RCP8.5 (Kwiatkowski *et al.*, 2017). Using satellite-based observations of ocean–colour and an emergent-constraint relationship, the uncertainties in the decline of tropical NPP have been reduced with an estimated decline of -11±6% in 2100 for a business-as-usual scenario (Kwiatkowski *et al.*, 2017).

In the Arctic, some models project an increase in NPP because of the loss of perennial sea-ice and an increase of light availability, whereas other models simulate a decrease due to increasing ocean stratification and decreasing nitrate availability (Vancoppenolle *et al.*, 2013). In the Southern Ocean, models project a zonally-varying response of NPP to climate change, with a decrease in the subpolar band (50°S and 65°S), but increases in the Antarctic (south of 65°S) and in the transitional band (40°S-50°S) (Leung *et al.*, 2015). Mechanisms at play are changing light availability and iron supply by sea ice melting (Wang *et al.*, 2014).

Under the SRES A1B scenario, the reduction in zooplankton biomass was projected to be higher than for primary production in 47% of the ocean surface particularly in the tropical oceans, implying negative amplification of ocean warming through bottom-up control of the food web (Chust *et al.*, 2014). This impact differs regionally with positive amplification of zooplankton biomass in response to the increase of NPP in the Arctic and Antarctic oceans, thereby increasing the efficiency of the biological pump in those regions. Other changes in species composition can be expected under future climate change, such as shifts from diatom-dominated phytoplankton assemblages with high POC export efficiencies to smaller, picoplankton communities characterized by low export efficiencies (Morán *et al.*, 2015; Smith *et al.*, 2008).

In addition to warming and changes in ocean stratification/circulation, ocean acidification is also expected to influence metabolic processes in phytoplankton and zooplankton species. Laboratory and mesocosm experiments have shown contrasting responses for different plankton types under elevated CO_2 concentrations, with a stimulating influence for nitrogen-fixing cyanobacteria (Hutchins *et al.*, 2007, 2013) and pico-eukaryotes (Bach *et al.*, 2017), but potential detrimental effects on growth and calcification rates for some of the main calcifying phytoplankton (Meyer & Riebesell, 2015). Other potential effects of ocean acidification include a reduction in microbial conversion of ammonium into nitrate (Beman *et al.*, 2011), which could have major consequences for oceanic primary production and potentially less carbon export to the deep sea. A recent modeling study incorporating differing growth responses of phytoplankton types to increased $p\mathrm{CO}_2$, has suggested that acidification effects may even outrank the effects of warming and of reduced nutrient supply on phytoplankton communities over the 21st century (Dutkiewicz *et al.*, 2015).

Higher trophic levels

Most published global scale scenarios of change in higher trophic levels in response to climate change rely on correlative models examining changes in species' spatial distribution (64% of publications on the effect of climate change on marine biodiversity at global scale in our literature database*,* Appendix A4.1.1). These "Species Distribution Models" (SDMs) (also called ecological niche models or climate envelope models) analyze the statistical relationship between species occurrences and a set of environmental variables (Araújo & New, 2007; Thuiller *et al.*, 2009). SDMs do not typically consider species adaptation nor the effects of species interactions.

Using species distribution models for projecting future climate-induced changes, the main findings at the global scale are that species will shift their distribution poleward (Cheung *et al.*, 2009), likely resulting in an increase in species richness and species invasions in high latitude regions (the Arctic and Southern Ocean) and conversely a decrease of species richness in the tropics and the equator (García Molinos *et al.*, 2016; Jones & Cheung, 2015; Pörtner *et al.*, 2014) and in semi-enclosed seas (e.g., Mediterranean Sea, Ben Rais Lasram *et al.*, 2010). A mean latitudinal range shift of 25.6 km per decade to 2050 was projected under the high emission scenario RCP8.5, which reduced to 15.5 km per decade under RCP2.6 (Jones & Cheung, 2015).

Distributional shifts of marine species are the most clearly detectable pattern that can currently be assigned to climate change, or more specifically to sea surface temperature change (García Molinos *et al.*, 2016). This is related to the sensitivity of marine ectotherms, which constitute the bulk of high trophic level species, to temperature change. But ocean warming can trigger additional adaptive responses such as phenological shifts and physiological changes in growth and reproduction. It is expected that animals inhabiting temperate latitudes, where seasonality is strong, will better adapt to a changing climate whereas polar stenotherm species will be more vulnerable to warming (Pörtner *et al.*, 2014). Tropical species, in addition to having narrow thermal windows, inhabit the warmest waters and are thus near physiological temperature tolerance limits that lower their adaptive capacity (Storch *et al.*, 2014)
At low latitudes, open-ocean oxygen-minimum zones (OMZ) constitute an additional threat to marine organisms, especially in the eastern tropical Pacific (Cabré *et al.*, 2015)

and along major eastern boundary upwelling systems (Gilly *et al.*, 2013). The horizontal and vertical expansion of already large OMZs will potentially affect marine populations dramatically, through shifts in their spatial distribution and abundance, as well as altered microbial processes and predator-prey interactions (Breitburg *et al.*, 2018; Gilly *et al.*, 2013). The shoaling of the upper boundary of the OMZs can also trap fish in shallower waters, compressing their habitat, and thereby increasing their vulnerability to predation and fishing (Bertrand *et al.*, 2011; Breitburg *et al.*, 2018).

In addition to correlative species distribution models, there are recently developed integrated modelling approaches (e.g., end-to-end models combining the physics of the ocean to organisms ranging from primary producers to top predators) considering the multiple responses of marine populations to climate change (based on e.g., physiological rates, trophic interactions, migration behavior), as well as essential food web knock-on effects and adaptive mechanisms to move towards more realistic projections of marine biodiversity (Payne *et al.*, 2016; Rose *et al.*, 2010; Stock *et al.*, 2011; Tittensor *et al.*, 2018a; Travers *et al.*, 2007). At regional and local scales, such models have been developed with more detailed representation of multiple taxa of commercial interest or of conservation concern than at the global scale, where the few existing end-to-end models represent ecosystems and biodiversity through large functional groups (e.g. fish biomass, pelagic biomass, biomass in different size classes) or are focused on single key species. A global scale end-to-end model run under the worst-case scenario (RCP8.5) projected that the biomass of high trophic level organisms would decrease by 25% by the end of the century (Lefort *et al.*, 2015). This first estimate, which has been recently confirmed by an ensemble of global marine ecosystem models **(Box 4.2.2)**, suggests that the response of high trophic levels amplifies the decrease of biomass projected for phytoplankton and zooplankton.

Global scale models project that ocean warming may shrink the mean size of fish by the end of century (Cheung *et al.*, 2013; Lefort *et al.*, 2015) and lead to smaller-sized infaunal benthos globally (Jones *et al.*, 2014). This trend is very robust to the model used in the different studies, as well as to the mechanisms involved: the decrease in mean size could be either due to the combined effects of future warming and deoxygenation on animal growth rates (Cheung *et al.*, 2013), the combined effects of warming and food limitation (Lefort *et al.*, 2015), or to the limiting flux of particulate organic matter from the upper ocean to the benthos (Jones *et al.*, 2014).

Air-breathing marine species

Marine turtles are particularly vulnerable to climate change as, being ectotherms, their behavior, physiology, and life traits are strongly influenced by environmental factors (Janzen, 1994; Standora & Spotila, 1985). Arguably, the most detectable impacts will occur during the terrestrial reproductive phase: incubating eggs are vulnerable to sea-level and extreme weather events (Fish *et al.*, 2005; Fuentes *et al.*, 2010), while future changes in temperature and rainfall at nesting beaches will likely reduce hatching success and emergence, cause a feminization of turtle populations, and produce hatchlings with higher rates of abnormalities (Fisher *et al.*, 2014; Mrosovsky & Yntema, 1980). Future changes in temperature are expected to impact the frequency and timing of nesting (Fuentes & Saba, 2016; Limpus & Nicholls, 1988; Saba *et al.*, 2007), as well as marine turtle distribution (McMahon & Hays, 2006; Pikesley *et al.*, 2015; Witt *et al.*, 2010). Foraging specialists (i.e. leatherbacks) might be more susceptible to climate change impacts on the marine food web relative to foraging generalists (i.e. loggerheads) due to a lesser ability to switch prey type (Fuentes & Saba, 2016). Ultimately, impacts will depend on populations' resilience and ability to adapt. Some marine turtle populations are already responding to climate change by redistributing their nesting grounds and shifting their nesting phenology (Pikesley *et al.*, 2015). However, it is still unclear whether marine turtles will be able to fully adapt since climatic changes are occurring more rapidly than in the past and are accompanied by a variety of anthropogenic threats (e.g., fisheries by-catch, pollution) that make them more vulnerable and decrease their resilience (Fuentes *et al.*, 2013; Poloczanska *et al.*, 2009).

Seabirds responses to future climate change are commonly predicted using species distribution models. Shifts and contractions in foraging habitat could be particularly problematic for seabirds by increasing energetic expenditures. For example, the summer foraging areas for king penguins are predicted to shift southward in response to an intermediate warming scenario (SRES A1B), doubling the travel distance to optimal foraging areas for breeders with likely negative consequences for population performance (Peron *et al.*, 2012). Poleward shifts in foraging areas are also projected for seven Southern Ocean albatross and petrel species under a range of emission scenarios, with associated range contractions of up to 70% for wandering and grey-headed albatross by 2050 (Krüger *et al.*, 2018). For other species (e.g., the endangered Barau's storm petrel), climate-driven shifts and contractions in wintering range are predicted but the overall population consequences are unclear (Legrand *et al.*, 2016). Fewer studies have coupled mechanistic population models with climate projections to estimate future population trajectories. Cassin's auklets are predicted to decline by 11-45% by 2100 under a mid-level emission scenario, due to increased sea surface temperatures and changes in upwelling dynamics within their foraging range (Wolf *et al.*, 2010). Contrasting responses to future climate scenarios were reported in three seabirds (albatrosses and petrel),

Box ④ ② ② Ensemble model projections of marine ecosystem futures under climate change.

Model intercomparison studies use a common set of input conditions to force a suite of potentially very different models to then produce an 'ensemble' of outputs. These outputs can be compared to examine differences among models, and provide a multi-model mean and range of uncertainty for end users. While such studies are a common tool in the Earth system and climate modelling communities, their application to biodiversity and ecosystems, particularly in the marine realm, remains relatively new.

Fish-MIP (Tittensor *et al.*, 2018b) is the first model intercomparison project examining the impacts of climate change on fisheries and marine ecosystems at regional to global scales using a common set of climate change scenarios. There have been many different attempts to model the ocean ecosystem resulting in a large diversity of models with various purposes – from examining species distributions to ecosystem structure to fisheries catch potential (Tittensor *et al.*, 2018b). Fish-MIP provides a common simulation framework and standardized forcing variables to provide consistent inputs to these models and prescribes a common set of consistent outputs for analysis. In the first round of Fish-MIP, the focus was on examining climate change (rather than fisheries) both regional and global scales. Here, marine animal biomass includes mostly fish, but in some models, invertebrates and marine mammals are also considered.

The results across six global marine ecosystem models (APECOSM, BOATS, DBEM, DPBM, EcoOcean, Macroecological) that were forced with two different Earth-system models (ESMs) and two emission scenarios (RCPs 2.6 and 8.5) show that ocean animal biomass will likely to decline over the coming century under all climate change scenarios (Figure 4.2.4; Lotze *et al.*, 2018; Tittensor *et al.*, 2018b). The ensemble model means show steeper declines under RCP8.5 (highest emission scenario) than RCP2.6 (high mitigation scenario), and steeper declines when forced with the ESM IPSL-CM5A-LR than GFDL-ESM2M. The trajectories from different ESMs and RCPs remain relatively similar until about 2030 to 2050, after which they begin to diverge markedly. Thus, by 2100, the model-mean animal biomass is projected to decline between 3% and 23% **(Figure 4.2.4)**. These declines are largely driven by a combination of increasing water temperature and declining primary productivity, and are likely to impact ecosystem services including fisheries (Blanchard *et al.*, 2017).

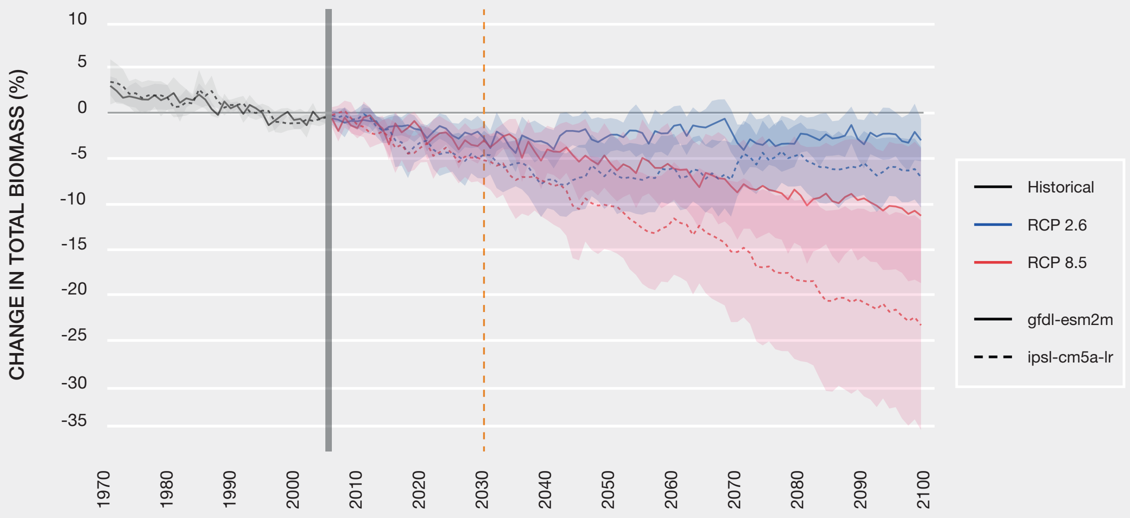


Figure ④ ② ④ **Ensemble projections of global ocean animal biomass under different scenarios of climate change.**

Projections represent the multi-model means of six global marine ecosystem models forced by marine environment change projected by two different Earth-system models: GFDL-ESM2M (solid lines) and IPSL-CM5A-LR (dashed lines) and two greenhouse gas emission scenarios: RCP2.6 (low emissions; blue) and RCP8.5 (very high emission; red) with no fishing signal imposed (i.e., changes are due only to climate). Shaded areas represent one inter-model standard deviation (ecosystem models). All percentage changes are relative to a 1990-1999 baseline. The vertical grey line separates historical and future projections for climate forcing; the vertical dashed orange line represents the 2030 target year for the Sustainable Development Goals. Data source: Tittensor *et al.* (2018b); Lotze *et al.* (2018).

Spatial maps of ensemble projections (**Figure 4.2.5**; Lotze *et al.*, 2018; Tittensor *et al.*, 2018b) show broad-scale decreases in animal biomass in tropical and many temperate regions, and potential increases in polar regions. While ensemble projections across many models are more likely to capture plausible trends than any single model, there was more variation among models in polar and some coastal regions, suggesting that there is greater uncertainty about projected outcomes.

The results shown here for global marine ecosystem models are helpful for describing the global trends but may not capture the complex dynamics at local and regional scales. Forthcoming analyses should therefore compare regional projections based on regional scale models and global models and examine the variability between regional models to provide projections and measures of uncertainty at scales better matched to the needs of resource managers. Moreover, different scenarios of fishing pressure need to be incorporated to examine interactions between fishing and climate change impacts.

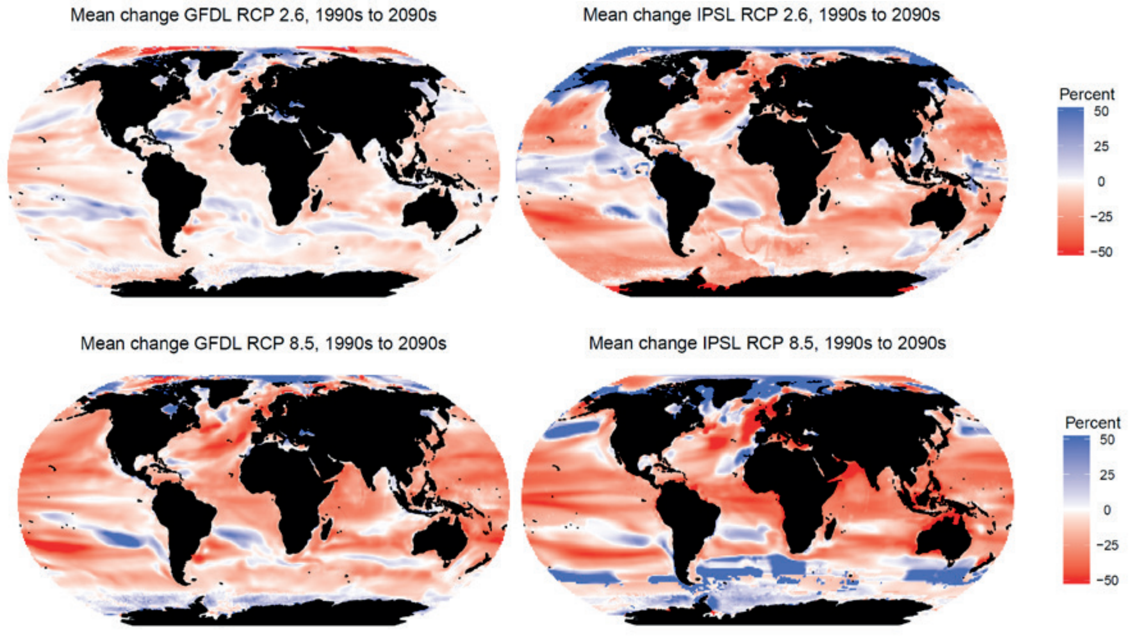


Figure ④ ② ⑤ **Global ensemble mean spatial patterns of change in global ocean animal biomass under RCP2.6 (low greenhouse gas emissions; top) and RCP8.5 (very high emissions; bottom) forced by GFDL-ESM2M (left) and IPSL-CM5A-LR (right) Earth System Models.**

Percentage changes are relative to a 1990-1999 baseline. Data source: Tittensor *et al.* (2018b); Lotze *et al.* (2018).

owing to differences in life histories and distribution area (Barbraud *et al.*, 2011). These studies have identified strong non-linearities in demographic responses, suggesting the potential for threshold effects under future climate extremes (Pardo *et al.*, 2017).

Marine mammals, as homeotherms, are physiologically buffered from some direct effects of temperature rise. Rising ocean levels from ocean warming and ice melt will likely lead to a loss of land or ice-based habitat available for breeding or pupping, particularly for marine mammals on low-lying atolls or ice-dependent breeders (Baker *et al.*, 2006; Laidre *et al.*, 2015). A global assessment of climate change effects on marine mammals used a range of climate scenarios (warming between 1.1°C and 6.4°C) to qualitatively rank negative population effects for all marine mammal species (MacLeod, 2009). It showed that species tied to land, ice, or facing geomorphic barriers were most likely to be affected.

4.2.2.2.2 Climate change impacts in shelf ecosystems

Tropical Coral Reefs

An unprecedented 3-year (2014-2017) marine heat wave have damaged most of coral reefs on Earth (75%) with still unassessed social-ecological consequences (Eakin *et al.*, 2018). Thermal stress disrupts the relationship between corals and their algal symbionts, with bleached corals being physiologically damaged and suffering severe mortality rate. The number of years between recurrent severe coral-bleaching events has diminished fivefold in the past four decades, from once every 25 to 30 years in the early 1980s to once every 5.9 years in 2016 (Hughes *et al.*, 2018). A full recovery of mature coral assemblages, source of reef biodiversity and productivity, generally takes from 10 to 15 years for the fastest growing species (Hughes *et al.*, 2018). Many reefs, including those of the iconic and well-protected Great Barrier Reef, have experienced a shift from dominance of branching tabular species that build 3-dimensional habitats, towards corals with simpler morphological characteristics (Hughes *et al.*, 2018). A trophic model showed that a loss of coral complexity could cause more than a 3-fold reduction in fishery productivity (Rogers *et al.*, 2014), due to the preferential settling of juvenile fishes in unbleached coral habitat (Scott & Dixson, 2016).

In addition to thermal stress, ocean acidification represents a major threat to marine calcifier organisms like corals, particularly those building large but low-density skeletons. A decrease of pH by 0.4 units (expected under RCP8.5; Hoegh-Guldberg *et al.*, 2014) would translate into a coral habitat complexity loss of 50%, inducing a decrease in species richness by 30% for both fish and invertebrates (Sunday *et al.*, 2017). A seawater pH lowered by just 0.14 units (RCP2.6) would induce a loss of 34% net community calcification (Albright *et al.*, 2018). Projections anticipate a shift from a state of net accretion to net dissolution before the end of the century (Eyre *et al.*, 2018). Anoxic events are also rapidly increasing in prevalence worldwide and cause underestimated mass mortality on coral reefs (Altieri *et al.*, 2017).

To better anticipate and simulate the potential futures of coral reef habitats, two complementary approaches have been used. First, laboratory and field experiments try to estimate the tolerance, acclimatization and adaptability of coral species and their symbionts to environmental changes. One of the most striking studies demonstrates that progressive acclimatization, even to temperatures up to 35°C, can achieve the same heat tolerance as expected from strong natural selection over many generations (Palumbi *et al.*, 2014). This suggests that at temperatures beyond the thermal limits of coral species, the rate and speed of temperature change is key to explain coral bleaching. Experiments also allow testing of the interactions of multiple stressors. For instance, a 3-year field experiment deciphered the mechanisms by which elevated temperatures exacerbate overfishing and nutrient pollution effects on corals by increasing coral–algal competition and reducing coral recruitment, growth and survivorship (Zaneveld *et al.*, 2016).

Second, models attempt to simulate the futures of tropical coral reefs under various scenarios. A simulation based on genomic models predicting future evolution and persistence in a high-latitude population of corals from Cook Islands (South Pacific) showed a rapid evolution of heat tolerance resulting in population persistence under mild warming scenarios (RCP2.6 and RCP4.5) though this adaptation would not be rapid enough to prevent extinction under more severe scenarios (RCP6.0 and RCP8.5; Bay *et al.*, 2017). Other studies based on niche models, that can also integrate adaptation capacity related coral cover to environmental variables allowing for projections at global (Logan *et al.*, 2014) and regional (Ainsworth *et al.*, 2016) scales. For instance, coral cover on the Great Barrier Reef was projected to remain lower than 5% before the end of the century under a high emission scenario (RCP8.5) (Ainsworth *et al.*, 2016).

Rocky and sandy shores

Straddling the intersection between land and ocean, rocky and sandy shores are the dominant components of coastlines globally, are the most accessible of the marine biomes and supply services in terms of coastal protection, direct provisioning (food and materials), recreation (tourism, fishing), spiritual and cultural purposes, and substrate for aquaculture and infrastructure.

These ecosystems are vulnerable to sea-level rise which adds to the height of sea-level extremes, such as during storm surges, and can exacerbate projected changes in wave impacts (Hemer *et al.*, 2013). Sea level rise can affect the dynamics of the morphology of beach systems, as well as increasing coastal inundation risk, leading to erosion in many cases, as well as increasing threats to nesting beaches for turtles and seabirds, dune vegetation and coastal infrastructure and assets (e.g., de Winter & Ruessink, 2017; Jevrejeva *et al.*, 2016; Pike *et al.*, 2015).

Evidence of species responses to warming oceans are recorded from sandy and rocky shores globally, showing that barnacles, mollusks, crabs and macroalgae have shifted their distributions in response to recent warming (e.g., Johnson *et al.*, 2013; Pitt *et al.*, 2010; Poloczanska *et al.*, 2013; Schoeman *et al.*, 2015; Wethey *et al.*, 2011). For example, the cold-water barnacle *Semibalanus balanoides* may disappear from south-western English shores by 2050 (Poloczanska *et al.*, 2008). The frequency of temperature extremes is projected to increase in the next few decades, particularly during summer in regions such as the Mediterranean (Kirtman *et al.*, 2013), with potential high ecosystem impact as large-scale mortalities of intertidal species have been recorded during extreme heat events (Garrabou *et al.*, 2009; Wernberg *et al.*, 2013). In south-east Australia, the temperature-driven range extension of the sea urchin *Centrostephanus rodgersii* has led to the loss and overgrazing of kelp beds and a reduction in associated biodiversity (Johnson *et al.*, 2011; Ling *et al.*, 2015).

Forests of kelp, large brown temperate-coast marine algae, are themselves directly impacted by climate change. Under RCP2.6 and RCP8.5 scenarios, models of kelps in the

North Atlantic incorporating changes in temperature, salinity, and sea ice cover predict northern movement and range contraction by 2090 (Assis *et al.*, 2017a, 2017b, 2016; Raybaud *et al.*, 2013). Under RCP8.5, areas such as the Gulf of Maine, Southern Europe, and the northwestern coast of Africa would be bereft of kelps (Assis *et al.*, 2017a), a trend which in some of these systems is already observed now (Filbee-Dexter *et al.*, 2016; Krumhansl *et al.*, 2016). The Arctic, conversely, is projected to gain kelps, which is consistent with observations of kelp increases in areas that are decreasing in sea-ice cover and hence increasing in light availability (Bartsch *et al.*, 2016). The area gained is not projected to counterbalance the area lost. Similarly, in Japan, models project its southernmost species, *Ecklonia cava*, to colonize new northern habitats that are currently occupied by colder water kelps, due to a combination of shifting temperatures and increases in grazing by warm water fishes under all RCP scenarios by 2090. Further scenario-based modeling efforts are needed for Australia, New Zealand, the Southern Atlantic, and the Pacific Coasts of the Americas, where models of climate change's future impacts on kelps have been less explored. While modeled predictions typically report declines or polar movement, the observed long-term trajectories of kelp forests are currently mixed (Krumhansl *et al.*, 2016). In some cases, such as South Africa, this is due to local cooling (Blamey *et al.*, 2015; Bolton *et al.*, 2012). In others, climate driven range expansions of urchin predators has also driven local increases (Fagerli *et al.*, 2014), although the longevity of this trend is unclear as they can be overridden by physical drivers (Moy & Christie, 2012).

Coastal wetlands

Coastal wetlands are found along coastlines globally, and include salt marshes (mostly found along temperate, boreal and arctic coastlines), mangroves (mostly found in tropical and subtropical areas), tidal flats, and seagrasses. They form essential marine vegetated habitats for carbon sequestration, and coastal protection against increased sea level rise (SLR) and natural hazards (Alongi, 2008; Duarte *et al.*, 2013; Fourqurean *et al.*, 2012). They also host a great diversity of species, playing a major role as nursery and breeding areas for a wide variety of marine fauna organisms (Heck Hay *et al.*, 2003), including migratory ones such as coastal birds (Nuse *et al.*, 2015) or coral reef fish species (Harborne *et al.*, 2016). Climate changes in the form of warming, sea level rise and increased extreme events (e.g. hurricanes) may increase the vulnerability of these ecosystems in the future. Vegetated coastal habitats are already declining globally (Duarte *et al.*, 2005), and many species are threatened with extinction (Polidoro *et al.*, 2010; Short *et al.*, 2011). The recent IPCC report on « Global warming of 1.5°C » (IPCC, 2018) assessed that at global warming limited to 1.8°C above the pre-industrial level, the risks to mangroves will remain medium (e.g., not keeping pace with SLR; more frequent heat stress mortality) whereas seagrasses are projected to reach moderate to high levels of risk (e.g., mass mortality from extreme temperatures, storm damage) (Hoegh-Guldberg *et al.*, 2018).

Sea level rise can have large impacts on coastal ecosystems because of the flat, gentle slope of much coastal land. Although coastal wetlands are dynamic ecosystems that can adapt to sea level rise, their capacity to do so is limited, regionally differentiated and is affected by many human activities (Kirwan & Megonigal, 2013; Schuerch *et al.*, 2018; see 4.2.2.5). The response of wetlands to sea level rise involves landward migration of vegetated areas, and submergence at lower elevations (Wong *et al.*, 2014). Acceleration of sea level rise threatens future wetlands capacity to adapt with occurrence of horizontal retreat, and vertical drowning, when accretion of sediment and organic matter cannot keep pace with SLR (Spencer *et al.*, 2016). A meta-analysis estimated that under RCP2.6, 60% of the saltmarshes will be gaining elevation at a rate insufficient to keep pace with SLR by 2100, and the loss could reach 90% under high SLR (RCP8.5) (Crosby *et al.*, 2016). Such high SLR (1m by 2100) could put at risk 68% of coastal wetlands in developing countries (Blankespoor *et al.*, 2014). By contrast, a just published integrated model, taking into account the capacity of wetlands to both expand horizontally by inland migration and build up vertically by sediment accretion, projected less pessimistic impacts of SLR with the loss of global coastal wetlands area ranging between 0 and 30% by 2100, depending on the RCP considered (Schuerch *et al.*, 2018). Sea level rise and storm surges cause salinity intrusion inland, that can impact coastal and freshwater wetlands, with various effects such as decreased inorganic nitrogen removal, decreased carbon storage, and increased generation of toxic sulphides (Herbert *et al.*, 2015). Increased salt and sulphide concentrations induce physiological stress in biota and ultimately can result in large shifts in communities and associated ecosystem functions. Because impacts of sea level rise are so prominent in coastal wetlands (Jennerjahn *et al.*, 2017), the impacts of temperature rise have been relatively less explored despite their importance in terms of ecosystem structure and function (Gabler *et al.*, 2017).

Submerged plants such as seagrass are highly impacted by temperature extremes. Warming-induced deterioration of seagrass ecosystems has been observed over recent decades in the West Atlantic, Mediterranean, and Australia, with summer temperature spikes often leading to widespread seagrass mortality (Fraser *et al.*, 2014; Jordà *et al.*, 2012; Moore & Jarvis, 2008; Short & Neckles, 1999). In the western Mediterranean Sea, a model relating mortality rates to maximum sea temperature projected that seagrass meadows may become functionally extinct by 2050–2060, under the SRES A1B emission scenario (Jordà *et al.*, 2012). Climate warming is also affecting other components of seagrass ecosystems, notably via 'tropicalization'—increasing representation of tropical species—among seagrass-associated fish communities (Fodrie *et al.*, 2009), with the potential to reduce seagrass biomass and habitat complexity as tropical herbivorous fishes increase (Heck *et al.*, 2015). Among the most serious concerns is rising frequency of disease epidemics and prevalence of pathogens, which are associated with warming in many systems, and that could trigger widespread die-offs of seagrass (Altizer *et al.*, 2013; Harvell *et al.*, 2002; Kaldy, 2014; Sullivan *et al.*, 2013).

Under elevated mean global temperatures, mangroves are expected to displace salt marshes in many areas as the limits to mangrove growth imposed by cold events decrease (Short *et al.*, 2016). Mangroves in the southeastern US have been projected to expand in area (Osland *et al.*, 2013), consistent with observed trends across five continents over the past 50 years (Cavanaugh *et al.*, 2014; Saintilan *et al.*, 2014). These projections overlook important differences among mangrove species, and also depend on mangroves' ability to successfully migrate landward (Di Nitto *et al.*, 2014), and to build up sediment or continue to receive allochtonous sediment inputs from estuarine or freshwater sources at rates apace with SLR (Lovelock *et al.*, 2015; Parkinson *et al.*, 1994). In coastal settings experiencing erosion, an expansion of mangroves is highly unlikely. On the other hand, expansion is seen in areas of accelerating sediment deposition due to upstream land-use changes (Godoy & de Lacerda, 2015). Species distribution modeling studies have projected geographically dependent shifts in community composition and species richness under climate change scenarios (Record *et al.*, 2013). While species richness is projected to increase in SE Asia, South America, eastern Australia and parts of the African coasts, it will likely decline in Central America and the Caribbean, partly linked to increased intensity and frequency of tropical storms, as well as in northern Australia (Record *et al.*, 2013).

Under increased CO_2, the productivity of wetlands vegetation (seagrass, mangrove trees, saltmarsh plants) is expected to increase in the future (Wong *et al.*, 2014). Seagrasses are likely to be among the species that perform better in a more acidified ocean, because their growth can benefit from increasing dissolved CO_2 (Koch *et al.*, 2012). This simulation result is supported by greater growth rates reported around natural marine CO_2 seeps, where seagrass sequestered considerably more carbon below-ground under acidified conditions, suggesting a possible feedback to reduce the impacts of CO_2 injection into marine waters (Russell *et al.*, 2013). However, there is limited evidence that elevated CO_2 will increase seagrass resistance to warming (Jordà *et al.*, 2012). For mangroves, increased CO_2 has been linked to variable responses in net primary productivity, with decreased NPP projected for *Laguncularia racemosa* and increased NPP for *Rhizophora mangle* (Farnsworth *et al.*, 1996; Snedaker & Araújo, 1998). Such variation may be due in part to methodological differences, but may also reflect important variations in regional conditions (McKee, 2011).

4.2.2.2.3 Climate change impacts in deep seas

The deepsea (defined here as >200m depth) covers about 60% of global ocean area and represents the largest ecosystem in the world (Smith *et al.*, 2009; Watling *et al.*, 2013), accounting for more than 95% of the volume of the Earth's oceans. Deep sea ecological processes and characteristics (e.g., nutrient cycling, productivity) underlie the healthy functioning of ocean ecosystems and provide valuable services to mankind (Thurber *et al.*, 2014).

Many observational studies have shown that present-day climate change is already impacting deep sea environments due to increased temperature (Purkey & Johnson, 2010), deoxygenation (Helm *et al.*, 2011; Keeling *et al.*, 2010; Stramma *et al.*, 2008, 2012), lowered pH of intermediate deep-waters (Byrne *et al.*, 2010), and altered particulate organic carbon (POC) flux to the seafloor (Ruhl & Smith, 2004; Smith & Stephenson, 2013). Elevated seafloor temperatures (3.7°C at the bathyal seafloor by 2100 under RCP8.5; Mora *et al.*, 2013b; Sweetman *et al.*, 2017) will lead to warming boundary currents which has the potential to massively release methane from gas hydrates buried on margins (Johnson *et al.*, 2015; Phrampus & Hornbach, 2012), especially in the Arctic, with simultaneous effects on water column de-oxygenation and ocean acidification (Biastoch *et al.*, 2011; Boetius & Wenzhöfer, 2013). Along canyon-cut margins such as those that occur in the western Mediterranean, warming may additionally reduce density-driven processes, leading to decreased organic matter transport to the seafloor (Canals *et al.*, 2006).

Climate change is also likely to increase wind-driven upwelling in eastern boundary currents, stimulating photosynthetic production at the surface (Bakun, 1990; Bakun *et al.*, 2015; Wang *et al.*, 2014). This new production may, however, decay as it sinks and increase biogeochemical drawdown of O_2. Upwelling may also bring low-O_2, high-CO_2 water onto the shelf and upper slope (Bakun, 1990; Bakun *et al.*, 2010; Feely *et al.*, 2008; Sydeman *et al.*, 2014; Wang *et al.*, 2014). The expansion of hypoxic zones is expected to affect many aspects of deep-sea ecosystem structure and function (Gooday *et al.*, 2010).

As O2 levels decline, many species of deep water octocorals (including gorgonians and pennatulaceans) which provide habitat for a diverse array of invertebrates, are expected to decrease in abundance (Buhl-Mortensen *et al.*, 2010; Etnoyer & Morgan, 2005; Murray Roberts *et* *al.*, 2009). Acidification of deep waters has been projected to negatively impact cold-water stony corals (Scleractinia), particularly in the North Atlantic (Tittensor *et al.*, 2010). Single stressors like warming will also limit tolerance windows for other stressors such as low O2 or low pH (Pörtner, 2012; Pörtner & Knust, 2007).

With the projected global reduction in the biomass of phytoplankton in the upper ocean (Bopp *et al.*, 2013; section 4.2.2.2.1), the flux of particulate organic carbon (POC) to feed open ocean seafloor communities is expected to decrease, causing potential alterations of the biomass, composition and functioning of the benthic communities. Reductions in seafloor POC flux will be most drastic in the oceanic gyres and equatorial upwelling zones, with the northern and southern Pacific Ocean and southern Indian Ocean gyres projected to experience as much as a 32–40% decline in POC flux by the end of the century (CMIP5, RCP8.5; Mora *et al.*, 2013b; Sweetman *et al.*, 2017). Recent studies have suggested that the NE Atlantic Ocean could also undergo similar reductions in POC flux (Jones *et al.*, 2014). The abyssal ocean is highly sensitive to changes in the quantity and quality of POC flux that could affect the biomass of benthic microbial and faunal biomass, and cause dramatic reductions in the sediment mixed-layer depth, benthic respiration, and bioturbation intensity (Jones *et al.*, 2014; Smith *et al.*, 2008; Sweetman *et al.*, 2017). These changes have the potential to feed back on global carbon cycling and ultimately C-sequestration (Thurber *et al.*, 2014).

4.2.2.2.4 Climate change impacts in polar seas

Rising temperatures are projected to reduce sea ice extent and volume in the Arctic and Antarctic, some of the fastest warming places on Earth (IPCC, 2013). The rapid rate at which sea ice retreats in polar seas implies major changes to be expected in the future for biodiversity and ecosystem function (Gutt *et al.*, 2015; Larsen *et al.*, 2014; Wassmann *et al.*, 2011). All components of the food webs will potentially be impacted, from phytoplankton to top predators, and from pelagic to benthic species.

Multiple lines of evidence show that ice-melting is likely to increase primary productivity in polar seas due to increased light availability, although this could be dampened by a decrease in nutrient supply due to enhanced water column stratification that is expected from warming and freshening of surface waters (section 4.2.2.2.1; Hoegh-Guldberg *et al.*, 2018; Larsen *et al.*, 2014). It has also been shown that the increased production of floating icebergs, enriched with terrigenous material, might significantly elevate nutrient levels and primary production (Smith *et al.*, 2007). However, while primary production may increase in polar seas in the future, warmer waters can cause a shift in the composition of the zooplankton community, such as the shift from *Calanus glacialis* towards dominance of the smaller, less energy-rich *Calanus finmarchicus* in Arctic waters (Kjellerup *et al.*, 2012), with potential huge consequences up the food chain. By contrast, in coastal areas, the production and transport of organic matter to the seafloor may decline because glacial meltwater and erosion of melting tundra (Węsławski *et al.*, 2011) will likely enhance water column turbidity, which results in decreased water column light levels (Grange & Smith, 2013; Sahade *et al.*, 2015). The increased sedimentation in deep coastal areas, particularly in Arctic fjords, may also smother or clog the breathing and feeding apparatus of sessile suspension-feeders (e.g., corals and sponges), induce O_2 stress, but may favour ophiuroids and capitellid polychaetes (Sweetman *et al.*, 2017; Wlodarska-Kowalczuk *et al.*, 2005).

Changes in primary production and resulting POC flux to the seafloor will have impacts on ecosystem structure and function. Elevated POC flux increases the abundance and diversity of benthic communities, the prevalence of habitat-forming taxa (sponges, benthic cnidarians), and the extension of species ranges into deeper waters (De Rijk *et al.*, 2000). It could also trigger the switch from dominance by bacteria to dominance by metazoans for processing benthic organic matter with bottom-up consequences on the food-web (Sweetman *et al.*, 2014). Changing ice regimes may also result in physical disturbance of the deep sea, as large icebergs can scour the sediment down to 400m on the Antarctic shelf, enhancing seafloor heterogeneity and creating hard substrates for sessile megafauna (Meyer *et al.*, 2015, 2016; Schulz *et al.*, 2010). In the longer term, iceberg scouring and dropstone deposition will tend to elevate diversity on regional scales through (re)colonization processes, although the immediate effect of scouring will be local elimination of many species (Gutt & Piepenburg, 2003; Gutt *et al.*, 1996; Thatje *et al.*, 2005).

Sea ice melting is also expected to impact species up the food-web, and especially those marine mammals and seabirds depending on ice as haul-outs, but future scenarios are available for just a few emblematic species. Demographic models predict that changes in Antarctic sea ice will substantially reduce the abundance of global emperor penguin (*Aptenodytes forsteri*) by 2100 under a mid-range emission scenario (Jenouvrier *et al.*, 2014), even when complex dispersal processes are included (Jenouvrier *et al.*, 2017). A high probability of extinction is foreseen for the polar bear (*Ursus maritimus*) subpopulation of southern Beaufort under SRES A1B scenario by the end of the century, due to the decrease in the cover, the duration and the thickness of sea ice (Hunter *et al.*, 2010), but low probability of extinction has been attributed for all polar bears in the Arctic (Larsen *et al.*, 2014). However, a recent study showed that the high-energy requirements of polar bears could endanger their survival in extended ice-free periods (Pagano *et al.*, 2018).

Ocean acidification is another major stressor which will be enhanced in polar regions because of the higher capacity of seawater to absorb CO_2 at low temperatures, resulting in lower pH and under-saturated waters in aragonite and calcite (Hoegh-Guldberg *et al.*, 2014; Orr *et al.*, 2005). This may impact the growth and survival of calcifying shelled organisms such as Arctic pteropods, foraminifera in the Southern Ocean, and the recruitment of Antarctic krill *(Euphausia superba)*, all of those species being essential prey species at the basis of food-webs (Kawaguchi *et al.*, 2013; Larsen *et al.*, 2014; Trathan & Hill, 2016). Adding to the negative impacts of acidification, a combination of ice retreat and changes in primary production is projected to decrease Antarctic krill suitable habitat and survival rate (Piñones & Fedorov, 2016) with potential cascading effects on their many predators (Trathan & Hill, 2016).

4.2.2.3 Future impacts of fisheries exploitation on marine ecosystems

In addition to exposure to climate change, marine animal populations will likely undergo increased fishing pressure as a result of increasing demand for fish products (World Bank, 2013) particularly in the developing world (**Figure 4.2.6**; FAO, 2016). This will largely be driven by growth of human population that is projected to reach 9.8 billion people by 2050 (UNDESA, 2017) and by income growth in low- and middle-income countries (Vannuccini *et al.*, 2018). The rate of increase in demand for fish has been more than 2.5 per cent per year since 1950 and is likely to continue in the future (HLPE, 2014). The world fish production (capture and aquaculture) was projected to increase by 17% between the base period (2013-2015) and 2025 (FAO, 2016). With the growing demand, commercial fishing activities are likely to expand to all areas of the globe.

Scenarios that include governance in fisheries management, human consumption of seafood, and advancement of fishing technologies (Squires & Vestergaard, 2013) are starting to be integrated into global scale projections. For example, a simple surplus production model applied to a set of 4713 fisheries worldwide showed that a business-as-usual fisheries management scenario would increase the proportion of overexploited populations by ca. 30% in 2050 (Costello *et al.*, 2016). In contrast, in a scenario where long-term economic benefits are optimized, such as through rights-based fisheries management, the majority of exploited fish populations (98%) would recover to a healthy status, with a median time of recovery of about 10 years. Similarly, under the high emission scenario RCP8.5 and the SSP3 scenario (characterized by low economic development and a large increase in human population), maximizing the long term economic yield of the fishery was projected to increase the biomass of the skipjack tuna population (Dueri *et al.*, 2016). Recently, it was shown that reforming fisheries by adopting an optimal harvest policy that maximizes long-term economic benefits and that adapts its management strategy to climate-induced changes in fish biomass and spatial distribution could offset the detrimental impacts of climate change on future fish biomass and catch under most RCP greenhouse gas emission scenarios, except RCP8.5 (Gaines *et al.*,

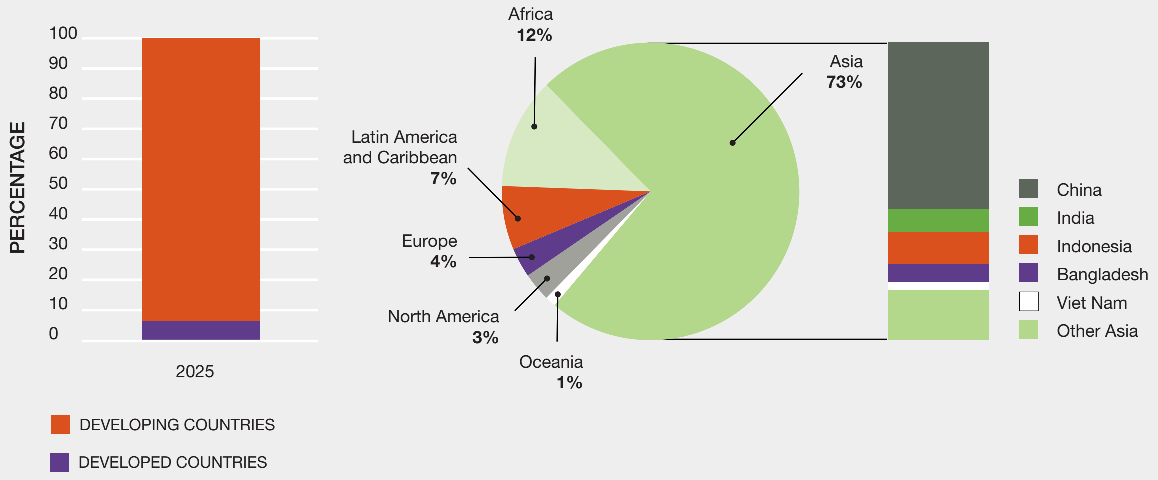


Figure ④ ② ⑥ **Projections of additional fish consumed in 2025 (from fisheries and aquaculture) per world region.**

Developing countries are projected to eat 93 percent of the additional fish available for human consumption. Source: OECD and FAO (FAO, 2016).

2018). This important finding needs to be consolidated by further investigations in a context where fisheries maximum catch potential is projected to decrease by 2.8-5.3% and 7-12.1% by 2050 relative to 2000 under RCP2.6 and RCP8.5, respectively (Cheung *et al.*, 2018).

In addition to climate change (see 4.2.2.2.1), heavy fishing also impacts fish size, decreasing both the maximum size of species and the biomass of large-sized species because (i) high-value target species are generally larger, (ii) fishing gear is size-selective and often designed to remove larger fish, (iii) older and larger fish in a population become fewer as a result of accumulation of fishing mortality rate through time, and (iv) large species are more vulnerable because their life-history traits are generally linked to lower potential rates of increase (Shin *et al.*, 2005). Under heavy fishing, a SRES A1B climate change scenario was reported to magnify the reduction in fish size (Blanchard *et al.*, 2012). This shift towards smaller fish size and higher growth rates could ultimately increase the variability of fish biomass (Hsieh *et al.*, 2006).

Species targeted by fisheries are not the only species impacted by different fishing scenarios. Long-lived and vulnerable species such as marine mammals, turtles and birds suffer from direct impact of fish harvest though bycatch, and so their future is tightly linked to the long-term fishing strategies adopted. The interaction with climate change is complex to resolve but some studies have started addressing the potential synergistic effects. Some models based on species distribution projected that climate change will alter the future distribution of both fisheries and seabird populations, altering the rates of future bycatch and hence seabird mortality rates (Krüger *et al.*, 2018). For some species, spatial overlap with fisheries may decline, reducing rates of incidental mortality associated with human activity. However, for two highly threatened seabird species (grey-headed and wandering albatross), severe range reductions and increased overlap with fisheries are projected.

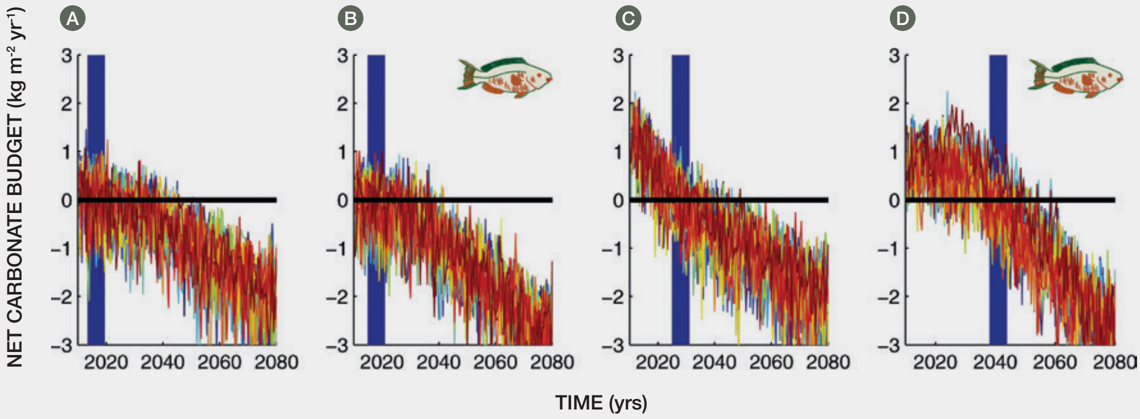
In addition to scenarios of fishing management, the future status of wild fish populations cannot be envisaged without considering alternative scenarios of aquaculture development which will play a major role in sustaining the supply of seafood products and the maintenance of per capita fish consumption (Delgado *et al.*, 2003; FAO *et al.*, 2018). But the development of aquaculture is partly dependent upon the exploitation of low trophic level fish species which supply fishmeal for farmed fish.

Aquaculture development could potentially reduce fishing pressure on wild fish populations, but not to an extent that could compensate for projections of increases in demand for seafood products and fishing technology, both of which result in increased fishing pressure (Quaas *et al.*, 2016). Taking into account projections in human population, climate change (IPCC A1B scenario), and technological development in aquaculture, a bio-economic model projected that if fishmeal prices increase, this would encourage fishers to maximize their short-term economic profits and exceed yearly quotas, leading to collapse of exploited fish populations (Merino *et al.*, 2012). Given the current increasing trends of fishmeal prices (Merino *et al.*, 2010), this implies that compliance to strict fisheries management and market stabilization measures need to be seriously considered to maintain exploited populations at sustainable levels. Likewise, another bio-economic model run under contrasted archetype scenarios suggested that relative to climate change impacts, fisheries regulation is the most important factor in determining the future of fish populations (Mullon *et al.*, 2016). However, the interplay between drivers of change cannot be ignored in fisheries management strategies (see example in **Box 4.2.3**). A multi-model ensemble approach allowed to show that the risk of negative synergistic effects between changes in primary production and in fishing effort was higher for small forage fish species (Fu *et al.*, 2018).

Box ④②③ Synergistic impacts of multiple drivers on tropical coral reefs.

Tropical coral reefs share a history of strong dependence on natural and human systems (Maire *et al.*, 2016) that must be accounted for in attempts to maintain long-term human development and well-being, and marine biodiversity (Cinner *et al.*, 2016). Indeed, coral reefs support the nutritional and economic needs of people in many developing countries. Their exceptional biodiversity translates directly into biomass production and thus food security (Duffy *et al.*, 2016). However, coral reefs face multiple and considerable challenges from ocean warming (see 4.2.2.2.2), ocean acidification, pollution, overexploitation and destructive fishing practices. More than 80% of the world's coral reefs are severely over-fished or have degraded habitats, thus imperiling the livelihood and sustenance of coastal human populations (McClanahan *et al.*, 2015). This negative spiral is likely to accelerate in the future due to the synergistic effects of climate change and direct human impacts. For example, nutrient loads from the land increases the vulnerability of corals to bleaching (Vega Thurber *et al.*, 2014). Plastic debris were estimated to increase coral susceptibility to diseases from 4% to 89% with structurally complex corals being eight times more likely to be affected by plastic (Lamb *et al.*, 2018) inducing a loss of fish productivity (Rogers *et al.*, 2014). Tipping points exist at which coral reef ecosystems can shift to being dominated by macroalgae (Holbrook *et al.*, 2016), with low resilience, reductions in biodiversity and degradation of the many ecosystem services they provide, such as reef-associated fisheries and tourism. However, there are opportunities for improving the status of coral reefs by the combined action of reducing both greenhouse gas emissions and overfishing of species which help the recovery of coral reefs by grazing their algal competitors (**Figure 4.2.7**; Kennedy *et al.*, 2013). Robust, integrated models that can account for combinations of multiple impacting drivers are still lacking, but these are needed to simulate the dynamics of coral reef social-ecological systems on a long-term basis and better anticipate their futures. This challenge is even more difficult given the multispecies nature of fisheries, the complexity of trophic interactions, and the time scales on which different processes determine the trajectories of coral reef social-ecological systems and the boundaries beyond which they collapse.

BUSINESS-AS-USUAL GHG EMISSIONS



LOW GHG EMISSIONS

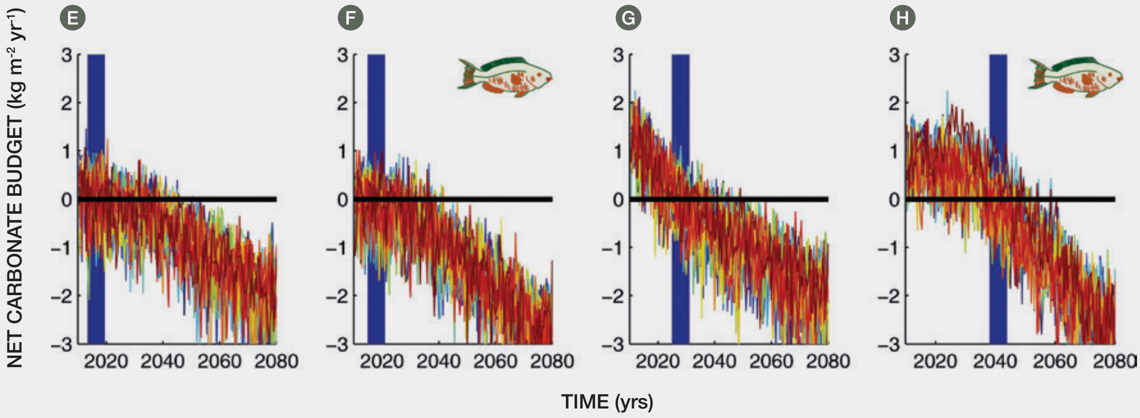


Figure ④ ② ⑦ **Future carbonate budgets (proxy for net production of corals skeletons) of Caribbean coral reefs under climate change and acidification scenarios (top panel: high RCP8.5 greenhouse gas emission scenario, bottom panel: strong mitigation RCP2.6 emission scenario), without or with local conservation of grazing fish (parrot fish symbol in Ⓑ, Ⓓ, Ⓖ, Ⓗ).**

Initial conditions of reefs are either degraded with 10% coral cover (Ⓐ, Ⓑ, Ⓔ, Ⓕ) or healthier with 20% coral (Ⓒ, Ⓓ, Ⓖ, Ⓗ). Vertical blue bars indicate point at which the projected budget becomes negative (erosion of corals skeleton exceeds production). Source: Kennedy *et al.* (2013).

4.2.2.4 Future impacts of pollution on marine ecosystems

4.2.2.4.1 Persistent organic pollutants and plastics: another 'Silent Spring'?

Over the last century the human enterprise has fundamentally altered the planet by releasing large quantities of persistent organic pollutants (POPs) into the environment. These synthetic organic compounds have harmful and toxic properties and are not readily metabolized by bacteria or other life forms, thus prolonging their presence in the environment. Concerns about their effects on wildlife and people were first raised by Rachel Carson's book 'Silent Spring' (Carson, 1962), highlighting the devastating effects of organochlorine POPs on birds and aquatic animals in particular. As a result, many POPs were tightly regulated or banned under the Stockholm Convention (UNEP, 2001), and their production has ceased or decreased for most listed substances. Large historical burdens of these pollutants still circulate in the environment however (Harrad, 2009), and novel substances get synthesized at a rapid pace, with potentially harmful effects.

Synthetic organic polymers (plastics) form another class of pollutants that share certain properties with POPS in that they persist and accumulate in the environment, can be transported over long distances (reaching remote polar regions for example; Science for Environment Policy, 2017), and can have harmful effects on wildlife and people. In contrast to POPs, their production numbers are much higher overall and still increasing, thus global concerns about plastic pollution now match or exceed those for other POPs, particularly with respect to the marine environment which forms a sink for discarded plastic waste (Jambeck *et al.*, 2015; Worm *et al.*, 2017). Annual plastic production now exceeds 330 million metric tons (Mt) (PlasticsEurope, 2015), with a cumulative burden of 8300 Mt produced since 1950 (Geyer *et al.*, 2017), approximately 6300 Mt of which has been discarded (9% recycled, 12% incinerated, and 79% ended in landfills or the natural environment). If current production and waste management trends continue, roughly 12,000 Mt (million tons) of plastic waste will be in landfills or in the natural environment by 2050 **(Figure 4.2.8)**. If evenly spread around the globe, this would equal a burden of ~24 tons of plastic waste for each square kilometre of land and sea surface. This level of pollution in terms of volume and persistence has no previous analogue in human history.

Negative impacts on the planet and people are becoming more profound **(Figure 4.2.9)** as exposure to plastic pollutants intensifies. As an example, about 90% of seabirds examined today have plastic in their gut, with

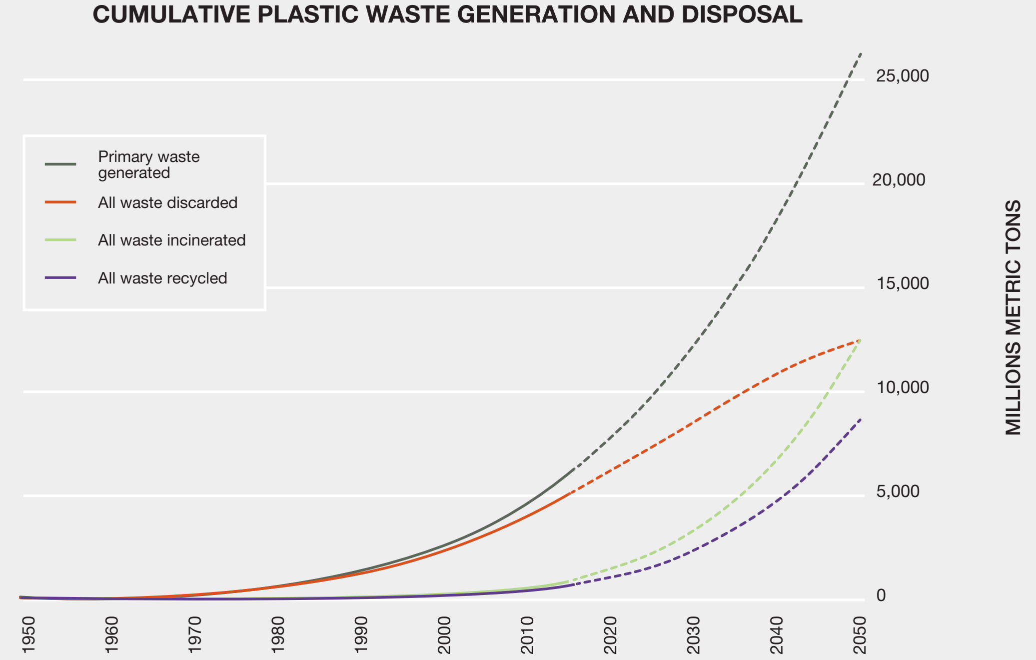


Figure ④ ② ⑧ **Current global trends and likely future trajectories of total plastic waste generation and management.**

After data in Geyer *et al.* (2017).

100% expected to be exposed by 2050 (Wilcox *et al.*, 2015). Sea turtles are similarly affected (Schuyler *et al.*, 2015), as are at least 693 other marine species that have been recorded to be compromised by plastic pollution (CBD, 2016). Much of the plastic is released as or broken down into small microplastic (1 µm-1mm) or nanoplastic (<1µm) particles. While the harmful effects of microplastic debris are well understood, the long-term effects of the smallest fragments are only now emerging (Galloway & Lewis, 2016), including their tendency to interact with other pollutants (GESAMP, 2015), facilitate diseases (Lamb *et al.*, 2018), and transmit through the food chain **(Figure 4.2.9)**.

Clearly, another 'Silent spring' scenario seems plausible, if effects on numerous wildlife species continue to accelerate further. Because plastic persists and accumulates in the environment in similar ways POPs do, a zero-net-release policy that builds upon the successful Stockholm Convention (SC) on Persistent Organic Pollutants (POPs) may be a promising strategy to mitigate the risk posed by current and future levels of plastic pollution. Yet, in contrast to traditional POPs, which are largely emitted by industry, plastic pollution touches every person's life, and requires a broader societal effort including designers, producers, regulators, and consumers of plastic products to engage in comprehensive solutions (GESAMP, 2015; Worm *et al.*, 2017).

4.2.2.4.2 Nutrient loads and eutrophication

Numerous model projections show that coastal zones in many world regions are almost certain to see increases in nitrogen (N) and phosphorus (P) from increasing river loads in the coming decades (Sutton *et al.*, 2013; **Figure 4.2.10**). In contrast, silica (Si) river export is decreasing globally as a result of retention in the increasing number of reservoirs in the world's river systems and this trend will also continue in many parts of the world. The result of these simultaneous changes of N, P and Si will continue to alter nutrient stoichiometry, affecting not only total algal growth but also biodiversity in coastal waters, including the propensity for harmful algal blooms (HABs). The enhanced primary production in coastal surface waters can cause eutrophication, with subsequent sinking of excess degradable organic matter to bottom waters where aerobic microbial decomposition reduces oxygen concentration. The decline in oxygen concentrations due to nutrient loads in coastal waters will likely be exacerbated with climate change, due to decreased oxygen solubility in warmer waters and decreased oxygen transport to deeper waters because of stronger stratification of the water column (Breitburg *et al.*, 2018). The expansion of areas of low oxygen will impact marine biodiversity at all levels from individuals' physiology and behavior, to populations' demography and range shifts with consequences for

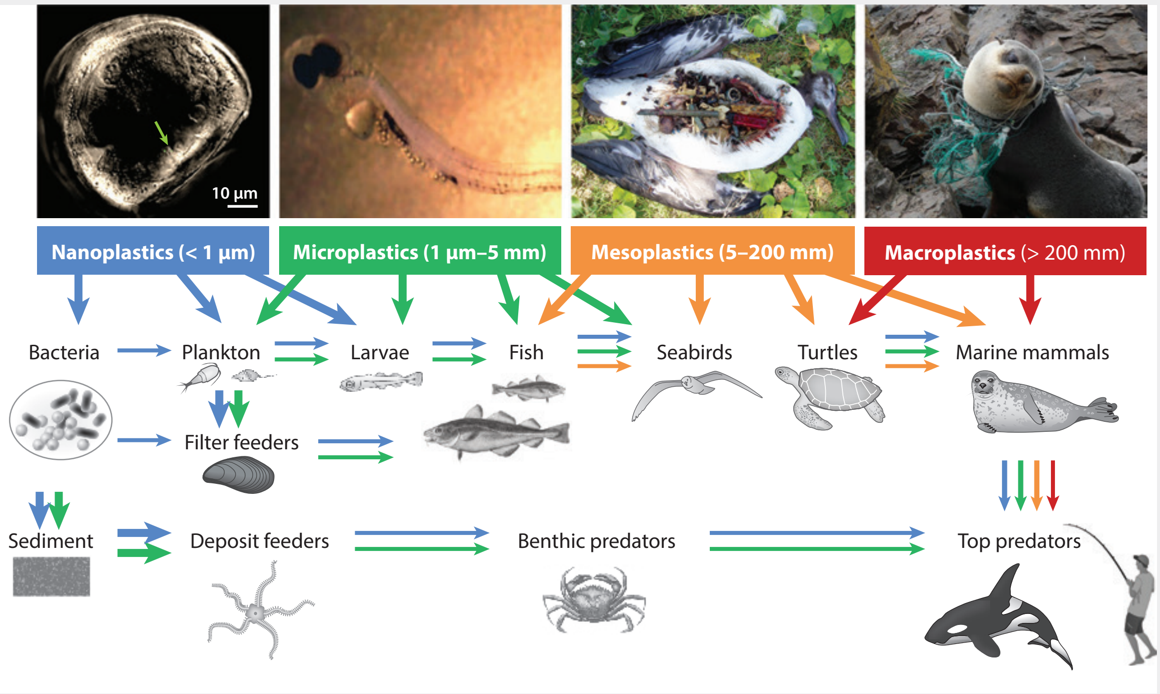


Figure ④ ② ⑨ **Possible pathways by which plastic pollutants of different size classes enter the food chain and propagate to higher trophic levels, including humans.**

After Worm *et al.* (2017).

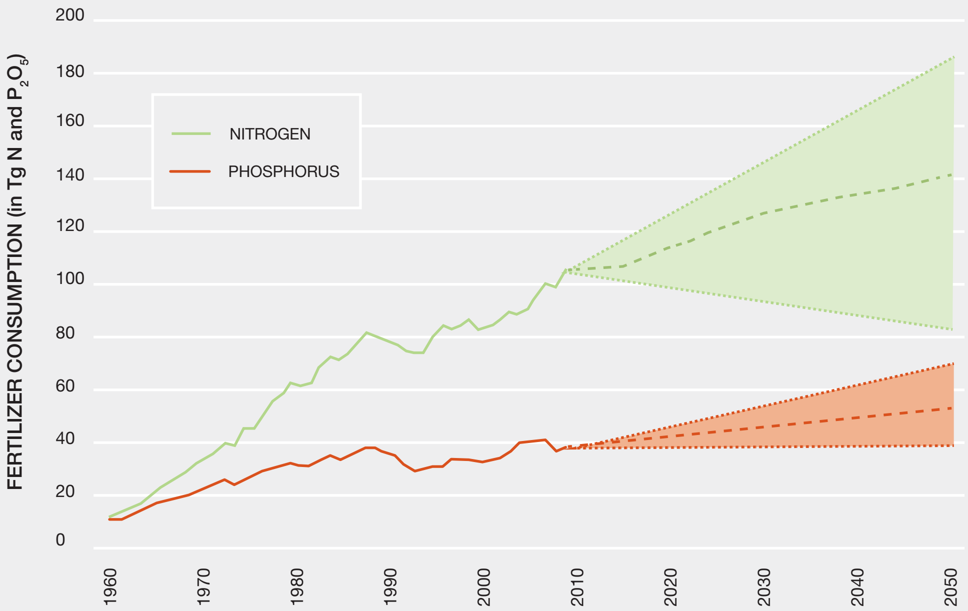


Figure ④ ② ⑩ **Trends in global mineral fertilizer consumption for nitrogen and phosphorus and projected possible futures.**

Source: Sutton *et al.*, 2013.

species assemblages and food-webs (Levin *et al.*, 2009; Pörtner *et al.*, 2014).

Storylines developed by the IPCC and the Millennium Ecosystem Assessment and translated into changes of the main anthropogenic drivers, i.e. economic development, demography and land use (Alcamo *et al.*, 2007), have been applied to project conditions to 2050. Although each storyline has different assumptions, they show major increases in N and P river export especially in South and Eastern Asia, in South America and Africa where fertilizer use will likely increase to support the population, and where urbanization and lagging treatment of wastewater and sewage connection will lead to increasing nutrient discharge to surface water (e.g., Glibert *et al.*, 2018). In contrast, stabilized or decreasing trends in nutrient loads are projected in Europe, North America and Australia owing to the development of improved wastewater treatment systems, and improved nutrient management reducing NH_3 volatilization, leaching and run-off. In these regions, improvements in hypoxia and frequency or magnitude of HABs may be realized.

However, the trajectory of nutrient loads is additive with other global changes, such as temperature rise, which will alter stratification of the water column, availability of nutrients and their forms and ratios, and pCO2, among other factors (e.g., Boyd & Doney, 2003). Recent models supported evidence for increased eutrophication together with climate changes, and therefore the propensity for the worsening of HABs and/or hypoxia by the end of the century (Sinha *et al.*, 2017). Multiple combined changes such as increases in nutrient pollution, in global temperature and in reservoir capacity resulting in increased retentiveness of rivers, require proactive management to stabilize or reduce the impacts of eutrophication, including hypoxia and the frequency of HABs.

4.2.2.5 Future impacts of coastal development on marine ecosystems

Direct human-related drivers of change such as urbanization, coastal development, and land-use change will bring challenges to coastal ecosystems in addition to climate change. Coastal populations are increasing disproportionately relative to the global population increase. Many of emerging cities are on the coast and their growth will add to the 75% of the world's mega-cities which are already coastally located (World Economic Forum's Ocean Programme, 2017). Over 2.6 billion people live on or near the coast, many in developing countries where dependence on coastal resources may be high and demand for multiple benefits such as food, coastal protection and income, will continue to grow as human populations expand (Bell *et al.*, 2009; Sale *et al.*, 2014). Some 1.36 billion live on tropical coasts, and this is projected to grow to 1.95 billion by 2050, with associated pollution and eutrophication of coastal waters and degradation of coastal ecosystems (Sale *et al.*, 2014). Urbanization and coastal development can restrict the capacity of coastal ecosystems to adapt to rising sea levels e.g. through the "coastal squeeze" (Wong *et al.*, 2014). Along urbanized coastlines, the resilience of wetlands to SLR will depend on the availability of accommodation space (Schuerch *et al.*, 2018) and sediment supply (Lovelock *et al.*, 2015) which are reduced by anthropogenic infrastructure barriers (e.g., flood protection structures, roads, settlements). Future expansion of coastal development will also bring risks to iconic and threatened species. For example, the expansion of artificial lighting at night from coastal development interrupts the sea-finding behaviour of sea turtle hatchlings and ultimately survivorship (Gaston & Bennie, 2014; Kamrowski *et al.*, 2014).

Future projections show a multiplicity of human stressors acting simultaneously with direct climate-induced changes on social-ecological systems. Stressors from population growth and coastal development such as nutrient run-off, urbanization, and land-use change are expected to increase and combine with climate stressors such as sea level rise and warming to exacerbate risks for rocky and sandy shores, and seagrasses **(Box 4.2.4)**. Models show that mangroves are particularly threatened by projected coastal development, with the main direct drivers including the expansion of aquaculture (prevalent in both Asia and Latin America) and agriculture (mostly rice cultivation and pasture), extraction of timber and related forest products (e.g., for charcoal and domestic construction), and infrastructure development and alterations of freshwater flows (e.g., for due to settlements, transportation networks or dams) (Roy Chowdhury *et al.*, 2017). Under projected changes, coastal adaptation options will involve increasingly difficult trade-offs in future among multiple development and biodiversity objectives (Mills *et al.*, 2015).

4.2.3 Freshwater ecosystems

4.2.3.1 Freshwater biodiversity and current threats

Freshwater ecosystems provide fundamental services to humans such as food, water, nutrient retention, recreation, and climate regulation. Globally, freshwaters (i.e. rivers, lakes, wetlands) represent less than 0.02% of Earth's water volume and cover only about 0.8% of Earth's surface (Dawson & Dawson, 2012). However, an estimated 129,000 species live in freshwater ecosystems, representing ~8% of Earth's described species (Balian *et al.*, 2008; **Figure 4.2.11**). The relative contribution of freshwater ecosystems to global biodiversity is thus extremely high (Tedesco *et al.*, 2017; Wiens, 2016). Climate, productivity and area size drive freshwater diversity patterns globally despite profound functional differences between taxa (Moomaw *et al.*, 2018; Tisseuil *et al.*, 2013).

Box ④ ② ④ Synergistic impacts of multiple pressures on seagrass meadows.

Direct human-related drivers of change such as urbanization, coastal development, and land-use change will bring challenges to coastal ecosystems. For seagrasses, key threats include sediment and nutrient run-off from upstream land-use change, physical disturbance, algal blooms, and invasive species, as well as climate warming and disease (Orth *et al.*, 2006; Waycott *et al.*, 2009). Requirements for clear water and low nutrient concentrations make seagrasses vulnerable to eutrophication, as nutrient and sediment loading reduce light availability and favor faster-growing algae (Burkholder *et al.*, 2007; Duffy *et al.*, 2013). The protected embayments in which seagrasses grow best are also prime real estate for coastal and harbor development. As a result seagrasses are declining worldwide, and roughly 30% of global seagrass cover has been lost since the first estimates were made in the late 19th century, with loss rates increasing in recent decades (Waycott *et al.*, 2009). Ten of the 72 known seagrass species on earth are at elevated risk of extinction and three species are classified as Endangered (Short *et al.*, 2011).

Perennial organisms such as seagrasses are vulnerable to human disturbance and, under repeated impacts, often yield dominance to faster growing, opportunistic species such as fleshy and filamentous algae. In the Baltic Sea, for example, dominance by eelgrass and rockweed has yielded over recent decades to accumulations of ephemeral algae (Bonsdorff *et al.*, 1997). Long-term field monitoring suggests that exploitation of piscivores such as cod in offshore waters has released the smaller inshore fishes—mesopredators—from top-down control, and their consumption of grazing invertebrates indirectly led to algal blooms and decline of perennial seagrasses (Eriksson *et al.*, 2011). Coastal vegetation, including seagrasses, protects coastal human communities against storm damage, and the continuing decline of these natural barriers will likely be aggravated by SLR. Coastal habitat loss exacerbates damage from storms and flooding in coastal communities (Gedan *et al.*, 2011). Mapping the risk of such hazards along the coastline of the USA shows that, under several projected climate scenarios, the number of people, especially the poor and elderly, and the total value of residential property exposed to hazards could be reduced by half by preserving existing coastal habitats (Arkema *et al.*, 2013).

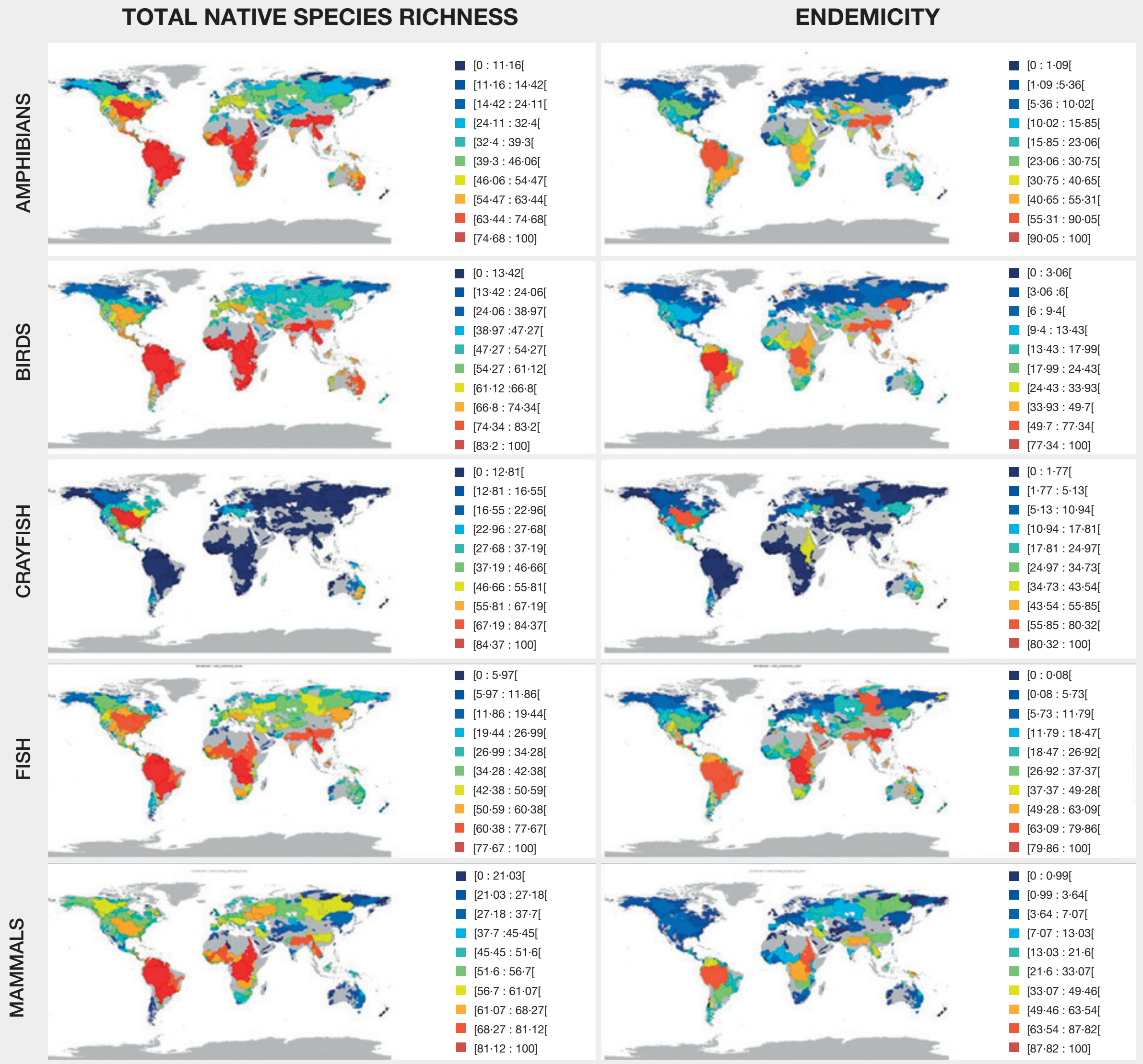


Figure 4 2 11 **Global diversity maps (species richness and endemicity) for freshwater fishes, aquatic amphibians, aquatic mammals, crayfish and aquatic birds.**

For comparison purpose, the diversity descriptor values of each taxon are rescaled between 0 and 100. Study based on the global distributions of 13, 413 freshwater species among five taxonomic groups (i.e. 462 crayfish, 3263 amphibians, 8870 fish, 699 birds and 119 mammals) and conducted on 819 river drainage basins covering nearly 80% of Earth's surface. After Tisseuil *et al.* (2013).

Current major threats to freshwater biodiversity include climate change, habitat modification and pollution from land-use, habitat fragmentation and flow regime homogenization by dams, non-native species, increased eutrophication resulting from nutrient and organic discharges, water abstraction, and overexploitation (Young *et al.*, 2016). Those threats currently affect freshwater biodiversity and functioning to varying degrees (Carpenter *et al.*, 2011; Vörösmarty *et al.*, 2010), and their additive and potentially synergistic effects may further threaten future freshwater biodiversity and resources (Collen *et al.*, 2014; Knouft & Ficklin, 2017).

4.2.3.2 Future climate change impacts on freshwater biodiversity and ecosystem functioning

The lowest greenhouse gas emissions scenario is the only scenario not expected to threaten much of global freshwater biodiversity in 2050 through direct effects of climate change. Under all other scenarios, freshwater biodiversity is expected to decrease proportionally to the degree of warming and precipitation alteration. All water body types on all continents are likely to be affected. Warmer waters will alter community structure, food webs, body sizes, and species ranges — especially in regions where semi-arid and Mediterranean climates currently occur as well as high-mountain ecosystems. In addition to reduced biodiversity and ecosystem functioning, warmer and less water will lead to species extinctions because of habitat shrinkage.

Scenarios of climate change impacts on global freshwater ecosystem biodiversity and functioning were reviewed by Settele *et al.* (2014). Climate change alters freshwater ecosystems and their biodiversity by changing (1) temperatures, (2) water availability and (3) flow regimes through changes in precipitation (Döll & Zhang, 2010; Knouft & Ficklin, 2017) and/or temperature (Blöschl *et al.*, 2017). Increased water temperatures often lead to progressive shifts in the structure and composition of assemblages because of changes in species metabolic rates, body size, migration timing, recruitment, range size and interactions (Daufresne *et al.*, 2009; Myers *et al.*, 2017; Parmesan, 2006; Pecl *et al.*, 2017; Rosenzweig *et al.*, 2008; Scheffers *et al.*, 2016). There is already evidence of regional and continental shifts in freshwater organism distributions following their thermal niches (Comte *et al.*, 2013), local extirpations through range contractions at the warm edges of species' ranges (Wiens, 2016), and body size reductions (Daufresne *et al.*, 2009). Warmer water temperatures also enhance microorganism metabolism and processing of organic matter (unless dissolved oxygen is limiting), causing eutrophication when nutrient levels are high (Carpenter *et al.*, 2011; Mantyka-Pringle *et al.*, 2014) as well as increased omnivory. Warming also induces phenological mismatches between consumers and resources in highly seasonal environments, potentially destabilizing food-web structure (Woodward *et al.*, 2010a).

The strongest temperature increases are projected for eastern North America (0.7 to 1.2 °C under RCP2.6 and RCP8.5, respectively, by 2050), Europe (0.8 to 1.2 °C), Asia (0.6 to 1.2°C), southern Africa (>2.0°C under RCP8.5) (van Vliet *et al.*, 2016b) and Australia (CSIRO & Bureau of Meteorology, 2015). Moderate water temperature increases (<1.0°C) by 2050 are predicted for South America and Central Africa (Van Vliet *et al.*, 2013; van Vliet *et al.*, 2016b). Changes in water temperature are projected to lead to local or regional population extinctions for cold-water species because of range shrinking especially under the RCP 4.5, 6.0 and 8.5 scenarios (Comte & Olden, 2017). Most lowland-tropical freshwater species are expected to tolerate warmer conditions where water is sufficient (Comte & Olden, 2017).

Decreased water availability and altered flow regimes reduce habitat size and heterogeneity. This increases population extinction rates because the probability of species extinctions increases with reduced habitat size (Tedesco *et al.*, 2013). Climate change can also alter flow regime seasonality and variability (e.g., Blöschl *et al.*, 2017; Döll & Zhang, 2010) and increase flow intermittency (Pyne & Poff, 2017). This would lead to decreased food chain lengths through loss of large-bodied top predators (Sabo *et al.*, 2010), altered nutrient loading and water quality (Woodward *et al.*, 2010b), and/or pushing taxa into novel trajectories from which they may not recover (Bogan & Lytle, 2011). However, whatever the RCP scenario, climate change impacts on the timing of seasonal streamflow are found to be generally small globally (Eisner *et al.*, 2017). Yet, relative to water availability and according to the wet-wetter/dry-dryer mechanism (Gudmundsson *et al.*, 2017; Held & Soden, 2006; Wang *et al.*, 2017), more severe water stress in current drylands is expected in the future. Although under RCP2.6 the distributions of water availability may change little by the end of the 21st century, RCP4.5, 6 and 8.5 scenarios are expected to induce substantial shrinking of water drainage where semi-arid and Mediterranean climates currently occur. Reduced water availability in those regions, including shifts from permanence to intermittency, will generate population extirpations of all types of freshwater organisms (Jaeger *et al.*, 2014), leading to global net biodiversity losses because endemism is usually high in those regions. For example, projected fish extinction rates from drainage shrinking under the high emission SRES A2 scenario in river basins worldwide show that among the 10% most-altered basins, water availability loss is likely to increase background extinction rates by 18.2 times in 2090 (Tedesco *et al.*, 2013; **Figure 4.2.12**). Also, in glacier-fed high-mountain ecosystems, significant changes to snow and glacier melt regimes, including glacier disappearance, have already been observed (Leadley *et al.*, 2014) and are expected to continue (Kraaijenbrink *et al.*, 2017). This leads to reduced water availability and declines in biodiversity through local population extirpations and species extinctions in regions of high endemicity in all water body types. Besides biodiversity losses, losses of glacial ice in closed drainages and flows in semi-arid regions (Vörösmarty *et al.*, 2010) will substantially decrease water for agriculture, power and public water supply, thereby increasing economic vulnerability in the affected regions (e.g., Moon, 2017).

Wetlands, including peatland and permafrost regions, sequester carbon in their soils. But when confronted to warming, drying and conversions to agriculture, wetlands are expected to release CO_2, CH_4, and N_2O. Global warming alone is projected to contribute 1.6 x 10^8 kilotons of carbon from melting permafrost to the atmosphere and CH_4 emissions from freshwater wetlands are projected to nearly double by 2100 (Moomaw *et al.*, 2018). Such changes are very likely to impact biodiversity negatively due to habitat loss and reduced water quality, which increase the risk of extinctions and extirpations of wetland endemic and dependent species (Segan *et al.*, 2016).

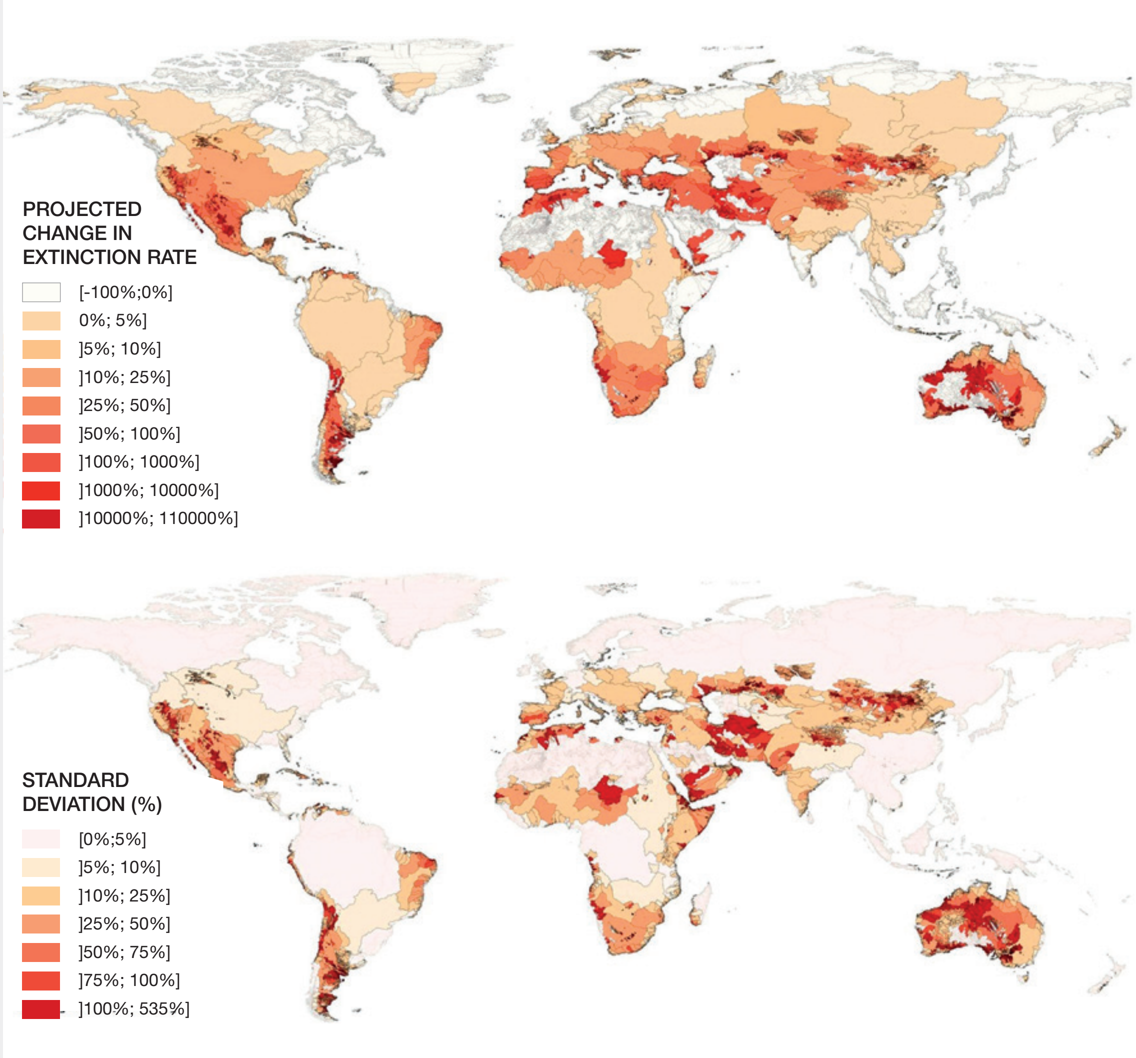


Figure ④ ② ⑫ **Global patterns of proportional increase or decrease in freshwater fish extinction rates between current climatic conditions and future (2090) under the most ‘pessimistic’ IPCC SRES scenario (A2).**

Negative values of projected change in extinction rate depict drainage basins where extinction rates may decrease, while positive values depict drainage basins where extinction rates may increase. 91 949 river drainage basins covering ~99% of the terrestrial surface. After Tedesco *et al.* (2013).

4.2.3.3 Future land-use change impacts on freshwater biodiversity and ecosystem functioning

Land use will likely increase the risk of eutrophication, leading to local population extinctions, changes in community structure and consequent modification of the food-web, ecosystem temporal instability, and establishment and spread of pathogens and toxic cyanobacteria blooms globally. Land use will become especially problematic in the emerging tropical economies because of increased human population density and weak pollution controls. Increasing pollution and eutrophication will degrade water quality, impair biological resource availability, reduce nutrition in developing countries, and reduce recreational opportunities and tourism income. Globally increased toxic cyanobacteria blooms and pathogens will increase health risks for people and livestock. These risks will most affect closed water bodies and estuaries, but rivers will also be threatened. The additional impact of future increasing use of pesticides in agriculture is hard to quantify due to a lack of scenario studies.

Land use, especially croplands, mining and urbanization, will affect freshwater ecosystems and associated biodiversity through two main pathways. First, further increased water and groundwater withdrawals are expected to decrease habitat (water) availability for freshwater organisms leading to increased population extinction rates in rivers and lakes or direct extinctions from wetland conversions (Gardner *et al.*, 2015; Tilman *et al.*, 2001). The problem is exacerbated in semi-arid regions where water withdrawals lead to some rivers and lakes drying routinely, with ensuing species extinctions (Foley *et al.*, 2005). Second, water quality is usually degraded by land use, and this trend is likely to continue. Intensive agriculture increases sediment, nutrient and pesticide loads to ground and surface waters (Lotze *et al.*, 2006; Vasconcelos *et al.*, 2017). The continuing, rapid urbanization also will substantially degrade water quality in many regions mostly through organic or phosphorous loadings, especially where wastewater treatment is absent. Mining leads to increased loadings of toxic metals, salts and acids (Daniel *et al.*, 2015; Hughes *et al.*, 2016). Such pollutants induce direct local mortality, impaired individual development and health, and altered community structure (Muturi *et al.*, 2017), particularly for predators through bioaccumulation (Carpenter *et al.*, 2011). Since nutrient loadings progressively lead to increased eutrophication, oxygen depletion, animal mortality, extirpation of submerged macrophytes and the production of algal blooms (including toxic varieties of cyanobacteria) (Foley *et al.*, 2005; Paerl & Paul, 2012), efforts to wastewater treatment related to all anthropogenic activities will need to increase. Pollutants affect in particular the biodiversity and functioning of closed systems and estuaries (Lotze *et al.*, 2006). For example, urban point sources have been the leading cause of hypoxia across European lakes since 1850 (Jenny *et al.*, 2016). Furthermore, continued deforestation, a key component of land-use change, will further disrupt organic matter processing and food webs, exacerbating the establishment and spread of pests and pathogens, especially in tropical regions (Morris *et al.*, 2016).

Future scenarios of changes in cropland area, pasture, forest and other natural land diverge widely depending on the underlying socio-economic assumptions (see sections 4.1 and 4.2.4) (Alexander *et al.*, 2017c; Popp *et al.*, 2017; van Vuuren *et al.*, 2011). For the RCP4.5 scenario, a decrease of cropland and pasture was projected in one study (van Vuuren *et al.*, 2011), which is expected to minimize future freshwater biodiversity disturbances. However, the global scenarios mask regional dissimilarities. For example, projections of future primary vegetation show major decreases in western and middle Asia (RCPs 2.6, 6.0 and 8.5), Australia (only RCP2.6) and North America (only RCP 8.5) (Settele *et al.*, 2014).

Water pollution has been considerably reduced in Australia, North America and Western Europe (Vörösmarty *et al.*, 2010), except for pharmaceuticals, biocides and plastics because of ineffective treatment (Ebele *et al.*, 2017). Reduced water pollution will benefit freshwater biodiversity. However, Sinha *et al.* (2017) projected increased eutrophication induced by increased precipitation from climate change in some regions, and Oliver *et al.* (2017) projected no decrease in nitrogen and phosphorus concentrations for most USA lakes despite attempts to reduce diffuse pollution. If there is little technology transfer to developing countries, then water pollution may increasingly threaten freshwater ecosystems, particularly in tropical regions because of increased human density notably in Asia and Africa, that are expected to account for over half of global population growth between 2015 and 2050 (UNDESA, 2015). Under RCP2.6, if much agricultural, mineral and bioenergy production relocates from high-income to low-income regions, pollution, freshwater biodiversity and aquatic ecosystem functioning will further worsen in those regions.

4.2.3.4 Future impacts of habitat fragmentation on freshwater biodiversity and ecosystem functioning

Hydropower is expected to increase worldwide whatever the RCP scenario unless other renewable energy sources are installed. Regions where significant losses in streamflow and decreased capacity production are projected, or where human population is expected to continue to increase (such as in many countries of Africa), should be most affected. Fragmentation of rivers by dams increases species extinction risks by blocking spawning/rearing migrations and/or reducing population sizes and gene flow.

Hydropower infrastructures alter rivers, floodplain lakes, wetlands and estuaries. Dams transform river basins by creating artificial lakes locally, fragmenting river networks, and greatly distorting natural patterns of sediment transport and seasonal variations in water temperatures and flows (Latrubesse *et al.*, 2017). Altered flow seasonality in rivers has led to less diverse fish assemblages, decreased inland fisheries production, less stable bird populations and lower riparian forest production (Jardine *et al.*, 2015; Kingsford *et al.*, 2017; Sabo *et al.*, 2017). Sediment retention by dams leads to delta recession (Luo *et al.*, 2017), decreased coastal fisheries catches, and degraded tropical mangrove forests that are major carbon sinks (Atwood *et al.*, 2017).

Dams also prevent upstream-downstream movement of freshwater animals, facilitate settlement of non-native species, cause local species extirpations and replacements and increase risk of water-borne diseases in reservoirs and highly altered environments by modifying productivity (Fenwick, 2006; LeRoy Poff & Schmidt, 2016). Dams have also caused a significant displacement of IPLCs around the world and projected expansion of dams, as shown in **Figure 4.2.13**, suggest significant overlap with areas held and/or managed by IPLCs (Garnett *et al.*, 2018). The fragmentation of river corridors also reduces population sizes and gene flows of aquatic species, increasing species extinction risks (Cohen *et al.*, 2016; Dias *et al.*, 2017). Dams are mainly concentrated in highly industrialized regions, but future hydropower development will be concentrated in developing countries and emerging economies (Grill *et al.*, 2015; Zarfl *et al.*, 2015). Hydropower is expected to expand worldwide whatever the RCP scenario **(Figure 4.2.13)**. Most hydropower plants are currently situated in regions where considerable declines in streamflow are projected, resulting in mean reductions in usable hydropower capacity (Turner *et al.*, 2017; van Vliet *et al.*, 2016b). Those regions may increase dam building to compensate for the losses unless other energy options are implemented (Zarfl *et al.*, 2015). Also, growing population density is expected to also increase demands for hydropower globally, especially in tropical regions (Winemiller *et al.*, 2016) where freshwater biodiversity is concentrated (Tisseuil *et al.*, 2013; UNDP, 2016).

4.2.3.5 Future impacts of non-native species on freshwater biodiversity and functioning

Future threats to freshwater ecosystems from non-native species will be greater in emerging economies because of accelerated economic growth, whatever the scenario. Non-native species often compete with and prey upon native species, generating occasional local population extirpations (Carpenter *et al.*, 2011), altering ecosystem structure and function (e.g., Blanchet *et al.*, 2010; Toussaint *et al.*, 2018), spreading infectious diseases (Gagne *et al.*, 2018) and sometimes degrading ecosystem services and economies (Leung *et al.*, 2002). They are a key contributor to biotic homogenization of aquatic ecosystems globally (Rahel, 2007; Villeger *et al.*, 2011). Anthropogenic disturbances coupled with introductions of non-native fish (particularly piscivores) are associated with native species extirpations and range reductions, especially in lakes and reservoirs (Whittier & Kincaid, 1999), as well as rivers (Hughes & Herlihy, 2012). In addition, reduced ecosystem services, particularly water quality, are likely to deteriorate as a result. Although policies have been implemented to prevent new introductions globally (McGeoch *et al.*, 2010 see chapter 6), the increase in the numbers of non-native species shows no sign of saturation over time. Also, many non-native species are predicted to spread worldwide in the next decades, mainly because of climate change, accelerated economic exchanges among countries, construction of new transportation corridors and increased aquaculture (Seebens *et al.*, 2017). These projections seem to occur in all RCP scenarios but especially so under the RCP 4.5, 6.0 and 8.5.

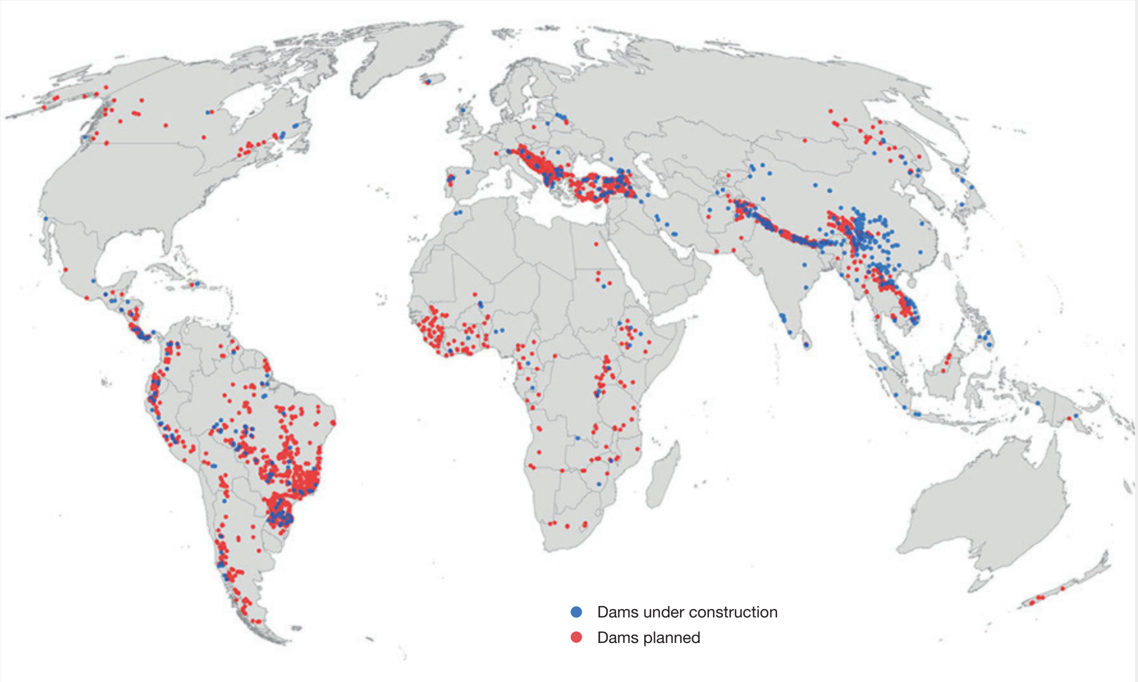


Figure 4 2 13 **Distribution of future hydropower dams, either under construction (blue dots 17%) or planned (red dots 83%).**

Source: Zarfl *et al.* (2015).

4.2.3.6 Future impacts of harvest on freshwater biodiversity and functioning

Irrespective of the exact type of scenario, given that human population density is continuously growing, increased harvesting is expected. Tropical ecosystems are of greatest concern. Intensive harvesting will deplete large-bodied fishes with consequent shifts toward harvests of smaller species and younger individuals with potential top-down effects on food web dynamics.

Current estimates of inland fisheries harvest are greatly underestimated (Deines *et al.*, 2017), but inland fisheries provide food for billions and livelihood for millions of people worldwide (FAO, 2016), and will continue to do so especially in developing countries. Low-income food-deficient countries account for ~80% of the total reported harvest from inland capture fisheries (Lynch *et al.*, 2016). Most global harvesting is concentrated in 16 countries, which have annual inland catches >200,000 tons and together represent 80% of the world total (FAO, 2016). Asian countries represent 63% of global total catches and African nations >13%. Harvests in African and Asian water bodies are already declining, probably because of environmental degradation and overexploitation (FAO, 2016). Given expected human population increases in Africa and Asia, increased harvesting is expected in both continents, whatever the RCP scenario. Because harvesting decreases population densities and large-bodied species, increased fishing pressure will lead to local extirpations of these species and will alter community structure and food web dynamics (Allan *et al.*, 2005; McIntyre *et al.*, 2016). These effects will be magnified by interactions with the other anthropogenic stressors listed above, including climate change. Because contributions of inland fisheries to economic security are inversely proportional to development level, rural economies in developing countries will be most affected.

4.2.3.7 Future impacts on peatlands

Peatlands are important flor global carbon cycling projections because they account for about one-third of the total carbon stored in soil organic matter (Page *et al.*, 2011) and also because many peatlands are an important source of methane (CH_4) (Kirschke *et al.*, 2013; Saunois *et al.*, 2016). Peatlands are threatened by future agriculture, forestry, peat extraction and dam construction activities (Minayeva *et al.*, 2017), which already over recent decades have begun transforming peatlands from greenhouse gas sinks to sources (Frolking *et al.*, 2011; Strack, 2008). For example, 15% of global peatlands have been drained worldwide and these drained peatlands are currently responsible for ~5% of all global anthropogenic CO_2 emissions (Strack, 2008).

While some regions appear to be improving peatland protection, others are increasing peatland destruction (Giam *et al.*, 2012; Hooijer *et al.*, 2010; Jauhiainen *et al.*, 2012; Koh *et al.*, 2011). Climate change is projected to possibly amplify shifts of peatlands from GHG sinks to sources, especially in regions where water tables are highly sensitive to local precipitation and where permafrost is melting (Dargie *et al.*, 2017; Turetsky *et al.*, 2015). A model intercomparison experiment showed that both peatland area and CH_4 emissions were less sensitive to potential future changes in precipitation than to increases in either atmospheric CO_2 or temperature (Melton *et al.*, 2013), but models disagree widely in both the magnitude and sign of potential climate effects on peatlands.

Where demands for water, food and energy put increasing pressure on the land resources, it is likely that peatland area will continue to decline (http://luh.umd.edu). Consequently, CO_2 emissions from peat decomposition and oxidation will expectedly persist well beyond the 21st century. Tropical regions are projected to be most affected under scenarios where much agriculture and bioenergy production relocate from high-income to low-income regions (Lawrence *et al.*, 2016). Considering the over proportional warming projected for subarctic and arctic ecosystems and the large amount of carbon stored in peatlands on permafrost soils, large climate warming feedbacks have been projected (Koven *et al.*, 2011; Page & Baird, 2016).

While plant and animal taxonomic diversity in peatland ecosystems is apparently low, highly specialized species predominate, with 5–25% of peatland plant species being endemic (Minayeva *et al.*, 2017). Many animal species occupy peatlands only at certain life stages or during particular seasons (but see Giam *et al.*, 2012 for some narrowly adapted fish species). Because of their unique flora, projected lost peatland area has implications for global biodiversity. In all scenarios, and without peatland conservation practices, climate change and other anthropogenic drivers are expected to disrupt peatland biodiversity to varying degrees, ranging from decreased population sizes to altered species composition and regional or global extinctions (Fraixedas *et al.*, 2017; Giam *et al.*, 2012; Hedwall *et al.*, 2017). For example, in Southeast Asia, if current rates of peatland conversions to agriculture continue through 2050, several fish species will become globally extinct (Giam *et al.*, 2012).

4.2.4 Terrestrial ecosystems

4.2.4.1 Future climate change and atmospheric CO_2 impacts on habitats, biodiversity, and ecosystem state and functioning

4.2.4.1.1 Climate change impacts on vegetation cover

Global vegetation and Earth system models all project substantial climate change driven shifts of natural vegetation cover over the next century (Davies-Barnard *et al.*, 2015; Gonzalez *et al.*, 2010; Ostberg *et al.*, 2013; Pereira *et al.*, 2010; Reu *et al.*, 2014; Sitch *et al.*, 2008; Wårlind *et al.*, 2014; Warszawski *et al.*, 2013). Area losses of natural vegetation are estimated to be 2-47% of terrestrial ecosystems for even relatively small temperature increases (<2°C above pre-industrial; Warren *et al.* (2011), and references therein). Other analyses confirm the risk of changes in vegetation cover (e.g., forest to non-forest or vice versa) for relatively small global temperature increases, especially in tundra, tropical forest and savanna regions but with changes within a given biome likely to occur in all regions (Gonzalez *et al.*, 2010; IPCC, 2018, Chapter 3.4.3; Ostberg *et al.*, 2013; Scholze *et al.*, 2006; Warszawski *et al.*, 2013). Biome shifts and associated impacts on ecosystem functioning increase notably in higher-warming scenarios (Ostberg *et al.*, 2013; Scholze *et al.*, 2006; Warren *et al.*, 2011; Warszawski *et al.*, 2013). Enhanced tree mortality from wildfires and increased drought and heatwaves can amplify vegetation responses to climate in models (Allen *et al.*, 2010; Lasslop *et al.*, 2016; Tietjen *et al.*, 2017).

4.2.4.1.2 Climate change impacts on species diversity

In principle, climatic changes could be favourable to some species in cases when a new climate can provide more resources for species growth, reproduction and distribution (Bellard *et al.*, 2012). However, even by the middle of the 21st century, or for relatively minor temperature changes, indices for animal and plant species richness have been projected to decline, and indices of species losses, enhanced (Alkemade *et al.*, 2013, 2009; Bellard *et al.*, 2012; Gonzalez *et al.*, 2010; IPCC, 2018, Chapter 3.4.3; Pereira *et al.*, 2010; Settele *et al.*, 2014; Warren *et al.*, 2011). Climate change has also been identified as a major driver of terrestrial species loss across all IPBES regional assessments (Bustamante *et al.*, 2018; Elbakidze *et al.*, 2018; Nyingi *et al.*, 2018; Wu *et al.*, 2018). A recent meta-analysis of studies reported that a global mean temperature increase of 2°C would threaten one in 20 species (for 5.2% of species, the distributional range falls below a minimum threshold), increasing to one in 12 and one in 6 species for 3°C and 4.3°C, respectively (Urban, 2015). Model projections across a range of scenarios show regionally highly variable extinction risks for terrestrial species on average between ca. 5-7% (Europe, Northern America) to ca. 25% (South America), ca. 9% in the tropics, and ca. 5% in temperate, polar and boreal environments, by 2100 (Maclean & Wilson, 2011; Urban, 2015). The projected extinction risk increases strongly with degree of global warming (Urban, 2015). Large uncertainties exist: for instance, extinction risks estimates when based on extrapolation of past observed trends have been found to be higher than the estimates based on model projections (Maclean & Wilson, 2011).

Climate change will impact biodiversity hotspots. Two contrasting future scenarios at the end of the 21st century have been estimated to negatively influence 25% of endemic species on average per hotspot, with largest effects in low latitudes, island locations and in Mediterranean type climates (Bellard *et al.*, 2014). Nearly all of the 143 investigated terrestrial regions in the Global 200 list of ecoregions that have been identified to support maintaining a broad diversity of Earth's ecosystems, will likely experience by the end of the 21st century moderate-to-pronounced climate change impacts, across a range of climate change scenarios (Li *et al.*, 2013).

Since the magnitude but also the velocity of climate change are chief determinants of whether (and which) terrestrial animal or plant species will be able to follow shifting habitats (Foden *et al.*, 2013; Gonzalez *et al.*, 2010; Keenan, 2015; Loarie *et al.*, 2009; Pecl *et al.*, 2017; Pereira *et al.*, 2010), the combination of abiotic and biotic characteristics that have not been observed in the past might be increasingly common in the future (Murcia *et al.*, 2014; Ordonez *et al.*, 2016; Radeloff *et al.*, 2015). Projected future changes in species ranges, species extinctions and community diversity therefore may be under– or overestimated by models that do not explicitly account for species interactions such that loss (or gain) of one species would trigger loss (or gain) for others (Bellard *et al.*, 2012; Schleuning *et al.*, 2016). As a consequence, new approaches to conservation are warranted that are designed to adapt to rapid changes in species composition and ensuing conservation challenges.

4.2.4.1.3 The combined impact of atmospheric CO_2 concentration and climate change on projected vegetation cover

Increasing atmospheric CO_2, the chief driver of climate change, also enhances relative competitiveness of plants of the C3 photosynthetic pathway by fostering carboxylation reactions in the leaf and allowing plants to operate at reduced stomatal conductance (Higgins & Scheiter, 2012; Pugh *et al.*, 2016b; Walker *et al.*, 2015). Whether or not enhanced photosynthesis or enhanced water use efficiency translates also into enhanced plant growth is not yet unequivocally established (Higgins & Scheiter, 2012; Pugh *et al.*, 2016b; Walker *et al.*, 2015). Globally, increased forest cover over the 21st century has been projected across a range of scenarios (Davies-Barnard *et al.*, 2015; Reu *et al.*, 2014; Sitch *et al.*, 2008; Wårlind *et al.*, 2014). Typically, forest cover increases in northern latitudes (Davies-Barnard *et al.*, 2015; Reu *et al.*, 2014; Sitch *et al.*, 2008; Wårlind *et al.*, 2014). A shift from grass- to increasingly woody-dominated vegetation (see Nyingi *et al.*, 2018) is simulated in semi-arid regions (Knorr *et al.*, 2016; Lehmann *et al.*, 2014; Lehsten *et al.*, 2009; Moncrieff *et al.*, 2014, 2016; Scheiter *et al.*, 2015). Impacts of enhanced CO_2 on canopy structure and combustible biomass alter fire regimes, with complex ecosystem feedbacks (Harris *et al.*, 2016; Jiang *et al.*, 2017; Kim *et al.*, 2017; Knorr *et al.*, 2016; Loudermilk *et al.*, 2013; Turco *et al.*, 2014; Wu *et al.*, 2015). Large-scale forest "die-back" emerges only in relatively few simulation experiments that examined future climate change and CO_2 impacts in tropical forest regions, especially the Amazon (Aragão *et al.*, 2014; Duran & Gianoli, 2013; Gumpenberger *et al.*, 2010; Malhi *et al.*, 2009, 2008; Nobre *et al.*, 2016; Poulter *et al.*, 2010; Rammig *et al.*, 2010; Schnitzer & Bongers, 2011). These model outcomes are supported by analyses that attributed the observed greening trends in many regions and (C3) shrub encroachment in C4-dominated grasslands chiefly to CO_2 fertilisation effects (Donohue *et al.*, 2013; Schimel *et al.*, 2015; Stevens *et al.*, 2016; Zhu *et al.*, 2016). Increases in woody vegetation in grass-dominated regions are expected to negatively impact grassland-related biodiversity (Barbosa da Silva *et al.*, 2016) but intermediate levels of woody cover might in some cases be beneficial for ecosystem functioning such as carbon storage, reduction of soil erosion and overall plant and animal species diversity (Barbosa da Silva *et al.*, 2016; Eldridge & Soliveres, 2014; Soliveres *et al.*, 2014).

4.2.4.1.4 Projected changes in ecosystem state and function

The uptake of CO_2 in land ecosystems is large, with 20-25% of anthropogenic emissions being removed from the atmosphere each year (Le Quéré *et al.*, 2018; see also Chapter 2.2, section 2.2.5.2.2). The future persistence of this land carbon "sink" is one of the largest uncertainties in climate research. It is important to address because of the potentially large warming feedback associated with a loss of the land sink (Arneth *et al.*, 2010; Ciais *et al.*, 2013). The direction (but not the magnitude) of the change in global terrestrial carbon uptake and pool sizes in response to climate change alone vs. increased CO_2 concentration alone is modelled relatively robustly (Ciais *et al.*, 2013; Hajima *et al.*, 2014; Nishina *et al.*, 2015; Sitch *et al.*, 2008; Walker *et al.*, 2015; Zaehle, 2013). However, when effects of climate change and CO_2 concentration are considered jointly, the rate and even the sign of change in simulated trajectories of future ecosystem C pools and related fluxes are highly inconsistent between ecosystem carbon cycle models (Ciais *et al.*, 2013; Eglin *et al.*, 2010; Friend *et al.*, 2014; Nishina *et al.*, 2015; Piao *et al.*, 2013; Sitch *et al.*, 2008). The latest IPCC report places low confidence on how stocks and fluxes will evolve over the coming decades (Ciais *et al.*, 2013).

Evapotranspiration (ET) from ecosystems is greatly altered by changes in leaf area, functional vegetation type, precipitation and atmospheric dryness, and the response of stomatal conductance to CO_2. Whether or not global or regional run-off (which affects availability of water for irrigation but also floods) will increase in the future due to enhanced water cycles in a warmer climate, or possibly reduced ET in a higher CO_2 world is unresolved. Similar to projections of ecosystem productivity and carbon balance, uncertainty arises from both variability in climate change projections and from process descriptions in impact models (Döll & Schmied, 2012; Piao *et al.*, 2007; Zhang *et al.*, 2014).

Overall, climate change, and change in atmospheric CO_2 levels will strongly impact productivity and other important ecosystem processes, vegetation cover, and habitat structure over the next decades, with the relative importance of these drivers differing between biomes/regions (see **Figure 4.2.2** and Table A4.2.1).

4.2.4.2 Future land-use and land-cover change impacts on habitats, biodiversity, and ecosystem state and functioning

Nearly 40% of the land surface today is used as croplands or pastures, and humans have transformed the vegetation structure and species composition in an area far greater still (Ellis, 2013; Ellis *et al.*, 2012; see also Chapters 2.1 and 2.2). Local within-sample richness, rarefaction-based species richness, and total abundance have all been shown to be generally lower in areas under different types and intensity of land use, compared with natural vegetation (Alkemade *et al.*, 2009; Newbold *et al.*, 2015; Wilting *et al.*, 2017; Chapter 2.2.). In some cases, species richness, at least for plants, can also increase under land use, such as documented in local management systems for agriculture and agroforestry, forests, meadows and grasslands found around the world (Ellis *et al.*, 2012; Gerstner *et al.*, 2014; see also Chapter 2.2). Both, changes in land cover and land use, are known to impact biodiversity and ecosystem functioning globally (Foley *et al.*, 2011; Kleijn *et al.*, 2009; Pywell *et al.*, 2012). But across large scales, studies typically assess impacts of land cover changes, rather than intensification of management at a given area of land which limits our ability to understand the combined effect of land-use and land-cover change (de Chazal & Rounsevell, 2009; Titeux *et al.*, 2017).

Humid or mesic savannas and woodlands seem particularly vulnerable to future conversion of natural vegetation into cropland or pasture, because of their climate suitability for agriculture. Land-use changes have been very pronounced in recent decade; for example, in the Cerrado or Chaco regions of South America, but also in African savannas (Aleman *et al.*, 2017, 2016; Cavender-Bares *et al.*, 2018; Nyingi *et al.*, 2018; Searchinger *et al.*, 2015; see also Chapter 2.1).

Land conversion pressure is large both in scenarios that explore high population growth and lack of consideration for sustainable development (e.g., lack of conservation efforts, little consumption change), as well as in strong mitigation scenarios that require land for bioenergy or afforestation (Popp *et al.*, 2017; see also section 4.2.4.3). Due to large land area requirements, maintaining or enhancing biodiversity and ecosystem functionality (such as productivity and changes in carbon pools or changes in water cycling) would be challenging under such socio-economic projections (Krause *et al.*, 2017, 2018; Popp *et al.*, 2017; Ryan *et al.*, 2016; Searchinger *et al.*, 2015).

Projections of future biodiversity at the global level have until recently been biased towards climate change related questions (Titeux *et al.*, 2016, 2017). Anthropogenic land-cover changes have been relatively well studied at the regional and local levels, particularly but not only in tropical forests regions, but are only slowly beginning to be considered in global scenario projections. Declining forest cover and/or reduced average local species richness, for 2050 and until the end of the 21st century have been found under “economic optimism” scenarios, such as the SSP5/RCP8.5 which projects large greenhouse gas emissions and climate change effects along with substantial expansion of cropland or pastures (Davies-Barnard *et al.*, 2015; Newbold *et al.*, 2015), or under scenarios that assume the absence of a REDD scheme (Strassburg *et al.*, 2012). Interactions of future climate change with land-cover change were shown to enhance risk of biodiversity loss by up to 43% for birds and 24% for mammals, compared to land-cover change impacts only (Mantyka-Pringle *et al.*, 2015). By 2050 in a business-as-usual scenario, climate and land-cover change were shown to lead to a decline in mean terrestrial carnivore and ungulate population abundance by 18-35%, and to an increase in extinction risk for 8-23% of species (Visconti *et al.*, 2016). Negative impacts are also projected to arise from land-cover and land-use changes on a range of threatened carnivores in an OECD Environment Outlook scenario (Di Minin *et al.*, 2016). Taken together these studies demonstrate that across a range of scenarios, expansion of managed land is projected to pose additional pressure on biodiversity. The relative impacts of climate change versus land-use change on biodiversity, however, are context-specific and vary between scenarios and regions, and depend on the biodiversity indicator or facet of biodiversity under scrutiny, as emphasised by the four regional IPBES assessments (e.g., Bustamante *et al.*, 2018; Elbakidze *et al.*, 2018; Nyingi *et al.*, 2018; Wu *et al.*, 2018) and also by very recent results emerging from the BES-SIM study (Kim *et al.*, 2018; **Box 4.2.5**; see also section 4.1).

Future anthropogenic land-cover change will also impact protected areas and the associated protected species range (see section 4.6). Even when implemented efficiently, the percentage area protected would have to increase to capture a similar rage of terrestrial vertebrate species range in simulations that include projections of land cover change over the next two decades, compared with land-cover change remaining at present-day levels (Montesino Pouzols *et al.*, 2014).

Box ④ ② ⑤ Biodiversity and nature’s contributions to people in the Shared Socio-economic Pathway scenarios: a model inter-comparison.

Background. In 2016, IPBES created a task force to support the scientific community in developing scenarios and models to provide IPBES and other stakeholders with greatly improved capacity to assess the future impacts of global environmental change on biodiversity and nature’s contributions to people (IPBES, 2016b; Rosa *et al.*, 2017). This work focuses on two complementary tasks. The first task is to work closely with the climate change community to analyze and extend the ‘Shared Socio-economic Pathways (SSP)’ scenarios and associated climate change projections that have been developed in support of the IPCC (Rosa *et al.*, 2017). The results presented below are the first outcomes from this task referred to as BES-SIM (Kim *et al.*, 2018). The second task is to develop a set of multi-scale, participatory based scenarios that explicitly account for nature conservation objectives. This task is ongoing, and the outcomes will only become available for future assessments.

The results presented below are from the first-ever comparison of multiple models of terrestrial biodiversity, ecosystem functioning and ecosystem services at the global scale using a common set of inputs for climate and land-use change drivers (Kim *et al.*, 2018), addressing shortcomings in previous comparative attempts that have been hampered by the lack of a common methodology (Bellard *et al.*, 2012; Pereira *et al.*, 2010; Settele *et al.*, 2014; Urban, 2015; Warren *et al.*, 2011). Using a total of 14 participating models, ten different indicators of biodiversity were simulated and six models contributed simulations of ecosystem function and ecosystem services (Kim *et al.*, 2018).

All models of biodiversity, ecosystem function and ecosystem services used harmonized land-use inputs from three SSP scenarios in combination with three scenarios of greenhouse gas emissions (RCP) and corresponding projected climate change (Kim *et al.*, 2018):

- SSP1 x RCP2.6 – is a 'global sustainability' scenario archetype (SSP1) combined with low GHG emissions (RCP2.6),
- SSP3 x RCP6.0 – is a 'regional competition' scenario archetype (SSP3) combined with high GHG emissions (RCP6.0), and
- SSP5 x RCP8.5 – is an 'economic optimism' scenario (SSP5) combined with very high GHG emissions (RCP8.5).

Climate and land-use change projections from these three sets of scenarios (see section 4.1.4, and Appendix A4.2.3) were evaluated for their consequences for biodiversity, ecosystem functions and ecosystem services. In addition, some of the participating models evaluated the impacts of climate change and land-use change individually, as well as in combination. Outputs from ecosystem functioning and ecosystem services models have been grouped into categories of nature's contributions to people as defined in Diaz *et al.* (2018).

Biodiversity and regulating nature's contributions to people are projected to decline while material contributions to people increase by 2050. The global average of projected impacts on biodiversity and on nature's contributions to people are shown in **Figure 4.2.14**. The combined impacts of climate and land-use change on biodiversity include large declines in local species richness, increases in regional to global scale species extinction and declines in biodiversity intactness. Several important regulating ecosystem services, such as coastal protection, soil erosion protection and crop pollination, are projected to decline in the 'regional competition (SSP3xRCP6.0)' and 'economic optimism (SSP5xRCP8.5)' scenarios.

In contrast, food, feed, timber and bioenergy production services are projected to substantially increase in these scenarios. This pattern of trade-offs between declining biodiversity and regulating contributions on one hand vs. increasing material contributions on the other hand are coherent with recent patterns (Carpenter *et al.*, 2009; see Chapters 2 and 3) and with a wide range of studies of biodiversity and ecosystem services evaluated in this chapter (sections 4.3 and 4.5).

Not all of the metrics follow this general pattern. One important example is ecosystem carbon storage at the global scale, which is an indicator of the capacity of ecosystems to contribute to climate change mitigation. Global scale ecosystem carbon storage is projected to be stable or increase in nearly all scenarios and in all ecosystem models by 2050 (see Table A4.2.2 in Appendix A4.2.3). This occurs in part because rising atmospheric CO_2 concentrations and rising temperatures (up to certain point) stimulate modeled plant productivity and ecosystem carbon storage, as well as the result of land-use change in the scenarios.

There are large regional differences in the patterns of biodiversity loss and changes in nature's contributions to people with the largest projected impacts in the global south **(Figure 4.2.15)**. The projected effects of land use and climate change on three metrics of biodiversity, material nature's contributions to people and regulating nature's contributions to people for the IPBES subregions are shown in Figure A4.2.1 in Appendix A4.2.3. The general patterns at the global level – i.e., declines in biodiversity and regulation contributions vs. increases in material contributions – are evident in nearly all subregions. Biodiversity in South America, Africa and Asia (with the exception of northeast Asia) is much more heavily impacted than in other regions, especially in the regional competition and economic optimism scenarios. Ecosystem carbon storage shows particularly contrasted regional responses, with very large declines projected for Africa. These regional differences occur in part because scenarios foresee the largest land-use conversions to crops or bioenergy in these regions (see section 4.1.5 and Appendix A4.1.2). Other regions such as North America and Europe are foreseen to have low conversion to crops and continued trends of afforestation which minimizes declines in biodiversity, or even increases in some regional biodiversity metrics. Regional differences in climate change impacts also play a major, and sometimes dominant role in regional contrasts.

The magnitude of impacts and the differences between regions are much greater in scenarios of regional competition and economic optimism than in a scenario of global sustainability. Biodiversity loss at the global scale is much lower in the global sustainability scenario (SSP1xRCP2.6) than in the regional competition and economic optimism scenarios and even improves for the biodiversity intactness metric. Several regulating services, such as crop pollination and soil protection, increase at the global scale in the global sustainability scenario instead of declining as in the other two scenarios, and in general, the impacts of land use and climate change are much greater in the regional competition and economic optimism scenarios **(Figure 4.2.14)**. In contrast, the global sustainability scenario results in substantially lower projected food, feed and timber production, but it is important to note that this arises primarily from lower demand rather than insufficient supply of food and timber to people. The regional competition and economic optimism scenarios also are projected to generate much greater regional contrasts in biodiversity and nature's contributions than the global sustainability scenario **(Figure 4.2.15)**. But caution should be exercised when generalizing from these three scenarios because there is substantial variation in land use and other drivers within each of the main Shared Socio-economic Pathway classes (Popp *et al.*, 2017).

Figure ④ ② ⑭ **Global means of projected percent changes in biodiversity Ⓐ and nature's contributions to people Ⓑ between 2015 and 2050.**

Biodiversity metrics include changes in local species richness (= number of species in a small area), regional species richness (number of species at regional or global scales, the opposite of which is regional or global extinction), and biodiversity intactness (i.e., abundance of plant and animal communities in disturbed compared to undisturbed natural ecosystems). Values are averages across models, which number is indicated by N. Standard errors across models are indicated by whiskers when more than one model projection was available. The three SSPxRCP scenarios are defined in the box text.

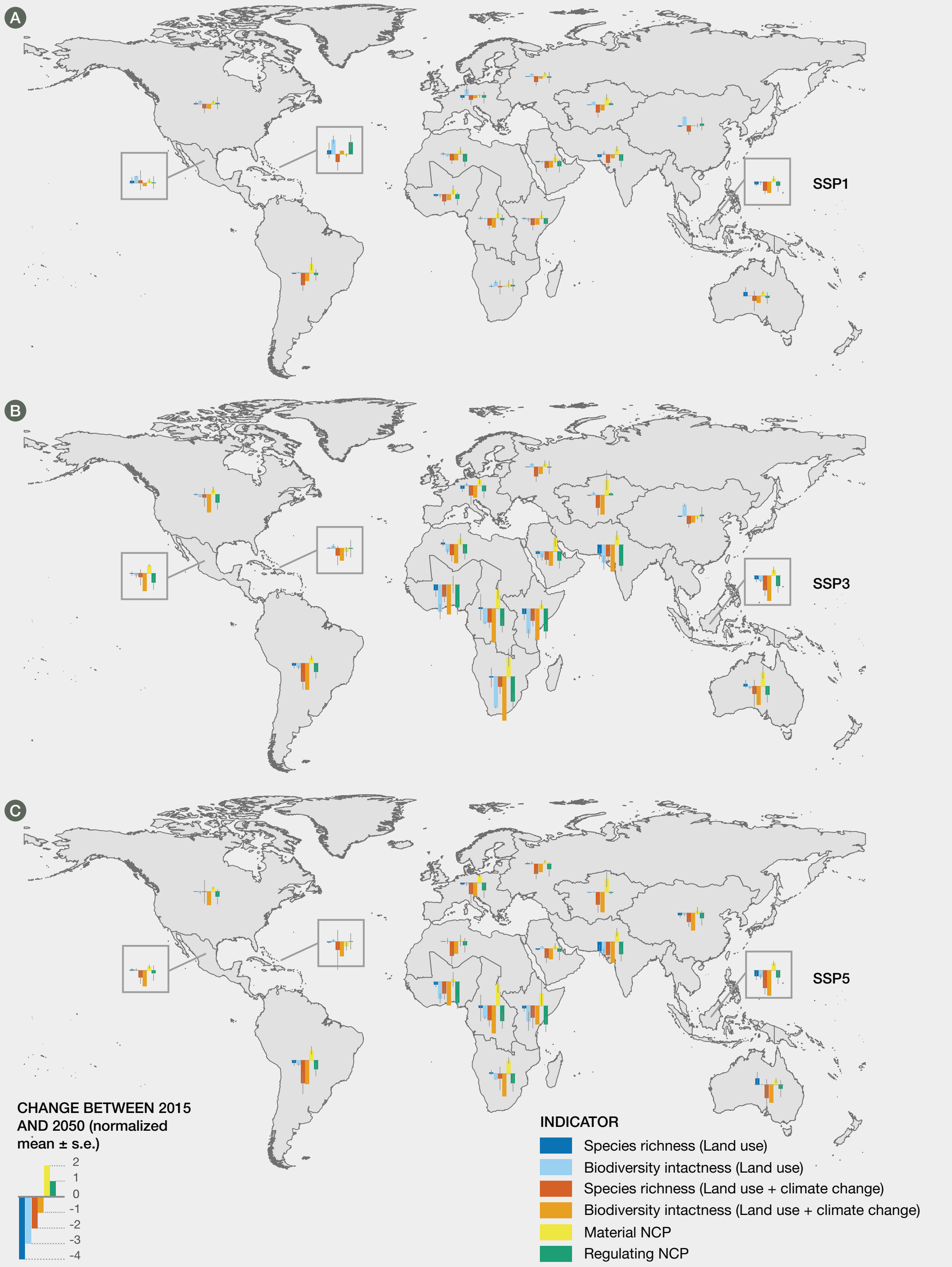


Figure ④ ② ⑮ **Projected changes in biodiversity and nature's contributions to people for the IPBES subregions for a 'global sustainability' scenario (SSP1), a 'regional competition' scenario (SSP3) and an 'economic optimism' scenario (SSP5) between 2015 and 2050.**

To allow for direct comparison across scenarios and subregions, absolute mean values of change have been standardized by dividing the individual value of a metric by the standard deviation of all the values of that metric for all subregions in all scenarios (see Appendix A4.2.3 for details).

The projected impacts of climate change on biodiversity are much greater than land-use change in this study, but there is large uncertainty in this result. There is considerable debate concerning the relative sensitivity of species response to land use vs. climate change (Bellard *et al.*, 2012; IPBES, 2018g, 2018j, 2018h, 2018i; Pereira *et al.*, 2010). This multi-model study suggests that climate change will dominate biodiversity responses as early as 2050 for all biodiversity metrics, but this outcome needs to be treated with considerable caution for several reasons including i) very high uncertainty in models of climate change impacts on biodiversity (see error bars in **Figure 4.2.14**, and Settele *et al.*, 2014 for a discussion of uncertainties), ii) there are small differences in projected land-use change across the three scenarios compared to the range in a wider set of plausible futures (Alexander *et al.*, 2017c; Pereira *et al.*, 2010; but see Popp *et al.*, 2010) showing that the three scenarios used here cover nearly the full spectrum of land-use change in the SSP scenarios set), iii) issues related to defining land-use classes and using a very small set of land-use classes and iv) optimistic assumptions about food production increases that contribute to relatively small land-use changes while neglecting impacts of agricultural intensification (see drivers section 4.1.4).

There are high levels of uncertainty associated with these projected impacts, as is the case in other studies. There are a number of general and specific limitations to the BES-SIM results. Generally, the models used to foresee future land-use change, as well as the models of climate change impacts on biodiversity and most ecosystem services have not been well evaluated with data (Alexander *et al.*, 2017c; Ferrier *et al.*, 2016; Settele *et al.*, 2014; van Vliet *et al.*, 2016a). In addition, all models have intrinsic limitations due to underlying hypotheses and simplifications (Ferrier *et al.*, 2016). For example, none of the models of species response to climate change used in the BES-SIM study explicitly accounts for the capacity of organisms to adapt to climate change, or for species-interactions (Kim *et al.*, 2018). Model outputs have been grouped into categories of metrics, but these groupings mask important differences in interpretation of metrics from the various models (Kim *et al.*, 2018). For example, interpretation of ecosystem service indicators is challenging because they are expressed in very different units. Nevertheless, besides constituting the first comparison of a broad range of models using a common set of climate and land-use scenarios, one of the benefits of the BES-SIM study was to help to quantify some of the components of uncertainty, and while the difference between models was large for all metrics **(Figure 4.2.14)**, the overall qualitative trends were similar.

Likewise, future land-cover change scenarios and different spatial patterns that have been projected for each of the four RCPs will affect buffer zones that surround existing protected areas (Beaumont & Duursma, 2012). In most biomes modelled in this study (Beaumont & Duursma, 2012), previously unused land in buffer areas is projected to decline considerably by 2050 and more so by 2100. The projected decline in local species richness might be similar for low and high emissions scenarios, if the low emissions scenario necessitates large conversion of primary vegetation, for instance for bioenergy crops (RCP2.6; Newbold *et al.*, 2015). In contrast, a scenario focusing on globally sustainable resource use, consumption change, and associated habitat restoration indicated that both extinction risks and species losses would strongly be reduced over the next decades (Visconti *et al.*, 2016). Likewise, scenarios of increasing carbon prices as incentives to increase return from maintaining forested areas under a REDD mechanism drastically reduced local extinctions, especially in regions with high species richness (Strassburg *et al.*, 2012).

Estimates of impacts of land-use change on ecosystems and biodiversity need to consider urban areas and landscapes. Over the coming decades, some ecoregions and biodiversity hotspots will lose remaining undeveloped area through urban development, with localised large pressures on rare species and protected areas (Güneralp & Seto, 2013; McDonald *et al.*, 2008; Seto *et al.*, 2012). Nonetheless, a number of indicators of bird biodiversity differed little between urbanised and non-urbanised environments (Pautasso *et al.*, 2011). In Australia, some cities support a relatively larger number of threatened plant and animal species compared to non-urban landscapes (Ives *et al.*, 2016). With ongoing and future projected urbanisation of human societies, impacts of cities, larger urban areas and land transportation networks clearly must be included in scenarios of future biodiversity at different spatial scales.

Projected anthropogenic land-cover change and intensification of agriculture and pastures will enhance emissions of greenhouse gases. Future emissions of N_2O from terrestrial ecosystems in response to deposition and fertiliser use and climate change are projected to be enhanced by ca. 20% to threefold by the middle of the 21st century across a range of RCP (2.6, 8.5) and SRES scenarios (A1, B1, A2, B2) (Bodirsky *et al.*, 2012; Kanter *et al.*, 2016; Stocker *et al.*, 2014). Other gaseous forms of N losses (NOx and NH3) and their atmospheric reactions affect secondary organic aerosols, the lifetime of methane, or formation of tropospheric ozone (Bodirsky *et al.*, 2012; Butterbach-Bahl *et al.*, 2011; Kanter *et al.*, 2016; Lassaletta *et al.*, 2016; Zaehle *et al.*, 2015), and pollute waterways (section 4.2.3). On the other hand, land management practices in cropland, pastures and managed forests have been estimated to potentially contribute to emissions reductions by 1.5-4.8 Gt CO_2eq a-1 (Griscom *et al.*, 2017; Smith *et al.*, 2014a) achievable over few decades at carbon prices up to 100 $ US, without detrimental side effects on productivity, water use or biodiversity. This greenhouse emissions reduction potential might be tripled if food demand-side measures are also taken.

4.2.4.3 Future global ecosystem functioning and biodiversity in strong climate change mitigation scenarios

Land use is becoming increasingly central in future scenarios that target strong climate change mitigation (Popp *et al.*, 2017). Avoided deforestation (in conjunction with afforestation and reforestation, AR) is seen as one possible option (Angelsen, 2010; Chazdon *et al.*, 2016; Cunningham *et al.*, 2015; Smith & Torn, 2013; Strassburg *et al.*, 2012), which is also low-cost (Griscom *et al.*, 2017; Humpenoder *et al.*, 2014). Co-benefits of avoided deforestation for biodiversity (see **Figure 4.2.2**, Table A4.2.1 in Appendix 4.2) and local communities can be large, whereas the environmental impacts of large-scale afforestation and reforestation depend to a large degree on prior vegetation cover and the tree species planted for reforestation. Under the Paris COP21 climate agreement, forest-based climate mitigation targets feature prominently in several countries' Nationally Determined Contributions (Grassi *et al.*, 2017). Likewise, bioenergy in combination with carbon capture and storage (BECCS) has been put forward as a major land-based climate change mitigation approach in many scenarios that achieve a target of 2°C warming or below (Fuss *et al.*, 2016; see IPCC, 2018, Chapter 4.3.7; Popp *et al.*, 2014; Smith *et al.*, 2016). In Integrated Assessment Models (IAMs), the global cumulative C-uptake potential has been estimated to be ca. 55-190 GtC for avoided deforestation and AR at the end of the 21st century, and between ca. 125-250 GtC for BECCS (Humpenoder *et al.*, 2014; Tavoni & Socolow, 2013). Annual carbon uptake in 2050 for BECCS (1-2.2 GtC a^{-1}) and AR (0.1-1 GtC a^{-1}) is equivalent to up to one third to three quarters of today`s land carbon sink (IPCC, 2018, Chapter 4.3.7; Le Quéré *et al.*, 2018). In absence of carbon capture and storage, IAM projections may indicate even higher use of bioenergy (although it remains unclear how the required land area could be made available in an overall environmentally sustainable manner), unless the IAM scenarios are based on reduced energy consumptions and/or availability of cheap renewable energy, which reduces the need for land-related climate change mitigation (IPCC, 2018, Chapter 2.3). Analyses of ecosystem carbon uptake with dynamic global vegetation models (Fisher *et al.*, 2010) have arrived at consistently lower numbers than land-use models in IAMs when confronted with similar land-use change projections (Krause *et al.*, 2018). The reasons for the discrepancies in carbon uptake potential calculated with IAMs and DGVMs are not yet fully resolved. Indirect land-use changes complicate projections further. For instance, Popp *et al.* (2014) argued that stringent forest conservation policies could well lead to a spill-over effect such that land transformation for agriculture is shifted to other carbon-rich and biodiversity-rich ecosystems such as savannahs or temperate grasslands. Stringent climate change mitigation affects ecosystem productivity through bounded temperatures (and precipitation), but also via lower CO_2 in the atmosphere. Stabilizing or reducing the atmospheric concentration of CO_2 is expected to stabilize or reduce the fertilization effect of photosynthesis and is likely to also stabilize or reduce productivity compared to present-day levels (Jones *et al.*, 2016; Pugh *et al.*, 2016b).

Growth of bioenergy in simulation studies is in some cases restricted to marginal lands to avoid competing with food production, with the implicit assumption that these marginal lands would also be diversity-poor, which is not necessarily the case (Plieninger & Gaertner, 2011). The published studies mostly lack a clear definition and do not quantify the criteria used for classifying marginal or degraded land (de Jong *et al.*, 2011). Schueler *et al.* (2016) mapped the sustainability criteria, which include biodiversity protection, of the European Renewable Energy Directive to the global land area and found, for present-day environmental conditions, a potential for an additional bioenergy generation of around 80-90 EJ a-1 on ca. 430 Mha land. A large proportion of this land area is classified as low yielding (low productivity). Regions of high-yield potential that are currently under natural vegetation would be at risk for development unless protective sustainability measures are applied. In a stylised scenario experiment based on data for *Miscanthus* as a bioenergy crop species, half the potential for global bioenergy production was found to lie within the top 30% of land area classified of highest priority for biodiversity protection (Santangeli *et al.*, 2016). In a recent simulation of future land-use impacts on extinction risk of endemic species, and applying land-use change projections adopted from (Popp *et al.*, 2014), the RCP2.6-SSP1 scenario was identified as causing the least loss of natural vegetation cover by 2050 and the least extinctions of endemic mammals, birds and amphibians, compared with the – in this study – "worst case" RCP3.4-SSP4 (Chaudhary & Mooers, 2017). Climate change was not considered as an additional factor, which likely would have enhanced the projected biodiversity risk in the stronger climate change cases. The published literature overall suggests that only protective mechanisms that account for carbon storage potential and biodiversity at the same time could yield the intended carbon-mitigation objectives while avoiding degradation of diversity.

Uncertainties regarding impacts on biodiversity and ecosystems arising from different land-use change projections cannot be assessed yet. It was shown that structural differences (for instance, the type of economic model) that exist between different land-use change models can have a similarly large impact on future land-use change projections than the underlying socio-economic scenario (Alexander *et al.*, 2017c; Prestele *et al.*, 2016). However, only one Integrated Assessment Model provides the so-called marker scenario per RCP/SSP combination (Popp *et al.*, 2014; see **Box 4.2.5**). Without a larger set of harmonised historical to future land-use change projections for each of the RCP/SSP, from a wide range of different land-use change models, the degree to which impacts on biodiversity and ecosystem state and function are related to scenario archetypes remains unresolved.

4.2.4.4 Invasive alien species

Invasive alien species are a major driver of biodiversity loss today (see Chapter 2.2, section 2.2.5.2; see Bustamante *et al.*, 2018; Elbakidze *et al.*, 2018; Nyingi *et al.*, 2018; Wu *et al.*, 2018). Projections of invasive alien species all foresee continued substantial changes in biological invasion state and pressure with significant consequences for both biodiversity and human well-being. These projections have until recently been biased towards climate change related questions, but increasingly also consider how land use and trade patterns might affect future distribution of invasive alien species. Future changes of invasive alien species distributions are still uncertain, but several generalizations can be made from modelling work.

The pressure on biodiversity, and ecosystem function from biological invasions is expected to continue to grow in the coming decades in most parts of the world (Bellard *et al.*, 2013; Gallardo *et al.*, 2017; Hulme, 2009), as well as the economic damage caused by invasive alien species to society (Bradshaw *et al.*, 2016). Extrapolations of cumulative introduction events over Europe suggest that the number of invasive species will continue to increase (CBD, 2014; Elbakidze *et al.*, 2018). This trend is likely to be accentuated at a global scale, as trade between climatically and environmentally similar regions are predicted to increase and habitats continue to be disturbed (Chytrý *et al.*, 2012; Seebens *et al.*, 2015). For example, future hotspots of naturalized plants are predicted to occur mostly in North America, Australia, and South America, followed by Europe, South Africa and China (Seebens *et al.*, 2015). An analysis conducted on the IUCN "100 of the world's most invasive alien species" suggests future expansion of these species especially in cool temperate areas. The biomes with the highest expected expansion are temperate mixed forest, temperate deciduous forests and coniferous cool forests but also southern Australia, Argentina, as well as Pacific and Caribbean islands due to climate and land-use changes (Bellard *et al.*, 2013). Tropical forest and tropical woodland are projected to be less favorable for those "top invasive" species by 2080. Moreover, some regions will offer more suitable environmental conditions for survival and spread of invasive species compared to current conditions in the eastern part of the United States, northern Europe, Argentina, southern China and India (Bellard *et al.*, 2013). Indeed, poleward migrations of species are expected for many invasive alien species, leading to shifts at higher latitudes of species (Bellard *et al.*, 2013), especially in Europe where shifts are anticipating to reach unprecedented rates of 14-55km/decade (Gallardo *et al.*, 2017). Climate change might also affect establishment of new invasive species indirectly, for instance through changing patterns of human transport or by rendering existing management strategies to defend against invasive species less efficient (Hellmann *et al.*, 2008).

The potential consequences for biodiversity of these future invasions are various. One of the most dramatic consequence is local extirpation of native populations but also species extinctions on islands (Clavero *et al.*, 2009). Invasive mammal species have been a primary cause of extinctions on islands and future impact of those species on insular threatened vertebrates are predicted to increase, if no management measures are undertaken (McCreless *et al.*, 2016). A recent study focusing on Europe showed that protected areas within Europe may offer effective protection to native species against future invasions (Gallardo *et al.*, 2017). Another substantial consequence of biological invasions is the homogenization of fauna and floras which is likely to continue in the future. For instance, continental islands are projected to homogenize greatly beyond current levels of mammal assemblages, while oceanic islands are simulated to experience little additional homogenization of their mammal assemblages (Longman *et al.*, 2018). How many of future introduced species will become invasive is difficult to assess because there is generally a time lag of several decades between introduction, establishment and impact. This time lag also offers a time window for opportunities and actions to mitigate invasions.

4.2.4.5 Pollution impacts on terrestrial ecosystems: Ozone (O_3) and Nitrogen

In response to tropospheric ozone exposure, net photosynthesis declines, either due to the energy needed to produce defence compounds, or the direct damage to the photosynthetic apparatus (Feng *et al.*, 2008; Wittig *et al.*, 2009). Simulations studies result in damage of the order of approximately 10% in annual gross primary production (Franz *et al.*, 2017; Li *et al.*, 2017; Lombardozzi *et al.*, 2012; Sitch *et al.*, 2007) with feedbacks to climate by reduced terrestrial carbon sink strength (Ciais *et al.*, 2013; Sitch *et al.*, 2007). Changes in future species community composition arising from differences in species' vulnerability to ozone is not possible to project with current modelling tools, although some evidence exists that ozone indeed can affect species composition and richness (see Fuhrer *et al.*, 2016 and references therein). Large regional differences regarding ozone's future impact on plant communities, carbon or water cycling, or crop yields are to be expected (Franz *et al.*, 2017; Fuhrer *et al.*, 2016; Li *et al.*, 2017).

Eutrophication of terrestrial ecosystems has been found to affect a wide range of ecosystem functioning and community composition across ecoregions (Clark *et al.*, 2017). Nitrogen addition in experimental grassland plots reduced species richness (DeMalach *et al.*, 2017), whereas aboveground plant productivity increases across ecosystems (Greaver *et al.*, 2016). While the key processes operating in the interplay of climate change, N deposition and plant and soil physiology are rather well known, today's modelling tools are inadequate to provide process-based future projections (Greaver *et al.*, 2016). Global projections of the future C sink strength of the terrestrial biota have demonstrated large differences in models that account for C-N interactions, compared to models that ignore these (Arneth *et al.*, 2010; Wårlind *et al.*, 2014; Zaehle, 2013; Zaehle *et al.*, 2015).

4.2.5 Challenges in linking biodiversity and ecosystem functioning at the global level

Linking biodiversity quantitatively to ecosystem function, globally and across large regions, is still a challenge. Species diversity was found to correlate with productivity in (semi) natural systems and in land managed for food or timber (Duffy *et al.*, 2017; Isbell *et al.*, 2011; Liang *et al.*, 2016; Visconti *et al.*, 2018). Likewise in tropical and temperate rivers fish biodiversity correlated positively with fish yields (Brooks *et al.*, 2016). In Amazon forests, carbon storage and turnover were shown to be impacted significantly by tree-mammal interactions (Sobral *et al.*, 2017). In boreal forests, diversity and tree productivity were also correlated (Paquette & Messier, 2011). But global modelling tools to explore in marine, terrestrial and freshwater systems the futures of biodiversity or the futures of ecosystem function are still mostly disconnected (Cabral *et al.*, 2017; Mokany *et al.*, 2016, 2015; Snell *et al.*, 2014; Visconti *et al.*, 2016). This gap reflects the need for connecting model development efforts across scientific disciplines. In the marine field, for example, global scale models of ecosystem function have been mostly developed by physicists, in the form of coupled physics-biogeochemical models representing carbon and nitrogen fluxes between low trophic level functional groups (e.g., phytoplankton, zooplankton), while at the other end of the food web, fish and higher trophic level models have been developed by biologists with far more focus on life history and biodiversity, but embodying simplified forcing of climate, and less global scale perspective (Rose *et al.*, 2010; Shin *et al.*, 2010; Travers *et al.*, 2007).

Global-scale biodiversity modelling has been concerned with a sub-set of challenges, focusing on how future warming will affect the distribution or extinction of species. Interspecific interactions and multi-driver interactions are typically ignored, which can result both in over- and underestimation of risks in diversity losses (Alkemade *et al.*, 2009; Bellard *et al.*, 2012, 2013; Carpenter *et al.*, 2011; Mokany *et al.*, 2016; Pacifici *et al.*, 2015; Pereira *et al.*, 2010; Snell *et al.*, 2014; Visconti *et al.*, 2015). Little attention has been paid to global scale projections of functional, phylogenetic or genetic diversity, even though fast adaptation to environmental changes are possible through microevolution or phenotypic plasticity (section 4.2.1.2; Bellard *et al.*, 2012; Pelletier & Coltman, 2018). Likewise, DGVMS simulate ecosystem state and function, expressed as the stocks and flows of carbon, water and nitrogen (Le Quéré *et al.*, 2018), but with little consideration for interactions between and within groups of plants, or across multiple trophic levels. Potential ways forward to overcome barriers in bridging between models of ecosystem state and functioning, and models that simulate changes in diversity are being proposed in the terrestrial domain (Mokany *et al.*, 2016, 2015; Snell *et al.*, 2014). In the marine domain, integrated end-to-end models start to emerge, resulting from the coupling of disciplinary models of ocean physics, ocean biogeochemistry and fish biodiversity (Fulton, 2010; Rose *et al.*, 2010; Travers *et al.*, 2007). It is expected that approaches towards integrating models of biodiversity and ecosystem function will flourish in the future, despite the multiple technical and conceptual challenges they entail.

Large uncertainties exist both in how impact models respond to climate change and associated environmental drivers (e.g., CO_2 fertilisation, N limitations/fertilization; Ahlström *et al.*, 2012; Ciais *et al.*, 2013; Friend *et al.*, 2014; Gonzalez *et al.*, 2010; Heubes *et al.*, 2011; Huntingford *et al.*, 2009; Rammig *et al.*, 2010; Warszawski *et al.*, 2013; see also section 4.7). Regarding land-use change projections, impacts on biodiversity and ecosystems received so far much less attention compared to climate change (see 4.2.4.2, 4.2.4.3). Futures of other drivers still need to be explored despite of their known large impacts on biodiversity and ecosystems in the past, and today (pollution, invasive species). Moreover, model experiments as well as observational studies tend to concentrate on single-driver responses, despite indications that combined effects cannot be predicted from the sum of single-factor responses (Alkemade *et al.*, 2009; Fu *et al.*, 2018; Langley & Hungate, 2014; Visconti *et al.*, 2015).

Clearly, improvements of scenarios and modelling tools are still needed to be able to represent the future environmental conditions (i.e. the range of conditions that will impact on biodiversity) in a way that is comparable across direct drivers and that enable us to make a fair comparison of their expected impact in the future. For that reason, the overall issue of the relative and combined expected impacts of different drivers in the future remains unresolved.

4.3 PLAUSIBLE FUTURES FOR NATURE'S CONTRIBUTIONS TO PEOPLE

4.3.1 Nature's contributions to people across scenario archetypes

Scenarios and models are important tools for understanding how the multiple contributions of nature to people (NCP) might unfold in the future. Scenarios that are adverse for biodiversity and ecosystem function are likely to be adverse for NCP because of known links between biodiversity, ecosystem function and the material, regulating and non-material benefits to humans (Mace *et al.*, 2012). Nonetheless, there is still a lack of robust knowledge and quantitative estimates of these relationships, and thus how they might impact future changes in NCP.

Scenario archetypes were used to examine the relationship between different socio-economic development pathways and their impacts on the three broad categories of nature's contributions to people (regulating, material and non-material contributions), as interpreted mostly from the ecosystem services literature. Results from the systematic literature review of global and continental-scale scenarios (see Appendix A4.1.1) were classified as falling under "economic optimism" (75 = number of results), "global sustainability" (35), "regional competition" (59), "business-as-usual" (34), "regional sustainability" (14), and "reformed markets" (31) (**Figure 4.3.1**; see also section 4.1.3 for archetype descriptions). Overall, global and continental-scale scenarios addressing NCP are scarce and biased towards a few categories. Some NCP are relatively frequently analyzed such as food and feed, regulation of freshwater and climate;

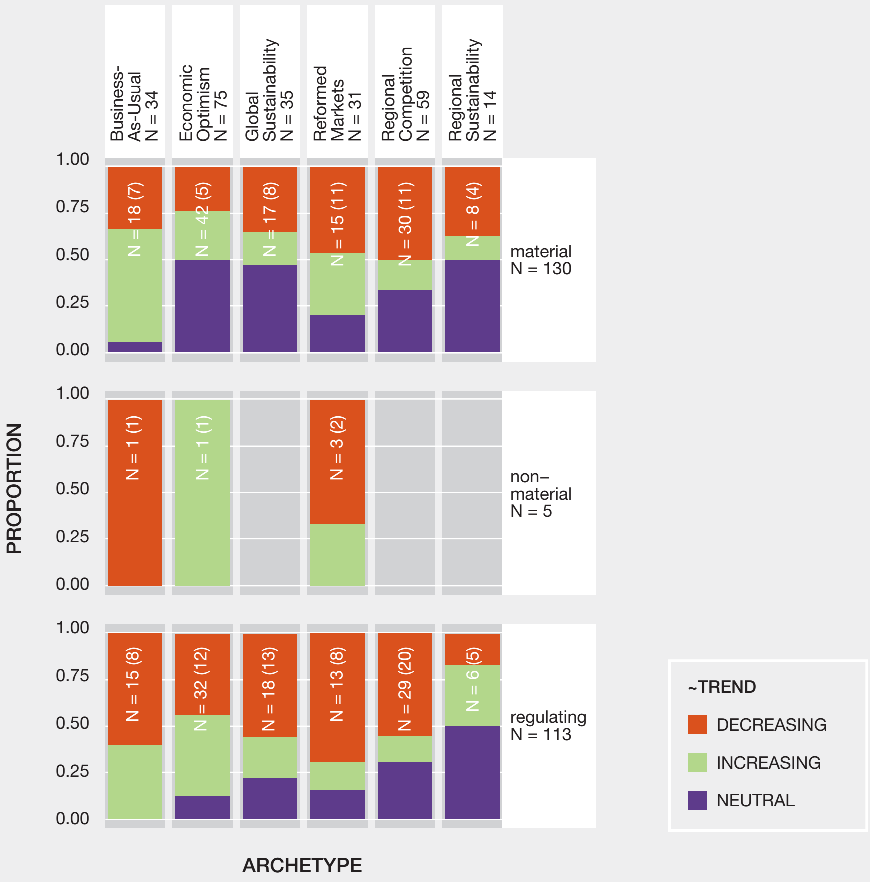


Figure ④ ③ ① **Results of the systematic literature review (Appendix A4.1.1) showing the three broad groups of nature's contributions to people (material, non-material, and regulating NCP) for each of the six scenario archetypes.**

The y-axis indicates the proportion of negative and positive trends reported in the literature review. Numbers (N) indicate the number of results, followed by the number of articles that report those results in parentheses.

while non-material NCP or some regulating NCP such as regulation of the impacts of hazards and extreme events and regulation of ocean acidification are covered by a very low number of studies at continental or global scales.

It should be noted that the reviewed literature usually uses the terminology of "ecosystem services" or reports on aspects of ecosystem services without making explicit reference to the ecosystem services framework. Chapter 1 presents a detailed discussion about the relationship between ecosystem services and NCP categories. The literature has been interpreted accordingly, and ecosystem services have been reclassified into IPBES NCP categories. In this section, the term "ecosystem service" is, however, used instead of NCP when it is helpful for clarity and understanding.

4.3.2 Changes in nature's contributions to people

Regulating NCP show decreasing trends in the future in most scenario archetypes **(Figure 4.3.1)**, with only "regional sustainability" and "economic optimism" scenarios showing mixed trends for regulating NCP. "Reformed markets" and "business-as-usual" scenarios present the highest proportion of declining trends for regulating NCP. Material NCP show mixed trends along scenario archetypes. "economic optimism" is the scenario that shows the lowest number of negative trends for material NCP followed by "business-as-usual" and "Global Sustainability". In all cases, published studies focused on the supply of NCP (which is not deconvoluted with the demand of NCP) and did not take into account flows, uses, beneficiaries or values.

Figure 4.3.2 shows the trends for three NCP with the most entries in the systematic literature review database. Food and feed show a mixed picture, while regulation of climate shows a more positive picture and regulation of freshwater a very negative one. This is especially worrisome, because water is the basis for the generation of all other NCP and the direct well-being of humans.

4.3.2.1 Nature's contribution to people – regulating contributions

Habitat creation and maintenance

Habitat creation and maintenance has crucial importance for facilitating all NCP. Considering the projected increasing loss of natural vegetation cover in nearly all future land-use

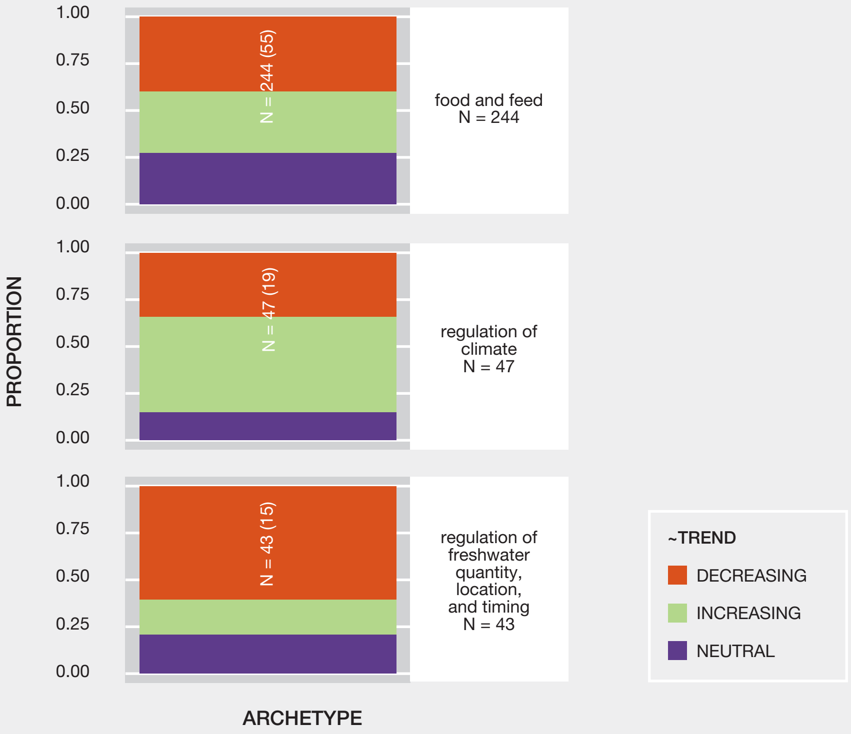


Figure ④ ③ ② **Results of the literature review showing the trends for three NCP categories that are the most frequently represented in studies.**

There were insufficient entries to differentiate between archetype scenarios so this figure shows the general patterns over all scenario types.

change scenarios and the climate change induced shift in natural vegetation distribution (see section 4.2.4), it is to be expected that species with specific habitat requirements will be under increasing pressure. Homogenization of communities and habitats is expected to have negative consequences on the ability of ecosystems to maintain multiple ecosystem functions. In addition to habitat specialists, species that can be classified as being intermediate between specialists and generalists will be under increasing pressure, since these species tend to rely on intact metapopulations and are vulnerable to increasing degradation of landscapes. Their loss would have a particularly large impact on genetic diversity since generalist species tend to have more genetic variability compared to specialists (Habel & Schmitt, 2018).

Projections of future interactions between changes in terrestrial habitats and biodiversity focus either on climate change impacts, or on the transformation of natural ecosystems into agricultural systems as main drivers (section 4.2.4; Alkemade *et al.*, 2009; Bellard *et al.*, 2012; Jantz *et al.*, 2015; Mantyka-Pringle *et al.*, 2015; Pereira *et al.*, 2010; Visconti *et al.*, 2016; Warren *et al.*, 2011). At the global scale, little attention has been paid to restoration scenarios. Likewise, most biodiversity and ecosystem models do not have the capacity to represent habitat degradation and fragmentation (Bonan & Doney, 2018). Beyond the use of species distribution models, actual movement of species, either as individuals or as groups is often not taken into account in models used to project interactions between changing environments and populations (Holloway & Miller, 2017), which implies large uncertainty regarding the future vulnerability and/or resilience of habitats and their interactions with the populations these habitats sustain.

Pollination and propagule dispersal

Animal pollination and propagule dispersal play a vital role as a regulating NCP, including for food production and many other ecosystem services. Projected loss of diversity of pollinators and alteration of their communities generate risks for food security, human health and ecosystem function. Pollinators and the provision of pollination will be negatively impacted by land-use change (habitat destruction, fragmentation and degradation), intensive agricultural management and pesticide use, environmental pollution, invasive alien species, pathogens and climate change (Chagnon *et al.*, 2015; IPBES, 2016a; Vanbergen *et al.*, 2018). For instance, the spread of invasive ants that can deter pollinators and seed dispersers is anticipated to continue (see also section 4.2.4) and projected to substantially impact future pollination services (Vanbergen *et al.*, 2018). Impacts of climate change on pollinators are the most commonly reported scenario results. Under all climate change scenarios, pollinator community composition is expected to change. The projected velocity of climate change, especially under mid- and high-end emission of greenhouse gas scenarios, exceeds the maximum speed at which several groups of pollinators (e.g., many bumble bees or butterflies) can disperse or migrate (IPBES, 2016a). Differential phenological shifts can cause mismatches between plant and pollinator populations and lead to the extinctions of plant or pollinator species, with expected consequences on the structure of plant pollinator networks (Hegland *et al.*, 2009; Lavergne *et al.*, 2010; Memmott *et al.*, 2007). However, the inherent plasticity of plant–pollinator interactions suggests that many species may be able to persist, even though their mutualistic partners may change (Burkle & Alarcón, 2011).

Many management responses are available that can reduce the risks of pollination deficit in the short term, including land management to conserve pollinator resources, decreasing pollinator exposure to pesticides, and improving managed pollinator techniques (IPBES, 2016b). The disruption of propagule dispersion due to biodiversity loss is also expected to disturb ecological communities and threaten important ecosystem functions and NCP. For example, frugivore defaunation in tropical forests can lead to local extinction of trees depending on them to reproduce and the induced changes in tree species composition will likely result in the loss of carbon storage capacity of tropical forests (Bello *et al.*, 2015).

Regulation of air quality

Terrestrial ecosystems are large emitters of substances that are relevant for air quality, in particular biogenic volatile organic compounds (BVOC) and emissions from wildfires. Several studies using coupled vegetation and BVOC models show that climate change alone enhances emissions due to their temperature-dependent response (Arneth *et al.*, 2011; Niinemets *et al.*, 2010). However, land-use change is simulated to counteract these effects, in particular for compound groups isoprene and monoterpenes, since woody vegetation tends to emit more BVOC than crops. The effects of rising atmospheric CO_2 are difficult to quantify, because CO_2 enhances productivity which increases emissions, but on the other hand high CO_2 concentrations have been shown to reduce leaf-level emissions – at least for isoprene (Hantson *et al.*, 2017; Heald *et al.*, 2008; Squire *et al.*, 2014; Szogs *et al.*, 2017; Tai *et al.*, 2013). Wildfire emissions, similar to BVOC, are expected to increase in a warmer climate as fire-prone conditions are enhanced (Hantson *et al.*, 2016). In case of fire, atmospheric CO_2 enhances plant productivity, and hence combustible litter, but also leads to a shift towards more woody vegetation, which slows fire spread compared to grasslands (Hantson *et al.*, 2016; Knorr *et al.*, 2016; Rabin *et al.*, 2017). How BVOC and wildfire emissions will affect future air quality and climate regulation will depend not only on how climate change will affect biogenic emissions, but also on how anthropogenic air pollutants will alter biogenic emissions and chemical reactions in a future atmosphere (Shindell & Faluvegi, 2009; Shindell *et al.*, 2009; Tsigaridis *et al.*, 2014; Young *et al.*, 2009). Anthropogenic emission controls are much more important than biogenic emissions for air quality. However, assessments of impacts of bioenergy, reforestation and afforestation efforts on air quality and climate regulation must consider side effects of biogenic emissions on human health and on climate-related substances, as well as (in case of wildfire) the risk of forest loss (Ashworth *et al.*, 2013; Rosenkranz *et al.*, 2015; Simpson *et al.*, 2014).

Regulation of climate

Oceans and terrestrial ecosystems currently take up around 50% of anthropogenic CO_2 emissions each year (sections 4.2.2, 4.2.4; Le Quéré *et al.*, 2016). In the future, these carbon sinks may weaken, resulting in amplifying feedbacks to climate change (Arneth *et al.*, 2010; Ciais *et al.*, 2013; section 4.2.4). In oceans, warmer temperature, increased stratification of the water column, deoxygenation, and acidification, as well as sea level rise in coastal wetlands, might lead to a reduction of the sink (see 4.2.2.2.1, 4.2.2.2.2), while in terrestrial ecosystems, the interplay between CO_2-fertilisation of photosynthesis, heterotrophic respiration stimulated by warmer temperatures, and episodic events such as fire, insect outbreaks, or heat waves are controversially debated with respect to their impacts on future carbon uptake and climate regulation (Ciais *et al.*, 2013; Kautz *et al.*, 2017). Reducing greenhouse gas emissions from land cover change and land use, mostly related to human conversion of forests to crops and pastures, fertilizer use, rice production and animal husbandry could contribute notably to mitigate climate warming (Bustamante *et al.*, 2014; Smith *et al.*, 2014b, 2013; Tubiello *et al.*, 2015). Changes in vegetation cover would impact also regional temperature and precipitation. In tropical regions, deforestation is simulated to lead to local warming, as croplands tend to have considerably lower evapotranspiration. By contrast, in boreal regions changes in surface reflectance is the predominating factor and deforestation results in local cooling (Alkama & Cescatti, 2016). Therefore, in tropical regions, avoiding deforestation will contribute to reduce CO_2 emissions, as well as contribute to moderate the impact of regional warming – supporting also the maintenance of biodiversity (Alkama & Cescatti, 2016; Perugini *et al.*, 2017; Quesada *et al.*, 2017a).

Regulation of ocean acidification

Increasing atmospheric CO_2 concentrations will increase the partial pressure of CO_2 (pCO_2) and its dissolution in the surface ocean (section 4.2.2; Le Quéré *et al.*, 2016). It is expected that pCO_2 might double its pre-industrial value within the next 50 years (Eyre *et al.*, 2018; Hoegh-Guldberg *et al.*, 2017). Decreased calcification in calcified organisms due to increased acidification of the ocean is likely to impact marine food webs and, combined with other climatic changes in temperature, salinity, and nutrients, could substantially alter the biodiversity and productivity of the ocean (Dutkiewicz *et al.*, 2015; Kawaguchi *et al.*, 2013; Larsen *et al.*, 2014; Meyer & Riebesell, 2015). How species will respond to these changes depends on their capacity for adaptive responses. Many studies project the degradation of a large percentage of the world's tropical coral reefs (Albright *et al.*, 2018; Eyre *et al.*, 2018; Sunday *et al.*, 2017 section 4.2.2.2.2) and calcifying marine species like bivalves, might as well be significantly endangered due to ocean acidification (Hendriks *et al.*, 2010; Kroeker *et al.*, 2010). This is projected to impact many regulating ecosystem services and entire sectors of human activities and millions of livelihoods, both in developed and especially in developing countries that depend on fish and other marine products for their daily sustenance (Hilmi *et al.*, 2015; Mora *et al.*, 2013a). Moreover, recreational activities, as well as tourism which are among the world's most profitable industries (Rees *et al.*, 2010) are projected to decline by up to 80% in some areas due to climate change (Moreno & Amelung, 2009; USGCRP, 2008). Although local and regional-scale management strategies may build resilience in the short term, longer term resilience will further require a successful shift to a low greenhouse gas emissions scenario, e.g., RCP2.6 or RCP4.5 (Anthony, 2016).

Regulation of freshwater quantity, location and timing

Today, two-thirds of the global population live under conditions of severe water scarcity at least one month of the year and half a billion people face severe water scarcity all year round (Mekonnen & Hoekstra, 2016). World water demand is estimated to increase significantly, up to 50% by 2030 (UNDP, 2016), mostly due to population growth and lifestyle choices, such as shifting diets towards highly water-intensive foods (see section 4.5.3). Scenarios of water use foresee overexploitation, pollution or degradation of aquatic ecosystems (see 4.2.3) and the ecosystem services they provide or produce together with other ecosystems (Molle & Wester, 2009). Societal problems and new inequalities will also emerge as a result (Bruns *et al.*, 2016). The projected increases in human population and per capita consumption will likely lead to a sharpening of already existing water shortages if the demand of freshwater cannot be satisfied (Alcamo *et al.*, 2007; Murray *et al.*, 2012; Pfister *et al.*, 2011). Some estimates put demand surpassing supply significantly already in 2030 (Mekonnen & Hoekstra, 2016). Changing climate is progressively modifying all elements of the water cycle, including precipitation, evaporation, soil moisture, groundwater recharge, and run-off. But it is also expected to change the timing and intensity of precipitation, snowmelt and run-off (Murray *et al.*, 2012). Indirect effects of land-use change, such as deforestation, is also expected to increasingly affect water quality, water quantity and seasonal flows, especially in the tropics (Piao *et al.*, 2007). Many of the world's most water-stressed areas will likely get less water, and water flows will become less predictable and more subject to extreme events (Mayers *et al.*, 2009; Mekonnen & Hoekstra, 2016). The additional challenges for water security posed by poor management are expected to first become apparent in mega-cities. Increasing demands for water by agricultural, industrial and urban users, and water for the environment will intensify competition (Mayers *et al.*, 2009; Murray *et al.*, 2012; Pfister *et al.*, 2011). In order to address these challenges, water needs to be used more efficiently in agriculture (Fraiture & Wichelns, 2010) and caps to water consumption by river basin have been proposed (Mekonnen & Hoekstra, 2016).

Formation, protection and decontamination of soils and sediments

The Sustainable Development Goals related to food, health, water supply, biodiversity and climate all rely on healthy soils (Arcurs, 2017). Human activity has increased the erosion rates well above natural levels, degrading soils structurally and nutritionally and generating a surplus of sediment transport to rivers, which damages infrastructure, aquatic habitats and deteriorates water quality (Bouchoms *et al.*, 2017; Doetterl *et al.*, 2016; Li & Fang, 2016). Whether or not the eroded material decomposes rapidly or even acts as a carbon sink is still being debated (see Doetterl *et al.*, 2016 and references therein). Climate change is expected to globally exacerbate erosion rates in the future although exact rates and magnitude are poorly understood and large regional variability is to be expected (Li & Fang, 2016). Water erosion caused by overall enhanced precipitation in some regions or by extreme precipitation can be expected to increase (Bathurst, 2011; Bussi *et al.*, 2016; Hu *et al.*, 2013; Shrestha *et al.*, 2013). In a recent compilation of erosion model studies, most at catchment scale, Li & Fang (2016) found enhanced future erosion in response to climate change in 136 of 205 listed studies. Soil erosion can be effectively reduced by land management practices (reduced tillage, vegetation cover) (Doetterl *et al.*, 2016; Poesen, 2018). However, models that combine soil organic carbon cycling with modelling of degradation processes at regional to global scales do not yet exist. Therefore, scenarios of possible futures are virtually absent, and global or sub-global studies could not be found on future soil degradation, nor on soil restoration (IPBES, 2018f).

4.3.2.2 Nature's contributions to people – changes in material contributions

Energy

Ecosystems provide relatively inexpensive and accessible sources of traditional biomass energy, and therefore have a vital role to play in supporting poor populations. Bioenergy draws on a wide range of potential feedstock materials: forestry and agricultural residues and wastes of many sorts, as well as crops or short-rotation forests grown specifically for energy purposes (Smith *et al.*, 2016). The raw materials can be converted to heat for use in buildings and industry, to electricity, or into gaseous or liquid fuels, which can be used in transport. Today's global supply of bioenergy is around 10% of the total demand (Smith *et al.*, 2016). The global demand for primary energy is projected to grow across future scenarios, unless the world's energy system were to transformatively change within the coming two or three decades (IPCC, 2018, Chapter 2.3). Bioenergy is estimated to provide ca. 100-300 EJ a^{-1}, accounting for 15-25% of global future energy demand in 2050, but concerns about the sustainability have been raised even for amounts of 100 EJ a-1 or well below (Beringer *et al.*, 2011; IPCC, 2018, Chapter 2.3; Smith *et al.*, 2016). Deriving about 20-60% of total energy from energy crops would require up to a doubling of land and water resources (Beringer *et al.*, 2011).

Recent scenarios in Integrated Assessment Models that explore options to achieve global warming of 2°C or less include large-scale bioenergy for climate change mitigation (see 4.2.4.3; Bonsch *et al.*, 2016; Smith *et al.*, 2014b, 2016). Combining bioenergy with carbon capture and storage (BECCS) may offer the prospect of energy supply with large-scale net negative emissions, which plays an important role in many low-emission scenarios (Bruckner *et al.*, 2014; IPCC, 2018, Chapter 2; Tavoni & Socolow, 2013). However, there are challenges and risks entailed, as shown by an increasing number of studies, especially around potential conflicts with biodiversity and other NCP (Fuss *et al.*, 2016; Humpenoder *et al.*, 2014; Santangeli *et al.*, 2016; Smith *et al.*, 2016). The use of different sources for bioenergy production will have large impacts on the capacity of energy crop production, climate change mitigation and thus on the trade-offs with other NCP (Gelfand *et al.*, 2013). The trade-offs most often cited are with food production, biodiversity and terrestrial carbon storage (Beringer *et al.*, 2011). Food production will be impacted not only by conflicts in land use as such, but also because of rivalling water use through irrigation of bioenergy crop production (Beringer *et al.*, 2011). Also, the future benefit of CO_2 savings of bioenergy crops is not completely clear, as many studies do not include the emissions of N_2O in crop production that could offset CO_2 savings (Don *et al.*, 2012), or the long-term CO_2 emitted by land conversion or deforestation of natural vegetation to bioenergy crop areas (Don *et al.*, 2012; Krause *et al.*, 2017, 2018).

Food and feed materials

The largest anthropogenic use of land and water is related to the production of food. Also, food production is the largest component of human domination of the global nitrogen and phosphorus cycles (Bouwman *et al.*, 2013).

The drivers are both the food demand (type of diets, wealth and population size) and the food production system (productivity of the agricultural, aquaculture and livestock systems, exploitation of wild species, transport, waste). Rapid changes in dietary patterns since the end of 20th century (mainly in transitioning countries: Latin America, East Asia, others) have become a major factor in global land-use change pressures, mainly related to the increase of animal products consumption (Kastner & Nonhebel, 2010; Kastner *et al.*, 2012). In the coming decades, the increase in consumption of animal products is expected to play the strongest role in the demand of land, water, nutrients (N, P, K) and energy (and related CO_2 emissions) for food production (Alexander *et al.*, 2016; Peters *et al.*, 2016; Ranganathan *et al.*, 2016; Wirsenius *et al.*, 2010), due to the poor resource efficiency in the production of animal, especially ruminant protein. Therefore, land degradation and its impacts on food security are likely to increase, especially in developing regions with high and increasing demographic pressure, pressures from export-oriented commodity production expansion, scarce land and water resources and weak governance structures. Importantly, effects of land degradation on food security are not considered in any global scenario study (IPBES, 2018f). For sufficient land and water resources being available to satisfy global food demands during the next 50 years, water will have to be managed much more effectively in agriculture (Fraiture & Wichelns, 2010). Supplying sufficient calories and an overall healthy diet to feed the global population with sustainable production systems is a recognized challenge and will require solutions from local to global levels, addressing both food production, distribution and trade, and consumption (Foley *et al.*, 2011; Godfray *et al.*, 2010; Tilman & Clark, 2015). Closing yield gaps in many regions of the world may play a major role if done using sustainability principles for land management. This poses a large challenge as climate change has been projected to reduce crop yields in tropical and semi-arid regions; regions in which already today large yield gaps exist (Pugh *et al.*, 2016a; Rosenzweig *et al.*, 2013) and which include countries with projected fast changes in diets and population growth. There is large uncertainty in how extreme weather events, pest and diseases and atmospheric CO_2 levels will interact with yields (Deryng *et al.*, 2014; Gornall *et al.*, 2010; Rosenzweig *et al.*, 2013). Thus, it is necessary to increase productivity sustainably and at the same time reduce the vulnerability of agricultural production systems to climate change impacts.

Medicinal, biochemical and genetic resources

Because genetic diversity of crops and their wild relatives is a product of both the natural process of evolution and the biocultural process of evolution under domestication, genetic diversity is a source of, and a proxy for options for the future, and hence maintains options for the supply of ecosystem services (Bellon *et al.*, 2018; Faith *et al.*, 2017). However, if yields continue to be increased by means of intensive agriculture, then the environmental consequences would be substantial (Tilman *et al.*, 2001) and to the detriment of other NCP (section 4.5). The current diet worldwide is based on only 150 of the more than 7,000 plant species that humans have utilized historically for food (Gepts, 2006) and food supplies have become increasingly similar in composition across the globe (Khoury *et al.*, 2014).

The conservation of genetic resources from local varieties and crop wild relatives plays an important role in increasing productivity sustainably, maintaining local food security and quality, as well as in providing adaptive options for agricultural systems to grow diverse and nutritious food with fewer resources in harsh environments. For instance, cultivars based on local varieties can be grown in marginal conditions where commercial varieties do not perform well (Ceccarelli, 2009), and crop wild relatives harbor genetic adaptations to drought, pest and diseases resistance (Maxted *et al.*, 2013). Therefore, genetic diversity represents a source of options to face the increasingly uncertain and variable patterns of biotic and abiotic changes (Bellon *et al.*, 2017). Similarly, deploying sufficient genetic diversity decreases the risk of pathogens reaching epidemic levels and causing large-scale crop failure (Heal *et al.*, 2004).

Indigenous Peoples and Local Communities play an essential role in this regard both in managing key agrobiodiversity areas around the world and holding the knowledge that gives meaning to the value of such diversity. Maintaining in-situ crop genetic diversity is at present done mostly by smallholders and indigenous communities, cultivating local varieties individually in small-scale mosaic production systems, but these constitute in many regions large effective systems in providing food to large regional populations within a wide range of environmental conditions and cultural preferences (Bellon *et al.*, 2018; Enjalbert *et al.*, 2011). If trends towards replacing local varieties with genetically homogeneous materials of the private sector continue (Heal *et al.*, 2004; Howard, 2009), evidence suggests that while crop production yield may increase (particularly for crops destined to industrial uses and fodder), food security may be compromised not only in terms of lower crop production of food crops, but also in the form of higher risk and vulnerability of farmers and the food system to future challenges.

4.3.2.3 Nature contributions to people – changes in non-material contributions

The results of the systematic literature review highlight the scarcity of global or continental scale scenarios addressing non-material contributions to people: these have received far less attention than material and regulating NCP. Even on the local scale, the number of scenario studies dealing with the category of cultural ecosystem services is limited.

The sections below describe how different non-material NCP might unfold in the future based on scenario studies at different scales, including some local studies. In order to arrive at a better understanding on how changes in nature and changes in people's demands interact for all NCP, future studies that target non-material NCP are needed.

Learning, artistic, scientific and technological inspiration

The published literature on the future evolution of this category of NCP is scarce with most studies focusing on the current state of nature-inspiration for learning, the arts, science and technology. Nature inspiration for the arts, including music, painting and literature comes ultimately from the fact that we are part of nature, and that when we are amazed by certain aspects of nature, this inspires individuals to express their creativity (Komorowski, 2016). Whether the ongoing disconnection of humans from nature (Soga & Gaston, 2016) will affect how art is inspired by nature in the future is unresolved. Nature-inspiration has advanced technology in multiple ways, the Lotus effect or the shark skin effect being some of the most common examples (Bhushan, 2016). Nature inspiration has played a significant role in computation and communication and it is likely that it will continue doing so (Vinh & Vassev, 2016). The self-organized architecture of nature can play a major role in nature-inspired algorithms and computing (Yang, 2014, 2010). Bioinspiration and biomimetics in engineering and architecture has a long history of application, but its future development is uncertain (Ripley & Bhushan, 2016).

Physical and experiential interactions with nature

Connections to nature have been classified as being material, experiential, cognitive, emotional, and philosophical (Ives *et al.*, 2018). Partially as a result of rapid urbanization (see section 4.3.3 and Jiang & O'Neill, 2017) some argue that urbanites are undergoing an "extinction of experience" resulting from decreasing contact with nature in everyday live (Soga & Gaston, 2016). Although varying significantly across and within regions, interactions with nature have been changing from direct subsistence interactions (i.e. through agriculture, farming, fishing, hunting, herding, foraging) to sporadic subsistence, leisure, education, or as health-recommendation. This trend is expected to continue in the future although other forms of interaction with nature are also emerging, such as increasing attention to urban parks, river and lake restoration projects, urban gardens, and increasing green infrastructure in cities (Grimm & Schindler, 2018; Shanahan *et al.*, 2015; Thompson *et al.*, 2008). Indicators to assess interactions with nature are scarce. Visits to protected areas have been estimated at 8 billion per year (Balmford *et al.*, 2015) with a generally increasing trend (except for some developed countries (Balmford *et al.*, 2009), but it is unclear how this figure will evolve under different scenarios. Apart from protected areas, direct interactions with nature occur in many non-protected landscapes, from urban parks, to rural areas and remote landscapes. These interactions are more widespread than visits to protected areas and happen continuously.

The main drivers expected to affect future physical and experiential interactions with nature through nature tourism are demographics, urbanization, climate change, technology, psychological drivers, health care trends and development (Frost *et al.*, 2014). A warmer future may increase the visits to protected areas, especially to mountain protected areas where temperatures are cooler (Fisichelli *et al.*, 2015; Steiger *et al.*, 2016). In some areas, a business-as-usual scenario might reduce our interactions with nature due to the loss of natural ecosystems through deforestation. Local scenarios in the Eastern Arc Mountains in Tanzania show that non-sustainability pathways would also reduce ecotourism (Bayliss *et al.*, 2014). Participatory scenario planning approaches in which stakeholders co-develop different scenarios have been used in several local studies and assessed future trends of diverse non-material NCP such as interactions with nature (Oteros-Rozas *et al.*, 2015). Future trends for ecotourism, for example, were analyzed through the integration of ILK and scientific knowledge for a case study in Papua New Guinea (Bohensky *et al.*, 2011b).

Symbolic meaning, involving spiritual, religious, identity connections, social cohesion and cultural continuity

Among the very few existing scenario-based studies that specifically focus on this nature's contribution to supporting identities (Díaz *et al.*, 2018), some focus on sense of place, which is highly relevant for ecosystem service stewardship and for human well-being, particularly of IPLCs (Masterson *et al.*, 2017). Some analyses suggest that climate change might negatively affect sense of place (Ellis & Albrecht, 2017), an issue of concern to an increasing number of people living in coastal areas and under increasing risks such as floods and sea level rise will increase (Neumann *et al.*, 2015). Sense and forms of attachment to place are also negatively affected by changes caused by infrastructural responses, such as the need to construct flood defenses (Clarke *et al.*, 2018).

Identities that are linked to nature, such as those related to cultural keystone species, will probably decline under certain scenarios (Garibaldi & Turner, 2004). In business-as-usual scenarios indigenous identities are expected to decrease, as these are often linked to nature, and Indigenous People´s spiritual beliefs (Dudgeon *et al.*, 2010). Hunting practices that have deep cultural meanings for some local communities and help to bound some societies might be affected as well (Luz *et al.*, 2017). In cities, declining green space might produce feelings of loneliness and shortage of social support (Maas *et al.*, 2009). Connecting theories and tools related to sense of place within broader socio-ecological systems research is expected to enhance our understanding as to how and why people engage in solving challenges related to sustainable use of ecosystems (Masterson *et al.*, 2017).

Preservation of biodiversity and ecosystems, as options for the future

One of the challenges posed by the expected continuous degradation of ecosystems and loss of biodiversity in most scenario archetypes is to assess the implications of these trends in terms of options for the future (Pereira *et al.*, 2010 and see section 4.2). Local level examples **(see Box 4.3.1)** highlighting the interdependence between nature, indigenous and local knowledge, and local livelihoods provide powerful stories about economic-environmental trade-offs and the importance of maintaining options, including in terms of complementary knowledge systems, in times of accelerated environmental and social changes.

Box 4 3 1 An example of the role of Indigenous Local Knowledge in sustaining ecosystem services.

The shea tree is highly valued by rural households in Western and Central Africa. The shea fruit is a non-timber forest product that is indigenous to ecosystems in semi-arid regions of Africa (Jasaw *et al.*, 2015). Shea is exported as raw kernels or as shea butter to serve the high-value cosmetic and personal care industry and the wide range of food products in USA, Europe, and Japan. It currently grows throughout semi-arid northern Ghana (CRIG, 2007; Naughton *et al.*, 2015), with almost every rural household in the region engaging in shea fruit picking, and processing into shea kernels (shea nuts) and/or shea butter. For years, local populations have followed local knowledge, norms and practices including not using shea for fuelwood and integrating it into farmlands to preserve and manage it (Jasaw *et al.*, 2015). In recent years however, high disregard for indigenous knowledge practices, degradation and subsequent scarcity of traditional fuelwood tree species, and fluctuating world market prices for shea products, have pushed locals being faced with the dilemma of still preserving the tree to enable them earn income or cut the trees for fuelwood (Boafo *et al.*, 2016; Jasaw *et al.*, 2017). If current trends continue, the co-production of the shea butter will continue eroding indigenous and local knowledge (ILK), the management of common resources, as well as regulating and non-material contributions from nature to people. Both technological improvements (such as improved stoves) and the strengthening of community-based woodland management (such as harvesting tree branches instead of whole trees) need to be put in place to revert this trend (Boffa, 2015; Jasaw *et al.*, 2017, 2015).

Figure 4 3 3 **Woman taking shea harvests home to process.**

Photo credit: Yaw Boafo, 2014.

Figure ④ ③ ④ **Woman sorting shea kernel for sale in Northern Ghana (left); Shea kernel being dried after picking from the wild in Northern Ghana (right).**

Photo credit: Yaw Boafo, 2014.

Future scenarios of climate change predict in this case an increased climate suitability for the shea tree (Platts *et al.*, 2010). This could open certain opportunities to adapt to climate change and at the same time reinforce the value of ILK in landscape management. Since the traditional form of Shea butter production also requires large amounts of energy (Jasaw *et al.*, 2015), six scenarios of future development of technologies were developed for Burkina Faso (Noumi *et al.*, 2013). The improvement of the energy systems would result in better incomes for women and reduced vulnerabilities of rural families whilst minimizing land degradation and enhancing carbon sequestration potential of savannah landscapes.

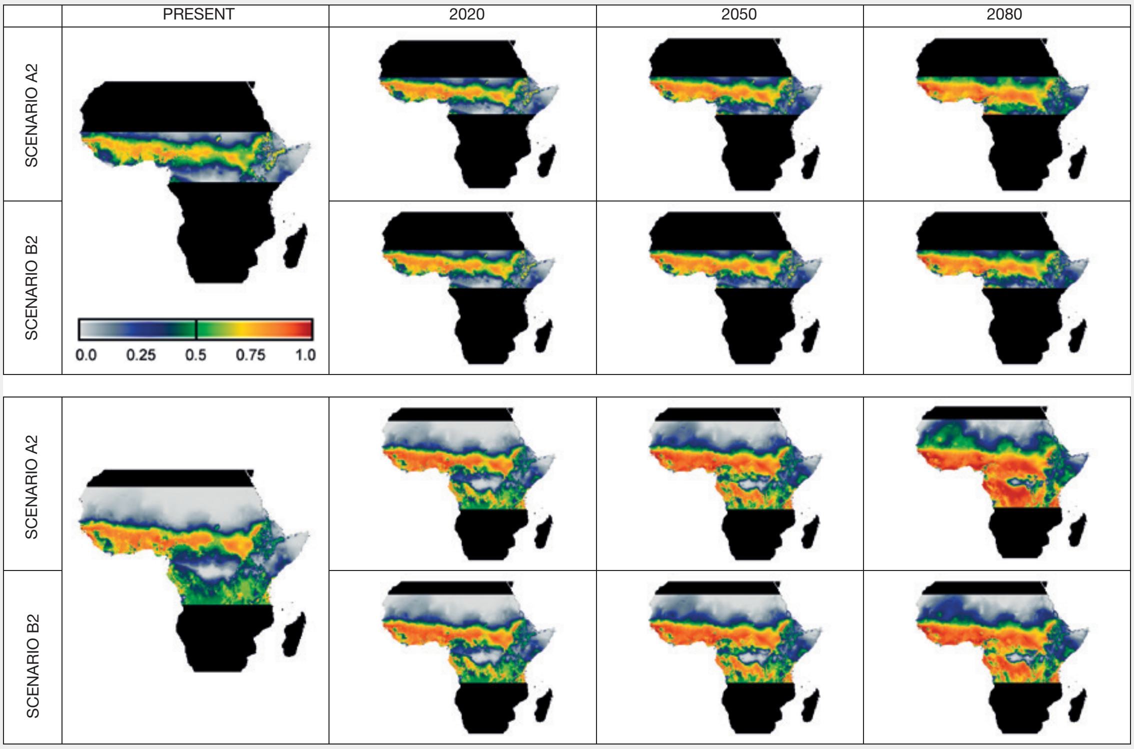


Figure ④ ③ ⑤ **Present situation and future scenarios of the climatic suitability for the distribution of the shea tree.**

In both scenarios, niche-based models predict an enhanced climatic suitability for the shea tree during the 21st century (Platts *et al.*, 2010). Top panels are projections based on a restrained geographical range for model calibration and lower panels are based on a broader geographical range. The suitable habitat for the shea tree in central Africa is projected to increase in two explored IPCC scenarios (A2 and B2) in 2020, 2050 and 2080. According to these scenarios, the maximum suitability is predicted for 2080.

4.3.3 How changes in nature's contributions to people will manifest in different regions, including teleconnections across regions

Ecosystems and biomes (or IPBES units of analysis) are interconnected, influence each other and thus many NCP are also interconnected in space (Álvarez-Romero *et al.*, 2018; Liu *et al.*, 2015). These interactions can occur in the natural system (e.g., via the atmosphere, or through river flows), often called teleconnections. In socio-economic and socio-ecological systems the telecoupling concept considers interactions, feedbacks and spillover between different and typically distant system components (e.g., by trade or migration; Güneralp *et al.*, 2013; Liu *et al.*, 2013; Melillo *et al.*, 2009). Through those mechanisms, resource use and ecosystem management in some regions affects NCP from other regions (Pascual *et al.*, 2017; see section 4.5 and Chapter 5). For example, the displacement of timber extraction from Finland to Russia has created environmental impacts in Russia that in turn affected migratory birds in Finland (Mayer *et al.*, 2005).

Knowledge about the interaction, feedback and spill-overs among regions, and implementation in future global scenarios is needed for better projections and management of NCP including flow-based aspects of governance beyond the classical territorial approaches (Liu *et al.*, 2013; Sikor *et al.*, 2013). Without such knowledge, decisions on the management of NCP in one region will lead to incomplete and skewed conclusions that affect sustainability at the global level (Schröter *et al.*, 2018). For example, telecoupling is linked to remote, large-scale investment in land purchase or lease and freshwater demand, which is happening in all continents except Antarctica (Rulli *et al.*, 2013). Also in context of urban-rural relations this consideration can help to better understand interactions with systems beyond their boundaries (Seto *et al.*, 2012).

Urbanization is one of the global development trends that has large impacts on local and distant socio-ecological systems. The global urban population represents now 55% of the total population and is projected to reach 6.6 billion by 2050 (68% of the total population) (https://population.un.org/wup/).

In the vicinity of cities, urban growth leads to the loss of agricultural land and hence agricultural production, and associated land-use displacement to other regions as compensation. Overall it is estimated that, due to urban build up, 1.8–2.4% of the global croplands will be lost by 2030 (Bren d'Amour *et al., 2017)*. On local and regional level urban areas modify climate through the urban heat island effect, impacting also human health. In combination with altering of precipitation patterns, the heat island effect will possibly also have significant impacts on net primary production, functions of ecosystems, and biodiversity in larger urban regions (Seto *et al., 2013)*. Urbanization also frequently correlates with lifestyle and dietary changes towards more meat and fish (Satterthwaite *et al., 2010)*. As a result, long-distance connections intensify as demand for resources increases to support these urban lifestyles and activities. Often such change in demand is not only met by intensification but also by cropland expansion into semi-natural or natural vegetation (DeFries *et al., 2010)*, which in turn may lead to the displacement of local farmers due to loss of land and increases migration to urban areas.

There are very few global scenario studies of telecouplings, and the related interactions between nature and NCP. For instance, most forward-looking studies on impacts of urbanization on ecosystems focus on impacts on biodiversity and habitats (Güneralp *et al.*, 2013). There are no quantitative studies and scenarios that assess interactions of urban areas with ecosystem services at global and large spatial scales and there are only a few, mostly scenario-based, regional studies from developed countries (Deal & Pallathucheril, 2009; Eigenbrod *et al.*, 2011; Norman *et al.*, 2010; Pickard *et al.*, 2017). Virtual water import/export has been explored under future scenarios under climate change, stressing local water losses due to trade links (Konar *et al.*, 2013; see also Chapter 5). For instance, continued increased consumption of meat or milk in China would have negative consequences on the virtual water imported by the country (Zhuo *et al.*, 2016), as well as higher greenhouse gas emissions and land use in milk exporting regions (Bai *et al.*, 2018). Results from the systematic literature review regarding future trends of various NCP in different world regions and the interlinkages between them do not show clear trends for many NCP because of the limited number of studies **(Figure 4.3.6)**. Mixed trends prevail for regulating NCP in most parts of the world, with slightly more increasing trends in North America, Europe, and Australia. Material NCP are expected to mainly decrease in Central America, in Southeast Asia and Australia, stabilize in South America, South Asia and East Asia; a higher proportion of increasing material NCP are expected in Europe and North America. Not much data on non-material NCP is available but positive trends in Africa and Asia could emerge, while in South America the expected trends were mostly negative.

In addition to the systematic literature review, we reviewed the IPBES regional assessments (IPBES 2018a, b, c, d) for relevant information of future trends of telecoupled interactions.

The IPBES regional assessment for Europe and Central Asia (IPBES, 2018i) highlights a variable but generally decreasing supply of regulating NCP in Europe (Harrison

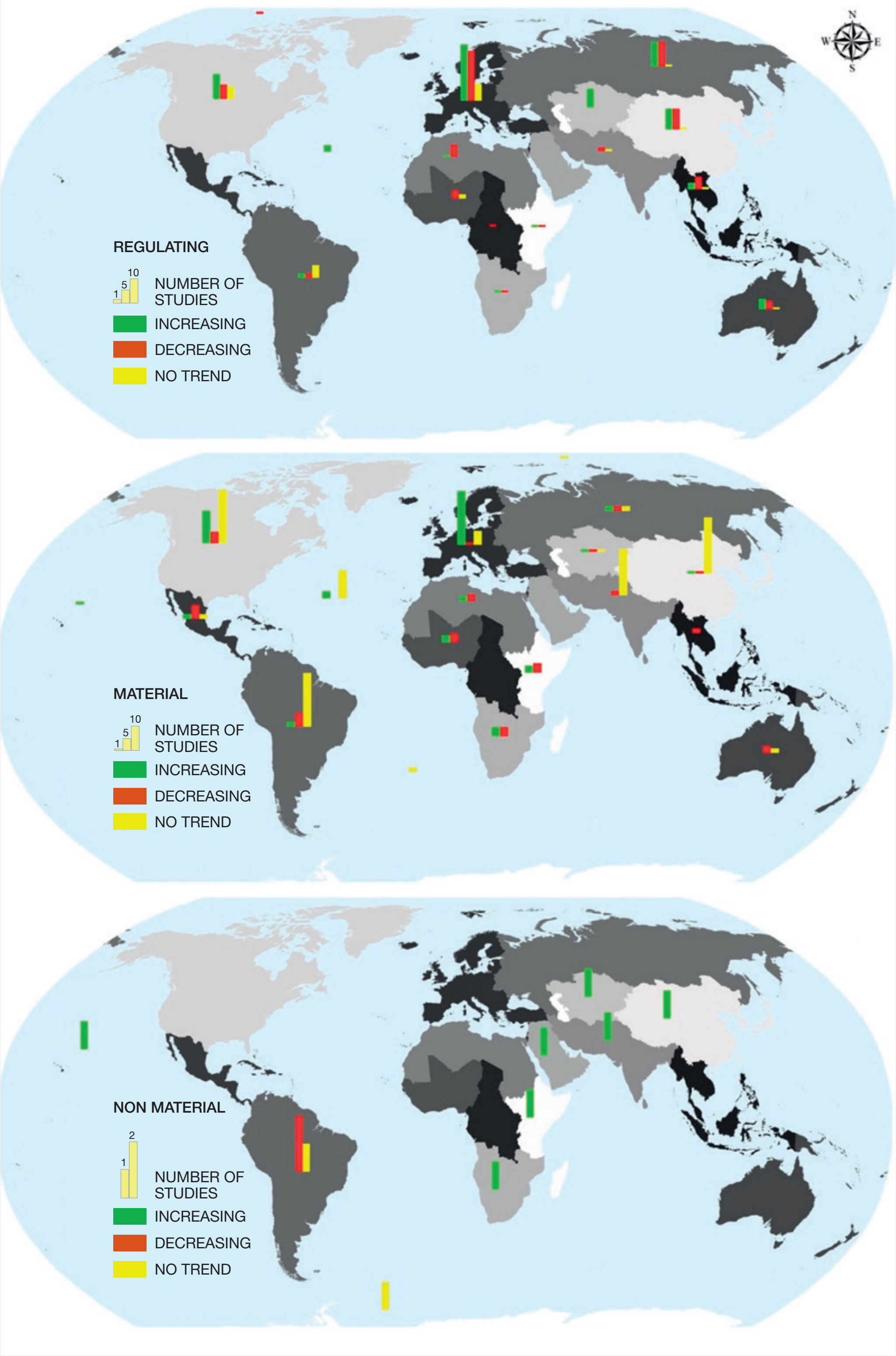


Figure ④ ③ ⑥ **Future trends of NCP in the different world regions.**

The height of the bars indicates the number of studies. The color of the bars shows the sign of future trend of NCP in the different world regions (IPBES regions shown in grey scale). Results are based on the systematic literature review of future scenarios (Appendix A4.1.1) at the continental scale. Only the studies with an explicit distinction of NCP trends between countries or regions were selected.

et al., 2018). Hazard regulation, climate regulation, water quality and quantity regulation show stable or increasing trends, whereas regulation of freshwater quantity, location and timing decreases, especially in Southern Europe. Pollination and pest regulation indicate mixed trends. Regarding material NCP, the results vary across subregions. An increase of food and feed is expected in western Europe due to increasing imports from other world regions (Dunford *et al.*, 2015). Eastern Europe and Russia show increasing trends in food production, due to the increase in suitability for food production following climate change (Zabel *et al.*, 2014). Information on non-material NCP is scarce (Harrison *et al.*, 2018).

The demand of material NCP in Europe, especially food and feed, materials and energy could increase up to 1.5-2 times, which not only means an increase in material NCP but will have considerable trade-offs with biodiversity and regulating NCP (Harrison *et al.*, 2018). According to the BAU scenario, food production will be the economic sector with the largest impact on biodiversity, possibly contributing to 60-70% of terrestrial biodiversity loss and 50% in freshwater systems (Kok *et al.*, 2014; van Vuuren *et al.*, 2015). Other scenarios, such as the global technology, decentralized solutions and consumption change would result in preventing more than half of the loss of the biodiversity that is projected for 2050. Other models show that domestic greenhouse emissions can be reduced affordably by 40% in 2030, but would require strong policies and binding targets, and possibly the use of biofuels, which have associated negative effects on biodiversity (Harrison *et al.*, 2018).

In Africa a lack of studies that assess the future of NCP is apparent and the few existing ones focus on Southern and East Africa (Biggs *et al.*, 2018). The systematic literature review shows that in different regions of Africa, the demand for food and feed will lead to an increase of this NCP, despite the pressure arising in many regions from climate variability and change (Palazzo *et al.*, 2017). Scenarios show that increased water stress will have most adverse effects on food production, as areas suitable for agriculture along the margins of semi-arid and arid areas are expected to decrease (Biggs *et al.*, 2018). An estimated 600,000 km^2 of arable land could be lost with 800 million people facing physical water scarcity. Rising sea levels will pose threats to Gambia around to the Gulf of Guinea and a predicted band of desiccation will wrap around the Congo Basin from the Gambia to Angola (Biggs *et al.*, 2018). Given the general trade-off between material and regulating NCP, a decrease in the supply of regulating NCP is expected. In Sub-Saharan Africa, bans on food imports would negatively impact poverty (Bren d'Amour *et al.*, 2016).

Existing scenarios with information for NCP in the Americas focus on the strong competition among land uses, primarily agricultural lands and natural land cover (Klatt *et al.*, 2018). The demand for food and feed will increase in the future with strong trade-offs for regulating NCP (e.g., water quality, increased greenhouse gas emissions, disruptions of natural pest control, pollination, and fertility and nutrient cycling; Diaz & Rosenberg, 2008; Matson *et al.*, 1997). Co-benefits may occur, like e.g., incorporating biodiversity in agricultural production systems (Baulcombe *et al.*, 2009; Chappell & LaValle, 2011; Clay *et al.*, 2011; de Schutter, 2011; Perfecto & Vandermeer, 2010). The supply of regulating NCP provided by natural ecosystem decreases under all scenarios (even under conservation scenarios), especially through tropical deforestation in Latin America, which is projected to continue. A similar pattern can be observed also for other ecosystems, like tundra, mangroves or wetlands. The decrease in supply of regulating NCP means that the tundra may convert from a carbon sink into a carbon source under the temperature increase that thaws the permafrost, leading to a feedback to accelerated climate change and sea level rises. The same applies for the prevention of soil erosion, coastal protection and fisheries support of mangroves. Also, the regulating services of wetlands may get traded by agricultural productions under the strong increase of population and other market forces. An example is the Amazon forest, where especially cattle ranching together with agriculture leads to deforestation, leading to a synergistic drying up of large parts of the watershed due to climate change (Klatt *et al.*, 2018).

In the Asia-Pacific region, expansion of urban industrial environments, consumption patterns and transformation of agriculture in favor of high yielding varieties and cash crops are the main drivers for changes in NCP, considering the current rate of human population growth (Gundimeda *et al.*, 2018). The demand for material NCP is projected to increase, especially for food and feed in Southeast Asia and South Asia, leading to deforestation for monocrop plantations of oil palm, rubber or timber trees. This may lead to a decrease in the supply of some regulating NCP, and natural habitats in the Asia Pacific Regions are likely to be adversely affected in the coming decades (Gundimeda *et al.*, 2018). Telecouplings are very pronounced, especially within Southeast Asia (e.g. Vietnam- Laos) and between mainland Southeast Asia and North Asia, as between Southeast Asia and Latin America and Africa. Regarding other regulating NCP the results are mixed with increases and decreases in all subregions (IPBES, 2018h).

4.4 PLAUSIBLE FUTURES FOR GOOD QUALITY OF LIFE

4.4.1 Linking good quality of Life to nature and nature's contributions to people

Global scenarios of biodiversity and ecosystem services have paid scarce attention to plausible futures for people's good quality of life (GQL), relative to those for nature and nature's contributions to people (but see Butler & Oluoch-Kosura, 2006). This gap is further pronounced for the analysis of future trends for the quality of life of Indigenous Peoples and Local Communities (IPLCs), who have been addressed typically at local and subnational scales rather than at the regional to global scales. However, a recent assessment of scenarios and models of ecosystem services and biodiversity brought to light some of the plausible futures of GQL (IPBES, 2016b), while earlier assessments highlighted the dependency of human beings on ecosystems for well-being and socio-economic development (MA, 2005; UK National Ecosystem Assessment, 2011).

To complement these efforts, in this section we seek to show how good quality of life has been integrated in the assessment of plausible futures of nature and nature's contributions to people. To this end, we address how eleven key material and non-material dimensions of GQL (see also Chapter 1) are expected to evolve under the different archetype scenarios, and highlight the role of access, social values and other factors mediating the relationship between nature's contributions to people and good quality of life.

4.4.1.1 Key Dimensions of good quality of life and their links to nature and nature's contributions to people

4.4.1.1.1 Material dimension of good quality of life

In future scenarios governed by market forces (e.g., economic optimism, business-as-usual; see Section 4.1), multiple dimensions of good quality of life (GQL), both material and non-material, can be expected to decline **(Figure 4.4.1)**. These projections are based on narratives associated with specific archetype scenarios, with numeric scores above zero indicating an anticipated positive (increased) GQL for the selected indicator, and negative indicating a decline. Projected declines are particularly pronounced for material indicators relative to livelihood and income security. The regional competition scenario, in particular, is assumed to be associated with the lowest expected GQL outcomes. On the other hand, the regional sustainability and reformed economic markets scenarios are expected to result in improved GQL outcomes across a large cross-section of material and non-material indicators. Overall, the global sustainable development and regional sustainability scenarios are associated with the most desirable GQL outcomes. Scenarios of direct and indirect drivers of change are expected to have regionally differentiated impacts on GQL, including where Indigenous Peoples and Local Communities (IPLCs) are located (see examples below). Many IPLCs are found in protected areas and indigenous areas where dimensions of a GQL such as food and energy security play out in context-specific ways. Indirect drivers of change such as climate mitigation policy (e.g., REDD+) disproportionately impact the possible trajectories towards achieving GQL by IPLCs (sections 4.1.4, 4.1.5).

Food and nutritional security

The 2018 annual report on the State of Food Security (http://www.fao.org/state-of-food-security-nutrition/en/), assessed that world hunger is on the rise again with the number of undernourished people having increased to an estimated 821 million (2017), compared with 804 million in 2016 and 784 million in 2015, although still below the 900 million reported in 2000. Future projections raise important concerns about global food security and indicate widespread disparity in its outcomes, estimating that between 5 million and 170 million people will be at risk of hunger by 2080 (Schmidhuber & Tubiello, 2007). With continuing urbanization of the global population (see section 4.3.3), much of this burden can be anticipated to be borne by the urban poor, especially in the developing south. Food security is related to cultural rights and human rights, and to processes of community change such as out-migration and livelihood shifts (e.g., changing migration patterns may leave fewer young people to hunt and fish, and elders often too old to engage in these activities). Access to resources (including financial resources) are also needed to participate in traditional activities securing access to food. Future food security scenarios refer to at least one of the four key dimensions of food security: availability, access, utilization and stability (FAO, 1998). All four dimensions are expected to be affected by climate change, although only food availability is commonly considered by simulation studies with a wide projected range of impacts across regions and time depending on the socio-economic context (Brown & Funk, 2008; Schmidhuber & Tubiello, 2007). The systematic literature review conducted in this chapter (Appendix A4.1.1) portends strong negative trends for food security in future scenarios **(Figure 4.4.3)**. The IPCC Special Report on Emission Scenarios (SRES) depicted cereal production, cereal prices and food security under three conditions: no climate change, climate change with CO_2 fertilization effects, and climate change without

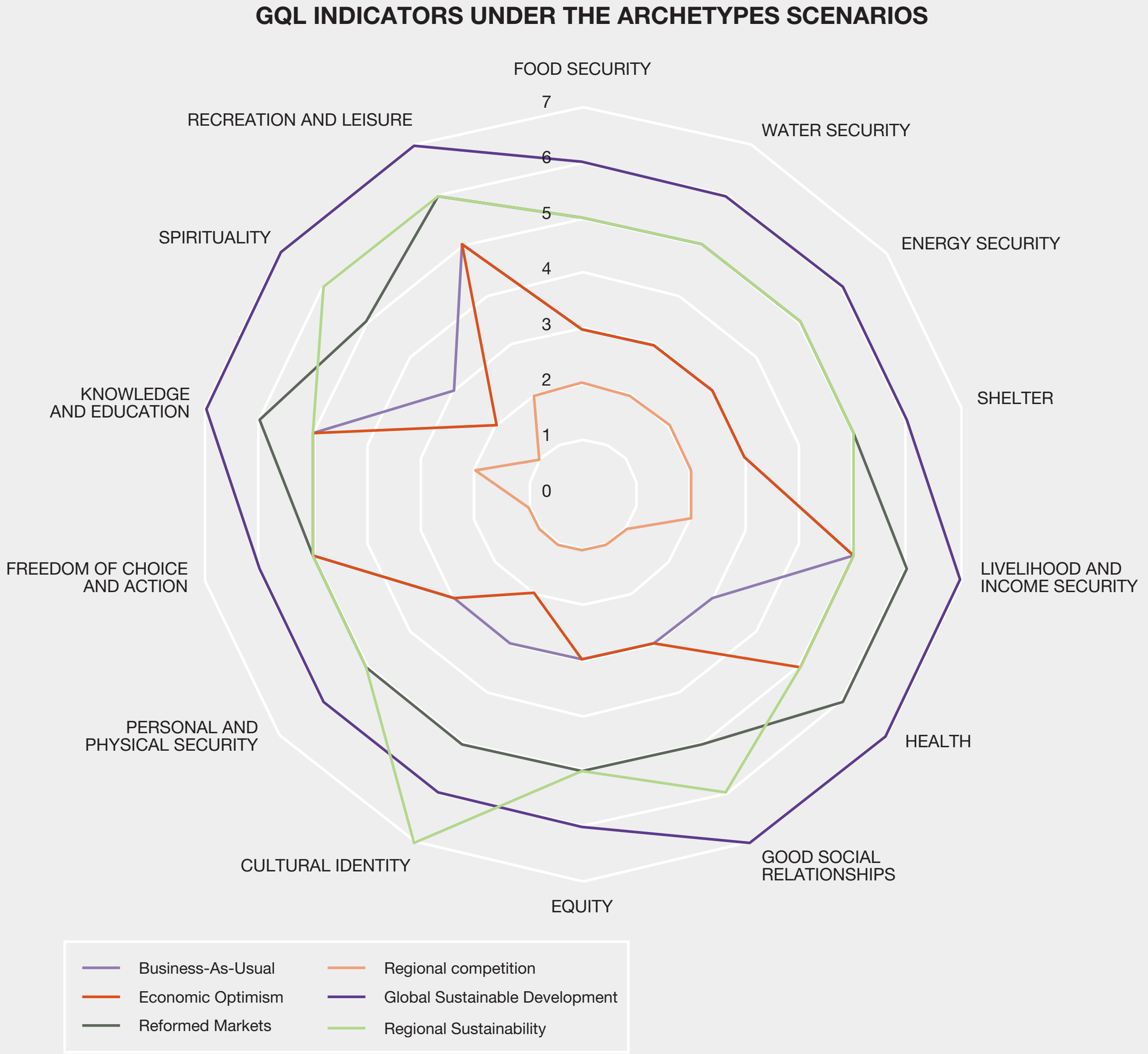


Figure ④ ④ ① **Dimensions of Good Quality of Life (GQL) under archetype scenarios based on the narratives of these scenarios, taking also into account Table 6.3 of the IPBES methodological assessment of scenarios and models of biodiversity and ecosystem services (Cheung *et al.*, 2016).**

The numerical scale (-2.5 to 2.5) refers to the progress from "very negative" status = -2.5 to "very positive" status = 2.5 in the corresponding GQL indicator. More detailed information is found in the Appendix, Table A4.4.1.

CO_2 fertilization effects (Parry *et al.*, 2004). Under the assumption of no climate change and increasing yields due to technological change, it was estimated that cereal prices would increase due to an increase in global income. With climate change, food shortages were expected to drive up food prices. The MA scenarios projected an increase in total and per capita food production but variation in food prices, calorie availability and child malnutrition were also to be expected (Carpenter *et al.*, 2006). More recent work agrees that the impact of climate change on food security varies across time, space and subpopulations. For instance, food insecurity is expected to be more severe in the Amazon floodplains (Oviedo *et al.*, 2016; Vogt *et al.*, 2016), polar regions such as the Arctic Bay (Pearce *et al.*, 2015) and the Pacific Islands (McMillen *et al.*, 2014). Small-scale farming, fishing and other communities that depend directly on local environments for food production (McDowell & Hess, 2012) especially in developing countries, indigenous communities (Huntington *et al.*, 2016), or First Nations (Golden *et al.*, 2015) are particularly vulnerable to climate-related food insecurity. A synthesis across a number of international assessments integrated and grouped factors impacting food security **(Figure 4.4.2)** and identified that in these assessments the individual factors underpinning food security were mostly not linked to other relevant factors, i.e. indicating substantial gaps in our understanding of the food

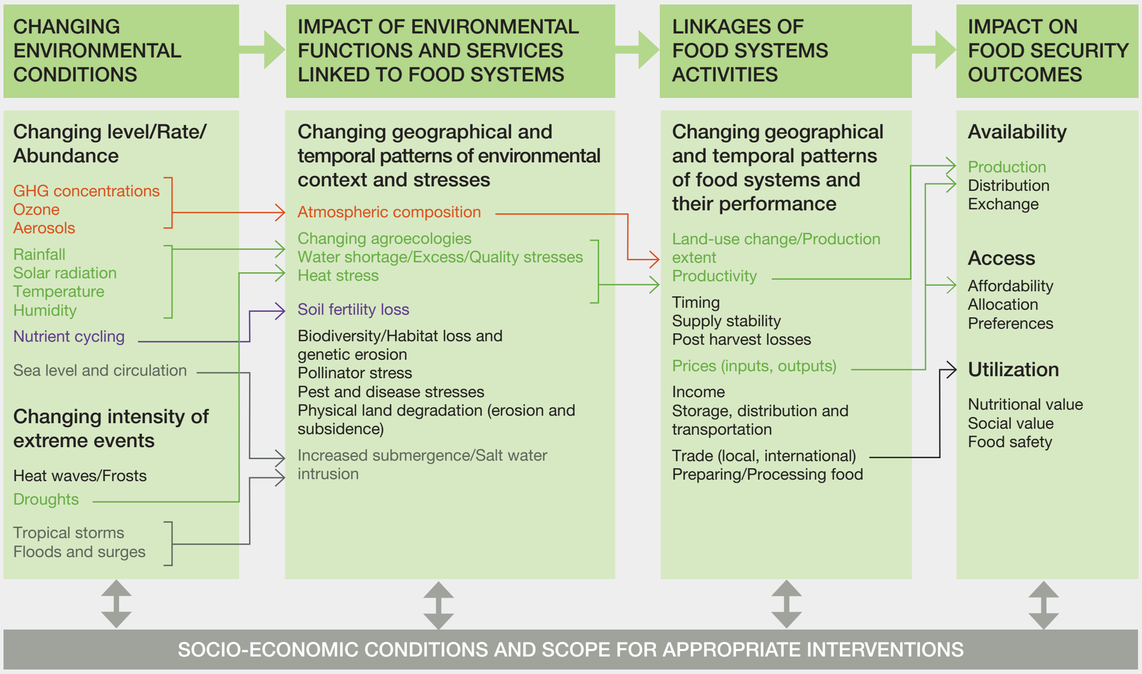


Figure ④ ④ ② **Food Security impacts pathways in different global environmental assessment studies.**

Arrows indicate links in the analyses from environmental conditions (left) to food security outcomes (right). Source: Wood *et al.* (2010).

system, in particular how natural and socioeconomic system components interact.

Water security

Regular access to clean water is a growing concern across multiple regions of the world, affecting two-thirds of the population (see 4.3.2.1). Water scarcity is strongly driven by behaviour driving overconsumption, infrastructure, and climate change. Climate projections indicate that a global temperature increase of 3-4°C could cause altered run-off patterns and glacial melt that will force an additional 1.8 billion people to live in a water scarce environment by 2080 (UNDP, 2007). Other drivers such as rising populations in flood-prone lands, climate change, deforestation, loss of wetlands and rising sea levels are expected to increase the number of people vulnerable to floods to 2 billion in 2050 (WWAP, 2012). Drylands are particularly vulnerable to changes in rainfall (Carpenter *et al.*, 2006), and with climate change, drought impacts are anticipated to intensify across increasing extents of the world's drylands (IPCC, 2013). The world's megacities are already facing increasingly frequent and acute water shortages, which can be expected to worsen in the future (Li et al 2015a). Similarly, in coastal regions, decreases in precipitation and fresh water supplies, along with projected increases in sea level, sea surface and air temperatures, and ocean acidification are projected to have major negative effects on water security for societies (McMillen *et al.*, 2014). The 'fresh water planetary boundary' is approaching rapidly (Dearing *et al.*, 2014; Rockström *et al.*, 2009), and sustainability of water use will likely be difficult to achieve in the near future (Gosling & Arnell, 2016). According to the results of the systematic literature review, water security indicators show negative trends in global and continental scale scenarios **(Figure 4.4.3)**.

Energy security

Ensuring the global population's access to modern and sustainable energy services in consideration of environmental integrity remains a major challenge for policymakers and practitioners worldwide. According to the systematic literature review, energy security derived from nature appears to be the only indicator with no identified negative trends in global scale scenarios **(Figure 4.4.3)**. However, scenarios such as decarbonisation ones, appear to also provide other benefits in addition such as lower energy market risks (Jewell *et al.*, 2014). However, energy security faces several other challenges. Energy security has both producer and consumer aspects (UNDP, 2004). Access to sustainable energy, which can include bioenergy sources, is critical in enabling people to meet essential needs linked with good quality of life as energy security encompasses availability, affordability, efficiency and environmental acceptability. The development of energy models in the 1970s in response to the energy crisis has provided relevant insights into the consumption and management patterns towards a sustainable energy for all future. On the other hand, current uneven global consumption coupled with the dearth of studies and quantitative data on energy use, especially from developing economies, presents a challenge for developing effective forecasting models. Scenarios based on non-linear energy consumption consider limiting overconsumption can keep 2040 energy consumption at 2010 levels, while increasing energy-for-life efficiency can keep 2040 energy use at 2010 levels (Pasten & Santamarina, 2012).

Livelihood and income security

While global scenarios lack sufficient attention to livelihood impacts, the results of the systematic literature review indicate regionally differentiated negative trends projected for livelihood and income security in the future **(Figure 4.4.3)**. Employment and incomes derived from nature are indicative for value derived in cash or direct use that impact good quality of life. Nature-based income, as part of environmental income, includes that derived from resources such as fish, timber, and non-timber forest products such as fuel wood, game, medicinals, fruits and other foods, and materials for handicrafts or art. It also includes income from nature-based tourism, as well as payments that rural landowners might receive for environmental services such as carbon storage or preservation of watershed functions. Also included is income from aquaculture as well as from small-scale agriculture, including commodity crops, home gardens, and large and small livestock. Nature-based livelihoods may become precarious with intensifying future trends in environmental change and its drivers (Hopping *et al.*, 2016). Climate change-induced depletion of household assets may have especially negative impacts on the future welfare of populations already fighting poverty. For example, farmers in Sub-Saharan Africa will spend an increasingly high share of their income on securing basic needs such as food, while housing and related needs also intensify (Enfors & Gordon, 2008).

Health

The future of biodiversity and ecosystem services is inextricably linked to that of human health and well-being, for instance, through supporting healthy diets to mitigating the health impacts of climate impacts or pollution. Many health benefits are related to the conservation or use of specific elements of biodiversity such as species or genetic resources. Indigenous communities increasingly anticipate, and are impacted by, changes to traditional practices and pathways of food, toxicity impacts from distant (e.g., pesticides) and local (e.g. mining) sources, hunting and gathering of medicinal plants, and experience their consequences for local diets and resistance to diseases, as exemplified in Queensland Australia (McIntyre-Tamwoy *et al.*, 2013), by Arctic Bay Inuit (Ford *et al.*, 2006), and across North American and Russian indigenous populations. As

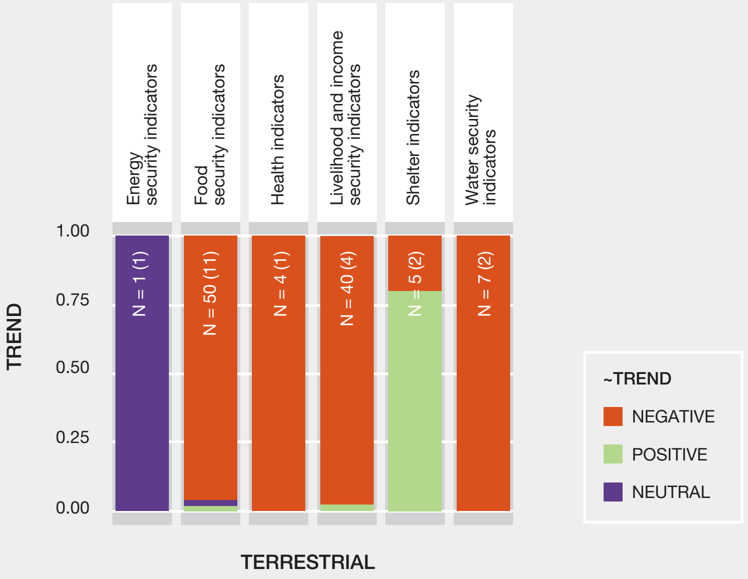


Figure 4 4 3 **Trends in selected indicators of GQL in terrestrial ecosystems. Colors indicate the value trend of the indicator.**

"N" indicates the number of results reported per facet, with the number of papers indicated in parentheses.

environmental hazards and extreme weather events increase in frequency, intensity or duration, they are expected to have increasingly visible consequences for health (Bai *et al.*, 2016).

Projected increases in the production of biofuel crops, in particular in case of woody bioenergy species (eucalypt, poplar) which emit more isoprene than traditional crops, suggest important impacts on ground-level ozone concentrations, and consequently on human health and mortality (Ashworth *et al.*, 2013). On the other hand, projected reductions of anthropogenic air pollutants point towards a widespread decline of small aerosol particles; projected future wildfires may not alter this general trend except for some parts of the wildfire season (Knorr *et al.*, 2017). Projected environmental changes are also expected to impact the prevalence of vector borne diseases such as malaria. Of the four MA scenarios, health under the "techno garden" scenario was expected to ameliorate due to technological advancements (Butler & Oluoch-Kosura, 2006; Carpenter *et al.*, 2006). Likewise, climate change under the five shared socio-economic pathways affects health outcomes (Ebi, 2014). Some health indicators can be expected to decline according to the systematic literature review **(Figure 4.4.3)**, however, more comprehensive global scenarios need to address various dimensions of health impacts.

4.4.1.1.2 Non-material dimensions of good quality of life

Along with material needs, human well-being depends profoundly on non-material and experiential factors (Butler & Oluoch-Kosura, 2006). However, narratives around good quality of life in global scenarios typically ignore such non-material dimensions which include but are not limited to: social relations, equity, cultural identity, values, security, recreation, knowledge and education, spirituality and religion, and freedom of choice and action.

Good social relations

Social relations refer to the degree of influence, respect, co-operation, and conflict that exists between individuals and groups (MA, 2005). Good social relations underlie the development of strong institutions and collective action, providing routes for sustainable use and management of nature and nature's contribution to people. The natural environment has important influences not only on individual well-being, but social relations as well (Hartig *et al.*, 2014). Good social relations also include mutual respect, social cohesion, and good gender and family relations. The linkages between good quality of life, nature and nature's contribution to people were explicitly identified in the Millennium Ecosystem Assessment, with an emphasis on cultural and spiritual values (MA, 2005). Even though the world is more connected than ever before, social differentiation remains a major constraint to social relations at multiple scales and in many cases is closely associated with inequality in access to nature and natural resources. Thus, it is crucial to address disparities among stakeholders in and across socio-ecological systems and the role of social relations in negotiating such disparities, in order to more fairly and equitably address how nature and NCP can be leveraged to promote a good quality of life. The degradation of ecosystems, highly valued for their aesthetic, recreational, or spiritual benefits, can also damage social relations, by introducing or exacerbating disparities among social groups and reducing the bonding value of shared experience, including resentment towards and resistance against groups that disproportionately profit from their damage. While global scenarios of future trends in social relations are elusive, climate and land-use changes in the future are highly likely to accentuate social inequity in use of and access to resources, in the absence of changes in governance arrangements to address current disparities.

Equity

Equity broadly concerns an even distribution of nature's contributions to people, and access to natural resources and rights (see also section 4.4.3). Typically three dimensions of equity are considered: (1) distribution, (2) procedure, and (3) context, access and power (McDermott *et al.*, 2013). Equity concerns evidence of parity in processes and outcomes across gender, age, race and ethnicity, income and other social indicators or axes of difference. It is fundamental to human rights, including the rights of IPLCs (see also **Box 4.4.1**), and implicitly influence nature, its contributions to people and good quality of life (Breslow *et al.*, 2016). Equity addresses fairness or justice in the way people are treated. In principle, equity concerns pertain to at least three domains –international, intra-country, and inter-generational. Social justice (equity) constitutes one of the three pillars of sustainable development, along with economic prosperity (development) and ecological integrity (sustainability) (Banuri *et al.*, 2001). Equity may increase in scenarios where the consumption of material goods is reduced relative to that of services and intangibles, such as the new welfare scenario (Sessa & Ricci, 2014). Equity is also expected to increase in global sustainable development scenarios such as SSP1, B1 (A1T), B2, sustainability first, global orchestration and techno garden, and some economic optimism scenarios such as SSP5. In regional competition scenarios such as SSP3/4, A2, security first and order from strength, equity is expected to be low (see section 4.1).

Cultural identity

Cultural identity includes concerns related to the terms, language, activities and practices that embody the relationships of people and nature. The cultural identity of IPLCs is particularly linked to long-term material and non-material relationships to nature and place, with direct and sustained physical and experiential interactions (e.g., see section 4.3.2.3 above). As indicated earlier, among the direct and indirect drivers of changes to such interactions, and to fundamental aspects of IPLCs cultural identity, are urbanization, climate change, demographic changes, technology, psycho-social or cultural factors, and health and development. Future threats to biodiversity and ecosystem services also constitute imminent challenges to the cultural identity of communities, particularly when faced with environmental degradation. For example, "blue-ice," as a term inherent to First Nation languages and as the material formation on lakes and rivers, links transportation to access to food and energy. It is thus central to First Nations' cultural identity and traditional activities, and their future well-being (Golden *et al.*, 2015). Such relations are at once material and symbolic. As section 4.3.2.3 also highlights, symbolic meaning is intimately tied to spiritual, religious and cultural identity, and strongly shapes social cohesion, and future trends in these relations are central to IPLC futures.

Personal and physical security

Future climate change poses physical risks with implications for human safety and security. Such risks emanate from multiple dimensions, including those linked to increased exposure to episodic stress (e.g., extreme climate events) as well as chronic pressures (e.g. related to warming temperatures and sea level change). For instance, climate change scenarios in the Great Barrier Reef indicate marked declines in security that accompany declines in ecosystem services, along with indicators of equity, education, health and shelter (Bohensky *et al.*, 2011a). In other examples, projections of future population dynamics have indicated that more people may live in areas that are prone to both floods and wildfires in the future (Knorr *et al.*, 2016). In northern regions, among other risks, for some populations, traveling on thinning ice in winter is becoming more dangerous, restricting movement of people and goods (Ford *et al.*, 2006).

Recreation and leisure

There is considerable research from environmental psychology on the human health and well-being benefits from recreation in nature (Barton & Pretty, 2010; Marselle *et al.*, 2014). The Millennium Assessment Technogarden scenario (see section 4.1) argues for the multifunctionality of land-use including recreational opportunities, seen as an affordable luxury in e.g., the Order from Strength scenario (MA, 2005; see also Appendix 4.4). Similarly, the SRES B1 (A1T) mentions the preservation of recreational spaces (Nakicenovic *et al.*, 2000; see also Appendix 4.4). Loss of coral reefs under the RCP2.6 and RCP8.5 scenarios (section 4.2.2.2.2) could cost between U.S. $1.9 billion and U.S. $12 billion in lost tourism revenues per year, respectively (Gattuso *et al.*, 2015). The loss of recreational areas such as camping sites is signaled as a regional concern by indigenous participants in case studies in Australia (McIntyre-Tamwoy *et al.*, 2013).

Knowledge and Education

Knowledge and education related to biodiversity and ecosystem services are essential for ensuring good quality of life. The taxonomic records of world fauna and flora indicate 8.7 million known species (Mora *et al.*, 2011), which represent only a fraction of the species that may exist (WRI *et al.*, 1992), indicating a large knowledge gap on fundamental aspects of biodiversity. It has been estimated that 86% of existing species on Earth and 91% of species in the ocean still await description (Mora *et al.*, 2011). Much of the knowledge used in scenarios of biodiversity and ecosystem services is derived from biology, ecology and related disciplines.

Yet, a variety of conceptualizations of biodiversity are embedded in local knowledge and cultural memories directly relevant to regional and global resource and food production systems (Nazarea, 2006), but poorly represented in future scenarios. Additional perspectives could be derived from work on human cognition, decision-making, and behavior. For example, ethnobiology of agricultural diversity, cultural ecology of plant genetic resources, participatory conservation, politics of genetic resources, and legal dimensions of biodiversity conservation are very poorly represented in scenario development. The role of education has been to some extent explored in global scenarios. Specifically, the narratives of scenarios SSP1 and SSP5 assume that the human capital component of education is highest compared to SSP2, SSP3 and SSP4 (KC & Lutz, 2017). Schools play an important role in educating pupils and students to be active and responsible towards the environment, and the challenge of biodiversity conservation (Torkar, 2016; Ulbrich *et al.*, 2010).

Spirituality, religion

A number of studies highlight the ways in which spirituality is related to good quality of life. Spirituality has been considered in a variety of ways, ranging from the traditional understandings of spirituality as an expression of religiosity in search of the sacred, to humanistic views of spirituality not specifically anchored in religion, or at least, ecclesiastical religion. Fisher (2011) noted that the spiritual health of individuals has four important domains: personal, communal, environmental and transcendental. Many religions emphasize a deep connection or oneness with nature, including Hinduism, Buddhism, Jainism, Christianity and Islam. For example, in India, patches of forest frequently constitute sacred groves of varying sizes, which are communally protected with significant spiritual connotations.

The rapid retreat of the Gangotri Glacier, the sacred source of the Ganges, is alarming for Hindu religious practitioners (Verschuuren *et al.*, 2010). The landscape that surrounds sacred groves has a vital influence on biodiversity within them (Bhagwat *et al.*, 2005). Similarly, sacred sites in Italy often display ecological features that highlight their important conservation role (Frascaroli, 2013). These sacred places are, symbolically, repositories of knowledge of our planet as 'home.' Our relationship with nature and GQL, where the spirit of nature and culture meet, and are additionally memorialized and maintained by rituals and festivities performed there. However, most of the current archetype scenarios of biodiversity and ecosystem services fail to incorporate the spiritual and cultural significance of nature.

Freedom of choice and action

Freedom emphasizes a person's social, political, economic, and personal rights, and whether one is actually able to exercise these rights. Freedom of choice and action is a vital pre-requisite to GQL. In practical terms, freedom can promote or inhibit access to nature and its multiple benefits needed to sustain life. Human and natural constraints prevent different groups of people around the world from having or exercising freedom of choice and action to access nature and its benefits needed for good quality of life. Thus, even though nature and its contributions to good quality of life may be abundant in certain areas, lack of freedom may impede access. Projected changes to climate, biodiversity and ecosystem services can be expected to directly impact social access to nature and its benefits. In addition, future changes can strongly impact the institutions shaping freedom and choice. For instance, experience has shown that sociopolitical institutions and environmental regulatory regimes tend to favour certain groups over others. In the Doñana protected area from Southern Spain, freedom of action and choice is completely reduced in a future scenario of market liberalization (Palomo *et al.*, 2011). Similar trade-offs with GQL are evident in the varying degrees of environmental protections at the global scale. For instance, different IUCN categories in protected areas, from the most stringent preservationist approaches excluding human use, to the more integrated protection categories incorporating some (sustainable) use, have vastly different implications for GQL in different communities living in those regions.

4.4.1.2 Good quality of life across worldviews and knowledge systems

GQL conceptualizations across worldviews and knowledge systems vary considerably due in part to values, beliefs and worldviews, as well as social and political contexts. What GQL entails is highly dependent on place, time and culture, with different societies espousing different views of their relationships with nature and placing varying emphasis on collective versus individual rights, or the material versus the spiritual domain. Understanding and appreciating plausible GQL scenarios require an integrative assessment of subjective and objective approaches and indicators for quality of life, including quantitative or qualitative social and economic measures (Cummins *et al.*, 2003; Diener *et al.*, 1999; Easterlin, 2003; Haas, 1999). Over the past half century, increasing research and policy attention has been directed to socio-ecological concerns relevant to Indigenous People and Local Communities (IPLCs) (e.g., **Box 4.4.1**), with recognition of long histories and ongoing processes of exclusion and marginalization of IPLCs in ecosystem and biodiversity conservation and management across socio-ecological regions. The IPBES framework acknowledges the varying perspectives of GQL across knowledge systems, cultures and societies (Díaz *et al.*, 2015).

While indigenous worldviews differ from one community to another, indigenous understandings of well-being are also frequently intertwined with understandings of nature; the relationship between people and their environment happens not only at a cognitive level. In many societies, "prestige and satisfaction are gained through relationships and generosity rather than in accumulation of personal wealth. A good life is one spent in service to one's community, in living in balance with the other lifeforms of one's homeplace. Responsibilities extend not just to the present, but to many generations into the future" (Turner & Clifton, 2009). Different understandings also exist around the notion of 'time'. In Iñupiaq and Siberian Yupik culture, for instance, it is important for hunters to avoid speculating about the future, reflecting the belief that one should be humble about one's abilities to predict it, and not expect any one particular outcome over another (Voorhees *et al.*, 2014). Addressing quality of life under different plausible futures will benefit from bridging indigenous and local epistemologies with scientific knowledge systems (Tengö *et al.*, 2017), such as initiatives addressing mitigation and adaptation from a local perspective (UNU-IAS & IGES, 2015).

4.4.2 Linking good quality of life to nature and nature's contributions to people across future scenarios

4.4.2.1 Mediating factors of future GQL and NCP

Future quality of life and its relation to nature and its contributions to Ppeople (NCP) is expected to be mediated by a bundle of overlapping factors across socio-ecological systems at local and global levels, from the individual or the household to the system **(Figure 4.4 4)**. These mediating factors are fundamental to shaping the productive base of a society, including substitutable capital assets, i.e. natural,

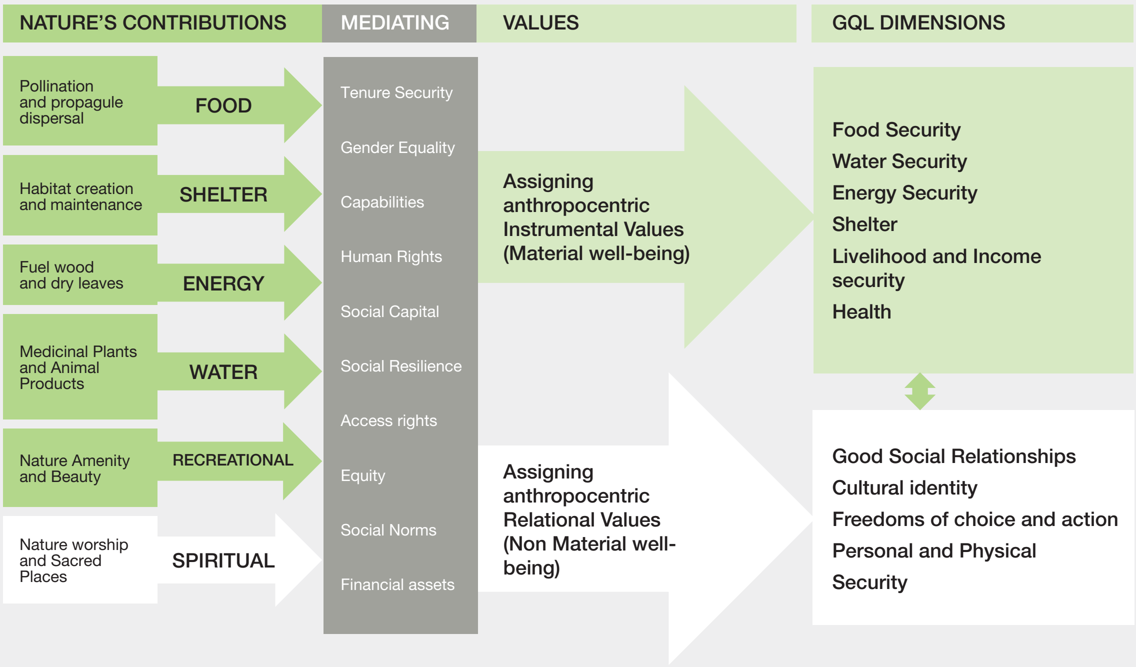


Figure ④ ④ ④ **Conceptual model of mediated relationships between nature's contributions to people and good quality of life (GQL).**

produced, and human capital (Duraiappah *et al.*, 2014). They are akin to indirect drivers of changes to nature, NCP and GQL, and include tenure security (e.g., use and access rights), equity concerns, power relations, formal and informal institutions and human rights, technology access, financial assets, and social capital and social resilience (Horcea-Milcu, 2015; Shapiro & Báldi, 2014; Spangenberg *et al.*, 2014). However, inequities, political challenges and distributional issues are seldom discussed by scenarios considering implications for GQL.

Social groups have distinct ways to derive well-being from NCP, as a result of a range of interlinked mediating factors (Horcea-Milcu, 2015). For example, policies such as the European Common Agriculture Policy rural development program of agri-environment schemes may increase nature's contributions to people, but because it does not holistically engage with mediating factors it will not equitably increase access to benefits (Horcea-Milcu, 2015). Although people's values and attitudes are crucial in shaping the future, they are rarely central to scenario exercises. Novel methods, such as the three horizons approach (Sharpe *et al.*, 2016) have been developed to fully integrate people's worldviews into scenario planning, however transcendental values held by the social groups are only beginning to be considered (Kass *et al.*, 2011). For example, the ethnographic futures framework focuses on how changes in the natural environment take place through human agency and how society will act as recipient in the future (Kass *et al.*, 2011). Importantly, the process of elaborating scenarios is increasingly taking into account participatory approaches and corresponding value negotiations around the meaning of good quality of life. Consequently, ethical questions emerge regarding how to build scenarios so that local knowledge and IPLCs are not coopted in ways that may exacerbate processes of their social marginalization (but see also **Box 4.4.1**).

How mediating factors may be expected to change in magnitude and direction across different archetype scenarios remains to be explored. Scenarios of regional sustainability seem more suited for mitigating the negative influence of mediating factors (Hanspach *et al.*, 2014). Mismatches among mediating factors, nature and NCP may pose challenges. For instance, Duraiappah *et al.* (2014) identified mismatches of individuals' values (e.g., of ecosystem services within different social contexts), mis-matches in ecosystem services and ecosystem scales (at which levels of biodiversity, ecosystem processes and functions operate to produce the bundle of provisioning, regulating, and cultural services), and mis-matches of institutions (those that account for spatial, temporal, and functional fit in managing ecosystem services).

The way NCP components will be filtered and transformed to GQL components and reach beneficiaries such as individual, social groups or societies will be highly influenced by mediating factors such as: access arrangements, assets, institutions, values and norms. One avenue to incorporate this variability is integrating more participatory, deliberative or transdisciplinary processes into scenario building endeavors towards improved considerations of GQL in its variety of components, whether material or non-material, of local or global concerns. Storylines of socio-economic development used in global scenarios include few indicators of GQL, typically predicated on its material aspects. Given these limitations, lessons learnt from the current assessment is that indicators of GQL in global scenarios generally improve in the future in the "global sustainability", "regional sustainability", and "economic optimism" scenario archetypes. However, continued degradation of nature and non-provisioning NCP in the "Economic optimism" scenarios suggests that the decoupling of GQL from Nature and non-provisioning NCP that is often currently observed could potentially continue into the future. Indicators of GQL have the poorest future trajectories in the "regional competition" scenarios and do only slightly better in "business-as-usual" scenarios at the global scale.

4.4.2.2 Future scenarios of GQL and NCP

Key characteristics of GQL indicators are assumed to substantially improve in the future with a reduction in global poverty in the "global sustainability" archetype and to a lesser extent in the "regional sustainability," but with recognizable regional differentiation (section 4.1). These improvements in GQL in sustainability scenarios go hand-in-hand with the most favorable projections of future dynamics of nature and NCP. However, continued degradation of nature, especially in developing economies of the tropics, and the consequences on NCP in the "economic optimism" scenarios suggest that the decoupling of economic growth on the one hand and nature, NCP and GQL on the other hand (see Chapters 2 & 3, and sections 4.2.2-4.2.4) could potentially continue into the future.

Indicators of GQL (Table A4.4.1, Appendix 4.4) have the poorest future trajectories in the "regional competition" scenarios and do only slightly better in "business-as-usual" and "economic optimism" scenarios at the global scale with substantial geographical differentiation. One of the underlying components of these storylines (particularly in the regional competition archetypes) is fragmentation, and large geographical variation in indicators of GQL. These scenarios also lead to the least optimistic future projections of nature and NCP (sections 4.2 & 4.3). These scenarios suggest that many of the current trends in socio-economic development (see Chapters 2 & 3) are projected to lead to lose-lose-lose responses of nature, NCP and GQL in the future (section 4.5) with inhabitants of developing economies expected to be severely impacted.

The literature review also finds that plausible scenarios are more likely to recognize the importance of nature for fulfilling material dimensions rather than the non-material ones. Similarly, there is a gap in the literature on the extent to which GQL dimensions depend on nature's contributions, and how they fit together. The literature clearly documents a strong correlation between nature's contributions and good quality of life (**Figure 4.5.2b** in section 4.5). Notably, positive trends in NCP are correlated with corresponding positive trends on GQL (top right of **Figure 4.5.2b**). Negative trends in NCP and GQL are similarly correlated (bottom left of **Figure 4.5.2b**) and comprise the bulk of the correlations reported as scenarios' outcomes. Nevertheless, analyses of such NCP-GQL relations could be further specified for scenarios exploring how those relations are mediated by contextual factors. For instance, future scenarios voiced by Amazonian communities reveal concerns with regard to livelihoods, equity aspects and the long-term impacts for communities and nature (Evans & Cole, 2014).

A challenge to the assessment of NCP and GQL under different future scenarios is their socially differentiated nature. This means that different groups may experience changes in NCP differently and with distinct impacts on GQL, so that a given change scenario usually implies winners and losers. People vary in their access to ecosystem services, exposure to disservices, dependence on ecosystems, and needs and aspirations for NCP. These are influenced by societal structures and norms as individual characteristics (Daw *et al.*, 2011) and power relations (Berbés-Blázquez *et al.*, 2017; Horcea-Milcu, 2015). Access shapes the transformation of ecosystem services to human well-being. For example, the perception of, dependence on and access to ecosystem services are strongly gendered. Men and women participate in different ecosystem-based livelihoods due to gendered roles and responsibilities gendered access to physical space, and gendered knowledge systems about ecosystems and NCP.

Thus, decision-making about environmental management with implications for different bundles of ecosystem services is an intently political process, with different stakeholders favouring different outcomes and holding different levels of power within those processes (Schoon *et al.*, 2015). Value systems and societal preferences for example evolve through globalisation of culture, or from burgeoning environmental consciousness in society (Everard *et al.*, 2016). Thus, changes in NCP and GQL are affected by social, economic, institutional change as well as biophysical change. Also how GQL of particular groups of people will respond to changes in biophysical conditions will be influenced by a wide range of factors (Daw *et al.*, 2016); see also section 4.4.2).

Evaluating GQL under different scenarios of change can benefit from deliberative and participatory approaches that consider a wide range of stakeholder views, and disciplinary perspectives (e.g., Brand *et al.*, 2013). Such a diversity of perspectives is necessary to take account of the multiple interacting factors and socially differentiated experiences, vulnerabilities and preferences for NCP (Barnaud *et al.*, 2018) as well as complexity and uncertainties in how NCPs evolve (Lele & Srinivasan, 2013).

Narrowly informed assessments of change may overlook socially differentiated outcomes. For example, aggregate analysis of a small-scale fishery in Kenya showed a win-win opportunity to improve profitability and conservation outcomes by reducing fishing effort and the use of small meshed beach seine nets. However, an inclusive participatory modelling approach showed that the livelihoods of certain groups, such as women traders would be negatively impacted by such a change due to the gendered nature of the value chain (Daw *et al.*, 2015). Likewise, in southern India, a disaggregated economic analysis shows how different stakeholder groups would experience different benefits and costs from the implementation of a forest conservation area (Lele & Srinivasan, 2013). For example, non-indigenous groups would suffer from curtailment of firewood and grazing benefits while indigenous groups would also lose out on these services but benefit to a greater extent from increased opportunities and sale of non-timber forest products. Importantly, from the perspective of developing scenarios, these wins and losses are shown to be highly contingent on complex institutional, technical and ecological dynamics in terms of access arrangements, irrigation methods and invasive species, respectively (Lele & Srinivasan, 2013).

Trade-offs between the good qualities of life of particular societal groups might easily be overlooked due to the complexity of ecological and social relationships, because the 'losers' of such trade-offs are marginalised or lack a voice in assessment processes and because of the psychological and political biases towards 'win-win' narratives that overlook uncomfortable or inconvenient trade offs (Daw *et al.*, 2015). A limitation with participatory approaches is the difficulty of imagining future scenarios of changes in the 'demand side' of NCP. So, a group may discuss how changes in a resource might be affected by climate change, but it is often framed in terms of current social conditions. Social, economic and political changes can have major impacts on NCP and subsequent effects on GQL.

Perspectives on GQL are also disputed and dynamic amongst modern and urban populations in wealthy countries. Increasing interest in well-being by Western governments (e.g., the OECD better life index http://www.oecdbetterlifeindex.org/) is critical for future scenarios because development trajectories, informed by the pursuit of economic growth are a major driver of ecosystem change. The possibility of a broader conceptualisation of well-being informing economic and development policy could have a major impact on the drivers behind environmental change.

Different conceptualisations or subjective experiences of GQL extend into relationships with ecosystems. While dominant economic framings in modern societies have emphasised instrumental values of nature, spiritual and aesthetic-cultural

Box 4 4 1 Climate Futures and Rural Livelihood Adaptation in Nusa Tenggara Barat, Indonesia.

What different futures are plausible for Indigenous People and Local Communities (IPLCs)?

Nusa Tenggara Barat (NTB) Province in the island archipelago of Eastern Indonesia is one of the country's poorest regions, and highly vulnerable to climate change due to dependence on rural, ecosystem-based livelihoods (Kirono *et al.*, 2016). It is therefore representative of other island regions in the tropical Asia-Pacific, which share the challenges associated with rapid change and entrenched poverty intertwined with complex traditional culture (Butler *et al.*, 2014, 2016a).

To assist communities to navigate future changes, from 2010-14, the Australian Government funded a series of scenario planning workshops with multiple stakeholders to investigate alternative development pathways and potential impacts on ecosystem services (Butler *et al.*, 2015). The project's Theory of Change assumed three evolutionary stages of adaptive co-management that would be triggered: 1) capacity building, 2) policy and program development and 3) implementation, adoption and scaling out. A participatory evaluation was carried out to test these assumptions and measure outcomes (Butler *et al.*, 2016c).

A key principle of the scenario planning process is that multiple stakeholders must be engaged through collaborative learning and knowledge co-production (Butler *et al.*, 2016c). Scientific and local knowledge was integrated in an interactive and iterative process throughout the workshops with the goal of co-producing knowledge via a 'learning spiral' **(Figure 4.4.5)**. Stage 1 scenario workshops were carried out with provincial level stakeholders, and then repeated in Stage 2 for five sub-districts and their community level stakeholders; Stage 3 then integrated the outputs of Stages 1 and 2 **(Figure 4.4.5)**. Stages 1 and 2 were structured around four questions: 1) What are the drivers of change for livelihoods? 2) What is the desired future for livelihoods? 3) What are the possible futures for livelihoods? and 4) What are the priority 'no regrets' adaptation strategies required to achieve the desired future in spite of future uncertainty?

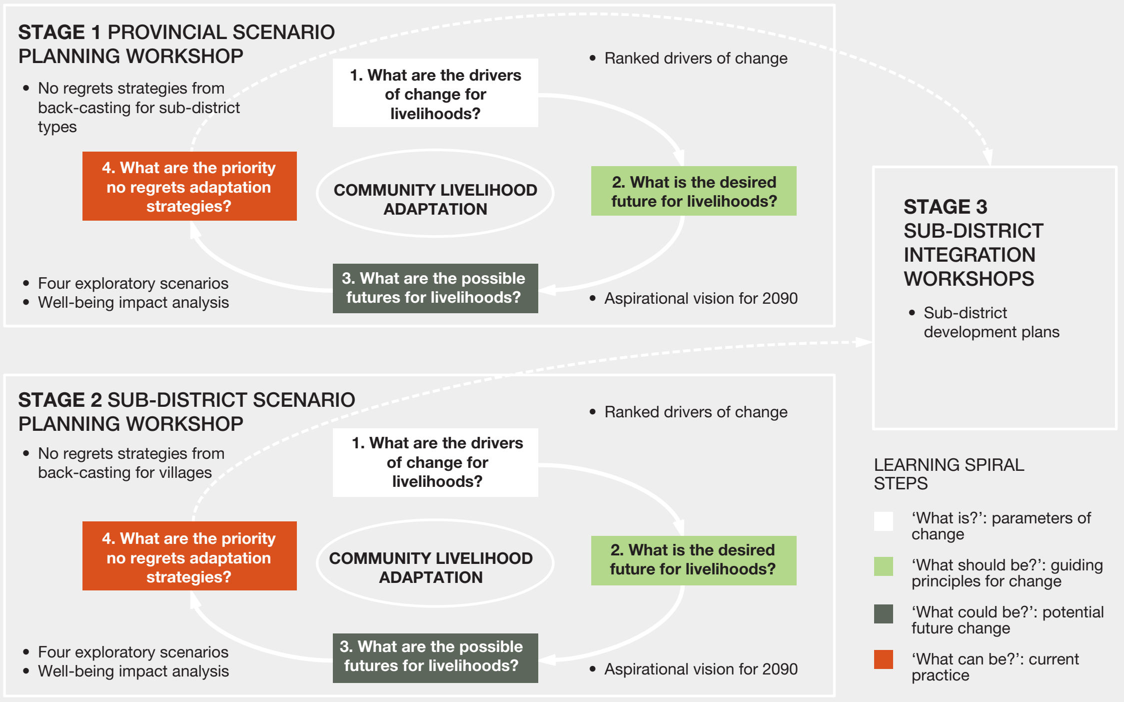


Figure 4 4 5 Stage 1 and 2 scenario planning workshop steps and primary outputs (bullets) based on Brown's (2008) learning spiral.

Also shown are the links to the subsequent Stage 3 sub-district integration workshops. From Butler *et al.* (2016b).

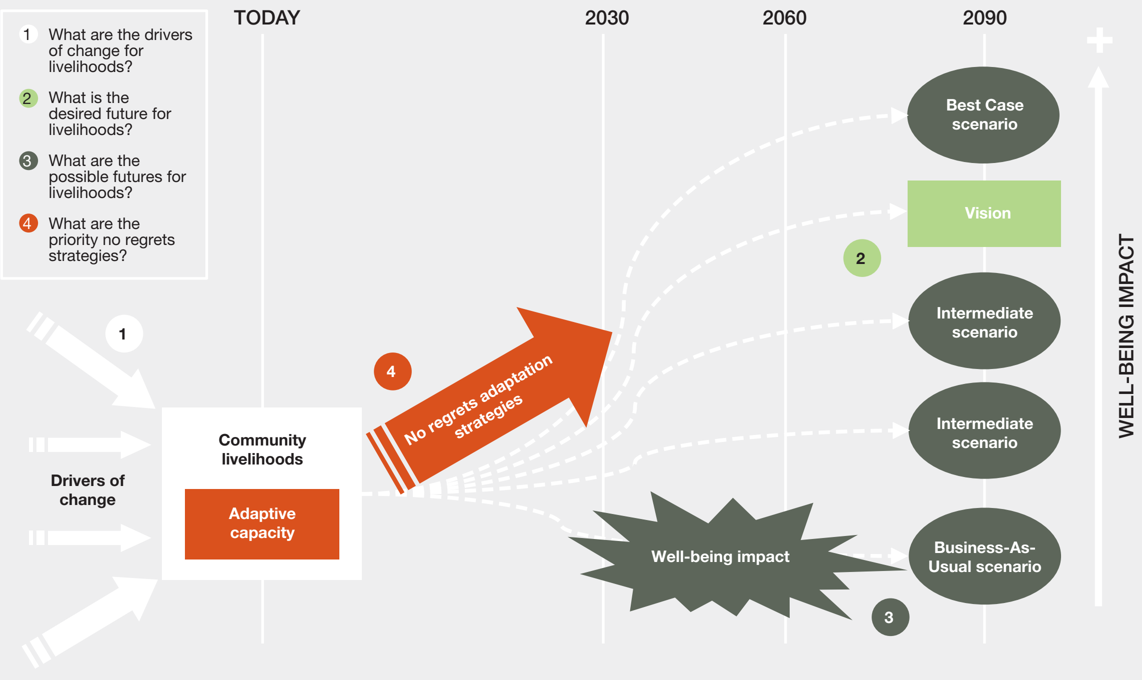


Figure 4 4 6 The 'roadmap' used in the Stage 1 and 2 scenario planning workshops to explain the sequential learning steps.

The step numbers and questions correspond to the learning spiral in Figure 4.4.5. From Butler *et al.* (2016b).

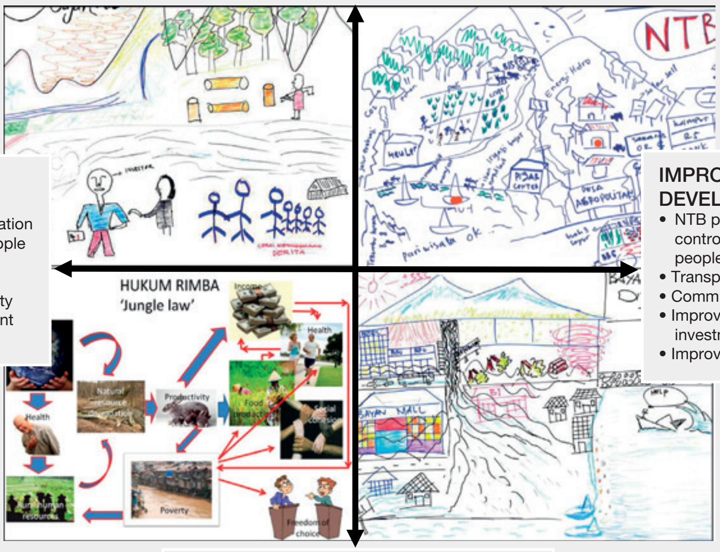
Participants in Stage 1 identified two key drivers from a list of 50 current drivers of change: development of human resources and climate change. They described a desired future vision for NTB rural livelihoods in 2090 based on adequate income, health, food security, social cohesion and freedom of choice for a good life. A matrix of four possible future scenarios was created from better or worse extremes of human resources development and climate change. Participants created narratives and illustrations for each scenario **(Figure 4.4.7)**.

An ecosystem goods and services typology and model was used to project future ecosystem goods and services and impacts on human well-being in 2030 for the business-as-usual scenario **(Figure 4.4.6)**. The most affected ecosystem types were rice and bandeng (fish) ponds, diverse cropping and coastal activity, diverse agriculture and forest use, and rice and tobacco (Skewes *et al.*, 2016). However, communities dependent on these ecosystem types for their livelihoods have varying levels of adaptive capacity. Hence, an adaptive capacity index was developed to rank vulnerability of NTB livelihoods, which identified the diverse cropping and coastal activity livelihood as most vulnerable. This assessment helped the participants to select sub-districts for community case studies in the next phase. Based on ecosystem goods and services and human well-being impacts and adaptive capacity for each typology, participants designed adaptation strategies for livelihoods to steer them away from 'business-as-usual' towards the NTB vision and the 'Best Case' Well-being Village scenario.

The same process was undertaken for each case study sub-district in the Stage 2 workshops, with more focus on local issues, knowledge and ecosystem goods and services.

Through the process, surveys identified distinct 'knowledge cultures' amongst stakeholder types in this region (e.g. government, communities and NGOs), with differing perceptions of future time horizons, climate change and development priorities (Bohensky *et al.*, 2016; Butler *et al.*, 2015). This finding justified the project design, which intentionally carried out the process at multiple scales in Stages 1 and 2, and then finally integrated the results by bringing stakeholders representing different scales together in Stage 3 (Figure 4.4.5). As a consequence, learning and innovation was one of the primary outcomes of the process (Butler *et al.*, 2016c).

LESS EXTREME CLIMATE CHANGE
- Temperature increase 0.3°C
- 3% increase in annual rainfall
- No change in wet season rainfall
- No change in drought risk area
- Sea level rise 23 cm
- Less severe wind and storms



POOR HUMAN DEVELOPMENT
- Uncontrolled NTB population growth with 7 million people by 2050
- Corrupt government
- Disempowered community
- Little education investment
- Little health investment

IMPROVED HUMAN DEVELOPMENT
- NTB population growth controlled with 5 million people by 2050
- Transparent government
- Community empowerment
- Improved education investment
- Improved health investment

EXTREME CLIMATE CHANGE
- Temperature increase 1.2°C
- 19% decline in annual rainfall
- Declining wet season rainfall
- Increased drought risk area
- Sea level rise 1.3 m
- Severe wind and storms

Figure 4 4 7 **Driver themes, sub-themes and exploratory scenarios for 2090 from the Stage 1 provincial workshop.**

From Butler *et al.* (2016b).

How indigenous and local knowledge (ILK) can be integrated with scientific knowledge in scenario-based projects towards Sustainable Development Goals (SDGs)

Participatory scenario planning has become a popular tool for navigating changes faced by many Indigenous Peoples and Local Communities. Integrating knowledge and multiple perspectives on change drivers, how the future might look and how stakeholders might respond, can potentially catalyse single-, double- and triple-loop learning that enable adaptation (Butler *et al.*, 2016c; Totin *et al.*, 2018).

The power of scenario planning to effect real change may be limited, however. While such scenarios present local visions for alternative futures in ways that conventional models, projections and forecasts cannot (Peterson *et al.*, 2003; Wollenberg *et al.*, 2000), their widespread adoption has not been matched by adequate resources. A review of place-based participatory scenarios found that very few projects complete a rigorous evaluation of outcomes (Oteros-Rozas *et al.*, 2015). Even in well-funded, multi-year projects such as the project in NTB, scenarios have only catalysed partial learning and change (Butler *et al.*, 2016a). In particular, the adoption of incremental rather than transformative adaptation strategies suggest that root causes of community vulnerability were not fully acknowledged, although numerous systemic drivers were identified. Scenario planning should be considered as only one tool in a process of capacity-building. This is particularly important in developing country contexts where capacity of stakeholders is low (Chaudhury *et al.*, 2013; Vervoort *et al.*, 2014). One-off scenario planning can generate enhanced learning and social networks but is unlikely to create transformational change needed to address systemic issues such as politics and institutions (Totin *et al.*, 2018). Ideally, the principles of futures analysis and learning should also be integrated within existing decision-making or development planning processes (Butler *et al.*, 2016c). If sustained, such grassroots platforms may catalyse and implement transformation, and ultimately enable vulnerable communities to leap-frog the SDGs (Butler *et al.*, 2016b).

values, whether of indigenous or modern societies, are hard to capture by instrumental thinking that underlies economic ecosystem service approaches. Instead, they are grounded in conceptions of nature that differ from the ecosystem services conceptual framework (Cooper *et al.*, 2016).

4.5 TRADE-OFFS, CO-BENEFITS AND FEEDBACKS BETWEEN NATURE, NATURE'S CONTRIBUTIONS TO PEOPLE AND GOOD QUALITY OF LIFE

4.5.1 Analysis of interactions from the Systematic Literature Review

Very few models and scenarios have been developed that simulate the complex interactions between nature, nature's contributions to people and good quality of life at continental or global scales, although such interactions are qualitatively well described and documented in the literature. As a result, scenario outcomes developed so far do not cover the full range of plausible futures. In the systematic literature review conducted for this chapter (Appendix A4.1.1), only 14 papers (out of a total of 572 papers), reporting a total of 41 different scenarios outcomes, addressed interactions between nature and NCP and GQL **(Figure 4.5.1)**. Advancing scientific knowledge about such interactions is crucial because of their relevance for identifying feedback effects, understanding trade-offs or win-win solutions and the risk of breaching thresholds and so–called "tipping points".

Analyses of the systematic literature review **(Figure 4.5.2)** suggest further that while relationships between nature, NCP and GQL are both positive and negative, the reported results indicate that the majority of indicators' trends are correlated either positive-positive or negative-negative. For instance, if a trend in a nature indicator is positive, there is more chance that a trend in an associated NCP is also positive **(Figure 4.5.2a)**, and conversely for negative/negative relationships. 62% of the simulated interactions between nature and NCP indicators' trends are correlated that way (excluding cases where both indicators of Nature and NCP have null trends). Likewise, the majority of relationships between NCP and GQL are positive-positive or negative-negative (80%; **Figure 4.5.2b**). The high proportion of such correlations suggests the existence of opportunities and potential co-benefits of measures aimed at preserving a specific nature's component, or a specific ecosystem service (section 4.5.3). However, the literature analysis does not allow to decipher whether there are causal relationships behind the positive correlations, and whether there are differences across regions or changes in trend over time (near vs. longer-term future). In addition, the level

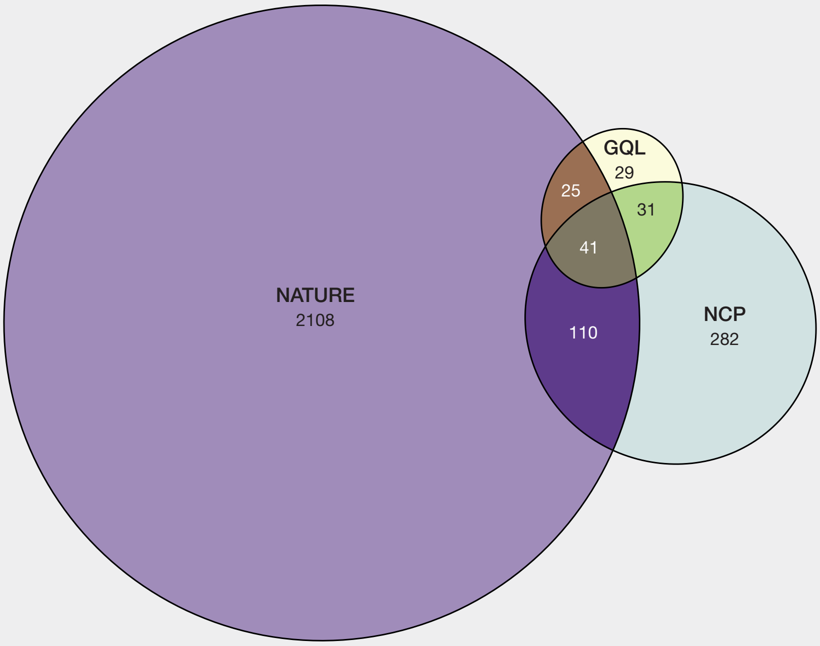


Figure ④ ⑤ ① **Number of results in the systematic literature review (for details, see Appendix A4.1.1) which report projections of at least one indicator of nature, nature's contributions to people (NCP), or good quality of life (GQL).**

Overlaps of the circles indicate that a result contained trends for more than one group of indicators.

Figure 4 5 2 **Heatmap showing the relationship between negative, neutral and positive trends reported for indicators of nature and nature's contributions to people (NCP, left panel, A), and NCP and good quality of life (GQL, right panel, B) in output of the models and global scale scenarios from the systematic literature review (Appendix A.4.1). n: number of results reported (i.e. multiple results per paper possible).**

of correlation is neither quantified, nor linked to any potential feedback effects that can dampen or amplify the drivers impacts on nature, NCP and GQL (section 4.5.1). There are a few numbers of negative correlations between nature, NCP and GQL indicator trends, which, although found in a lower proportion, can represent difficult trade-offs between different policy targets, e.g. between conservation and food provisioning targets (section 4.5.2).

4.5.2 Feedbacks

Feedbacks are processes that either reinforce or degrade the resilience of a stable state (Briske *et al.*, 2006), with both damping (also known as negative or balancing) and amplifying (also known as positive or reinforcing) feedbacks acting together or separately in a complex system to hold it in a particular state. A compilation of studies illustrative for feedbacks can be found in the Appendix (Table A4.5.2).

Feedbacks are well documented in the climate system (Ciais *et al.*, 2013). For example, increases in atmospheric concentration of CO_2, warmer temperatures and/or altered precipitation impact uptake and release of CO_2 in vegetation and soils, which in turn amplifies or dampens the original forcing via feedbacks on atmospheric CO2. Along coastlines, global sea level rise, temperature extremes and storm surges are projected to damage marine vegetated habitats and decrease wetlands area (Crosby *et al.*, 2016; Hoegh-Guldberg *et al.*, 2018), with potential negative feedbacks on climate change as these areas play key role in carbon burial and sequestration (Duarte *et al.*, 2013; section 4.2.2.2.2). In terrestrial systems, shifts in vegetation cover associated with climate change and atmospheric CO_2 (such as changes in woody type and cover, reduction of permafrost and peatlands, or shifts in fire regimes) play additional important roles in these dynamics (see section 4.2.4.1; Achard *et al.*, 2014; Arneth *et al.*, 2010; Davidson *et al.*, 2012; Lenton *et al.*, 2008; Lenton & Williams, 2013; Pearson *et al.*, 2017; Stocker *et al.*, 2013). In addition, reduced evapotranspiration due to climate change (or deforestation) feeds back on surface humidity, formation of regional cloud or rainfall which could also enhance forest vulnerability to fire and drought (Avissar & Werth, 2005; Devaraju *et al.*, 2015; Lenton & Williams, 2013; Quesada *et al.*, 2017b; Ray *et al.*, 2006). However, there remain large uncertainties in the magnitude and direction of feedbacks (Arneth *et al.*, 2010; Friedlingstein *et al.*, 2014; Raes *et al.*, 2010; Roy *et al.*, 2011; Stocker *et al.*, 2013).

Feedbacks also exist in coupled socio-ecological systems (and hence between nature, NCP and GQL; Hersperger *et al.*, 2011; Hull *et al.*, 2015; Robinson *et al.*, 2017). For instance, infrastructure used for extraction and use of natural resources generates wealth, which amplifies technological development and further extraction of resources. As the demand of a natural resource intensifies, its economic value increases. To seek monetary profits, exploitation increases as well and as long as the demand is high, economic value and exploitation continue to increase (Cinner *et al.*, 2011; Leadley *et al.*, 2010, 2014; Walker *et al.*, 2009. A social driver like market demand increases the value of natural resources with increasing scarcity of the resource. This negative feedback starts to be accounted for in fishing scenarios, with for example, high short-term economic incentives to exceed sustainable exploitation targets of marine resources, potentially leading to increases in fishing capacity and rapid depletion of fish stocks (Merino *et al.*, 2012). This often happens with large predatory fishes that are of high monetary value (Tsikliras & Polymeros, 2014). Overfishing leads to their depletion, new global markets develop for alternative species in turn (Quaas *et al.*, 2016), often their own prey, which leads to further depletion of marine resources (Steneck *et al.*, 2011). In addition, economic market feedbacks in response to a conservation intervention can hinder conservation efforts (Lim *et al.*, 2017). In this case the price increase of e.g., timber following future logging bans or other protective measures such as protected areas might be counterbalanced by illegal trade and enhanced logging elsewhere ("leakage") and these unintended feedbacks on timber supply via market responses could be amplified even further if interventions shift the competitive ratio of efficient to non-efficient producers. Leakage effect from protected areas could also take place, when protected areas reduce threats within their boundaries by displacing a part of these threats into adjacent areas (Renwick *et al.*, 2015).

One of the key interactions between climate change and socio-economic changes is human population distribution and mobility. Climate change-induced migration, also referred to as "environmental migration" (Black *et al.*, 2011), can exert additional pressure on the environment in regions of migratory influx of people, which in turn exacerbates degradation of resources. Likely, migrants would choose urban or developed areas as their destinations (Tacoli, 2009). Enhanced pressure on resources around cities (see 4.3.3) following the influx of large number of people might lead to further environmental degradation, and pressure of people to move elsewhere. There are inherent difficulties in explicitly monitoring and predicting the effects of environmental migration caused by migration due to lack of comprehensive data (Kniveton *et al.*, 2008). However, evidence from the past (including non-environmental migration) can already illustrate the potential impacts (Reuveny, 2007).

Changes in value systems and lifestyle, sense of nature and loss of indigenous or local knowledge can be side effects of globalization and commercialization that ultimately impacts the GQL which in turn leads to more exploitation of natural resources (Hubacek *et al.*, 2009; Reyes-García *et al.*, 2013; Uniyal *et al.*, 2003; Van der Hoeven *et al.*, 2013). Robust identification and quantification of feedbacks is a challenge for future scenario projections, in part because of teleconnections and telecoupling that need to be considered (Liu *et al.*, 2013). Both are interactions over distances; teleconnections refer often to interactions in the natural environment such as through atmospheric transport or ocean currents, while telecoupling explicitly acknowledges that in today's world interactions occur in coupled human-environment systems (Liu *et al.*, 2013; Robinson *et al.*, 2017). Global scale scenarios and models that would allow to assess the complex interactions between nature, NCP and GQL, and to identify the role of amplifying or damping feedbacks not only locally but also between regions do not yet exist.

4.5.3 Trade-offs

The use of a given ecosystem service by human societies affects in most cases the availability of other ecosystem services. In many cases trade-offs arise, especially between material NCP vs. regulating NCP and biodiversity (see sections 4.3.2 and 4.3.3; Bennett *et al.*, 2009; Bonsch *et al.*, 2016; Carpenter *et al.*, 2017; Clark *et al.*, 2017; Di Minin *et al.*, 2017; Krause *et al.*, 2017; Lafortezza & Chen, 2016; Powell & Lenton, 2013; Seppelt *et al.*, 2013; Tscharntke *et al.*, 2012; Vogdrup-Schmidt *et al.*, 2017). Similar results have been found across all the IPBES regional assessments (IPBES, 2018b, 2018e, 2018c, 2018d) and UNEP's Global Environmental Outlooks (e.g., UNEP, 2012). In most future scenarios, the demand for material NCP increases because of population growth and consumption pattern changes (Popp *et al.*, 2017), which can be considered principal drivers for the declines in regulating NCP and biodiversity. In absence of targeted policy, future global demand for food, energy, climate and biodiversity may be very difficult to achieve simultaneously (e.g., Henry *et al.*, 2018; Obersteiner *et al.*, 2016; von Stechow *et al.*, 2016). Trade-offs (but also co-benefits) in ecosystem service supply can be considered important components of feedback loops (see 4.5.2), since in the long term a substantial decrease in regulating services will also negatively affect provision of material services that depend on the regulating ones (Cavender-Bares *et al.*, 2015). For instance, the destruction of pollinator habitat as part of agricultural expansion or intensification, can lead to declines in food production (IPBES 2016b), resulting in the need for further agricultural expansion (and associated further loss of pollinator habitat). The implications of future trade-offs will be influenced by regionally specific biophysical settings in combination with cultural preferences and thus should be considered in decision-making (Cavender-Bares *et al.*, 2015) (see chapter 6). However, since scenarios and models for many NCP are non-existent or incipient, many trade-offs and synergies remain unknown (Mach *et al.*, 2015). In particular cultural services are usually not considered in scenarios development or in models (see section 4.3), therefore future trade-offs with material and non-material aspects are poorly understood.

Food, bioenergy and water

Increasing consumption of food, and associated terrestrial and marine food production sectors, are seen as a main driver of biodiversity loss. Overexploitation of wild marine resources is expected to increase in the future under current management schemes (Costello *et al.*, 2016; see section 4.2.2.3.1) but could be alleviated by the growth of the aquaculture sector (Merino *et al.*, 2012; Quaas *et al.*, 2016). However, aquaculture development is challenged by a number of trade-offs related to fishmeal provisioning (Blanchard *et al.*, 2017) from wild marine resources (and potential further decline of marine populations, especially those serving as prey for already overexploited marine predators) or from cereal and soya production affecting land-based food production. Terrestrial ecosystems are impacted through cropland expansion as well as intensification on existing agricultural land and associated inputs of water and fertilizer (Foley *et al.*, 2011; Tilman *et al.*, 2011; Tilman & Clark, 2015). The pressure on agricultural systems will be increasing not only due to the continued population growth but also due to projected changes in dietary preferences towards meat-based protein intake in many countries. Under continuation of current trends, global food, water or timber demands are estimated to increase by 30% (timber), 65% (food and feed) and 75% (water) by 2050 (van Vuuren *et al.*, 2015).

Land-based climate change mitigation requires additional land area (e.g. for bioenergy or reforestation), which is projected to be lowest in sustainability scenarios that assume changes in consumption patterns (e.g., 250-530 Mha, SSP1/RCP2.6), and highest in scenarios that describe a world with large regional competition (e.g., 250-1500 Mha, SSP4/RCP2.6) (Popp *et al.*, 2017). In view of food and water demands of a growing human population, the question remains whether (and where) the required land area would be available for large bioenergy plantations or afforestation/reforestation efforts. Likewise, large direct or indirect side effects have been shown to arise for the global terrestrial ecosystem carbon balance, and hence climate regulation, other ecosystem functionality and biodiversity (Bird *et al.*, 2013; Jantz *et al.*, 2015; Krause *et al.*, 2017; Kraxner *et al.*, 2013; Melillo *et al.*, 2009; Plevin *et al.*, 2010; Santangeli *et al.*, 2016). It is well documented that the use of ecosystem services regionally will impact ecosystem functioning and services in other regions (Jantz *et al.*, 2015; Krause *et al.*, 2017; Seppelt *et al.*, 2013; and see section 4.3.3). For tradeable goods, and in absence of changing demand, land-use change in a given region (for instance, converting land to bioenergy rather than food production) will result in compensatory land-use changes elsewhere (for instance, conversion of natural habitat to food production) (Bird *et al.*, 2013; Krause *et al.*, 2017; Kraxner *et al.*, 2013; Melillo *et al.*, 2009; Plevin *et al.*, 2010).

Future land-use change scenarios with Integrated Assessment Models (Popp *et al.*, 2017) assume that land for bioenergy growth or afforestation and reforestation can be freed up through continued strong increases of crop yields (Bijl *et al.*, 2017; Bonsch *et al.*, 2016; Humpenoder *et al.*, 2015; see also **Table 4.1.6**, section 4.1), but the environmental and societal issues associated with the intensification of agricultural production are insufficiently considered in these scenarios. For an end-of-century 300 EJ bioenergy target to be produced from plants, Bonsch *et al.* (2016) found a doubling of global agricultural water withdrawal and a bioenergy production area of 490 Mha, or a land requirement of 690 Mha if no irrigation of bioenergy plants is considered. The latter increased to approximately 1000 Mha land for bioenergy if technology effects on increased yields would be only half of those in bioenergy than in food crops (Bonsch *et al.*, 2016). Krause *et al.* (2017) found both increases and decreases in different ecosystem functioning in response to scenarios under a RCP2.6 umbrella that included large-scale land-related climate change mitigation efforts, with large variability across regions and land-use scenarios. Large nitrogen losses were simulated in response to fertiliser needs to support yield increases, indicative of air and water pollution. Competition for land in climate change mitigation scenarios based heavily on bioenergy production has also been shown to increase food prices (Kreidenweis *et al.*, 2016). Detrimental societal impacts will arise if these price increases cannot be met by economic growth. It has now been consistently demonstrated that regional surface temperature can be strongly affected by land cover change, arising from altered energy and momentum transfer between ecosystems and atmosphere, with either an increase or decrease in temperature depending on the geographic context (Alkama & Cescatti, 2016; Li *et al.*, 2015; Perugini *et al.*, 2017; Quesada *et al.*, 2017a). Thus, changes in surface climate arising from large-scale land cover change in mitigation efforts can regionally amplify or reduce climate change. Large-scale land-based climate change mitigation efforts need to take account of unintended consequences on ecosystems that could undermine climate regulation or provisioning of a range of important ecosystem services.

An important element of the SSP1/RCP2.6 scenarios which limit global warming to about 2°C is that much of agriculture and bioenergy production relocates from high-income temperate regions to low-income tropical ones (van Vuuren *et al.*, 2011) where most of freshwater diversity is concentrated (Tisseuil *et al.*, 2013). Deforestation, extraction of high amounts of water withdrawal for irrigation, and use of pesticides and fertilizers to increase productivity in expanding bioenergy croplands are known to adversely affect natural aquatic systems and their biodiversity, notably fishes through local extinctions and alteration of their community structure (sections 4.2.3.2; 4.2.3.3). Inland fisheries are particularly important in tropical developing countries and currently provide the major dietary protein source for well over half a billion people (FAO, 2016; Lynch *et al.*, 2016). An increase in bioenergy production in these low-income food-deficit countries is thus expected to strongly impact fisheries and compromise further their food security.

4.5.4 Co-benefits

In order to sustain and enhance the future supply of NCP, in particular between regulating and non-material contributions (Ament *et al.*, 2017; Hanspach *et al.*, 2017; Potts *et al.*, 2016; Vogdrup-Schmidt *et al.*, 2017), changes in consumption patterns, globally, alongside changes in supply has emerged as crucial in scenarios of ecosystem change, NCP and GQL. In this context, reduction of food waste and shifts in diets are most illustrative.

Enhancing efficiencies in the food system, including the reduction of food losses and waste that occurs at several stages in the food production system, has large potential to enhance food security in a world where still every third person is malnourished, and 815 million people are hungry (FAO *et al.*, 2018). It may also free up land for other uses such as for biodiversity conservation, and entail additional co-benefits such as reduced greenhouse gas emissions from the land sector, and reduced irrigation water needs which will also release pressure on freshwater pollution and biodiversity (Alexander *et al.*, 2017b; Godfray *et al.*, 2010; Kummu *et al.*, 2012; Pfister *et al.*, 2011; Smith *et al.*, 2013). Nearly one-quarter of total freshwater used today in food crop production could be spared if wastes and losses in the food system were minimized (Kummu *et al.*, 2012). Nearly 10% of the agricultural land area could be spared globally through halving consumer waste arising from over-consumption in some sectors of society (Alexander *et al.*, 2017b). For the period 1961-2011, waste and losses in the food system were estimated to sum to approximately 68 $GtCO_2$ equivalents (Porter *et al.*, 2016).

A number of studies address the potential of reducing future expansion of croplands and/or reducing environmental impacts from agriculture and pastures (especially climate regulation related to reduced greenhouse gas emissions) through changes in diets. Studies that explore dietary scenarios of either reduced consumption of animal protein (combined with a globally more equitable distribution of animal protein) or no consumption of animal protein estimate that between about 10% and 30% of today's area under agriculture could be freed for other purposes (Alexander *et al.*, 2016; Bijl *et al.*, 2017; Ridoutt *et al.*, 2017 and references therein; Roos *et al.*, 2017; Tilman & Clark, 2014; Wirsenius *et al.*, 2010). A further positive side effect of these dietary shifts are health benefits in overweight population categories (Roos *et al.*, 2017; Tilman & Clark, 2014). The evidence base on impacts of diets on biodiversity, arising from reduced agricultural expansion is limited and context specific; however, a consumption-change scenario that included, among other changes in lifestyle, a shift towards a more vegetarian diet found positive effect on biodiversity of terrestrial mammals, in particular those with large ranges (Visconti *et al.*, 2015).

Additional cost-efficient measures to address environmental challenges have been demonstrated in studies that investigated optimizing crop distribution or the combination of several climate change mitigation options, while respecting food and fiber demand and conservation needs (Davis *et al.*, 2017; Griscom *et al.*, 2017). Through the globally optimal distribution of major crops, agricultural water use could be reduced by 12-14%, in a process-based crop-water-model combined with spatial information on yields, with large co-benefits for calorie and nutrient supply (Davis *et al.*, 2017). In particular, a move from some of the main cereal and sugar crops to e.g. roots, tubers and nuts underpinned these positive impacts. While cultural barriers, such as dietary preferences, will prevent to reach these potential gains of reduced water loss and enhanced food security, the analysis nonetheless puts forward a cost-efficient strategy towards sustainable intensification that could maintain small-holder farm systems and avoid large investments in technology-driven agriculture. From the perspective of contributing towards the achievement of the 2°C warming goal, economically-constrained greenhouse-gas reduction measures in the agriculture and livestock sector were estimated to contribute 1.5-4.3 Gt CO_2-eq. a^{-1} emission reductions (Bustamante *et al.*, 2014; Smith *et al.*, 2013; Tubiello *et al.*, 2015), which can be substantially enhanced further if consumer demand measures were also included. Recently, a combination of 20 different management measures in forests, agricultural land and wetlands achieved a maximum reduction of ca. 11 Pg C_{eq} a^{-1} when constrained by food security, conservation considerations and cost-efficiency (Griscom *et al.*, 2017). In addition, the future of land use and its impacts on biodiversity and ecosystem services depends on opportunities for building climate-resilience across sectors, including fisheries and aquaculture production systems (Blanchard *et al.*, 2017). As fish production has been the fastest growing food industry for the last 40 years, outpacing growth in all other livestock sectors (Béné *et al.* 2015), adaptive sustainable fisheries management (Costello *et al.*, 2016; Gaines *et al.*, 2018) combined with the development of sustainable low input and low impact aquaculture could generate co-benefits for food security, conservation of biodiversity, and climate regulation.

4.5.5 Regime Shifts, Tipping Points and Planetary Boundaries

There is a growing body of evidence that socio-ecological systems can be pushed past certain limits, beyond which they are profoundly altered in their structure and functioning. These are variously referred to as "regimes shifts", "tipping points" and "moving beyond planetary boundaries" and can be caused by a number of mechanisms (see Table A4.5.3 in Appendix 4.5). In some cases, these shifts occur rapidly and are difficult to reverse (Hughes *et al.*, 2013). The term "regime shifts" encompasses most of the concepts found in the definitions of tipping points and planetary boundaries, and so it will be used throughout this section except in cases where the distinction between concepts is important (Hughes *et al.*, 2013; Leadley *et al.*, 2014).

In some cases, regime shifts arise from relatively well understood physical and biological processes or feedbacks (Table A4.5.2) and have been included in models. In many cases, however, regime shifts arise from the complex interplay and feedbacks between people and nature (Table A4.5.3), and in general have not been well accounted for in scenarios and models. In addition to the underlying mechanisms, the spatial and temporal scales of regime shifts are extremely important when assessing the importance of their impacts and the evidence base for their past, current and possible future occurrence (Hughes *et al.*, 2013; IPCC, 2018; Steffen *et al.*, 2018).

Regime shifts that occur over the span of several years to several decades are well documented at local to small regional scales and occur frequently in response to increasing human pressure. In some cases, these can be reasonably well foreseen with scenarios and models. These regime shifts have large impacts on nature, nature's contributions to people and good quality of life at local scales, but may also have important impacts at much larger scales when they occur in many places at the same time (Leadley *et al.*, 2014). The collapse of local and regional fisheries is a salient example in marine ecosystems. The accumulation of these collapses at local to regional scales has reached a point where a substantial fraction of the world's fisheries is either collapsed or near the limits at which they could collapse (section 4.2.2.3.1). Land degradation is a good example in terrestrial socio-ecological systems. Land degradation is often the result of complex human-nature interactions and therefore the causes of land degradation are not the same everywhere in the world (Table A4.5.3). Land degradation is, however, sufficiently widespread that it is "negatively impacting the well-being of at least 3.2 billion people" (IPBES, 2018a). The increasing widespread phenomena of eutrophication of ponds and lakes by excess nutrient input is an excellent example in freshwater ecosystems (section 4.2.3.3). The common characteristics of these examples are that i) there is a rapidly increasing number of areas affected by these regime shifts, to the point that they now have global scale implications for nature and people, ii) scenarios and models of business-as-usual trajectories indicate that the pressures driving these regime shifts will increase over the coming decades in many regions and iii) scenarios and models suggest there are plausible alternative pathways that avoid aggravation of these regime shifts and, in many cases, lead to partial restoration of these systems (e.g., land restauration scenarios in IPBES, 2018f; Leadley *et al.*, 2010).

There are several regime shifts at large regional scales underway that have been initiated by human disturbance and are projected to have direct impacts on biomes over the next several decades (Leadley *et al.*, 2010; Steffen *et al.*, 2018). There is strong evidence that large-scale regime shifts have begun for tropical coral reefs (section 4.2.2.2.2, **Box 4.2.3**), large-scale changes in marine communities and ecosystem function due to the loss of summer sea ice in the Arctic Ocean (sections 4.2.2.2.1 and 4.2.2.2.4); and degradation of permafrost and increasing woody vegetation in arctic tundra systems (Settele *et al.*, 2014; section 4.2.4.1.1). Models foresee rapid aggravation of these regime shifts over the coming century (IPCC, 2018; Leadley *et al.*, 2010; sections cited above). Further rapid, global-scale degradation of tropical coral reefs — which are driven by the combined impacts of climate change, ocean acidification, sea level rise, pollution and overexploitation — is of particular and immediate concern because of the severe impacts on biodiversity and because large human populations depend on coral reef ecosystems for food, income and shoreline protection (IPCC, 2018; see **Box 4.2.3** and section 4.3.2.1). Several other postulated regime shifts at large regional scales are more uncertain. For example, the large-scale collapse of the Amazonian rainforest has been postulated due to the combined effects of deforestation and climate change and regional scale feedbacks, but observational and experimental evidence, as well as modeling studies are equivocal about the likelihood of a large-scale regime shift (Settele *et al.*, 2014; section 4.2.4). There are also early signals of tree dieback in boreal forests due to climate change, and some models project large-scale boreal forest degradation over the coming century, but the spatial scale and magnitude of this regime shift remains speculative (Settele *et al.*, 2014). A key feature of these regime shifts is that they are driven in large part by climate change and/or rising atmospheric CO_2 concentrations and therefore require strong international actions to reduce greenhouse gas emissions (IPCC, 2018). However, adaptation to and attenuation of climate change impacts also require additional local and national scale efforts to reduce other pressures under biophysical and economic limits (e.g., Smith *et al.*, 2016).

The likelihood of the occurrence of regime shifts, tipping points, or boundaries being exceeded for biodiversity and ecosystem services at global scales are speculative. The planetary boundaries literature posits that there are a few indicators that can be used to identify boundaries beyond which the planet will leave the relatively stable "safe operating space" that it has operated in over the last 10 millennia (Hughes *et al.*, 2013). There is growing evidence that some indicators, especially for climate change, are useful for identifying potential global scale regime shifts (Steffen *et al.*, 2018), but there is little evidence yet for a global scale indicator for biodiversity loss or degradation of ecosystem integrity (Mace *et al.*, 2014). It has also been postulated that the Earth is approaching a global scale regime shift that would lead to a massive loss of biodiversity and incalculable impacts on people (Barnosky *et al.*, 2012; Brook *et al.*, 2013; Steffen *et al.*, 2018). The mechanisms for these Earth scale tipping points are not well defined and not included in any models (Hughes *et al.*, 2013), but the combined effects of several large-scale regime shifts including the irreversible melting of the Greenland ice sheet, the loss of the West Antarctic ice sheet and several other regime shifts could plausibly combine to create a shift to a very hot global climate regime once moderate levels of global warming have been exceeded (Steffen *et al.*, 2018). There are also plausible mechanisms leading to telecoupling between regions such as atmospheric transport, movements of organisms, or human migrations that can greatly increase the spatial extent or impact of regime shifts (Leadley *et al.*, 2014). While these global scale regime shifts and planetary boundaries are speculative, the potential magnitude and scale of the impacts are so large that further work to understand and model the underlying mechanisms is essential.

4.6 LINKS TO SUSTAINABLE DEVELOPMENT GOALS, AICHI BIODIVERSITY TARGETS AND OTHER INTERNATIONAL OBJECTIVES FOR NATURE AND NATURE'S CONTRIBUTIONS TO PEOPLE

4.6.1 How good will we be at reaching international biodiversity and sustainability targets beyond 2020?

Scope: How are scenarios and models addressing international biodiversity targets and sustainability goals and what insights do they provide? This section builds on Chapter 3 (Progress towards Aichi Biodiversity Targets) by looking at projections beyond 2020.

The Aichi Biodiversity Targets agreed to in the Strategic Plan for Biodiversity 2011–2020, targets in other multilateral environmental agreements, and the Sustainable Development Goals (SDGs) have been adopted to motivate actions to sustain nature and its contributions to the promotion of human well-being and sustainable development (Chapter 3). Although many of the SDGs do not explicitly focus on nature, with the notable exception of goals related to life below water and life on land (SDGs 14 and 15), the supply of multiple ecosystem services is critical to achieving many SDGs. And despite the fact that relatively few SDG targets (as currently expressed) map directly onto nature or its contribution to people, most Aichi Biodiversity Targets are clearly related to SDGs.

Analysis of the data sourced from the systematic literature review (Appendix A4.1.1) shows that despite the importance of SDGs and Aichi Biodiversity Targets for sustainability and human well-being, few scenario analyses have a specific focus on achieving them, at least at global scale. Scenarios of biodiversity and ecosystem services can contribute significantly to policy support in all the major phases of a policy cycle, including agenda setting and policy design (Ferrier *et al., 2016; IPBES, 2016b, figure SPM3)*. Several scenario and modeling analyses provide useful indications related to policy targets, albeit indirectly **(Figure 4.6.1)**, but the vast majority of these relate to species declines and extinctions, therefore informing only on Aichi Target 12 (conservation of threatened species) and

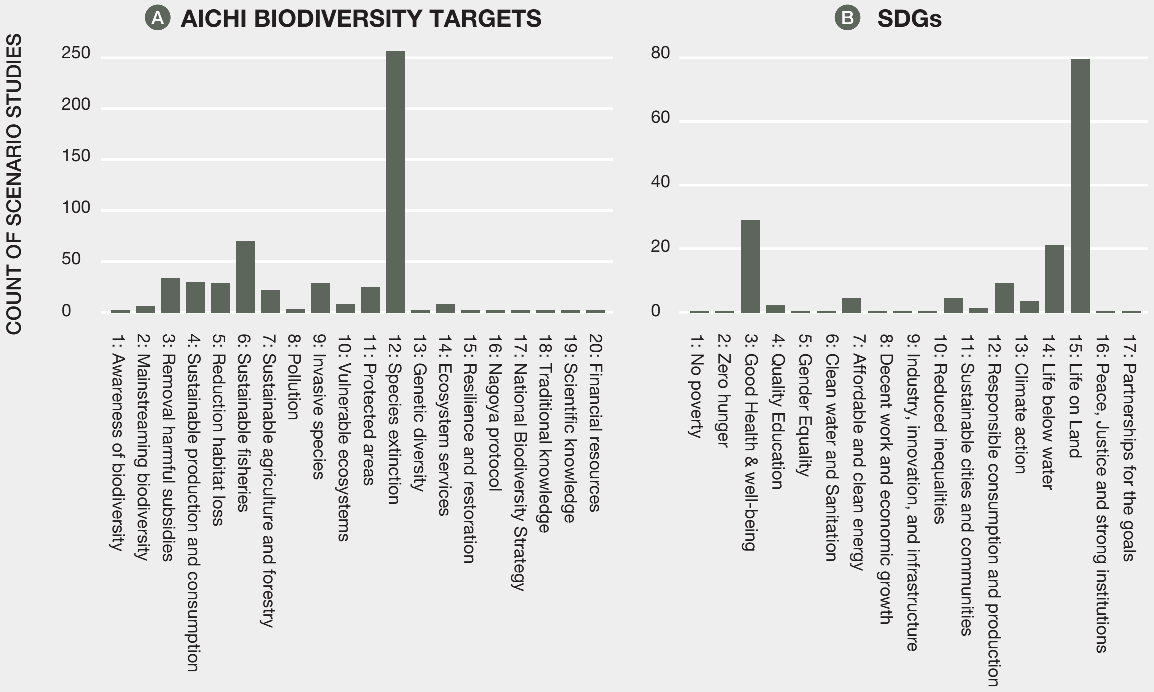


Figure ④ ⑥ ① **Number of scenario studies addressing Aichi Biodiversity Targets Ⓐ and SDG thematic policy areas Ⓑ based on the systematic literature review carried out for this chapter (Appendix A4.1.1).**

a small subset of targets related to SDG 15 (life on land). The reason for this imbalance probably lies in the different level of development of methods in the research community. Models for projecting species distributions under climate scenarios (which relate to Target 12 and SDG 15) are well established and widely used in the literature, while the exploration of other targets and goals is hampered by the scarcity of appropriate models at global scale. Global scale scenarios specifically addressing Aichi Biodiversity Targets are scant **(Figure 4.6.1A)**, and most of them relate to Target 12 (conservation of threatened species) and 6 (sustainable fisheries). Scenarios addressing SDGs focus mostly on SDG 15 (life on land), 2 (zero hunger) and 14 (life below water), but this also reflects the fact that the focus of the systematic literature review for this chapter was restricted to biodiversity and ecosystem services, rather than encompassing other societal goals. Therefore, the SDGs other than 14 and 15 represented in **Figure 4.6.1B** were addressed in conjunction with SDG 14, 15 or both.

For Sustainable Development Goals, scenario analyses are usually sector-specific (Obersteiner *et al., 2016)*, and a review of 22 modelling case studies has shown that it would be unlikely that any scenario modelling exercise could cover all (Allen *et al., 2017)*. Most studies focus on environment-economy interactions, such as greenhouse gases (GHG) reduction and impacts of this on growth and employment, and consideration of broader social issues is limited (Allen *et al., 2017)*. Various models have been used to assess SDGs including top-down system dynamics, macro-economic and hybrid models as well as bottom-up sectoral models across multiple sectors such as energy, agriculture, transport, land use, etc. (Allen *et al.,* 2016, 2017).

Biodiversity targets have been missed in the past for 2010 (Butchart *et al.*, 2010), and the mid-term progress towards Aichi Biodiversity Targets for 2020 was insufficient (Tittensor *et al.*, 2014). The world is still far or very far from achieving most of the Aichi Biodiversity Targets by 2020 (Chapter 3). Evidence from the limited number of scenario analyses from the systematic literature review (Appendix A4.1.1) shows that these targets are unlikely to be achieved even at some point in the future in most scenarios (2050 and beyond). However, for most targets, delayed achievement in the future is possible under some scenarios **(Figure 4.6.2)**. Recent scenario research has explored the likelihood that global biodiversity targets can be achieved by steering from business-as-usual to more sustainable socio-economic development trajectories. For example, Visconti *et al.* (2016) have projected policy-relevant indicators (Living Planet Index, LPI, and indicator of species abundance, and Red List Index, RLI, an indicator of extinction risk) for large mammals to 2050, comparing a reference scenario to sustainability scenarios (van Vuuren *et al.*, 2015).

Figure ④ ⑥ ② **Achievement of Aichi Biodiversity Targets in the future (2050 or beyond) across different global scale scenarios, based on our systematic literature review (Appendix A4.1.1).**

N: number of indicators, with number of studies in parenthesis, each study usually projecting several different indicators. Many scenarios dealing with Aichi Biodiversity Targets did not report results in a way that could be interpreted as level of target achievement, therefore the numbers here are smaller than in **Figure 4.6.1**.

They showed that after a mid-term increase until 2030, biodiversity indicators would decline again afterwards as the projected effects of climate change outpace mitigation actions. This analysis showcases how scenario modelling links long-term results to short- and medium-term action. It has been proposed that for achieving future targets, bold goals like the CBD 2050 Vision be adopted, and that integrative policies for sustainable production and consumption (e.g., a shift towards a more balanced diet, Chapter 5) be adopted (Mace *et al.*, 2018).

The global results on achievement of biodiversity targets do not scale down to the IPBES regions where the same topic has been addressed. The IPBES regional assessment for Africa (IPBES, 2018g) found low likelihood to ever achieve most Aichi Biodiversity Targets, except Target 1 (awareness of biodiversity) and 14 (ecosystem services), for which the regional trend is positive. Under the “fortress world” archetype scenario (similar in characteristics to the “regional competition” archetype defined in this chapter, section 4.1), the trend in Africa is negative for all Targets. For Europe and Central Asia, sustainability scenarios are expected to achieve most Aichi Biodiversity Targets, but still fail a few (in particular Targets 1, awareness of biodiversity, and 17, national biodiversity strategies) (IPBES, 2018i). The information is not available for other IPBES regions.

If the global socio-economic development continues according to a business-as-usual scenario, it is likely that we will fail to achieve several biodiversity-related SDGs (SDG 14, Life below water, and 15, Life on land). Three-quarters of the scenario and models that address SDG 15 project that we will be far or very far from achieving it. A similar outcome is projected for SDG 14 **(Figure 4.6.3)**. In Europe and Central Asia scenarios of sustainable production and consumption are expected to achieve most SDGs (IPBES, 2018i). In this region, the economic optimism archetype scenarios are expected to achieve most SDGs, but notably fail SDG 14 and 15. A recent study stressed that under the current trajectory of socio-economic development, progress in SDGs related to poverty and social inclusion happens at the expense of the environment, and this will lead to missing environmental SDGs in most of the world countries (**Figure 4.6.4**; Spaiser *et al.*, 2017). This is attributed to the

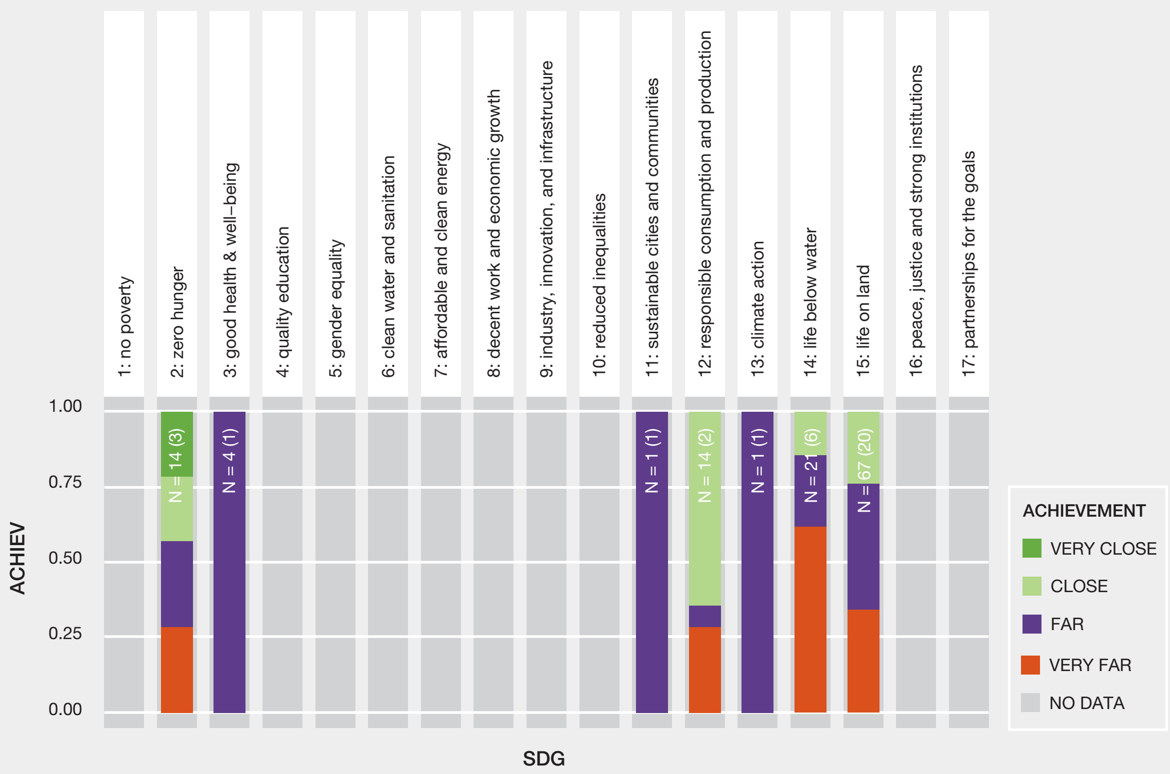


Figure 4 6 3 **Achievement of SDGs across different global scale scenarios, based on our systematic literature review (Appendix A4.1.1).**

N: number of indicators, with number of studies in parenthesis, each study usually projecting several different scenarios. Many scenarios dealing with SDGs did not report results in a way that could be interpreted as level of target achievement, therefore the numbers here are smaller than in **Figure 4.6.1**. The systematic literature review focused on Nature and NCP, therefore SDGs other than 14 and 15 were captured only if they were assessed in conjunction with them.

focus on economic growth and consumption as means for development.

Several emerging issues have been identified as influential to the achievement of the SDGs. These include new scientific knowledge, new technological development, new scales or accelerated rates of impact, a heightened level of awareness and new ways to respond to a known issue (UN, 2016). Despite the uncertainty associated with these emerging issues, various aspects have been identified as necessary to achieve the SDGs. First, measuring progress at all scales, and integrating global targets with local policies is fundamental towards achieving the SDGs (Biermann *et al., 2017)*. Goal 17 on revitalizing the “global partnership”, for example, will require increased funding and clear leadership (Biermann *et al., 2017)*. Increased funding is also one of the fundamental needs to achieve the SDGs in some regions within the African Continent (Kedir, 2017). Controlling consumption and demand remains an important issue. A recent work combing literature review and a comparison exercise of integrated energy-economy-climate models, AMPERE, found out that in order to achieve a 2ºC scenario, lowering the global growth of energy demand is key according to energy-economy-climate models (von Stechow *et al., 2016)*. Several local scenario studies provide useful insights towards achieving SDGs. In South Asia, industrial transformation, sustainable agriculture and innovations have been identified as key aspects to achieve SDGs (Kumar *et al., 2016)*. Participatory scenarios to achieve visions coherent to SDGs and to adequately adapt to future climate change impacts have also been applied with local communities in Indonesia (Butler *et al., 2015)*.

Scenarios have proven useful to identify and analyze synergies and trade-offs among biodiversity targets and SDGs. Glover and Hernández (2016) applied foresight techniques with experts in international development studies and found out that SDGs are not necessarily harmonious and mutually reinforcing but that trade-offs exist. According to this study, without strategic planning, advances towards one SDG might lead to negative consequences to others. Sustainable Consumption and Production policies (SDG 12), assessed through the GLOBIOM model, shows the need of inclusive policies among global development and

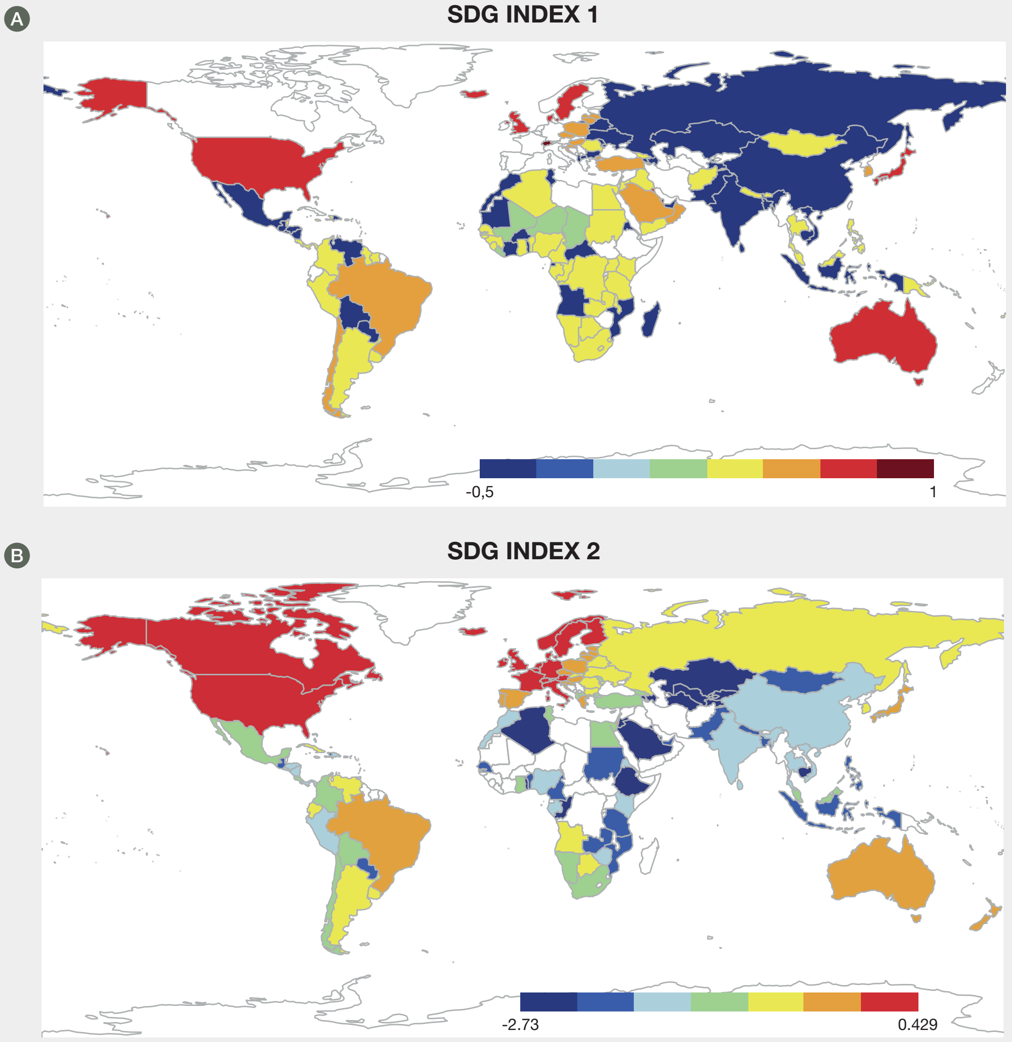


Figure ④ ⑥ ④ **World maps with countries being coloured based on an SDG index scores for 2011.**

The SDG index is based on the rate of change in three variables (child mortality, education, and CO_2 emissions) representing the three SDG pillars (ending poverty, social inclusion and environment). White colour indicates that an index could not be calculated, due to missing data in one or several of the predictors in the modelling approach used by Spaiser *et al.* (2017). The modelling approach used country and year specific data and combined confirmatory and exploratory factor analysis with dynamic systems modelling. Panel Ⓐ reveals which factors are associated with development and negative values indicate a reduction of child mortality and an increase in education, as well as increased CO_2 emissions. Countries such as Russia, China and India present negative values of this indicator, showing that they perform well at reducing poverty and increasing socio-economic inclusion, but with associated environmental trade-offs in terms of CO_2 emissions. Developed countries like Australia, USA and UK, which have high levels of socio-economic development, show little room for improvement of their SDG index. Other countries like Brazil, Thailand and South Africa seem to present slower socio-economic development trends. Panel Ⓑ uses a similar modelling approach but combined with a Bayesian model. The SDG index shows similar results for rich countries. However, contrarily to results of panel Ⓑ, in panel Ⓐ Russia presents a slow socio-economic development and several African countries such as Angola and Kenya make some progress in terms of socio-economic development. The trade-off between environment and the other two SDG pillars means that based on this analysis, under the current socio-economic model not all SDGs can be achieved together.

conservation agendas to minimize trade-offs and foster synergies (Obersteiner *et al.*, 2016). In another recent study using the IMAGE integrated assessment model, van Vuuren *et al.* (2015) have shown that achieving 2050 goals for both biodiversity and hunger would require a substantial increase in agricultural productivity per hectare, to accommodate a 50-70% increase in demand for food while halting the conversion of natural habitats. Another study found that implementing ambitious protected area expansion plans, under business-as-usual socio-economic trends, may result in a shortfall in productive land, as well as displacement of agricultural areas with consequential socio-economic

Table ④ ⑥ ① Synergies and trade-offs between different sustainability objectives.

Colours indicate synergies (green) and trade-offs (red) in various intensities. Source: van Vuuren *et al.* (2015).

	Eradicate hunger	Halting biodiversity loss	Access to energy	Reduce air pollution	Mitigate climate change	Access to clean water	Balance nitrogen cycle
Eradicate hunger					More emissions from increased production (fertiliser. land expansion tractors) (*)	Increased water use for agriculture (*)	More emissions from increased production (fertiliser, manure (*)
Halting biodiversity loss	Less land for food production (*) Preservation or ecosystem services helps safeguard long-term food supply			Intact ecosystems contribute to better air quality	Fewer CO_2 emissions from land conversion and agriculture, new CO_2 sinks (*)	More gradual and uniform water flow, cleaner water Increased water use by permanent vegetation	More contribution of ecosystems in balancing nitrogen cycle
Access to energy	Increases income opportunities due to reduced time for fuel collection, better health	Less disturbance of local biodiversity for food collection		Less indoor and urban air pollution (*)	New emissions from modern energy offset by reduced traditional energy emissions (*)	Water requirement for power generation (small) (*)	
Reduce air pollution	Less negative impact of air pollution on crop yields	Less air pollutions impacts on biodiversity (*)	Higher energy prices		Depends on which air pollutants are reduced (*)	Less water pollution	Helps to reduce nitrogen deposition (*)
Mitigate climate change	Reduces negative impacts on yields (but also positive impacts) (*) Bio-energy competes for land with food production	Reduces negative impacts of climate change (*) Additional land for bio-energy (*)	Higher energy prices (*)	Less emissions of air pollutants due to lower fossil fuel use (*)		Negative impacts on precipitation patterns and evapotranspiration reduced (*)	Some positive impact N_2O emission reduction (*)
Access to clean water	Improved water for cooking Competition between agriculture and domestic purpose						
Balance nitrogen cycle	Reduction of fertiliser use (but also prevents toxic fertiliser levels)	Reduces pollution		Reduces air pollution	Some reduction of N_2O emissions		

Note: *denotes that the linkages is addressed quantitatively by the modelling framework.

impacts (Visconti *et al.*, 2015). Eradicating extreme poverty however, does not necessarily mean jeopardizing climate targets, even in the absence of specific climate policies and technological innovations (Hubacek *et al.*, 2017). Di Marco *et al.* (2016) explored the interactions between Aichi Biodiversity Targets 5 (reducing the loss of natural habitat), 11 (expanding the global coverage of protected areas) and 12 (conserving threatened species). They showed that the expansion of the global protected areas to 17% of land area resulted in different priorities of sites depending on whether the goal was to reduce habitat loss or conserve species. In addition, expanding protected area coverage to 17% to conserve threatened species would result in safeguarding 30% more carbon stock than targeting areas under high deforestation rates. The reason is that areas under rapid deforestation are not necessarily those with the highest capacity to stock carbon. While the figures relate to the Aichi Biodiversity Targets for 2020, the same trade-offs are likely to apply to post-2020 biodiversity targets. **Table 4.6.1** highlights some of the most significant synergies and trade-offs between different objectives associated with the Sustainable Development Goals.

Further modelling on policy targets that explicitly embodies nature into scenarios is of outmost importance. Scenarios developed for global environmental assessments have explored impacts of direct and indirect drivers on nature but have not embedded nature in the scenario itself. The effects of alternative pathways of socioeconomic development on nature have thus been assessed as one-way outcomes, ignoring the possible feedbacks of nature on the system (Rosa *et al.*, 2017). Existing scenarios ignore policy objectives related to nature protection. As targets for human development become increasingly connected with targets for nature, such as in the SDGs, the next generation of scenarios should explore alternative pathways to reach these intertwined targets and address feedbacks between nature, nature's contributions to people, and human well-being. Several desirable properties of this new generation of scenarios have been identified, including the use of participatory approaches, the integration of stakeholders from multiple sectors (for example, fisheries, agriculture, forestry) (Rosa *et al.*, 2017), and addressing decision makers from the local to the global scale (Biermann *et al.*, 2017).

4.6.2 How can the evidence from scenarios contribute to the development of future biodiversity targets and the 2050 vision?

Scope: How can scenarios and models help to reformulate the new set of targets? To address this issue, this section uses the Aichi Biodiversity Targets for 2020 as templates for setting the next generation of targets. Only a subset of the targets is discussed, with the purpose to demonstrate the type of considerations that should underpin the new targets. Existing scenarios and models for biodiversity and ecosystem services are used to explore: i) how targets can be formulated in ways that can more easily be understood and evaluated by both policymakers and practitioners; ii) which kinds of indicators, that come from observations and scenarios, can be used to evaluate progress towards the objectives of this target; and iii) what scenarios and models tell us about ambitious vs. aspirational targets, i.e. whether they can be achieved under plausible conditions represented by a variety of exploratory scenarios of societal and economic development.

4.6.2.1 Habitat loss and degradation (Target 5)

"By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced."

Analyses based on satellite remote sensing identified over the period 2000-2012 a net global loss of ca. 1.5 million km^2 of forest (Hansen *et al.*, 2013), including substantial loss of structurally intact pan-tropical forests (Tyukavina *et al.*, 2016). At current trends, even the target specified in the New York Declaration of Forests (to halve the rate of natural forest loss by 2020) is highly unlikely to be achieved (Zarin *et al.*, 2016). Under most future scenarios, the future net loss of natural habitats is partly counterbalanced by secondary regrowth. This is true for both forest and non-forest natural habitats (Hurtt *et al.*, 2011). Secondary habitat types typically host a fraction of the biodiversity present in primary habitats of the same type (Alkemade *et al.*, 2009; Newbold *et al.*, 2013), and this fraction depends on the integrity and age of the secondary vegetation. Therefore, numeric targets for the rate of loss of natural habitat are insufficient to capture the complex dynamics of habitat change, and the proportion of biodiversity that they retain compared to pristine habitats should also be considered.

From a scenario and modelling perspective, assessing the current and future state of forest globally is challenging for a number of reasons: 1) very different classifications as to what is a forest and which forest is considered intact, which one degraded (Alexander *et al.*, 2017c; Thompson *et al.*, 2013); 2) Most land-use change scenarios do not yet tend to consider environmental policies such as the Aichi Biodiversity Targets, the SDGs or REDD+ (Alexander *et al.*, 2017c; Eitelberg *et al.*, 2016, 2015; Popp *et al.*, 2017); 3) Integrated Assessment models that are often used to produce scenarios typically do not have the forest sector explicitly included at their core (Schmitz *et al.*, 2014); 4) Models that seek to assess future ecosystems from state of, e.g., carbon cycle and climate regulation perspective do not yet account well for forest (or other habitat) management (Arneth *et al.*, 2017).

In principle, activities to achieve Target 5 could have large co-benefits with achieving Targets 11 and 17, if protected area expansion could be dedicated to cover habitats of both high species density (in particular threatened or rare species) and regions of high carbon density (Di Marco *et al.*, 2016). Under otherwise unchanged conditions, scenarios in which multiple demands for land resources are aimed to be met resulted in intensification of croplands (adding to the "land sharing/land sparing" debate) and enhanced areas with tree cover (Eitelberg *et al.*, 2016). However, accounting for demand for protected area had no effect on reducing the projected loss of grassland, compared to business-as-usual (Eitelberg *et al.*, 2016). Maximizing forest habitat conservation as well as forest species conservation was estimated to be possible in 73% of the area identified to be also most appropriate for expanding the current protected area to meet Target 11 (Di Marco *et al.*, 2016).

Recent and projected trends in population growth and lifestyle (e.g., dietary changes), jointly with enhanced requirements for bioenergy crops are expected to maintain large pressures on further cropland expansion (Alexander *et al.*, 2017a; Eitelberg *et al.*, 2015). Agriculture is one of the largest drivers of biodiversity loss, and a large source of greenhouse gases and pollutants (McLaughlin & Kinzelbach, 2015; Newbold *et al.*, 2015). Therefore, achieving conservation goals alongside meeting demand for food and fibre, water, bioenergy and climate mitigation will require a dedicated effort that considers both changes in supply and demand, as well as equitable trade (Alexander *et al.*, 2017a; McLaughlin & Kinzelbach, 2015).

4.6.2.2 Sustainable fisheries (Target 6)

"By 2020, all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches [...]."

Whilst the objectives of Target 6 are relatively clear, some terms remain imprecise. The primary facets of the Target which remain loosely defined are the concepts of 'safe ecological limits' and 'no significant adverse impacts' (also the issue of 'vulnerable' ecosystems; see Target 10). 'Safe ecological limits' as a term lacks indication of whether these limits are structural (e.g. maintenance of facets such as ecosystem trophic structure or species composition) or functional (e.g. continued provision of goods and services). Without clarification, it is then difficult to know what aspects of ecosystems should be maintained, nor the level of degradation that is to be tolerated. Furthermore, the margins of safety are not clearly specified –how are these limits to be measured, quantified, and monitored? How close to the 'safe ecological limit' is acceptable? Finally, the term "safe limits" has been used with many contexts including the planetary boundary framework (Steffen *et al.*, 2015) and, therefore, might benefit from clarification. It is important not to confound 'safe ecological limits' and 'safe limits for humanity' since these refer to very different reference baselines, as well as very contrasted spatial and temporal scales.

Regarding 'no significant adverse impacts', the lack of specificity here is to do with the meaning of the word 'significant' (note that Target 5 also includes this terminology). Scientifically, 'significant' generally has a statistical meaning, indicating evidence at some level of likelihood that an effect is not attributable to chance. It seems unlikely that this is the intended meaning here, but significant can be so broadly interpreted as to make consistency of application across national and regional scales extremely challenging.

Quantification of progress towards this target through appropriate indicators has shown that at least some indicators exist for monitoring resource state (e.g., the proportion of fish stocks within safe biological limits), the pressures on it (e.g., global effort in bottom trawling), and fisheries responses to pressures on fish stocks (e.g. Marine Stewardship Council certified fisheries). However, indicators of whole ecosystem (as opposed to stock) status and recovery plans remain limited or absent, and the scope and alignment of existing indicators varies. Recent focus has been put on ecosystem-based indicators for assessing the state of exploited species and the ecosystems they are embedded in (Coll *et al.*, 2016; Shin *et al.*, 2012), some of which have been retained in the list of IPBES "Highlighted indicators" but still lack global scale coverage for nations to be able to report routinely (proportion of predators, mean fish size).

Projecting plausible futures for marine and aquatic biological resources is aided by the fact that there has been a long history of model development for these systems, with a particular profusion of models emerging over the past decade or so (Fulton, 2010). Models range from single species stock assessment models to whole ecosystem approaches, and in some cases such models incorporate large parts of the socio-economic and management components as well as the biological ones (Nielsen *et al.*, 2018). The heterogeneity of models is also beginning to be addressed by applying standardised ensemble modelling approaches across specified scenarios (Tittensor *et al.*, 2018b), akin to model intercomparison studies in the climate and earth science communities. Perhaps more challenging is the specification of socio-economic storylines that can then be translated into projections that can be used to force ecosystem models. While storylines have recently been in development at both regional (CERES, 2016) and global (Maury *et al.*, 2017) scales, specifying how the developments in economics, management, and governance that are outlined in scenarios can then be used to force models, especially spatially explicit models, is difficult. Furthermore, management and stewardship of marine resources remain varied among nations in terms of capacity, approach, and effectiveness (Bundy *et al.*, 2017). Management regimes can also change radically and rapidly in response to changes in national policy environments (e.g., the enactment and amendments of the U. S. Magnuson-Stevens Fishery Conservation and Management Act), and resource management plays an integral role in terms of the status of both target species and ecosystems (FAO, 2016), and furthermore adaptation to a changing climate. Nonetheless, the continued development of scenarios, together with the broad and growing range of marine ecosystem models at multiple scales, suggests that Target 6 can be usefully and increasingly informed by their application.

Broadly speaking, the development of future policy targets needs to further incorporate the role of climate change on the sustainability and use of aquatic resources. Furthermore, objectives may need to be reframed or at least clarified in order to address the challenges of measuring 'significant adverse impacts' and 'safe ecological limits' whilst still allowing for national level variation in how objectives are attained and recognizing differences in capacity for stewardship of aquatic resources. When specifying targets, it also needs to be made clear whether the goal is maintaining ecosystem structure, the provision of goods and services (including contributions to food security), or both. Currently, there is also potential overlap between Targets 6 and 7, in that Target 6 includes the management and harvest of fish and invertebrate stocks and aquatic plants, which will be increasingly linked to the development of aquaculture in the future that is addressed in Target 7 (section 4.2.2.3.1). Given the continued growth in the importance of aquaculture, its impacts on broader ecosystem health, including indirect effects such as fishing wild stocks to provide fishmeal for aquaculture (not explicitly mentioned in Target 7, but implicitly included in Target 6) needs to be further integrated into future targets. Similarly, at present there is overlap with Target 10, since anthropogenic impacts on coral reefs (and other vulnerable aquatic ecosystems) include those integrated into Target 6.

4.6.2.3 Sustainable agriculture (Target 7)

"By 2020, areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity."

The scientific community has been engaged in a controversial debate about whether biodiversity conservation can better be achieved by improving habitat availability and quality on farmland (i.e. through wildlife-friendly farming – "land sharing"), or whether it is dependent on natural habitat and thus requires high-yielding agriculture to reduce land requirements (i.e. sparing land for nature – "land sparing") (Fischer *et al.*, 2014; Phalan *et al.*, 2011). But recently consensus has started to develop that convey that different strategies are needed in different contexts and for different conservation goals (Ramankutty & Rhemtulla, 2012) and that sustainable agricultural management includes both measures to increase on-farm habitat quality, as well as increasing productivity while minimizing harm to biodiversity (Clough *et al.*, 2011; Kremen, 2015; Seppelt *et al.*, 2016).

Currently, however, it appears unlikely that we will achieve either wildlife-friendly farming or stop the conversion of natural habitats by 2050 if current trends continue. Crop production is projected to increase by 50-100% by 2050 to meet future demand under current population and diet trends (Alexandratos & Bruinsma, 2012; Tallis *et al.*, 2018; Tilman *et al.*, 2011). According to a comparison of the best state-of-the-art land-use models, the combined effect of projected climate change, as well as middle of the road population and economic development projections, would result in an expansion of global cropland by about 20% by 2050 (Schmitz *et al.*, 2014). Business-as-usual trends would also result in the further conversion of >50% of natural habitats to croplands in important ecoregions like Mediterranean forests and temperate grasslands (Tallis *et al.*, 2018). In addition to this conversion of natural habitats, fertilizer use, which has large negative impacts on biodiversity and ecosystem services especially in freshwater systems, is projected to increase by 58% by 2050 (Alexandratos & Bruinsma, 2012). Wildlife-friendly farming methods are still restricted to comparatively small areas: only about 1% of global agricultural land is, for example, managed organically (Willer & Lernoud, 2017), and approximately 7.5% of it is managed with agroforestry with more than 50% tree cover (Zomer *et al.*, 2009).

Numerous analyses show, however, that achieving sustainable agriculture that produces enough food for everyone while ensuring conservation of biodiversity is possible, if far-reaching food system changes are implemented. Recent scenario analyses have shown that globally enough food could be produced for everyone in 2050 on existing agricultural land, while halting deforestation and protecting 17% of the world's terrestrial habitats if we shifted towards more sustainable diets, reduced food waste and closed yield gaps (Erb *et al.*, 2016; Foley *et al.*, 2011; Muller *et al.*, 2017; Tallis *et al.*, 2018; West *et al.*, 2014). A recent study, for example, estimated that by closing yield gaps and optimizing where crops are grown, >50% of each of the world's biomes could be set aside, while still producing enough food for all people in 2050 (Tallis *et al.*, 2018). Similarly, organic agriculture could be used as a wildlife-friendly agricultural management strategy, if combined with other food system strategies, e.g. reductions in food waste and changes in livestock feed composition, to provide enough food for people in 2050 on current agricultural land while also reducing pesticide use and nitrogen pollution (Muller *et al.*, 2017). These various scenarios show that both land-sharing and land-sparing strategies would be possible to help conserve biodiversity while feeding humanity if broad food system changes were implemented.

4.6.2.4 Vulnerable ecosystems (Coral Reefs) (Target 10)

"By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning."

The Global Biodiversity Outlook 4 (GBO-4), which evaluated progress towards the Aichi Biodiversity Targets in 2014, focused on the aspects related to climate change impacts on tropical coral reefs and the importance of reducing multiple pressures to minimize these impacts – and concluded that this target had been missed. Observations, experiments and models provide sound arguments for maintaining a strong priority on tropical coral reefs due to their exceptional vulnerability to climate change (IPCC, 2018). Warm-water coral reefs are one of the most biodiverse marine ecosystems in the world and provide a wide range of ecosystem services, especially to people living in tropical regions (CBD, 2014). They are also one of the most rapidly degrading ecosystems globally due to a combination of many pressures including pollution, overexploitation and ocean warming (see sections 4.2.2.2.2, **Box 4.2.3** in section 4.2.2.3.1; Butchart *et al.*, 2010; CBD, 2014; IPCC, 2018). Models and observations indicate that tropical coral reefs are exceptionally vulnerable to future ocean acidification and warming due to their very high sensitivity to these factors compared to most other systems (Bay *et al.*, 2017; Gattuso *et al.*, 2015; IPCC, 2018). Models project that there will be significant negative impacts even if the most ambitious targets of the Paris agreement of limiting global warming to 1.5°C are achieved (IPCC, 2018). For higher CO_2 emissions and warming scenarios, models project severe degradation of nearly all tropical coral reefs and the limits of natural adaptation and ecosystem management to preserve the integrity of these ecosystems will be exceeded (Bay *et al.*, 2017; Gattuso *et al.*, 2015).

Observations and models also indicate that all ecosystems are vulnerable to climate change or acidification to some extent (IPCC, 2014). Some ecosystems are projected to be particularly vulnerable because exposure to climate change is high – these include Arctic tundra and ocean ecosystems where warming is projected to be higher than elsewhere on the globe (Settele *et al.*, 2014). Other ecosystems are projected to be especially vulnerable due to their high sensitivity to climate change or acidification, and little space for adaptation – in addition to coral reefs, these include mountain terrestrial and freshwater ecosystems, tropical ecosystems, and deep oceans (section 4.2.2.2.3; Settele *et al.*, 2014). All ecosystems of the world are projected to experience changes in species composition and abundance due to species ranges shifts and modifications of ecosystem function caused by rising CO_2 and climate change (IPCC, 2014). A consensus ranking of ecosystem vulnerability to climate change is not available due to unsettled scientific debates and uncertainty in modelled impacts (e.g., Settele *et al.*, 2014).

Because there is a lack of consensus on the vulnerability of ecosystems to climate change outside of coral reefs, this target currently suffers from a lack of clarity. This target has been dubbed "Vulnerable Ecosystems" for shorthand (Aichi Passport, UNEP-WCMC) and covers "other vulnerable ecosystems", which poses problems of definition because all ecosystems are vulnerable to climate change or acidification to a greater or lesser extent (IPCC, 2014). As such, this target has been associated with a loosely related set of indicators, some very narrow and others overly broad, that are used to assess progress towards this target; for example, the Biodiversity Indicators Partnership lists the Ocean Health Index (extremely broad), Climatic impacts on European and North American birds (taxonomically and spatially restricted), Red List Index for reef-building corals (not well targeted for climate change impacts), and Cumulative Human Impacts on Marine Ecosystems (exceptionally broad) as indicators for this target.

There is strong evidence that reducing other stresses on ecosystems will generally improve the capacity of ecosystems to adapt to climate change. For tropical coral reefs, reducing nutrient loading and maintaining or reinforcing herbivorous fish populations helps reduce the competition by algae and these and other measures are projected to substantially improve the capacity of coral reefs to maintain their integrity in the face of climate change (Box 4.3.2 in section 2.2.3.1; Gattuso *et al.*, 2015; Kennedy *et al.*, 2013). Other examples include the importance of halting terrestrial habitat fragmentation and increasing connectivity between natural habitats to allow species to move so that they can track favourable climates (Imbach *et al.*, 2013).

Public policy and ecosystem management strategies for adaptation to climate change are being developed and deployed for some ecosystems. Forest managers, for example, have been very active in developing climate adaptation strategies based on projected impacts of climate change on trees, some of which depend on maintaining or reinforcing genetic and species diversity of trees and protecting ecosystem integrity (Keenan, 2017). However, not all climate change adaptation strategies for ecosystems are biodiversity friendly; for example, some forest adaptation strategies put an emphasis on the introduction of fast-growing alien tree species (Keenan, 2017). Evidence-based action plans for tropical coral reefs are in place for some reef systems, and most put an emphasis on maintaining ecosystem integrity as a key to enhancing resilience and resistance to climate change and acidification (e.g., Great Barrier Reef Climate Change Adaptation Strategy and Action Plan, see also Gattuso *et al.*, 2015; Kennedy *et al.*, 2013). Scientists are also actively exploring other strategies requiring much more active intervention such as protective sun screens, cultivation of warming adapted corals and climate geoengineering (Kwiatkowski *et al.*, 2015; van Oppen *et al.*, 2015).

These considerations suggest that future policy targets could highlight the relationships between climate change adaptation and biodiversity protection. They could include relatively broad objectives that are common to all climate adaptation strategies for ecosystems, as well as a particular emphasis on tropical coral reefs, focusing on: the vital importance of meeting the 2°C goal, and if possible the 1.5°C goal of the Paris Agreement in order for adaptation to be effective in highly vulnerable ecosystems *(new emphasis)*; the need to reduce multiple pressures on all vulnerable ecosystems, so as to improve their resistance and resilience in the face of climate change and acidification *(maintained emphasis)*; the key role of developing and implementing climate change adaptation measures for all ecosystems with a wide range of stakeholders that take into account the protection of biodiversity and emphasize the importance of nature-based adaptation strategies *(new emphasis)*; the need to develop strategies of societal response to projected inevitable changes in highly vulnerable systems *(new emphasis)*; and the special and urgent need to develop protection and adaptation measures for tropical coral reefs *(maintained emphasis).*

Models and other considerations also suggest that a more focused set of indicators would be helpful for monitoring progress towards such a target. For example, trends and projections of sea surface temperatures, ocean acidity, coral reef bleaching events, proxies of marine nutrient loading in coral reef areas, etc. are readily available from observations and models and may be much better adapted to monitoring progress towards a component focusing on tropical coral reefs than very broad indicators of ocean health or human impacts on marine ecosystems.

4.6.2.5 Protected Areas and other Effective Area-based Measures (Target 11)

"By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved [...]"

While the world may be on track to meet or exceed the numeric target of protecting globally 17% of the land and 10% of the oceans by 2020 (Chapter 3), other aspects of the target, including the global connectivity and representativity of protected areas, and their coverage of areas important for biodiversity (including Key Biodiversity Areas), have made little or no progress (Butchart *et al.*, 2015; Santini *et al.*, 2016). These aspects may be more important that numeric targets per se, as demonstrated by the evidence that if new protected areas between 2004 and 2014 had targeted unrepresented threatened vertebrates, it would have been possible to protect >30 times more threatened species for the same area or cost as the actual expansion that occurred (Venter *et al.*, 2014).

In theory, it would be possible to hit much larger numeric targets for protected areas in the future. Depending on scenarios, between 30-40% of the land would remain primary (forest or non-forest) habitat in 2050, and artificial land-use types (urban, cropland and pasture) would occupy 30-40% of the land (Hurtt *et al.*, 2011). In practice, much land is already degraded by processes that can spread globally including climate change and invasive species, thus restoration will be required in addition to protection (IPBES, 2018a).

The uneven distribution of biodiversity (Butchart *et al.*, 2015), projected expansion of human population, and regional differences in projected land-use change (Hurtt *et al.*, 2011) suggest that global percentage targets do not necessarily achieve effective biodiversity conservation. Indeed, an analysis looking at Target 11 for 2020 (Visconti *et al.*, 2015) showed that expanding protected areas to protect 17% of the land while minimizing the opportunity cost for people (i.e. by prioritizing protection of unpopulated areas) would reduce habitat available to threatened mammals. The reason is that threatened mammals occupy areas densely populated by humans, and protecting unpopulated areas displaces further land conversion in highly populated areas. In addition, climate change may change dramatically the suitability of protected areas for their native biodiversity in the future (Hole *et al.*, 2009; Loarie *et al.*, 2009). Therefore, dynamic scheduling (Wilson *et al.*, 2007) based on scenarios of climate and land-use change and allowing species to move across landscapes to track suitable habitat and climatic space should be used to translate numeric targets into allocation of protected areas in space and time (Pressey *et al.*, 2007).

4.6.2.6 Preventing Extinctions and Improving Species Conservation Status (Target 12)

"By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained."

Forecasts of species decline are blurred by several sources of uncertainty. While scenarios exist for climate change and land-use change (which can be used to derive habitat loss), for other direct drivers of species loss, including invasive species, overexploitation, disease spread, scenarios are lacking. These drivers and their impacts start being projected into the future though rarely at global scale and with wide coverage of species biodiversity, but they will interact with or add up to land use and climate change, intensifying species declines. Interactions among drivers have only partly been explored (e.g., climate and land-use change; Mantyka-Pringle *et al.*, 2015). Even projections based on the same driver can differ widely. For example, the proportion of species that is projected to go extinct based on climate change varies with model assumptions (amount of extinction debt, species' ability to disperse) and modelling technique (species-area curves: 22% extinctions; mechanistic or correlative models: 6-8% extinctions) (Urban, 2015). Uncertainty on the species' response to global change (adaptation / plasticity, dispersal, or local extinction) is also reflected in uncertainty in the scenario outcome (Rondinini & Visconti, 2015). Finally, extinctions are fundamentally stochastic events caused by extinction vortexes (Soulé, 1986), which are difficult to predict and prevent.

Despite wide uncertainty in the projections, business-as-usual scenarios produce substantially different outcomes compared to scenarios having a strong focus on sustainability typically (Alkemade *et al.*, 2009; Newbold *et al.*, 2015; Visconti *et al.*, 2016). Assuming that species can cope with climate change, sustainability scenarios can almost halt their decline due to land-use change (Rondinini & Visconti, 2015). This, in addition to the evidence that conservation action alone is insufficient (Butchart *et al.*, 2010; Hoffmann & Sgrò, 2011; Tittensor *et al.*, 2014) suggests that halting biodiversity loss for some indicators such as population size or average conservation status is within the boundaries of scenarios, provided that a mixed strategy of stepped up conservation action and societal changes is adopted. However, the stochasticity of extinctions means that even in the best-case scenario, considering the current depauperate state of biodiversity, some extinctions may still occur.

4.6.2.7 Ecosystem Restoration and Resilience (Target 15)

"By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through [,,,] restoration of at least 15 per cent of degraded ecosystems [...]"

The main issue with quantifying degradation and restoration is the lack of a clear baseline (IPBES 2018e). Several possible baselines can be chosen as a reference for restoring degraded land, including pre-modern (<10,000 years BCE), historical (typically between 300 and 50 years ago), counterfactual (how an ecosystem would look like in the absence of human pressures). For this reason, the scientific community has not been able to provide a detailed global assessment of land degradation, and different models estimate the proportion of degraded land between 7-40% (Gibbs & Salmon, 2015; Van der Esch *et al.*, 2017).

Given the uncertainty in the quantification of current land degradation, scenario analysis cannot provide strong quantitative predictions around restoration, but boundaries for restoration opportunities can be identified. According to the World Resource Institute, over 20 million km^2 of degraded tropical and temperate forests would be suitable for restoration (Laestadius *et al.*, 2011). Extending afforestation further, to non-forest biomes, would have significant negative effects on ecosystem services (Veldman *et al.*, 2015) as well as inevitably on the biodiversity adapted to these biomes. A trade-off between restoration of natural ecosystems and bioenergy production exists, since under a business-as-usual scenario, limiting warming to 2° C will require an expansion of bioenergy production to abandoned and degraded land (Dauber *et al.*, 2012; Nijsen *et al.*, 2012) to achieve negative emissions from biofuels (van Vuuren *et al.*, 2011).

4.7 DEALING WITH UNCERTAINTY, SPATIAL SCALE AND TEMPORAL SCALE ISSUES WHEN MOBILIZING SCENARIOS AND MODELS FOR DECISION-MAKING

4.7.1 Scenarios and models help prepare decision makers for uncertainty and long-term thinking

In the IPBES methodological assessment of scenarios and models, Ferrier *et al.* (2016) provide several examples of the use of scenarios and models in support of decision-making and policy. The methodological assessment highlights, in particular, the importance of matching the spatial and temporal scales of scenarios and models to the needs of the specific policy and decision context, and of identifying sources of uncertainty, communicating uncertainty in a transparent way to decision makers and providing tools to deal with uncertainty.

When these issues are dealt with appropriately, scenarios and models can help people prepare for future uncertainty, promote long-term thinking and broaden perspectives. For example, Johnson *et al.* (2016) found that reading scenarios of future land-use changes increased the willingness of a wide range of stakeholders to participate in land-use planning. Scenarios and models have also proven to be effective tools for engaging indigenous and local knowledge holders in planning management of socio-ecological systems (Ferrier *et al.*, 2016; Hartman *et al.*, 2016; Oteros-Rozas *et al.*, 2015). Ground truthing through monitoring, especially with engagement of stakeholders, is a valuable approach for reducing uncertainties (Robinson *et al.*, 2017). **Box 4.7.1** provides examples of the use of scenarios and models in support of decision-making, with a focus on the role of uncertainty and scale.

Box ④ ⑦ ① Case studies of uncertainty and scale in decision-making using models and scenarios.

Example 1: Forest management and climate change – Forest managers are very actively using scenarios and models to develop management strategies for dealing with climate change because tree growth is very sensitive to climate and because trees generally live a long time, often more than a century, before they are harvested (Keenan, 2015). Forest managers often desire very fine spatial resolution climate projections (ca. 1 km^2) in order to make site-based management decisions, and the climate modeling community has made tremendous efforts to downscale global scale climate projections in order to meet this type of demand from a wide range of stakeholders (Giorgi *et al.*, 2009). However, downscaling introduces new sources of uncertainty that can degrade the quality of climate projections (Stefanon *et al.*, 2015) and often contribute little to improving management strategies (Keenan, 2015). Forest managers are also often presented with projections of climate impacts on trees and forests based on a single type of impact model. However, several model inter-comparisons show that different types of models – for example, correlative and mechanistic models – often give very contrasting projections of tree growth and distributions in response to future climate change (Cheaib *et al.*, 2012). High uncertainty in future global climate projections, high uncertainty in modeling impacts on trees and uncertainties introduced when downscaling climate projections have left many forest managers in a quandary about how to plan for climate change. Current recommendations focus on managing for uncertainty by employing forest management schemes that are robust under a broad range of climate and impact projections, for example by increasing resilience, by managing for higher genetic and species diversity, or by promoting natural regeneration (Cheaib *et al.*, 2012; Keenan, 2015). More importantly, there is a growing recognition that adaptive strategies for dealing with an uncertain future must be developed much more inclusively by creating partnerships between researchers from multiple disciplines, forest managers and local actors including indigenous communities in many cases (Keenan, 2015).

Example 2: Climate change and biodiversity at national and regional scales – The PARCC West Africa Project (Belle *et al.*, 2016) conducted a biodiversity risk and adaptation assessment using a combination of IPCC AR5 global scale climate projections, together with finer scaled assessments driven by higher resolution climate downscaling for five focal countries. While uncertainty in temperature projections was reduced through confirming consensus between local and global model projections, uncertainty in rainfall projections remained high in many areas, even though only one general circulation model was applied. A representative range of scenarios was used to assess risks to biodiversity especially in the context of protected area networks, and from this to design adaptation strategies and build regional capacity to enhance implementation. Multi-country efforts were integrated from local to regional scales to develop policy recommendations for climate change adaptation and management at national and regional levels.

Example 3: Participatory scenarios at local scales – Oteros-Rozas *et al.* (2015) reviewed 23 case studies of place-based participatory scenarios to assess the characteristics, strengths and weaknesses of participatory modeling. All but one study involved local communities, most included members of local governments and sixteen involved indigenous communities. Qualitative storylines in the form of drawings, or illustrations were the most common output (Figure 1), but most participatory processes also produced reports and scientific publications. Local communities were the most common primary audience, and fifteen studies had the explicit objective of informing policy or decision-making. Uncertainty was examined in sixteen of the studies, most focusing on uncertainty in drivers. Only six studies explicitly accounted for drivers or impacts at spatial scales above the local scale under consideration. The authors concluded that well-designed participatory processes enriched both local environmental management and scientific research by generating shared understanding and fostered thinking about future planning of social-ecological systems. Unfortunately, in most cases there was insufficient follow-up to determine the contribution to long-term policy or management outcomes. Numerous additional examples can also be found at the consortium of 'companion modeling' (www.commod.org).

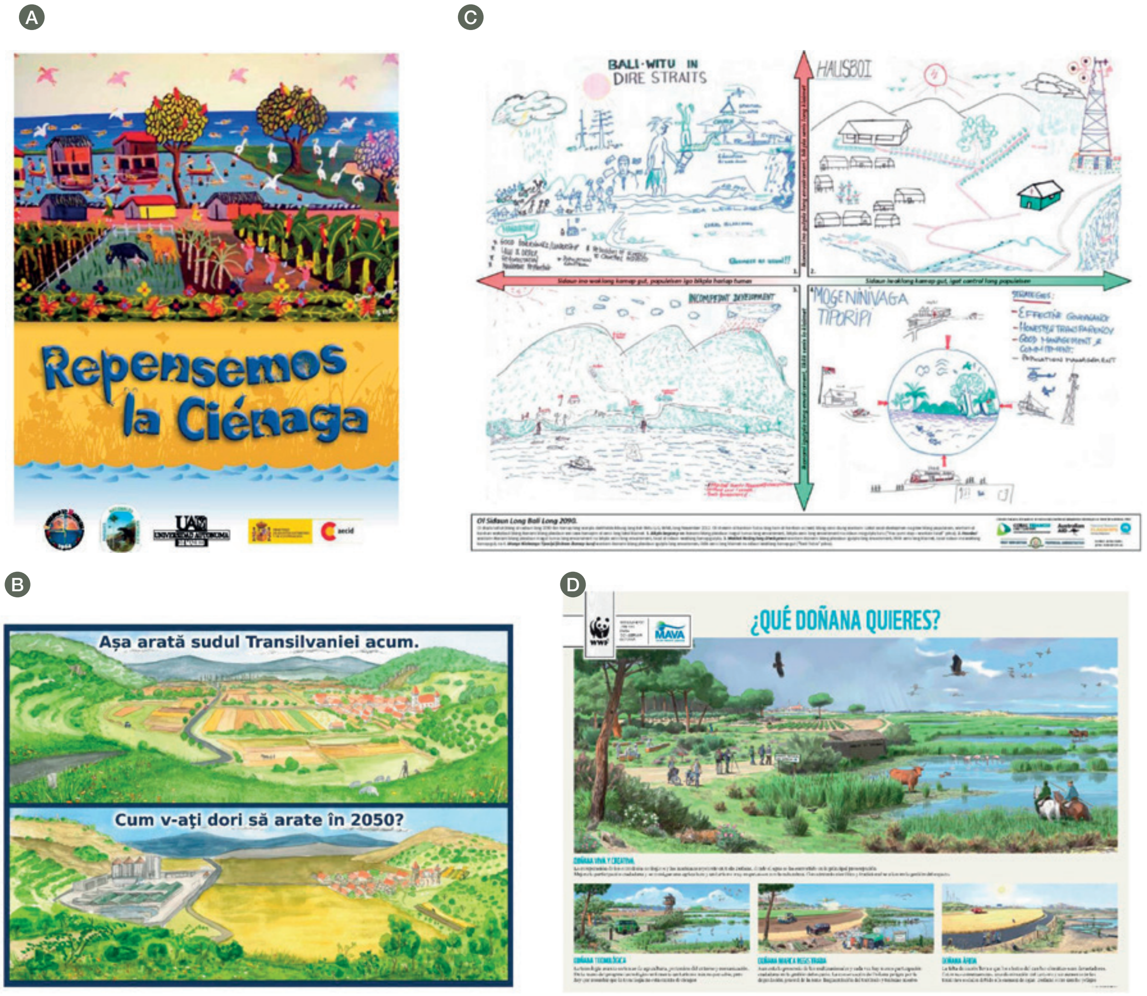


Figure ① **Examples of outreach material used for communicating scenarios results:**

Ⓐ leaflet of the Ciénaga Grande of Santa Maria case in Columbia; Ⓑ postcard of the Southern Transylvania case in Romania; Ⓒ poster of the drawing of the four scenarios of the Papua New Guinea case; and Ⓓ poster of the socio-ecological system of Doñana Protected Area case in Spain (from Oteros-Rozas *et al.*, 2015).

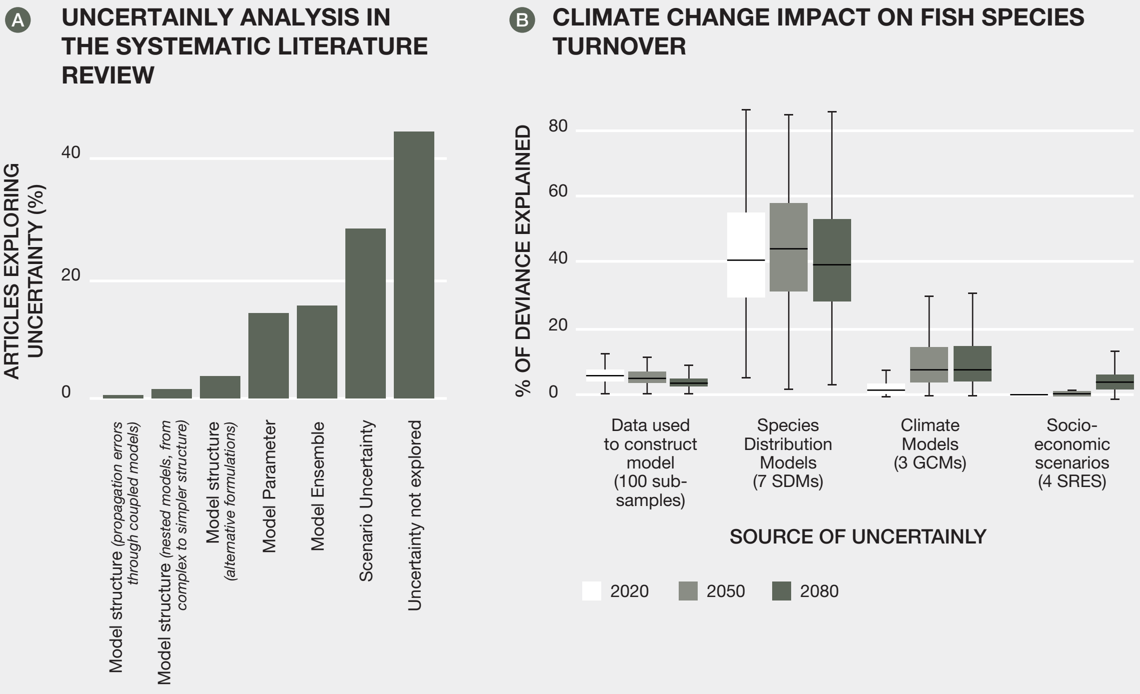
4.7.2 Dealing with uncertainty when using scenarios and models to support decision-making

Uncertainty in scenarios and models arises from many sources including insufficient data for development and testing, inadequate representation of complex socio-ecological systems and intrinsically low predictability of the system being analyzed (Ferrier *et al.*, 2016). The importance of these sources of uncertainty differs greatly between scenarios of direct and indirect drivers and models of impacts on nature and NCP (Brotons *et al.*, 2016; Ferrier *et al.*, 2016). As noted in the introduction of this chapter, the exploratory scenarios assessed in this chapter can help address the high level of uncertainty in many components of direct and indirect drivers by exploring a wide range of plausible futures (Pichs-Madruga *et al.*, 2016). Evaluation of uncertainty in models of nature and NCP are typically addressed using comparisons of model outputs with data, intercomparisons of multiple types of models, sensitivity analyses and measures of error propagation in coupled models (Brotons *et al.*, 2016).

Uncertainty in scenarios and model projections is not necessarily a major obstacle to acceptance by stakeholders, especially if it does not directly conflict with their recent experiences (Kuhn & Sniezek, 1996). Indeed, despite the common perception that communication of uncertainty can lead to confusion for decision makers, recent studies show that most audiences value the communication of uncertainty in scientific evidence as opposed to oversimplification (Fischhoff & Davis, 2014; Rudiak-Gould, 2014). This highlights the importance of transparency as well as sustained, effective communication between scientists and decision makers throughout the processes of using models for decision support (Acosta *et al.*, 2016; Ferrier *et al.*, 2016). There are also a wide range of qualitative and quantitative decision support mechanisms that can help decision makers deal with uncertainty, even though these tools are underexploited in many decision-making contexts (Acosta *et al.*, 2016).

The literature survey carried out for this chapter (Appendix A4.1.1) highlights the challenges facing the scientific community in dealing with uncertainty. The majority of studies did not include an analysis of uncertainty **(Figure 4.7.2a)**. Of those that did include an analysis, most focused on uncertainty associated with different scenarios of direct and indirect drivers and less than half provided quantitative analyses of uncertainty. Relatively few studies examined multiple sources of uncertainty. This analysis shows that significant progress needs to be made in understanding, quantifying and communicating uncertainty in order for scenarios and models to be more widely used in decision-making.

In the small number of studies that have assessed uncertainty across a wide range of sources, the relative contribution of sources of uncertainty varies substantially over time, space and different measures of nature or NCP (e.g., **Figure 4.7.2b**; Payne *et al.*, 2016). These analyses also indicate that currently the largest sources of uncertainty arise from differences in model structure or application rather than data, scenarios or models of direct drivers (e.g., **Figure 4.7.2b**; Payne *et al.*, 2016). It is important to note as well that the range of scenarios typically used in many analyses may not cover plausible extremes and potential regime shifts (Leadley *et al.*, 2010; Pereira *et al.*, 2010; Prestele *et al.*, 2016).


Ⓐ UNCERTAINLY ANALYSIS IN THE SYSTEMATIC LITERATURE REVIEW

Ⓑ CLIMATE CHANGE IMPACT ON FISH SPECIES TURNOVER

Figure ④ ⑦ ② **Treatment and importance of different sources of uncertainty in models and scenarios of nature and NCP.**

Ⓐ Outcome of the analysis of the systematic literature review (Appendix A4.1.1) showing the percentage of studies that include estimates of different sources of uncertainty. Ⓑ An example of systematic analysis of sources of uncertainty in models of fish community responses to climate change (from Buisson *et al.*, 2010).

Comparisons of models and observations provide a powerful means of evaluating uncertainty in models of impacts on nature and NCP, and for communicating with decision makers. Considerable work has been done to evaluate models of ecosystem functions and some categories of NCP (e.g., ecosystem carbon stocks and fluxes; Zaehle, 2013), that indicated large variation between models, and helped improving the understanding of the capacities and limits of these models. On the other hand, models of global change impacts on species diversity, species range, habitat change and many NCP suffer from a chronic deficit of comparison with independent datasets (i.e., datasets that are entirely independent from the data used to develop and calibrate the model (Araújo & Guisan, 2006; Settele *et al.*, 2014). Those studies that have made robust comparisons between models and data indicate that agreement between models and data varies greatly between species, habitats and NCP (Araujo & Rahbek, 2006; Sitch *et al.*, 2008). It is widely acknowledged that significant progress needs to be made in comparing models and data in order for scenarios and models to be more widely used in decision-making (Araújo & Guisan, 2006; Dawson *et al.*, 2011).

There is a growing consensus that triangulation of multiple approaches, e.g., ecosystem and species models, projections based on trend extrapolation, in situ observations and experimentation, should be used to increase confidence in models (Dawson *et al.*, 2011). There are a number of efforts underway to improve international collaboration to including efforts being supported by IPBES (Rosa *et al.*, 2017; Tittensor *et al.*, 2018b).

4.7.3 The challenge of spatial and temporal scales in using scenarios and models to support decision-making

The IPBES conceptual framework emphasizes the importance of considering multiple temporal and spatial scales (e.g. local, national, regional and global scales) in understanding, assessing and managing nature and nature's contributions to people (Diaz et al 2015a, b) note that "although the biodiversity crisis is global, biodiversity distribution and its conservation status is heterogeneous across the planet; therefore, the solutions will have to be scalable to a much finer level". As such, scenarios and models used for assessments and decision support need to be developed at a wide range of spatial and temporal scales and relationships between scales need to be explicitly accounted for (Ferrier *et al.*, 2016; Rosa *et al.*, 2017).

The IPBES methodological assessment of scenarios and models highlighted the strong relationships between spatial and temporal scales, types of scenarios employed and decision-making contexts (Ferrier *et al.*, 2016; **Figure 4.7.3**). Participation of stakeholders in developing scenarios is more common and better formalized at the local scale than at regional or global ones. Local scale scenarios and models also often focus on projections over much shorter time horizons, several years to a few decades, whereas supra-national scenarios and models are often multi-decadal (Ferrier *et al.*, 2016). Local policy and decision-making more often mobilize intervention scenarios to examine policy design and implementation with the objective of providing input to decision support. At the other end of the spectrum of spatial scales, global policy and decision-making tend to rely on exploratory scenarios for agenda setting or policy review **(Figure 4.7.3)**. These relationships between spatial and temporal scale with their use within different parts of the policy cycle are important to keep in mind as a context for interpreting the analyses presented earlier in this chapter.

Explicitly accounting for linkages across spatial and temporal scales can, in some decision contexts, enhance the ability of existing scenarios and models to address the multi-scale nature of environmental policy and decision-making (Cheung *et al.*, 2016; Rosa *et al.*, 2017). For example, studies undertaken at larger scales lose the site specificity that policymakers and managers often desire. On the other hand, local case studies provide a refined understanding of local issues based on long term investigation at specific locations, but the possibility of generalizing findings is limited by the geographic coverage of the studies and the locality-specific conditions (Castella *et al.*, 2007). These are common and well-known trade-offs among precision, realism and generality one faces when constructing and analyzing models (Levins, 1966).

Existing scenarios and modeling tools and approaches typically do not capture, or poorly capture the linkages across scales, including interactions and feedbacks between them (Carpenter *et al.*, 2009; Cheung *et al.*, 2016). This is in large part due to methodological limitations that are difficult to overcome, although ambitious efforts are now addressing solutions (e.g., Purves *et al.*, 2013). The IPBES methodological assessment report on scenarios and models of biodiversity and ecosystem services explored how to address societal and ecological processes that act at multiple spatial scales, and the challenges they present for decision-making (Cheung *et al.*, 2016). Multi-scale processes can be forecasted by linking (coupling) across scales, scenarios and models developed at particular scales. This process often requires some harmonization of scenarios across spatial scales.

Harmonization across spatial scales involves upscaling (summarizing fine-scale information at coarser scale) and/or downscaling (inferring fine-scale information from coarser scale). Existing applications have greater emphasis on downscaling than upscaling. Downscaling provides information for local-scale policy making using the large

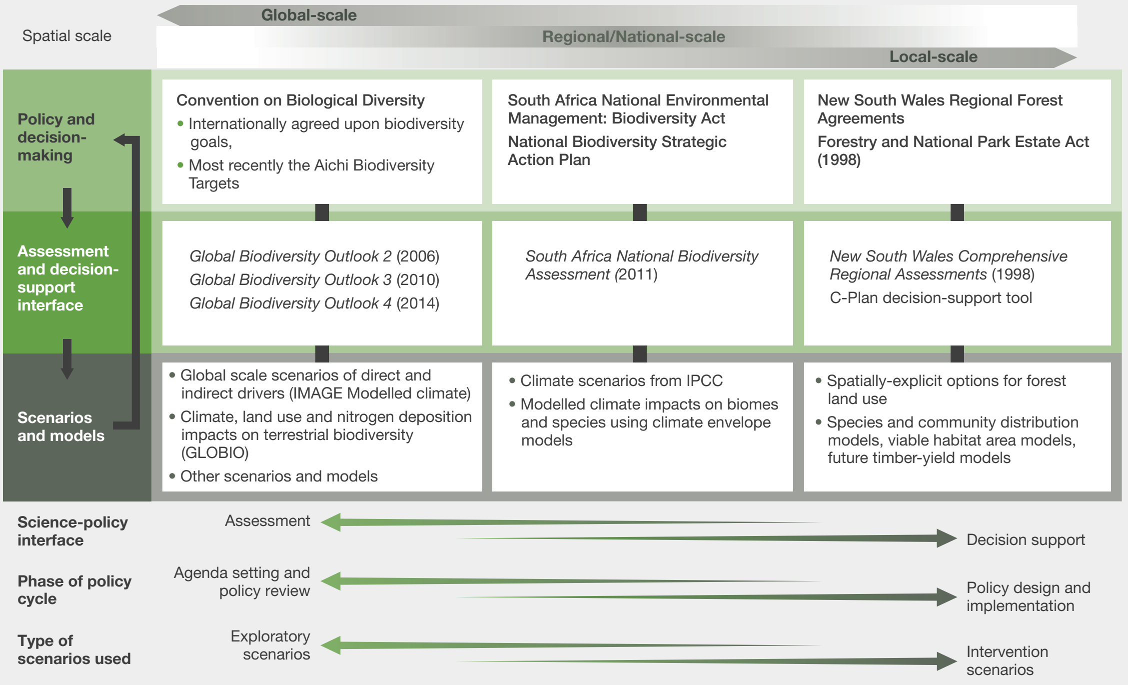


Figure ④ ⑦ ③ **Examples of the use of scenarios and models in agenda-setting, policy design and policy implementation relating to the achievement of biodiversity targets across a range of spatial scales.**

The diagram indicates the typical relationships between spatial scale (top arrows), type of science-policy interface (upper set of arrows at bottom), phase of the policy cycle (middle set of arrows at bottom) and type of scenarios used (lower set of arrows at bottom). Source: IPBES (2016b).

scale information and projections as boundary conditions and using the most refined local information to represent local processes more reliably. However, while the objective is to decrease process uncertainty at the local scale, the change of scale can introduce new sources of uncertainty, because downscaling is usually done through modelling or heuristic rules that introduce errors. Models and scenario comparison across multiple sites is another means to upscale scenarios and infer generalities, and there is a growing number of applications of this approach: Fish-Mip (Tittensor *et al.*, 2018b); IndiSeas (Fu *et al.*, 2018; Shin *et al.*, 2018); Madingley Model (Bartlett *et al.*, 2016; Harfoot *et al.*, 2014). Technical progress is being made in downscaling and upscaling, in particular by integrating data from a wide variety of sources and using powerful mathematical tools that combine spatial interpolation, upscaling, downscaling, data fusion, and data assimilation (Hoskins *et al.*, 2016; Yue *et al.*, 2016).

Despite these methodological challenges, there are substantial potential benefits of using multi-scale scenarios and models for improving understanding of system dynamics and for providing better support for decision-making. Ferrier *et al.* (2016) recommend that the scientific community works “on methods for linking […] scenarios and models across spatial and temporal scales” and in particular that IPBES works with the scientific community to “develop a flexible and adaptable suite of multi-scaled scenarios” (see also Rosa *et al.*, 2017). Approaches for developing multi-scale scenarios include using global-scale scenarios as boundary conditions for regional-scale scenarios, translating global-scale storylines into regional storylines, using standardized scenario families to independently develop scenarios across scales, and the direct use of global scenarios for regional policy contexts. These methods of upscaling can minimize inconsistencies between local scale contexts with larger scale assumptions, while also representing a diversity of local scale contexts (see Biggs *et al.*, 2007 for an example). However, substantial resources and effort are needed to coordinate the development and aggregation of multiple local scale scenarios, so it is rarely done. Of particular importance, is the post-hoc approach to scaling used in Chapter 5 of this assessment and the IPBES regional assessments that have used common (or “archetype”) scenarios in order to make qualitative linkages across spatial and temporal scales (see also Biggs *et al.*, 2007; Kok & van Delden, 2009).

However, multi-scale scenarios and models are not appropriate in every decision context, particularly when error propagation increases uncertainty to an unacceptable level. When system processes interact across scales resulting

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in nonlinear dynamics, harmonizing of models and their outputs across these scales is more prone to scaling error, therefore the uncertainty resulting from model linkages should be quantified (Cheung *et al.*, 2016), but the literature survey suggests this is rarely done (see section 4.7.2).

4.7.4 Improving communication and building capacity to enhance the use of scenarios and models in decision-making

The IPBES methodological assessment of scenarios and models highlighted cases in which scenarios and models have been successfully mobilized for policy and decision-making (Ferrier *et al.*, 2016). It also, however, identified several key factors that have limited the mobilization of scenarios and models for policy and decision-making (Acosta *et al.*, 2016). Many of these factors are related to insufficient communication between scientists and decision makers and the willingness and capacity of scientists and decision makers to engage in long-term interactions but may also run into more fundamental problems such as complex political agendas that are not compatible with the transparency associated with good scientific practice (Acosta *et al.*, 2016).

The IPBES methodological assessment of scenarios and models made several recommendations for improving the use of scenarios and models in decision-making to address these deficiencies (Ferrier *et al.*, 2016). One of the most important keys is to establish and maintain interactions between policymakers, stakeholders and scientists (see also Fiske & Dupree, 2014; Scheufele, 2014). In most successful applications, this typically involves many cycles of feedback between these groups during the development and use of scenarios and models. Sustained interactions between these groups help ensure that a relationship of trust is built between modelers and decision makers, that scenarios and models are adapted to the decision-making context, and that all parties understand the capacities and limits of scenarios and models.

Human and technical capacity for scenario development and modeling needs to be enhanced in order to address these shortcomings (Lundquist *et al.*, 2016). Recommendations for capacity building include promoting of open and transparent access to scenario and modelling tools, to data required for the development and testing, and to training programs on scenarios and models for scientists and stakeholders (Biggs *et al.*, 2018; Lundquist *et al.*, 2016).

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