Ecosystems

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Climate and Environmental Change in the Mediterranean Basin – Current Situation and Risks for the Future

First Mediterranean Assessment Report (MAR1)

Chapter 4 Ecosystems

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Executive summary

Marine ecosystems

Despite covering only 0.82% of the ocean's surface, the Mediterranean Sea supports up to 18% of all known marine species, with 21% being listed as vulnerable and 11% as endangered. The accelerated spread of tropical non-indigenous species is leading to the "tropicalization" of Mediterranean fauna and flora as a result of warming and extreme heat waves since the 1990s. The acidification rate in the Mediterranean waters has ranged between 0.055 and 0.156 pH units since the pre-industrial period, affecting the marine trophic chain, from its primary producers (i.e., coccolithophores and foraminifera) to corals and coralline red algae.

Projections for high emission scenarios show that endemic assemblages will be modified with numerous species becoming extinct in the mid 21st century and changes to the natural habitats of commercially valuable species, which would have many repercussions on marine ecosystem services such as tourism, fisheries, climate regulation, and ultimately on human health.

Adaptation strategies to reduce environmental change impacts need effective mitigation policies and actions. They require anticipatory planning to enable them to tackle problems while they are still manageable. Given the diversity of each Mediterranean sub-basin, wider monitoring coverage is needed to strengthen our knowledge about the different adaptation processes that characterize and best suit each geographical zone. Adaptation implies the implementation of more sustainable fishing practices as well as reducing pollution from agricultural activity, sustainable tourism or developing more effective waste management. Marine protected areas can potentially have an insurance role if they are established in locations not particularly vulnerable to ocean acidification and climate change.

Coastal ecosystems

The coastal zone, i.e. the area in which the interaction between marine systems and the land dominate ecological and resource systems, is a hotspot of risks, especially in the south-eastern Mediterranean region. Alterations to coastal ecosystems (lagoons, deltas, salt marshes, etc.) due to climate change and human activities affect the flow of nutrients to the sea, the magnitude, timing and composition of potentially harmful/toxic plankton blooms. They also significantly increase the number and frequency of jellyfish outbreaks, and could have negative impacts on fisheries. 1.2 to 5% of seagrass meadows in the Mediterranean Sea, which represent 5 to 17% of the worldwide seagrass habitat, are lost each year. Among them, almost half of the surveyed *Posidonia oceanica* sites have suffered net density losses of over 20% in 10 years. As for fish, non-indigenous species and climate change cause local extinction.

Projected temperature increases combined with a decrease in nutrient replenishment and ocean acidification, are expected to cause changes in plankton communities, negative impacts on fish, corals, seagrass meadows and propagation of non-indigenous species. Projected sea level rise will impact coastal wetlands deltas and lagoons. Extensive urbanization added to climate change is also expected to threaten coastal ecosystems, human health and well-being.

A nexus approach is required when trying to establish adaptation methods for the entire Mediterranean, while taking into account ecosystem-based management, synergies and conflicts, integrating local knowledge and institutions. Suitable adaptation policies include reducing pollution runoff, both from agriculture and industry and waste management, and policies to limit or prevent acidification. Conservation planning and management should focus on cross-cutting approaches and building resilience between structural and functional connectivities of various fields.

Terrestrial ecosystems

> Biodiversity changes in the Mediterranean over the past 40 years have occurred more quickly and been more significant than in other regions of the world. Urbanization and the loss of grasslands are key factors of ecosystem degradation across the region. Since 1990, agricultural abandonment has led to a general increase in forest areas in the northern Mediterranean, while in the southern Mediterranean, ecosystems are still at risk of fragmentation or disappearance due to human pressure from clearing and cultivation, overexploitation of firewood and overgrazing. Drylands have significant biodiversity value, with many of the plants and animals highly adapted to water-limited conditions. They are undergoing an overall increase in response to climate change and extensive land abandonment. 48% of Mediterranean wetlands were lost between 1970 and 2013, with 36% of wetland-dependent animals in the Mediterranean threatened with extinction. Because of the reduction in river flows, 40% of fish species in Mediterranean rivers are endangered.
>
> Projections for the 21st century indicate drier climate and increased human pressure, with negative impacts on terrestrial biodiversity, forest productivity, burned areas, freshwater ecosystems and agrosystems. Future projections indicate that burnt areas can increase across the region by up to 40% in a 1.5°C warming scenario and up to 100% from current levels for 3°C warming at the end of the century. Mediterranean drylands will become drier and their extent is expected to increase across the region. Projections suggest decreased hydrological connectivity, increased concentration of pollutants during droughts, changes in biological communities as a result of harsher environmental conditions, and a decrease in biological processes such as nutrient uptake, primary production, and decomposition.
>
> Promotion of ‘climate-wise connectivity’ through permeability of the landscape matrix, dispersal corridors and habitat networks are key to facilitating upward the migration of lowland species to mountains in order to adapt to new climate change conditions. Promotion of mixed-species forest stands and sylvicultural practices such as thinning, and management of understory can promote the adaption of Mediterranean forests to climate change. Promotion of the spatial heterogeneity of the landscape matrix can help reduce fire impacts. The preservation of the natural flow variability of Mediterranean rivers and streams and wide riparian areas, along with reductions in water demand are key to the adaptation of freshwater ecosystems to future climate change.

4.1 Marine ecosystems

4.1.1 Current condition and past trends

4.1.1.1 Observed changes

Despite only covering 0.82% of the ocean surface, the Mediterranean Sea supports a high level of biodiversity, including about 18% of all known marine species (~ 17,000) (Bianchi and Morri 2000; UNEP/MAP-RAC/SPA 2009; Coll et al. 2010). The Mediterranean Sea is biologically diverse because it is a warm sea at temperate latitudes, and is thus home to both temperate and subtropical species, and has been further diversified by its complex geological history (Bianchi and Morri 2000; Merheb et al. 2016). As a result, the present marine biota of the Mediterranean is composed of species belonging to: (1) temperate Atlantic-Mediterranean species; (2) cosmopolitan species; (3) endemic elements, comprising both paleoendemic (Tethyan origin) and neoendemic species (Pliocenic origin); (4) subtropical Atlantic species (interglacial remnants); (5) boreal Atlantic species (ice-age remnants); (6) Red Sea migrants (especially into the Levantine basin); (7) eastern Atlantic migrants (especially into the Alboran Sea) (Bianchi and Morri 2000).

In marine ecosystems, specific drivers of environmental change include: i) the increasing temperature and salinity of surface waters (Coma et al. 2009; Conversi et al. 2010; Calvo et al. 2011) and the deep-sea (≥400 m) (Béthoux et al. 1990; Rixen et al. 2005; Vargas-Yáñez et al. 2010; Skliris et al. 2014; Schroeder et al. 2016), ii) enhanced thermal stratification (Powley et al. 2016), which can increase eutrophication and O_2 consumption due to increasing dissolved organic carbon (DOC) concentrations in the mixed layer (Ferreira et al. 2011; Santinelli et al. 2013; Ngatia et al. 2019), and iii) decreasing ocean pH fundamentally changing ocean carbonate chemistry (Calvo et al. 2011; The MerMex Group et al. 2011; Flecha et al. 2015; Hassoun et al. 2015, 2019; Merlivat et al. 2018). Detailed information about these drivers, namely temperature and salinity changes, Mediterranean hydrology and ocean acidification can be found in Sections 2.2.4, 2.2.7.2 and 2.2.9. Risks and vulnerabilities caused by these drivers are also affected by non-climate related anthropogenic stressors, such as industrialization, urbanization and agriculture, fishing, maritime traffic, harbor activities, tourism (Macías et al. 2014; Thiébault et al. 2016) and floating plastics and other polymers (Fossi et al. 2012, 2018; Suaria et al. 2016). These non-climate drivers are thoroughly described in Chapters 2 and 3.1 (Section 3.1.2.3) and can be classified as pollution (Section 2.3) and land and sea-use changes (Section 2.4).

The interconnected effects of climate change and several non-climate related drivers, covered in Chapter 2, Section 2.6 affect the way the Mediterranean marine ecosystem functions at all levels, from primary producers to upper trophic-levels (The MerMex Group et al. 2011; Doney et al. 2012; IPCC 2014) (Figure 4.1). Consequences include enhanced mortality of key marine habitat species, e.g., coralligenous outcrops, maërl beds (Pairaud et al. 2014; Molina et al. 2016) and the bivalve *Pinna nobilis* (Vázquez-Luis et al. 2017), as well as the increased establishment of new communities and disease outbreaks (Rubio-Portillo et al. 2018; Berzak et al. 2019). Impacts of warming on marine biota not only result from the direct impact of increasing temperature on organism physiology, but also from the effect of warming on other biological (e.g., microbial activity, metabolic rates) and abiotic (e.g., oxygen solubility) components of ecosystem functions (Vaquer-Sunyer and Duarte 2013).

Since the mid-1980s, regime shifts in the Mediterranean Sea have impacted different ecosystem components (e.g., diversity and abundance of zooplankton, abundance of anchovy stocks, frequency of harmful algal blooms, mucilage outbreaks), possibly due to regional effects of climate modes (Section 2.2.2), such as a positive state of the North Atlantic Oscillation (NAO) that affects the physical properties of the water column (Conversi et al. 2010; Barausse et al. 2011). The recent study by Fortibuoni et al. (2017), while confirming the existence of some regime shifts, does not support the hypothesis of climatic change as a main driver for these, and rather points to the impact of local pressures, i.e. overexploitation and nutrient loads.

Increasing temperatures are driving the northward spread of warm-water species (Sabatés et al. 2006; Tsikliras 2008; Bianchi et al. 2018), and have contributed to the spread of the non-indigenous Atlantic coral *Oculina patagonia* (Serrano et al. 2013). The recent spread of warm-water species that have entered from Red Sea and Atlantic Ocean into cooler northern areas is leading to the “tropicalization” of Mediterranean fauna (Vergés et al. 2014; Bianchi et al. 2018; Galil et al. 2018). Non-indigenous species are extensively detailed as a driver in Section 2.5. Species that need certain temperature ranges cannot migrate further, as the different areas in which they usually live and span are becoming more and more restricted, e.g., the anchovy *Engraulis engrasicolus* (Sabatés et al. 2006). Warming water may also have strong effects on deep Mediterranean areas of the two zones were cold water is formed, as increasing temperature may slow the potential downwelling and the provision of oxygen both in the Gulf of Lions and in the Adriatic Sea, leaving the cold-water coral communities exposed to a certain degree of hypoxia (Taviani et al. 2016).

In addition to the general warming patterns, periods of extreme temperatures have had large-scale and negative consequences for Mediterranean marine ecosystems (Sections 2.2.1 and 2.2.2). A link between positive thermal anomalies and observed invertebrate mass mortalities has been observed in the Mediterranean Sea (Rivetti et al. 2014). Also, unprecedented mass mortality events, which affected at least 25 prominent sessile metazoans, occurred during the summers of 1999, 2003, and 2006 across hundreds of kilometers of coastline in the northwest Mediterranean Sea (Cerrano et al. 2000; Calvo et al. 2011). These events coincided with either short periods (2 to 5 days: 2003, 2006) of high sea temperatures (27°C) or longer periods (30 to 40 days) of less extreme temperatures (24°C: 1999) (Crisci et al. 2011). Impacts of these events on marine organisms have particularly been reported between 0 and 35 m depths, such as gorgonian coral mortality (Coma et al. 2009) or shoot mortality and flowering of seagrasses (Díaz-Almela et al. 2007; Marba and Duarte 2010). A collaborative database for tracking mass mortality events in the Mediterranean Sea has been recently launched to support the analysis of relationships between thermal conditions and/or other environmental drivers (Garrabou et al. 2019), and can be helpful for better detecting changes across the Mediterranean Basin.

In addition, ocean acidification is an emerging human health issue, that also threatens the marine realm (Falkenberg et al. 2020) (Section 2.2.9). Studies of the consequences of ocean acidification on marine Mediterranean ecosystems report diverse responses (Martin and Gattuso 2009; Rodolfo-Metalpa et al. 2010; Movilla et al. 2012; Bramanti et al. 2013; Gazeau et al. 2014; Lacoue-Labarthe et al. 2016). Insights have been gained by studying natural CO_2 seeps at Mediterranean sites such as Ischia and Vulcano in Italy, where biodiversity decreases with decreasing pH toward the vents, with a notable decline in calcifiers (Hall-Spencer et al. 2008; Prada et al. 2017). Transplants of corals, mollusks, and bryozoans along the acidification gradients around seeps reveal a low level of vulnerability to CO_2 levels expected over the next 100 years (Rodolfo-Metalpa et al. 2010, 2011). However, periods of high temperature increase vulnerability to ocean acidification, thereby increasing the long-term risk posed to Mediterranean organisms and ecosystems as temperatures rise (Gazeau et al. 2014; Lacoue-Labarthe et al. 2016). Ocean acidification seems to have a slower but unstoppable effect on several organisms, the increase of temperature being a more immediate stress factor in most species (Lejeusne et al. 2010). A recent overview (Gao et al. 2020) showed that the combination of ocean acidification and warming may affect food webs from different directions; ocean acidification is more likely to follow bottom-up controls (resource driven), while temperature drives top–down controls (consumer driven).

Key habitats undergoing change

Rapid warming of the Mediterranean Sea, in synergy with other climate and non-climate related drivers (see Chapter 2), threatens marine biodiversity, and particularly some key ecosystems that have high vulnerability to such pressures, as presented below.

Coralligenous

The coralligenous is a typical Mediterranean underwater seascape, present on hard bottoms from ~15 to 120 m depths and is mainly produced by the accumulation of calcareous encrusting algae (*Lithophyllum, Lithothamnion, Mesophyllum* and *Peyssonnelia*) growing in dim light conditions and relatively calm waters (Ballesteros 2006; Boudouresque et al. 2015). These outcrops foster one of the richest assemblages found in the Mediterranean, harboring approximately 10% of Mediterranean marine species (Ros et al. 1985; Boudouresque 2004; Ballesteros 2006; Casas-Güell et al. 2016), most of which are long-lived algae and sessile invertebrates (sponges, corals, bryozoans and tunicates) (Garrabou et al. 2002; Ballesteros 2006). The different habitats that make up these biogenic formations are mainly determined by light exposure, so that some coralligenous habitats can be dominated by calcareous algae and others completely dominated by macroinvertebrates with almost no algae (Gili et al. 2014; Casas-Güell et al. 2016). Red coral, *Corallium rubrum*, is one of the habitat-forming species that plays a key role in the functioning of coralligenous habitats because of its trophic activity, biomass and perennial biogenic structure, like other Mediterranean gorgonian species (Gili et al. 2014; Ponti et al. 2014b, 2016, 2018). Red coral is a slow-growing, long-lived species that grows in dim light habitats (e.g., caves, vertical cliffs and overhangs) between 10 and 200 m depths. Despite its essential ecosystemic role, little is known about the geographical distribution of red coral up to 400 km offshore the coastline due to its large bathymetric range and afferent constraints (Casas-Güell et al. 2015, 2016), and the major studies focus on the phytobenthic component (Piazzi et al. 2009, 2012; Boudouresque et al. 2015). Studies at an intermediate scale (tens of km) have been conducted with key species, pinpointing the fact that their distribution may be very heterogeneous depending on the environmental factors (Gori et al. 2012; Coppari et al. 2014, 2016). Due to this lack of baseline data, the structure of coralligenous outcrops is still poorly understood, preventing a proper assessment of its current state of biodiversity and the potential impacts of harvesting, and other disturbances related to global change, on red coral assemblages. A recent study (Mallo et al. 2019) based on historical red coral data from the north western Mediterranean Sea, documented the halt in the *C. rubrum* decrease and the first recovery response due to effective protection measures in some areas.

Coralligenous outcrops are affected by several consequences of global change such as nutrient enrichment, non-indigenous species, increased sedimentation, mechanical impacts, mainly from fishing activities, e.g., mechanical injuries and sediment re-suspension (Cebrián et al. 2012; Piazzi et al. 2012; Gatti et al. 2015), as well as sea warming (e.g., massive mortalities related to temperature anomalies) and the potential effects of ocean acidification (Bramanti et al. 2013; Cerrano et al. 2013; Gili et al. 2014). Recently, potential synergies between these stressors have been hypothesized (Section 2.6), especially in shallow areas were heat waves may have a large impact on several organisms (Galli et al. 2017), resulting in a fragmentation of the habitat that can open new space for non-indigenous species (Vezzulli et al. 2013). It has also been demonstrated that a decrease in the abundance of coralligenous habitat-forming species leads to a rapid fragmentation in community structure and a loss of species benefiting from the structural complexity these species provide (Ponti et al. 2014b; di Camillo and Cerrano 2015; Valls et al. 2015).

In addition to marine heat waves (Garrabou et al. 2001, 2009), one of the main past threats for the red coral *Corallium rubrum* has been intensive harvesting (see section 2.4), which has caused an overall shift in population structure, resulting in a decrease in both biomass and colony size (Tsounis et al. 2010; Bramanti et al. 2014; Montero-Serra et al. 2015). Moreover, its Mg-calcite skeleton makes it vulnerable to ocean acidification (Bramanti et al. 2015). Bramanti et al. (2013) experimentally evaluated the effects of low pH on *C. rubrum* over a 314-day period under two pH levels (8.10 and 7.81). This study concludes that exposure to lower pH conditions negatively affected skeletal growth and spicule morphology (i.e., abnormal shapes).

Mediterranean gorgonian “forests” (e.g., *Paramuricea clavata*, *Eunicella cavolinii*) are threatened by several human activities and are affected by climatic anomalies that have led to mass mortality events in recent decades (Ponti et al. 2014b, 2018; Verdura et al. 2019). Observed mortality events have been linked to temperature-dependent bacterial pathogens (Bally and Garrabou 2007). Also, diverse responses to thermal stress have been shown in gorgonians (Pivotto et al. 2015; Crisci et al. 2017). This may condition the future response of these species to climate change.

The ecological role of these habitats and the possible consequence of their loss are still poorly understood. The experimental study of Ponti et al. (2014b) reports a significant effect of gorgonians (*E. cavolinii*, and *P. clavata*) on the recruitment of epibenthic organisms and their presence mainly limits the growth of erect algae and enhances the abundance of encrusting algae and sessile invertebrates. This effect could be due to microscale modification of hydrodynamics and sediment deposition rate by i) a shading effect that reduces light intensity, ii) intercepting settling propagules, iii) competing for food with filter-feeders and/or iv) competing for space by producing allelochemicals. Although the biological interaction between gorgonians and other species deserves further study, changes to the edaphic conditions caused by gorgonian forests influences the larval settlement and recruitment processes of benthic assemblages (Ponti et al. 2014b, 2018).

In addition to the long-term effects of global change and its consequences on the Mediterranean coralligenous, short-term extreme events may be even more devastating than heat waves. Teixidó et al. (2013) show how an extreme storm event affected the dynamics of benthic coralligenous outcrops in the northwestern Mediterranean Sea using data acquired before (2006–2008) and after the impact (2009–2010) of a major storm. The most exposed and impacted site experienced a major shift immediately after the storm and over the following year. This impact consists of changes in the species richness and diversity of benthic species such as calcareous algae, sponges, anthozoans, bryozoans and tunicates. In this site, benthic species recorded a 22% to 58% loss of cover on average, with those with fragile forms showing cover losses up to 50 to 100%. Small patches survived after the storm and began to grow slightly during the following year, and the sheltered sites showed no significant changes in all the studied parameters, indicating no variations due to the storm (Teixidó et al. 2013).

Deep sea ecosystems

Although poorly known, deep seafloor ecosystems provide services that are vitally important to the entire ocean and biosphere, and play a particularly major role in climate change mitigation. For instance, by storing a large amount of anthropogenic CO_2 and by absorbing heat accumulated from the greenhouse effect, the deep sea Mediterranean waters and ecosystems capture large quantities of carbon and, as such, slow down the warming of surface waters and land (Luna et al. 2012; Palmiéri et al. 2015) (Sections 2.2.7 and 2.2.9). Rising atmospheric greenhouse gases are affecting water column oxygenation, temperature, pH and food supply (Section 3.2), with similar impacts on deep-sea ecosystems (Sweetman et al. 2017). As they are deprived of light, deep-sea ecosystems are greatly dependent on surface primary production: “marine snow” (Gambi et al. 2014). Surface water also oxygenates deep-sea environments when they sink to form deep and intermediate water masses. When surface water is warmer, it does not mix well with deep water (Section 2.2.7).

In the Mediterranean, the deep sea covers about 79% of the Mediterranean Basin, including habitats potentially able to deliver multiple ecosystem services and numerous resources of high economic value (Manea et al. 2020). Despite this fact, very few studies address the response of deep-sea ecosystems to ongoing climate change in this sea. In contrast with most oceans where the flux of particulate organic matter to the seafloor is likely to decline significantly in response to climate change (Sweetman et al. 2017), a study from the eastern Mediterranean shows that climate change has caused an immediate accumulation of organic matter on the deep-sea floor in recent decades (Danovaro et al. 2001). This led, together with deep-sea warming, to alteration of carbon and nitrogen cycles and has had negative effects on deep-sea bacteria and benthic fauna (Danovaro et al. 2001, 2004). For instance, the observed salinity and temperature changes in eastern Mediterranean deep and bottom waters from 1987 to 1994 (Roether et al. 1996; Theocharis et al. 2002) led to the uplift of these water masses by several hundred meters, reaching shallower depths (100–150 m; i.e. close to the euphotic zone) under the influence of cyclonic circulation. This resulted in increased biological production and therefore enhanced flux of organic carbon to the deep sea, thereby significantly and quickly changing the way deep-sea ecosystems function (Psarra et al. 2000; Danovaro et al. 2001). The review of Yasuhara and Danovaro (2016) on temperature impacts on deep-sea Mediterranean biodiversity shows that minor temperature shifts of around 0.1°C or less are sufficient to cause significant changes in biodiversity and the community structure of deep-sea nematode assemblages.

Planktonic ecosystems

Several studies have addressed the possible impact of climate change on marine phytoplankton diversity and distribution in the Mediterranean Sea, highlighting highly contrasting regional patterns (Duarte et al. 2000; Goffart et al. 2002; Marty et al. 2002; Bosc et al. 2004; Ribera d’Alcalà et al. 2004; Marty and Chiavérini 2010; Herrmann et al. 2014; Oviedo et al. 2015; D’Amario et al. 2017). Some studies from the northwestern Mediterranean have reported a positive trend in phytoplankton biomass in response to the expansion of the summer stratification. This trend was accompanied by an increase in picoplankton and nanoflagellates (i.e. small-sized phytoplankton) and a decline in diatoms, which are responsible for new production (Goffart et al. 2002; Marty et al. 2002; Mena et al. 2019; Ramírez-Romero et al. 2020). However, other studies report that the spring bloom in many Mediterranean regions tends to occur earlier in the year, possibly in relation to earlier water warming and high irradiance, in contrast with the autumn bloom that tends to disappear because of a longer stratification period (Bosc et al. 2004). Bosc et al. (2004) also reveal significant interannual variations in biomass and primary production, not only in the northwestern basin (e.g., the exceptional bloom in spring 1999), but also, and more surprisingly, in the oligotrophic waters of the eastern basin (e.g., the 9%

decrease in primary production from 2000 to 2001). In this latter basin, phytoplankton shifts seem to be concurrent with rising winter precipitation and sea surface temperature (Mena et al. 2019) (Section 2.2.4, 2.2.5 and 2.2.7).

In some Mediterranean settings, such as the central Ligurian Sea, increased deep-water convection (as deep as 2000 m) has been attributed to greater surface salinity causing increased nutrient supply near the surface, and thus more primary production (Marty and Chiavérini 2010). In contrast, in the productive northwestern Mediterranean Sea, deep convection could significantly decrease under the influence of climate change (Herrmann et al. 2014), impacting pelagic planktonic ecosystem, which are strongly influenced by these hydrodynamics. The weakening of deep convection and surface warming modifies the pelagic planktonic ecosystem and associated carbon cycle indirectly only: the spring bloom occurs one month earlier, and the bottom up control of phytoplankton development and bacteria growth by nitrogen and phosphorus availability strengthens, and the microbial loop intensifies as the small-sized plankton biomass increases (Herrmann et al. 2014). Net carbon fixation and deep export do not change significantly. In the Tyrrhenian Sea, Ribera d'Alcalà et al. (2004) explain the significant changes in the long-term patterns of rare copepod species as a symptom of large-scale meteorological phenomena of the North Atlantic sector.

In the NW Mediterranean Sea, decadal climatic oscillations linked to the NAO forcing of the precipitation regime led to an increase in the upper salinity in the 1980s and in the late 1990s and early 2000s (Chapter 2, Section 2.2.7). In saline years, the annual abundance of zooplankton is higher than otherwise (Fernández de Puelles and Molinero 2007). According to Molinero et al. (2008), large-scale climate forcing has altered the local environment and the pelagic food-web dynamics in the NW Mediterranean Sea through changes in biological interactions, competition and predation. The authors also suggest that warming, the dominance of small phytoplankton and predation pressure by jellyfish negatively affected copepod populations (recruitment, life-history traits and physiological thresholds) in the early 1990s, whereas chaetognaths were surpassed by jellyplankton as the most frequent copepod prey. A more recent study from the same Ligurian time-series updated with ten more years (up to 2003) revealed that the zooplankton, mainly copepods, recovered their initial concentrations after 2000, suggesting a quasi-decadal cycle (Coma et al. 2009). This illustrates the difficulty in identifying long-term changes from decadal oscillation in short time-series in plankton. However, surface salinity appears to be a common physical indicator of changes in the pelagic ecosystem of the NW Mediterranean Sea for jellyfish (Buecher et al. 1997), crustaceans (García-Comas et al. 2011) and phytoplankton (Marty and Chiavérini 2010).

Gallisai et al. (2014) report that aerosol deposition from the Sahara may explain 1 to 10% of seasonally detrended chlorophyll variability in the nutrient-low Mediterranean with main effects in spring over the eastern and central Mediterranean, corresponding to dust events fueling needed nutrients for the planktonic community (Ternon et al. 2011). The areas showing negative effects on chlorophyll from dust deposition are regions under significant influence from European aerosols. Anthropogenic aerosol deposition of nitrate and phosphate largely influence primary production in the northern Mediterranean Sea (Richon et al. 2018a, 2018b) (Section 2.2.3). This response of chlorophyll dynamics to dust deposition is important when knowing that future scenarios predict increased aridity and shallowing of the mixed layer (Gallisai et al. 2014) (Section 2.3.2).

From around the island of Lampedusa (central Mediterranean), the multi-year evolution of biogenic dimethylsulfide (DMS) production in the marine surface layer and the resulting methanesulfonate on the atmosphere are mainly attributed to phytoplankton physiology (Becagli et al. 2013). High phytoplankton productivity can also be the expression of stressed cells, especially during summer when high irradiance and the shallow depth of the upper mixed layer prevails. This therefore leads to higher methanesulfonate concentrations in the atmosphere. These dynamics can be further controlled by the North Atlantic Oscillation, and related oceanic and atmospheric processes (Becagli et al. 2013).

Large vertebrates

One of the biggest threats to large marine vertebrates is litter debris, such as fishing gear or other large items (Galgani et al. 2014) (Figure 4.1). Regularly, in the Mediterranean Sea and worldwide, large vertebrates such as sea birds (van Franeker et al. 2011), cetaceans (de Stephanis et al. 2013; Notarbartolo di Sciara 2014) and marine turtles (Lazar and Gračan 2011; Campani et al. 2013; Camedda et al. 2014) accidentally swallow micro and macro-plastic debris that is often found in their digestive tracts. The plastic debris (Section 2.3.2.3) affects the marine biota of the Mediterranean at macro, micro- and nano-levels.

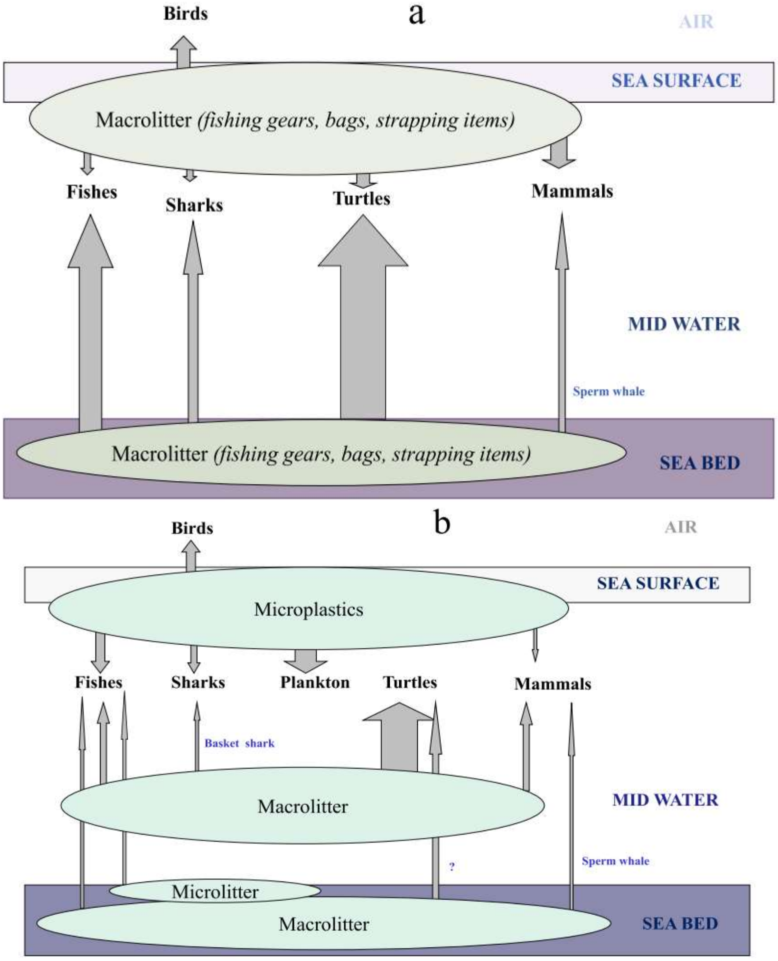

Figure 4.1 | Summary of interactions between large marine vertebrates and marine litter (Galgani et al. 2014). Fluxes of litter in the life cycle and intensity of its effects on large marine vertebrates, (a: entanglement; b: ingestion), depending on various factors such as ingestion mechanisms (predation, active or passive filter feeding), development stage (benthic or pelagic phases for sea turtles), behavior and foraging strategy (feeding on the sea floor, in the water column or on the surface, selectivity according to color, shape etc., ecological plasticity in diet and habitat), types of litter (micro/macro litter) and types of fishing gear (nets, hooks and lines). The thicker arrows indicate key processes. Although trophic transfer from one level to another has been demonstrated in vitro for microplastics in plankton, it remains controversial in situ, as most ingested litter is excreted in feces.

Sperm whales (*Physeter macrocephalus*) in the Mediterranean Sea, which are believed to be fewer than 2500 mature individuals, are endangered world-wide (Notarbartolo di Sciara 2014). A decline in sperm whales in the Mediterranean has been observed over the last half-century. In addition to ingestion of solid debris, other anthropogenic activities at sea are suspected to have caused the decline of this species and continue to threaten its survival in various ways: bycatch, collisions with vessels, debilitation by chemical pollution, anthropogenic noise, disturbance from irresponsible whale watching and most likely climate change, and prey depletion (Notarbartolo di Sciara 2014). Regarding specifically ingested debris, ingestion rates are as high as 31% in some marine mammal populations, and sublethal effects could result in impacts at the population level (Baulch and Perry 2014). Campani et al. (2013) and Camedda et al. (2014) investigated the interaction between loggerhead sea turtles (*Caretta caretta*) and marine litter in the northern Tyrrhenian Sea and around Sardinia, respectively. In thirty-one *C. caretta* individuals found stranded or accidentally bycaught in northern Tyrrhenian Sea, marine debris, mainly plastics, were present in 71% of specimens (Campani et al. 2013). In Sardinia, only 14% of the 121 monitored turtles had debris in their digestive tracts but plastic was the main physical category (Fossi et al. 2013; Camedda et al. 2014).

Sharks and rays are also seriously threatened by anthropogenic pressures, mainly as a result of overfishing (Dulvy et al. 2014) (Figure 4.2), as described in Section 2.4.2 in the context of the increasing sea use changes. Some sharks live in narrow climatic ranges (Chin et al. 2010), putting them at risk in a climate change hotspot such as the Mediterranean (Ben Rais Lasram et al. 2010). Microplastic (<5 mm) ingestion has been recorded in 16.8% of the analyzed specimens of the blackmouth catshark *Galeus melastomus* around the Balearic Islands, with higher quantities of filament-type microplastics (Alomar and Deudero 2017). In three striped dolphin populations living in the Pelagos Sanctuary (bordered by western Italy, southern France and northern Sardinia), the highest toxicological stress was from PBT (persistent, bioaccumulative and toxic substances) chemical levels, combined with correlated biomarker responses (Fossi et al. 2013). More on chemical pollution is covered in Section 2.3.

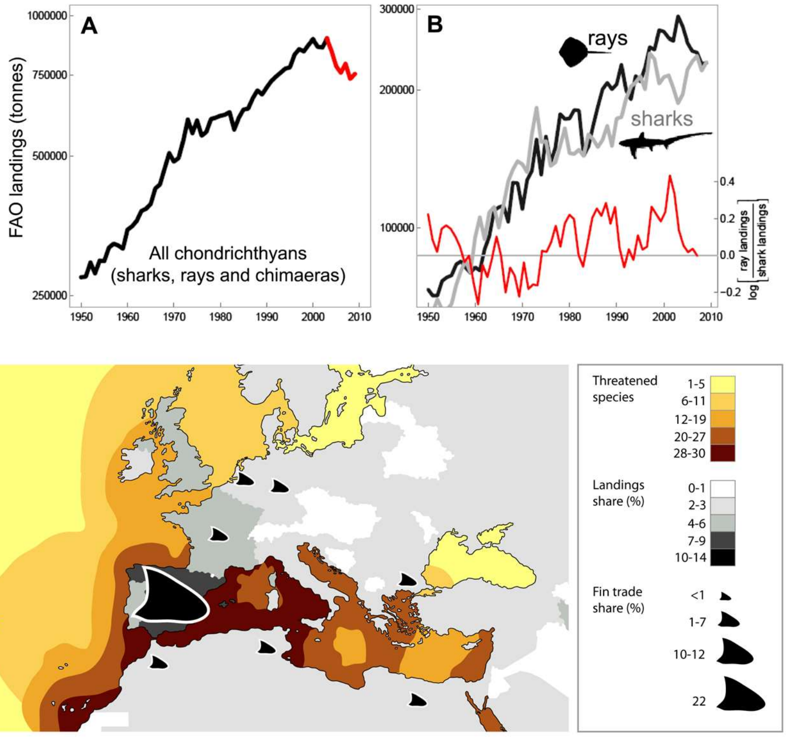

Figure 4.2 | The trajectory and spatial pattern of chondrichthyan (cartilaginous fishes that include sharks, skates, rays and chimaeras) fisheries catch landings and fin exports. (A) The landed catch of chondrichthyans reported to the United Nations Food and Agriculture Organization from 1950 to 2009 up to the peak in 2003 (black) and subsequent decline (red). (B) The rising contribution of rays to the taxonomically-differentiated global reported landed catch: shark landings (light gray), ray landings (black), log ratio [rays/sharks], (red). Log ratios >0 occur when more rays are landed than sharks. The peak catch of taxonomically-differentiated rays peaks at 289,353 tonnes in 2003. (C) The main shark and ray fishing nations are gray-shaded according to their percentage share of the total average annual chondrichthyan landings reported to the FAO from 1999 to 2009. The relative share of shark and ray fin trade exports to Hong Kong in 2010 are represented by fin size. The taxonomically-differentiated proportion excludes the 'nei' (not

elsewhere included) and generic ‘sharks, rays, and chimaeras’ category (adapted from Dulvy et al. 2014).

Changes in biodiversity

To date, changes in Mediterranean marine biodiversity are essentially driven by human activities (Mannino et al. 2017), i.e. pollution (Section 2.3), sea use changes (Section 2.4.2), the introduction of non-indigenous species (Section 2.5), together with climate change (Section 2.2) (Lejeusne et al. 2010; Zenetos et al. 2012; Katsanevakis et al. 2013, 2014b). In general, the Mediterranean Sea represents the highest proportion of threatened marine habitats in Europe (32%, 15 habitats) with 21% being listed as vulnerable and 11% as endangered (see review in Mannino et al. 2017). This threat includes several valuable and unique habitats (e.g., seagrasses and coralligenous), supporting an extensive repository of biodiversity (Gubbay et al. 2016).

The shallow depth (on average 1450 m) of the Mediterranean Sea and the relatively fast deep-water turnover in comparison to the open ocean, coupled with a high degree of endemism (about 20% of Mediterranean marine species; Coll et al. (2010)) point to a potential amplification of climate change impacts. These are expected to cause earlier changes in biodiversity in comparison with other seas, thus making this system a model for investigating biodiversity response to direct and indirect effects of temperature changes and other climate-related and non-related drivers (Chapter 2).

Species with low dispersal ability are particularly affected by climate change, which may also lead to local extinctions, greatly contributing to biodiversity loss (Mannino et al. 2017). Any change in biodiversity may affect ecosystem functioning, even in the case of the establishment of a single species and may lead to important consequences both for nature as well as for society. However, the extreme richness of microclimates in the Mediterranean (ranging from climate conditions similar to those of the Northern Sea in the Adriatic to an almost tropical condition in the eastern basin) makes prediction at large spatial scales difficult. Most effects of climate change (or climate anomalies) on marine biodiversity have been so far identified at regional scales (Philippart et al. 2011).

During recent decades, Mediterranean marine communities have shown significant changes in taxa composition and distribution. In the western Mediterranean, climate change is influencing the boundaries of biogeographic regions and thus warm water marine species are extending their ranges and colonizing new regions where they were previously absent (Katsanevakis et al. 2014a). For instance, mucilages have appeared more frequently (associated with a malfunctioning of the microbial loop) in the Adriatic Sea, where it was documented for the first time, and in several regions beyond, in recent decades, concomitantly with a significant increase in sea surface temperature (Danovaro et al. 2009). Mucilage is not closely associated with the presence of eutrophic conditions, as several mucilage outbreaks have been recently observed in oligotrophic seas, such as the Aegean Sea (Danovaro et al. 2009). The Ligurian Sea, one of the coldest areas of the Mediterranean Sea, displays a low number of subtropical species and a higher abundance of cold-temperate water species. However, the recent warming of Ligurian seawater has favored the penetration of warm-water species (e.g., *Thalassoma pavo*), which from 1985 onward, established large and stable populations (Parravicini et al. 2015).

Temperature anomalies, even of short duration, can dramatically change Mediterranean faunal diversity. The largest mass-mortality event recorded in the Mediterranean Sea so far occurred in 1999 along the French and Italian coasts (Cerrano et al. 2000; Perez et al. 2000; Garrabou et al. 2001). That year was characterized by a summer with a positive thermal anomaly that extended the thermocline down to a depth of 40 m (Romano et al. 2000) and resulted in the extensive mortality of 28 epibenthic invertebrate species (Figure 4.3) (Perez et al. 2000; Rivetti et al. 2014). Among benthic organisms, sponges and gorgonians were most severely affected (Cerrano et al. 2000; Perez et al. 2000; Romano et al. 2000; Garrabou et al. 2001; Rivetti et al. 2014). The shortage of food over several weeks is a common phenomenon in the Mediterranean Sea due to summer water stratification, but very long periods with high temperatures may explain such mass mortalities (Rossi et al. 2017a).

In the eastern Mediterranean, the rise of seawater temperatures may also be partly responsible for the entrance of non-indigenous species (Section 2.5), mostly from the tropical Indo-Pacific (Galil 2000; Por 2009; Zenetos et al. 2012; Rilov 2016). The increased introduction and spread of non-indigenous species may be a supplementary stress factor for native species already weakened by climate variations resulting in the dislocation of indigenous species' niches and possibly cascade effects on the food webs (Rilov 2016; Corrales et al. 2018). Non-indigenous species are a recognized threat to diversity and the abundance of native species as well as a threat to the ecological stability of the infested ecosystems. Despite the overall tendency towards ocean warming, the eastern Mediterranean also experiences occasional climate anomalies, for example between 1992 and 1994, when temperatures dropped by about 0.4°C (Danovaro et al. 2001). This caused a drastic decrease in nematode abundance and overall faunal diversity (e.g., a roughly 50% decrease in nematode diversity, Danovaro et al. 2004). After 1994, when the temperature gradually recovered, biodiversity started to reverse to previous conditions but had not recovered fully in 1998 (Danovaro et al. 2004).

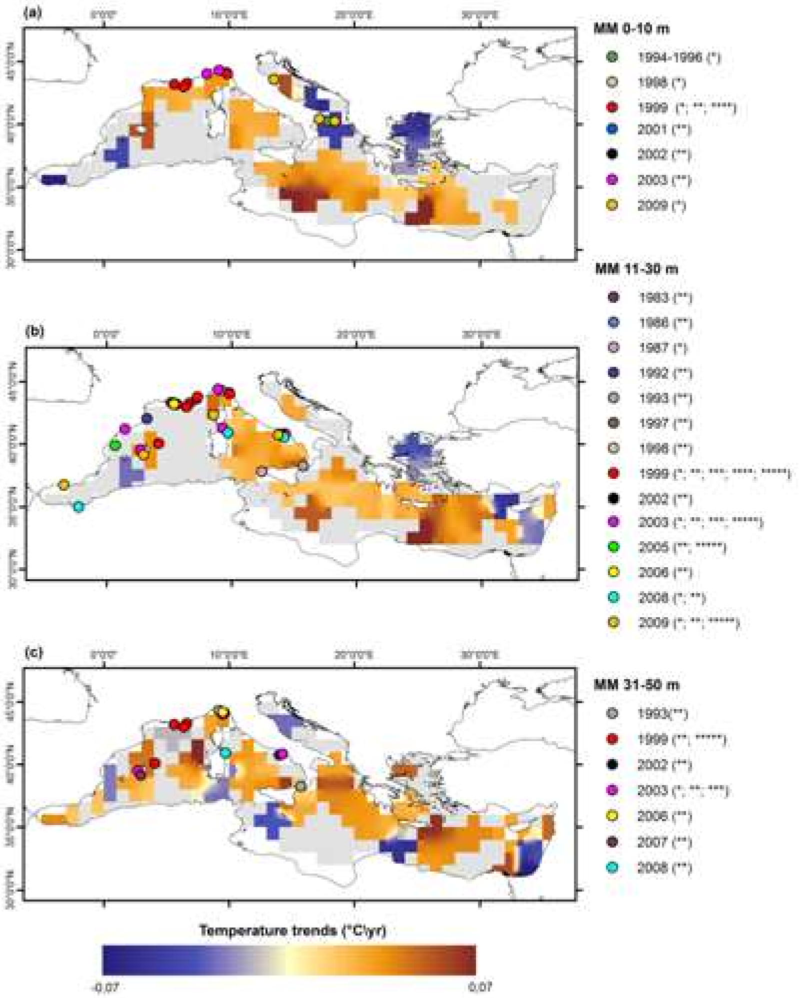

Figure 4.3 | Temperature trends across the Mediterranean Basin. Temperature trends at 0–10 m (a), 11–30 m (b), 31–50 m (c) depth layers for the period 1945–2011 in July-November. Linear regressions have been calculated on grids of 1° latitude by 1° longitude and tested for statistical significance at the 90% level. Significant increased/decreased temperature trends are reported as colored cells, non-significant increased/decreased temperature trends are reported as grey areas. Dots show the locations of documented mass mortalities for a depth layer, each color represents a single event. The asterisks in the legend of mass mortalities (MM) events refer to the taxa affected: * stands for sponges, ** for cnidarians, *** for bryozoans, **** for ascidians, ***** for bivalves (Rivetti et al. 2014).

Sea warming may also have effects on the virulence of pathogens, favoring the frequency of epidemiological events, as most pathogens are temperature sensitive (Bally and Garrabou 2007; Vezzulli et al. 2013). Mass mortalities of the gorgonian *Paramunicea clavata*, scleractinian corals, zoanthids, and sponges observed in 1999 in the Ligurian Sea were indeed promoted by a temperature shift, in conjunction with the growth of opportunistic pathogens (including some fungi) (Cerrano et al. 2000).

Increased surface temperatures and altered circulation and precipitation regimes have been evoked to explain the increased frequency of bottom water hypoxia or anoxia in coastal areas of the northern Adriatic. These phenomena, often associated with mass mortalities of fish and benthic fauna, alter food webs and might have important cascade effects on biodiversity (Coll et al. 2010). The Adriatic Sea can undergo dramatic change in the lower part of its temperature ranges. In winter 2001, the Adriatic Sea experienced a period of abnormally low surface temperatures (from 9°C to freezing) that led to mass mortalities of sardines (*Sardinella aurita*) (Guidetti et al. 2002), with resulting alteration of the food webs. The Adriatic basin is also the site for deep-water formation, as a result of the bora winds associated with decreased temperatures, but recent studies have reported the shift of this water formation site towards the Aegean Sea by a phenomenon known as eastern-Mediterranean Transient (EMT), related mainly to climatic sea and atmosphere conditions (Hassoun et al. 2015). EMTs change the salinity distribution with surface water freshening linked to enhanced deep-water production and in turn to strengthened Mediterranean thermohaline circulation (Incarbona et al. 2016). This phenomenon can thus affect the marine biodiversity not only in the Adriatic and Ionian Seas but much further, as documented by Ouba et al. (2016), who have correlated the salinity variations and increase in total zooplankton abundance in Lebanese waters to the activation of the Aegean Sea as a major source of dense water formation as part of an “eastern Mediterranean Transient-like” event (see Section 2.2.7 for more details about Mediterranean circulation changes).

In response to ocean acidification, calcifying organisms (planktonic and benthic) such as corals, foraminifera, coccolithophores and coralline red algae, important contributors of marine calcium carbonate production, may be greatly affected (Langer et al. 2009; Moy et al. 2009; Bramanti et al. 2013; Cerrano et al. 2013; Kroeker et al. 2013). Based on experiments, the impact of ocean acidification on Mediterranean corals was examined and a significant decrease in calcification rates in most tested species was reported (Movilla et al. 2012, 2014). In the latter study, there was a heterogeneous effect of low pH on the skeletal growth rate of the organisms depending on their initial weight, suggesting that those specimens with high calcification rates may be the most susceptible to the negative effects of acidification. Also, a significant effect on benthic foraminiferal communities of low-pH seawaters around the island of Ischia (Italy) has been demonstrated as a result of volcanic gas vents with significant changes in distribution, diversity and nature of the fauna (Dias et al. 2010).

Coccolithophores, which are the primary calcifying phytoplankton group, and especially the most abundant species, *Emiliania huxleyi*, have shown a reduction of calcification at increased CO_2 concentrations for the majority of strains tested in culture experiments (Meyer and Riebesell 2015). Meier et al. (2014) analyzed in situ *E. huxleyi* coccolith weight from the NW Mediterranean Sea in a 12-year sediment trap series, and surface sediment and sediment core samples. Their findings clearly show a continuous decrease in the average coccolith weight of *E. huxleyi* from 1993 to 2005, reaching levels below pre-industrial (Holocene) and industrial (20th century) values recorded in the sedimentary record, as most likely a result of the changes in the surface ocean carbonate system. Also, a drastic decrease in production, species diversity and anomalous calcification in coccolithophores has been shown along a natural pH gradient caused by marine CO_2 seeps off Vulcano Island (Italy) (Ziveri et al. 2014).

To conclude, (1) Mediterranean fauna is highly vulnerable to human activities and climate change; (2) both structural and functional biodiversity of continental margins are significantly affected by very small temperature changes; and (3) the impact of human activities and climate change on marine biodiversity might be non-reversible. Since there are close interactions between deep and shallow systems, the vulnerability of deep-sea ecosystems to climate change might also have important implications on the biodiversity and functioning of continental shelves.

The extent of changes caused by climate and non-climate drivers, the responses of Mediterranean marine biota to these changes and their local-regional consequences are yet to be investigated, as slow but significant transformations that may modify the neritic, pelagic, and benthic zones are still ongoing.

4.1.1.2 Past changes

Understanding the degree to which changes in Mediterranean marine ecosystems point to a directional trend driven by global warming remains a challenge for marine ecology (Bertolino et al. 2017a). Reconstructing the temporal variability of Mediterranean marine ecosystems on time scales longer than a few centuries beyond the instrumental records, crossing relevant climate variations and historical periods, can be critical for interpreting these changes.

Climate forcings of Mediterranean marine ecosystems over the past thousand years have occurred on different time scales (Abrantes et al. 2005; Hennekam et al. 2014; Xoplaki et al. 2018). During the Holocene, rapid warming and cooling events have occurred which can, to some degree, provide analogues for the projected changes for the coming centuries (Blois et al. 2013; Benito-Garzón et al. 2014; Raji et al. 2015). In the Mediterranean, these past climate changes impacted the marine physico-chemical parameters of surface and deep waters (e.g., salinity, temperature, oxygenation, pH) which in turn affected marine ecosystems (Frigola et al. 2008; Schmiedl et al. 2010a; Mojtahid et al. 2015; Bertolino et al. 2017b).

Response of marine ecosystems to past temperature changes

In the Mediterranean region, the most abundant Holocene temperature proxy data, especially for the Common Era (the last 2000 years) are alkenone-derived records (Abrantes et al. 2012; Jalali et al. 2016; Sicre et al. 2016). These studies document natural long-term trends superimposed on a multidecadal variability in response to external (e.g., solar) and internal forcings (e.g., NAO) which might explain some recently observed sea surface temperature trends (Versteegh et al. 2007). These studies also reveal a strong regional component. For example, a high resolution study from the Gulf of Lion shows an overall sea surface temperature cooling trend since the mid-holocene followed by a rapid warming from ~1850 AD onwards that may parallel recent climate change (Jalali et al. 2016). In contrast, south of Sicily and in the eastern Levantine basin, sea surface temperature records show progressive warming since the early Holocene without a clear signature of the recent anthropogenic change (Castañeda et al. 2010; Luterbacher et al. 2012; Jalali et al. 2017). The planktonic ecosystem in the Siculo–Tunisian Strait responded to this progressive warming of the sea surface temperature by increasing the abundance of warm dinocyst species (*Spiniferites mirabilis* and *Impagidinium aculeatum*) and planktonic foraminifera (*Globorotalia inflata* and *Globigerinoides ruber*) (Rouis-Zargouni et al. 2010).

The Holocene was interrupted by at least four brief cooling events at ~9.2 ka, ~8 ka, ~7 ka and ~2.2 ka cal. BP, which may be correlated to climate events recorded elsewhere, including in Greenland ice cores and in Atlantic Ocean sediments. Investigations on cetacean bones from the Grotta dell'Uzzo in northwestern Sicily (Italy) show that the rapid climate change around 8 ka coincided with increased strandings in the Mediterranean Sea (Mannino et al. 2015). Also, the diversity of sponge species living in coralligenous habitats from the Ionian and Ligurian was strongly affected by Holocene warming episodes with a significant loss of their biodiversity in recent decades (Bertolino et al. 2017b, 2019).

In the eastern Mediterranean, multiproxy records derived from sediments from the southeastern Levantine (Schilman et al. 2001b; Mojtahid et al. 2015) and the Adriatic Sea (Piva et al. 2008) reveal complex paleo-oceanographic changes during the late Holocene, with pronounced anomalies during the Medieval Warm Period (MWP) (ca. AD 1150) and the Little Ice Age (ca. AD 1730). These temperature anomalies were accompanied in the eastern Levantine basin by a drastic change in planktonic foraminiferal successions indicating periods oscillating between cold and warm surface waters in opposite phase with the western Mediterranean records (Mojtahid et al. 2015). This east–west contrast in the climate signals has been confirmed by other proxy data (Jalali et al. 2016, 2017).

These findings imply that long-term and short-term climate-driven environmental changes, caused by global warming, will likely impact the entire food chain from planktonic ecosystems to large mammals (e.g., cetaceans) in the near future.

Response of marine ecosystems to past changes in stratification and ventilation

Throughout the Pleistocene, the eastern Mediterranean experienced numerous anoxic events recorded by the cyclical deposition of organic-rich layers or sapropels (Rossignol-Strick et al. 1982; Rohling 1994), the most recent being Sapropel S1 from ~10 to 6 cal ka BP. Maximum insolation due to the Earth's orbital precession minimum significantly intensified the northeast African monsoon, leading to enhanced discharge of fresh and nutrient-rich Nile River water into the eastern Mediterranean (Rossignol-Strick et al. 1982; Emeis et al. 2000). In the Levantine basin, sea surface salinity during S1 dropped by about 2.0 to 4.0 units compared to present values (Kallel et al. 1997; Myers et al. 1998). This led to severe water column stratification and organic enrichment from the Nile river water. In the Ionian Sea, the correspondence of recent sapropel layers with peaks of the lower photic zone coccolithophore species *Florisphaera profunda* indicated the development of a deep chlorophyll maximum, due to the pycnocline/nutricline shallowing in the lower part of the photic zone (Incarbona et al. 2011). In the SE Levantine basin, a severe drop in planktonic foraminiferal diversity was recorded in response to the water column stratification and expressed by the near exclusive presence of the euryhaline tropical-subtropical species *Globigerinoides ruber* and the disappearance of deep-dwelling species (Mojtahid et al. 2015).

The combination of higher organic matter remineralization and decreased ventilation resulted in widespread bottom water anoxia (Rohling 1994; Hennekam et al. 2014). In the Southern Aegean and Levantine Seas, there was a gradual increase in deep-water residence times, preceding S1 formation by approximately 1–1.5 kyr. Once oxygen levels fell below a critical threshold, the benthic ecosystems collapsed almost synchronously with the onset of S1 deposition. The recovery of benthic ecosystems during the terminal phase of S1 formation is controlled by subsequently deeper convection and re-ventilation over a period of approximately 1500 years. After the re-ventilation of the various sub-basins during the middle and late Holocene, deep-water renewal was more or less similar to recent rates (Schmiedl et al. 2010b). Several species of deep-water ostracods that are still common in the western Mediterranean became extinct in the eastern Mediterranean Basin at the onset of early Holocene S1 sapropel deposition and the related anoxia (Van Harten 1987). The deep-water ostracode *Bythocypris obtusata* apparently survived the oxygen crisis in the eastern basin itself. This suggests that full oxygen depletion may not have affected the bottom of all deep sub-basins and supports a midwater oxygen-minimum model for these sub-basins (Van Harten 1987; Schmiedl et al. 2010b).

These paleoclimatic findings suggest that eastern Mediterranean pelagic and benthic marine ecosystems are capable of abrupt transitions in response to gradual forcing. This is crucial for the projection of whether an increase in oceanic moisture availability under current and future warming could trigger a sudden intensification of monsoon rainfall further inland from today's core monsoon region (Schewe and Levermann 2017).

Response of marine ecosystems to past changes in productivity

In the western Mediterranean, productivity has shown an overall decreasing trend since the early Holocene with a marked fall in productivity after the 8.2 ky BP dry-cold event (Ciampo 2004; Jiménez-Espejo et al. 2007; Melki et al. 2009). Superimposed on this long-term pattern, some studies show millennial–centennial time scale variability linked with weakening and strengthening of upwelling conditions that have been simultaneous to changes in Western Mediterranean Deep Water (WMDW) formation in the Gulf of Lions and by extent to the NAO over the past 7.7 ka (Ausín et al. 2015). These changes were accompanied by re-organization in coccolithophore assemblages showing in particular, several high-amplitude oscillations of the productivity indicator species *F. profunda* (Ausín et al. 2015).

In the eastern Mediterranean, several proxy data support overall increased productivity during Sapropel S1 in a high-nutrient stratified environment (Gennari et al. 2009; Castañeda et al. 2010; Mojtahid et al. 2015). This period is characterized by the highest accumulation rates of planktonic foraminifera together with the productivity indicator coccolithophore species *F. profunda* (Incarbona et al. 2011; Mojtahid et al. 2015). After Sapropel 1, a progressive decrease in surface water productivity was recorded and surface and deep-sea ecosystems were driven by short-term changes in food quantity and quality as well as in seasonality, all of which are linked to millennial-scale changes in river runoff and associated nutrient input (Kuhnt et al. 2008; Schmiedl et al. 2010b). Particularly, the last 2.9 ka encompassed a succession of three ecosystem states characterized by nutrient-limiting surface waters from 2.9 to 1.1 ka, and during the Little Ice Age, and by nutrient-rich waters from 1.1 to 0.54 ka (Medieval Climate anomaly) (Mojtahid et al. 2015). These conditions were linked to periods of low and high Nile River runoff respectively, in line with arid and humid climate conditions in the Levant and Nile headwaters.

These findings imply that surface productivity in the overall oligotrophic Mediterranean Sea responds rapidly to short and long-term changes in nutrient input, either via rivers, winds or upwelling activity, modifying the benthic-pelagic ecosystems by extending into the entire food chain (Marino and Ziveri 2013), ultimately increasing eutrophication.

Response of marine ecosystems to past changes in pH

Holocene reconstructions of paleo-pH have yet to be undertaken in the Mediterranean. There is a promising raw data record of planktonic foraminiferal (*Neogloboquadrina incompta*) $\delta^{11}B$ and B/Ca. These geochemical proxies can be used for paleo-pH and show an overall decreasing trend in both sub-basins of the Mediterranean Sea during the last deglacial episode of glacial-interglacial CO_2 rise (Grelaud et al. 2012; Marino and Ziveri 2013). The response of marine calcifiers to this trend can be estimated via planktonic foraminifera shell weight that shows overall decreasing planktonic calcification in response to this variability. In addition to this general trend, periods of changing seawater carbonate chemistry can be observed, which could be linked to low/high primary production activity such as the anomaly observed during Sapropel 1 period, which can be linked to enhanced mineralization of organic matter.

These first studies show that Mediterranean marine calcifiers responded to past changes in surface seawater carbonate chemistry conditions. The extent to which this affects marine ecosystems needs to be analyzed in the context of the current acidification in the Mediterranean's surface and deep seawaters.

4.1.2 Projected vulnerabilities and risks

4.1.2.1 Projected impacts and risks

As already discussed in Section 2.2.4.1, annual mean temperatures in the Mediterranean are now 1.5°C above late 19th century levels with magnitudes that vary locally depending on the period of analysis, the region and the type of dataset. The diurnal temperature range has also changed in some parts of the Mediterranean (Section 2.2.4.1). In absolute terms, the warmest parts are the southern and eastern Mediterranean and the major impact in these parts is the immigration of Indo-Pacific species (around a thousand species), which has accelerated in recent years, mainly for thermophilic species, due to rapid warming conditions (more than 50% of Mediterranean non-indigenous species are in the eastern Mediterranean) (Azzurro et al. 2011; Marbà et al. 2015; Kletou et al. 2016; Bariche et al. 2017). All Mediterranean waters, even the deepest, are affected by ocean acidification driven by Mediterranean Sea uptake of atmospheric CO_2 (Flecha et al. 2015; Hassoun et al. 2015; Palmiéri et al. 2015; Ingrosso et al. 2017) (Section 6.11). In addition, the effects of climate change are amplified by other major non-climate-related anthropogenic forcings, as the Mediterranean has one of the most populated coastlines with a long human history of exploitation of marine resources (with presently one of the world's most intense coastal and maritime tourism areas), habitat degradation and plastic pollution (Cózar et al. 2015; Compa et al. 2019). More information about sea use changes and pollution are covered in Chapter 2.

The combination of various ongoing climate change processes (e.g., sea warming, ocean acidification, and sea level rise; Section 2.6) has caused detectable effects on marine organisms at individual, population, and ecosystem scales (Fig. 4.4). Future risks of seal level rise, marine heat waves, and ocean acidification are also highlighted in Sections 6.9, 6.10 and 6.11 respectively. In fact, sponges, gorgonians, bryozoans, molluscs, and seagrasses are all affected by these drivers (Cerrano et al. 2006; Garrabou et al. 2009; Bensoussan et al. 2010; Marba and Duarte 2010), but primary producers, mainly calcifiers such as coccolithophores, are among the most vulnerable organisms (Meier et al. 2014). The impacts are expected to affect endemic and iconic ecosystems including major reorganizations of the biota distribution, species loss, marine productivity, increases in non-indigenous species, and potential species extinction (Malcolm et al. 2006; Ramírez et al. 2018; Gao et al. 2020).

Projected impacts on microbes

Sea warming may have effects on the virulence of pathogens (viruses, parasites, etc.), favoring the frequency of epidemiological events, as most pathogens are temperature sensitive (Vezzulli et al. 2013) (see Section 4.1.1 and Section 2.3.4 in Chapter 2 for more information about biological pollutants), as observed for *Vibrio shiloi*, responsible for the whitening of the coral *Oculina patagonica* in the eastern Mediterranean (Kushmaro et al. 1998). This warming is also responsible for the expansion of harmful and/or toxic microalgae, mainly dinobionts such as *Ostreopsis ovata*, which produces palytoxins, a serious public health hazard (Accoroni et al. 2016; Vila et al. 2016). Temperature anomalies also seem to negatively affect the chemical defenses of marine organisms (Thomas et al. 2007), allowing pathogens to act undisturbed. Given the predicted rise in temperatures over the coming decades, a better understanding of the factors and mechanisms that affect the disease process will be of critical importance in predicting future threats to temperate gorgonians communities (Bally and Garrabou 2007), and other affected species in the Mediterranean Sea.

In deep waters, a recent study has shown that deep-sea benthic Archaea can be more sensitive to temperature shifts than their bacterial counterparts. Changes in deep-water temperature may thus alter the relative importance of Archaea in benthic ecosystem processes (Danovaro et al. 2016). With rising deep-water temperatures, the predicted positive response of prokaryotic metabolism to temperature increases may accelerate oxygen depletion in deep Mediterranean waters, with domino effects on carbon cycling and biogeochemical processes across the entire deep basin (Luna et al. 2012). Along canyon-cut margins (e.g., the western Mediterranean), warming may additionally reduce density-driven domino effects, leading to decreased organic matter transport to the seafloor (Canals et al. 2006), though this very process is also likely to reduce physical disturbance on the seafloor and therefore affect deep-sea ecosystems.

Projected impacts on primary and secondary production

Climate change affects the functioning of the biological components of ecosystems, from the basis of the food webs (plankton) to the higher trophic levels (e.g., predator fish). Phytoplankton constitutes the autotrophic primary producers in the pelagic food chains in marine waters and their annual cycle is affected by many physical features that in turn control nutrient levels. These include large horizontal gradients in temperature (Izrael 1991). Due to their rapid turnover and fast responses to environmental changes, plankton is considered a suitable proxy to highlight either environmental changes circumscribed in space and/or time or wider climatic variations. Warming, for example, is responsible for the expansion of harmful and/or toxic microalgae, mainly the dinobionts such as *Ostreopsis ovata*, which produces palytoxins, a serious public health hazard (Accoroni et al. 2016; Vila et al. 2016). A new study in the Eastern Mediterranean has shown the occurence of important concentrations of biotoxins (domoic acid, gymnodimines and spirolides) in various marine organisms sampled from the Lebanese shores (Hassoun et al. 2021). These concentrations were correlated with the abundance of biotoxins' producers such as *Pseudo-nitzschia*, *Prorocentrum*, *Alexandrium*, and other species that could be occurring more frequently due to climate change (Hassoun et al. 2021).

Moreover, phytoplankton species responsible for bloom at late winter and at the beginning of spring (like *Skeletonema costatum, Nitzschia* spp, *Leptocylindrus danicus* and *L. minimus* and others) could start earlier, because features of temperate marine planktonic ecosystems are not only sensitive to annual variations in weather, but also any trends that might result from greenhouse warming or other factors that affect the climate system and both the density and timing of spring blooms will be altered in some regions (Townsend et al. 1994).

The taxonomic compositions of phyto- and zooplankton may change under the influence of changes in ocean structure (Kawasaki 1991; Berline et al. 2012; Howes et al. 2015) (Section 2.2.7). A thermophilic phytoplankton species could proliferate especially in some enriched areas and could be ichthyotoxic or even toxic for humans (Abboud-Abi Saab 2008, 2009; Accoroni et al. 2016; Abboud-Abi Saab and Hassoun 2017). Some examples can explain such variations. In the Mediterranean Sea, phytoplankton biomass abundance and sea surface thermal stratification show a strong inverse relationship at seasonal and sub-basin scales. At inter-annual and sub-basin scales, a gradual decline of the phytoplankton biomass across the entire central Mediterranean occurs with a delay of one year (Volpe et al. 2012). In the Adriatic Sea, during the past decade, the community structure and seasonality of phytoplankton have changed significantly. The phytoplankton annual cycle has become more irregular with sudden diatom blooms, reflecting the variability of meteorological events in recent years (Totti et al. 2019).

Only a few regional studies have investigated the sensitivity of the oligotrophic Mediterranean Sea to future climate change. The first investigations considered only the changes in circulation. For instance, a regional model of the northwestern Mediterranean domain found that the effect of local stratification due to climate change would have no drastic effect on the pelagic ecosystem (Herrmann et al. 2014). However, one study investigated the overall effects of a moderate climate change scenario (A1B SRES) on Mediterranean biological productivity and plankton communities and found an overall decrease in phytoplankton biomass in response to the stratification simulated in their dynamic climate change scenario (Lazzari et al. 2014). A simulation was carried out for an increase in integrated primary productivity across the eastern Mediterranean Basin as a result of changes in density (decreased stratification) (Macías et al. 2015). However, conclusions from these studies remain limited by the fact that they are based on non-transient simulations and present-day nutrient inputs.

A new study has investigated the influence of both changes in circulation and biogeochemical forcings (rivers and input at Gibraltar) (Richon et al. 2019). It suggests that climate change and nutrient inputs from river sources and fluxes through the Strait of Gibraltar have contrasting influences on Mediterranean Sea productivity (Section 2.3.3). Increased future stratification globally reduces surface productivity in the eastern basin, but the biogeochemistry in the western basin is strongly controlled by nutrient input across the Strait of Gibraltar, while the Eastern basin is more sensitive to vertical mixing and river inputs. In the near future, longer water stratification and warmer conditions may be essential clues to understanding local stress and species mortalities, especially because of the changes in primary productivity and lack of enough resources to face starvation periods (Rossi et al. 2017a). Once some species are eradicated, other species, pre-adapted to the new conditions, can replace locally extinct species, thus hampering ecosystem resilience to pre-impact conditions.

Unfortunately, the future evolution of nutrient atmospheric deposition has not been considered in modelling studies so far, despite the important role of nutrients for marine primary production (Richon et al. 2018b, 2018a). Using aerosol sampling and microcosm experiments performed during the Trans-Med BOUM cruise (June–July 2008), Ternon et al. (2011) showed that primary production significantly increased at all tested stations after aerosol addition collected on-board and after Saharan dust analog addition, indicating that both additions relieved on-going co-limitations, whereas a decline in the future of primary production is predicted and associated with trophic amplification toward phytoplankton and zooplankton (Richon et al. 2019).

Abrupt community shifts are expected in plankton communities coinciding with climate changes that alter local thermal regimes, which in turn interact with the thermal niche of species to trigger long-term and sometimes abrupt shifts at the community level (Beaugrand et al. 2019). Planktonic calcifying organisms such as foraminifera, pteropods and coccolithophores are expected to be particularly affected by ocean acidification and climate change. A recent study in the Mediterranean found foraminifera to be highly susceptible to temperature-induced surface water stratification and food availability. In correlation with these results, rapid warming increased surface ocean stratification impacting food availability and changes in trophic conditions could be the causes of reduced foraminiferal abundance, diversity, and species-specific changes in planktic foraminiferal calcification (Mallo et al. 2017). Coccolithophores, an abundant unicellular calcifying phytoplankton, are known to have a haplo-diploid life cycle with environmental affinities. This dimorphic life cycle might provide the ability to adapt to the ‘tropicalization’ of Mediterranean environments under climate change, in conditions characterized by surface water with a relatively high calcite saturation state, high temperature, stratification and nutrient limitation (D’Amario et al. 2017).

Projected impacts on macrobenthic and pelagic species

Sea warming, ocean acidification, sea level rise and changes in circulation patterns will likely change Mediterranean benthic and pelagic ecosystems, as shown in Figure 4.4, where potential impacts related to climate change are presented (Rossi et al. 2019). Repercussions will be different depending on the region and on the dominance of certain benthic organisms, species interactions, metabolic constraints, dispersion capability, and the presence of alien species that may take advantage of the new physical, chemical, and biological conditions in the future oceans. Changes in trophic relationships will likely change biodiversity, both in the water column and in the different benthic communities. The nursery effect (i.e. sheltering and feeding grounds for juveniles and larvae) may be compromised if massive mortalities or deoxygenation affect the three-dimensional live structures (i.e., Posidonia meadows, gorgonian forests, pluriannual macroalgae, cold water corals, etc.). Also, changes in river runoff due to different rain frequency/quantities will also have effects on coastal nutrient and pollutant inputs. Nutrient and pollutant equilibria will also have direct or indirect effects on pelagic and benthic communities (Rossi et al. 2019).

Corals

Environmental change-driven modifications to the environment of gorgonian forests influence the larval settlement and recruitment processes of the benthic assemblages (Ponti et al. 2014b). In experiments, red coral (*Corallium rubrum*) showed a 59% decrease in its calcification rate at lowered pH based on experimental studies (Bramanti et al. 2013). Disturbances such as harvesting pressure could act in synergy with ocean acidification bringing local populations to extinction. The changes in the likelihood of occurrence obtained by differences between present conditions and future scenarios show that the projected extent of potential mortality zones is higher than in the current climate for red coral and that coralligenous formations along the Mediterranean sites are less likely, mostly due to acidification increase. However, the adverse impact is localized to certain regions: an increase in likelihood is also reported for the presence of coralligenous in the North Aegean and northern Adriatic Seas (MEDSEA 2015; Gómez-Gras et al. 2019).

The effects of *in situ* exposure to different pH levels (7.4–8.1) and temperatures (15.5–25.6 °C) on mortality and net calcification rates have been assessed for Mediterranean scleractinian corals transplanted near a volcanic CO_2 vent off Panarea Island (Prada et al. 2017). Results of this *in situ* study show a synergistic adverse effect on mortality rates for all corals (up to 60%), suggesting that high seawater temperatures may have increased their metabolic rates which, in conjunction with decreasing pH, could have led to rapid deterioration of cellular processes and performance. The same study (Prada et al. 2017) suggests that symbiotic corals may be more tolerant to increasing warming and acidifying conditions compared to asymbiotic corals. Laboratory cultures of coralline algae under conditions of elevated temperature and pCO_2 revealed effects on photosynthesis, growth and calcification. Crustose coralline algae (*Neogoniolithon brassica-florida*) sensitivity to ocean acidification examined in CO_2 seeps confirmed that calcifying algae are likely to be threatened by ocean acidification, especially species living near their thermal limit. Further *in situ* and laboratory experiments indicate that *N. brassica-florida* may not be able to contribute to reef accretion under the levels of seawater warming and ocean acidification projected by the end of this century (Fine et al. 2017).

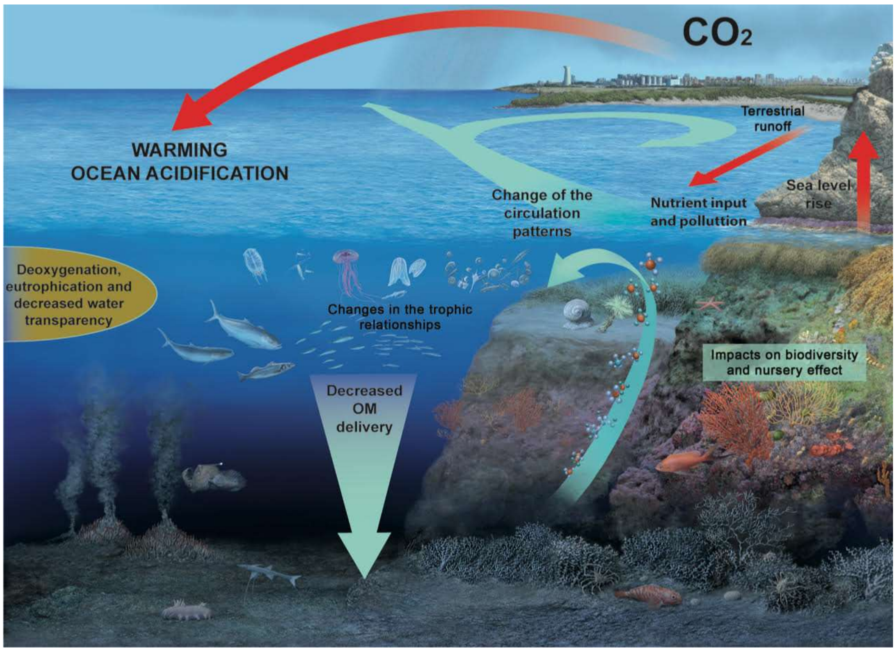

Figure 4.4 | Different drivers potentially affecting marine pelagos and benthos in the Mediterranean Sea (original diagram in Rossi et al. 2019, artwork by A. Gennari).

Seagrass

Warming can induce declines in abundance through increased shoot mortality in Mediterranean *Posionia oceanica* meadows. Younger life stages (i.e., seedlings) of *P. oceanica* may be particularly vulnerable to climate change. Insights into acidification effects on seagrasses have come from CO_2 vent surveys showing consistent loss of crustose coralline algal epiphytes on seagrass leaves, and greater seagrass density close to seeps with a lower pH (Hendriks et al. 2017). Lower epiphyte loads can have positive impacts for seagrass as it reduced shading and nutrient uptake by the epiphytes.

In the case of seagrass such as *P. oceanica,* projections show a negative impact due to the effects of global warming over the next century and to significant climate change challenges posed to an endemic system that is already suffering losses from anthropogenic impacts (Jordà et al. 2012). The trajectory of *P. oceanica* meadows under the warming expected in the western Mediterranean was studied through the twenty-first century and warming seems to likely lead to the functional extinction of *P. oceanica* meadows by the middle of this century (year 2049 ± 10) even under a relatively mild greenhouse-gas emissions scenario (Jordà et al. 2012). Similarly, the distribution of two seagrass species under different scenarios was forecasted (Chefaoui et al. 2018), and the results found that, in the worst-case scenario (RCP 8.5 scenario), *P. oceanica* might lose 75% of suitable habitat by 2050 and is at risk of functional extinction by 2100, whereas *Cymodocea nodosa* would lose around 46.5% of suitable habitat by 2050. The same study (Chefaoui et al. 2018) also predicts that erosion of present genetic diversity and vicariant processes can happen, as all Mediterranean genetic regions could decrease considerably in extension in future warming scenarios. The functional extinction of *P. oceanica* would have important ecological impacts and may also lead to the release of the massive carbon stocks these ecosystems stored over millennia.

Mussels

The Mediterranean mussel *Mytilus galloprovincialis* is particularly sensitive to warming. A significant decrease in growth (total weight, shell length, shell weight) has been found in warmer conditions as well as clear dissolutions of the shells exposed to low pH conditions (-0.3 compared to ambient). Mussels exposed to low pH showed a clear loss in the organic layer covering the shell in summer, explaining the clear dissolution signal measured on these mussels (Gazeau et al. 2014). Non-climate drivers, such as chemical pollution (Section 2.3.3), are exacerbating the pressures on this important commercial mussel (*M. galloprovincialis*).

Jellyfish

The sensitivity and specificity of the Mediterranean Sea to climate change and other human-related stressors have been unequivocally documented in recent years by a significant increase in the number and frequency of jellyfish outbreaks (Coll et al. 2010; Canepa et al. 2014). Ocean warming and acidification may favor the dissemination of the non-indigenous *Cassiopea andromeda* that seems to benefit from the changed conditions (Fuentes et al. 2018). Also, seawater temperature, together with the quantity and quality of available food resources, are known as major drivers of gonadal outputs (Harland et al. 1992; Ben-David-Zaslow and Benayahu 1999). Some studies showed that elevated temperature by itself or in combination with high feeding frequency (due to raised zooplankton prey abundance) increased the budding rate and bud size in *Aurelia* polyps populations worldwide (Hočvar et al. 2018). Thus, more food and warmer waters may be the key to understanding proliferation of jellyfish in general and non-indigenous tropical species in particular. In a sea highly impacted by the alteration of the trophic chains due to overfishing, seawater warming favors the successful dispersion and growth of jellyfish.

Winners and losers

The western basin is acidifying faster than the eastern basin (Goyet et al. 2016). A first tipping point has already been reached, since anthropogenic CO_2 is already over 82 µmol kg^{-1} in many Mediterranean areas (Hassoun et al. 2015). The exact timing of the tipping points (Section 2.2.9.2) will strongly depend on the policies controlling human activities, which will impact both global warming and the anthropogenic CO_2 increase both in the atmosphere and into the ocean. The results of these projections raise concerns about how marine organisms will respond in the context of each scenario, after reaching every tipping point.

Mean warming, acidification and associated non-climatic stressors will have varying impact across the Mediterranean marine ecosystems, and result in both potential winners and losers. Impacts of ocean acidification and warming may extend to several Mediterranean marine and coastal ecosystem services, food provision, recreational activities, carbon absorption, climate regulation, coastal protection, and ultimately affecting human health (Falkenberg et al. 2020). Marine areas with economic activities directly depending on marine resources may face serious impacts on employment and benefits in sectors like aquaculture, open sea fisheries and tourism, which is relevant to many Mediterranean countries. Tourism may be affected by acidification and warming through the impact of degraded marine ecosystems (loss of iconic coralligenous species, such as gorgonians - soft coral (Bramanti et al. 2013) from diving experiences and through jellyfish outbreaks). Sensitivity of shell-forming species such as bivalve mollusks to changes in temperature and acidity presents a threat to the aquaculture sector (Rodrigues et al. 2015).

A reduction in primary production linked to an increase in sea surface temperature could have negative impacts on fisheries catches and could exacerbate current overfishing trends (Cheung et al. 2010). Projections of biomass and fisheries catches across the Mediterranean under the high emission scenario RCP8.5 suggest a 5 and 22% increase in total fish and macroinvertebrate biomass , and a 0.3 and 7% in fisheries catches by 2021–2050 and 2071–2100, respectively, combined with changes in primary and secondary production (Moullec et al. 2019). Winners were mainly small pelagic species, thermophilic and/or exotic species, of smaller size and of low trophic levels. Loser species are generally large-sized, some of which are of great commercial interest. The bulk of the increase is expected in the southeastern part of the basin whereas significant decreases are most likely in the western Mediterranean Sea. Read more about projections for marine food resources and fisheries in Sections 2.4.2.3 and 3.2.2.2.

Since temperature seems to be a main environmental parameter driving the cnidarian community composition, abundance and spatial distribution patterns in the Mediterranean Sea, jellyfish are considered a possible group of winners under warming (Guerrero et al. 2018). The structure and phenology of the Mediterranean hydrozoan community displayed significant changes in species composition, bathymetric distribution, and reproductive timing over the last decades. When the Scyphozoa group is considered, *Pelagia noctiluca* (among the most abundant jellyfish in the Mediterranean Sea and Eastern Atlantic waters) has increasingly frequent massive outbreaks associated with warmer winters (Milisenda et al. 2018). Swarms of the Portuguese Man-of-War (*Physalia physalis*), in summer 2010 were the result of an unusual combination of meteorological and oceanographic conditions during the previous winter and not a permanent invasion favored by climate changes (Prieto et al. 2015). However, many studies have attributed the increase in *Pelagia noctiluca* outbreaks to the alteration of the trophic structure of ecosystems due to overfishing and/or eutrophication on the one hand, and by sea warming and changes in surface hydrography on the other (Licandro et al. 2010; Canepa et al. 2014). Water temperature affects sexual reproduction through changes in the energy storage and gonad development cycles and it is still expected that the species composition and biogeography of jellyfish communities will change under global warming.

4.1.2.2 Vulnerabilities

Climate-related vulnerabilities

It is expected that the ocean’s primary production will, in general, be reduced with environmental change. As a result, production zones may be redistributed and the natural habitat of commercially valuable species of fish may change (Izrael 1991). On the other hand, climate change can also lead to changes in the composition of the bottom of marine food webs. The rise in water temperature has already increased jellyfish population outbreaks in the Mediterranean Sea (Section 4.1.2.1), such as *Pelagia noctiluca*, a planktonic predator of fish larvae and of their zooplankton prey. The outbreaks of this species, along with other jellyfish species, may become more frequent in the Mediterranean Basin in the future and extend over a longer period of the year than previously, causing changes to the pelagic food web and thereby reducing fishery production (Licandro et al. 2010). Rising seawater temperatures might also trigger the increased spread of pathogens throughout the Mediterranean in the future, affecting both marine organisms, and human health (Danovaro et al. 2009) (see Section 5.2.3 on heat-related impacts).

Temperature has a major direct impact on the physiology, growth, reproduction, recruitment and behavior of marine organisms such as fish. Warming associated with climate change already affect the Mediterranean ecosystem for some benthic and pelagic species (Marbà et al. 2015). Warming combined with a decline in oxygen and resource availability reduces fish body size, with the average maximum body weight of fish expected to shrink by 4% to 49% from 2000 to 2050 (Cheung et al. 2013). Also, fish tend to adapt to local environmental temperatures. Therefore, among the most perceptible large-scale consequences of climate change is the shift in spatial distribution range of marine organisms, which will make some Mediterranean sub-basins more vulnerable to drivers than the others.

Seawater warming will induce a loss of climatically suitable habitats for various organisms, causing distribution shifts, as well as species extinction. The diversity of fish assemblages is predicted to be severely affected due to their loss of suitable climatic niches. Demersal species will suffer regional impacts associated with the expected changes in primary production, thermohaline circulation, and the severity of winter weather (Section 2.2.2). Warming and the expected increase in Atlantic water entering into the Mediterranean will likely affect migrations and spawning behavior in large pelagic fish (Barange et al. 2018). In recent decades, several mass mortality events of invertebrates have occurred in the Mediterranean which have been linked to the documented rise in seawater temperatures (Rivetti et al. 2014).

Projections for the global warming scenario (SRES A2) for the potential future distribution of 75 Mediterranean endemic fish species have shown that by 2041-2060, 31 species are projected to extend their geographic range, whereas the geographic range of 44 species is projected to reduce (Ben Rais Lasram et al. 2010). Also, 25% of the Mediterranean continental shelf is predicted to undergo an overall change to endemic assemblages by the end of the 21st century, where the survival of 25 species is threatened and six species would become extinct (for example, starry sturgeon *Acipenser stellatus* and European sturgeon *Huso huso*). For “narrow” endemic species found strictly in the Mediterranean Sea that do not reach the neighboring Atlantic Ocean and Black Sea, their extinction would be irreversible. By the middle of the 21st century, the coldest areas of the Mediterranean Sea, namely the Adriatic Sea and the Gulf of Lion, would act as a refuge for cold-water species, but by the end of the century, those areas are projected to become a ‘cul-de-sac’ that would drive those species towards extinction (Ben Rais Lasram et al. 2010). Another study concludes that 54 species will have lost their climatically suitable habitat at the end of the century and that species richness will decrease across 70.4% of the continental shelf area (Albouy et al. 2013). Information about the trends of fisheries in the Mediterranean is detailed in Sections 2.4.2.1 and 3.2.1.2.

Anthropogenic vulnerabilities

Ship collisions and harmful fishing practices are among the non-climate drivers exacerbating the vulnerability of marine cetaceans in the changing Mediterranean. The previous analysis of ship collision records for the relatively isolated population of fin whales in the Mediterranean Sea from 1972 to 2001, indicated that the fatal collision rate increased from 1 to 1.7 whales per year during this period mainly in the Pelagos Sanctuary (the largest marine protected area created for marine mammals in the Mediterranean; see Section 4.1.1) due to high levels of traffic and whale concentrations (Panigada et al. 2006). Studies by Pennino et al. (2016, 2017) assessed the risk exposure for high intensity vessel traffic areas for the three most abundant cetacean species (*Stenella coeruleoalba*, *Tursiops truncatus* and *Balaenoptera physalus*) in the Southern area of the Pelagos Sanctuary. They modeled both the occurrence of three cetacean species and marine traffic intensity, and identified two main hot-spots of high intensity marine traffic in the area, which partially overlap with the area where the studied species are present. International shipping, although considered as an environment-friendly form of transportation, directly and indirectly impacts cetaceans in many ways, particularly in the Mediterranean Sea, one of the world's busiest waterways (Bray et al. 2016; Coomber et al. 2016). More recent data about maritime traffic in relation to cetaceans, investigated through direct observations (July 2013–June 2015) and along three fixed transects in western Mediterranean areas, showed seasonal maritime traffic intensity with the highest vessel abundance impacts on cetaceans in most offshore sub-areas in the spring and summer, especially for the species *B. physalus* and *S. coeruleoalba* (Campana et al. 2017).

4.1.3 Adaptation

4.1.3.1 Long-term monitoring and adaptation strategies

Temperature significantly affects eukaryotic phytoplankton metabolism, increasing the demand for nitrogen with consequences for the marine carbon cycle due to shifts towards N-limitation (Toseland et al. 2013). Experiments reveal that some taxa of marine phytoplankton may adapt to ocean acidification, and there are also strong indications from studies of variation and structure in natural populations that selection on standing genetic variation is likely (Collins et al. 2014).

To better evaluate the adaptation strategies of plankton communities to the diverse climate and non-climate related drivers, it is necessary to assess the vulnerabilities of the pelagic ecosystem, including both plankton and nekton communities, to the impacts of climate change, including temperature and pH variations. Taking into consideration the diversity of plankton communities' responses to climate change and other stressors in every Mediterranean sub-basin (Crise et al. 1999; Psarra et al. 2005; Vadrucci et al. 2008; Calvo et al. 2011; Marić et al. 2012; Ouba et al. 2016; Danovaro et al. 2017; Benedetti et al. 2018), wider monitoring coverage is needed to improve our knowledge about the different adaptation processes that characterize and best suit each geographical zone. Since each Mediterranean Basin represents a unique set of interrelated physical, biological and human components and processes, the extent and nature of impacts in one basin will differ between sub-basins as well.

Also, an assessment of the implications of accelerated temperature increase in the Mediterranean Sea and the identification of the types of problems that a marine area will face is crucial in order to anticipate the need for action (Lacoue-Labarthe et al. 2016). In their review about the biodiversity and ecosystem functioning, Danovaro and Pusceddu (2007) have recommended enhanced strategies for protecting the Mediterranean Sea, such as monitoring environmental quality, grey and black lists of chemicals, utilizing the best available technologies once they have been tested for their ecosustainability, applying precautionary principles (e.g., reducing pollution emissions), monitoring biodiversity and long-term temporal changes in community structure. The suggestion has been made to assess not only the apparent changes, but also the potential biodiversity (cyst banks) while paying particular attention to species replacement in relation to functional biodiversity (Danovaro and Pusceddu 2007).

4.1.3.2 The role of Marine Protected Areas (MPAs) for adaptation

Marine Protected Areas cannot halt climate change and impacts such as ocean acidification, but they are an important tool for enhancing the resilience and adaptive capacity of ecosystems. A topical example in this sense are Mediterranean coralligenous reefs, critical for life-supporting and ecological functions and providing a natural capital like biomass production, erosion control, historical, economic and intrinsic value. Marine protected areas provide refuge and replenishment zones for this heavily exploited and vulnerable species. Moreover, they protect, aid recovery and are home to a large number of species (Rodríguez-Rodríguez et al. 2015; Pascual et al. 2016). Few MPA designs account for official MPA boundaries regardless of boundary overlaps and their ecological implications (Gabrié et al. 2012), whereas others account for MPA boundary overlaps and thus foster a more ecologically meaningful, functional spatial approach through “protected polygons” (Foster et al. 2014).

The effectiveness of MPAs can be improved if they form part of a system of protected areas geared towards ensuring ecological representativeness and creating networks. Nevertheless, subdividing an area into zones with varying intensities of use (zoning), ranging from total protection (marine reserves where extractive use is prohibited) to areas serving primarily to uphold sustainable and/or traditional use of marine resources, and areas that are closed to fishing activities, is increasingly recognized as a useful instrument for sustainable, ecosystem-based fisheries management, particularly artisanal fisheries (Pascual et al. 2016). MPA ecological effectiveness, defined as species self-replenishment and colonization through dispersal, depends, among other variables, on MPA design factors such as size, shape, spacing and location (Shanks et al. 2003; OSPAR Commission 2007; Roberts et al. 2010; Sciberras et al. 2013). Different categories of MPAs often sit side by side with core areas under strict protection and peripheral zones with fewer restrictions relating to use.

An example of these MPAs is the Mediterranean Marine Protected Area of Medes Islands, in the north-western Mediterranean Sea. This area was recently the subject of an economic assessment, using monetary valuation, of changes in the quality of highly biodiverse coralligenous systems (Rodrigues et al. 2015). The Adriatic Sea, for example, has the largest number of MPAs and also the smallest, least spaced and least compact designated and functional MPAs. The design pattern in this ecoregion seems to have randomly followed a design approach of “several small” MPAs (Rodríguez-Rodríguez et al. 2015). The establishment of a transboundary Large Marine Protected Area (LMPA) and Fisheries Restricted Area (FRA) can reverse ecological and socio-economic losses in the Adriatic, one of the most exploited areas of the Mediterranean as shown in a study that presents current opportunities and expected benefits of LMPAs (Bastari et al. 2016). This demonstrates that the establishment of MPAs should take into consideration the structural and functional links between key organisms within the ecosystem and between ecosystems to guarantee a sustainable adaptation strategy.

The majority of MPAs are located along the basin's northern shores, highlighting the lack of MPAs in the south and east coasts (Abdulla et al. 2008). Coll et al. (2012) studied the interaction between marine biodiversity and threats (including climate change) across the Mediterranean and assessed their spatial overlap with current marine protected areas. They identified areas of conservation concern where future protection activities should be targeted through spatial prioritization. Spatial prioritization in conservation is commonly employed to direct limited resources to where actions are most urgently needed and most likely to produce effective conservation outcomes. Resilience is increasing through building PA networks and setting priorities at the regional level. Examples for the conservation of three key Mediterranean habitats, i.e. seagrass *Posidonia oceanica* meadows, coralligenous formations, and marine caves, were determined through a systematic planning approach (Giakoumi et al. 2013).

4.1.3.3 Management of fisheries and adaptation

Fisheries is one of the main sectors related to resource-based growth activities in the Mediterranean area (Section 5.1.1.3). Sustainable development of fishing activities and the management of their impacts require better constructive collaboration between scientists, industry and government agencies. The analysis of all available stock assessment and effort data for the most important commercial species and fleets in the Mediterranean Sea since 2003, demonstrated a significant decline for red mullet and giant red shrimp stocks (Cardinale et al. 2017). This latter study concluded that the European Common Fisheries Policies have failed to achieve the maximum sustainable yield before 2015 for the Mediterranean Sea and will face large difficulties to reach maximum sustainable yield and Marine Strategy Framework Directive targets before 2020 under the current management system, due to many factors, such as the ineffectiveness of the putative effort reductions to control fishing mortalities, the continuous failure to follow scientific advice, and the existence of ineffective national management plans as a primary management measure (Cardinale et al. 2017).

The establishment and implementation of management plans that could efficiently help fisheries to adapt relies on knowing each species properties and demographic features. For example, the local and isolated groups of the European anchovy (*Engraulis encrasicolus*) may have unique demographic properties and should be managed separately since they may react independently to exploitation (Jemaa et al. 2015a). The same conclusion has been highlighted for pelagic species (such as the European sardine, *Sardina pilchardus*) with high gene flow to ensure sustainable fishery benefits and efficient conservation as they also may have unique demographic properties and responses to exploitation (Jemaa et al. 2015b).

The current knowledge on Mediterranean fisheries and ecosystems is limited. In fact, the effect of poorly regulated fisheries, in combination with ongoing climate forcing and the rapid expansion of non-indigenous species, are rapidly changing the structure and functioning of ecosystems with unpredictable effects on the goods and services provided (Colloca et al. 2017). Although this would call for urgent conservation actions, the management system implemented in the region appears too slow and probably inadequate to protect biodiversity and secure fisheries resources for future generations. This is why some studies are encouraging the adoption of other management approaches such as the establishment of a transboundary Large Marine Protected Area (LMPA), specifically a no-trawl area LMPA or Fisheries Restricted Area (FRA), which is a promising and feasible approach for reversing ecological and socio-economic losses in some Mediterranean sub-basins such as the Adriatic (Bastari et al. 2016), as mentioned earlier in Section 4.1.3.2.

Maintaining ecosystem services (through efficient fisheries management, sustainable and ecofriendly aquaculture industry (Section 3.2) is crucial for the food security, economic growth and well-being of neighboring populations (Section 5.1.1.3). Developing practical management actions that take into consideration the uniqueness of each species and their responses towards different drivers is crucial to increasing their resilience and plasticity in the context of climate change.

4.1.3.4 Adaptation strategies for ocean warming and ocean acidification in the Mediterranean Sea

Studies on adaptation to climate change in the Mediterranean Sea are still very limited. Actions considered are mostly supply-side oriented, aimed at restoring or protecting the production of marine goods and services harmed by ocean warming and acidification for example (Ziveri et al. 2017). The demand-side dimension can ultimately produce economic consequences of the same or greater magnitude than adaptation through supply-side strategies and actions.

The Mediterranean Sea is a marine biodiversity hotspot (Coll et al. 2010) and ecosystems with high biodiversity and/or redundancy of functional groups (for example, several species fill the role of algal grazers) tend to be more resilient and recover more quickly following disturbance. This implies that biodiversity preservation and improvement are logical methods for sustaining ecosystems responding to rapid environmental stressors. In practice this means, on the one hand, exploiting the acclimation potential of many calcifying species of the Mediterranean and, on the other hand, protecting other species. The recommendation of a drastic reduction of local drivers is strictly connected with mitigation and adaptation strategies, since reducing local stressors – such as land-based pollution, coastal development and overharvesting – is the most common strategy for improving or maintaining ecosystem resilience (read more on economic vulnerabilities/risks and the adaptation measures in Sections 5.1, 6.10.2 and 6.11.2).

In the Mediterranean, commercial fisheries are economically important on a regional and local scale or for some specific communities and user groups. Although there is still limited knowledge on the combined direct impacts of ocean acidification and warming on fish, there are indications of the physiological and behavioral effects of CO_2 on fish (Nilsson et al. 2012; Milazzo et al. 2016). However, the two phenomena could indirectly impact fisheries affecting phytoplankton community structures at the bottom of the food web (Nagelkerken et al. 2016). In general, when fishing activity is more 'sustainable', it tends also to be more resilient to negative shocks. All the measures working in this direction are thus also suitable for mitigating the adverse economic effects of ocean acidification.

Aquaculture is a key economic sector of fisheries affected by ocean acidification and warming. In the Mediterranean, detrimental effects on bivalve mollusk species might arise from the associated increase in sea surface temperature, ocean acidification and possible synergies with other non-climate drivers (Gazeau et al. 2014). A study suggests that the increase in frequency and duration of summer heatwaves are perceived as the highest threat, having been observed in a majority of the studied production sites in past years, with effects on seed (spat), adult mortality and byssus attachment (Rodrigues et al. 2015). Ocean acidification knowledge transfer and monitoring programs are essential for the development of appropriate strategies to counteract the effects of these phenomena, which are still poorly known by stakeholders. Adaptation in this sector tends to be particularly expensive since it usually requires costly investment in new machinery or in modifying existing machinery so as to reduce negative environmental impacts. A recent study emphasizes that the cost of management and mitigation strategies and actions will be dependent upon the socio-economic context. Specifically, costs will likely be greater for socio-economically disadvantaged populations, exacerbating the current inequitable distribution of environmental and human health challenges (Falkenberg et al. 2020).

Some species are proven to improve the resilience of their habitat to various drivers. For example, *Paramuricea clavata* forests may enhance bioconstruction processes and increase the resistance and resilience of benthic assemblages in Mediterranean coralligenous habitats (Ponti et al. 2014b, 2018). The lack of available food, rising temperature and decreasing pH trends will be essential to understanding future population dynamics. Bioengineering as a possible adaptation strategy includes techniques to mitigate chemical effects of increased atmospheric CO_2 concentrations on the oceans. These chemical changes may have a variety of important biological consequences, including some potentially negative impacts, which are controversial and surely require further consideration. These ideas have never been tested *in situ* (Ziveri et al. 2017).

In conclusion, any kind of action that improves marine ecosystem health, resilience or biodiversity could delay and reduce the adverse effects of climate drivers. This includes the implementation of more sustainable fishing practices as well as reducing pollution from agricultural activity, sustainable tourism and developing more effective waste management. Marine protected areas can potentially have an insurance role if they are placed in locations not particularly vulnerable to ocean acidification and climate change. However, the detrimental effects of these global phenomena on certain habitats in vulnerable regions do not make MPAs easily effective in improving species resilience to environmental change when considering long-term strategies. Developing specific adaptation options, for example new practices in aquaculture or improving marine and coastal protection against storm surges and coastal erosion, could be effective but particularly costly (see Sections 3.3.2.2 and 3.3.2.3 for more information about the use of marine energy as a renewable energy resource and the vulnerability of coastal energy systems to climate extremes).

Adaptation strategies must have medium- to long-term effectiveness. They thus require careful and anticipatory planning to enjoy their benefits reasonably soon, and especially to enable them to tackle problems while they are still manageable. Overall, adaptation strategies are a necessary to response to ongoing and expected Mediterranean environmental changes. However, the necessary strategy for reducing climate change impacts needs effective mitigation policies and actions to be implemented.

4.1.3.5 Regional observation networks as a tool for adaptation

Another aspect that might improve the effectiveness of managed adaptation strategies is the establishment of active regional and local observation networks. Local observation programs and regional networks that include scientists from different Mediterranean countries/sub-basins create a solid platform for peers to collaborate in monitoring climate change drivers and impacts, enhance data sharing policies and accessibility, and improve capacity-building among the members of their scientific community.

Furthermore, long-term active hubs would definitely help in deriving more robust findings about the different environmental trends in the Mediterranean, which will provide more comprehensive and conclusive results for decision makers. Within the Global Ocean Acidification-Observing Network (GOA-ON), a regional Mediterranean hub has been recently established, called the Ocean Acidification Mediterranean Hub[1] (OA Med-Hub). This hub could be an important platform for providing robust ocean acidification-related results for the scientific community, the general public and decisions-makers, which would help to create relevant future adaptation actions in Mediterranean countries.

Moreover, these regional networks could be an effective tool for improving public awareness and enhancing capacity-building among scientists who are not able or do not have the appropriate tools/equipment to monitor specific phenomenon. Thus, such hubs could unify the methodologies and tools adopted to measure and monitor short- and long-term climate change trends, and their effects on local and regional marine resources and ecosystems. They could also help obtain funds for laboratories that do not have the capacities to properly survey climate change drivers and impacts. For example, GOA-ON published recently an implementation strategy document[2] to provide guidance that will harmonize sampling and analysis procedures, to compare results and trends. Creating similar

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[1] http://goa-on.org/regional_hubs/mediterranean/about/introduction.php

[2] http://goa-on.org/documents/general/GOA-ON_Implementation_Strategy.pdf

networks that could target other phenomena, such as deoxygenation, warming, etc., and good communication between these hubs will be crucial for developing suitable and holistic key messages that could be provided to policy-makers.

4.2 Coastal ecosystems

4.2.1 Current condition and past trends

4.2.1.1 Observed changes

The coastal zone refers to the area in which the interaction between marine systems and the land dominate ecological and resource systems. These rather complex systems consist of coexisting biotic and abiotic components which interact with human communities and their socio-economic activities (UNEP/MAP/PAP 2008). In sensitivity studies concerning expected sea level rise, the term "low-elevation coastal zone" (LECZ) has been used for the specific area up to an elevation of 10 m (Vafeidis et al. 2011). Ecosystems in this zone are referred to hereafter as "coastal ecosystems". The natural coastal systems include distinct coastal features and ecosystems such as rocky coasts, coral reefs, beaches, barriers and sand dunes, estuaries, lagoons, and generally the land located at the lower end of drainage basins, where stream and river systems meet the sea and are mixed by currents and tides (i.e. deltas, river mouths, wetlands) (Convertino et al. 2013).

To evaluate the risks that could affect Mediterranean coastal systems, their natural habitats, or particular events that could occur on its shores, many indexes have been developed, such as the Environmental Sensitivity Indices (ESI) developed for the evaluation of oil spill risk in Mediterranean coasts (Gugliermetti et al. 2007), the participatory multicriteria analysis (MCA) for a multidimensional assessment of coastal erosion risks (Roca et al. 2008), and the coastal dune vulnerability index for Mediterranean ecosystems (Ciccarelli et al. 2017). Also, a Coastal Risk Index (CRI-Med) has been developed to assess coastal vulnerabilities associated with the physical and socio-economic impacts of climate change in all Mediterranean coastal zones (Satta et al. 2017; Fig. 4.5). By applying the CRI-Med on 21 Mediterranean countries, coastal risk hotspots are found to be predominantly located in the South-Eastern Mediterranean region. Countries with the highest percentage of extremely high-risk values are Syria (30.5%), Lebanon (22.1%), Egypt (20.7%), and Palestine (13.7%). Coastal hotspots are designated to support the prioritization of policies and resources for adaptation and Integrated Coastal Zone Management (ICZM).

Natural Mediterranean habitats under severe degradation

Climate change affects marine biodiversity, especially in coastal habitats (Gatti et al. 2015; Bertolino et al. 2016, 2017a; Betti et al. 2017; Longobardi et al. 2017). Mediterranean shores have unique and highly diversified landscapes, which harbor a high level of diversity in fauna and flora (Section 2.4). Directly or indirectly, sea level rise (Section 2.2.8), global warming (Section 2.2.4) and changes in rainfall patterns (Section 2.2.5) would greatly modify coastal ecosystems and rivers with significant impacts on the way they function. For instance, in response to relative sea level rise, coastal wetlands and river mouths would be affected, while reduced precipitation and prolonged droughts will reduce the water discharge of Mediterranean rivers and catchments (Merheb et al. 2016). Dynamic coasts are likely to retreat or disappear because of the effects of erosion due to the accelerated rise in sea level. With the accelerated rise in sea level, the least mobile species will likely be the most severely impacted in contrast to mobile organisms such as fish and water-birds which are more able to adapt.

Coastal ecosystems function in a way that maintains the preservation of energy and matter transfer and plays an important role in global cycles (such as carbon and nitrogen global cycles) between the continental and marine realms. Furthermore, in addition of hosting a wide diversity of wild faunal and floral species, coastal ecosystems are also often used as aquaculture platforms (i.e. fish, shellfish cultures, etc.), and the pressures on them may have significant impacts on their uses.

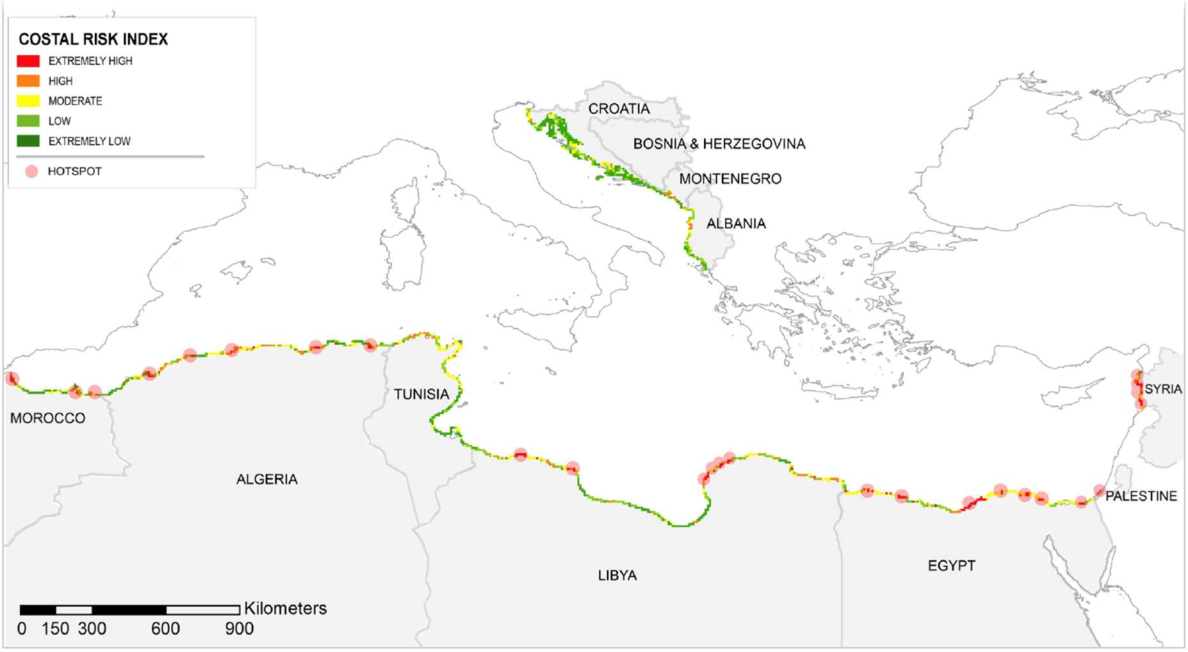

Figure 4.5 | Coastal Risk Index (CRI-MED) map of the Mediterranean, indicating five levels of vulnerability (Satta et al. 2017).

Sandy beaches and sand dunes

Escalating pressures caused by the combined effects of population growth, demographic shifts, economic development and global climate change pose unprecedented threats to sandy beach ecosystems across the world (Velegrakis et al. 2016; Vousdoukas et al. 2016). From the sub- to the supralittoral, sandy habitats are important in preventing coastal erosion and flooding, but their value may be enhanced by the many biological processes that complement or even increase their role in coastal defense. For example, in addition to their role in nourishing other sandy systems, shallow, sub-tidal sands also support seagrass beds, a habitat increasingly recognized as important for coastal protection due to their ability to stabilize and accumulate sediment, and attenuate and dissipate waves (Christianen et al. 2013; Ondiviela et al. 2014).

In addition to direct anthropogenic impacts (e.g., urban and agricultural development, Sections 2.3 and 2.4), climate change is predicted to have dramatic, widespread and long-lasting consequences on sandy coastal ecosystems. For instance, sea-level rise increases the phenomenon of 'coastal squeeze', but increased storm intensity and frequency are likely to be the major challenges faced (Feagin et al. 2005; Harley et al. 2006; Velegrakis et al. 2016; Vousdoukas et al. 2016).

As ecosystems, sandy beaches play varying important roles, and interact closely with coastal dunes both physically and biologically. The typical coastal dune system is composed mainly of three components: the submerged beach, the emerged beach and the dune (Figure 4.6). For thousands of years, human activities have been impacting the sandy beaches and coastal sand dunes of the Mediterranean Basin through agriculture, husbandry and the deliberate use of fire (Lavorel et al. 1998; Cori 1999; Falcucci et al. 2007). In recent decades, tourism has caused important damages as the main driver of coastal urbanization, the increase of summer visitors, and the introduction of non-indigenous or exotic species (Tzatzanis et al. 2003). The pedestrian and motorized pathways all over dunes lead to vegetation destruction and therefore enhanced weathering and erosion. Waste and non-indigenous species introduction are also destruction factors among many other drivers highlighted in Sections 2.3 and 2.5.

The impact of human pressure on landscape patterns and plant species richness in Mediterranean coastal dunes was assessed and a general simplification was observed in the natural dune spatial pattern with a decline in plant richness where human pressure is important (Malavasi et al. 2018). Assessing the conservation status of coastal dune systems in Tuscany (Italy), Ciccarelli (2014) showed that the general spatial pattern of vegetation there was close to the natural zonation (Acosta et al. 2007; Forey et al. 2008; Miller et al. 2009; Fenu et al. 2013), with a variable sequence of coastal dune plant communities, ranging from the disappearance of the foredune habitats to the presence of the complete sequence. Vegetation of the driftline disappeared in this study's transects where erosion was significant, while embryonic shifting dunes with *Elymus farctus* were well represented in the area. However, embryonic dune habitat showed a decrease in coverage in transects belonging to coastal tracts affected by erosion. Although vegetation of the driftline and embryonic shifting dunes have few taxa with low coverage, they represent an important element that must be considered in conservation programs (Ciccarelli 2014).

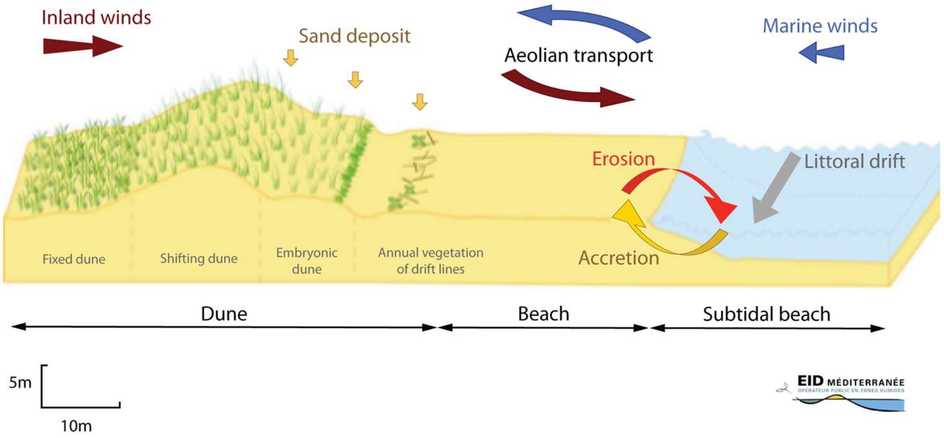

Figure 4.6 | Cross-section of a sandy dune system[3]

Rocky coasts

Rocky intertidal shores play an important role for marine habitat functioning (Sugden et al. 2009). They are a destination for leisure, exploration and relaxation (Hall et al. 2002; Sugden et al. 2009), and are subject to extractive activities for food, bait or ornamental purposes (Murray et al. 1999; Hall et al. 2002). Hence, the benefits provided by the Mediterranean intertidal area, and the associated economic assets, are relevant in the perspective of integrated coastal zone management and under the current climate change scenario. Rocky shores are biologically rich and diverse habitats (Benedetti-Cecchi et al. 2003; Ceccherelli et al. 2005; Schembri et al. 2005; Mangialajo et al. 2008a). Their diversity is multiplied from the local scale up to the basin-wide level by the interplay of biotic and abiotic factors such as interaction among species, exposure, microtopography, island or mainland location and latitude. The presence of ecosystem engineering species also plays a crucial role here, where their disappearance is typically associated with radical changes in the structure of the assemblages. The genus *Cystoseira*, for example, includes low-shore canopy algae responsible for habitat formation for sessile invertebrates requiring high humidity and low light levels, such as sponges and tunicates (Benedetti-Cecchi et al. 2001; Ceccherelli et al. 2005; Mangialajo et al. 2008b). The loss of *Cystoseira* canopies results in disruptions to habitat complexity and species diversity patterns, and in the development of algal turfs associated with low-abundance invertebrate assemblages (Benedetti-Cecchi et al. 2001; Mangialajo et al. 2008b).

Mytilus galloprovincialis is an edible mussel, widespread in intertidal and shallow subtidal areas throughout the Mediterranean Sea, where it is also aquacultured. The loss or reduction of natural *M. galloprovincialis* populations (Sarà et al. 2011) and of another autochthonous element of bivalve fauna, *Mytilaster minimus* (Sarà and de Pirro 2011), has been shown to produce changes in the associated assemblages, enhanced when the contemporary reduction in canopy-forming algae occurs (Maggi et al. 2009). Although not an endangered species, its presence and abundance might be altered by the establishment of non-indigenous species like the mussel *Brachidontes pharaonis* (Sarà et al. 2008).

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[3] http://moving-coast-med.fr/map-pins-nature/vent-et-transport-%C3%A9olien/

Mediterranean vermetid reefs are found where the temperature of surface coastal waters is no lower than 14°C (in winter) (Chemello and Silenzi 2011). A typical vermetid reef is the outcome of complex synergistic building activity by the vermetid mollusc *Dendropoma (Novastoa) petraeum* and the encrusting red algae *Neogoniolithon brassica-florida* (Chemello and Silenzi 2011). Other species, such as the red algae *Lithophyllum* spp., support the process of bioconstruction. Vermetid reefs develop in the lower mesolittoral and upper infralittoral, on rocky coasts only, and precipitate an aragonite shell (Silenzi et al. 2004; Sisma-Ventura et al. 2009). In addition to temperature and the type of substrate, the hydrodynamism of shallow water layers influences the distribution and size of these structures on a small scale, because vermetid platforms are rare along sheltered coasts (Chemello and Silenzi 2011). As for ocean acidification (Section 2.2.9), although *D. petraeum* were able to reproduce and brood at high levels of CO_2, recruitment success was found to be adversely affected (Milazzo et al. 2014).

Experimental work on the Mediterranean subtidal red alga *Peyssonnelia squamaria* shows that this species may benefit from ocean acidification, as its own nitrogen metabolism will be regulated (Yıldız 2018). Among the engineering species, although vermetids are resilient to near-future pCO_2 levels, it is likely that their reefs will not be able to withstand levels of acidification predicted for the end of this century, and the associated community will change as a result (Section 2.2.9).

Mediterranean subtidal rocky ecosystems have not been well studied. An initial current baseline and gradient of ecosystem structure was established for nearshore subtidal rocky reefs on a Mediterranean scale, at 8 to 12 m water depths (Sala et al. 2012). This baseline study showed remarkable variation in the structure of rocky reef ecosystems and suggested that the healthiest shallow rocky reef ecosystems in the Mediterranean have both large fish and algal biomass. Protection level and primary production were the only variables significantly correlated to community biomass structure. Fish biomass was significantly larger in well-enforced no-take marine reserves, but there were no significant differences between multi-use marine protected areas (which allow some fishing) and open access areas at the regional scale (Sala et al. 2012).

Overall, intertidal systems are poorly represented in the socio-economic literature, and there appears to be low awareness of their value among stakeholders. Subsequently, conservation efforts for intertidal communities are minimal.

Coastal wetlands

Mediterranean coastal wetlands include a wide variety of natural habitats such as river deltas, coastal lagoons and salt marshes, intertidal wetlands, and coastal aquifers. Global warming and direct anthropogenic impacts, such as water extraction (more on land and sea use changes and practices in Sections 2.4 and 3.1.5.2), largely affect water budgets in Mediterranean wetlands, thereby increasing wetland salinities and isolation, and decreasing water depths, water quality and hydroperiods (duration of the flooding period) (Mediterranean Wetlands Observatory 2018). These wetland features are key elements that structure waterbird communities for instance (Ramírez et al. 2018). However, the ultimate and net consequences of these dynamic conditions on key species assemblages are largely unknown, although recent work indicates that waterbird communities, for example, tend to adapt to climate conditions (Gaget et al. 2018).

Seagrass meadows

Seagrass meadows in the Mediterranean Sea cover 1.35 to 5 million hectares, between 5 and 17% of the worldwide seagrass habitat. The rate of loss of seagrass is above 1.2% each year and up to 5% in the Mediterranean (Marba and Duarte 2010). The Mediterranean Sea is home to four seagrass genera (*Cymodocea, Halophila, Posidonia and Zostera*) encompassing four indigenous species (*C. nodosa, P. oceanica, Z. marina and Z. noltii*) and one non-indigenous species (*H. stipulacea*) (Sghaier et al. 2011). The largest Mediterranean sublittoral area occupied by seagrasses is dominated by the endemic *P. oceanica* (neptune seagrass). It clonally reproduces in meadows and can spread up to 15 km while being hundreds to thousands of years old. In certain areas its populations are significantly declining due to a combination of human-induced factors (Vohník et al. 2016). *Posidonia oceanica* is the most threatened seagrass habitat: almost half of the surveyed sites in the Mediterranean have suffered net density losses of over 20% in 10 years. These *P. oceanica* losses are directly linked to human activities: eutrophication from nutrient pollution, alteration of coastal sediment balance and physical disturbances from trawling, anchoring of yachts, dredging, and other activities highlighted in Sections 2.3 and 2.4. Other non-human impacts include rising sea temperature (Section 2.2.8) and non-indigenous species (Section 2.5) (Claudet and Fraschetti 2010; Crooks et al. 2011).

The non-indigenous variety of *Caulerpa racemosa*, currently spreading in the Mediterranean Sea, was first discovered in the early 1990s near Tripoli Harbour in Libya (Nizamuddin 1991). It spreads rapidly, and is now found off the coasts of multiple countries around the Mediterranean Sea (Lebanon, Tunisia, Libya, Egypt, Cyprus, Turkey, Greece, Malta, Croatia, Italy, France and Spain), and has reached the Canary Islands in the Atlantic Ocean (Verlaque et al. 2004; Bitar et al. 2017). Recent work has shown that this non-indigenous variety is *C. racemosa* var. *cylindracea*, which was introduced from southwestern Australia (Verlaque et al. 2003). Long-range dispersal of the algae seems to be a result of human activities (e.g., dispersal by anchors, fishing). *C. racemosa* can inhabit a wide range of subtidal substrata (sand, mud, rocks, dead matte of seagrass, from 0 to 50 m depths), and has the potential to expand its range around the entire coastline of the Mediterranean Sea. *C. racemosa* modifies the density and diversity of benthic communities (Capiomont et al. 2005).

Early studies on spatial distribution and expansion of *H. stipulacea* had been focused on the northern Mediterranean - the spreading of *H. stipulacea* along the southern and southeastern Mediterranean coasts was only noticed later (Lipkin 1975; Bitar et al. 2000, 2017; Abboud-Abi Saab et al. 2003; Lakkis and Novel-Lakkis 2007; Verlaque et al. 2015).

Coastal lagoons and deltas

The small range of tides associated with low-speed currents has encouraged the establishment of lagoon or endogenic systems around much of the Mediterranean. Lagoons and deltas provide support for rich biodiversity, including vital habitats for bivalves, crustaceans, fish and birds. They provide a physical refuge from predation and are used as nursery and feeding areas for some endangered species (Franco et al. 2006; Le Pape et al. 2013; Escalas et al. 2015; Isnard et al. 2015). There are more than 100 coastal lagoons in the Mediterranean, holding an important ecological role, and also providing essential goods and services for humans (read how the effects of climate change on coastal ecosystems could affect livelihood, culture and human rights in Section 5.3). Over 621 macrophyte species and 199 fish species are present in Atlantic-Mediterranean lagoons (Pérez-Ruzafa et al. 2011).

Due to their location between land and open sea, coastal lagoons are subject to strong anthropogenic pressures in parallel with climate change. Habitat destruction, pollution, water withdrawal, overexploitation and non-indigenous species are the main causes of their degradation (Newton et al. 2018). These pressures are responsible for major ecosystem alterations such as eutrophication, bacterial contamination, algal blooms (toxic, harmful or not), anoxia and fish killings. Furthermore, additional problems arise from coastal erosion, subsidence and effects related to extreme meteorological events, typical for the Mediterranean (Aliaume et al. 2007).

In the Mediterranean, the largest coastal wetlands are found in delta areas like that of the Po (Italy), Nile (Egypt), Rhône (France) and Ebro (Spain) rivers (Section 3.1.1.3). Like most wetlands, deltas are diverse and ecologically important ecosystems (UNEP-MAP-RAC/SPA 2010). Deltas absorb runoff from both floods (from rivers) and storms (from lakes or the ocean). Deltas also filter water as it slowly makes its way through the delta's distributary network. This can reduce the impact of pollution flowing from upstream. Plants such as lilies and hibiscus grow in deltas, as well as herbs such as wort, which are used in traditional medicines. Many animals are indigenous to the shallow, shifting waters of a delta. Fish, crustaceans such as oysters, birds, insects are also part of a delta's ecosystem (UNEP-MAP-RAC/SPA 2010; Mediterranean Wetlands Observatory 2012). In deltaic areas and low-lying coastal plains climate change, particularly sea-level rise, is already considered as an important issue (Nicholls and Hoozemans 1996), that significantly decreases the return periods of maximum water levels due to storm surges (Section 2.2.8 and Box 3.1). This has been discussed by Sánchez-Arcilla et al. (2008) for the Ebro delta where other phenomena are affecting deltaic behavior such as increases in inundation/flooding, coastal erosion, salinity intrusion, and changes in wave climate (wave height, direction, and storminess).

Salt marshes

Salt marshes cover low-energy, intertidal shorelines worldwide and are among the most abundant and productive coastal ecosystems. Salt-marsh ecosystems provide a wide array of benefits to coastal populations, including shoreline protection, fishery support, water quality improvement, wildlife habitat provision, and carbon sequestration (Hansen and Reiss 2015). These are specialized habitats, characterized by high primary productivity and species diversity, which support a wide variety of native flora and fauna, and also constitute important areas for wintering aquatic birds (Simas et al. 2001). Salt marshes protect lowlands from marine flooding by damping storms and waves and by slowing flows pushing inland (Allen 2000).

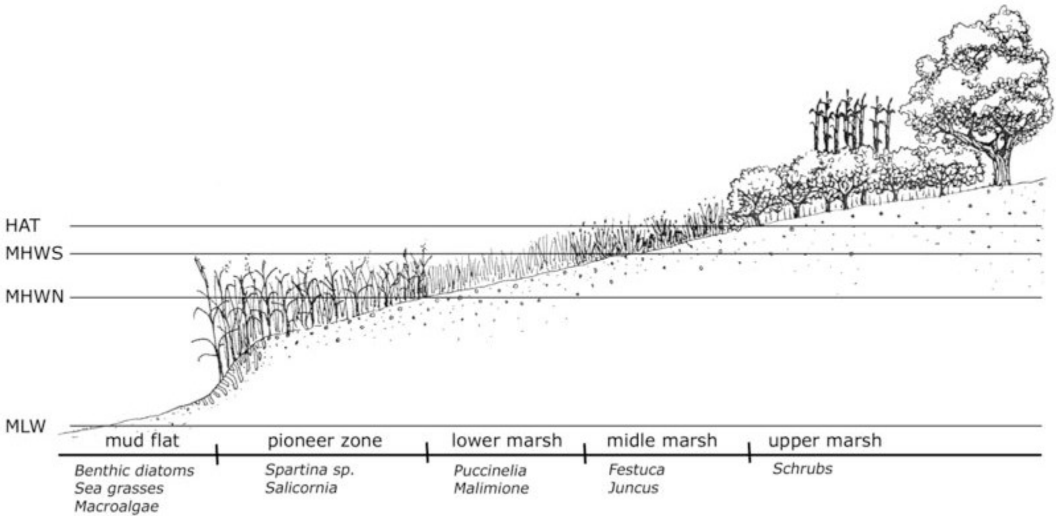

Figure 4.7 | Typical salt marsh zonation (modified from Bertness et al. 2002). Species along the tidal elevation gradient are adapted to the inundation frequency, including extreme flooding and storm events. MLW: Mean Low Water; MHWN: Mean High Water of Neap tides; MHWS: Mean High Water Spring tide; HAT: Highest Astronomical Tide.

In the Mediterranean Basin, coastal salt marshes (Figure 4.7) include various Mediterranean plant communities of the classes *Juncetea maritimi* and *Salicornietea fruticosae* which are under the influence of saline seawater. The vegetation is dominated by perennial and shrubby halophytes growing on the extreme upper shores of low sedimentary coasts, sheltered from the mechanical action of waves. Their habitat is especially diverse in the Iberian Peninsula and in southern Italy (Sicily, Apulia, Calabria) (Molina et al. 2003; Cutini et al. 2010; Sciandrello and Tomaselli 2014). In these parts, the habitat forms a mosaic of tall rushes mixed with shrubby and other herbaceous species, often with succulent stems and/or leaves, forming halophytic shrublands and thickets. In soils with brackish water, beds of reed and other tall helophytes grow. The habitat further includes Mediterranean halo-psammophile meadows (*Plantaginion crassifoliae*), humid halophilous moors with the shrubby stratum dominated by *Artemisia coerulescens*, halo-nitrophilous shrubby seablite thickets of *Suaeda vera* rarely inundated, shrub communities of Limoniastrum sp, and communities in the Dalmatian coastal region, in somewhat drier habitats with less salt, which are not directly affected by waves and tides. On intertidal muds, cord grasses (*Spartinion maritimae*) may grow, but these are relatively rare in the Mediterranean (Molina et al. 2003; Cutini et al. 2010; Sciandrello and Tomaselli 2014). Saltmarshes include a wide diversity not only of vegetation (Ayyad and El-Ghareeh 1982), but also of plankton, crustaceans and fish species. The zooplankton structure and dynamics in permanent and temporary

Mediterranean salt marshes were investigated, and the results found that although temporary and permanent basins show no differences in the most abundant zooplankton species, they differ in their zooplankton diversity, temporal pattern and size structure (Brucet et al. 2005). The same study also found that the presence of a stable fish population in the permanent salt marshes may explain the high values of zooplankton diversity and the low densities of large zooplankton (Brucet et al. 2005).

In an assessment of the threat level of non-indigenous species on salt marshes of the southeastern Iberian Peninsula, the status of its habitats has been categorized as 'particularly critical' as many of them were destroyed in the past, due to their transformation into cropland or by desiccation for fear of malaria (Al Hassan et al. 2016). In the Valencia region (Spain), the coastline supports virtually all farming, much of the region's industrial activity, and shelters large population centers. This, along with huge pressure from tourism (i.e., pressures related to water consumption, Section 3.1.2.3), have highly impacted salt marshes. These ecosystems house a specific flora of halophytes and their high specialization contributes to their vulnerability (Pétillon et al. 2005).

Coastal aquifers

In Mediterranean coastal systems, seawater intrusion is an important consequence of climate change and human action in coastal aquifers (Sherif and Singh 1999). Salinization alters the fundamental physicochemical nature of the soil-water environment, increasing ionic concentrations and altering chemical equilibria and mineral solubility (Herbert et al. 2015). Increased concentrations of solutes, especially sulfate, alter the biogeochemical cycling of major elements including carbon, nitrogen, phosphorus, sulfur, iron, and silica (Herbert et al. 2015), which has negative impacts on photosynthetic pigments and global biomass (Parihar et al. 2015). More about seawater intrusion in coastal aquifers can be found in Section 3.1.2.2. Three realistic scenarios (no change, sea rise of 0.5 m, land side lowered by 0.5 m by water pumping) were considered by Sherif and Singh (1999), who found that the Nile Delta aquifer is vulnerable to climate change and sea level rise. However, salinization tolerance can be found in some soil inhabitants like the spider *Arctosa fulvolineate* and the beetle *Merizodus soledadinus*, which survived salinity levels up to 70‰ (Pereira et al. 2019). Also, some littoral and terrestrial amphipod species can survive salinity levels up to 900 mOsm external concentration (Morritt 1988).

Risks from non-indigenous species

Phytoplankton

The Mediterranean Sea has experienced particularly strong algal bloom events over the past 50 years, mostly near the coast, in bays, lagoons, ports, beaches and estuaries, leading to deterioration in water quality, increasing the mortality of fish and risks to human health due to specific toxins that could be released into the marine environment and bio-accumulated through the marine trophic chain (Aligizaki 2009; Vlamis and Katikou 2015; Griffith and Gobler 2020). Climate-induced changes in water temperature, stratification and other physical properties appear to strongly impact the physiology and behavior of harmful algae bloom species, in terms of occurrence, physiology and toxin production (Section 2.3.4) in Mediterranean coastal areas where already more frequent Harmful Algal Blooms "HABs", and "novel" nuisance species have been recorded (Legrand and Casotti 2009).

The potential impact of climate-induced changes to phytoplankton, and especially HABs, has raised attention in the scientific communities and directed their research in this field, mainly driven by human health concerns due to the potency of some algal toxins that are transferred through the marine food web (Turki et al. 2006; Drira et al. 2008; Mabrouk et al. 2012; Estevez et al. 2019). In monitoring southern Mediterranean countries, more than 64 dinoflagellate species were identified with a remarkable increase in spring and summer (Feki et al. 2016). Dinoflagellate abundance between tidal periods was variable and the highest abundance was detected in the slack period in the Gulf of Gabès (southern coast of Tunisia), suffering from the pressure of high urbanization and industrialization rates, as well as rapidly increasing population growth rates. The dinoflagellates represent a major part of the eukaryotic primary production in marine ecosystems and the ability of several strains to cause shellfish poisoning and/or to form resting cysts has led to considerable attention being paid to the diversity and distribution of planktonic dinoflagellates in relation to environmental factors, hydrodynamism, nutrients and microalgae/biotoxins (Monti et al. 2007).

Ostreopsis (a dinoflagellate) blooms have become common in temperate areas as well, and regularly occur in the Mediterranean Sea in the summer and autumn (Vila et al. 2001; Selina et al. 2014). In these areas, *Ostreopsis* was well-known as its blooms were often associated with harmful effects on the health of both humans and benthic marine organisms (Vila et al. 2001; Aligizaki and Nikolaidis 2006). Additionally, *Ostreopsis* often appeared in association with other toxic or potentially toxic benthic dinoflagellates such as *Prorocentrum* spp, *Amphidinium* spp. and *Coolia monotis* in several Mediterranean areas (Monti et al. 2007; Mabrouk et al. 2012; Selina et al. 2014). The genus *Ostreopsis* includes several species producing various palytoxin-like compounds with harmful effects on humans and marine fauna (Scalco et al. 2012). Species of this genus are regular members of the epiphytic community in tropical seas but their geographic range has shown an apparent expansion towards temperate regions of the Mediterranean Sea.

Jellyfish

Cassiopea andromeda is a non-indigenous jellyfish species that possibly takes advantage of the warming tendency in the Mediterranean Sea. Recently detected in Malta, Sicily and other areas beyond the Eastern Mediterranean Sea, this benthic jellyfish seems to be well adapted to mesotrophic waters near harbors and closed bays in where water has low hydrodynamism (Yokeş et al. 2018). Due to the fact that many coastal areas all over the Mediterranean are no longer oligotrophic, the dispersion of this species may accelerate its path. The huge biomass reached in certain zones and its fast-growing features (Deidun 2018) may be a problem for fisheries, coastal tourism and management.

Other non-indigenous jellyfishes are also increasing their abundance in the warming Mediterranean Sea. The dreaded cubomedusa is typical of tropical seas, such as around the Australian Great Barrier Reef or the Philippines. Cubozoans, or 'box jellyfish', are considered to be the cnidarian group's most dangerous, with an extremely painful sting that changes the lives of dozens of Australians every year. During the summer of 2008, a great abundance of a cubomedusa occurred along the Spanish coast, spotted off the beaches of Denia, Alicante, particularly *Carybdea marsupialis*, a small jellyfish species. It is rare in the Mediterranean Sea, so had never been considered to be a species that would form a major proliferation, yet during the summer of 2008 the Red Cross reported a high number of stinging incidents in this area due, no doubt, to this almost imperceptible, transparent and seemingly harmless jellyfish forming dense swarms in the breakers (Kingsford et al. 2018).

It is possible that *C. marsupialis* appeared because of the changing conditions throughout the water column, but the information about its distribution or what factors influence its life cycle is still limited (Canepa et al. 2017). Sea warming seems to be one of the key factors explaining its acute proliferation, and is likely the reason of the already changing trophic interaction map of the Mediterranean Sea due, in part, to climate change. Water temperature, together with quantity and quality of available food resources, are known as major drivers of gonadal outputs (Harland et al. 1992; Ben-David-Zaslow and Benayahu 1999). In general, jellyfish sexual and asexual reproduction is known to be influenced by warming. Some studies have shown that elevated temperature by itself or in combination with high feeding frequency (due to raised zooplankton prey abundance) increased budding rate and bud size in *Aurelia* polyps populations worldwide (Hočvar et al. 2018).

Fish

The establishment of the lionfish (*Pterois* sp.) and the blowfish (*Lagocephalus* sp.) in Mediterranean waters can be envisaged as a paradigm of how climate change helps the dispersion of tropical species in a warm temperate sea (Section 2.5.1). The lionfish, for example, is a predator that has almost no controlling species (other fishes, sharks, etc.) and is a generalist, living in all shallow and mesophotic zones. The species has high reproductive and dispersal capacities, a massive production of well-protected eggs all year long and a fast spread and high larval survival rates (Betancur et al. 2011). In the Eastern Mediterranean Sea, these non-indigenous species have been introduced from the Red Sea, and have been documented in many areas (Bariche et al. 2013, 2017; Kletou et al. 2016). The blowfish (*Lagocephalus sceleratus*) is another example, with dispersion apparently faster in certain zones (Boustany et al. 2015; Kara et al. 2015). One of the first records indicate fast mobility from the original source (the Suez Canal) (Akyol et al. 2005; Kara et al. 2015), and since its first recorded sighting, it has been detected even in coastal waters in southern Italy (Azzurro et al. 2014). All the non-indigenous vagile fauna would have a very restricted dispersion if the water column temperature conditions were stable, with a clear marked low temperature during the autumn and winter, but changing conditions may be favoring its definitive establishment in Mediterranean waters (Bianchi and Morri 2003). The rapid spread of some of these species will be a serious problem for fisheries and trophic relationships in coastal areas, causing the likely local extinction of some species that may be preys of these generalist fish species (Coro et al. 2018).

Plants

Among numerous threats, the pressure of non-indigenous plants has strong effects in these fragmented and linear ecosystems. *Dittrichia vi*scosa is a perennial, 40-130 cm high plant, very common in the western Mediterranean but also found in its eastern part (Al Hassan et al. 2016). The species shows a remarkable pioneering character, and in recent decades largely expanded its range in Mediterranean countries, possibly due to increased human disturbances (Wacquant 1990; Mateo et al. 2013). Its capability to colonize new habitats and threaten biodiversity has been well-documented (Wacquant 1990) and related to characteristics such as its phenotypic plasticity (Wacquant and Bouab 1983), high stress tolerance (Curadi et al. 2005) and resistance to chemical pollution (Murciego et al. 2007; Fernández et al. 2013), as well as to its allelopathic effects (Omezzine et al. 2011). In the last 50 years, *D. viscosa* has become an invader in the NW Mediterranean region, since it increased its ecological range under disturbance pressure and is colonizing new habitats (Wacquant 1990; Boonne et al. 1992; Wacquant and Picard 1992; Mateo et al. 2013). The species' recent expansion in the Iberian Peninsula has also been correlated to temperature increases due to accelerated global warming (Vesperinas et al. 2001). Although *D. viscosa* cannot directly compete with true halophytes in highly saline environments, it is nevertheless quite stress tolerant and therefore presents a threat to the vegetation located on salt marsh borders, where several endemic and threatened species are found in the area of study conducted by Al Hassan et al. (2016).

Other non-indigenous species

Some non-indigenous eco-engineering species are also favored by sea warming in the Mediterranean Sea (Section 2.5). Sea forests are living three-dimensional structures, similar to terrestrial forests but basically comprised of seaweeds, seagrasses, sponges, cnidarians, bryozoans, ascidians and other sessile organisms in the ocean benthos (Rossi et al. 2017a). These forests are dominated by ecosystem engineering species, organisms which directly or indirectly modulate the availability of resources to other species, causing changes to the physical condition of biotic or abiotic materials (Jones et al. 1994). In the case of the non-indigenous species, we can highlight two different case studies of non-indigenous eco-engineering species that may be already changing the benthic seascape in many areas of the Mediterranean Sea due to the suitable conditions for their expansion. The first case is represented by the seaweed *Caulerpa cylindracea*. This chlorophyte has been identified as one of the most successful bioinvaders (Montefalcone et al. 2015). This species has been much more successful with respect to *Caulerpa taxifolia*, the so-called killer algae. In several areas, it is replacing other algae, phanerogams and sessile animals creating a new seascape in which the biodiversity and biomass are rapidly changing (Alomar et al. 2016).

A different case study is that of myxotrophic scleractinians that are more present toward the northern and western Mediterranean Sea. *Oculina patagonica* has been recorded in many Mediterranean areas but was not present in northern Spanish coastal areas until recently (Leydet et al. 2018). This species seems to be rapidly adapting to new temperature trends, a factor that drives new populations to quickly move north. Originating from the south, this species is now an invader that proliferates, replacing other native species in shallow waters, where sea urchin barrens may be essential to understand their capability to cope with new spaces (Coma et al. 2011). The presence of this species in northern areas seems to be correlated with sea warming, but has clear photobiological limits due to temperature factors that have to be considered (Rodolfo-Metalpa et al. 2014).

4.2.1.2 Past changes

Although human activity is considered as a major driving force affecting the distribution and dynamics of Mediterranean ecosystems, the full consequences of projected climate variability and relative sea-level changes on fragile coastal ecosystems for the next century are still unknown. It is unclear how these waterfront ecosystems, as well as the services they provide, can be sustained, when relative sea-level rise and global warming are expected to exert even greater pressures in the near future (drought, habitat degradation and accelerated shoreline retreat) (Kaniewski et al. 2014). The most suitable archives for such paleoreconstructions are located in coastal wetlands that are highly vulnerable to global warming and the rapid rise in sea level, as they are highly exposed to processes such as flooding, subsidence, sediment scarcity and coastline erosion (Anthony et al. 2014; Wong et al. 2014) (Chapter 2).

Response of coastal ecosystems to past changes in sea level

During the Holocene period, sea level worldwide exhibited significant fluctuations, mostly responding to the advance and retreat of the continental ice sheets (Lambeck et al. 2010). In the Mediterranean, regional sea level has risen continuously for the whole Holocene with a sudden slowdown at ~ 7.5 ka BP and a further deceleration during the last ~4.0 ka BP, after which time observed regional sea-level changes have mainly related to variability in isostatic adjustment (Galili et al. 2005; Vacchi et al. 2016). This mid-Holocene sea level stabilization had a major influence on fluvial sedimentation in coastal regions, and the formation of many coastal wetlands such as flood plains, coastal lagoons and deltas can be traced back to this period (Pavlopoulos et al. 2006; Fouache et al. 2008; Carmona et al. 2016). Although the stabilization of sea level and availability of sediment are important variables in this process, the palaeogeography of the lagoon environment and specific fluvial and marine sedimentary dynamics are also important variables in each case study (Huntley 2001).

Coastal wetland-based archives have an exceptional scientific potential to observe past ecosystem changes and identify key thresholds for particular ecosystems facing sea-level rise. In northeastern Spain, a reconstruction of past ecosystems was presented for the Castelló lagoon using ostracods, diatoms, and pollen and non-pollen palynomorphs as bio-indicators of marine vs freshwater influences (Ejarque et al. 2016). This study pointed out the strong link between the lagoon ecosystem, changing sea-level and the historical anthropogenic pressure. From the late Neolithic to the medieval period, dynamics of the lagoon ecosystem were mainly driven by changing marine influence. From ~1550 and ~150 cal BC maximum marine flooding hampered agropastoral activities. After the Medieval period, societies actively controlled lagoon dynamics and became a major agent of landscape transformation. The removal of littoral woodlands after the 8th century was followed by the expansion of agrarian and industrial activities. The expansion of the milling industry and of agricultural lands led to the channelization of the river Muga into the lagoon after ~1250 cal AD, which caused its transformation into a freshwater lake ecosystem (Ejarque et al. 2016).

In the Corsican back-barrier wetlands, a study of the fossil Coleoptera in two sediment cores supplemented by pollen and geochemical data show that 60% of past wetland beetle fauna at the Grecu pond became locally extinct because of the increase in salinity caused by marine intrusions (Poher et al. 2018). Most of this diversity loss occurred 3700 years ago, when relative sea-level reported in the region rose. Regarding the Cannuta marsh, results of the same study show evolution from a brackish lagoon to a freshwater environment marked by diversification of wetland beetle fauna 1000 years ago, which is possibly due to relative sea-level stability and floodplain progradation (Poher et al. 2018).

In the Eastern Mediterranean, the rapid response of the Alikes Lagoon of Zakynthos Island ecosystem has been attributed from 8540 and 3400 BP to climate-related events such as storms or tsunamis where marine characteristics dominate (Avramidis et al. 2013). Furthermore, the study of Kaniewski et al. (2014) shows that Haifa Bay underwent a landward sea invasion, with a maximum sea penetration 4000 years ago. The main consequences of the sea invasion were a retreat of the coastal forest, a loss of resilience and disappearance of the initial local biogeographic zonation. The forest replacement by a thorny shrub-steppe and then by an open-steppe appeared to follow, rather than cause, failure of tree regeneration. Due to the intrusion of the saline water table in freshwater streams, the freshwater wetland associated with the Na'aman River was deeply impacted after 4000 cal. years, with a fall in hygrophilous-hydrophilous herbs. The subsequent coastal progradation that started at 3400-3300 cal yr BP left an eroded sandy-salty area, colonized by a steppe vegetation that became dominant until the end of the shoreline retreat (2900 cal yr BP). A similar process was observed in the Salt Lake of Larnaca, Cyprus, where a shift from sheltered marine to lagoon environments produced an ecological change with a strong increase in xerophytic vegetation-types colonizing the shores that were no longer washed by seawater (Kaniewski et al. 2013b). The 3.2 kyr BP drought event (Schilman et al. 2001a; Kaniewski et al. 2013a) caused a dramatic demise in wooded ecosystems unrecovered until after 2850 cal yr BP. Due to human-induced modification during the last ~200 years, it retreated by 15 m (Zviely et al. 2009). This new phase of sea-level invasion is well attested by high values of steppe vegetation and a renewed drop in hygrophilous-hydrophilous herbs. This suggests that similar stresses generate analogous biological processes, whatever the period under consideration.

Mediterranean coastal ecosystems, including fauna and flora components, are very sensitive to sea-level changes. The mid- to late Holocene changes in sea-level caused strong biological stresses and major ecological alterations providing a foreshadowing of potential future diversity and community changes along Mediterranean coastal wetland ecosystems and a model to understand the consequences of sea-level rise for the next century (Marcos and Tsimplis 2008). This retrospective approach encompassing historical anthropogenic pressure also clearly indicates that for efficient management of coastal habitats, anthropogenic pressures linked to urban development (Section 5.1.1.3) must be reduced in order to mitigate the predicted effects of global change.

Response of coastal ecosystems to past climate variability

In addition to eustatic sea level curves and coastal geomorphological dynamics, recent research has revealed that the climate variability of the Holocene impacted the Mediterranean coastal ecosystems particularly through storms and floods (Carmona et al. 2016). Storm surges, leading to catastrophic coastal flooding, are amongst the most feared natural hazards due to the high population densities and economic importance of coastal areas. Strong evidence has been provided for enhanced periods of storms that caused coastal flooding over the last 4500 years as a result of solar activity, acting on cycles of around 2200-yr and 230-yr (Kaniewski et al. 2016). These storm surges were characterized by inland intrusion of ostracods and dinoflagellate cysts, while the intrusion of saline water into the freshwater-fed plains greatly affected terrestrial ecosystems leading to land fragmentation by salt encroachment (Kaniewski et al. 2016). An important part of paleoenvironmental Holocene research has focused on identifying phases of high frequency and magnitude of floods related to climate variability (Thorndycraft and Benito 2006; Benito et al. 2008, 2015). In the Western Mediterranean region, periods with more frequent heavy-rain flooding events coincide with transitions to cooler and wetter climates, while flood frequency in northern Africa is linked to drier climate and in the eastern Mediterranean, to wetter conditions (Benito et al. 2015). Some studies hypothesized that these have been driven by internal modes of atmospheric and oceanic changes such as the EA, the NAO, and thermohaline circulation (Degeai et al. 2017). However, the effect of these flood episodes on coastal ecosystems is much less documented and needs to be further addressed in high time-resolution studies.

Based on these findings, short-term climate events, which are driven by high frequency external and internal forcing factors, are superimposed on anthropogenic-driven factors. This complex interaction may either accentuate or attenuate the effect of current and future global warming. For instance, climate models are predicting a decrease in Mediterranean storms in the second half of the 21st century while the study by Kaniewski et al. (2016) suggests that a decrease in solar activity will increase and intensify the risk of frequent flooding in coastal areas.

4.2.2 Projected vulnerabilities and risks

4.2.2.1 Projections and risks based on biological groups

Phytoplankton

Climate change consequences, particularly increasing temperatures (Section 2.2.4), decreasing nutrient replenishment (Section 2.3), and ocean acidification (Section 2.2.9), are expected to cause changes in plankton communities at different levels, from phenology and biomass to community structure. For example, a shift in phytoplankton community, dominance of smaller species (picophytoplankton and nanoflagellates) and a decrease in diatoms, with an expected decrease in the biomass of calcifying organisms such as coccolithophorids are some of the expected outcomes (Dias et al. 2010; The MerMex Group et al. 2011). There are still many uncertainties when it comes to the impact of sea warming and acidification on primary production in the Mediterranean, but it is clear that physico-chemical changes will affect the magnitude, timing and composition of phytoplankton blooms, with associated changes in the seasonal distribution of zooplankton (Moullec et al.) (Section 4.1.2). Ocean acidification combined with warming and deoxygenation, has been shown to cause negative effects on marine animals and to stimulate the production of primary producers, particularly in coastal waters that do not experience stratification or nutrient limitation (Gao et al. 2020). The associated decreased predatory pressure has the potential to further increase primary production. The increased primary production will stimulate the respiration of bacteria and thus intensify the hypoxia and low pH zone (Gao et al. 2020).

To understand how climate variation controls phyto-and zooplankton dynamics and possibly affects artisanal and small-scale fisheries exploiting areas near the coast, Goffart et al. (2017) used a unique long-term (1979-2014) time series obtained from a Mediterranean coastal area unbiased by local anthropogenic pressure in the Bay of Calvi, Corsica. They identified threshold values of physical variables below and above which they strongly impact nutrient availability, phyto- and zooplankton bloom characteristics and seasonality succession of plankton functional groups, stressing the importance of winter conditions in determining the state of Mediterranean pelagic ecosystems, and highlighting that the thresholds obtained from their long-term time series provide key information for improving model scenarios of the impact of climate change on Mediterranean ecosystems.

The functional traits and geographic distribution of 106 copepod species were used to estimate the zooplankton functional diversity of Mediterranean surface assemblages for the 1965–1994 and 2069–2098 periods (Benedetti et al. 2018). Multiple environmental niche models were trained at the global scale to project species habitat suitability in the Mediterranean Sea and assess their sensitivity to climate change predicted by several scenarios. A relatively low decrease in species richness is predicted for 97% of the Mediterranean Basin, with higher losses in the eastern regions (Benedetti et al. 2018). The results of the same study show that climate change is not expected to alter copepod functional traits distribution in the Mediterranean Sea, as the most and the least sensitive species are functionally redundant. Such redundancy should buffer the loss of ecosystem functions in Mediterranean zooplankton assemblages induced by climate change. Since the most negatively impacted species are affiliated with temperate regimes and share Atlantic biogeographic origins, the results of Benedetti et al. (2018) are in line with the hypothesis of increasingly more tropical Mediterranean communities (Section 2.6.2.3).

Fish

In the Mediterranean Sea, a reduction in primary production linked to an increase in sea surface temperature (see previous sub-sections in this chapter and Section 3.2.2.2) could have negative impacts on fisheries catch and could exacerbate current trends of overfishing. Projected changes in primary and secondary productions suggest that trophic mismatches between fish pre-recruits and their prey could increase in the future, with negative consequences for recruitment success, sustainable fisheries and conservation of biodiversity (Lejeusne et al. 2010; Stergiou et al. 2016). Also, jellyfish outbreaks (e.g., *P. noctiluca*) may become more frequent in the Mediterranean Basin (see previous sub-sections) and may extend over a longer period of the year than previously, causing alteration of the pelagic food web and thereby reducing fishery production (Licandro et al. 2010) .

Using Bioclimatic Envelope Models (BEMs), the potential future climatic niches of 288 coastal Mediterranean fish species were projected based on a global warming scenario, then the species-level projections were geographically aggregated to analyze the projected changes in species richness and composition (Albouy et al. 2012). The results show that projected changes in assemblage composition are caused by different processes (species replacement vs. nestedness) in several areas of the Mediterranean Sea, and that the coastal fish fauna in several regions of the Mediterranean Sea could experience a ‘cul-de-sac’ effect if exposed to climate warming (Albouy et al. 2012) (Section 4.1.2.2.).

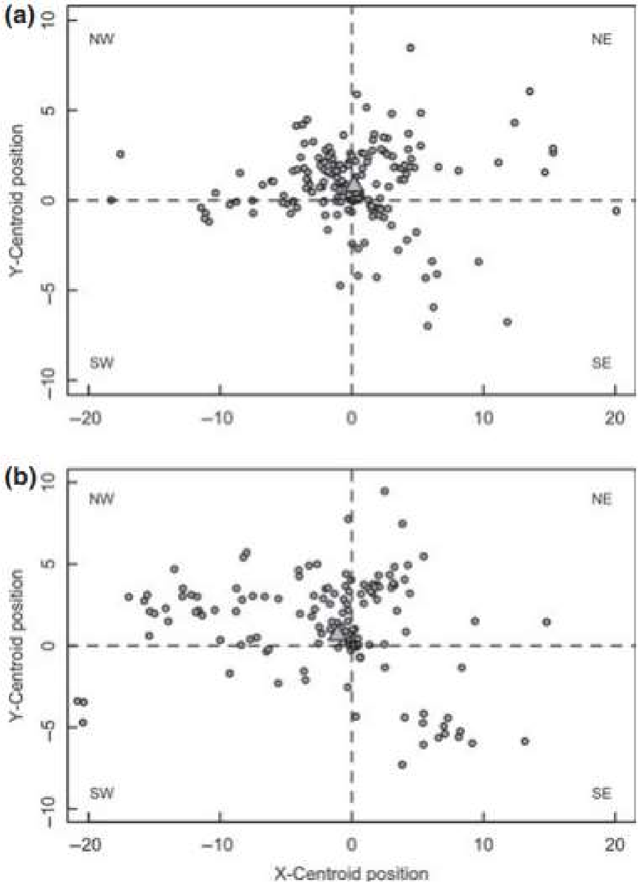

Figure 4.8 | Expected geographical species shifts for the 288 coastal Mediterranean fish species for both periods (a: 2040–2059; b: 2080–2099). Shifts were calculated from geographic range centroids and are expressed in degrees (NW: North West, NE: North East, SE: South East, SW: South West). Triangles represent the overall displacement of the fish assemblage calculated as the average centroid geographical shift (Albouy et al. 2012).

Fish species ranges are expected to move northwards and eastwards, and most of the Gulf of Lion as well as the Adriatic and Aegean Sub-basins are projected to experience a net decrease in species richness resulting from a loss of thermal niches for numerous fish that are not balanced by the arrival of other species from the south by the end of the 21st century (Fig. 4.8). In a best-case scenario, remaining or arriving species in local assemblages after global change impact would share combinations of functional traits with lost species, thereby maintaining ecosystem functioning, while in the worst-case scenario, lost species would have functional traits distinct from those of remaining or arriving species, hence the functions they support would go extinct and would imperil ecosystem functioning. Overall, the joint exploration of changes in species richness and composition coupled with the distinction between species replacement and nestedness bears important information for understanding the nature of climate change impacts on biodiversity (Albouy et al. 2012).

Lionfish (*Pterois* sp.) and blowfish (*Lagocephalus* sp.) in Mediterranean waters reflect how climate change helps the dispersion of tropical species in a warm temperate sea (Section 4.2.1.1). The question is when these species will migrate to the Western Mediterranean Sea. Johnston and Purkis (2014) argue that this is unlikely to happen. The connectivity between different areas is not the same as in the Caribbean for example, and the shifts in temperature need to be much more dramatic to make this invasion possible during the coming decades. However, the generalist behavior of this species may be one of the essential points to understanding its future success if a shift of 1 to 1.5 °C occurs in surface waters in some areas. This alien species from warm tropical waters may present a problem for other species because of the sea warming tendency (Section 2.2.8). It is not only a problem of the alien species per se, but a problem of shift from warm temperate to tropical trophic chain structures. The rapid spread of some of these species will be a serious problem for fisheries and trophic relationships in coastal areas, as the local extinction of some species that may be preys of these generalist fishes is very likely (Coro et al. 2018).

All the above-mentioned projections highlight the pressures that could increase the risk to fish and their habitats, namely for commercially valuable euryhaline coastal fish species (i.e., sole, seabass, seabream, mullet, eel) in coastal nursery sites such as lagoons, estuaries, and deltas. These pressures vary from rising temperatures that could exacerbate the occurrence of HABs (i.e., ciguatoxins, produced by dinoflagellates) and thus the distribution of biotoxins and pathogens (i.e., *Vibrio* bacteria) (Lloret et al. 2016), to the depletion of oxygen that may cause suffocation, which kills fish, to plastic pollution (Barange et al. 2018). A likely decrease in connectivity between neighboring ecosystems within the Mediterranean is expected because of a decrease in the size of spawning areas and an increase in larval retention on smaller areas of the continental shelf (Barange et al. 2018). In addition to warming and ocean acidification, changes to fisheries' structures will contribute to the disappearance and modification of fragile and long-lived species that create biogenic structures or seagrass meadows, which provide important ecosystem services as well (Jordà et al. 2012).

Seaweed

Other non-indigenous species are also favored by sea warming in the Mediterranean Sea. For the non-indigenous seaweed *Caulerpa cylindracea* (section 4.2.1.1), the effects on the quality and quantity of available seston may be positive, also changing biogeochemical cycles and benthic-pelagic coupling relationships (Rizzo et al. 2017). Warming and acidification may be a perfect match for these fleshy algae (Comeau and Cornwall 2017), promoting its proliferation, in part, by the resistance to sedimentation processes, increased in many areas due to direct human impact (Alomar et al. 2016). Several algae species' dispersal may be thus enhanced not only by direct climate change effects but also by direct human impacts in coastal areas.

Corals

As for myxotrophic scleractinians, such as *Oculina patagonica*, projected global warming is likely to cause a gradual contraction in their distribution zones, where temperatures are too high due to their temperature constraints (Rodolfo-Metalpa et al. 2014). Higher transparency of water and rising temperatures in surface waters may also be the key to understanding its successful proliferation. Myxotrophy needs a photosynthetic component that is not present in Mediterranean waters as it is in tropical waters: light (water transparency) and temperature, combined, are the key to understanding high photosynthetic performance (Schubert et al. 2017). Some of these species from tropical or subtropical waters, may be favored by the new conditions of the water column in the Mediterranean Sea, but not all of them. It is possible that Indo-Pacific species (Lessepsian species), adapted to higher temperatures, have more potential of spreading over Mediterranean shallow areas over the coming decades.

The reproductive features and the larval release strategy of octocorals species make them highly sensitive to global climate change (especially in shallow areas, above 40 m depths). Recent models highlight that water stratification in the Mediterranean Sea may last for longer periods and warmer waters may stress non-mobile organisms (Galli et al. 2017). Whether the food availability for benthic suspension feeders would be affected by elevated temperatures is not clear, but it has been shown that under anomalous warming episodes in shallow water adults of *P. clavate, E. singularis* and *C. rubrum* suffer from partial or total tissue loss (Garrabou et al. 2001; Linares et al. 2005, 2008; Rossi and Tsounis 2007). Also, the new recruit mortality in shallow populations is very high when compared to deeper populations (Bramanti et al. 2005; Linares et al. 2008; Coma et al. 2009). Furthermore, mother care (i.e. the energy invested by mother gorgonian colonies to the offspring) will be crucial to understanding the potential survival in a warmer and less productive ocean (Viladrich et al. 2016, 2017). Within the context of global change, there is a risk that the period of trophic crisis might be significantly prolonged to the point that the capacity of energy reserves in lecitotrophic larvae would not last until the arrival of favorable feeding conditions in early autumn. This situation could be even worse if the spawning of these species is triggered earlier by the increase in temperature. Asexual reproduction may enable some individuals to survive catastrophic mortality events such as warming episodes and then expand following the disturbance (Lasker and Coffroth 1999). However, chronic stress that reduces recruitment will have less obvious effects on these clonal taxa and may be the key to understanding future composition of benthic communities. Climate change could lead to partial recruitment failure in the affected species, with major changes in the population structure and dynamics, and a drastic change in ecosystem functioning. These combined factors may be crucial to understanding how seascapes will change in shallow Mediterranean benthic communities.

Non-indigenous species in the Mediterranean Sea may be invasive or simply immigrant species (Section 2.5). The new suitable conditions are key to understanding the transition observed in coastal and offshore areas. Higher temperatures that may be bad for native species (adapted to clear seasonal trends and certain limits of temperature and light), may be positive for the incoming species that are stressed by the same rising temperature phenomenon in their native areas.

4.2.2.2 Projections and risks based on key natural habitats

Sandy beaches/dunes

The impacts from reshaped coastlines as a result of sea level rise and changes in wave climate were assessed via regional climate models, indicating that beaches of the Balearic Islands (western Mediterranean) would suffer a coastal retreat of 7 to 50 m, equivalent to half of the present-day aerial beach surface, under the RCP4.5 and RCP8.5 climate scenarios (Enríquez et al. 2017). Also, beach erosion due to sea level rise in the Aegean archipelago (eastern Mediterranean) was evaluated: under a mean sea level rise of 0.5 m (RCP4.5), a storm-induced sea level rise of 0.6 m is projected to result in complete erosion of 31 to 88% of all beaches (29 to 87% of beaches are currently fronting coastal infrastructure and assets), at least temporarily (Monioudi et al. 2017). The projections of the same study suggest a very considerable risk, which will require significant effort, financial resources and policies/regulation in order to protect/maintain the critical economic resources of the Aegean archipelago (Monioudi et al. 2017). Biodiversity loss will be the outcome of the negative pressures driven by climate change consequences, which would hamper beach ecosystem resilience (Scapini et al. 2019). The specificity of sandy beaches as narrow ecotones between sea and land may be lost under climate change pressure, adversely affecting fine-tuned macrofaunal adaptations and therefore ecosystem functioning

(Scapini et al. 2019). In comparing two coastal plant communities, one in Montenegro and another in Albania, it is demonstrated that the less disturbed beach had zonation very similar to potential vegetation, while plant communities of the touristic beach were fragmented or even substituted by replacement communities (Šilc et al. 2016).

The way habitat distribution will be altered under the effects of two climate change scenarios were analyzed, and the efficiency of the current Italian network of protected areas in the future after distribution shifts was evaluated in Prisco et al. (2013). According to this latter study the range of habitats is currently sufficiently covered by protected areas, achieving the conservation target. However, according to their predictions, protection levels for mobile and fixed dune habitats is predicted to drop drastically under climate change (Prisco et al. 2013).

After combining a digital terrain model with 5 years of nest survey data describing location and clutch depth, Varela et al. (2019) identified (a) regions with the highest nest densities for the loggerhead (*Caretta caretta*) and green turtle (*Chelonia mydas*), (b) nest elevation by species and beach, and (c) the estimated proportion of nests inundated under each sea-level rise scenario. On average, green turtles nested at higher elevations than loggerheads. However, because green turtles dig deeper nests than loggerheads, these were at similar risk of inundation. For a sea-level rise of 1.2 m, a loss of 67.3% for loggerhead turtle nests and 59.1% for green turtle nests were estimated (Varela et al. 2019). Existing natural and artificial barriers may affect the ability of these nesting habitats to remain suitable for nesting through beach migration.

Rocky shores

The prolonged desiccation events on the southeastern rocky shores of the Mediterranean were characterized, and their potential ecological impacts on the unique intertidal Mediterranean Sea ecosystem of vermetid reefs were examined (from 2012 to 2014) by Zamir et al. (2018). This study shows that desiccation stress has already increased on southeastern Mediterranean vermetid reef ecological communities, and if this trend continues, further increases in aerial exposure and desiccation stress could be expected, which could have long-term impacts on this fragile ecosystem (Zamir et al. 2018). For the vermitid *Dendropoma petraeum*, long-term exposure to acidified conditions predicted for the year 2100 and beyond caused shell dissolution and a significant increase in shell Mg content. Unless CO_2 emissions are reduced and conservation measures taken, these reefs are in danger of extinction within this century (Zamir et al. 2018), with significant ecological and socio-economic ramifications for coastal systems (Milazzo et al. 2014).

The narrow range of the intertidal in the Mediterranean has particular implications for its resilience to climate change and sea level rise. For example, in the Mediterranean, the potential harsh effects of tidal aerial exposure on the ecological responses of intertidal organisms is, in fact, usually buffered by wave splashing (Sarà et al. 2011). This might help limit the otherwise detrimental impacts of increasing aerial temperature and dryness on organisms. However, the limited amplitude of the Mediterranean intertidal area implies a very small optimal range of environmental features. As such, some species may be able to adapt and migrate as sea levels rise, but others will not. This knowledge is not definitive as historical exploitation of the Mediterranean Sea and the absence of rigorous baselines makes it difficult to evaluate the current health of these ecosystems and the efficacy of conservation actions at the ecosystem level (Sala et al. 2012).

Coastal wetlands

The extent of wetland salinization and thus its effect on Mediterranean wetlands are still poorly known. Typically, increased salt and sulfide concentrations induce physiological stress in wetland biota and ultimately can result in significant shifts in wetland communities and their associated ecosystem functions (Herbert et al. 2015). In a large-scale outdoor mesocosm experiment, the effects of salinity on successional patterns, diversity, and relative abundance of Camargue (southern France) temporary pool crustaceans were studied (Waterkeyn et al. 2010). Salinity significantly altered crustacean communities hatching from the resting egg bank through a number of direct and indirect effects. Salinity had a significant negative effect on the establishment of large branchiopods and copepods. Both the diversity and density of cladocerans, especially chydorids, were positively related to salinity, possibly due to the absence of biotic interactions with large branchiopods at the highest salinity values (Waterkeyn et al. 2010). In the same study, the authors hypothesize that the salinity-mediated presence of the large branchiopod keystone group can shift the whole wetland regime from a zooplankton-rich clear-water state to a zooplankton-poor turbid state. Crustacean succession was significantly altered by salinity, by slowed development rates, population growth or maturation rates of some species. This suggests that in addition to salinity changes, any alteration of wetland hydroperiod (e.g., through aridification or poor water management) could have a synergistic effect on community structure and the diversity of invertebrate communities, including some keystone species.

Based on Multi-Criteria Decision Analysis techniques, it is documented that wetlands and terrestrial ecosystems have the highest relative risk scores in the Tunisian coastal zone of the Gulf of Gabes (Rizzi et al. 2016). A combination was made for regular sampling of waterbird presence through one annual cycle with in-situ data on relevant environmental predictors of waterbird distribution to model habitat selection for 69 species in a typical Mediterranean wetland network in southwestern Spain (Ramírez et al. 2018). Species associations with environmental features were subsequently used to predict changes in habitat suitability for each species under three climate change scenarios (encompassing changes in environmental predictors that ranged from 10% to 50% change as predicted by regional climate models). Waterbirds distributed themselves unevenly throughout environmental gradients and water salinity was the most important gradient structuring the distribution of the community. Environmental suitability for the guilds of diving birds and vegetation gleaners will decline in future climate scenarios, while many small wading birds will benefit from changing conditions. Resident species and those that breed in this wetland network will also be more negatively impacted than those using this area for wintering or stopover (Ramírez et al. 2018).

Seagrass meadows

Considering the rapid loss of seagrass habitat in the Mediterranean Basin and its capacity to capture carbon, preventing seagrass degradation by developing blue carbon projects is a major opportunity for financing seagrass protection. The avoided degradation of *Posidonia oceanica*, for instance, could provide an extra source of CO_2 capture of 4 tons per hectare per year (Sifleet et al. 2011). More importantly, avoiding their destruction would also prevent the washing away of carbon stored in the sediments under the seagrass beds and thus avoid the release of more than 500 tons of CO_2 per hectare, stored over millennia (Chefaoui et al. 2018).Temperate seagrass ecosystems as the thermal regime of the Mediterranean Sea, are sensitive to ocean warming and will exceed the upper thermal limit of the endemic *P. oceanica* in some areas (Marba and Duarte 2010; Jordà et al. 2012).

Using *Cymodocea nodosa* as a model species, Ontoria et al. (2019) assessed the joint effects of warming (at 20°C, 30°C and 35°C) with two potential outcomes of eutrophication. They found that in addition to the possibility of the persistence of *C. nodosa* being directly jeopardized by temperature increase, the joint effects of warming and eutrophication may further curtail its survival (see projected impacts on seagrasses in Section 4.1.2.1, whereas the drivers “Eutrophication” and “Warming” are detailed in Sections 2.2.7 and 2.3.2).

Coastal lagoons

Coastal lagoons are sentinel systems that are highly vulnerable to potential impacts associated with climate change, particularly, as these systems have a key role in regulating the fluxes of water, nutrients and organisms between land, rivers and the ocean (Newton et al. 2018).

The effects of seawater acidification were assessed on a number of biological responses for one mussel and two clam species, including growth and calcification, at two locations, namely a coastal lagoon in southern Portugal and in the Northern Adriatic Sea (Range et al. 2014). In this study, the CO_2

perturbation experiments produce contrasting responses depending on the species and location. Whereas the effects of acidification on growth and calcification in water of the Adriatic Sea were significant, in the coastal lagoon, these effects were much less evident, probably buffered by a high carbonate content in the water (Range et al. 2014). The same study reveals major variations in macroinvertebrate response to the imposed changes (in temperature and pH, respectively), underpinning the need for species-specific and location-specific adaptation measures. A reflection on threats to integrated management of the Thau coastal lagoon (France) due to climate change and the multi-scalar water scarcity adaptation strategy underlines that although water uses are currently secured thanks to the regional transfer of water, they are not coherent with local water management and create new vulnerabilities in the context of climate change (La Jeunesse et al. 2016). Climate change scenarios predict intensified terrestrial storm runoff, providing coastal ecosystems with large nutrient pulses and increased turbidity, with unknown consequences for the phytoplankton community. In the same lagoon (Thau), a 12-day mesocosm experiment shows that pulsed terrestrial runoff can cause rapid, low quality (high carbon: nutrient) diatom blooms (Deininger et al. 2016). However, bloom duration may be short and reduced in magnitude by fish. Thus, climate change may shift shallow coastal ecosystems towards famine or feast dynamics.

Covering most of the bottom of the Mar Menor lagoon (southeastern Spain), *Caulerpa prolifera* has probably increased the resistance of the lagoon to eutrophication processes through the high uptake of nutrients from the water column and their retention in the sediments, avoiding high phytoplankton densities (Lloret et al. 2008). Nevertheless, if climate change predictions prove true, the current status of the lagoon is likely to collapse, since future environmental conditions could make *C. prolifera* unable to reach values of net photosynthesis greater than zero, and eutrophication processes are expected to appear (Lloret et al. 2008).

Deltas

A comprehensive overview of the status and sustainability of the Ebro, Rhône, and Po Deltas and Venice Lagoon has been published by Day et al. (2019), showing that all of these systems have been strongly modified by human activities. However, each system has a unique combination of impacts that informs management and restoration approaches (see the interaction among drivers in Section 2.6).

As an example, the Ebro Delta is a diverse area in terms of wetland habitat types and has high ecological (e.g., it is the second most important ‘Special Protection Area’ for birds in Spain) and economic value (e.g., third largest producer of rice in Europe) (Fatorić and Chelleri 2012). In the last 150 years, the Ebro delta has been largely transformed into rice fields, which now cover 70% of the total area (Cardoch et al. 2002) and have both direct and indirect effects on the ecology of the area, such as salt infiltration in ground water. The main impact is the destruction of natural habitats, but even the remaining deltaic ecosystems have been affected by rice production, through alteration of the natural hydrological cycle as a result of freshwater inputs during the rice growing season (April to September). In addition, large amounts of nutrients and pesticides are delivered for the fertilization and care of the rice paddies (Forès 1992). The planned construction of 49 new reservoirs mainly for irrigation purposes and withdrawal of water upstream from the delta are forecasted to have drastic ecological consequences such as reductions in sediment and freshwater inputs into the delta and detrimental side effects on deltaic ecosystems fauna and flora (Prat and Ibáñez 1995). Overall, because of its morphology, relative sea-level rise will become the most important climate-induced potential hazard for the Ebro delta (Sánchez-Arcilla et al. 2008).

Sea level rise may severely threaten many key coastal ecosystems such as the Nile delta and may cause the loss of important habitats such as the loggerhead (*Caretta caretta*) nesting beaches (UNEP/MAP-RAC/SPA 2009). Projection of averaged sea-level rise trends by El Sayed Frihy et al. (2010) indicates that the coastal plain of the Nile Delta and Alexandria is vulnerable to accelerated sea-level rise but not at the same level due to wide variability of the land topography, which includes low-lying areas, high-elevated coastal ridges and sand dunes, accretionary beaches, and artificially protective structures. Similarly, based on Earth System model simulations, the sea-level variation along the Egyptian coasts is significantly affected by other factors such as sea-level variation West of the Gibraltar Strait, steric sea level, and sea-surface temperature (Shaltout et al. 2015).

Coastal aquifers

Several recent studies have considered the possible impacts of climate change and seawater level rise on seawater intrusion in coastal aquifers (Sefelnasr and Sherif 2014). All have revealed the severity of the problem and the significance of the landward movement of the dispersion zone under seawater level rise. Most of the studies did not consider the possible effects of seawater rise on the inland movement of the shoreline and the associated changes in the boundary conditions at the seaside and the domain geometry. Such effects become more evident in flat, lowland, coastal alluvial plains where large areas might be submerged with seawater under a relatively small increase in seawater level. None of the studies combined the effect of increased groundwater pumping, due to the possible decline in precipitation and shortage in surface water resources, with the expected landward shift of the shoreline. Using a two-dimensional horizontal model, the study of Sefelnasr and Sherif (2014) investigated the possible effects of seawater level rise in the Mediterranean Sea on the seawater intrusion problem in the Nile Delta Aquifer. They concluded that large areas in the coastal zone of the Nile Delta will be submerged by seawater, and the coastline will shift landward by several kilometers on the eastern and western sides of the Delta (Figure 4.9). By using an equivalent porous continuous medium to represent a karstic Apulian aquifer (southern Italy), an evident piezometric drop was confirmed for the past period (until 1999) and a likely similar dramatic drop in the future was projected (Romanazzi et al. 2015). All phenomena considered in this study's models (e.g., sea level and sea salinity) showed non-negligible effects on coastal groundwater (Romanazzi et al. 2015).

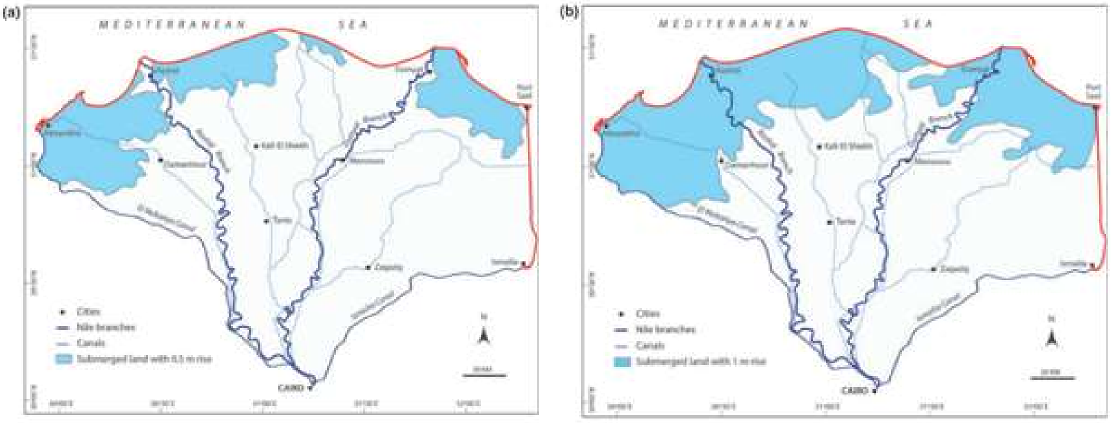

Figure 4.9 | Nile Delta, Egypt, (a) Submerged land in the coastal zone under 0.5 m seawater rise. (b) Submerged land in the coastal zone under 1.0 m sea-level rise (Sefelnasr and Sherif 2014). Red line indicates the border of the basin.

The effects of salinization on coastal aquifers' biogeochemistry typically include decreased inorganic nitrogen removal (with implications for water quality and climate regulation), decreased carbon storage (with implications for climate regulation and wetland accretion), and increased generation of toxic sulfides (with implications for nutrient cycling and the health/functioning of wetland biota) (Herbert et al. 2015). In agriculture, studies on the salinization effects on soil organisms are scarce, but negative effects of saline conditions on survival and reproduction of soil invertebrate species (Owojori et al. 2008, 2014) or on avoidance behavior of earthworms (Bencherif et al. 2015) have been reported. Deleterious effects of soil salinization on diverse life stages of agriculture plants have also been described (Wichern et al. 2006), including decreased and/or delayed germination and/or effects on seedling physiognomic state, deficient growth, as well as a decrease in photosynthetic pigments, and global biomass (Parihar et al. 2015). However, some littoral and terrestrial amphipod species can survive salinity levels of up to 900 mOsm external concentration (Morritt 1988), as along with other spiders mentioned earlier (Pereira et al. 2019).

4.2.2.3 Vulnerabilities

Coastal urbanization

The Mediterranean bioregion is currently suffering severe disturbance due to intensive urbanization and climate change effects (Adloff et al. 2015)[4]. The situation is expected to worsen as land availability decreases (due to the global warming and infrastructure impairment), while demographic growth and migration flows are likely to pursue (Burak and Margat 2016) (read more on land and sea use changes in Section 2.4, water management and infrastructure in Box 3.1, and the vulnerability of coastal energy systems to climate extremes in Section 3.3.2.3). In the future, hydroclimatic hazards, probably more frequent and intense, will have adverse impacts on ecological balances and human health and well-being, particularly in coastal Mediterranean cities where almost one-third of the population lives (Hallegatte et al. 2009; Magnan et al. 2009; Adloff et al. 2015; Im et al. 2018). However, Mediterranean coastal cities seem to lack a long-term vision (i.e., establishing smart cities, green cities, etc.) for planning future urban development and valuable policies and social-economic resources for establishing participative governance (Mazurek 2018) (Section 5.1.3.1).

Social-economic contexts and urban growth rates, trends and phases are quite variable from North to East and South and even across each country (Im et al. 2018). While the size of these urban settlements varies from North to South, most have a historic urban center developed around a harbor near the sea, which makes the different biological species living in or near these areas highly vulnerable not only to human stressors, but also to global phenomena like climate change (Chapter 2).

The survey of juvenile fish populations across various infrastructures and natural sites along a 100 km shoreline of the French Mediterranean coast demonstrated that anthropogenic structures can play an important role as potential juvenile fish habitats, particularly in harbors where highly variable densities were found, with densities on ripraps or jetties that were equivalent to those of natural sites (Mercader et al. 2018). This is the case of the herbivorous fish *Siganus rivulatus* in Lebanon-Eastern Mediterranean where it settled in protected shallow areas offering hard substrates and algal communities such as muddy harbors (Bariche et al. 2004).

One of the most frequently documented negative impacts of the high density of harbors and boats, are the collisions and disturbance of large mammals (dolphins, whales, sea turtles). In particular, the destructive impact of fishing practices on dolphin populations has reached international news headlines (see Section 4.1.2.2 for more information about ship collision effects on cetaceans). Another threat for marine mammals in harbor areas is marine dredging with high concern and impact on the marine life of cetaceans, pinnipeds, and sirenians with effects largely unknown (Todd et al. 2015). The leisure activities, habitat degradation, noise, contaminant remobilization, suspended sediments, and sedimentation may impact marine mammals indirectly through changes to prey. Dredging has the potential to impact marine mammals with specific species and location effects, depending on the type of dredging equipment. In harbor areas, marine mammals continue to be impacted by many anthropogenic activities and almost all marine mammal species have been reported to face at least one threat in the Mediterranean (Laran et al. 2017; Avila et al. 2018).

Reduced primary production (Section 4.1.2.2) and the possible increase of local blooms of toxic algae in some urbanized coastal areas could have many repercussions on marine ecosystem services such as tourism and fisheries, and ultimately on human health (read how the effects of climate change on coastal ecosystems could affect livelihood, culture and human rights in Chapter 5.3). The impact of eutrophication is largely observed in many Mediterranean systems, namely in harbors. For example, in Punic harbors of Carthage, the oldest and most well-preserved in the Mediterranean Basin (Gulf of

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[4] http://www.medqsr.org/

Tunis, South Mediterranean Sea), the harmful blooms of *Dinophysis sacculus, D. acuminata*, *Alexandrium* spp., *Gymnodinium aureolum*, *Gymnodinium impudicum*, *Akashiwo sanguinea*, *Scrippsiella* spp. and *Prorocentrum gracile* were identified in correlation with water temperature and orthophosphate concentrations (Aissaoui et al. 2014) (Section 2.3.3).

The Mediterranean Basin is particularly exposed to biological invasions through shipping from maritime traffic and the high number of harbors constitute large areas for the extension of several non-indigenous species (Izquierdo-Muñoz et al. 2009). The species *Pseudonereis anomala* (Gravier 1900) (Polychaeta, Nereididae) first recorded in Alexandria (Egypt) by (Fauvel 1937), was recorded in several harbor areas in the Mediterranean within the period (2003 and 2005) (Kambouroglou and Nicolaidou 2006), indicating shipping transfer of benthic species (read more on biological pollutants in Section 2.3.4, on non-indigenous species in Section 2.5, and on future risks associated with non-indigenous species in Section 6.12).

In harbor systems, where macrofauna is scarce and difficult to sample, the study of meiofaunal assemblages is proposed as the most suitable instrument for monitoring purposes since, ports, ranging from large commercial harbors to small tourist marinas, are the main link between anthropized and natural coastal ecosystems, and should be taken as primary sources of coastal disturbances (Sedano Vera et al. 2014). Other Mediterranean species associated with marine fouling harbors have been described by Khedhri et al. (2016). The brachyuran decapod is associated with marine fouling in Egyptian Mediterranean harbors and nine species of 9 genera affiliated with 5 families have been recorded so far.

The spatial and temporal changes in climate attractiveness in the Mediterranean could have major impacts on the sustainability of tourism development as suggested by Amelung and Viner (2006), who used a Tourism Climate Index based on future climate change scenarios for the Mediterranean region. This intense tourism activity is harming the Mediterranean shores. Based on the type of garbage on 13 Mediterranean beaches, there are indications that most Mediterranean coastal litter is land-based (Gabrielides et al. 1991). In fact, based on beach cleanups organized over the summers of 2016 and 2017 in Cyprus, Loizidou et al. (2018) suggest that although these initiatives are quite successful at collecting large pieces of marine litter, small pieces of litter (such as cigarette butts and small pieces of plastic items related to recreational activities) remain, accumulating or buried over time, with some items becoming a nuisance to beach goers and a potential source of marine litter. This issue is already influencing coastal organisms such as loggerhead sea turtles (*Caretta caretta*) where, according to a survey by Tomas et al. (2002), the most frequent type of debris in their gastrointestinal tract is plastics (75.9%). Furthermore, the environmental impacts of sunscreen chemicals are likely to be exacerbated in the Mediterranean waters due to the massive influx of tourists and its densely populated coasts, the basin's limited exchanges with the ocean, the high residence time of surface waters, and its oligotrophic waters, which raises significant concerns about its toxicity on marine biota and its bioaccumulation in the marine trophic chain (Tovar-Sánchez et al. 2019) (Section 2.3).

Coastal cities with a sandy ground are often exposed to massive sand extraction. An investigation on the short-term effects of sand extraction on macrozoobenthic communities before and after beach dredging along the Emilia-Romagna coast (northern Adriatic Sea) showed no significant settlement of opportunistic species (Simonini et al. 2005). The same study suggested that the limited impact of sand extraction operations on the physical characteristics of sediment and hydrological-sedimentary characteristics in the relict sand area should aid its rapid recovery and the restoration of the original community in a short period of time (2–4 years after dredging). The same pattern was documented in the coastal ecosystems of the Bay of Blanes on the Catalan coast (Sardá et al. 2000) where recolonization in these dredged habitats was fast, and no changes in seasonal trends were detected after dredging. However, this latter study documented that the filter-feeder *Callista chione* and the carnivorous polychaetes *Protodorvillea kefersteini* and *Glycera* spp., were still significantly reduced after two years, suggesting that a longer period is needed to restructure dredged bottoms to their initial situation (Sardá et al. 2000).

Sea level rise

Similar to the impacts of sea level rise elsewhere (Bernstein et al. 2019; Mullin et al. 2019; Murfin and Spiegel 2020), many Mediterranean regions will be increasingly exposed to a major risk of submersion and erosion, affecting several parts of the coast (with extreme cases being Venice, Kerkennah archipelago in Tunisia, Alexandria and the Nile delta) (UNEP/MAP/PAP 2015; UNEP/MAP 2016). Future risks associated with sea level rise are detailed in Section 6.9. The main consequences on coastal ecosystems include more frequent and/or intensive flooding along low-lying coasts, particularly in delta areas, lagoon coasts, tideland and some islands (Sections 2.2.8.1 and 6.9.1). Slight increases in mean sea level will lead to relatively quick inundation, deterioration and displacement of significant areas of wetland vegetation. Severe losses of coastal wetlands are expected in the Mediterranean (McFadden et al. 2007). Apart from the actual loss of land area, these wetlands support rare and localized habitats containing highly specialized organisms, the degradation or loss of which will in turn impact migratory bird populations, particularly along main migratory routes (Cyprus, Malta, Palm Islands Nature Reserve in Lebanon). Mediterranean waterbird communities already show changes in community composition based on the recent changes in temperature and whether or not they have a strict protection status, greatly improves the adaptability of species and communities (Gaget et al. 2018). Future breeding suitability maps indicate that the little tern (*Sternula albifrons*) and the common tern (*Sterna hirundo*) could potentially face a drastic decrease in suitable breeding grounds even in protected areas (Ivajnšič et al. 2017).

A modelling study from the island of Zakynthos-Greece for the loggerhead sea turtle (*Caretta caretta*) suggests that even under the most conservative 0.2 m sea-level rise scenario, about 38% (range: 31 to 48%) of total nesting beach area would be lost, while an average of 13% (range: 7 to 17%) of current nesting beach area would be lost (Katselidis et al. 2014). For a sea-level rise of 1.2 m, they estimated a loss of 67.3% for loggerhead turtle nests and 59.1% for green turtle nests although suitability of nesting sites for future migration will also be dependent on existing natural and artificial barriers (Varela et al. 2019).

Accelerated cliff and beach erosion will result in habitat and species loss. For example many cliffs host chasmophytic endemics while many coastal habitats of priority importance at the European/global level (i.e., coastal Junipers, *Posidonia* meadows) (Gubbay et al. 2016; Janssen et al. 2016). In the western Mediterranean, seagrass could reach functional extinction under warming scenarios (Jordà et al. 2012; Telesca et al. 2015). The effects of sea-level rise on competition and the subsequent plant diversity decrease in Mediterranean-climate marshes (Noto and Shurin 2017). Increased salinization in the estuaries will result in species changes/structure, function, and occurrence of eutrophication (EEA 2004; Bernes 2005; Robinson et al. 2005; Smayda 2006).

4.2.3 Adaptation

4.2.3.1 Adaptation of different coastal systems

When it comes to adaptation strategies for coastal systems to environmental changes, different zones require specific actions. For example, shorelines are mainly affected by deterioration of engineering species such as corals, and vermetids forming reefs that protect coasts from erosion, regulate sediment transport and accumulation and provide habitat for other species. Estuaries are particularly vulnerable to pollution, including plastic from nearby human settlements and require different adaptation strategies. Thus, suitable adaptation policies include (i) reducing pollution runoff, both from agriculture and industry and waste management, (ii) policies to limit or prevent acidification and (iii) moving aquaculture operations to areas protected from critical acidification levels (Sections 3.1.5 and 6.11).

Keystone Mediterranean benthic species are vulnerable to ocean acidification and warming (Rodolfo-Metalpa et al. 2011, 2014; Milazzo et al. 2014; Zunino et al. 2017; Verdura et al. 2019). Therefore, spatial planning for these areas should include plans for coastal protection and different urbanization schemes (infrastructure sector), plans to enhance the attractiveness of these zones for tourists, and different regulations for recreational boats. The most likely mechanisms by which ocean acidification refugia (OAR) can mitigate ocean acidification impacts are reducing exposure to harmful conditions or enhancing adaptive capacity (Kapsenberg and Cyronak 2019). While local management options, such as creating OAR, can help coastal ecosystems to adapt, they present unique challenges, and reducing global anthropogenic CO_2 emissions remains a priority. Given the scale of ocean acidification impacts on human health and well-being, recognizing and researching these complexities may allow the adaptation of management such that both the harms to human health are reduced and the benefits enhanced (Falkenberg et al. 2020).

Deep waters are mainly impacted by changes in wild harvests so adaptation measures should focus on fisheries indirectly impacted by changes in phytoplankton production at the surface as well as ocean warming. Shallow coastal zones are exposed to changes in availability of fish and shellfish. Hence, here the most effective adaptive measures involve the management of both fisheries and aquaculture, and the wise use of coastal habitats. The resilience of socio-ecological systems to sea level rise, storms and flooding can be enhanced when coastal habitats are used as natural infrastructure since they provide similar services and added benefits that support short- and long-term biological, cultural, social, and economic goals (Powell et al. 2019). Better integration across policy and planning instruments is needed to enhance adaptive capacity at the interface of climate change adaptation, marine and aquaculture planning and management. This requires holistic and cooperative management tools, such as aquaculture management areas, that could support adaptation across wider spatial scales (Greenhill et al. 2020). This could be enabled by establishing links between existing and proposed collaborative groups to enhance development of adaptation responses and through co-ordination of monitoring and review processes to promote learning across scales (Kapsenberg and Cyronak 2019; Powell et al. 2019; Greenhill et al. 2020). Economic and financial tools to promote environmental management are detailed in Section 5.1.3.2.

4.2.3.2 Harmful algal bloom monitoring

In the last two decades, Harmful Algal Bloom (HAB) events have increased, with many species suddenly emerging in regions previously free from such toxic or potentially harmful algae. Along the Mediterranean coastline, several phytoplankton toxic networks have been established such as the French REPHY network[5]. The recent observations are quite atypical for phytoplankton blooms, and may be partially explained by exceptionally favorable new environmental conditions related to climate change (Draredja et al. 2019; Jenhani et al. 2019; Ninčević Gladan et al. 2020). Coastal HABs appear to have increased on a global scale and several reasons have been suggested: better knowledge of toxic species, better monitoring and alerting systems, the transport of algal cysts in ballast waters, the development of aquaculture, the stimulating effect of urban and industrial activities and/or atypical climate conditions (Glibert et al. 2005). The same trend has been observed in the Mediterranean (see sections 2.3.4 and 4.2.1.1). Thus, national and regional water quality assessment efforts and routine coastal monitoring programs intended to detect species, and the study their toxicities have increased worldwide and in the Mediterranean area as well (Nastasi 2010).

Similar to programs elsewhere around the Mediterranean, the Tunisian national monitoring network of phytoplankton and phycotoxins, has been implemented since 1995 to ensure public safety by establishing tools for early warning of bloom events. Also, a regional project “Risk-Monitoring, Modelling and Mitigation (M3-HABs) of benthic microalgal blooms across the Mediterranean regions” found that better awareness of the risks associated with the *Ostreopsis* blooms could be achieved, including appropriate diffusion of cautionary measures, the production of common monitoring protocols, the development of new technologies for species-specific identification, species counting, and the build-up of prediction models in order to prevent and reduce risk factors for the environment, human health and economic activities. Despite the efforts in management and monitoring work, predicting the

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[5] http://www.ifremer.fr/envlit/surveillance/phytoplancton_phycotoxines

impact of climate change on HABs in the Mediterranean is very challenging as it involves many combining factors. Thus, the calls for increased awareness in both HAB and seafood monitoring programs remain a high priority in the Mediterranean region (Turki et al. 2014; Visciano et al. 2016; Estevez et al. 2019; Ordoñez et al. 2019).

4.2.3.3 Early detection of potentially dangerous species

An updated list of introduced alien species in the Mediterranean (November 2018) shows that the non-indigenous species (NIS) number is close to 1000 species (Section 2.5.1.1). Adaptation to NIS requires protecting the coastal population against the possible risks associated with the establishment of NIS (Section 2.5). In fact, in the past two decades research interest in NIS has increased, mostly stimulated by evidence about their ecological and socio-economic impacts in the Mediterranean region. This has also raised the urgency of innovative approaches to forecast, track and manage these species (Corrales et al. 2018) (Section 6.12). For example, the Early Detection and Rapid Response (EDRR) has been recognized as a key aspect for NIS management and acknowledged by the European Commission, and has been included in the new European regulation (EU) No 1143/2014 on the prevention and management of the introduction and spread of NIS. Efficient public awareness campaigns disseminating information to local communities, also through “specific alerts”, was adopted as the key driver to quickly detect unwanted NIS and are still used in the last few years with several theoretical frameworks developed through formalized early warning systems (Azzurro et al. 2014).

The silver-cheeked toadfish *Lagocephalus sceleratus* (Gmelin, 1789) (Tetraodontidae) has expanded rapidly through the Western Mediterranean (Streftaris and Zenetos 2006; Jribi and Bradai 2012; Kara et al. 2015). Due to its toxicity, many Mediterranean countries have quickly responded by informing the general public about the risks associated to the consumption of this species. These awareness initiatives, necessary to limit the impacts of this invasion (Nader et al. 2012), have been carried out in countries such as Egypt, Turkey, Lebanon, Cyprus, Greece and Tunisia (Ben Souissi et al. 2014). The same strategy is being adopted for the common lionfish *Pterois miles* (Section 2.5.1.3). Hence, early detection and continuous monitoring of these species is a successful example of positive interaction between citizens, researchers, and policy-makers (Azzurro et al. 2016).

The current list of NIS provides a reliable updated database and basis to continue monitoring the arrival and spread of NIS in the Mediterranean, as well as to provide counsel to governmental agencies with respect to management and control. The current geographical, taxonomical and impact data gaps can be reduced only by instituting harmonized standards and methodologies for monitoring alien populations in all countries bordering the Mediterranean Sea.

4.2.3.4 Adaptation management strategies for the jellyfish *Pelagia noctiluca*

Long-term climate fluctuations have been correlated with jellyfish abundance in Mediterranean waters as revealed by (Molinero et al. 2005, 2008). The *Pelagia noctiluca* represents the most important jellyfish species in the Mediterranean Sea (an oceanic scyphozoan that has become very abundant along the coasts) with negative interaction and toxicity (Condon et al. 2013). In order to better monitor and track the dispersion of jellyfish in the Mediterranean and raise awareness about these species, many networks have been established. The CIESM JellyWatch Program was set up in 2009 to gather baseline data on the frequency and extent of jellyfish outbreaks across the Mediterranean Sea (CIESM 2009b, 2009a). The Medusa Project set up in Catalonia set out to understand the spatio-temporal dynamics of the jellyfish populations in the NW Mediterranean Sea by carrying out daily sampling during summer (May to September) of 243 beaches, covering more than 500 points. The recommendations of Medusa were to enhance similar sampling programs for all Mediterranean coasts to better understand changes in the distribution, abundance, and blooming patterns of dangerous jellyfish species (Canepa et al. 2014). The MED-JELLYRISK project “towards an early warning system to detect jellyfish swarms”, started with a campaign to better understand the movement of jellyfish blooms. Three sea drifters were deployed off the coast of Mellieha Bay. Based on satellite tracking, the information gathered by the sea drifters - including sea surface currents (direction and strength) and temperature allowed scientists to validate numerical models that can simulate the dispersion of jellyfish blooms and predict their incidence on coastal areas. These jellyfish dispersion models constituted the basic element of a prototype system intended to act as an early warning of jellyfish swarms impacting Mediterranean beaches. The mission of the Italian, Maltese, Spanish and Tunisian scientists behind the MED-JELLYRISK project (2014 – 2015) was making public authorities, local businesses and beachgoers ready to live together with jellyfish and adapting solutions to address the growth of such fascinating creatures (Lucas et al. 2014). The possible relationships between climate change and blooms of *P. noctiluca* have been studied and it has been proposed that *P. noctiluca* may be an indicator of climate variability in the Mediterranean Sea (Daly Yahia et al. 2010; Condon et al. 2013; Rosa et al. 2013).

Despite these studies and monitoring surveys, additional studies and long-term surveys are needed to improve knowledge of the eco-physiology of the marine species, which will help to better manage and maybe take advantage of NIS, harmful microalgal and jellyfish biomasses and/or their bioactive molecules as a resource for biotechnological applications, from biofuels to pharmacology, cosmetics, health products, food for humans and feed for livestock or aquaculture farms (de Domenico et al. 2019).

4.2.3.5 Ecosystem-based adaptation management

Given the already poor conditions of exploited resources, there is a need for fisheries management to adapt to future changes and to incorporate climate change impacts into future management strategy assessment (Moullec et al. 2019) (Section 3.2). Ecosystem-based adaptation is gaining attention as a cost-effective method for protecting human and ecological communities against the impacts of climate change. This approach has been supported by many studies for various Mediterranean habitats to improve their resilience against the consequences of various drivers. For example, an initial assessment of vulnerability to sea-level rise to help decision makers, and other relevant stakeholders, to develop appropriate public policies and land-use planning measures has been provided (Demirkesen et al. 2008). Also, potential strategies to ameliorate the impact of seawater inundation have been proposed, such as: wetland preservation, beach nourishment at tourist resorts and the afforestation of dunes (Snoussi et al. 2008). Protected areas can play an important role in safeguarding coastal dune plant communities against land-use transformations (Prisco et al. 2016). In this context, functional traits can guide conservation planning, helping to identify groups of species most at risk of population declines. Future conservation interventions need to be mindful to ensure that the natural disturbance regime of dune ecosystems is not disrupted. At the water body management level, scientific cooperation is necessary to deal with the conceptual and ecological difficulties derived from inter and intra-lagoon variability in hydrology and biological assemblages, which are inherent factors in the functioning of these complex ecosystems (Pérez-Ruzafa et al. 2011).

The multiple levels of land-sea interactions (Fang et al. 2018) require a new approach to Integrated Coastal Zone Management (ICZM) and marine spatial planning. The Mediterranean includes hotspots of global priority for land-sea integration (Halpern et al. 2009) and there are emerging Mediterranean case studies which embrace such an approach (Ramieri et al. 2019). Conservation planning and management should focus on cross-realm processes and building resilience between realms. In this respect, connectivity between processes and structural elements is of the utmost importance. Single realm connectivity is inadequate (Fang et al. 2018) since it cannot account for cross-system threats (Beger et al. 2010) and multi-realm species (Giakoumi et al. 2019). Interaction between realms should be translated into structural connectivity (see for example the framework proposed by (Beger et al. 2010) or functional connectivity (Magris et al. 2018) where multi-realm species are taken into consideration. In addition, integrating connectivity and climate change (Magris et al. 2014; Keeley et al. 2018) can be used in various spatio-temporal scales and could be fully applied to maintain and restore land–sea processes. Case studies and innovative approaches are highlighted in Sections 6.9.3 and 6.9.4.

4.2.3.6 The role of institutions/actors and local communities: recommendations

Adaptation efforts often focus on one species, or species group (e.g., jellyfish, algae) or on a specific land-sea ecosystem (e.g., sandy beaches). A more holistic approach is required when trying to establish adaptation methods for the entire Mediterranean, which could include:

- Ecosystem Based Management (EBM) of coastal areas: ecosystem-based approaches (the integrated management of land, water and living resources) to climate change adaptation and mitigation.
- Identify adaptation and mitigation interactions (synergies and conflicts) and assumptions related to adaptation/mitigation.
- Building institutional capacity (governance - adaptive management/monitoring) to improve governance over land/sea natural resources and climate change adaptation. A Nexus approach has been proposed by the IUCN Commission on Ecosystem Management - coastal ecosystem group (CEM/CEG) based on three complementary approaches: (i) sectoral adaptation, (ii) cross-sectoral governance and (iii) territorial strategic planning (Krchnak et al. 2011; Ozment et al. 2015).
- Adaptation and coastal community resilience: Impacts will directly affect coastal communities. A participatory research approach may assist in assessing vulnerability of socio-economic groups, their current adaptation strategies and their adaptive capacity to cope with the impacts of climate variability and extremes and sea-level rise. Such an approach may identify barriers and opportunities for community response to climate change and place emphasis on any existing knowledge/practices promoting adaptation.
- Integrating local knowledge and institutions with respect to risk management and adaptation is part of an ecosystem-based approach (Colls et al. 2009).
- At a practical/management level, adaptation to coastal risks can be achieved via hard structures but also soft protection including beach nourishments and dune rehabilitation, in addition to improved spatial planning regulations, and anticipatory and reactive adaptation.

Overall, information, data, adaptation techniques and networks do exist, but collaboration can be enhanced. Also, public involvement in the development and implementation of adaptation strategies for these ecosystems can be improved. Awareness campaigns on the importance of adaptation measures for these ecosystems may help policy-makers to make the necessary steps to ensure their protection and conservation against potential climate change impacts.

4.3 Terrestrial and freshwater ecosystems

4.3.1 Current conditions and past trends

The Mediterranean region is recognized as a global biodiversity hotspot (Myers et al. 2000; Mittermeier et al. 2005), representing one of the Earth's most geologically, biologically, and culturally rich and complex regions (Blondel and Aronson 1999; Blondel et al. 2010; Visconti et al. 2018). More than 50 plant refuges during recent ice ages (Médail and Diadema 2009) have allowed plant diversity to be exceptionally high (Mittermeier et al. 2005), with about 25,000 plant species today (Myers et al. 2000) and 60% endemism (Thompson 2005). There are 290 tree species (Noce et al. 2016), of which 200 are endemic (Quézel and Médail 2003; Gauquelin et al. 2018). Two thirds of Mediterranean amphibian species, 48% of reptiles, a quarter of mammals, 14% of dragonflies, and 3% of birds are endemic (Mittermeier et al. 2005; Lefèvre and Fady 2016; Paine and Lieutier 2016). However, the diversity of several taxa has not been analyzed in detail under the taxonomic and ecological angles (Azam et al. 2016).

4.3.1.1 Past climate variability and its impact on terrestrial ecosystems

Regional temperatures in the Mediterranean Basin are now ~1.4°C higher than during the 1880-1920 period, higher than the increase in global temperature of 0.85°C (Chapter 2). During the Holocene (especially in the second half of this period), periods of precipitation deficits have occurred, but in contrast to the 21st-century situation, temperatures did not rise above the present average. These periods of precipitation deficits (~6 to ~5.2, ~4.2 to ~4, and ~3.1 to ~2.9 thousand years before the present, yr B.P.) have been identified as possible causes of declines or collapses in civilization in the eastern Mediterranean region (Guiot and Kaniewski 2015).

Information from tree rings from different tree species growing at high elevation can provide annually-resolved, absolutely dated climate information across the Mediterranean covering the past centuries. Reconstructions from locations across the Mediterranean reflect different climate conditions during different times of the year. Recent tree ring-based climate reconstructions reflect not only seasonal temperatures but also parts of the hydrological cycle including drought stress. Only few tree-ring based climate reconstructions go beyond the past 600 years and this limits our understanding of drought variability, the magnitude and timing of long-term trends and centennial-scale variability across the Mediterranean back to medieval times.

The most detailed depiction of Mediterranean drought variability over the last 900 years is presented in the Old World Drought Atlas (OWDA), a tree-ring-based field reconstruction of warm-season drought severity (Cook et al. 2015). Summer drought reconstructions from high-elevation sites from Mount Smolikas in the Pindus Mountains (Northern Greece) go back to AD 730 (Konter et al. 2017; Klippel et al. 2018). Drought variability displays significant East-West coherence between the western (Spain, Morocco, Algeria, and Tunisia) and eastern (Balkans, Greece, and Turkey) Mediterranean Basin on multi-decadal to centennial timescales (Cook et al. 2016). There appears to be a north-south contrast in the eastern Mediterranean, with a tendency for wet anomalies in Greece, Anatolia, and the Balkans while Libya, the southern Levant, and the Middle East are dry and vice versa associated with North Atlantic Oscillation (NAO) and other atmospheric circulation dynamics (Cook et al. 2016). The recent droughts in north-western Africa (Morocco) and the Levant are unusual in the context of the past 900 years (Esper et al. 2007; Cook et al. 2016). In the Pyrenees, the Alps, the northern Apennines, the Balkans, the north-western and southern Carpathians long tree ring width formation are mostly controlled by summer temperature (Buntgen et al. 2007; Büntgen et al. 2009, 2017; Popa and Kern 2009; Panayotov et al. 2010). Reconstructions from the Iberian Peninsula and northern Africa for the past 900 years reflect overall warmer conditions around 1200 and 1400, and again after around 1850. Cooler conditions are reconstructed for the mid 13th century and between the 15th and 18th century.

The only winter/spring $\delta^{13}C$ from tree ring-derived temperature reconstruction from southwestern Turkey indicates warmer conditions during the early 12th century and the late 15th century and lower temperatures from the early 16th century to the late 19th century (Heinrich et al. 2013). Tree ring-based climate reconstructions account for a maximum of 35% explained variance and thus are associated with large uncertainties.

Pollen-based reconstructions for the entire Holocene confirm the picture that significant switches between drier and wetter conditions have occurred around the Mediterranean Basin, even if temperatures have never reached current levels. Vegetation has switched between major biome categories in up to 10% of the land area from one century to another, with only slightly higher values during the particular shifts identified above (Guiot and Cramer 2016).

4.3.1.2 Direct human impacts on ecosystems in the past

The Mediterranean Region is also one of the regions with the longest and most intense human occupation in the world (Underwood et al. 2009), and its diversity is the result of co-evolution between human societies and their environment, characterized by constantly evolving land use practices over at least the past 300 generations of human occupation (Blondel 2006). The presence of many endemic species is closely related to extensive use of Mediterranean landscapes, particularly agro-silvo-pastoral mosaic systems (Médail and Quézel 1999) and wetlands (Cuttelod et al. 2009). Reconstructing landscapes and ecosystems over the course of time remains difficult. Knowledge of the human history of the region is therefore still limited. Most changes in land cover appear to have been due to change in human activities, often inducing diversity through changes in different taxonomic groups and their interactions (Sirami et al. 2010). Throughout the Holocene, Mediterranean ecosystems appear to have been rather resilient to perturbations (Blondel 2006; Underwood et al. 2009). Together with urbanization and agriculture intensification (Myers et al. 2000), land abandonment and the decrease in open habitats are key trends in several countries of the northwestern part of the Mediterranean Basin (Portugal, Spain, France and Italy) (Mazzoleni et al. 2004).

Since about 1980, biodiversity changes are faster and greater across different Mediterranean taxonomic groups and habitats (Blondel et al. 2010; Vogt-Schilb et al. 2016; Delpon et al. 2018). Species loss is marked by a general trend of homogenization (loss of vulnerable and rare species) recorded in several taxonomic groups and by a general simplification of biotic interactions (loss of specialized relationship) (Blondel et al. 2010; Visconti et al. 2018).

The most detailed land use map of the Mediterranean indicates a highly heterogeneous spatial structure of land use systems (Figure 4.10) (Malek and Verburg 2017; Malek et al. 2018). In a coarser reconstruction of land use change during recent decades, cropland was found to be the dominant land use (35.2%), grassland was the second most common land cover (26% of plots), followed by forest (20.7%) and other lands (13.4%). Settlement and wetlands accounted for the smallest number of plots, with 3.3% and 1.4% respectively (Martín-Ortega et al. 2018).

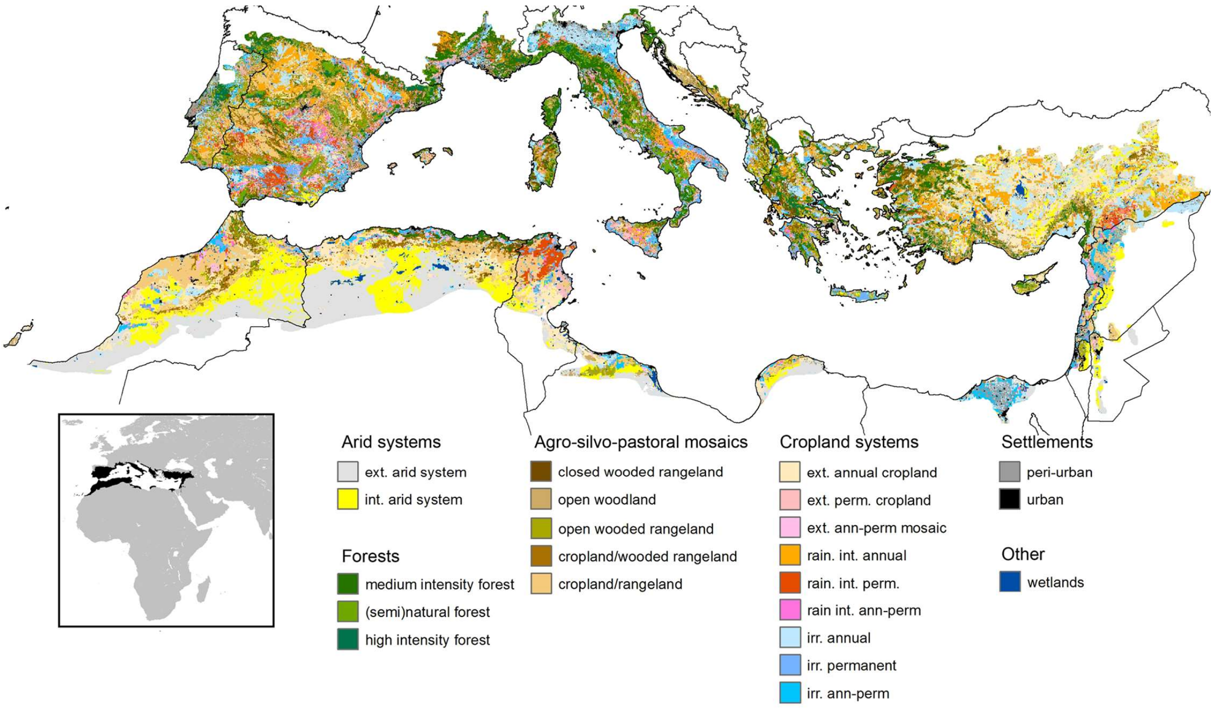

Figure 4.10 | Mediterranean land systems (Malek et al. 2018).

During the period 2000-2015, human activities have intensified in the Mediterranean region, particularly in Spain, France, Turkey and most North African countries (Figure 4.11), where an increase in cropland was recorded. An intensification of agricultural activities in the region is associated with marked transitions from non-irrigated or heterogeneous cropland to permanently irrigated cropland (Ruiz-Benito et al. 2012), leading to an increased use of freshwater resources, with similar projections being made for the future (Malek et al. 2018). New areas containing settlements occurred concurrently with this regional expansion in cropland because of urbanization and tourism, indicating an important trend of urbanization across the region (Martín-Ortega et al. 2018), impacting landscape character, resources use and ecosystem services capacity (Martínez-Fernández et al. 2009; Ruiz-Benito et al. 2012; Balzan et al. 2018).

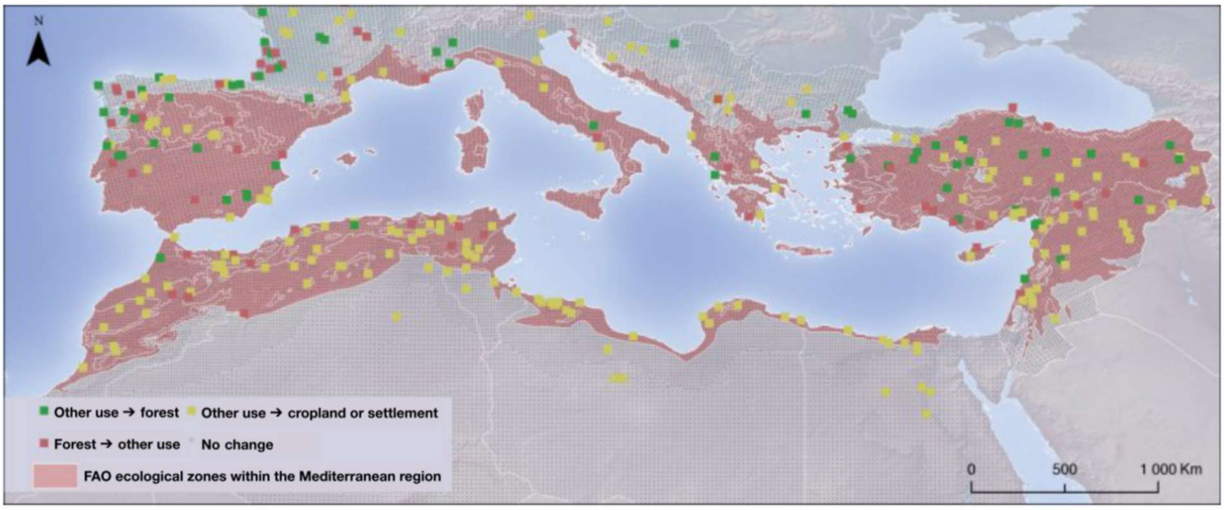

Figure 4.11 | Map of Global Dryland Assessment (GDA) plots showing main changes in land use over the years 2000 to 2015. Non-forest land uses are shown in green. Land use shifting from forests to other uses is shown in red. Changes from other land to cropland and settlements are shown in yellow. Plots that did not change are shown in black (Martín-Ortega et al. 2018).

Forests

In the Mediterranean region, the term ''forest'' comprises a variety of vegetation types interleaved with one another in complex patterns created by variations in soil, topography, climate, human use and fire history, among other factors. This includes dense stands with a closed canopy, as well as pre-forest or pre-steppe structures with lower tree density and tree height (Gauquelin et al. 2018) such as the human-shaped Mediterranean mosaic landscapes, including *dehesas* or *montados*, and multifunctional agro-silvopastoral systems of pastures with scattered oaks, typical of the Iberian Peninsula. The forests of the Mediterranean Basin cover more than 48.2 million ha of which 35 million are in southern Europe, 8.8 in the Middle East and 4.4 in North Africa (Quézel and Médail 2003; Fady and Médail 2004). However, based on the FAO definition of forests ("Land spanning more than 0.5 ha with trees higher than 5 m and a canopy cover of more than 10%"), there were an estimated 88 million ha of forest area in Mediterranean countries in 2015, representing 2.2% of the world's total forest area (FAO and Plan Bleu 2018).

Despite the small extent of the Mediterranean forest area compared to rest of the world, it is a floristic global hotspot with ca. 25,000 flowering plant and fern species (4.5% of the world´s endemics), which represent approx. 10% of the world's flowering plants (Myers et al. 2000). It is also the world's second highest region in terms of in plant endemism, with 50-60% of the plants being found nowhere else, including emblematic species such as cork oak (*Quercus suber*), *argan (Argania spinosa), cypresses* such as *Tetraclinis articulata* or*, Juniperus thurifera*, or fir species such as *Abies pinsapo*, *A. marocana*, *A. nebrodensis*, many of them endemic to the different mountain ranges across the Mediterranean (Thompson 2005; Blondel et al. 2010). The Mediterranean Basin hosts 290 indigenous woody species and subspecies (in comparison to 135 for non-Mediterranean Europe), 201 of which are endemic (Fady-Welterlen 2005). Intra-region variability in climate, soil and human factors result in a mosaic of forest types (Masiero et al. 2013). The relatively harsh climate conditions in arid zones prevent the existence of tall forests and lead to the formation of *maquis* and *garriga* shrublands, dominated by evergreen shrubs such as *Pistacia lentiscus, Quercus coccifera*, *Q. calliprinos* and *Cistus sp*. The semiarid zones are dominated by *Pinus halepensis* in the western part and *Pinus brutia* in the eastern areas. Sub-humid areas are the typical habitat for evergreen oaks such as *Quercus ilex* or *Q. suber*, but also *Pinus pinea* and other accompanying species such as *Arbutus unedo* or *Erica arborea*. Deciduous and marcescent oaks appear in the sub-humid to humid Mediterranean areas, with oak species such as *Quercus pubescens*, *Q. cerris*, *Q. pyrenaica*, *Q. faginea* or *Q. macrolepis*, among others, accompanied by conifers such as *Cedrus sp*. or by Mediterranean firs (*Abies pinsapo* or *A. cephalonica*). In mountain areas pines become the dominant species including *P. nigra* and *P. sylvestris*, and it is also possible to find islands of oceanic climate with *Q. robur, Q. petraea, Fagus sylvatica* or *Abies alba*. Along the rivers, forests of *Fraxinus sp, Populus alba* and *P; nigra* can prosper (FAO and Plan Bleu 2013). The wildlife diversity associated with this variety of forest environments is also high: 786 of 1,601 vertebrate Mediterranean species live in forest habitats and 792 of 1,184 terrestrial insects assessed by the IUCN Red List (as in 2018) are recorded as living in forests, 364 of which are endemic to the Mediterranean region (FAO and Plan Bleu 2018).

The human footprint in Mediterranean forests

The current composition, structure, dynamics and biological diversity of Mediterranean forests cannot be understood without considering the long history of uses and changes induced by human activities, which have contributed to shaping the Mediterranean landscapes as we know them today (Blondel 2006). Human influence in the Mediterranean dates back several thousand years, to the point that some authors argue that a "coevolution" has shaped the interactions between these ecosystems and the human societies that inhabited them (Blondel 2006).

Transformation into agricultural fields, over-exploitation, the prevalence of livestock grazing within forests, and the repeated occurrence of natural and human-caused fires led to a progressive reduction and fragmentation of vegetation cover, and forests are mainly confined into the less fertile slopes and occupy less than 15% of their potential area (Quézel and Médail 2003). In some areas, the loss of forest canopy on slopes and their associated understory after fire events has led to important soil erosion (Cerdà and Mataix-Solera 2009; Shakesby 2011). Nevertheless, most soil degradation in forests of the Mediterranean Basin is associated with overgrazing and trampling of the forest understory (Le Houérou 1990; FAO 2016; FAO and Plan Bleu 2018).

In many areas, however, the combination of forests, pastures and fields, together with the high variability in climate, relief and soil resulted in a mosaic-type landscape that greatly contributed to maintaining the biological diversity of Mediterranean landscapes. These landscapes are highly dependent on human stewardship to maintain their resilience to disturbances (e.g., by reducing fire risk through browsing the forest understory (Blondel 2006). The population increase and industrial development from the 18th century onwards led to an increase in the pressure on forests for wood and charcoal on the northern shore of the Mediterranean Basin (Nocentini and Coll 2013). Forests were intensively cut and transformed into coppices, some species were overexploited, and many forests were transformed into uniform, even-aged systems managed through clearcutting or uniform shelterwood (Puettmann et al. 2008).

At the beginning of the 20th century, extensive reforestation plans were implemented in many European countries to reverse the trend. For example, 3.3 million ha were reforested between 1938 and 1984 in Spain, 460,000 ha in Portugal and around 1.3 million ha in Italy during the 20th century (Pemán and Serrada 2017). These large national reforestation programs mainly used conifers (mostly pines) due to their ability to grow in degraded soils and harsh environments. Many reforested areas contributed to a general improvement of environmental conditions, but the use of a single species over vast areas, together with the lack of subsequent management led to very homogeneous forests, often at excessive densities, with associated expansion of pests and a high risk of wildfires (Nocentini and Coll 2013; Guijarro et al. 2017; Martín-Alcón et al. 2017).

Since 1990, overall forest area has increased by 0.67% yr^{-1} across the Mediterranean Basin (FAO and Plan Bleu 2018). Despite this generally increasing trend, forest loss and degradation still prevail around most of the Mediterranean Basin, especially in coastal areas, due to population increase and urban expansion (FAO and Plan Bleu 2018). Sharp differences can be observed between sub-regions. Almost all countries in the North experienced a huge increase in forest area, with rates around 1% yr^{-1} in Italy, France and Spain (Masiero et al. 2013), to which afforestation only contributes 0.23% yr^{-1}. The major part of this trend is due to the decline of agriculture and grazing and the consequent abandonment of marginal lands that are colonized by forests, a process that has been stimulated by European Common Agricultural Policy subsidies (FAO and Plan Bleu 2013, 2018). In contrast, on the southern Mediterranean shore, forest ecosystems are still at risk of fragmentation or disappearance due to human pressure from clearing and cultivation, overexploitation of firewood and overgrazing (Gauquelin et al. 1999; Croitoru 2007; Palahi et al. 2008; Djema and Messaoudene 2009; Masiero et al. 2013; FAO and Plan Bleu 2018). For example, Algerian forests decreased at a rate of 0.5% from 1990 to 2010 (FAO and Plan Bleu 2013) and a decrease rate of ~126,000 ha yr^{-1} across North Africa has been estimated over the last 25 years (Keenan et al. 2015). This degradation continues despite forest representing 22% of the protected land area in North African countries (FAO and Plan Bleu 2018). However, many of these protected areas generally lack management plans or the resources to implement them (IPBES 2018).

Ecosystem services provision by Mediterranean forests

Mediterranean forests are complex and biodiversity-rich socio-ecological systems, resulting from the coevolution of plants and societies through millennia of human perturbations and management (Blondel 2006; Doblas-Miranda et al. 2015; Gauquelin et al. 2018). Currently, Mediterranean forests play a key role in the livelihoods of diverse communities across the Mediterranean by providing people with ecosystem services, food and products for home consumption and income generation. In particular, the provision of non-wood forest products (NWFPs) and other services (e.g., watershed protection, soil erosion mitigation) (Merlo and Croitoru 2005; Croitoru 2007) stands out from the provision of wood forest products (WFPs) (FAO and Plan Bleu 2018). Removal of WFPs represents about 20 to 40% of the estimated total economy value in most northern countries, but less than 15% in most southern and eastern countries (Croitoru 2007). Northern Mediterranean countries dominate all areas of wood production, especially roundwood, pulpwood and derived products, with countries in eastern Mediterranean making a significant contribution to fiberboard production (Turkey produces 50% of the fiberboard products in the region) (FAO and Plan Bleu 2013). Production is low in the southern Mediterranean countries except for wood fuel, which constitutes one-third of the total production in the sub-region (FAO and Plan Bleu 2013), with firewood reaching 80 to 100% of total removals in Tunisia, Morocco and Lebanon (Croitoru 2007). In any case, the overall production of WFP in the Mediterranean is insufficient to meet regional demands, making the region a net importer of wood and wood forest products (FAO and Plan Bleu 2013).

The main NWFPs of Mediterranean forests include cork, pine cones and pine nuts, mushrooms, chestnuts, honey, truffles, berries, acorns, carob, myrtle, rosemary, and other products. Most of these NWFPs are generally harvested, stored and consumed by local communities or constitute their main source of income (FAO and Plan Bleu 2018). There is a significant geographical variation in production and consumption of NWFPs, largely dependent on the tree species available in each country. Portugal is the main producer of cork (50 percent of total production), followed by Spain (30%), Morocco (6%), Algeria (5%), Tunisia (4%), France (3%) and Italy (3%) (APCOR 2015). In 2016, the estimated annual export value of cork by Mediterranean countries was €1295.8 million (APCOR 2015). Cork production is mostly concentrated in cork-oak savannas (also called ‘dehesas’ or ‘montados’) that result from an intentionally induced simplification (both in terms of structure and species diversity) of the Mediterranean forest: human intervention reduces tree density, removes shrub cover and fosters the growth of grass. These are considered “biodiversity-based product systems” by the Convention of Biological Diversity and have a multi-functional character, contributing to the provision of other services such as fuelwood, acorns (to feed animals), carbon storage and pasture, while supporting important habitats for biodiversity (Bugalho et al. 2011).

Pine nut extraction (from stone pine *Pinus pinea)* generates an income of about €50-60 ha^{-1} yr^{-1} (for a cone yield of 200 kg ha^{-1} yr^{-1}), which is higher than the revenue from timber (€20–30 ha^{-1} yr^{-1}), fuelwood, and other products or uses (FAO and Plan Bleu 2013), where the most productive areas are Portugal and Lebanon, producing 4–7 t ha^{-1} yr^{-1}. In Mediterranean Europe, mushroom picking is a recreational activity and mushrooms are marketed according to origin (e.g., France, Italy and Spain) rather than used solely as a direct food source for local communities (FAO and Plan Bleu 2018). The saffron milk cap (*Lactarius deliciosus*) and porcino (*Boletus edulis*) are the most important species commercially traded, along with the highly-valued black truffle (*Tuber melanosporum*) which is increasingly artificially inoculated on purposefully cultivated oak trees. In Turkey, where 90% of the total pine honey is produced (*Pinus brutia* and *P. halepensis*), the beekeeping sector is the main source of income for nearly 10,000 families in the region (Croitoru and Liagre 2013). In Morocco, for example, argan forests contribute to 7% of regional GDP and ensure subsistence for 14% of the rural population (Croitoru and Liagre 2013). Grazing in forested areas remains the main source of subsistence for local populations in Algeria, Morocco, Tunisia, Lebanon and Turkey (Daly Hassen 2016). One the most recent marketed values of Mediterranean forests is their attraction for tourism, sometimes to the detriment of other forest services (García-Nieto et al. 2013) or to the forest itself (Kuvan 2010).

Mediterranean forests also hold many important non-marketed values, mostly regulating services. Among these, one of the most relevant is watershed protection: forests regulate watershed hydrological regimes and protect against erosion and extreme flooding events while filtering and purifying water for its local consumption (Palahi et al. 2008; Guerra et al. 2016). Watershed protection is the single most valuable benefit from forests in Syria, for example, accounting for more than 50% of the total economic value of forests (US$100 ha^{-1} yr^{-1}). In the Maghreb countries, it is second in value only to grazing, varying within US$26-32 ha^{-1} yr^{-1} (Croitoru and Liagre 2013). Mediterranean forests play an important role in regulating micro-climatic conditions, atmospheric composition, water and biochemical cycles (Peñuelas et al. 2017). They also represent a net carbon sink, helping mitigate climate change impacts (Section 4.3.3). In 2005, the economic value of carbon storage in Mediterranean forests ranged between US$ 37 billion and US$ 63 billion, i.e., 13% of the forests' total economic value, when assuming the SERES IPCC scenarios A1 and B1, respectively, for the 2050 horizon (Ding et al. 2010). However, carbon storage capacity by Mediterranean forests is strongly modulated by management (Seidl et al. 2014; Bravo et al. 2017). In fact, forests can act as carbon sources if disturbed, poorly managed, overexploited or burnt (Ding et al. 2010; Peñuelas et al. 2017).

Other societal values of Mediterranean forests include their cultural, spiritual and religious importance (especially for the few remaining ancient forests) (Mansourian et al. 2013) and their attractiveness (aesthetics) for recreational activities and tourism (FAO and Plan Bleu 2013; Bernetti et al. 2019; Raviv et al. 2020). Recreational uses of Mediterranean forests can lead to trade-offs with other services: for example, in Tunisia, the number of visitors to parks demanding recreational services from forests increased from 93,000 to 110,000 between 1998 and 2014 (Daly-Hassen et al. 2017). In these areas, limiting the access to the public also limits soil erosion, one of the main explicit concerns of Tunisian forests (Daly-Hassen et al. 2017).

Mountains

According to the UNEP definition, Mediterranean mountains cover some 1.7 million km^2. Seven Mediterranean countries are among the top 20 mountainous countries in the world, and half of the countries in the region have at least 50% of their land classified as mountain areas (Regato and Salman 2008). Mediterranean mountains exhibit similarities in their biotic, ecological, physical and environmental characteristics but also significant differences (floristically, human colonization patterns, historic land uses and current anthropogenic pressures). Mediterranean mountains host many regional and local endemic species, some of which are relicts of past biogeographical patterns. Médail and Diadema (2009) identified 33 mountainous areas within 52 refugia in the Mediterranean Basin. Some of these mountains had already been identified as regional biodiversity hotspots (Médail and Quézel 1999) and global centers of plant diversity (Davis et al. 1994).

Historically, Mediterranean mountain forests have been crucial for the development of all civilizations and countries in the region. Most of the prehistoric populations of the Near East originated in Mediterranean mountain areas with very high plant and animal diversity, year-round water, shelter and suitable conditions for survival. Early mountain farmer-herders in the eastern Mediterranean and

North African mountains changed pastoral and cropping patterns leading to the domestication of major livestock and domestic species but also important crops, including barley and wheat.

Land use changes in mountain regions

The long history of human intervention has modified land cover and resulted in numerous land use changes over time. Although Grove and Rackham (2003) support the resilience of modern Mediterranean landscape to changes since ancient times, McNeill (1992) argues that for most of the mountains, the changes that destroyed the environment and left behind skeletal landscapes are comparatively recent (past 200 years). Major land uses included woodcutting, pastoralism, agriculture and mining. Drivers of land use changes (north vs. south) included socio-political, economic, environmental expansion vs. population decline, urbanization and industrialization. The beginning of the 20th century, marked the start of rural emigration from the mountains of northern Mediterranean countries that peaked after the Second World War, resulting in land abandonment and the remarkable increase of forest cover seen in recent years. In southern Mediterranean countries, in contrast, an opposite trend is recorded due to the substantial increase in the rural population and consequent pressure on the mountains for arable and grazing land.

Recent land use changes have disrupted the traditional agro-silvopastoral equilibrium of the Mediterranean mountains. The increased forest cover in the northern Mediterranean has resulted in a decline in species and especially of habitat diversity and an increase in natural hazards, especially forest fires, but increased carbon sequestration and decreased soil erosion and sediment transport to the lowlands. The decrease in forest cover in the southern Mediterranean, on the other hand, has led to severe soil erosion.

The intensity and therefore impact of the principal human activities, i.e. agriculture, grazing and tourism, vary significantly from north to south of the basin. For example, agriculture was historically more important in the Sierra Nevada, the Lucanian Apennines of Italy and the Rif mountains than in Taurus mountains or the Pindos mountains (McNeill 1992). In addition, in Morocco the extensive deforestation of mountains has been reported as a result of an increasing rural population, intensive grazing and the end of traditional pastoral nomadic migrations (Rejdali 2004). Regional differences have also been demonstrated in the case of Lefka Ori and the Psiloritis mountains in Crete where the opposite pattern was observed with abandonment mainly due to different grazing practices and number of stock density (Papanastasis 2012). This reflects the impacts of cultural practices (tradition) even within the same region (Regato and Salman 2008; Papanastasis 2012).

Land use change is still considered a more imminent threat in the short to medium term for mountain areas compared to climate changes (Tasser et al. 2017). Abandonment of agricultural activities (including grazing) emerges as a common trend that can be identified in most of the Euro-Mediterranean mountains as a result of rural depopulation which started in the 1950s, with plenty of examples from Spain, Italy and Greece (Papanastasis 2012). Typical countryside structures such as terraces and stonewalls are collapsing, soil erosion is increasing while secondary succession is taking place in addition to reforestation activities carried out to mitigate the effects of torrential floods within rivers and ravines and to reduce the siltation of reservoirs within the valley bottoms (López-Moreno et al. 2008). Abandonment continues to date with farmlands abandoned as a consequence of migration to urban settlements in lowland areas, resulting in significant reduction in cultivation in the northern Mediterranean (Vicente-Serrano et al. 2004; Lasanta-Martínez et al. 2005). However, reports from Turkey support the argument that 'people are still in the mountains' since the decline in forest cover is small compared to other areas in the Mediterranean (Kadioğullari and Başkent 2008; Günlü et al. 2009) while in the Taurus, livestock husbandry is still very active (Kaniewski et al. 2007).

Mountain biodiversity changes

Mediterranean mountains located on the borders of different biogeographical regions, and three continents, are biodiversity hotpots with a flora which comprises different phytogeographical elements ranging from Euro-Siberian to Arctic-Alpine, and Irano-Turanian in the eastern Mediterranean. This is particularly demonstrated in the mountain flora of Crete, Cyprus and Turkey. In the Mediterranean Basin, with its long history of human activity, mountains are considered to be some of the last remaining wilderness areas with high landscape and biodiversity value providing a wide range of ecosystem services within and beyond their boundaries. Geology, tectonic activity, isolation and limited human activity explain the current biogeographical patterns occurring in Mediterranean mountains. These patterns conform to theory, with mountains displaying low species richness but high endemism along altitudinal gradients. In recognition of their importance, a high number of protected areas and many mountainous areas are part of the UNESCO World Network of Biosphere Reserves (Sierra Nevada, Mount Olympus and Lefka Ori-Crete).

Community composition changes have been recorded both because of land use as well as climate change. The increased forest cover in the northern Mediterranean has resulted in a decline in species and especially of habitat diversity (Papanastasis 2012). Altitudinal shifts have been already reported from the Montseny mountains (Peñuelas and Boada 2003), as well as changes in the abundance of endemic species (Fernández Calzado et al. 2012). Community-level studies in Mediterranean mountain ranges indicate that there will be colonization of high altitudes by subalpine species (Stanisci et al. 2005; Kazakis et al. 2007) or what Gottfried et al. (2012) have termed 'thermophilization'. There is already evidence in Mediterranean mountain areas of an increase in the frequency of extreme events, a direct result of climate change, manifesting itself as droughts and, sediment transfer (Maas and Macklin 2002).

Drylands and shrublands

Drylands are characterized by low precipitation levels which do not compensate for the evaporative demands imposed by high temperatures and solar radiation, thereby exhibiting high aridity levels (MEA 2005; Reynolds et al. 2007). Dryland ecosystems have low productivity, which is often exacerbated by the highly irregular, low predictable pulses of rain, resulting in a long period during which soil moisture depletion with no restoration prevails. Low productivity generates relatively low plant biomass, which produces only small amounts of plant litter and leads to low soil organic contents (Safriel 2006).

Drylands are classified using the aridity index (AI related to the average ratio of annual precipitation over potential evapotranspiration) developed by the United Nations (Middleton and Thomas 1997) into four classes: hyper-arid ($AI<0.05$), arid ($0.05<AI<0.20$), semi-arid ($0.20<AI<0.50$), dry sub-humid ($0.50<AI<0.65$). In the Mediterranean, drylands represent almost 80% of its area (Figure 4.15a) from which 12.3% are classified as hyper-arid, 16.5% arid, 36.7% semi-arid and 14.5% dry-sub-humid (considering the boundaries of the Mediterranean SREX region defined in the 5th IPCC Assessment Reort). Climatic constraints limit the productivity of drylands, increasing their susceptibility to wind and water erosion. These climatic limitations, coupled with intense human activity (e.g, agriculture, grazing and deforestation) has been leading to desertification and land degradation particularly in the Mediterranean (Olsson et al. 2019) (Figure 4.12).

The Mediterranean domain has undergone an overall increase in arid area of almost 15% (from 64% to 78%) at the cost of the more humid aridity classes (Daliakopoulos et al. 2017; Elsen et al. 2017). Changes in land cover in drylands have both human-driven and climate variability as underlying causes and have resulted in extensive land abandonment especially after 1960 (Moreira et al. 2011; Stellmes et al. 2013). The depopulation of marginal areas includes the abandonment of extensively used agricultural areas, the discontinuation of traditional forms of land use, e.g., *dehesas* (wooded pastureland) and a decrease in livestock grazing (Delgado et al. 2010; Rescia et al. 2010). Moreover, extensive active afforestation measures resulted in an increase in forested areas (Valbuena-Carabaña et al. 2010).

Despite their relative levels of aridity, drylands contain a great variety of biodiversity, much of which is highly adapted to water-limited conditions. As a result, there are many animal and plant species and habitats found only in drylands: some semi-arid and dry sub-humid areas are among the most biodiverse regions in the world (Gudka et al. 2014). Diversity is also high in drylands, for example between ecotones, areas of different aridity, temperature or altitude. Species have adapted to these factors in many unique ways, creating a variety of habitats that are essential to the survival of species as well as to the livelihoods of people.

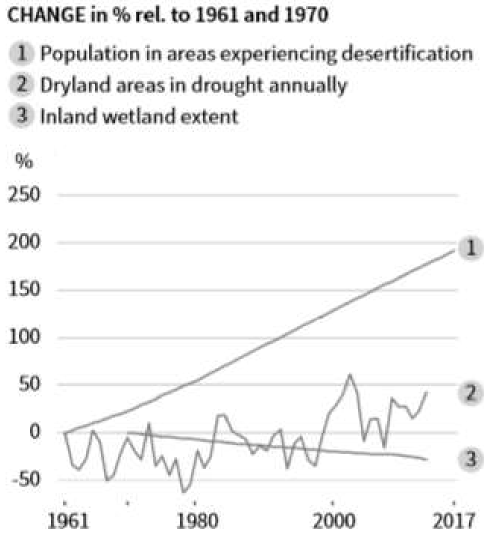

Figure 4.12 | Change in the percentage of land-use change from 1961 to present in relation to desertification and land degradation. Dryland areas were estimated using TerraClimate precipitation and potential evapotranspiration (1980–2015) to identify areas where the Aridity Index is below 0.65. Population data are from the HYDE3.2 database. Areas in drought are based on the 12-month accumulation Global Precipitation Climatology Centre Drought Index. The inland wetland extent (including peatlands) is based on aggregated data from more than 2000 time series that report changes in local wetland area over time (IPCC 2019).

Steppes occupy important areas (630,000 km^2) of the arid zones of the Mediterranean Basin, from the Red Sea to southeastern Spain (Maestre and Cortina 2005). Subjected to strong human impact for millennia, they have been greatly modified. Moreover, some of them, especially the graminean steppe of *Stipa tenacissima*, are the consequence of the degradation of the former open forests. In the past, particularly at the end of the last glacial episode and during the following glacial-interglacial transition, steppes were important biomes representing a significant part of the global continental carbon reservoir.

Shrubland ecosystems account for a substantial part of total land cover and are particularly relevant in arid, semi-arid, and dry-subhumid areas (Reynolds et al. 2007). Shrub-dominated ecosystems are increasing worldwide, a process with important implications for the structure and functioning of terrestrial ecosystems (Van Auken 2000; Berlow et al. 2002; Anthelme et al. 2007). In shrublands, the dominant maquis has many local names reflecting indigenous and local knowledge, such as macchia in Italy, matorral in Spain, phryganae in Greece or bartha in Israel. It is characterized by hard-leaved shrubby evergreen species of genera *Cistus*, *Erica*, *Genista*, *Juniperus*, *Myrtus*, *Phillyrea* and *Pistacia*. The term “garrigue” is restricted to the limestone, semi-arid, lowland and coastal regions of the basin and is maintained by grazing and fires.

Biological soil crusts are complex topsoil microbial assemblages composed of eukaryotic algae, cyanobacteria, mosses, liverworts, fungi and lichens (Velasco Ayuso et al. 2017). They cover the uppermost mm of the soil surface in most arid and semi-arid ecosystems throughout the globe and are one of the most conspicuous and important biotic components of these areas (Belnap and Lange 2013). They exert a strong influence on key ecosystem processes such as runoff and (Belnap 2006) (Alexander and Calvo 1990), soil respiration (Maestre and Cortina 2003), nitrogen fixation and transformations (Belnap 2002; Castillo-Monroy et al. 2010), establishment and performance of vascular plants (Defalco et al.

2001; Escudero et al. 2007) and act as habitats for a dependent food web of arthropods, fungi, bacteria, and other soil organisms (Belnap and Lange 2013).

Dryland biodiversity also provides significant global economic values through the provision of ecosystem services and biodiversity products. Many cultivated plants and livestock breeds originate in drylands, providing a genetic reservoir whose importance is increasing as climate change drives the demand for new adaptations and extinctions of wild breeds. These services, such as cultural identity and spirituality are central to dryland cultures and can be integral to the protection of dryland ecosystems. There has been an observable correlation between land degradation and cultural degradation in drylands demonstrating their interconnectedness (Davies et al. 2012).

Agroecosystems

Agroecosystems support high levels of biodiversity and then a rich diversity of habitats and landscapes because of traditional, low-intensity and diverse agricultural systems (Levers et al. 2016). However, this biodiversity has declined dramatically since the early 1950s due to the intensification of agriculture, leading to an increase in highly modified agroecosystems and simplified and agricultural landscapes (Poláková et al. 2011). The common farmland bird index indicates a reduction in agricultural biodiversity by 34% over the time period 1989-2016[6]. In agricultural landscapes, intensification of agricultural systems has generally induced decreased crop diversity, decreased coverage of natural and semi-natural areas (hedgerows, isolated trees, ponds, permanent grasslands) and lower connectivity between the remaining natural and semi-natural habitats (Stoate et al. 2001, 2009).

Agroecosystems provide important ecosystem services to society, but these are threatened by agricultural abandonment and intensification of agricultural practices. These threaten multifunctional landscapes and erode the capacity to deliver ecosystem services, particularly regulating, and cultural ecosystem services (Nieto-Romero et al. 2014; Balzan et al. 2020). Despite the increasing availability of literature about the topic, there are disparities in the availability of research about agroecosystem services within the Mediterranean region, in particular north-south trends. The limited availability of social research on the topic was identified in a review of Nieto-Romero et al. (2014). Most scientific studies focus on provisioning ecosystem services from intensely managed agroecosystems, whilst regulating and cultural ecosystem services were primarily studied in extensive agroecosystems (Nieto-Romero et al. 2014). These results are supported by recent literature, and a parallel can be drawn with the land-sparing vs. land-sharing debate (Phalan et al. 2011). Intensive agricultural districts, characterized by high landscape homogeneity, were shown to provide food products but are relatively poor in terms of capacity to deliver other services in Barcelona (Baró et al. 2017). Extensive agriculture and semi-natural habitats in Malta and in Sardinia were associated with ecosystem service synergies indicating high landscape multifunctionality (Bagella et al. 2013; Balzan et al. 2018).

Approaches that maintain farmland biodiversity have been linked with an improvement in the delivery of regulation ecosystem services in Mediterranean climates. There is evidence that habitat management through the provision of non-crop plant resources (e.g., floral) and conditions can contribute to increased abundance and diversity of natural enemies, biological control and suppression of crop pests when compared to lower biodiversity controls (Shackelford et al. 2017).

Agroecosystem development in different regions

The ongoing changes in Mediterranean agricultural ecosystems are driven by the dynamics of the global market of food, energy and technology (e.g., seeds, feeds, fertilizers and agrochemicals) and by regional societal changes (Debolini et al. 2018). A summary of these dynamics is necessary to

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[6] European wild bird indicators, 2018 update: https://pecbms.info/european-wild-bird-indicators-2018-update/

understand and frame the ongoing changes of Mediterranean terrestrial ecosystems associated with agricultural systems (Section 3.2.1.1).

Agricultural systems of northern and southern Mediterranean countries face contrasting challenges in relation to their diverse historical backgrounds and ecological constraints. On one side, particularly in southern Europe, farm abandonment in marginal land is associated with the bias introduced by the implementation of the Common Agricultural Policy (CAP) and post-socialism dynamics (Lasanta et al. 2017). Both external (migration, socio-economic model, public policies) and internal (local factors and characteristics of the agricultural holdings) factors trigger and control the land abandonment process, respectively. Biophysical and socio-economic drivers are interlinked, and the outcomes are therefore very site-specific, depending on local contexts. Abandonment of agricultural activities has many landscape, ecological and socio-economic implications particularly during the transition process, which is still under way in many mountainous areas of Mediterranean Europe (Sirami et al. 2010; Alary et al. 2019). In Mediterranean Europe, the agricultural subsidies related to the CAP and the agro-environmental measures had contradictory effects on land abandonment over the years, which in the long run resulted in a sharp decrease in the number of farms and the increase in average farm size (Papadopoulos 2015; Lowder et al. 2016). In contrast, between 1960 and 2000, average farm size in North Africa decreased, becoming less than 2 ha in around 70% of the farms operating less than 10% of farmland, as 50% of the land is farmed by holdings above 10 ha in size. This is consistent with a global trend of farm size reduction in low- and middle-income countries (Lowder et al. 2016).

The contrasting evolution of agriculture between the northern and southern Mediterranean shores indicates the profound differences in socio-economic and biophysical conditions driving agroecosystem change. While in northern Mediterranean countries abandonment of mountain and marginal land and intensification of lowland and coastal areas is clear, in southern countries there is still high pressure from agricultural and grazing systems on lands that are vulnerable to land degradation and desertification. In northern countries the role of grazing livestock is strategic to mitigate the negative impacts of abandoned farmland (e.g., wildfire prevention). In southern countries, overgrazing is still a core issue (Lasanta et al. 2015) with important impacts on biodiversity (Plieninger et al. 2013) and related ecosystem services (Hurni et al. 2015).

In many marginal agricultural areas of Mediterranean countries, particularly in the north west, the abandonment of agriculture and livestock activities and the consequent forest transition is leading to the rapid expansion of wild fauna (e.g., wild boars, wolves, wild dogs) which is in turn negatively impacting farming (e.g. increased production costs and lower competitiveness of the agricultural business) (Otero et al. 2015), and soil degradation (Mauri et al. 2019). The conservation of biodiversity and ecosystem services can emerge from the implementation of adaptive management approaches, including monitoring of population dynamics and related environmental indices (Katona and Coetsee 2019). Abandonment is also generating a loss of plant biodiversity and cultural landscapes associated with grasslands and farmland fields (Malavasi et al. 2018) and the loss of valuable plant and animal germplasm selected over centuries for their adaptive capacity to these marginal lands (Bullitta et al. 2017), that is not of interest for intensive farmland and hence is at risk of loss. The encroached abandoned croplands and grasslands become particularly vulnerable to wildfires, particularly during the transition from grassland to forest, which in Mediterranean countries is represented by pyrophilous shrubby vegetation, particularly in oligotrophic soils (López-Poma et al. 2014; Bagella et al. 2017). In southern Mediterranean countries, overgrazing still prevails with impacts on soil degradation (Martínez-Valderrama et al. 2018) that are compensated by the increasing import of feed for animal food supplementation, which has doubled in Northern African countries in the past two decades (FAO 2017).

A key issue related to the ongoing changes to Mediterranean agriculture is the impact of these changes on ecosystem water resources and the related hydrological cycle (Milano et al. 2013; Martínez-Valderrama et al. 2018). In silvopastoral ecosystems, the transition from grass to woody vegetation exacerbates the negative effects of increasingly frequent drought events and extreme heatwaves associated with ongoing climate changes (Rolo and Moreno 2019). Deep-rooted tall evergreen trees increase actual evapotranspiration beyond the expected increase of reference evapotranspiration due to increased temperatures. Land use abandonment therefore results in the loss of ground and surface water resources, which is expected just when more water is needed both for civil and agricultural uses (García-Ruiz and Lana-Renault 2011).

Agriculture absorbs 80% and 60% of total water demand in African and European countries surrounding the Mediterranean Sea, respectively. Under business-as-usual trends, this demand is expected to rise as a consequence of temperature rise and higher drought frequency, resulting in higher evapotranspiration, while at the same time, groundwater recharge and runoff are expected to be reduced as a consequence of the altered water balance due the above-mentioned land use changes in northern Mediterranean countries (García et al. 2017).

The intensification of agricultural activities in lowland and coastal lands is also impacting biodiversity and ecosystem services as is the abandonment of marginal land. The impact of such intensification processes goes beyond provisioning services and the impact on agricultural biodiversity and multiple regulating and cultural ecosystem services is one of the main focuses of the CAP reform debate in Europe (Nieto-Romero et al. 2014). In the following paragraphs we describe the dynamics and drivers of different Mediterranean agroecosystems to understand the implications for biodiversity and ecosystem services.

Perennial crops

In 2017, over 80% of the 10Mha of olive harvested area in Mediterranean countries was located in Spain (25%), Tunisia (17%), Italy (13%), Morocco (10%), Greece (9%), and Turkey (8%). The harvested area is steadily increasing at a rate of some 140 kha yr^{-1} because of the area increments in the MENA and North African countries, where many new plantations are increasing their productivity under introduced irrigation. However, the sustainability of such irrigated croplands is sometimes questioned in the arid lands of North Africa by the use of non-renewable deep groundwater and the high cost of non-conventional treated wastewater or seawater desalinization (Mualla 2018). Crop yield of perennial crops is instead stable and relatively high in European Mediterranean countries (Tanasijevic et al. 2014). In the case of grapes, the harvested area has declined from 200kha to 150kha in the past 2 to 3 decades, but production is stable as crop yield increased from 7.5 t ha^{-1} in the 1990s to some 8.0-8.5 t ha^{-1} in recent years, again as a consequence of the improvement of agronomic techniques and the use of irrigation (data from FAOSTAT).

Olives and vineyards are a fundamental part of the agricultural landscape and cultural heritage of Mediterranean croplands. In the traditional cropping systems, often based on some sort of agroforestry systems or, sometimes, agro-silvopastoral systems, are designed for a mix of provisioning services (food, wine, cork etc.), but they also provide unique habitats for agrobiodiversity and contribute to multiple ecosystem services (Cohen et al. 2015; Brambilla et al. 2017; Assandri et al. 2018). Almonds and other traditional fruit Mediterranean orchards, and agroforestry systems in many cases represent a traditional and cultural landscape (Moreno et al. 2018).

Vegetables

The production of fresh vegetables is increasing in some Northern African and Western Asian countries, particularly Egypt, Algeria, Israel and Turkey. In Egypt the area harvested has doubled in the last 10 years and now represents over 20% of the total harvestable area of fresh vegetables in the Mediterranean area, with just over 140 kha, slightly higher than Italy, traditionally the first country in the Mediterranean. In all other southern European countries, the harvestable area of fresh vegetables has remained stable during the past three decades. The cultivation of vegetables is related to a wide range of farming systems, ranging from very small family farms for subsistence, mainly in the northern African and Near East countries, to very well-organized industrial horticulture value chains. An extreme example of industrial vegetable production is that of Almeria, in Southern Spain, where some 30-40 kha of greenhouses for vegetables and ornamental plants in a very arid area (200 mm yr^{-1} rain) are producing a gross value of some €1.5-2.0 billion, 75% is generated through the export of fresh vegetables, primarily to northern Europe. These systems were developed relatively recently (the first greenhouse in Almeria was built in the 1960s) and rely on groundwater (80%), with potential overexploitation and salinization of aquifers under way (Custodio et al. 2016). However, a novel bioeconomy model is being developed in Almeria, to increase its sustainability (Egea et al. 2018). Such systems are increasingly growing in other countries, pushed by the demand for out-of-seasons vegetables across Europe, which is sometimes considered more sustainable than domestic production (Tobarra et al. 2018).

Intensive vegetable cropping systems increase the supply of provisioning ecosystem services but impact biodiversity and may lead to trade-offs with other regulating and cultural ecosystem services (Balzan et al. 2020). For example, the introduction of irrigation in arid and semi-arid agroecosystems generates a deep transformation of habitat, species composition and related ecosystem services. The mismanagement of irrigation can lead to soil salinization and impacts on agricultural biodiversity (De Frutos et al. 2015; Juárez-Escario et al. 2017). Intensive production sometimes includes the intensive use of agrochemicals with almost total control of weeds, pests and diseases. Furthermore, the industry includes investments in the development of new varieties characterized by tolerance or resistance to biotic and abiotic stress, reduction of harvesting costs, adaptation to long shelf-life and post harvest packaging, which in practice are reducing the diversity of varieties being grown. On the other hand, old varieties are often more suitable for organic farming systems and can provide valuable germplasm for future needs. Small holders therefore still represent a residual source of valuable germplasm that is at risk of extinction and deserves political attention. This is particularly true for Mediterranean germplasm that had been selected by farmers over centuries. Such farming systems therefore provide a valuable ecosystem service in terms of germplasm in situ conservation which is often linked to the cultural values of the traditional rural societies of the Mediterranean Basin.

Winter cereals

In Mediterranean countries, winter cereals often cover more than 50% of the arable land. Their impact on agroecosystems and ecosystem services is therefore very relevant. However, the winter cereals harvested area in the Mediterranean is generally declining, particularly in southern Europe (e.g., Italy) and is increasing in MENA countries (e.g., Egypt). In contrast, the grain yield is steadily increasing almost everywhere at an average rate ranging from less than +20 kg ha^{-1} in North African countries to +40 to 60 kg ha^{-1} $year^{-1}$ in southern and eastern Europe (FAOSTAT). Schils et al. (2018) have shown that the yield gap between actual and water limited yield potential for wheat is relatively low in central western Europe and is increasing in Mediterranean countries and eastern Europe, where crop stresses other than just water are still limiting actual yield. This decline in harvested area and increase in yield indicates that, in the past, winter cereals were grown on marginal land. This is certainly the case of EU countries where the CAP subsidies were coupled with winter cereal crops until the CAP reform in early 2000, with farmers also “growing the subsidies” in unsuitable areas (Balkhausen et al. 2007).

In Mediterranean agroecosystems, winter cereals are sown between early and late autumn and harvested in early summer. This guarantees soil cover and protection from erosion in winter and spring but as most winter cereal fields are tilled, soils are exposed to water erosion during the early stages of the crop, corresponding to the heavy rains that are frequent in the Mediterranean climate at the start of the season. Furthermore, with intensive crops the capacity of cereal seedlings in the early growth stages to uptake nitrate nitrogen is low, hence either nitrates derived from the natural mineralization of organic matter or from mineral fertilizers distributed before seeding can contribute to the contamination of groundwater. These processes are expected to increase due to increased temperatures leading to higher mineralization rates, and the higher frequency of heavy storms.

Grasslands and grazing systems

Mediterranean grasslands of the “old world” cover over 1 billion ha of land, mostly in the MENA regions. They host some 240 million dairy and meat sheep, 100 million dairy goats and 95 million beef cattle, mostly based on livestock-cereal, agricultural and agro-silvopastoral systems (Porqueddu et al. 2016). Large-scale grazing systems in southern Europe have almost completely abandoned the traditional transhumant system and only few short-distance vertical movements between lowland and upland pastures are maintained (Caballero et al. 2011). Large-scale grazing systems in rangelands and common grasslands are always associated with specific grazing institutions, regulating grazing management and different arrangements between landowners and pastoralists. Such arrangements shape the cohesion of the local rural society, thus generating complex relationships between biophysical and socio-economic processes leading to more or less desirable outcomes at environmental and social scales (Caballero et al. 2011). In MENA countries, the grazing systems are still shaped by such dynamics and overgrazing is among the main drivers impacting land degradation and desertification, with site-specific issues that call for the development of integrated policy implementation frameworks (Middleton 2018).

In Mediterranean countries, livestock grazing systems are often well integrated with winter cereal cropping systems. Grazing can stimulate tillering and hence contribute to increasing the number of cereal heads per unit area. This practice is coupled with early seeding, which can contribute to preventing soil erosion and nitrate leaching. Grazing is suspended before heading and resumed after grain harvest, when grains losses and straw greatly contribute to animal feeding.

Ecosystem services related to pollination

The decline of pollinators is largely seen across Europe (Biesmeijer et al. 2006; Potts et al. 2010; IPBES 2016), but it strongly contrasts with the steadily growing demand for pollination in crop production (Klein et al. 2006; Aizen and Harder 2009; Garibaldi et al. 2013; Breeze et al. 2014; IPBES 2016). Over the last five decades, agriculture has become increasingly pollinator-dependent, with a three-fold increase in the number of crops requiring the intervention of pollinators (Aizen and Harder 2009). The recommended number of honeybees and hives required to provide crop pollination (by considering the natural presence and action of wild pollinators) across 41 European countries rose 4.9 times faster than honeybee stocks between 2005 and 2010 (Garibaldi et al. 2013). As a result, 90% of the demand for honeybee stocks is not met in 22 out of the 41 countries studied (Breeze et al. 2014).

The Mediterranean climate zone has the highest bee species richness in Europe, with the Iberian, Italian and Balkan peninsulas being the most important areas of species richness. Southern Europe also has the highest concentration of endemism, and threatened species. The high diversity of bees in the Mediterranean region is a consequence of the climate of the region and the associated resource heterogeneity, which provide optimal conditions for bee diversity. Petanidou et al. (2008) provide evidence of high temporal plasticity in species composition and interaction identity, indicating that even flower visitation networks show high temporal variation. The main threat to European bees is habitat loss as a result of agriculture intensification and urban development, increased frequency of fires and climate change (Nieto et al. 2015). Fires considerably change vegetation and land cover conditions, and can therefore have an important effect on pollinators and plant pollination (IPBES 2016). For example, fires in Mediterranean oak-pine forests lead to an initial reduction of bee diversity in recently burnt areas. However, these areas recover in the following years, and this recovery is highly correlated to floral diversity (Potts et al. 2003). In Europe, 179 non-threatened species and two threatened species are regarded as under threat from an increased susceptibility to fire, whilst 113 non-threatened species and 23 threatened species are regarded as threatened by climate change (Nieto et al. 2015).

Traditional and non-intensive agricultural practices have a positive impact on agricultural biodiversity in the Mediterranean region (Sokos et al. 2013; Balzan et al. 2020). Similarly, several studies indicate a positive effect of diversified farming systems and organic management related to conventional monocultures (Kennedy et al. 2013). In the meta-analysis by Kennedy et al. (2013), Mediterranean organic fields were estimated to harbor 68% and 56% higher bee abundance and species richness respectively when compared to conventional fields. This study also recorded a significant positive effect of landscape composition, with average increases of 129% and 41% in bee abundance and richness, respectively, for each 0.1 unit increase in the Lonsdorf Landscape Index (an ecologically scaled index of landscape composition) (Kennedy et al. 2013).

The Middle East and Mediterranean Europe recorded higher monetary benefits in crop production that is directly linked with pollination services in comparison to other regions. This is mainly due to the cultivation of a variety of fruit and seed crops (IPBES 2016).

Freshwater ecosystems

Freshwater ecosystems, including streams, rivers, lakes, riparian areas and terrestrial wetlands, offer many important ecosystem services such as water supply for drinking, agriculture and industries (Brauman et al. 2007), water purification, erosion control (MEA 2005; de Groot et al. 2010), recreation, tourism and flood mitigation (Mediterranean Wetlands Observatory 2018). Humans have used these services for thousands of years, and in the process, have severely degraded these ecosystems (Zaimes and Emmanouloudis 2012; Geijzendorffer et al. 2019b). This is particularly true for the Mediterranean region that has been inhabited for thousands of years. The region is characterized by limited water resources and strong population growth (+70% increase in population since 1970 (UN 2013) and +30% in the last 20 years (Abis 2006). Furthermore, substantial increases in seasonal tourism are forecasted in many Mediterranean countries (Burak et al. 2004; Gober 2010), particularly in coastal regions, which can triple in population during the summer (Abis 2006; Collet et al. 2014). Overall, water demand has doubled in the second half of the 20th century in the Mediterranean (Blinda and Thivet 2009; Collet et al. 2013). Accelerated population growth, tourism and globalization are expected to further exacerbate agriculture, urbanization and subsequent pressures leading to an increase in water demand and to significant changes in water use patterns, thus affecting surface waters in the decades to come (Sala et al. 2000; Ferreira et al. 2019; Mack et al. 2019).

River regulation

The highly irregular rainfall patterns and strong seasonal and annual variability of the flow regimes of Mediterranean rivers and streams (Garofano-Gomez et al. 2011), along with the high topographic relief of many of its river (Grantham et al. 2013) have led to the building of more than 3500 dams in rivers during the 20th century (Cuttelod et al. 2009; Lobera et al. 2016). Large dams regulate river flow hydrology and influence water chemistry, sediment dynamics, channel form and biotic communities, act as barriers to sediments, fish migration and vegetation propagules (Brierley and Fryirs 2005; Charlton 2008), change the thermal regime, water quality and biogeochemical fluxes, thus impacting habitat availability and connectivity along the fluvial continuum (Van Steeter and Pitlick 1998; Gasith and Resh 1999; Brierley and Fryirs 2005; Nilsson et al. 2005; Garde 2006; Garofano-Gomez et al. 2011; Bernal et al. 2013; Bonada and Resh 2013; Mediterranean Wetlands Observatory 2018). Reservoirs can reduce the sediment load up to 90% and change the flow from a flashy Mediterranean river to a more constant flow regime below the dam. Downstream, the main consequences of water with reduced sediment supply from upstream include: i) river channel degradation (e.g., bed incision), coarsening of the surface layer and channel narrowing; ii) ecological degradation, damaging the availability and quality of habitat for both the aquatic and riparian biota; and iii) reduction of the sediment supply to the development of the river delta and hence accelerated coastal erosion (Kondolf 1997; Liébault and Piégay 2001; Simon and Rinaldi 2006; Vericat and Batalla 2006; Gendaszek et al. 2012; Lobera et al. 2016). The decline in river sediment inputs can be the result of human activities such as the stabilization of mountain slopes because of rural agriculture decline, rural exodus, reforestation and engineered torrent control (Provansal et al. 2014).

When considering all rivers, the total quantity of freshwater discharged into the Mediterranean each year (not including precipitation) has declined by about 45% during the 20th century (Section 3.1.3).

The reduction in river flows is a probable cause of the very unfavorable conservation status of the biodiversity dependent on rivers: 40% of the fish species found in Mediterranean wetlands are endangered (Mediterranean Wetlands Observatory 2018; Figure 4.13).

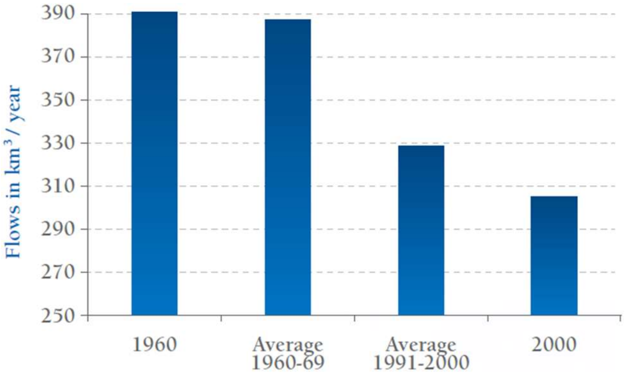

Figure 4.13 | Reduction in freshwater discharge flows into the Mediterranean for all rivers (Mediterranean Wetlands Observatory, 2018)

Groundwater depletion

The close connection between streams and aquifers is responsible for base flow during periods of scarce recharge, controlling stream discharge as well as other hydromorphological characteristics (Winter 1999; Woessner 2000; Menció and Mas-Pla 2010). Human activities, such as groundwater withdrawal or major changes in land cover primarily for agriculture, can result in a reduction of subsurface inflow to streams, changes in groundwater dynamics, and thus, loss of biological quality (Benejam et al. 2008; Menció and Mas-Pla 2010). The needs for water abstraction have risen and it has become difficult to meet water needs (Qadir et al. 2007; Collet et al. 2013; Mediterranean Wetlands Observatory 2018), which is also disrupting the functioning of surface irrigation (EEA 2018; Mack et al. 2019). This is particularly true for Mediterranean islands that are largely dependent on groundwater resources (MED-EUWI 2007; Koutroulis et al. 2013). Exploitation of water resources causes changes in the water balance which leads to low or zero flows, especially during the summer, but also decreases surface water quality (Baldock et al. 2000; Moustadraf et al. 2008; Menció and Mas-Pla 2010).

Hydrologic regimes

Rivers and streams in Mediterranean areas are subject to naturally occurring high variability in their flow, with predictable seasonal disturbances such as floods and droughts (Menció and Mas-Pla 2010; Zaimes et al. 2010) (Section 3.1.3). They can experience wet winters and consequent floods to severe droughts in the summer, when intermittency in otherwise perennial systems can occur (Cid et al. 2017) (section 2.2.5.3). In addition, Mediterranean regions are often rugged, marked by a notable altitudinal gradient between the headwaters and the outlet (Emmanouloudis et al. 2011). Mediterranean streams, located in high elevated areas experience annual rainfall exceeding 1000 mm, and are characterized by low temperatures in winter with the chance of snow accumulation. This creates a typically bimodal pattern in the flow regime, with the highest discharge following the onset of rain and following snowmelt in spring (Sabater et al. 1992), but maintaining a permanent flow throughout the year. In contrast, rivers located in semi-arid areas (low land areas), with mean annual precipitation ranging from 200 to 500 mm, show a less permanent flow regime (many are intermittent and ephemeral) (Lobera et al. 2016).

Human competition for water enhances the natural deficit in water resources of the region, due to mean annual precipitation lower than the mean potential evapotranspiration (Gasith and Resh 1999). Additionally, water diversion, damming, flow regulation, increased salinity, pollution and introduced species have also severely impacted Mediterranean freshwater ecosystems over time (Moyle 1995; Gasith and Resh 1999; Aguiar and Ferreira 2005; Hooke 2006). Overall, the discharge has decreased almost by half in many Mediterranean catchments in the second half of the 20th century (García-Ruiz et al. 2011; Mediterranean Wetlands Observatory 2018). Mediterranean river ecosystems also have a highly endangered biodiversity, with 40% of the fish species being endangered (Mediterranean Wetlands Observatory 2018), which cannot be dissociated from the long history of human disturbances (Zeder 2008; Feio et al. 2014).

Finally, while we are seeing decreases in water availability and runoff, urbanization and infrastructure have encroached river floodplains leading to higher exposure of both people and capital risk to flood damage (Geijzendorffer et al. 2019b). With climate change leading to more unpredictable flash floods, especially in intermittent and ephemeral torrents, the risk and potential damage of river floods has significantly increased in the Mediterranean Basin (Section 3.1.3.3).

Land-use changes, reduction of wetlands and riparian areas

In recent decades, coastal Mediterranean wetlands have suffered considerable pressures from land use change, intensification of urban growth, increasing tourism infrastructure and intensification of agricultural practices (Sanchez et al. 2015). The recent Ramsar Global Wetland Outlook (Ramsar Convention on Wetlands 2018) highlights that the Mediterranean region, where water shortages coincide with demography, had total wetland losses of 48% between 1970 and 2013 (significantly higher than other regions). In addition, 36% of assessed wetland-dependent animals in the Mediterranean are threatened with global extinction (Mediterranean Wetlands Observatory 2018). Special attention should be given to temporary aquatic habitats that are characteristics of the Mediterranean region that provide flood control, groundwater recharge, toxin removal and recycling of nutrients (Balzan et al. 2019). Some temporary wetlands in the Mediterranean region are a priority habitat under the Natura 2000 Network (Natura code 3170, 92 ⁄ 43 ⁄ CEE, 21 May 1992) (Waterkeyn et al. 2010). Land use intensification in/and adjacent to rivers and streams has eliminated or simplified riparian the structural diversity of ecosystems (Robinson et al. 2002; Corbacho et al. 2003; Kingsford and Thomas 2004). These ecosystems have substantial fragmentation in the lowlands of Mediterranean areas primarily due to agriculture, compared to the mountainous areas that can have detrimental effects on their functionality (Zaimes et al. 2011). The maintenance and re-establishment of riparian ecosystems is a difficult but also an important task in southern Europe where most riparian ecosystems have experienced an extensive history of intensive land-use changes and other human disturbances (Corbacho et al. 2003; Zaimes et al. 2010).

The most important parameters for riparian vegetation were the distance from dams, the sea and rivers (Zaimes et al. 2019). Overall, the riparian vegetation of a Mediterranean Basin decreased with increasing drought, flow regulation and agriculture (González et al. 2010; Bruno et al. 2016; Aguiar et al. 2018). Agriculture is the most important stressor for riparian functionality in the Mediterranean. Agricultural land use and hydro-morphological alteration intensification increases in Mediterranean and semi-arid areas (Nilsson and Berggren 2000; Allan 2004; Bruno et al. 2014a) led to a general decrease in both richness and ecological condition. Agricultural land use was the main pressure explaining riparian richness and quality, whereas the responses of aquatic communities were highly related to hydromorphological alteration. These basin-wide variables had a greater effect than variables operating on a local scale (Bruno et al. 2014b).

The riparian sites with the worst quality were near the river mouth and were characterized by an artificial and highly variable flow regime (Zaimes et al. 2011). This artificial flow variability as well as the presence of lateral structures in the river channel and geomorphological characteristics were the main factors driving the hydromorphological and floristic pattern in the regulated river. This flow-biota interaction is remarkable in Mediterranean rivers (Prenda et al. 2006) due to their high biological diversity and extremely variable flow regimes (Blondel and Aronson 1999; Naiman et al. 2008; Zaimes et al. 2010; Garófano Gómez 2013). Many native species of riparian vegetation exhibit life cycles adapted to seasonal peak flows, the loss of which may hinder the regeneration of these riparian communities, reducing their growth rates or favoring the invasion of alien species (Poff et al. 1997). Lateral connectivity is also altered by the reduction of the frequency, magnitude and duration of events that periodically flood banks and floodplains (Charlton 2008), causing loss of native riparian vegetation (Burch et al. 1987; Garofano-Gomez et al. 2011; Zaimes et al. 2019).

In conclusion, the loss of natural wetlands is a major concern, since their loss is nearly irreversible and leads to significant impacts on wetland biodiversity and ecosystem services. Restoration initiatives exist but have a low rate of success when it comes to re-establishing the same richness and stability that can be found in natural wetlands.

Water quality

Based on the Water Framework Directive (WFD; Directive 2000/60/EC), European Union (EU) countries are obligated to assess the ecological status of their freshwater ecosystems using biological indicators, as well as chemical, hydrochemical, and hydro-morphological parameters and to achieve good qualitative and quantitative status of all ground and surface water bodies (Van den Broeck et al. 2015). Additionally, the new Groundwater Directive (GD; Directive 2006/118/EC) considers groundwater as a valuable natural resource that should be protected from deterioration and chemical pollution (Menció and Mas-Pla 2010). However, a concern is that the WFD programs do not incorporate assessment techniques for temporary wetlands (Van den Broeck et al. 2015) whilst the links between water quality and ecosystem functions and services, and the implications of water management on ecosystem services are either implicit or overlooked (Acreman et al. 2017). Outside of the EU, data on water quantity and quality are sparse and often biased. This is a real problem, where countries can affect both water quality and quantity flowing downstream to another country, as is the case for some rivers in the eastern part of the Mediterranean Basin.

The global Sustainable Development of Agenda has included water as an important priority, and SDG6 emphasizes safe access to water and sanitation. Water quality is considered as a major environmental problem across the Mediterranean region (Table 3.4, Section 3.1.3.5), with recent assessment indicating that the WFD has improved water quality in the European countries of the Mediterranean Basin while water quality has degraded further in North Africa and the Middle East (Mediterranean Wetlands Observatory 2018).

Freshwater species

Declines in the Living Planet Index (LPI) of Mediterranean wetlands was continuously observed between 1990 and 2008, after which it increased, but with varying results depending on the group and sub-region. Between 1990 and 2013 waterbird numbers show a positive trend and increased in Western Europe (+101%) and Northern Africa. However, more moderate increases were observed in the eastern Mediterranean (+27%) while declines have been observed in the Middle East since 2008. Contrastingly, declines in amphibians, reptiles, mammals and fish have been observed since 1990 (Mediterranean Wetlands Observatory 2018).

Freshwater communities of the Mediterranean region have adapted to the natural variability in water flows through shorter life spans, mechanisms to resist or avoid desiccation, and higher colonization rates (Lytle and Poff 2004; Bonada et al. 2007; Stromberg et al. 2008; Santos 2010). These Mediterranean communities are, therefore, different from those of temperate rivers, showing interannual fluctuations in richness and composition and in trophic structure (Ferreira et al. 2001, 2002; Bonada et al. 2007; Feio et al. 2010, 2014). During dry seasons (predictable and periodical seasonal droughts), groundwater that flows towards streams is highly significant as it represents a unique input for water discharge, leading to steam/river reaches that are permanent, intermittent or ephemeral (Uys and O'Keeffe 1997; Argyroudi et al. 2009). Seasonal droughts can cause habitat loss, poor water quality and biotic interactions, but in severe droughts (longer, unpredictable, seasonal or supra-seasonal droughts) as expected due to climate change, major ecological effects will be observed, stressing and depleting both fauna and flora (Boulton 2003; Lake 2003; Bond et al. 2008; Menció and Mas-Pla 2010).

Mediterranean rivers present rich and dynamic riparian plant communities, which are highly interconnected with lateral and vertical ecotones and have multi-scaled biotic drivers that act in both space and time (Ferreira et al. 2019; Kontsiotis et al. 2019). Natural and human disturbances are entwined forces that shape riparian plant communities, to the point that undisturbed plant communities are difficult to find or characterize. Though there are few truly aquatic species, Mediterranean riparian plants nonetheless play an important role in stream functions (Zaimes et al. 2010; Magdaleno and Martinez 2014). The protection of these species should be a priority in the region, and many riparian areas are included in the Natura 2000 Network and the Ramsar Convention (Zaimes et al. 2010; Ferreira et al. 2019).

Freshwater ecosystems are under threat from the effects of multiple stressors, including non-indigenous species (Navarro-Ortega et al. 2015). Non-indigenous species are considered in the top five causes of biodiversity loss (Bruno et al. 2019) and result in the accelerated impairment of aquatic and riparian habitats and their ecosystem services worldwide (Saunders et al. 2002; Dudgeon et al. 2006; Van den Broeck et al. 2015; Rouissi et al. 2018; Fraixedas et al. 2019). Their increase and expansion are due to the alteration of their hydrologic regimes, and biological and morphological functionality due to agriculture on the floodplain, channel diversions and dams, and increased pollution (Jiménez-Ruiz and Santín-Montanyá 2016). Non-indigenous species in many cases can tolerate and adapt easier to the new conditions. Exotic species often thrive in Mediterranean rivers altered by human activity, further homogenizing river communities worldwide (Cooper et al. 2013). Alteration of the vegetative structure, competitive displacement of native riparian vegetation, reduction of arthropod and avian diversities and abundances are some of the major impacts of non-indigenous species (Saunders et al. 2002; Herrera and Dudley 2003; Dudgeon et al. 2006; Bruno et al. 2019). Examples of non-indigenous species that are serious threats and problems in Mediterranean riparian areas are the *Robinia pseudoacacia*, *Ailanthus altissima* and *Arundo donax* (Constán-Nava et al. 2015; Bruno et al. 2019; Nadal-Sala et al. 2019). Examples of non-indigenous species that are serious threats and problems in Mediterranean wetlands include *Myriophyllum aquaticum*, *Carpobrotus edulis* and *Cortaderia selloana* (Lastrucci et al. 2018; Chefaoui and Chozas 2019; Company et al. 2019).

Protected areas (Natura 2000 network and Ramsar Convention)

In the European Union, the importance of conserving and protecting freshwater ecosystems is recognized through the many that have been designated as Natura 2000 sites (Iakovoglou et al. 2013)(European Commission 2007). Wetland protection is also officially a priority for the 159 nations (as of 2009) that have ratified the Ramsar Convention, although wetlands still continue to be under threat of being drained and reclaimed (Ramsar Convention on Wetlands 2018; Geijzendorffer et al. 2019a). Degradation is closely related to the rapid increase in human population, and the increased input of nutrients, pollutants, and sediments, due to increases in urban development, industry, agricultural activities, and water abstraction. The most obvious effect is the loss of biodiversity as a consequence of a reduction in area and the deterioration in conditions, especially in arid and semiarid regions (Brinson and Malvárez 2002).

4.3.2 Projected vulnerabilities and risks

4.3.2.1 Forests

The interactions between different drivers of climate change (CO_2, warming, reduced rainfall, increase in drought frequency and intensity) are predicted to have multiple, and sometimes antagonistic effects on the future condition of Mediterranean ecosystems (Bussotti et al. 2014) (Figure 4.14). While increasing atmospheric CO_2 concentrations might directly promote forest productivity and growth (Sabaté and Gracia 2002; Keenan et al. 2011), this effect will likely be strongly modulated by increasing temperatures and drought conditions (Peñuelas et al. 2011; Bussotti et al. 2014; Doblas-Miranda et al. 2017; Lo et al. 2019). For pine and oak-dominated Mediterranean woodlands in Israel, Helman et al.

(2017) projected that warming scenarios of 1 and 2°C could lead to 16% and 31% reductions of annual gross ecosystem productivity, respectively, despite the increase in atmospheric CO_2.

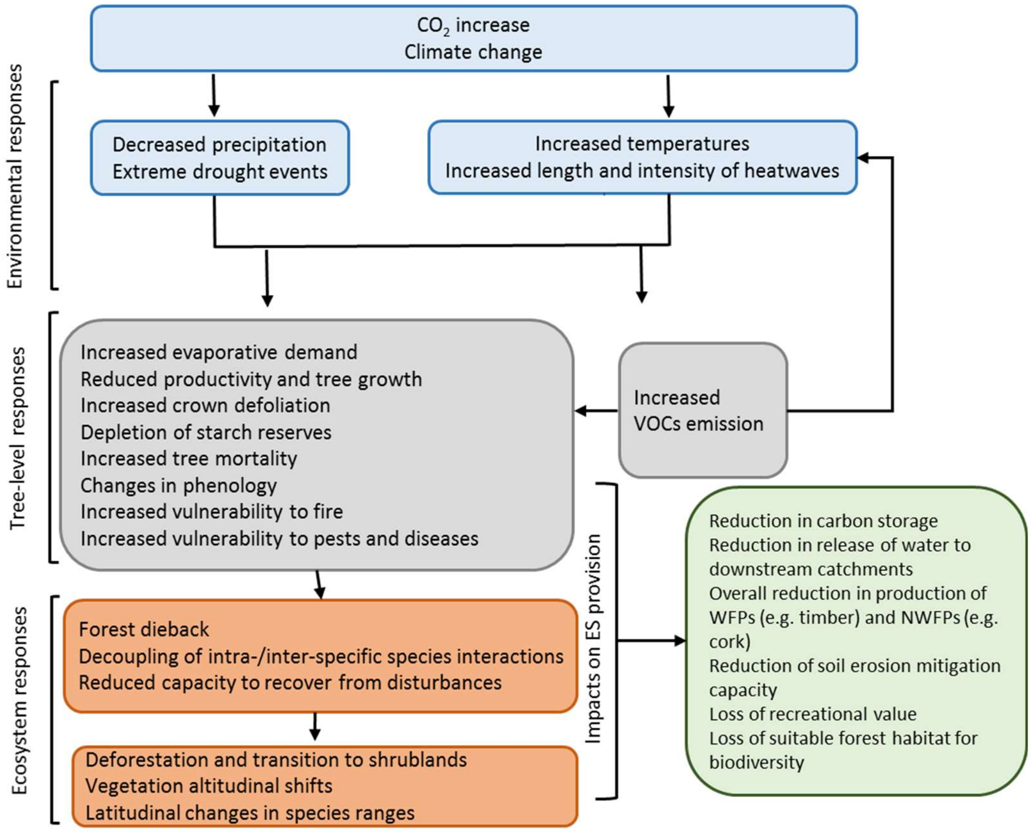

Figure 4.14 | Tree- and ecosystem-level responses in Mediterranean forests to environmental changes associated with climate change and their impacts on ecosystem service (ES) provision. Adapted from Figure 1 in Bussotti et al. (2014) and based on the reviews by Doblas-Miranda et al. (2017) and Peñuelas et al. (2017, 2018).

Warmer and drier conditions also alter plant phenology (i.e., leaf unfolding, flowering and fruiting), usually lengthening the growing season (Peñuelas et al. 2004), with direct consequences on forest productivity and growth (Kramer et al. 2000). Although a longer growing period may be positive for forest growth, advanced spring phenology may also cause higher risk of frost damage (Mutke et al. 2005), as well as increased transpiration. Moreover, global warming also changes bird migrations and dates of insect outbreaks, leading to a decoupling of species interactions (e.g., decoupling of predator-insect interactions reduces the effectiveness of pine processionary moth control by birds; (Barbaro and Battisti 2011). All factors considered, we can expect a general reduction of site productivity in the mid- and long-term, particularly for species or populations growing in water-limited environments, which constitute the majority of Mediterranean forests (Sabaté and Gracia 2002; Bravo-Oviedo et al. 2010).

Changes in forest ecosystem health and ecosystem services provision

There is evidence that Mediterranean forests and woodlands now experience climate-driven declines in growth and die-back episodes from drought and heat stress (Allen et al. 2010; Lindner et al. 2010; Anderegg et al. 2013; Gentilesca et al. 2017; Klein et al. 2019), similar to shrublands (Lloret et al. 2016; Sapes et al. 2017). Increasing crown defoliation and soil respiration may reduce net primary production and ultimately limit growth in Mediterranean forests, associated with higher mortality and dieback, especially if warming is combined with drought (Peñuelas et al. 2018). The combination of reduced water availability and increased respiration rates of tissues due to rising temperatures can result in hydraulic failure, the exhaustion of reserve carbohydrates and a general weakening of the trees, also making them more vulnerable to pests and pathogens (Rennenberg et al. 2006). Drought has been linked to the general dieback of *Quercus ilex* and *Q. suber* in southwestern Spain, where, known as "seca", the weakened trees are more susceptible to the attack of the fungus *Phytophthora* (Sánchez-Salguero et al. 2013). Although even drought-adapted ecosystems are influenced by growth reductions, these phenomena are expected to become particularly frequent in the trailing-edge of species distribution (Jump et al. 2006; Sarris et al. 2011), or for species found in the Mediterranean Basin, the southern limit of their European distribution (Linares et al. 2010; Dorman et al. 2013), which are particularly vulnerable (e.g., *Pinus pinaster, P. nigra, P. halepensis, P. sylvestris, Quercus ilex, Q. suber, Fagus sylvatica, Abies alba, A. pinsapo, Juniperus phoenicea, Cedrus atlantica*), and especially in dense, unmanaged forests (Lindner and Calama 2013) or in sites with shallow soils (Lloret et al. 2004).

This additional climate stress may lead to important changes in biotic interactions, affecting forest composition and species distribution (see next section). For example, in southern Spain, in a mixed *Abies pinsapo–Pinus halepensis* forest, A. *pinsapo* showed sudden growth reductions under drier conditions, while pine trees were able to maintain almost constant growth values and lower water costs under increasing long-term water stress (Linares et al. 2011). Similarly, Sarris et al. (2011) reported that where mixed *Abies cephalonica–Pinus halepensis* forests exist in southern Greece, *Pinus* did not experience any mortality at this altitude (800 a.s.l.) after drought events, unlike *Abies*. Prolonged droughts and hot spells will aggravate the risk of forest fires, which can further induce problems of soil erosion and fertility. In fact, fire-drought interactions can be complex and trigger vegetation transitions, disrupting ecosystem resilience and even leading to non-forest states (Batllori et al. 2019). In the driest areas, desertification can advance and become a major problem (Karavani et al. 2018a).

Climate change is also expected to affect host plant-pest interactions, favoring the establishment of new ones (Lindner and Calama 2013). Warming already causes changes in the distribution areas of pests, mostly upward and northward. An example is the pine processionary moth (*Thaumetopoea pityocampa*), which is expanding upwards in several mountain ranges due to milder winter temperatures, affecting tree populations that had previously never been exposed to this insect (Hódar and Zamora 2004; Battisti et al. 2005; Roques et al. 2015). The succession of several years of mild winters has favored unprecedented outbreaks in northeastern Spain (Roques et al. 2015). Opportunistic fungi and insects such as *Armillaria* or *Ips spp.* are also being favored by warmer temperatures, which induces better conditions for survival, allowing them to complete more than one generation in one year (Lindner and Calama 2013). The greatest impacts are expected to arise from the establishment of alien pests and diseases, i.e. those that are exogenous to a given environment. The number of alien pests is expected to increase under warmer and drier conditions, as has been the case with the pine nematode (*Bursaphelenchus xylophilus*), native from North America, and with the potential to spread across Europe (de la Fuente et al. 2018) and cause massive wilt and mortality in pine species (Vicente et al. 2012).

All these changes may ultimately lead to profound changes in ecosystem function and associated ecosystem services (Seidl et al. 2014; Peñuelas et al. 2017). Changes in carbon storage and water availability are especially important for their implications in all forest services, because they are the basis of the primary production that supports the services (e.g., timber production) and because of the effects they have on climate change (Peñuelas et al. 2017; Ruiz-Peinado et al. 2017). Increased plant evapotranspiration will decrease the movement of water from forest to downstream ecosystems (Peñuelas et al. 2018), compromising supporting services (e.g., water cycle), provision of habitat for aquatic species and water availability for consumptive uses. Severe summer droughts can reduce the yields of economically relevant NWFP such as cork (Oliveira et al. 2016) and pine nuts (Mutke et al. 2005)

The response of some forest ecosystem services to climate change drivers is still under debate. For example, despite the fact that some studies have highlighted that mushroom productivity in Mediterranean ecosystems may be experiencing a sharp drought-induced decrease (Boddy et al. 2014; Ágreda et al. 2015) due to delayed phenology in the autumn season under warmer and drier conditions (Kauserud et al. 2012; Büntgen et al. 2015), simulations by Karavani et al. (2018b) rather point towards an increase in production of edible and marketable species under climate change scenarios as a consequence of the longer mushroom season. The leisure use of Mediterranean pine forests (for walking, mountain biking hunting, etc.) will probably be negatively affected by the increasing incidence of pest outbreaks of the pine processionary moth (Morán-Ordóñez et al. 2019), as this species is responsible of strong allergic reactions in humans (Battisti et al. 2017). However, simulation studies in Mediterranean forests (Mina et al. 2017) suggest that forest management (i.e., silvicultural interventions) might have a prevailing role over climate in determining the future condition of forests and the provision of their associated ecosystem services. This has also been reported in other forest systems across the globe (Albrich et al. 2018; Schwaiger et al. 2019).

Besides the direct impacts of climate change drivers on the condition of tree species and ecosystem services provision, climate change drivers might push Mediterranean forests past critical thresholds (e.g, changes in community composition, loss of ecosystem functions), which could hamper their capacity to recover from disturbances in the future (Anderson-Teixeira et al. 2013). For example, Mediterranean water-stressed forests are likely to become more vulnerable to pests and pathogens (Lindner and Calama 2013; Gauquelin et al. 2018), as well as to other disturbances such as fire. Post-fire regeneration might be limited under water-limited conditions, ultimately leading to deforestation or transition from oak and pine forest to shrublands (Karavani et al. 2018a), thereby decreasing the overall capacity of the region to sequester atmospheric CO_2 and potentially losing the recreational value of affected areas (Peñuelas et al. 2017).

Changes in species range, abundance and extinction

Climate change is predicted to induce changes in the geographic ranges for many terrestrial species across the Mediterranean Basin (expansion, shrinkage, geographic shifts), with studies showing contrasting predictions depending on the modelling approach, the drivers and the scenarios considered, even when predictions are made for the same species and the same region. For example, on the basis of a process-based model, Keenan et al. (2011), predicted that around 40% of the current suitable stand locations of *Quercus ilex* in Spain will become unsuitable for the species during 2050-2080 under a non-Paris agreement compliance warming scenario (3.1°C) whereas Lloret et al. (2013), predicted an increase in climatic suitability for the same species, region, scenario and time horizon on the basis of a correlative model,.

The EU Mediterranean biome was predicted to be the most vulnerable region to plant species loss and turnover in a study by Thuiller et al. (2005), who simulated climatically determined geographic range loss of 1350 European plant species under seven climate change scenarios (IPCC AR4 SRES scenarios predicting temperature increases ranging from 1.8 to 3.6°C), with climate-related range contractions already reported in Mediterranean mountains (Pauli et al. 2012). Consistent patterns have been forecasted for other taxonomic groups. For example, using bioclimatic envelope models and ensemble forecasting of SRES scenarios, Levinsky et al. (2007) and Barbet-Massin et al. (2012), predicted losses up to 100% and 30% of current potential species richness of mammals and bird species in EU Mediterranean, respectively, for the end of the century. In general terms, species at the rear edge of their distribution in the Mediterranean (e.g, deciduous temperate species like *Quercus petraea*) and mountain species (e.g., *Pinus sylvestris*, *Abies alba*) will be the species most threatened by climate change (with ranges potentially shrinking), whereas the most xeric Mediterranean species, which are better adapted to drought, are those expected to encounter fewer problems for survival and range expansion under future climate change (Ruiz-Labourdette et al. 2012; Lindner and Calama 2013; Bussotti et al. 2014).

Projections of species range losses due to climate change across the Mediterranean cannot be taken as precise forecasts given the uncertainties in climate change scenarios. Only a few forecasting studies have assessed the interactions of climate change with other drivers (Morán-Ordóñez et al. 2019), there is therefore a risk that the vulnerability of species to other important disturbances, such as land use change, fires and their synergistic effects, is underestimated (IPBES 2019). An additional caveat for studies projecting changes to climatic range is that generally these do not incorporate the role of interactions between species or the effects of extreme weather events, the latter of which is of great relevance in the context of Mediterranean forest systems.

Fire activity and burnt areas across the Mediterranean

The Mediterranean Basin can be considered as a hotspot under future climate conditions conducive to extreme wildfire events, with significant potential impacts for human well-being (Bowman et al. 2017). How exactly climate change will influence future fire regimes is still under debate. While a warmer and drier climate will upsurge fire activity by increasing water demand and decreasing fuel moisture, increasing temperatures may also negatively affect ecosystem productivity and lead to an overall reduction of fuel biomass, which can counteract warming effects on fire activity (Batllori et al. 2013). Drought increases terpene emissions from Mediterranean plants, which are compounds that play a key role in the flammability of forests (Peñuelas et al. 2018). Warming conditions also increase emissions of other volatile organic compounds (VOCS) besides terpenes, with multiple physiological and ecological functions (e.g., plant defense, communication with other organisms) that, in a cascade effect, can affect communities of organisms, ecosystems, atmospheric chemistry and even meteorological conditions, even potentially generating feedbacks to warming (Doblas-Miranda et al. 2017; Peñuelas et al. 2017) (Figure 4.14).

The increase in exposure to large wildfires in recent years (Bowman et al. 2017), along with the effects of climate change, might still overcome current fire prevention efforts. More and different fire management approaches must therefore be considered in order to increase our resilience towards future Mediterranean forest fires (Moritz et al. 2014; Turco et al. 2018a). Projections indicate an increase of burned areas across the Mediterranean in the future, but it is difficult to compare estimates given the variation between scenarios, future periods and models used. For example, Amatulli et al. (2013) estimated increases of up to 66 and 140% in burnt area in EU-Mediterranean countries in 2071–2100 relative to 1985–2004 under the IPCC SRES scenarios B2 and A2, respectively, and Migliavacca et al. (2013) estimated a 34% increase in burnt area in southern Europe, in 2070–2100 relative to 1960–1990 under the A1B scenario. Turco et al. (2018b) projected future summer burned area in Mediterranean Europe under 1.5, 2, and 3 °C global warming scenarios, concluding that the higher the warming level, the greater the increase in burned area, ranging from a ~40% (1.5°C scenario) to ~100% (3°C scenario) increase from current levels across the scenarios. Although the future total burnt area could be smaller if a stationary relationship between drought and fires is assumed, in all the cases the burned area is still expected to increase with warming. Significant benefits (regarding burnt area reductions) would be obtained if warming were limited to well below 2°C (Turco et al. 2018b). These benefits extend beyond plant cover protection or human safety. A reduction of burned areas also reduces risks of soil erosion and desertification, especially in very dry areas (Shakesby 2011).

4.3.2.2 Mountains

Many of the key observed and projected climate changes identified for southern Europe by EEA (2017) apply in the case of Mediterranean mountains, including: (i) significant increase in heat extremes, (ii) decrease in precipitation and river flow, (iii) increasing risk of droughts, (iv) increasing risk of biodiversity loss and (v) increasing risk of forest fires. For Mediterranean mountains, projections indicate warming between 1.4°C and 5.1°C for 2055 (1.6°C and 8.3°C for 2085) and a decrease in precipitation, mainly during spring (−17% under A1fi and −4.8% under B1 for 2085) (Nogués-Bravo et al. 2008).

Mediterranean mountain environments seem to be accelerating towards uncertain ecological states because of changes associated with climate and land use changes (Nogués-Bravo et al. 2008). For the 21st century, projected warming and reduced rainfall are likely to affect: (i) snow pack and glaciers, which provide key habitats for alpine specialist species and, (ii) water availability and river discharge and therefore aquatic and wetland habitats and species. Beyond these indirect effects of climate change in biodiversity, climate change would, (iii) reduce habitat availability of alpine and sub-alpine belts, increasing the risk of extinction for endemic species or range-restricted species and may well disrupt the biological networks that ultimately support ecosystem functioning (Nogués-Bravo et al. 2008).

Mediterranean mountains are susceptible to forest fires and are vulnerable to hydro-geological risks (floods, landslides, infrastructure damage). Most of these hazards will be increased by the predicted rise in temperature and changes in precipitation patterns. Mediterranean mountains provide basic water-based ecosystem services. Therefore, water management and quality-assurance policies need to consider the specific features of mountains and predicted climate change trends. In mountain environments, changes in precipitation (amount and pattern) will be influenced by local geomorphology and therefore predictions are subject to high uncertainty and variation at local and regional scales. Local climate change adaptation strategies require careful consideration in order to counteract specific pressures.

The homogenization of Mediterranean mountain landscapes due to the abandonment of agropastoralism has negative impacts on biodiversity, water resources, soils and natural hazards (Vogiatzakis 2012). For many species in Mediterranean high-altitude zones, including cedars in Cyprus and Lebanon, migrating upwards is not an option (Fernández Calzado et al. 2012). For other species, such as junipers, climate change and increased fires have adverse effects on regeneration, which is already limited by the environment (Vogiatzakis 2012). Fire events are likely to increase in number and intensity in Mediterranean mountain forests and will be associated with elevational shifts of dominant tree species (Fyllas and Troumbis 2009; Pausas et al. 2009). Reduced depth and persistence of snow cover will also affect high mountain vegetation (García-Romero et al. 2010).

4.3.2.3 Drylands and shrublands

There is high confidence in observed drought increases in the Mediterranean and West Africa and medium confidence that anthropogenic climate change has contributed to increased drying in the Mediterranean region and that this tendency will continue to increase under higher levels of global warming (Koutroulis 2019). According to global warming levels of 1.5°C, 2°C and 4°C above pre-industrial temperatures, Mediterranean land will shift to drier types by 11.6%, 20.1% and 41.3% respectively (Fig.4.15b-d). With a 2°C global temperature rise, annual warming over the world’s drylands is expected to reach 3.2°C–4.0°C, implying about 44% more warming over drylands than elsewhere (Huang et al. 2016), thus potentially aggravating water scarcity issues through increased evaporative demand. The Mediterranean, North Africa and the eastern Mediterranean will be particularly vulnerable to water shortages, and expansion of desert terrain and vegetation is predicted to occur in the Mediterranean biome, an unparalleled change in the last 10,000 years (medium confidence) (Guiot and Cramer 2016). At 2.5°C–3.5°C, risks are expected to become very high with migration from some drylands resulting as the only adaptation option (medium confidence). Scarcity of water for irrigation is expected to increase, in particular in Mediterranean regions, with limited possibilities for adaptation (Haddeland et al. 2014; Malek and Verburg 2018).

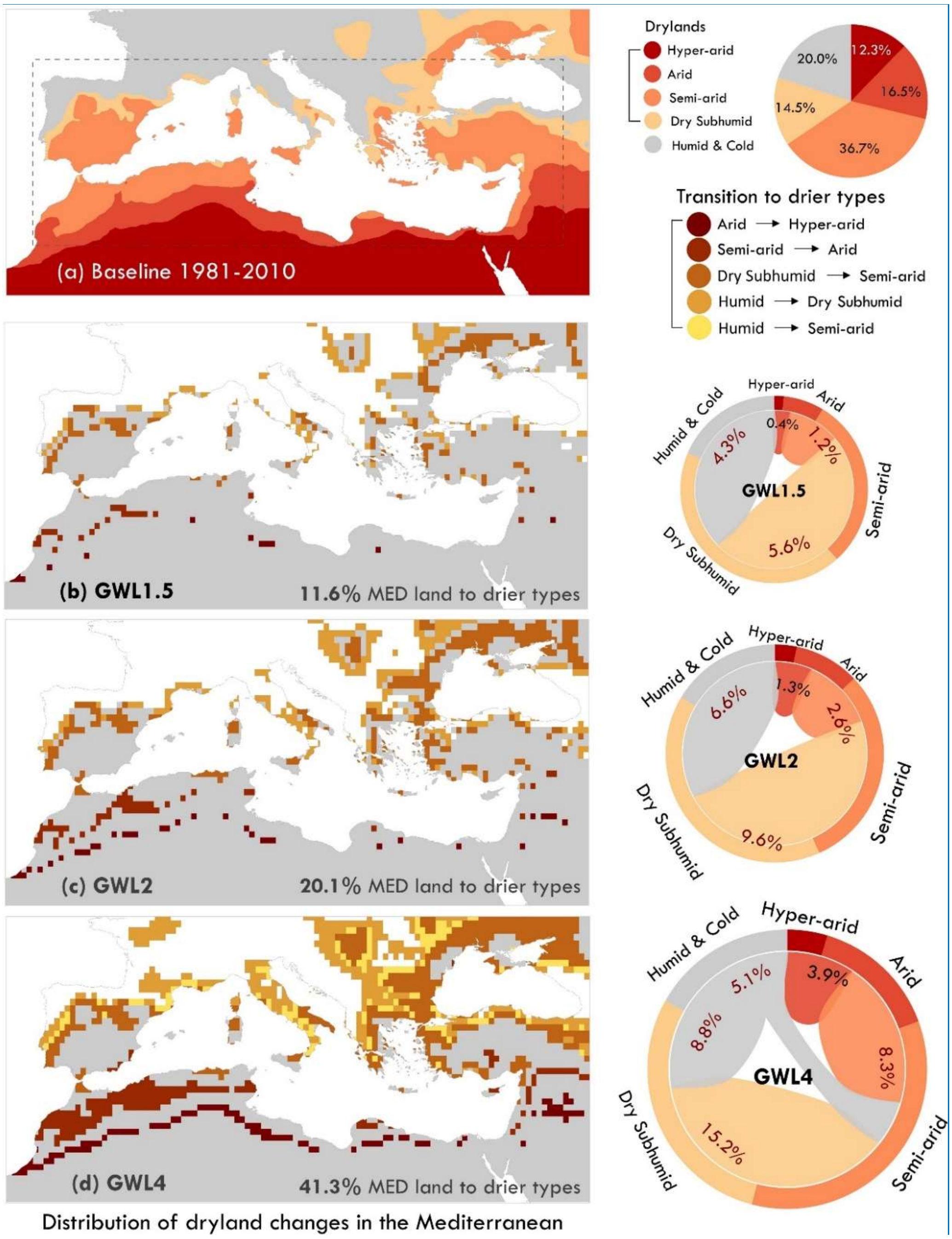

Figure 4.15 | (a) Distribution of drylands and their subtypes based on observations (reanalysis combined with station and satellite observations) for the 1981-2010 baseline period. Areal coverage (percentage) of drylands per subtype is presented in the form of pie charts and is calculated within the boundaries of the Mediterranean SREX region (dashed line). (b, c, d) **Distribution of projected dryland transitions according to RCP8.5 for three Global Warming Levels** (GWLs: +1.5°C, +2°C and +4°C from preindustrial levels), relative to the baseline (1981-2010) period, using the high-resolution atmosphere-only version of the HadGEM3A model. Grey shaded areas in (b), (c) and (d) are drylands of the baseline period. Chord diagrams denote the areal extent (fraction of MED SREX land area) of projected transitions in each dryland subtype under the three GWLs. The size of the chord diagrams is proportional to the total areal extent of the MED land changing to drier types. Figure adapted from (Koutroulis 2019).

Mediterranean drylands are relatively resilient systems with a certain capacity to recover from various forms of disturbance that have occurred for millennia, such as fires, overgrazing and drought (García-Romero et al. 2010). Nevertheless, these ecosystems face critical thresholds with potential catastrophic shifts that may trigger biological diversity losses and modifications in ecosystem functioning and services (Daliakopoulos and Tsanis 2014). The resilience of Mediterranean drylands is currently under pressure from various factors, such as high permanent and seasonal population density, abandonment of traditional practices, continued habitat conversion (2.5% of Mediterranean habitat was lost between 1950 and 1990) and loss of the typically high spatial natural heterogeneity (Fahrig 2003). Although vegetation recovery both from shrubs and forest can reduce soil erosion and enhance carbon sinks, it might change the cultural landscapes frequently developed from initial mosaics of different land-use types towards homogenized states with dense shrubs (Stellmes et al. 2013). Increases in woody biomass and the loss of gaps and breaks as well as edges between different fuel types make these landscapes more vulnerable to fires compared to disconnected patches of forest, shrub and cultivated fields or grasslands (Puigdefábregas and Mendizabal 1998; Viedma et al. 2006; Röder et al. 2008), in particular during drought (Viegas 1998; Duguy et al. 2007).

Since 1960, wildfire occurrence in many Mediterranean drylands has increased because of changes in land use. The level of soil degradation due to these fires depends on fire recurrence, topography, the intensity of soil erosion processes and post-fire plant cover regeneration rate (Caon et al. 2014). To promote the accumulation and retention of nutrients in soil after a fire, it is important to stabilize the burnt site by applying post-fire measures that limit soil erosion, surface runoff and loss of the ash due to wind. Depending on the plant species and the time that elapses between consecutive wildfires, fire is responsible for the transition from forests to shrublands, which are poorer in soil nutrient status. High fire frequency may cause the eradication of keystone species, which has consequences for soil nutrient pool recovery (Caon et al. 2014). In addition to the increased risk of soil erosion, frequent wildfires also induce changes in the water cycle by altering the infiltration capacity of soil and increasing soil hydrophobicity (Vallejo and Alloza 1998; Fernández et al. 2012; Carreiras et al. 2014).

Droughts

Mediterranean shrublands are experiencing important episodes of drought-induced die-back explained by the decrease in climate suitability for plants during the drought event (Lloret et al. 2016; Sapes et al. 2017). In anticipation of a drier climate and to project future changes in dryland dynamics, it is imperative to understand species-specific differences in drought resistance (Väänänen et al. 2020). In the long rainless eastern Mediterranean summer, it was found that the physiological traits of species exhibiting different levels of mortality and co-existing in the same habitat (*Phillyrea latifolia, Pistacia lentiscus and Quercus calliprinos*) were more associated with drought resistance strategies rather than actual drought stress experienced by the plants (Väänänen et al. 2020). The intensity of drought effects on shrub performance is thus species-specific, and plant species combination-dependent. High shrub richness levels modulate the negative impacts of aggravated drought conditions. Results point to a probable shift in interspecific relationships in response to water shortage. As drought impacts are not mediated in low-diversity communities, species-specific responses to drier conditions could lead to shifts in plant community composition favoring the most drought-resistant species such as oaks and rosemary. Maintaining high diversity appears critical to mediate drought effects for less resistant species (e.g., *Cistus* and *Ulex*) (Rodriguez-Ramirez et al. 2017).

4.3.2.4 Agriculture and pasturelands

The ongoing changes to agricultural systems in the Mediterranean Basin are the outcomes of a combination of climate and other drivers that influence farmers' perceptions and choices (Fahrig 2003; Nguyen et al. 2016), which in turn trigger changes in agroecosystems and their service provision at different scales. These processes have a strong site-specific component with a common background of multiple pressures including global and climate changes (Kummu et al. 2017) .

Mediterranean agriculture is generally vulnerable to shocks in the flow of agricultural commodities, particularly in southern countries, because of the heavy dependence on imports (Capone et al. 2014). This is coupled with expectations of future adverse climate impacts leading to the decrease of water-limited crop yields (Schils et al. 2018), increase in irrigated crop water demand, increasing risks in livestock production and mortality associated with heatwaves, expansion of habitats for southern disease vectors and increases in multiple climate hazards (EEA 2017). These impacts hamper the profitability and the ecosystem service provisioning of agricultural activities, particularly in the most vulnerable situations, such as those of farm enterprises relying on natural and semi-natural resources in marginal land or where farmers have made large investments and are therefore exposed to high capital risks (Dono et al. 2014). The latter is the case, for instance of intensive dairy farming, where lower animal productions caused by an increase in the frequency of heatwaves are not counterbalanced by potential benefits in terms of irrigated forage crop yield that can be expected as a consequence of higher temperatures in winter and/or CO_2 concentration rise (Dono et al. 2016).

Water represents a key factor for risk, vulnerability and the resilience of agriculture at farming system and landscape scales (Iglesias et al. 2007). Extreme drought and flooding are two side effects of the same climate pressure (Iglesias et al. 2007; Quintana-Seguí et al. 2016) and there is evidence that the Mediterranean Basin is highly vulnerable to reductions (up to -49%) in provisioning and regulating ecosystem services associated with agricultural water use and management (Jorda-Capdevila et al. 2018). Dramatic changes in the water balance of Mediterranean watersheds are ongoing, as a consequence of a combination of changes in rainfall regime, temperature rise and the increase in vegetation cover from the abandonment of agricultural and pastoral activities, particularly in northern countries (Serpa et al. 2015; Krause et al. 2016; Zeng et al. 2018), which can also result into increased soil erosion (Bussi et al. 2014).

These dynamics are affecting grassland-based systems (e.g., reduction of water pools for animal drinking) and large catchments based on mountain water resources and hence the availability and stability of groundwater and reservoir stocks to be potentially used for irrigation (Rolo and Moreno 2019). There is also evidence of increased vulnerability of water stocks potentially affecting agricultural systems in southern Europe and northern Africa, more than in eastern countries of the Mediterranean Basin, caused by both ongoing climate and socio-economic dynamics, which in most cases, cannot be totally eliminated by adaptation strategies (Koutroulis et al. 2019).

The expected impacts of climate change on croplands and grasslands are often assessed by modelling under current and future climate scenarios (Moriondo et al. 2010; Koutroulis et al. 2019), sometimes integrated with economic (Dono et al. 2016) and policy change analyses (Cortignani and Dono 2018). Unfortunately, only a few studies consider the combined effects of the other drivers of ongoing changes such as those related to technology development, consumer behavior, energy production etc. (Alexander et al. 2015; Doblas-Miranda et al. 2017). Differences in geographic, political and environmental context characterize agriculture and pastureland ecosystems of each sub-region of the Mediterranean Basin, resulting in diverse spatial distribution of vulnerabilities and risks (Prosperi et al. 2014).

In southern European countries the impacts of climate pressures (e.g., increased drought) are coupled with the ongoing transformation agroecosystems (e.g., abandonment in marginal lands and intensification of coastland agriculture). Forest wildfires, landslides and depopulation of marginal rural areas cause additional change (Nainggolan et al. 2012), just as water and air pollution in intensively cultivated areas. Overall, production (food or forage) is given priority over regulating or cultural services (Aguilera et al. 2013; Maes et al. 2018), except for the urbanization of agricultural fertile soils (Ceccarelli et al. 2014).

In the Middle East and northern African countries, multiple stressors include climate, groundwater overexploitation, seawater intrusion in coastal areas, water pollution, land degradation and desertification (Sowers et al. 2011; Schilling et al. 2012; Fouchy et al. 2019), impacting social and political stability (de Châtel 2014). Future trends in agricultural and pastoral land use are very much context-sensitive, as predictions of future dynamics are the consequence of multiple drivers beyond climate, which are much more uncertain in northern Africa than for southern European countries (Prestele et al. 2016).

Focusing on an agricultural and pastoral district (some 54,000 ha) characterized by a mosaic of many different crops and land uses in southern Europe, located in the mid-west Mediterranean Basin, Dono et al. (2016) have shown that the same climate and socio-economic pressures can generate a mosaic of different impacts on diverse farming systems even within the same production system. Several factors, including economic farm size, the dependence on external inputs and the availability of water, contributed to a range of situations: rice and vegetable farms were the "winners", as long as water is not a constraint in the near future, while "losers" were livestock farmers whose feeding system was based on rain-fed grasslands suffering from the increased frequency of extreme drought, hampering hay stock production (e.g, dairy sheep and beef cattle), or those heavily dependent on external inputs, such as the dairy cattle system. Net impacts were associated mainly with the increased frequency of heat waves with a high temperature-humidity index (Bernabucci et al. 2014).

Cropping systems

The potential higher resilience of irrigated cropping systems to increased drought must be managed in the context of +4 to +18% increased crop water requirements due to climate forcing under different scenarios, involving water resource availability in quantity and quality and water use efficiency, particularly in southern and eastern Mediterranean countries (Fader et al. 2016; Malek and Verburg 2018). At present, the Mediterranean region could save some 35% of irrigation water by implementing more efficient irrigation and water management systems, but southern and eastern sub-regions would need around 35% more water than today in the future, even after the implementation of some degree of modernization of irrigation and conveyance systems, taking into account increased CO_2 fertilization effects (Fader et al. 2016) and the need for supplemental irrigation for winter cereals (Saadi et al. 2015). Mediterranean irrigated croplands include a wide range of vegetable crops, including potato, orchards and grapes, forage crops and, in southern countries, sugar cane and cotton. Most C_3 irrigated crops (e.g., many vegetables and rice) would benefit from increased CO_2 fertilization effects and some C_4 from the increased temperature (e.g., sugar cane and maize) but others might be negatively affected (e.g., olives) (Makowski et al. 2020).

Studies on climate change impact on vegetable production are scarce. Bisbis et al. (2018) have shown that climate change may threaten vegetable crop yield and quality in response to rising CO_2 and O_3 concentrations as well as extreme events. Heat stress reduces fruit set of fruiting vegetables and accelerates the development of some crops, thus reducing assimilation, resulting in lower quality and higher product waste. Vernalization of some crops such as cauliflower can also be threatened by cool season temperature increase. Fruit crops such as apples may suffer significant delays to flowering dates due to temperature rise and the difficult achievement of chill requirement fulfillment in milder Mediterranean climates, which might threaten the cultivation of sensitive varieties in currently vacated areas (Funes et al. 2016). Most of these impact assessments are made without considering the threats related to increased incidence of pest, diseases and weeds (Bindi and Olesen 2011; Pautasso et al. 2012; Hulme 2017) and those related to extreme events such as flooding (Erol and Randhir 2012).

In the Mediterranean area, rain-fed croplands include mainly winter cereals, forage crops and other autumn-spring herbaceous crops, and perennials such as grapes and olives. Schils et al. (2018) showed that water-limited cereal yield gaps are still relevant, particularly in eastern European countries, as they are substantially higher than for irrigated crops. Unlocking the potential for production growth requires a substantial increase of crop N uptake and/or N use efficiency. Filling these gaps requires ecological or sustainable intensification of agricultural systems, and has many implications on innovation of cropping systems, their ecosystem services and impacts on GHG emissions and soil functions (Cassman 1999; Serpa et al. 2015; Hamidov et al. 2018; Serraj and Pingali 2019). Currently, the climate resilience of European wheat crops is declining because of the decline in the response diversity that is emerging both from farmers' fields and plot experiments also in southern European countries

(Kahiluoto et al. 2019). This suggests that current breeding programs and cultivar selection practices do not sufficiently prepare for climate uncertainty and variability and calls for more coordinated assessment and communication of response diversity among plant breeders,the recovery of old varieties that had been abandoned by seed producers to be considered and the need for domestication to broaden the germplasm pool (Langridge 2019).

Climate change will impact olive crop evapotranspiration (+8%) and irrigation requirements (+18.5%) and crop phenology, up to reducing the possibility of rain-fed cultivation (Tanasijevic et al. 2014). Furthermore, climate change will also impact the interaction of olive and the obligate olive fruit fly (*Bactrocera oleae*) and alter the economics of olive crop across the basin. Climate warming will affect olive yield and fly infestation levels resulting in economic winners and losers at the local and regional scales, that overall result in threatened biodiversity and soil conservation (Ponti et al. 2014a).

Wine grape production provides a good test case for measuring indirect impacts mediated by changes in agriculture, because viticulture is sensitive to climate and is concentrated in Mediterranean climate regions. At the global scale, the impacts of climate change on viticulture are expected to be substantial, leading to possible conservation conflicts in land use and freshwater ecosystems. The area suitable for viticulture is expected to drop up to 73% in major wine producing regions by 2050 in the worst scenario (RCP8.5), which could be partially compensated by upland or northward cultivation, or by irrigated crops, possibly resulting in land or water degradation (Hannah et al. 2013). The projected increasing temperatures will result in a general acceleration and shortening of the phenological stages compared to the present period. Accordingly, the reduction in time for biomass accumulation negatively affects the final yield. In the cooler subregions of the Mediterranean Basin such as southern France and western Balkans, climate conditions are not limiting and the crop benefited from enhanced atmospheric concentration of carbon dioxide (Schils et al. 2018). Impacts are also expected on grape composition and hence wine quality, in particular with respect to aroma compounds. Furthermore, the frequency of extreme climate events such as hail and flooding is likely to increase vulnerability and risks in some areas (van Leeuwen and Darriet 2016).

In the warmer areas, increasing temperature can have detrimental impacts on grape yield due to increased asynchrony between the larvae-resistant growth stages of the grapevine and the larvae of the grapevine moth. On the other hand, the increase in pest pressure due to the increased number of generations might not be as severe as expected, because of the advance in harvest dates limiting damages from late-season generations. Furthermore, powdery mildew is expected to decrease in disease severity, especially in years with a later onset of the disease symptoms and under the most extreme warming scenarios (Caffarra et al. 2012).

Grasslands and grazing systems

Mediterranean pastoral systems in drylands are expected to be severely impacted from climate change, mainly because of altered rainfall regime and grassland ecosystem water balance. Pastoral mobility, where possible, can mitigate the effects in terms of livelihood but not necessarily in economic terms (Martin et al. 2014). These impacts are expected to be site-specific as they are related mainly to the change in precipitation variability, which appears as the main determinant of degradation in terms of losses in fodder and livestock production in drylands. Perennial forage plants adapted to Mediterranean conditions are a fundamental resource (Lelièvre et al. 2011), providing that sufficient rest is allowed between two subsequent grazing periods. However, projected change is expected to outrange the adaptive capacity of pastoralists. Similar conclusions about the dependence of climate change effects on land use and subregions in grassland systems were achieved by Bütof et al. (2012), who showed how single plant species respond in many different ways to climate pressures because of the complex interactions of climate change with land use practices. More assessments are expected from the use of grassland modelling well-calibrated to Mediterranean-type ecosystems (Pulina et al. 2017; Langridge 2019).

Wooded pastures such as dehesa-type habitats, are a typical high nature value (Bernués et al. 2016) agro-silvopastoral vegetation of many Mediterranean countries, particularly in the western basin (Bagella et al. 2013; Torralba et al. 2016; Seddaiu et al. 2018). These types of ecosystems are already threatened by current management systems under present climate conditions (Rossetti et al. 2015) and by increased drought risks in relation to stocking rates and grazing management, leading to potential higher economic losses with high stocking rates (Iglesias et al. 2016). Tree survival in such ecosystems depends on deep water reserves throughout late spring and summer, which helps to avoid competition for water with herbaceous vegetation (Cubera and Moreno 2007).

Few studies have explored how climate change and grazing interactively affect the biodiversity, primary productivity and ecosystem stability of grassland ecosystems. A recent meta-analysis indicates that the effects of climate change on biodiversity and ecosystem functioning were largely dependent on grazing history within same climate conditions. However, more field studies are needed to test how different climate scenarios affect the biodiversity, functioning, structure and stability of grassland ecosystems, to address sustainable grassland management in different environmental and climate contexts (Kairis et al. 2015; Li et al. 2018).

Vulnerability and risks are mainly associated with the increased frequency of heat stress in summer, leading to heavy impacts on animal health and welfare, i.e. increased incidence of diseases and mortality or lower fertility (Lacetera 2019). Indirect effects of climate pressures increase vulnerability and risks associated with new vector-born infections such as bluetongue (driven by *Culicoides imicola*), or other direct parasites, whose spread can be facilitated by a milder winter climate in northern Mediterranean countries (Bosco et al. 2015). Other indirect effects can be related to the increased incidence of mycotoxins in fodders due to a higher incidence of pests and diseases in forage crops favored by increased temperature (Bernabucci et al. 2011).

4.3.2.5 Freshwater ecosystems

Rivers and streams

In most of the Mediterranean region average river discharge is predicted to decrease while both water temperature and the frequency of large floods are likely to increase (Calbó 2010). The projected decrease in rainfall and increase in temperatures will result in a 10 to 30% decrease in river discharge by the end of the 21st century and a significant reduction in the availability of freshwater (Allen and Ingram 2002; Milly et al. 2005; Lelieveld et al. 2012).

In the Eastern Mediterranean, many authors have detected negative trends in runoff. This was the case for rivers located in Greece (Giakoumakis and Baloutsos 1997) and the Balkans (Genev 2003; Rivas and Koleva-Lizama 2005; Frantar and Hrvatin 2006), Lebanon (Shaban 2009) and Turkey (Kahya and Kalayci 2004). In the western basin, the Duero Basin in the Iberian Peninsula is the most obvious example. Since 1960, Duero River discharges have decreased by 20 to 50% (Ceballos-Barbancho et al. 2008; Morán-Tejeda et al. 2010). Most Mediterranean catchment headwaters are in mountainous areas and are snow-fed. In various regions across the Mediterranean, snow-fed high mountainous springs are the only source of runoff during the long dry summer of the Mediterranean climate. Hence, an increase in temperatures cause less snow accumulation and an irregular and rapid snowmelt, which will result in turn in higher winter and lower spring discharges and decreasing summer low flows. These impacts are observed in various part of the Mediterranean such as the Pyrenees (López-Moreno and García-Ruiz 2004; López-Moreno 2005) and in Lebanon (Shaban 2009).

Overall, projections suggest decreased hydrological connectivity, increased concentration of pollutants during droughts, changes in biological communities as a result of harsher environmental conditions, and a decrease in biological processes like nutrient uptake, primary production, or decomposition. Furthermore, the increased pressure on shrinking water resources will compound the impacts on river ecosystems (Navarro-Ortega et al. 2015).

Wetlands

Mediterranean wetland water depths and hydroperiods (meaning the water inundation period) along with the increase in their salinity levels and isolation and fragmentation are affected by multiple human activities (e.g., water extraction) (Ramírez et al. 2018). These activities are altering the water budgets of wetlands and reducing their ecosystem services. Reed beds in the region have expanded by 89.3% and are the predominant aquatic plant of the all wetlands in the region, which is a major change. In contrast, open water areas and wet meadows have decreased by 53.7 and 96.5% respectively (Papastergiadou et al. 2007). The loss of these key wetland features (e.g., open waters and wet meadows) are impacting the structure of waterbird communities. The future conditions of climate change scenarios will further reduce the environmental suitability of Mediterranean wetlands for the guilds of diving birds and vegetation gleaners (Ramírez et al. 2018).

Freshwater biodiversity

The high intensity and large-scale water management alterations on rivers and streams of the region have had a particularly strong impact on these ecosystems, possibly the highest in the world (Grantham et al. 2013). A similar trend in fish biodiversity loss, also associated with water management pressures, has been reported for rivers in the Iberian Peninsula (Aparicio et al. 2000; Benejam et al. 2008; Clavero et al. 2010). The establishment of alien species in these ecosystems, which can alter natural processes and adversely affect native biota, has also been associated with numerous anthropogenic hydrologic infrastructure in the region (Elvira and Almodovar 2001; Clavero et al. 2004; Light and Marchetti 2007; Grantham et al. 2013).

The Mediterranean-climate freshwater ecosystems host fauna that have evolved and are adapted to the stresses of its streams and rivers. With climate change predicting longer or more extreme drying events (Lawrence et al. 2010; Filipe et al. 2013), their populations and communities will be highly stressed during dry years, thus reducing the resilience capacity of Mediterranean rivers and streams and compromising the survival of their biota (Magalhães et al. 2007). Under this situation, these new conditions will lead to irreversible, and undesirable, “regime shifts” in Mediterranean rivers (Cid et al. 2017).

Due to climate induced changes, stream biota tend to move towards higher elevations and upper latitudes, while the communities change and homogenize their composition (Filipe et al. 2013). Some life-history traits provide biota with resilience and resistance to adapt to the new conditions although it appears that in many cases, current and future environmental changes are exceeding the biota survival boundaries. The difficulty of distinguishing disturbances due to natural hydrologic variability from the effects of climate change in the region make adaptation forecasts even more challenging. Long-term studies are needed to improve knowledge regarding stream biota ecological responses due to climate change (Filipe et al. 2013).

The reduction of subsurface inflow to streams and the changes in groundwater dynamics that have degraded of their biological quality have already made these ecosystems highly vulnerable (Benejam et al. 2008). In addition, wastewater inflow (whether treated or not) into streams will further exacerbate the pressures on fluvial ecosystems, even though initially the induced drought impacts can be partially offset by these industrial discharges. The hydrological benefits of these discharges are compromised by declines in water quality and habitat quality. Moreover, the capacity of aquatic ecosystems to cope with droughts has been lost or significantly reduced in many regions (Andersen et al. 2004; Bond et al. 2008; Rault et al. 2019).

4.3.3 Adaptation

Communities continue to be significantly dependent on ecosystem services for their livelihoods and therefore the preservation of the livelihood and culture of communities together with its biodiversity, in these areas is considered as important to promoting sustainable development and adaptation to climate stresses within the region. The integration of humans, and human actions within the landscapes and seascapes of the Mediterranean region also embraces the IUCN Category IV - Protected Landscapes (Dudley 2008), which provides the flexibility to offer protection to entire landscapes, as shaped through the interaction of human actions and nature, as well as specific protection for specially defined purposes (e.g., habitats/species). Appropriate (integrated) landscape management can be used to promote heterogeneity compensating for the loss of habitat diversity:

- the protection of traditional food systems, conservation of species and functional agricultural biodiversity, and improvement in cropping and irrigation systems to adapt to a changing climate
- sustainable urban development that promotes the uptake of nature-based solutions that are suitable for a Mediterranean climate in urban areas to provide benefits to biodiversity, contribute to ecosystem services and increase resilience to climate change (Box 4.2)
- semi-natural ecosystems: adaptive management includes the implementation of habitat management, restoration and afforestation actions to provide benefits to biodiversity and human well-being, whilst using species adapted to expected future conditions,
- managing changing disturbance regimes: preventive (e.g., pest monitoring) and remedial (e.g., sanitation felling, pest control),

This section provides an overview of the opportunities for adaptation of ecosystems, through incremental (capacity-building) actions and impact-based actions, whilst considering the limitations to the adaptability of Mediterranean social-ecological systems and the impacts of these actions on biodiversity and the ecosystem services and benefits to human well-being. In this analysis, the role of human influences and inputs on ecosystem structure and functions is critical to promoting (or limiting) adaptation of Mediterranean social-ecological systems.

4.3.3.1 Forests

Mediterranean forests will need to adapt to a warmer and drier climate, which entails extended drought periods, long heat waves, increasing fire risk and exposure to increased intensity and frequency of biotic disturbances (e.g., pests). Mediterranean forests, as any other type of ecosystem, have an inherent adaptive capacity as a result of the co-evolution of plants with environmental conditions that have always changed (Valladares et al. 2014a). However, the speed of current environmental change is unprecedented and poses doubts concerning the ability of Mediterranean species to cope with the change to come, and in some cases might make it advisable to adopt planned adaptive measures.

Biological adaptation

The inherent adaptive capacity of forests includes *in situ* adjustments to new environmental conditions via phenotypic plasticity or natural selection, and migration to more suitable habitats (Matesanz and Valladares 2014). Climate envelopes are shifting polewards and upwards, and the easiest response to climate change may be a geographic shift in distribution into climatically suitable areas (Christmas et al. 2016). There is already evidence of some species responding to increasingly warmer and arid conditions through altitudinal or latitudinal migration. For example, Peñuelas et al. (2007) and Peñuelas and Boada (2003) showed a gradual upward shift of the temperate *Fagus sylvatica* species in north-eastern Spain and their gradual replacement by the xeric *Quercus ile*x in the mid- and low- altitudes. Similarly, Sanz-Elorza et al. (2003) reported an encroachment of sub-alpine grasslands by Mediterranean woody species characteristic of lower altitudes during the second half of the 20th century in mountain systems of central Spain. Upward migration of forest species has also been reported in the Italian Apennines (Palombo et al. 2013), the Spanish Pyrenees (Améztegui et al. 2010, 2016) or south-eastern France (Bodin et al. 2013). However, although these movements coincide with an increase in temperatures, changes in land use (agricultural abandonment and reduction of anthropic pressure on forests) seem to play a preponderant role in forest expansion (Améztegui et al. 2010). Most of the species altitudinal displacements have occurred via the colonization of open areas after their abandonment. Replacement of a given tree species by their low-altitude neighbors is only possible when there is a retraction in its trailing-edge distribution, such as in the case of the Montseny mountains studied by Peñuelas and Boada (2003). However, altitudinal range retractions have received much less attention than lead-edge expansions, particularly in Mediterranean mountains, although they seem to be occurring in many mountain areas worldwide (Jump et al. 2009).

Phenological observations since the 1950s show a fairly consistent response of Mediterranean vegetation to rising temperatures. Between 80 and 96% of the species studied advanced their leaf unfolding, delayed the leaf fall, or both, which resulted in an average extension of 30 days in the growing season between 1952 and 2000 (Peñuelas et al. 2002). The lengthening of the growing season could trigger increases in growth but can also cause higher frost damage risk and increased water transpiration. There is less consensus about the physiological plasticity of Mediterranean forest species to environmental stressors. The current available information reveals the potential for some Mediterranean plant species for significant plasticity and rapid evolutionary change, and epigenetic responses have also been documented (Madlung and Comai 2004). However, this information is fragmentary and suggests large differences among species, with some of them being quite vulnerable to fast rates of environmental change (Matesanz and Valladares 2014). Some studies reveal contrasting functional responses to disturbances among tree species and forest biomes. For instance, evergreen gymnosperms growing in drought-prone areas showed lower resistance but faster recovery after drought events than plants dominating in temperate or wet regions, which suggests different physiological strategies to cope with drought (Gazol et al. 2018). This may be of great importance in a changing future, as the response of vegetation may be different as droughts become more intense, more frequent, or both. Tree species with wide a distribution range also seem to display contrasting responses across their entire range (Benito-Garzón et al. 2011) that have been related to intraspecific plasticity and genetic differentiation among tree populations as a result of differences in the intensity of the environmental stresses (Benito-Garzón et al. 2013). In a drier environment, interactions between species may also be altered. According to the stress gradient hypothesis (Maestre et al. 2009), facilitative effects may become more frequent. In fact, the role of shrubs as nurse vegetation for pine seedlings has already been documented in semi-arid and arid Mediterranean regions (Castro et al. 2004; Gómez-Aparicio et al. 2008), and this role could become even more important in the future.

The degree to which physiological responses lead to vegetation shifts (i.e., changes in the composition of the vegetation) is fundamentally unresolved. When two or more species coexist and are differently affected by directional changes in climate and/or by disturbance events, demographic responses become fundamental to project the fate of woody plant communities (Martínez-Vilalta and Lloret 2016). Mortality and regeneration thus become the key processes, since a vegetation shift will only occur if the initially affected species is not able to regenerate and dominate again (Martínez-Vilalta and Lloret 2016). In a drier environment, interactions between species may also be altered. According to the stress gradient hypothesis (Maestre et al. 2009), facilitative effects may become more frequent. In fact, the role of shrubs as nurse vegetation for pine seedlings has already been documented in semi-arid and arid Mediterranean regions (Castro et al. 2004; Gómez-Aparicio et al. 2008), and this role could become even more important in the future.

Limits to adaptation

Forests are particularly sensitive to climate change, because the long life-span of trees does not allow for rapid adaptation to environmental changes (Lindner et al. 2010). Moreover, the Mediterranean region is likely to experience more adverse effects of climate change when compared to Europe, while being the least prepared to cope with such drastic changes (Lindner et al. 2010; Lindner and Calama 2013). On the one hand, the strong human impact on Mediterranean forests has led to high levels of fragmentation, which alter population genetics and species ecology, and affect the ability of populations to respond to environmental changes. Furthermore, adaptive capacity is usually the lowest at the rear edge of species ranges, where plants are growing close to their physiological limits, as is the case for many species dwelling in Mediterranean forests. These forests cannot benefit from gene flow from better adapted populations, so only short-term adaptation and plasticity are available to cope with the extinction risk (Lindner and Calama 2013).

Fragmented populations suffer from greater genetic drift, homozygosity and inbreeding within populations, and are less likely to benefit from the positive effects of gene flow (Valladares et al. 2014b; Christmas et al. 2016). Both the adaptive capacity to new environmental conditions and the ability to migrate are hampered by fragmentation. Shifts in species or population ranges to track optimal climate conditions can be limited by fragmentation, which acts as a barrier for the colonization of many species. Indeed, migrations are not as common as could be expected (Harsch et al. 2009; Zhu et al. 2012), potentially putting populations at higher risk of becoming increasingly maladapted over time (Christmas et al. 2016), particularly at the rear end of species distributions, where populations are deprived of gene flow from better adapted populations (Lindner et al. 2010). Under the projected rates of future climate change, migration will rely on the evolution of very long dispersal distances in order to enable species to reach suitable new habitats. Moreover, global warming will also lead to a decoupling of species interactions (pollinators, predator-prey, etc.) that can further complicate the migration of several forest species.

If species fail to migrate, then *in situ* adaptation will be the only strategy for persistence (Christmas et al. 2016). In this regard, fragmented populations also show lower capacity of adapting and responding to changing conditions. Forests with a greater diversity of response traits (i.e., traits that confer the organisms the ability to cope with disturbances), are indeed more resistant and resilient to disturbances, increasing the likelihood that such communities may persist under future conditions (Sánchez-Pinillos et al. 2016). Moreover, the effects of fragmentation on the genetic diversity of organisms can last for centuries, and some populations still hold legacies from the effects of human actions centuries ago, as is the case of *Fagus sylvatica* in northeastern Spain (Jump and Peñuelas 2005). In the absence of migration or evolutionary adaptation, the ability of populations to persist in the new environment will depend exclusively on their phenotypic plasticity, i.e. their ability to alter their phenotype with environmental conditions. Although Mediterranean forests have shown relatively high levels of plasticity, being plastic has an important metabolic cost, and there are universal physicochemical constraints that prevent the ability of a species to simultaneously tolerate several stresses (Laanisto and Niinemets 2015). In this sense, the succession of disturbances can cause an important limitation to the adaptation of the species. For example, the regeneration of *Pinus nigra* after wildfires depends both on the existence of nearby unburned vegetation patches and on climate conditions in the years following the fire (Martín-Alcón and Coll 2016; Sánchez-Pinillos et al. 2018). Distinct sequences of disturbance events can cause vegetation transitions, with non-linear responses and tipping points, even if the recurrence of individual disturbances is moderate (Batllori et al. 2019). Therefore, the succession of fires and droughts could trigger massive failures in regeneration, leading to a change in the ecosystem towards a greater dominance of oaks. In the driest areas, the combined effects of several disturbances is likely to exceed the response capacity of organisms, leading to the extinction of some species and even triggering shifts in ecosystem state (from forest to non-forest) (Batllori et al. 2019), which entails a high risk of soil erosion, degradation, and desertification.

Measures to promote adaptation

Whenever the inherent adaptive capacity of species is not sufficient, or too slow, planned adaptation measures can be implemented to decrease the known risks, increase forest resistance, or promote its recovery capacity (Lindner and Calama 2013). Adaptation measures in the Mediterranean commonly seek to address the two main disturbances in the region: drought and fire, and can be classified into five categories (Vilà-Cabrera et al. 2018).

Reducing tree density through thinning has the triple effect of increasing the growth and value of the remaining trees while also improving their water status and reducing fire risks. In a climate change context, thinning can diminish interception losses and reduce stand transpiration, increasing the amount of available water, which is apportioned among fewer trees (Sohn et al. 2016b). Some studies report a direct reduction of drought-induced mortality of Scots pine for high thinning intensities (Giuggiola et al. 2013), and an increase in the resistance and recovery of growth following drought events (Martín-Benito et al. 2010; Sohn et al. 2013, 2016a), which may be particularly important in dry areas or under sever climate change scenarios (Ameztegui et al. 2017; del Río et al. 2017). However, there is also evidence to suggest that, under extremely dry conditions, tree mortality risk may be density-independent, as all the available soil moisture can be lost to evapotranspiration before it can be harnessed by trees (Dorman et al. 2015).

The **reduction of the understory cover** has mainly been applied with the aim of reducing the risk of fire propagation by breaking the vertical and horizontal fuel continuity. Reduction of understory cover can be achieved either through mechanical treatments, prescribed burning or by promoting understory grazing in forest areas (Vilà-Cabrera et al. 2018).

The promotion of **mixed-species stands** (at the species or genotype levels) can increase resistance and recovery capacity to extreme droughts (Pretzsch et al. 2013), higher temporal stability (Jucker et al. 2014; Sánchez-Pinillos et al. 2016; del Río et al. 2017), and reduce the risk of biotic and abiotic disturbances (Guyot et al. 2016; Jactel et al. 2017) and the maintenance of ecosystem service provision (Gamfeldt et al. 2013).

The **change in species or genetic composition** seeks to replace the maladapted species or populations with species or genotypes better adapted to the forecasted climate conditions, and can include (i) assisted population migration (i.e. the active relocation of well-adapted populations of a given species within its current range); (ii) assisted range expansion (relocation of a species to an area adjacent to its current range); and (iii) assisted species migration (i.e., the displacement of a species beyond its current range, where the future climate is expected to be suitable for its development) (Williams and Dumroese 2013). Although the effectiveness of these practices seems apparent, there are still many doubts about their risks and consequences on the host environment, especially in the latter sense. At present, there is no consensus on their suitability, and they generate significant rejection both by a large part of the scientific community and by the general population (Lawler and Olden 2011), and in the Mediterranean they have not yet been applied beyond small-scale scientific experiments (Martín-Alcón et al. 2016).

The promotion of the **spatial heterogeneity of the landscape matrix** has mostly been advocated as a way to reduce the impacts of fire by slowing or preventing its expansion and allowing for greater effectiveness in firefighting (e.g., minimizing total burnt area) (Loepfe et al. 2012; Regos et al. 2016). A heterogeneous landscape also allows for the coexistence of different habitats (forests, open areas, etc.) each with different goals and providing different services. Moreover, greater heterogeneity can also contribute to enhancing gene flow and natural species migration, provided that enough corridors are available (Saura et al. 2018; Vilà-Cabrera et al. 2018). Fire risk management can also be achieved through the promotion of particular land covers/uses that reduce the risk of intense crown fires (e.g., fagaceae vs. conifers: (Moriondo et al. 2006).

However, the socio-economic adaptive capacity of the Mediterranean forest sector also has to face several constraints derived from the low economic incomes of many Mediterranean forests, mainly due to low fertility and water limitation, the lack of a developed road network and the limited implementation of technological advances (e.g., harvesting machinery), which results in a large part of Mediterranean forests not being managed at all (Lindner et al. 2010; Lindner and Calama 2013), limiting the capacity of forests to adapt to climate change. Moreover, managing forests to increase adaptive capacity can lead to trade-offs with other ecosystem functions and with biodiversity (Vilà-Cabrera et al. 2018).

4.3.3.2 Mountain ecosystems

Implementation of effective adaptation measures depends on the availability of human resources and expertise. However, the knowledge base about Mediterranean mountains varies significantly.

Enhancing connectivity is a key measure to facilitate expected range shifts (Keeley et al. 2018) which in mountain areas may be achieved by “building” linear and latitudinal corridors and taking advantage of the river network. In addition, due to interconnected risks, wider spatial frameworks are necessary, for instance at the watershed level, since upstream changes influence downstream. There are still pristine areas in many Mediterranean mountains that sustain a diversity of plant and animal species. However, this role is impeded by ongoing human activities and most importantly, climate change has pointed to the need to design a flexible reserve system along with conventional ex situ conservation measures. Such as reserve system may place emphasis on the permeability of the intervening landscape matrix, dispersal corridors and habitat networks (Jongman and Pungetti 2004; Watts and Handley 2010). Mountains have played a refuge role in geological history and to a certain extent they retain this role today, with many of their endemic species surviving in places located in such refugia (Vogiatzakis and Griffiths 2008; Vogiatzakis 2012). This is at the core of climate-wise connectivity as proposed by recent studies (Keeley et al. 2018). Mountain ecosystems may prove more resilient since upward migration of lower zone species will be conditioned by topography and geomorphology i.e., habitat suitability (Kazakis et al. 2007).

4.3.3.3 Drylands and shrublands

Plants exhibit a variety of mechanisms to avoid (e.g., annual life-cycle) or to tolerate drought (e.g., perennial shrubs), and to deal with disturbance pressures, such as fire and herbivory (Noy-Meir 1973; Davies et al. 2012). Dryland biodiversity interacts with abiotic factors to determine ecosystem functioning (e.g., productivity, nutrient fluxes) and resilience (i.e., the ability to return to a previous state after disturbance), both of which are critical to ensuring the provision of ecosystem services (MEA 2005). Climate change predictions point to an overall increase in aridity and in the variability of precipitation distribution in drylands (Dai 2013). Climate change is therefore expected to further reduce productivity over time.

Topography creates contrasting microclimates, especially between northern and southern slopes in drylands, which result in clear differences in tree cover patterns in the landscape. These local-scale differences in tree cover patterns may result from limitations occurring at different plant development stages, such as seed germination, seedling establishment, tree growth rate and survival, all crucial for the maintenance and expansion of plant populations through natural regeneration. Leaf physiological performance was similar under contrasting microclimatic conditions. However, in areas with higher Potential Solar Radiation, tree age and density were significantly reduced. These results suggest that microclimatic differences on southern slopes with high Potential Solar Radiation are limiting for germination and sapling establishment. Thus, forest regeneration, restoration and nature conservation practices aiming at increasing forest resilience in Mediterranean dryland climates should account for the importance of microclimate in defining the niche of seedlings and adult trees (Príncipe et al. 2019).

Shrub encroachment was largely predicted by topo-edaphic factors in Mediterranean dryland ecosystems subject to conventional low-intensity land use composed of savanna-like holm oak woodlands, along with a regional climate gradient (Nunes et al. 2019). Management strategies to reduce encroachment therefore need to take these drivers into account for efficient forecasting and higher cost-effectiveness. Climate had a stronger effect on a set of functional traits involved to a limited extent in shrub encroachment, related to flowering and dispersal strategies. These results suggest that climate change might not greatly impact shrub encroachment in the Mediterranean Basin, but may affect the functional structure and reduce the functional diversity of plant communities, thus affecting ecosystem functioning (Nunes et al. 2019).

Drylands are very susceptible to the effects of climate change due to water stress. One possible climate change adaptation measure is the construction of lakes to increase water availability for drinking and irrigation (food production) and decrease fire risk. These lakes can also increase local biodiversity and human well-being. However, other non-target services such as carbon (C) storage, water purification, and sediment retention might also change.

An evaluation of the trade-offs on non-targeted ecosystem services due to lake construction in drylands was carried out by Santos et al. (2018). This was done using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) modelling tools, comparing a Mediterranean area located in southwestern Europe, with and without artificial lakes. Results showed that the construction of artificial lakes caused an increase of 9.4% in carbon storage. However, the resulting increase in agricultural area decreased water purification and sediment retention services. This could diminish the lifespan of the lakes, changing the initial beneficial cost-benefit analysis on lakes as adaptation measures to climate change. As a global measure for mitigation and adaptation to climate change strategy, we consider lake construction in drylands to be positive since it can store carbon in sediments and reduces the vulnerability to water scarcity. However, as a general recommendation, and when built to support or increase agriculture in semi-arid landscapes, we consider that lakes should be supplemented with additional measures to reduce soil erosion and nutrient leaching, such as (i) locating agricultural areas outside the lake water basin, (ii) afforestation surrounding the lakes, and (iii) adopting the best local agriculture practices to prevent and control soil erosion and nutrient leaching.

4.3.3.4 Agriculture and pasturelands

Transformational adaptation in agriculture has been described as a redistribution of at least a third of the production factors and/or production output in a 25-year timeframe, which mostly involves qualitative changes in inputs and outputs (Vermeulen et al. 2013). Transformational adaptation can result into a radical change to the area of production, to the main crops or production types (e.g., shift from animal to crop productions, abandonment of a specific type of farming, reclaim of abandoned lands, shift from rain-fed to irrigated agriculture, from nomadic to settled grazing systems, from conventional to organic farming systems). Vermeulen et al. (2018) also pointed out that the success factors and drivers of positive transformational adaptation include changes in governance in favor of disadvantaged stakeholders. They also showed that the capacity of producers, processors and consumers to adapt is highly context-sensitive and depends on public policy, market drivers and cultural values.

Farmers’ long-term responses and investments in adaptation are constrained by barriers: (i) climate change signals can be biased by the perceptions of farmers (Nguyen et al. 2016), (ii) the projections are uncertain, (iii) climate change communication is difficult and often does not result in behavioral change (Wise et al. 2014), (iv) timescales for farm planning are relatively short and other priorities take precedence, (v) there are expectations of technology being able to cope with the negative effects of climate change, (vi) scientific knowledge and tools developed in agriculture rarely support long-term strategic decisions (Robertson and Murray-Prior 2016). Also, short-sighted state policies can increase the vulnerability of farmers to climate change and constrain their adaptive capacities (Turhan et al. 2015).

Opening new spaces for learning and generating enabling contexts through translating systems thinking into practice are fundamental steps for raising awareness about adaptation actions in a climate change world (Ison 2010). Vermeulen et al. (2018) examined several case studies on transformational adaptation in agriculture and showed how in practice it often emerges from a disorganized combination of responses from multiple actors to external pressures where climate change can only be an indirect driver. However, farmers rely on their experience to plan their practices, which will become obsolete if climate change occurs too quickly. They design their activities on the basis of a perceived probability distribution of their known drivers and performance indicators but, under climate change, probability distributions are shifted in mean, variance or both and are unknown, thus resulting in increased frequency of unexpected events. Dono et al. (2016) showed how these shifts between actual and expected probabilities can result in winners and losers in an affected area, depending on the type of farming system under the same expectations of climate pressures in a Mediterranean context.

Adaptation in agriculture requires customized support to the choices of any specific farm type in a given environmental situation or context, which is a challenge both for science and policy. The variety of environmental and socio-economic contexts and agricultural systems across the sub-regions of the Mediterranean Basin and across farming types within sub-regions, generate a great diversity of needs, adaptation strategies and have site-specific implications on biodiversity and the ecosystem services of agroecosystems.

Adaptation strategies and plans are being adopted at the continental, country or local level under the Cancun Adaptation Framework of the UNFCCC[7]. However, by December 2018, among the nine Mediterranean countries of the EU, only three (Cyprus, France and Spain) had already adopted a national adaptation plan, while eight had adopted a national strategy. Such plans should create the enabling environment for "last-mile" adaptation to occur, but moving from planning to implementation is a challenge because of the difficulties found in addressing capacity constraints, securing adequate financing and measuring the success of actions (Mullan et al. 2015). The following cases provide some examples of the specific adaptation needs of agriculture and pastureland systems in the Mediterranean context.

At present, cereal production is well below potential in southern Europe. Schils et al. (2018) demonstrated that the yield gap between actual and potential production would require good agronomy for sustainable intensification and thus increase the self-sufficiency food production of the entire Mediterranean area (Vermeulen et al. 2013). The self-sufficiency ratio in northern Africa is lower today than it was in the past as a consequence of demographic expansion, and is also resulting in low stability, not only in low GDP countries (Luan et al. 2013). Under pronounced drying trends documented by recent assessments, particularly in northwest Africa, a strategic objective is to move from maximized to stabilized production (Schilling et al. 2012). This can also be achieved through improved climate-proof agronomic practices such as the incorporation of crop residues combined with supplementary irrigation, where available (Benlhabib et al. 2014; Jacobsen 2014).

Adapting dryland agriculture to climate change in the Mediterranean requires substantial investments in plant breeding for heat and water stress tolerance and to increase yield and quality under conditions of high CO_2 concentration (Asseng and Pannell 2013). The same authors suggest investments in new species and cultivars of perennial plants. Such investments should be coupled with improved seasonal forecasting, which would enable farmers to make timely decisions about agronomic practices, thus improving resource use and crop yield. However, climate resilience is currently not receiving the necessary attention from breeders, seed and wheat traders, and farmers, while there are clear signals of declining resilience, at least for durum wheat, also in Mediterranean countries, including Spain (Kahiluoto et al. 2019). This latter assessment revealed that current breeding programs and cultivar selection practices do not sufficiently prepare for climate uncertainty and variability by applying a variety of responses to the same climate pressures by different wheat cultivars. In the case of barley, a pivotal crop in the Mediterranean area, Cammaron et al. (2019) demonstrated that, besides plant breeding, shifting sowing dates and improving soil organic carbon are viable adaptation strategies to mitigate the expected negative impacts of a future drier and warmer Mediterranean climate on barley grain yield.

Wheat protein yield gains are expected to be lower and more variable in most rain-fed low-input cropping regions, where nitrogen availability limits growth stimulus from elevated CO_2 (Asseng et al. 2019). This is particularly true for North African countries, where food demand is increasing due to population increase (Schils et al. 2018). Introducing wheat genotypes adapted to warmer climate may not result in increased protein production. Therefore climate adaptations leading to stabilized grain yield could not always be positive in terms of grain quality (Asseng et al. 2019). Adaptive pathways for cereal

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[7] https://unfccc.int/topics/adaptation-and-resilience/workstreams/national-adaptation-plans

productions also have impacts on integrated governance aiming to yield stability, such as the implementation of national action plans and policies to regulate and provide incentives for increasing diversity in crop responses to climate uncertainties (Kahiluoto et al. 2019).

Legumes represent a strategic resource for sustainable intensification of agricultural systems and climate change adaptation in the Mediterranean Basin. In addition to serving as a fundamental source of high quality food and feed, legumes contribute to net nitrogen inputs in cropping systems at low N_2O emissions and contribute to net soil carbon sequestration (Volpi et al. 2016; Stagnari et al. 2017). The environmental services provided by legume cultivation are still undervalued, while new opportunities for yield improvement are arising from the ongoing development of cost-efficient genome-enabled selection procedures, enhanced adaptation to specific cropping conditions and more thorough exploitation of global genetic resources (Annicchiarico 2017).

The cropping systems for the production of bioenergy and biomaterials are assumed to occupy part of the residual agricultural land abandoned in the past 50 years, but this will not be sufficient to meet the increased bioenergy demand associated with climate change energy policies in European countries (Cosentino et al. 2012). These crops may also find a strategic position in Mediterranean cropping systems to reclaim polluted arable land from industrial or mining wastelands (Fagnano and Fiorentino 2018).

Increased irrigation water efficiency and the design of climate-friendly agro-ecosystems are key adaptation strategies for Mediterranean agriculture, in particular for countries such as Algeria, Libya, Israel, Jordan, Lebanon, Syria, Serbia, Morocco, Tunisia and Spain, which are at high risk of not being able to meet future irrigation needs (Fader et al. 2016). A range of adaptation strategies are being studied or put in place either to store more water in hot and arid environments (e.g., with managed aquifer recharge) (Salameh et al. 2019), use and recycle non-conventional water sources (Ait-Mouheb et al. 2018; Elkiran et al. 2019), desalinate seawater (Stanhill et al. 2015) or improve irrigation efficiency (Tarjuelo et al. 2015; El Jaouhari et al. 2018). All these strategies have some potential side effects in terms of energy requirements (Rodríguez-Díaz et al. 2011), GHG emissions, high capital investments and social acceptance (Daccache et al. 2014; Chartzoulakis and Bertaki 2015). However, in arid zones, these are often the only alternatives to achieve sustainable agricultural intensification. Some solutions, like desalination, should be considered only where there is evidence that the natural recharge available in surface and underground storage might become limiting considering the economic and environmental dimensions of sustainability (Stanhill et al. 2015).

Supplementary irrigation of rain-fed crops is also crucial for increasing the productivity of traditional Mediterranean rain-fed cropping systems, including winter cereals or perennial crops such as olive and vineyards (Fraga et al. 2012; Tanasijevic et al. 2014). However, the introduction of new technologies for irrigation on traditional rain-fed or irrigated cropping systems has many systemic implications in the environmental, socio-cultural, institutional and economic domains (Ortega-Reig et al. 2017).

Increased water and soil salinity is also a threat for future Mediterranean cropping systems, particularly in coastal areas (Maggio et al. 2011; Pittalis et al. 2016). Adaptation strategies include the introduction of salt tolerant crop species for which there is increasing consumer demand. For example, there is increasing interest in quinoa germplasm and production in the Mediterranean Basin, in relation to its tolerance to salinity and water stress, high water use efficiency and the increasing demand of gluten-free food (Hirich et al. 2014; Lavini et al. 2014; Mahmoud 2017; Noulas et al. 2017).

Combined agro-ecological approaches to climate change adaptation in organic horticulture is suggested by Diacono et al. (2016), following long-term field experiments that showed that such cropping systems can sustain the yield of cash crops in rotation, in spite of changes in temperature and rainfall.

Pasturelands and rangelands will face multiple threats from expected drier and warmer climate in Mediterranean countries. These farming systems rely heavily on natural resources on marginal land, often characterized by shallow and oligotrophic soils, with low water holding capacity. Highland pastures and their biodiversity is being threatened by loss in biodiversity due to climate change (Dibari et al. 2015). Dono et al. (2016) showed that near future climate change will result in losses for rain-fed grazing systems, mainly because of the shift in the probability distribution of rain-fed pasture and hay crop production due to drier springs caused by reduced rainfall and higher evapotranspiration, resulting in higher costs for purchasing external feeds or renting more land. Adaptation strategies range from incremental to transformational strategies in this case. Silanikove and Koluman (2015) project an overall negative impact, but a positive role of dairy goats in adaptation to global warming when compared to dairy cows, given their higher tolerance to heat stress.

The savanna-type pastoral vegetation of the dehesa in Spain, montado in Portugal or pastures with scattered cork-oak trees in Sardinia and elsewhere in northern African countries are considered a multifunctional resource that can support adaptive responses to climate change and the provisioning of multiple ecosystem services (den Herder et al. 2017; Castro and Castro 2019). Mediterranean agro-silvopastoral systems generate unique habitats for plant and microbial diversity, resulting in a wide range of services such as forage, wood and non-wood products, soil organic carbon sequestration and landscape cultural values (Bagella et al. 2013; Seddaiu et al. 2013, 2018; Rossetti et al. 2015; Tardy et al. 2015; Torralba et al. 2016; Garrido et al. 2017). Adaptation strategies in pastoral systems based on wooded pastures include actions that can prevent the threats of degradation due to abandonment (e.g., wildfires, loss of cultural landscape and heritage, increased drought stress) or intensification (e.g., lack of tree regeneration) (Garrido et al. 2017; Rolo and Moreno 2019). Given the complexity of the factors driving the sustainability of agro-silvopastoral systems, adaptation strategies should be designed and implemented through systemic and integrated approaches and not by just targeting a specific service or pastoral activity (Hernández-Morcillo et al. 2018). However, more attention should be devoted to these agro-silvopastoral systems, as they are currently overlooked by rural development policies in Europe, while agroforestry systems can effectively contribute to maximizing the productivity of marginal land (Mosquera-Losada et al. 2018). Agro-silvopastoral systems in the Mediterranean area are under threat because the income of farmers that contribute to their maintenance does not acknowledge the many ecosystem services they provide (Fagerholm et al. 2016; Rodríguez-Ortega et al. 2018).

The adaptive capacity of grazing systems in the Mediterranean depends on local contexts, with contrasting trends in northern and southern countries, rain-fed or irrigated conditions. A general trend towards increased specialization and related environmental risks is occurring almost everywhere, which is in contrast with the need for increased resilience to climate pressures and reduced environmental impacts (Rodríguez-Ortega et al. 2017). Adaptive development strategies include enhancing the spatial dimension of grazing systems through increased animal mobility, increased feeding self-sufficiency and integration of crop-livestock integration at the regional and sub-regional levels (Alary et al. 2019). These livestock farming systems would also respond to the ongoing change in human dietary recommendations, which is one of the drivers of the meat crisis (D'Silva and Webster 2017). However, this may result in different environmental impacts in terms of greenhouse gas emissions, eutrophication and land use in different regions, depending on the income level (Behrens et al. 2017).

Adaptation strategies are more complex for farmers who have made large long-term investments following market pressures and productivity objectives. Dairy cattle farming systems rely mostly on irrigation water and are threatened by the increasing frequency of heatwaves, to which highly productive cows are very sensitive (Lacetera 2019). These farming systems are facing uncertainties caused by fluctuating world feed prices, climate, market and environmental normative pressures (e.g., nitrate vulnerable zones), which are gradually squeezing their marginal net returns (Dono et al. 2016). Adaptation strategies are constrained by multiple pressures and are often based on crop and animal diversification combined with improved animal feeding and genetics (Rojas-Downing et al. 2017; Henry et al. 2018).

4.3.3.5 Freshwater ecosystems

Successful adaptation measures need to follow a large-scale hydrological approach to determine the origin of variations, which are usually related to human pressures, and to provide further strategies for environmental management (Menció and Mas-Pla 2010). Conservation and restoration efforts traditionally carried out at a local scale need to be accompanied by land use and hydrological planning at a basin-wide scale in order to maintain stream ecosystem integrity and biodiversity (Bruno et al. 2014a, 2014b). In this sense, nature-based solutions can play an important role in maintaining freshwater biodiversity, and because of their multifunctionality, in providing critical ecosystem services (e.g., food provisioning, erosion regulation and cultural ecosystem services) (Balzan et al. 2019). Nature-based solutions (NbS) are defined by IUCN as "actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits"[8].

The Mediterranean Strategy for Sustainable Development 2016-2025 prioritizes national action to ensure that natural water resources are extracted sustainably. However, reliable methods to assess water demand are needed. Such methods assess the dynamics and determine the main drivers of each type of water demand, and project future water uses (Charlton and Arnell 2011; Collet et al. 2013; Griffin et al. 2013; Reynard et al. 2014), with interdisciplinary approaches that combine physical and human features and incorporate climate change impacts at the local scale (Grouillet et al. 2015). Local water management planning and adaptation strategies need to be improved and updated in order to attain future water security (Koutroulis et al. 2013) (more on human security related to water is highlighted in Chapter 5.3). Achieving integrated and sustainable water management will also require enhanced awareness of climate change effects and public demands for water-use efficiency and improved environmental quality (Grantham et al. 2013). Awareness of the practical implications of plausible hydro-climatic and socio-economic future scenarios will shift perceptions and preference towards a more sustainable model (Koutroulis et al. 2010).

The success of mitigation and adaptation policies to restore sustainability depends on implementation efficiency at local level, where awareness and perception often pose barriers (Betzold 2015; La Jeunesse et al. 2015). Communicating relevant and targeted climate change information to stakeholders and decision makers is crucial for gaining commitment in the field. The projected water scarcity in the region highlights the important role for development and deployment of water conservation technologies and practices (Hejazi et al. 2014) and the need for strategic resource planning from global to regional and local scales (Koutroulis et al. 2013). Stakeholders, the beneficiaries of ecosystem services from river bodies and landscapes, play a key role in interpreting the impact of climate change on water resources and usage. A mixed methodology based on a transdisciplinary approach and the involvement of academia, policy-makers, and local experts is suggested. Many physical models on the impacts of climate change and on water scarcity exist but approaches that are transdisciplinary with input from local stakeholders and interpretation of intermediary results are limited (Rault et al. 2019). Improving the understanding of ecosystem responses to multiple stressors and defining measures to improve the ecological status of water bodies are needed and sought by the WFD (Menció and Mas-Pla 2010).

Preserving the natural flow variability of rivers and streams is key in sustainable environmental management plans in the Mediterranean (Menció and Mas-Pla 2010) and critical to the long-term conservation of their unique biodiversity (Cid et al. 2017). The high variation in hydrological regimes in the region, however, tends to exacerbate the magnitude of negative responses to anthropogenic and climate impacts. For example, land use changes promote longer dry season flows, concentrating contaminants, allowing the accumulation of waste, algae, and plants, and fostering higher temperatures and lower dissolved oxygen levels, all of which may extirpate sensitive native species. Exotic species often thrive in rivers altered by human activity, further homogenizing river communities worldwide. Future research should rigorously evaluate the effects of management and restoration practices on river ecosystems, determine the cause–effect pathways leading from human disturbances to stream biological communities, and incorporate analyses of the effects of scale, land use heterogeneity, and high temporal hydrological variability on stream communities (Cooper et al. 2013).

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[8] https://www.iucn.org/commissions/commission-ecosystem-management/our-work/nature-based-solutions

The surface water-groundwater relationship is of major interest in the characterization of human pressures on stream hydrological dynamics and the ecological quality of Mediterranean reaches (Menció and Mas-Pla 2010). The ecological status of streams depends on an equilibrium between hydrological processes and biological dynamics. Water discharge is the main requisite for a rich riparian habitat and impacts upon aquifer water storage and on base flow generation have a significant effect on stream biology. Tools for managing these systems, such as those related to biomonitoring, climate change, and conservation, must be tailored to the seasonal and inter-annual variability of these systems.

Much of the ecological surface water monitoring under the WFD focuses on the assessment of biological structure, hydromorphological elements and chemical and physicochemical elements, all of which represent important information about ecosystem condition (Balzan et al., 2019). It is assumed that good ecological and chemical status have a positive impact on the capacity of ecosystems to provide ecosystem services and benefits to human well-being (Grizzetti et al. 2019). The link between waterbody condition, and ecosystem function and services has seldom been explored in detail, with the implication of water management for ecosystem services being either implicit or overlooked (Vlachopoulou et al., 2014). Thus, tools that work at the ecosystem function level, connecting ecosystem condition to services, are required especially in climate change scenarios where ecosystem condition is expected to change with the climate and other interacting drivers (Chapter 2). There are a number of studies from the Mediterranean Basin that demonstrate that such links can be made in practice, as demonstrated by the study by Acreman et al. (2017), which shows how the implementation of beach restoration is associated with ecological recovery and recreational ecosystem services.

Wetland management and conservation in semi-arid Mediterranean areas is necessary because they have been highly impacted by agriculture. This can be done with pressure and state indicators at landscape and wetland scales that reflect the status, condition, and trends of wetland ecosystems. Ortega et al. (2004) developed an ecological integrity index with 12 indicators (5 at the catchment scale and 7 at the wetland scale) based on the relationship between pressures from anthropogenic activities and the ecological state of wetlands and their catchments, integrating environmental, biological, economic, and social issues. Overall, a wide riparian zone acts as a buffer for wetlands, diminishing the effects of intensive agriculture. Provisioning services are more relevant in normal and wet years, while regulating service water purification provides higher benefits in dry years, when threats to water quality are increased because of a decreased dilution capacity (Terrado et al. 2014). Protecting water towers in semi-arid regions expected to experience dramatic changes is essential to ensuring water provisioning in dry years. However, the protection of water resources is not sufficient if consumption rates continue or increase in the future. Actions should be planned to enhance the provision of regulating services (Terrado et al. 2014). Overall, in semi-arid basins under continuous human impact, hydrological ecosystem services are very sensitive to climate extremes, and service supply and demand areas are usually spatially and temporally decoupled. Both aspects are relevant and need to be considered in basin management in semi-arid regions (Terrado et al. 2014).

Studies that consider the respective influences of climate, land cover (forest cover dynamics in hydrological processes) and water withdrawals on water availability (e.g., Chauvelon et al. 2003; Varela-Ortega et al. 2011) are required for proper adaptation measures but are still scarce, since the required data are often unavailable or not easily accessible over long periods (Sivapalan et al. 2003). Databases at large spatial and temporal scales are key to understanding the variability of hydrological systems, and in providing water managers with science-based decision-making support information (Hannah et al. 2011). However, such databases are still too scarce in the Mediterranean, despite efforts to maintain and develop data networks at the regional and global levels. At this scale, the physical and human characteristics of catchments, especially in the Mediterranean, are extremely heterogeneous since they encompass extreme contrasts in terms of climatic, topographic and geological characteristics, population distribution and water uses and are therefore difficult to define and grasp (Collet et al. 2014). The climate variability of Mediterranean river basins also makes it difficult to describe general patterns which explain and predict the relationships between runoff, erosion and sediment transport

(López-Tarazón et al. 2010; de Vente et al. 2011) and this is even more complicated in rivers affected by regulation. Time-series on sediment transport (Batalla and Vericat 2009) and on lake ecosystems, especially those predating anthropogenic influences, are very scarce (Papastergiadou et al. 2007). Long-term data sets are also important for understanding the interactions among native species and introduced species. These are particularly valuable in understanding the influence of extreme events such as drought and floods (Magalhães et al. 2007; e.g., Bêche et al. 2009). Such studies, particularly in regulated systems, help guide flow recommendations to benefit native species but have only begun, although their value is already clear (Kiernan et al. 2012; Resh et al. 2013). Long-term data sets that can reveal trends need to exceed the sub-decade scale for sustainable management of Mediterranean-climate streams and rivers (Cid et al. 2017).

Box 4.1: Bio-indicators for the assessment of changes in Mediterranean marine ecosystems

Awareness of recent changes in ecological conditions in many seas has fostered a need to assess increasing anthropogenic pressures and their consequences on sediment and water quality, and to suggest measures to reverse this trend. In this context, the European Commission has implemented the Water Framework Directive (WFD, Directive 2008/56/EC) with the aim to obtain (or to maintain) a "good status" for all European waters by 2015. To support this, a large number of monitoring tools have been developed, including several bio-indicators such as phytoplankton, macro-algae, seagrass, angiosperms, fish faunas and soft substrate benthic invertebrate fauna, which are benthic foraminifera.

Phytoplankton and zooplankton are ecologically important groups in most aquatic ecosystems and have been an important component of biological monitoring programs in the Mediterranean (Abboud-Abi Saab et al. 2008, 2012; Tunin-Ley et al. 2009; Gharib et al. 2011; Tunin-Ley and Lemée 2013; Abo-Taleb et al. 2016; Ouba et al. 2016; Abboud-Abi Saab and Hassoun 2017). The genus *Neoceratium* (planktonic dinoflagellates) in the NW Mediterranean is known to be particularly sensitive to water temperature, and is responsive to global warming (Tunin-Ley et al. 2009; Tunin-Ley and Lemée 2013). Moreover, the WFD mandates the use of biological quality element (BQE) phytoplankton to assess the ecological status of coastal and transitional water bodies. Alternatively, Camp et al. (2016) propose a methodology to assess water-quality based on the use of chlorophyll-a (Chl-a), as a proxy of phytoplankton biomass.

For soft-bottom marine habitats, macrofauna is traditionally used as a bio-indicator, and a wide range of different biotic indices have been developed (Borja et al. 2016). The use of meiofauna, occurring in higher densities, is less developed. Among these, benthic foraminifera appear particularly suitable for bio-monitoring in the Mediterranean (Barras et al. 2014; Jorissen et al. 2018). The abundant and diverse benthic foraminifera faunas in the Mediterranean react rapidly to environmental changes such as organic pollution, eutrophication and oxygen depletion. These characteristics led to the development of a standardized biotic index based on foraminifera (Jorissen et al. 2018).

For the coralligenous, several indices of its health status have been suggested. One is the INDEX-COR approach, based on long time series of photographic sampling, standardized and used as a large spatial comparison tool (Sartoretto et al. 2017). This type of index integrates the sensitivity of different coralligenous taxa to organic matter and sediment deposition, the observable taxonomic richness and the structural complexity of the benthic assemblages. With these approaches, the health status of this complex ecosystem can be assessed without invasive or directly impacting methods. When deeper areas are considered in the coralligenous assemblages (e.g., from 30-40 to 200 m depth), Remotely Operated Vehicle (ROV) approaches may be useful (Rossi et al. 2008). The Mesophotic Assemblages Ecological Status (MAES) has been suggested as a tool for conservation and management procedures (Cánovas-Molina et al. 2016). The MAES index is based on community structure, condition of the erect species and visible human impacts (Cánovas-Molina et al. 2016). A combined biomarker index can also be considered as a medium-long term monitoring approach. In selected populations of representative sessile species of the coralligenous, activity (e.g., polyp expansion), growth, reproductive output, stable isotopes, biochemical balance (protein-carbohydrate-lipids), fatty acids and C/N ratio may be used to estimate the nutritional condition and health status of populations or entire communities, considering the biology and ecology of each species (Rossi et al. 2017b).

For large vertebrates, cetaceans and seabirds are widely regarded as reliable indicators of the health of marine ecosystems due to their position near the top of the marine food web, conspicuous nature, and reliance on marine resources (Durant et al. 2009; Bossart 2011; Schwacke et al. 2013; Fossi and Panti 2017; Fossi et al. 2018). Some cetaceans and seabirds are reported as sentinels or indicators for the state of marine ecosystems because they are globally subject to multiple stress factors, such as the bioaccumulation of contaminants, infectious diseases, non-indigenous species, food depletion, and climate change (UNEP/MAP 2012; Poloczanska et al. 2013). The advantage of using cetaceans as sentinels is that they have physiology and/or diets similar to those of humans, so they can indicate earlier potential adverse health effects (Schwacke et al. 2013). For fish, red mullets (*Mullus barbatus* and *M. surmuletus*) have been widely used as quantitative bio-indicators of chemical contamination (Porte et al. 2002; Storelli and Marcotrigiano 2005; Martínez-Gómez et al. 2017). For instance, the recent study by Cresson et al. (2014) confirmed that red mullets are efficient bio-indicators of Mercury (Hg), one of the main chemicals currently altering Mediterranean ecosystems.

End Box 4.1

Box 4.2: Urban biodiversity in the Mediterranean Region

Further to being a hotspot of biodiversity, the Mediterranean region is also one of the most densely urbanized areas in the world (FAO and Plan Bleu 2018). The overall population in the region grew by 190 million people between 1970 and 2010, while the urban population increased by 163 million, with more 74% of population growth concentrated in the countries of the southern part of the Mediterranean (UNEP/MAP 2012). Despite substantially lower growth rates of urban populations in European countries of the Mediterranean, surfaces occupied by urban settlements have also increased considerably due to tourism and decentralization of population from high density core cities towards low density residential areas and along the coastlines, as for instance in Barcelona (Domene and Saurí 2007) or Athens (Cecchini et al. 2019).

Also, patterns of land use changes differ between southern and northern parts of the region. In European countries, urbanization has been accompanied by abandonment of agriculture since the post war period and extended mainly onto different types of cultivated areas, and onto shrubland and/or herbaceous areas (García-Nieto et al. 2018), leading to substantial losses of biodiversity and of agricultural area, while shrubland and forest land increased (Domene and Saurí 2007; FAO and Plan Bleu 2018; García-Nieto et al. 2018).

The growth of north African peri-urban areas in that period occurred in parallel with an increase in irrigated arable land, permanent crops, complex cultivation patterns and shrublands and/or herbaceous areas and pastures, at the expense of non-irrigated arable land and forest, both around peri-urban areas, as well as at the national level. In southern Mediterranean countries, environmental change contributed, for example, to a rural exodus in Morocco between 1980 and 1990, and in Algeria and Tunisia in 1999 (García-Nieto et al. 2018).

Consequences for biodiversity and ecosystem services in urban areas

Consequences for biodiversity and ecosystem services available for urban areas differ accordingly. In most cases, urban areas replace former agricultural land, but, while in the southern part of the Mediterranean this coincides with an extension of cultivated areas outside urbanized areas, in most European countries the contemporary abandonment of agricultural areas leads to an increase in shrubland and uncultivated areas and contributes to increasing vulnerability of surfaces, for instance with the threat of wildfires along the wildland-urban interface (San-Miguel-Ayanz et al. 2013; Lafortezza et al.

2015; Xanthopoulos 2015). and desertification (Salvati et al. 2015). Abandonment of agricultural lands also leads to the loss of cultural landscape management practices. For example, traditional Mediterranean agricultural landscapes are in many places characterized by terraces and dry walls which represent small scale practices of erosion prevention (Cecchini et al. 2019). Increasing soil sealing in urban areas and connected infrastructure and abandonment of historic techniques of landscape management lead to increasing risks from flooding and landslides (Salvati et al. 2015; García-Nieto et al. 2018).

The remaining ecosystems are increasingly under threat with particular risks for rare plants depending on small patch ecosystems with highly localized distributions even though they may be protected (Vimal et al. 2012). Peri-urban areas nevertheless provide relevant services for urban populations, for instance the increase in areas of natural or semi-natural vegetation such as ecosystem fragments, reserves, nature parks, forests, and river banks, which house varying amounts of native species, potentially provide cultural services. Formal and informal forms of urban agriculture and horticulture (Domene and Saurí 2007; Cecchini et al. 2019) present an opportunity for integration into of diets for urban residents and provide both provisioning and cultural services, as these areas are also used for leisure activities (Domene and Saurí 2007; Cecchini et al. 2019; Palau-Salvador et al. 2019).

The expansion of peri-urban agriculture, for instance, olive cultivation for self-consumption or small-scale economic production, have counteracted land use change in the peri-urban areas of several cities in the Mediterranean Basin (García-Nieto et al. 2018; Cecchini et al. 2019). Olive landscapes have a high tolerance to pests and are characterized by a stable trend of economic production, and abundant insect fauna contributing to biodiversity conservation. They furthermore show a higher resistance to wildfires than other Mediterranean vegetation. The economic stability of olive oil production in small groves, like other small-scale agricultural areas and the status of protected natural areas (e.g., coastal woods in the case of Rome) in the green belt around cities contributes to the ability of these areas to form an efficient barrier against urban dispersion and reduce the impacts of soil sealing on the hydrological cycle and on ecosystem services loss (Salvati et al. 2015; Cecchini et al. 2019).

Urban biodiversity and ecosystem services

Despite the important provisioning and cultural services and their increasing popularity in many cities, urban gardening seems to be not as effective in protecting peri-urban areas from land use changes as in the case of peri-urban agriculture observed in Athens (Domene and Saurí 2007; Heywood 2017; Cecchini et al. 2019). This may be due to less stable legal position of such areas which are often the result of squatting on private or, more often, public land (Domene and Saurí 2007) and their less consolidated economic status. In Rome, managed spaces with cultivated vegetation such as parks and gardens represent a lower level of resilience against transformation despite their importance for cultural and regulating services (e.g., leisure and heat mitigation) for urban residents (Salvati et al. 2015).

In many urban areas of the Mediterranean, street trees provide important regulating services for human well-being by offering shade and reducing heat impacts during summer due to their evaporation rates and the albedo created by foliage (Rana and Ferrara 2019). They also provide important cultural services as characterizing elements of Mediterranean urban landscapes (Heywood 2017).

Furthermore, urban wastelands and shrublands which are mainly colonized by weeds, ruderal plants and non-indigenous species, in many cases house considerable numbers of native plants and are potential places for valuable biodiversity (Heywood 2017). The importance of green spaces in urban areas is increasingly recognized by Mediterranean cities, which are increasingly engaging in urban green infrastructure projects, preserving remnants of biodiversity and natural areas within cities. These are expected to provide important regulating, cultural and provisioning services but there is a general lack of data on urban biodiversity in urban and peri-urban areas of the Mediterranean (Heywood 2017).

End Box 4.2

Box 4.3: Nitrogen deposition and ecosystems

Climate change contributes to an increase in dry deposition of nitrogen and increases the negative impacts of excess atmospheric nitrogen on biodiversity (Oliveira et al. 2020). Reactive nitrogen (Nr) impacts vegetation through direct foliar damage, eutrophication, acidification, and susceptibility to secondary stress depending on the nitrogen form and concentration (Krupa 2003). Grassland, heathland and forest ecosystems are recognized as habitats vulnerable to Nr in Europe (Dise et al. 2011). In Spain, natural grasslands, particularly in the northern alpine area, were found to be the most threatened habitat followed by mountain ecosystems (García-Gómez et al. 2014). At least 14% of the Natura 2000 sites in western Iberia are at risk of eutrophication (Oliveira et al. 2020).

It is not yet clear if different wet-deposited forms of Nr (e.g., nitrate, NO_3 − versus ammonium, NH_4^+) have different effects on biodiversity. However, gaseous ammonia (NH_3) can be particularly harmful to vegetation. The highest relative risk of biodiversity change in Natura 2000 sites due to NH_3 pollution in Portugal was found to be in peats, mires, bogs, and similar acidic and oligotrophic habitats (most located in the northern mountains), whereas in the Atlantic and Mediterranean climate zone (coastal, tidal, and scrubland habitats) they were deemed the least sensitive in Portugal (Pinho et al. 2018).

Exceedance of critical loads for nitrogen is linked to reduced plant species richness in a broad range of European ecosystems (Dise et al. 2011). Experimental evidence shows that species richness and abundance resulted in larger declines with greater amounts of annual N addition including in semi-arid areas (Midolo et al. 2019). Reductions in the abundance of individual species were greater for N-sensitive plant life-form types (legumes and non-vascular plants) (Ochoa-Hueso et al. 2014, 2017; Midolo et al. 2019).

Several conservation plants (e.g., orchids and carnivorous) and cryptogams are naturally adapted to low environment N supply. Thus, increasing Nr alters the natural ecological balance. This results in the loss of the most sensitive species, which are often a priority for protection, and their replacement by non-indigenous or other opportunistic species that prefer high rates of nitrogen supply (Bobbink et al. 2010). Lichens and bryophytes are among the most sensitive organisms to N pollution at the ecosystem level (Cape et al. 2009), having a different response depending on their functional response group (Pinho et al. 2008, 2009, 2011, 2012b, 2012a; Jovan et al. 2012).

There are some clear examples of reductions in faunal diversity that can be linked to Nr deposition, but overall, our knowledge of faunal effects is still limited (Dise et al. 2011). Changes to above-ground faunal communities probably occur primarily through changes in vegetation diversity, composition or structure (Murray et al. 2006). The evidence strongly suggests that ecological communities respond to the accumulated pool of plant-available N in the soil. Thus, the cumulative load of enhanced Nr impacting an ecosystem is probably important (Stevens et al. 2011). Because of this response to cumulative inputs, it is likely that biodiversity has been in decline in Europe for many decades due to enhanced Nr deposition (Bobbink et al. 2010). Equally, full recovery in response to reduced Nr deposition is likely to be slow, especially in highly impacted ecosystems. In some cases, recovery may require management intervention.

End Box 4.3

Box 4.4: Mediterranean islands

Islands as laboratories

The high concentration of islands (> 10,000) is one of the features which contributes to the Mediterranean's unique character, placing the region within the richest in the world in terms of islands and archipelagos (Médail 2017). In addition to being biological laboratories, the largest of the islands are also the centers of many of the world’s ancient civilizations (Patton 1996). On these islands, the diversity of biogeography, geology and human settlement has produced exceptionally high numbers of biodiversity and endemicity, earning them a place in the global biodiversity hotpots list (Médail and Quézel 1997; Vogiatzakis et al. 2016; Médail 2017). In addition to the role as tertiary and glacial refuges, islands have also contributed to more recent plant diversification (Médail 2017). Biodiversity on islands display an insular syndrome due to abiotic conditions (nature of isolation, particular climate) and their own biogeographical history, and are characterized by unique specific assembly (with several endemics) and biotic interactions (de Montmollin and Strahm 2005; Blondel et al. 2010; Médail 2013, 2017; Schatz 2017).

Recent evidence of change

Despite their relatively small contribution to greenhouse gas emissions, Mediterranean islands are likely to be adversely affected by climate change, in synergy with ongoing land use changes.

- **Land-use change:** The landscapes of Mediterranean islands have evolved as the result of similar pressures to the mainland generated by socio-economic and political factors but amplified on a 'matrix' of limited space. Land use changes and associated impacts differ significantly depending on the size of the island and therefore demographics, as well as its popularity as tourist destination (Vogiatzakis et al. 2008). Therefore, to date there is no consensus on the trends of changes since islands seems to respond/behave individualistically (Vogiatzakis et al. 2008; Harris 2012). Recent land uses are associated with temporal and spatial shifts in land-use systems with polarization of land-use intensity, particularly on small/medium size islands (Tzanopoulos and Vogiatzakis 2011; Balzan et al. 2018).
- **Climate evidence:** Observed trends for winter (Nov–Feb) precipitation (mm/50 years) and summer (Jun–Sep) temperature (°C/50 years) for Mediterranean island regions during the second half of the twentieth century do not show a consistent climate pattern (Vogiatzakis et al. 2016).
- **Biological/ecosystems evidence:** Documented evidence on plant and animal phenology changes (Peñuelas et al. 2002; Gordo and Sanz 2010), range shifts (Lenoir et al. 2008) and changes in the function, structure and dynamics of ecosystems e.g., temporal mismatches among mutualistic partners) (Visser et al. 2004), species loss and changes (+/-) in species richness (Kazakis et al. 2007). Plant communities are steadily changing, such as orchids in Corsica (Vogt-Schilb et al. 2016), as well as specialized plant-insect interactions (pollination, seed dispersal) (Traveset and Riera 2005; Blondel et al. 2010; Stefanaki et al. 2015).

Climate change projections and islands

- **Current scenarios:** Projected trends from various climate models agree as far as the direction of change in precipitation and temperature regimes are concerned (Table 2.1 in Section 2.2) (Vogiatzakis et al. 2016).
- **Sea level rise:** For the Aegean archipelagos, Monioudi et al. (2017) assume mean sea-level rise of 0.5 m for RCP4.5 and predict that a storm-induced sea level rise of 0.6 m would result in complete erosion of between 31 and 88% of all beaches, at least temporarily.
- **Island representation and model resolution:** In most modeling studies (niche models or GCMs), islands are simply a subset of the Mediterranean (Araújo et al. 2006; Settele et al. 2008). As a result, neither distribution nor climate data have sufficient resolution to allow climate envelope models for most endemic island taxa (Henle et al. 2010).
- **Synergies with land cover changes:** Changes in land use (Settele et al. 2005) coupled with climate models predict modifications to species climate space (Settele et al. 2008) and islands are no different. Documented land cover changes related to urban/tourism development and increasing linear infrastructure are already having an impact on island biodiversity (Zomeni and Vogiatzakis 2014).

Vulnerability/resilience

The vulnerability of Mediterranean island systems to past and recent extinctions has been well documented (de Montmollin and Strahm 2005; Foufopoulos et al. 2011). A recent assessment of global imminent extinctions includes two Mediterranean Islands (Ricketts et al. 2005). Compared to the rest of the Mediterranean Basin, islands have always been more vulnerable to invasion by exotic species (Hulme 2004). Human activity will be the limiting factor which will determine the future of island flora. Islands are representative examples in the Mediterranean of the co-evolution of social-ecological systems intensified by the element of insularity and which are currently under threat and more susceptible to externalities due to (i) limited resources/space, (ii) administrative/political leverage, (iii) institutional capacities. While many of the islands have experienced demographic losses, concerning their permanent inhabitants, they have become principal tourist destinations (Ioannides et al. 2001) and islands are being faced with a key challenge of balancing economic benefits from ecosystem services delivery (.., tourism, agriculture) with environmental pressures.

Conservation and adaptation

On islands, opportunities for (human assisted) adaptation are limited. The lack of available space for wildlife to shift presents significant barriers to the natural adaptation of species and habitats. It also leads to more intense land-use conflicts, therefore increasing size of protected areas, and connections might be problematic. In an attempt to promote ‘climate-wise connectivity’ (Keeley et al. 2018), areas not (significantly) affected by climate change could act as refugia to species in the future. Many of the island endemic species of today have survived past climate changes in places in such refugia (Vogiatzakis and Griffiths 2008; Vogiatzakis 2012). In the case of island clusters, the suitable climate space might be a neighboring island, although there are many examples, particularly in the Aegean, where neighboring islands have different floras (Kallimanis et al. 2010). Therefore, biogeography may be more important than climate *per se* in interpreting species distribution patterns (Whittaker and Fernandez-Palacios 2007). Building a coherent ‘‘network’’ of protected areas across islands (e.g., in the Aegean) might provide solutions to safeguarding common biotic elements (species or habitats). At the government level, adaptation should include increased institutional capacity for innovation, the increase of monitoring activities, adaptive management, and promotion of inter-island collaboration (Kark et al. 2009). In the case of managing island ecosystems and their services, the key priorities for the future must be to:

- identify ecosystem service capacity hotspots and how they can be affected under climate change scenarios;
- identify ecosystem service demand and flows in hotspots and manage green infrastructure and co-created nature-based solutions to provide synergies for biodiversity conservation and ecosystem services for human well-being;
- manage protected areas for ecosystem service provision given the fact that lack of space is also problematic;
- evaluate tradeoffs and thresholds of ecosystem service provision and assess the impacts from demographics and tourism, as well as land use and climate change projections.

End Box 4.4

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