2 DRIVERS OF CHANGE

Coordinating Lead Authors:
Semia Cherif *(Tunisia)*, Enrique Doblas-Miranda *(Spain)*, Piero Lionello *(Italy)*

Lead Authors:
Carlos Borrego *(Portugal)*, Filippo Giorgi *(Italy)*, Ana Iglesias *(Spain)*, Sihem Jebari *(Tunisia)*, Ezzeddine Mahmoudi *(Tunisia)*, Marco Moriondo *(Italy)*, Olivier Pringault *(France)*, Gil Rilov *(Israel)*, Samuel Somot *(France)*, Athanassios Tsikliras *(Greece)*, Montserrat Vilà *(Spain)*, George Zittis *(Cyprus)*

Contributing Authors:
Giovanni Argenti *(Italy)*, Marie-Anne Auger-Rozenberg *(France)*, Ernesto Azzurro *(Italy)*, Corina Basnou *(Spain)*, Sophie Bastin *(France)*, Mustapha Béjaoui *(Tunisia)*, Lorenzo Brilli *(Italy)*, Martina Carrete *(Spain)*, Emma Cebrian *(Spain)*, Hanene Chaabane *(Tunisia)*, Sílvia Coelho *(Portugal)*, Renato Colucci *(Italy)*, Styliani Dafka *(Germany)*, Sofia Darmaraki *(Greece/Canada)*, Camilla Dibari *(Italy)*, Donna Dimarchopoulou *(Greece)*, Jean-Claude Dutay *(France)*, Monia El Bour *(Tunisia)*, Antonietta Elia *(Spain)*, Elena Georgopoulou *(Greece)*, Sylvaine Giakoumi *(France)*, Juan Jesús González Alemán *(Spain)*, Pablo González-Moreno *(UK)*, Madeleine Goutx *(France)*, Olivier Grünberger *(Tunisia)*, Ivan Güttler *(Croatia)*, Nathalie Hilmi *(Monaco)*, Gabriel Jordà *(Spain)*, Stelios Katsanevakis *(Greece)*, Mehdi Lahlou *(Morocco)*, Manfred A. Lange *(Cyprus)*, Luisa Leolini *(Italy)*, Myriam Lopes *(Portugal)*, Annarita Mariotti *(USA)*, Ana Isabel Miranda *(Portugal)*, Meryem Mojtahid *(France)*, Alexandra Monteiro *(Portugal)*, Samuel Morin *(France)*, Pierre Nabat *(France)*, Anika Obermann-Hellhund *(Germany)*, Tuğba Öztürk *(Turkey)*, Androniki Pardalou *(Greece)*, Sandra Rafael *(Portugal)*, Francesca Raffaele *(Italy)*, Lena Reimann *(Germany)*, Alain Roques *(France)*, Asma Sakka Hlaili *(Tunisia)*, Alberto Santini *(Italy)*, Giuseppe Scarcella *(Italy)*, Katrin Schroeder *(Italy)*, Isla Simpson *(USA)*, Nicolina Staglianò *(Italy)*, Meryem Tanharte *(Morocco)*, Rob Tanner *(France)*, Rémi Thiéblemont *(France)*, Yves Tramblay *(France)*, Marco Turco *(Italy)*, Nassos Vafeidis *(Germany)*, Martin Wild *(Switzerland)*, Elena Xoplaki *(Germany)*, Argyro Zenetos *(Greece)*

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Executive summary

Climate drivers

During recent decades, observations of several variables provide evidence of the ongoing anthropogenic climate change in the Mediterranean region, particularly increase of mean and extreme temperatures, and dry environmental conditions. Climate projections show that the region will among the most affected regions by climate change, specifically regarding precipitation and the hydrological cycle, but also mean warming and heat extremes (in both the terrestrial and marine environment), sea level rise and sea water acidification.

Basin-wide, annual mean temperatures are now 1.5°C above the preindustrial level. In the last decades dry conditions have become more frequent and a large reduction of glaciers across high mountains of the Mediterranean has occurred at a progressively increasing pace. Mediterranean Sea waters have become warmer and saltier, Mediterranean sea level has risen at a rate (1.4 mm yr^{-1}) similar to the global trend at centennial scale.

In the future, the regional average warming will exceed the global mean value by 20% and it might reach 5.6°C at the end of the 21st century in the RCP8.5 high emission scenario. Heat waves and warm temperature extremes will intensify. Total annual precipitation is expected to decrease over most of the region (the average reduction rate is approximately 4% per each degree of global warming). However, magnitude and spatial distribution of changes are uncertain, because of differences among models. Dry conditions will be further enhanced by increasing evapotranspiration over land. At the same time, the inter-annual variability of the hydrological cycle will increase, with longer dry spells especially in the southern areas. Extreme precipitation events will become more intense over large parts of the northern Mediterranean areas.

Mediterranean mean sea level is projected to be at the end of the 21st century in the range from 20 to 110 cm higher than at the end of the 20th century, depending on the level of anthropogenic emissions. Sub-regional and local relative sea level rise will be further modulated by vertical land motions and regional circulation features (with deviations in the order of 10 cm from the basin average). Therefore, though in the future milder marine storms are expected, coastal hazards, floods and erosion will increase, because of mean sea level rise.

Widespread seawater warming will continue. Annual mean surface temperature will increase 2.7-3.8°C and 1.1-2.1°C in one century under the RCP8.5 and the RCP4.5 scenarios, respectively. Marine heat waves will become longer, more intense than today and their spatial extent will increase. Seawater acidification will continue, with a pH reduction that might larger than 0.4 units at the end of the 21st century.

Pollution

Across the Mediterranean Basin, ocean and inland pollution are ubiquitous, diverse and increasing in both quantity and in the number of pollutants, due to demographic pressure, enhanced industrial and agricultural activities, and climate change.

Mediterranean seawater is generally oligotrophic (low nutrient), with decreasing levels from Gibraltar eastwards to the Levantine Sea. Several coastal regions are hotspots of human-induced nutrient inputs. This nutrient enrichment causes eutrophication and may provoke harmful and toxic algal blooms, whose frequency will likely increase. Harmful algal blooms may cause negative impacts on ecosystems and may represent serious economic threats for fisheries, aquaculture, tourism and human health.

Emerging contaminants are well present across the Mediterranean Basin, and enhanced by increasing inflow of untreated wastewater. These substances may cause disorders of the nervous, hormonal and reproductive system. And the increasing frequency of extreme precipitation events in the north of the Mediterranean increases the supply of fecal bacteria and viruses to the coastal zone. The Mediterranean Sea is one of the most polluted large water bodies globally in terms of plastic and the level of this pollution is expected to increase in the future.

The Mediterranean Basin is among the regions in the world with the highest concentrations of gaseous air pollutants (NO_2, SO_2 and O_3). Its dry and sunny climate, and specific atmospheric circulation patterns enhance air pollution levels. Ships are among the major causes of increasing SO_2 and NO_x emissions in this region.

Emissions of aerosols and particulate matter (PM) into the atmosphere arise from a variety of anthropogenic activities. Particular meteorological conditions and natural sources, including the proximity of the Sahara Desert, create particular patterns of aerosol concentrations that may influence particulate matter PM concentrations. The occurrence of critically high PM concentrations associated with dust outbreaks is higher in the southern Mediterranean (>30% of the annual days) than in the northern area (<20% of the annual days).

Land and sea use change

Landscapes and their use have changed over millennia in the Mediterranean Basin, however the rate of change has increased substantially since the second half of the 20th century, with rapid growth of urban and peri-urban areas leading to loss of biodiversity and habitats. Abandonment of agro-pastoralism (which will likely continue in the future) is causing unmanaged shrubs forest development in marginal lands, arid and mountain areas in European countries, while land overexploitation is causing widespread forest degradation in areas of North Africa and the Middle East. Future land use trends depend strongly on regional policies for urbanization, agriculture, forestry and nature conservation.

Marine resource overexploitation and unsustainable fishing practices have increased in time and are the main drivers of the population decline of several species. Presently, more than 60% of marine stocks have collapsed or are overexploited. Sustainable management of marine resources requires the reduction of fishing pressure.

Non-indigenous species

The Mediterranean Sea (and particularly the Levantine Basin) is a hotspot for the establishment of many non-indigenous species (invertebrates, primary producers, and vertebrates), whose arrival and increase are linked to the decrease or collapse in populations of native species. Most marine non-indigenous species enter the Mediterranean Sea from the Red Sea and Atlantic Ocean, but those introduced by ships and aquaculture produce the largest impact on the environment. The number and spread of non-indigenous species will likely further increase with increasing shipping activity and impacts of climate change on the Mediterranean water masses.

Mediterranean land areas currently host a high number of non-indigenous species (mostly plants and invertebrates) in human-modified ecosystems and in regions with high infrastructure development. Most invertebrate species are phytophagous pests that cause damages to crops and forests. Future warming is predicted to induce a northward shift at a speed of 37-55 km decade^{-1} of current major non-indigenous species and determine a window of opportunity for new non-indigenous species adapted to dry environments. The presently increasing trend of the numbers of introduced invertebrates and vertebrates (the latter generally caused by accidental escapes) will very likely continue, as they can be easily transported also as stowaways in air and maritime cargo.

Interaction among drivers

When ecosystems and societal sectors are threatened by multiple, co-occurring drivers, climate change, pollution, land and sea use change, and non-indigenous species can interact. Interactions cause effects that can be additive/cumulative, synergistic or antagonistic and result in alteration, intensification, and even in generation of new impacts. Examples of new threats are increase of flood events, due to a combination of climatic and land use changes, desertification, which is the result of increasing aridity and exploitation of resources, and wildfires, affected by forest encroachment and heat waves, among many other interactions.

2.1 Introduction

This chapter describes characteristics and evolution of human-induced and natural factors that cause changes in the Mediterranean Basin ecosystems and human systems. In order to cover most major risks for people and biodiversity, four broad domains of change drivers are considered: climate change and variability, pollution, land and sea use changes and non-indigenous species. These factors correspond to the concept of "direct drivers", which was introduced in the Millennium Ecosystem Assessment (MEA 2005; Nelson et al. 2006), that unequivocally influence processes in ecosystems and can be identified and measured to differing degrees of accuracy.

Anthropogenic climate change is already affecting the environment and societies in the Mediterranean region. Warming is unequivocal, and there are emerging signs of changes of the hydrological cycle and other climate variables *(Section 2.2)*. Climate models indicate a trend towards a warmer and drier environment, seawater warming, with more intense warm extremes both over land and in the sea, and regional increase of sea level *(Section 2.2)*. The Mediterranean region is likely very vulnerable to climate change and many components of its terrestrial and marine environment are already under stress *(Section 2.2)*.

Atmospheric and water pollution can be driven by many factors, which affect all the compartments of the environment: water, air and soil/sediments. Pollutants can migrate from one media to another. There is a wide range of pollutants that can be biological (e.g., bacteria or insects), chemical (e.g., pesticides, trace metals) or physical (e.g., particulate matter) (*Section 2.4-6* and *Chapters 3* and *4* of this report).

Changes in land and sea use changes are considered among the major direct drivers of environmental change worldwide, but their characteristics vary, depending on each region, even at very local scale. Mediterranean terrestrial landscapes and ecosystems show different patterns of change on northern and southern shores, due to urbanization, coastal development, evolving agricultural and farming practices, including their abandonment. The overexploitation of the Mediterranean Sea resources poses a particular threat due to its intrinsic geographical limits *(Section 2.4)*.

Non-indigenous species are profoundly affecting terrestrial and marine ecosystems in the Mediterranean and their impact is not only measurable in biodiversity alterations, but also in human health and economic damages *(Section 2.5)*.

There is no strict consensus of the grouping of the drivers into the categories that have been adopted in this report. More condensed or more articulated lists can be found in the scientific literature. Our four categories include all physical, chemical and biological factors that directly act on the Mediterranean environment, with a substantial correspondence with those used in the recent IPBES 2018 regional reports (Bustamante et al. 2018; Elbakidze et al. 2018; Nyingi et al. 2018; Wu et al. 2018b).

The level or rate of change of direct drivers can be influenced or altered by indirect drivers (MEA 2005; Nelson et al. 2006). Indirect drivers are grouped in categories such as demographic, economic, sociopolitical, cultural, religious, technological, legislation and financial drivers. In turn, indirect drivers are distinguished in "endogenous" and "endogenous" drivers, whose magnitude can and cannot be influenced/altered by the decision-makers, respectively. Whether a driver is exogenous or endogenous depends on the organizational level and on the spatial and temporal scale. The concepts of indirect and direct drivers roughly match those of driving forces and pressures in the Drivers–Pressures–State–Impacts–Responses (DPSIR) framework, which was initially developed by the European Environmental Agency (EEA 1999).

Understanding of indirect drivers is essential for the benefit of the environment. It is the action on indirect drivers by policymakers and stakeholders that can effectively manage the risks posed to the environment and human societies by climate change, pollution, land and sea use changes and arrival of non-indigenous species. However, this is a quite different topic with respect to the content of this chapter. Mechanism and tools influencing direct drivers are considered in other parts of this report. The scope of this chapter is the assessment of the state of knowledge of physical, chemical, and biological factors, of their present status, past and future evolution in the Mediterranean Basin.

2.2 Climate change

2.2.1 Framing

There is no universal definition of the land boundaries of the Mediterranean region *(Chapter 1)*. In this section, and also for much of this report, we adopt a simple regular latitude-longitude box (29°N to 47.5°N and 10°W to 39°E, *Fig. 2.1*), which includes some regions with other than Mediterranean climates, such as in the Alps, the Eastern Balkans or part of the Sahara. This definition of Mediterranean region is similar to the MED zone adopted in IPCC-AR4 (IPCC 2007), and slightly larger than in IPCC-AR5 (IPCC 2013a).

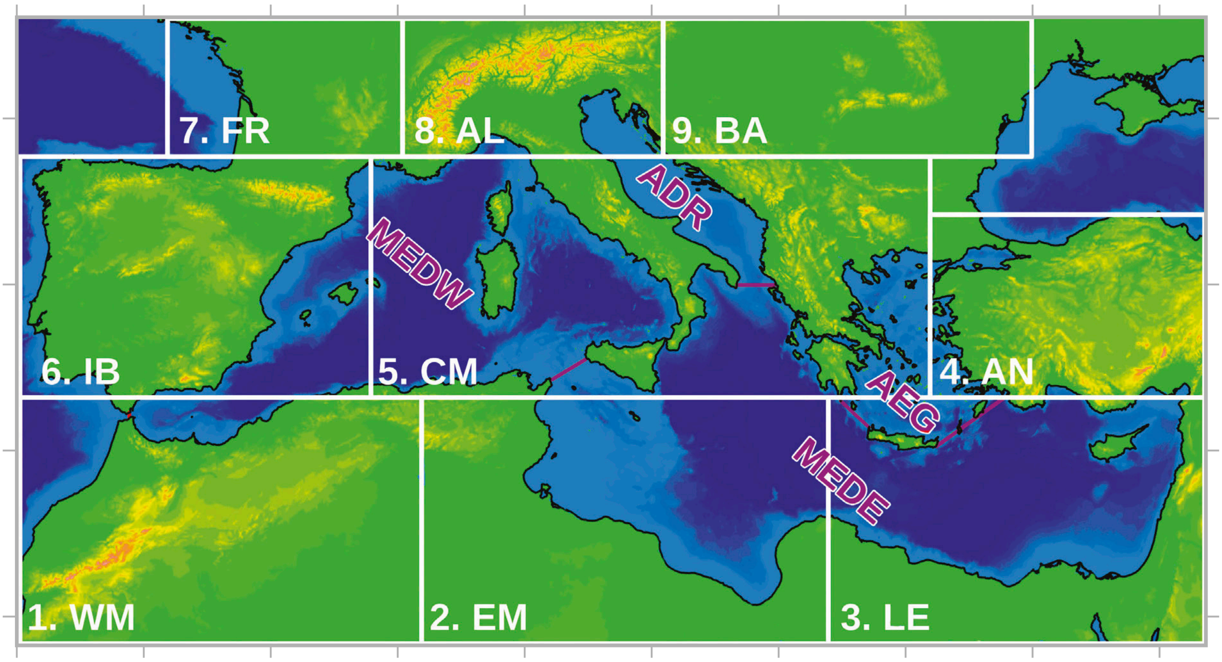


Figure 2.1 | Mediterranean coastline, topography over land and bathymetry over the sea plus the box definition. Relief data are derived from the ETOPO1 1 Arc-Minute Global Relief Model[5]. Sub-regions are defined as West Maghreb (WM), East Maghreb (EM), Levant (LE), Anatolia (AN), Central Mediterranean (CM), Iberia (IB), France (FR), Alps (AL) and Balkans (BA). The Mediterranean Sea is divided into 4 sub-basins, the Western Mediterranean Sea (MEDW), the Eastern Mediterranean Sea (MEDE), the Aegean Sea (AEG) and the Adriatic Sea (ADR).

In order to provide a spatially refined assessment, we define sub-regions over land and over sea by using smaller rectangular boxes. Over land, the European sub-regions follow the definition proposed during the PRUDENCE European project (Christensen et al. 2002) *(Fig. 2.1)*, and we define new sub-regions (inspired by Nabat et al. 2015b) for the Middle East and Africa. Over the sea, 4 sub-regions are defined using the natural strait delimitations *(Fig. 2.1)*, the Western Mediterranean Sea (MEDW) between the Gibraltar Strait and the Sicily Strait, the Adriatic Sea (ADR) north of the Otranto Strait, the Aegean Sea (AEG) north of the Cretan Arc Straits and the Eastern Mediterranean Sea (MEDE) for the remaining.

When assessing future climate change, it is important to specify the reference period to which climate projections are compared, along with future "time slices". In MAR1 we use 20-year long time periods following standard IPCC practice. This length of time period is sufficient to smooth part of the high-frequency natural climate variability that can otherwise mask the forced trend, but it is short enough to assume that climate does not change much during the 20 years covered. For the reference period, we choose the latest years of the 20th century (1980-1999). This choice is a compromise related to the observation and model data availability at the Mediterranean scale. IPCC has traditionally chosen the pre-industrial

[5] http://www.ngdc.noaa.gov/mgg/global/global.html

period (around 1850 or 1900) as a reference but regional climate models and regional high-quality observations are not available for that period. This choice (2 full decades at the end of the 20th century) also targets to facilitate the repeatability of the MAR1 computations made in future Mediterranean studies or reports.

In addition to the reference period, we define also the "present-climate" period (1995-2014), which defines the current climate conditions. The reference period is fixed in different reports to be able to intercompare results of simulations across different model generations, whereas the present-climate will move from one report to another. For example, the IPCC-AR4 (IPCC 2007) defined 1981-2000 as present climate, whereas the IPCC-AR5 (IPCC 2013a) used 1986-2005. For the assessment of past changes, the longest period available (generally 1950-2019 or 1900-2019, if possible) in the observations is used in this report and units such as °C per decade are used in order to compare past trends computed over different past periods.

For the future, we keep 20-year time slices in order to sample the same level of internal variability as in the reference period. We divide the 21st century in 20-year time slices with a present-climate period (2000-2019), a near-future period (2020-2039), a mid-term period centered in 2050 (2040-2059) and a far-future period close to the end of the 21st century (2080-2099). The mid-21st century period is arguably of particular interest for many stakeholders, especially for mid-term adaptation. The end of the 21st century period is also of interest for stakeholders working on mitigation targets and involved in very long-term planning (e.g., for the design and planning of dams, forests, cities).

For the future climate change assessment, an important part of the uncertainty is related to the future evolution of socio-economic development. To be able to propose future climate projections considering various possible socio-economic trajectories and climate policy pathways, we follow the IPCC scenario approach. Where more recent studies are not available, the assessment also considers studies based on the older IPCC SRES approach (Nakićenović 2000). Although results based on multiple IPCC scenarios are reported in the MAR1, we mostly focus on two options which encompass the range of IPCC-AR5, CMIP5 and CORDEX simulations: the "business as usual" scenario (RCP8.5, for an explanation of the RCPs see *Box 2.2*) and the optimistic scenario closest to the UNFCCC Paris Agreement target (RCP2.6). These scenarios have been chosen also due to model projection availability constraints at the regional scale.

Detecting trends and attributing regional climate change to human influence is challenging due to natural climate variability and the strong spatio-temporal dependency of different climate variables. For projections, uncertainty estimates are provided where this is possible. For small model ensembles, the total range is also given. For larger ensemble, 90% confidence intervals are used as much as possible or else "whisker plots" describing the various statistics of the distribution (median, 25th and 75th percentile, 90% interval, minimum and maximum values).

2.2.1.1 Observations and reanalyses

More and more observation datasets have become available at regional scales, either from satellite or *in-situ* observations, or from reconstructions and reanalyzes. This new generation of observation-based products are (1) long and homogeneous enough to allow trend studies (ESA-CCI) (Ribes et al. 2019); (2) of sufficiently high spatial resolution to capture complex topography and land-sea mask (SAFRAN, EURO-4M), thereby allowing regional to local studies; and (3) of sufficiently high temporal resolution (daily or hourly, COMEPHORE) (Fumière et al. 2019), to allow the study of regionally-relevant extreme events. High-resolution gridded products have also become available for southern Mediterranean countries e.g., Cyprus (Camera et al. 2017) or Tunisia (Tramblay et al. 2019).

Finding the best fit-for-purpose observation dataset is becoming a new challenge, given the large number of available products, often characterized by substantial differences. Results of past trend studies and model evaluations are sensitive to the choice of the reference dataset (Flaounas et al. 2012; Prein and Gobiet 2017; Zittis 2018; Fumière et al. 2019; Kotlarski et al. 2019; Peña-Angulo et al. 2020). Long-term, accessible, gridded, well-calibrated and homogeneous in time and space *in-situ* data are nonetheless still lacking, especially for the ocean or the high-frequency variables over land. In addition, regional model-based reanalyzes are still rare.

Various observation datasets are used to assess the past evolution of the different components of the Mediterranean climate system. Atmospheric dynamics are mostly assessed against atmospheric reanalyzes (ERA-Interim, ERA20C, 20CR) (*Section 2.2.2)*. For aerosols, clouds and surface radiation, both satellite products and station data (BRSN, GEBA) are used to estimate past evolutions *(Section 2.2.3)*. The trend evaluation over land relies on high-resolution observation-based gridded products, i.e., CRU, E-OBS for temperature *(Section 2.2.4)* and CRU, E-OBS, U. Del, GPCC for precipitation *(Section 2.2.5)*. The other water cycle components are evaluated against reconstructed products blending *in-situ* observations, satellite and models of river networks (Ludwig et al. 2009; Pellet et al. 2019; Wang and Polcher 2019) *(Section 2.2.6)*, while satellite data are used for sea surface temperature (Marullo et al. 2010; Pisano et al. 2016; Pastor et al. 2018) *(Section 2.2.7)*, and *in-situ* data for the deep water characteristics (Houpert et al. 2016; Schroeder et al. 2017; Testor et al. 2018; von Schuckmann et al. 2018).

2.2.1.2 Modelling

Complex and realistic global and regional climate models (GCMs and RCMs), based on fundamental physics, chemistry and biology equations are currently the standard tools to simulate the future evolution of the regional climate system. Different types of climate models are available to study the Mediterranean climate (past and future), often organized in large coordinated multi-model initiatives under the World Climate Research Programme (WCRP) umbrella CORDEX (Giorgi et al. 2009) and CMIP5 (Taylor et al. 2012). Combining the various sources of information or extracting the most credible (actionable) information is a new challenge, sometimes called the "distillation" problem (Hewitson et al. 2014; Fernández et al. 2019).

The MAR1 climate assessment concerning future climate evolution is based on four climate model ensembles, chosen for their good representation of the Mediterranean climate and for their good coverage of the various sources of uncertainty in future climate projections:

- CMIP3 and CMIP5 GCM ensembles with resolution ranging approximately from 300 to 100 km: they are the largest GCM multi-model ensembles available so far. They cover at a relatively low resolution all the uncertainty sources and can provide data for all the components of the climate system (atmosphere, land, ocean, marine biogeochemistry and aerosols). Some of the participating models share components and this may result in a redundancy in the ensemble results (Knutti et al. 2017).
- The Euro-CORDEX RCM ensemble is a large high-resolution ensemble at 12 km resolution (Jacob et al. 2014), which has clearly improved the representation of climate variables compared to coarse resolution GCMs over land, e.g., for extreme precipitation (Fantini et al. 2018), regional winds (Obermann et al. 2018), mountain climate (Torma et al. 2015; Torma 2019), and over the sea, e.g., for regional winds (Herrmann et al. 2011) and extreme winds ("medicanes") (Gaertner et al. 2018). An example for the strong modification of the future climate change signal by high-resolution RCM compared to GCMs has been found for summer precipitation over the Alps (Giorgi et al. 2016). Although the EURO-CORDEX ensemble is a large high-resolution multi-model dataset, it does not cover the entire uncertainty space of the CMIP5 ensemble.
- The Med-CORDEX RCSM ensemble (Ruti et al. 2016) is a relatively small ensemble which does not cover the CMIP5 uncertainty range particularly well, but is the best data source available to study the future evolution of the Mediterranean Sea (Darmaraki et al. 2019b; Soto-Navarro et al. 2020), its ecosystems (Moullec et al. 2019) and atmosphere-ocean interactions. The models of this ensemble have high-resolution in both the atmosphere (resolution range: 25-50 km) and the ocean (resolution range: 6-30 km) component of the regional climate system (Somot et al. 2018).
- The CORDEX FPS-convection CPRCM ensemble (Coppola et al. 2020): this mini-ensemble provides the highest spatial resolution (2-3 km) for the greater Alpine region, reaching convection resolving scales. It yields, in particular, a strong improvement in the representation of extreme precipitation at sub-daily time scales (Kendon et al. 2014; Ban et al. 2015; Fosser et al. 2015; Berthou et al. 2018; Fumière et al. 2019). Convection resolving models are very promising tools to study the future evolution of extreme precipitation associated with thunderstorms, medicanes or mesoscale convective systems in the Mediterranean region (Lenderink et al. 2019) or urban-climate interactions.

A large variety of downscaling methods are available to study the Mediterranean climate (COST VALUE) (Maraun et al. 2019), including on-line tools on web processing servers (Cofiño et al. 2007). Among these methods, correcting climate change simulations using statistical tools (bias correction methods) allows to improve the present-climate statistics of climate simulations, with clear benefits for studying threshold-dependent extremes or for forcing impact models. All statistical methods require long-term observations (stations, gridded-products or satellite data calibrated for climate applications) for the learning phase and their application is therefore limited in regions where observations are lacking.

In the MAR1 report, CMIP ensembles are mostly used for the assessment of atmospheric dynamics, aerosol, cloud and radiation, water cycle, sea hydrology, sea level and acidification, whereas the Euro-CORDEX ensemble is used for the assessment of wind, clouds, temperature, precipitation and the cryosphere. The Med-CORDEX coupled regional models are used for sea hydrology and sea level, and the FPS-convection ensemble for the assessment of extreme precipitation.

Despite the continuous improvement of climate model ensembles by increased resolution and ensemble size, increased complexity and improved model physics, these still suffer from deficiencies and systematic errors. In particular their poor representation of some key regional phenomena may limit confidence for some aspects of the MAR1 assessment. This is especially true for coastal sea level, medicanes, tornadoes, hail phenomena, lightning, city climate, sub-daily precipitation, glaciers, clouds or cloud-aerosol interactions, human influence on land and water use.

2.2.2 General circulation and dynamics

2.2.2.1 General circulation

The proximity of the Mediterranean to the Atlantic and Indian Oceans and the surrounding massive land areas, with diverse climatic characteristics, places the area at the crossroads of many global climate patterns and processes of tropical and extra-tropical origin (Xoplaki et al. 2003a; Lionello et al. 2006; Lelieveld et al. 2012; Lionello et al. 2012a; Lionello 2012; Ulbrich et al. 2012). Its location on the eastern edge of the Atlantic Ocean means it is particularly affected by variability and change in the North Atlantic jet stream (or Polar Front Jet) in both winter and summer (Dünkeloh and Jacobeit 2003; Hurrell et al. 2003; Athanasiadis et al. 2010; Bladé et al. 2012) and by stationary blocking patterns (Tyrlis and Hoskins 2008). The Mediterranean Basin is also influenced by semi-permanent large-scale anticyclones (e.g., the Azores anticyclone in the west during summer and the cold Siberian anticyclone in the northeast during winter), while mobile anticyclones play also important role throughout the whole year (Hatzaki et al. 2014). During the summer, the climate of the Mediterranean is further influenced by circulation patterns set up by the Asian summer monsoon (Rodwell and Hoskins 1996) and local orography (Simpson et al. 2015).

Observed trends

The wintertime large-scale circulation has exhibited a long-term trend toward increased sea-level pressure and anticyclonic circulation over the Mediterranean *(Fig. 2.2a and b)* (Mariotti and Dell'Aquila 2012). Aside from this long-term trend, the historical record has also exhibited sizable multi-decadal variability. This is illustrated for the sea-level pressure anomalies in *Fig. 2.2e* and has also been discussed widely in the contexts of trends in the North Atlantic Oscillation (NAO) and associated Mediterranean drying that occurred over the latter half of the 20th century (Hurrell 1995) in which internal variability is thought to play an important role (Kelley et al. 2012). CMIP5 models suggest that the externally forced contribution to sea-level pressure trends since 1900 consist of a pattern that resembles that of the observed trends but with a magnitude that is considerably smaller *(Fig. 2.2c)*. It is likely that both external forcing and internal variability have contributed to the observed long-term trends (Hoerling et al. 2012). During summer, it is challenging to assess the observed long-term trends, as there is no strong agreement in the pattern of sea-level pressure change *(Fig. 2.2f and g)*. There are indications of a summertime decline in sea-level pressure over North Africa and the southern Mediterranean and, indeed, the CMIP5 models suggest that external forcings have contributed to a decline in sea-level pressure in this region over the 20th century *(Fig. 2.2h)*.

Future changes

Under rising greenhouse gas concentrations, climate models project that the Hadley Cell circulation will change, the tropics will expand and the mid-latitude westerlies and associated storm tracks will likely shift poleward *(medium/high confidence)* (Yin 2005; Lu et al. 2007b, 2007a; Chang et al. 2012; Barnes et al. 2013; Shaw et al. 2016; D'Agostino et al. 2017, 2020). This is expected to enhance subsidence and reduce storminess at the latitudes of the Mediterranean region, with a resulting reduction in precipitation *(medium confidence)*. While there is considerable inter-model spread in the magnitude of these projected changes and the forced signal can be small compared to internal variability (Woollings and Blackburn 2012; Barnes et al. 2013; Zappa et al. 2015; Quan et al. 2018; Grise et al. 2019), the Mediterranean could be influenced by additional local circulation anomalies, leading to pronounced hydroclimate changes (Seager et al. 2014; D'Agostino and Lionello 2020). Future projections suggest that the wintertime trend toward increased anticyclonic circulation

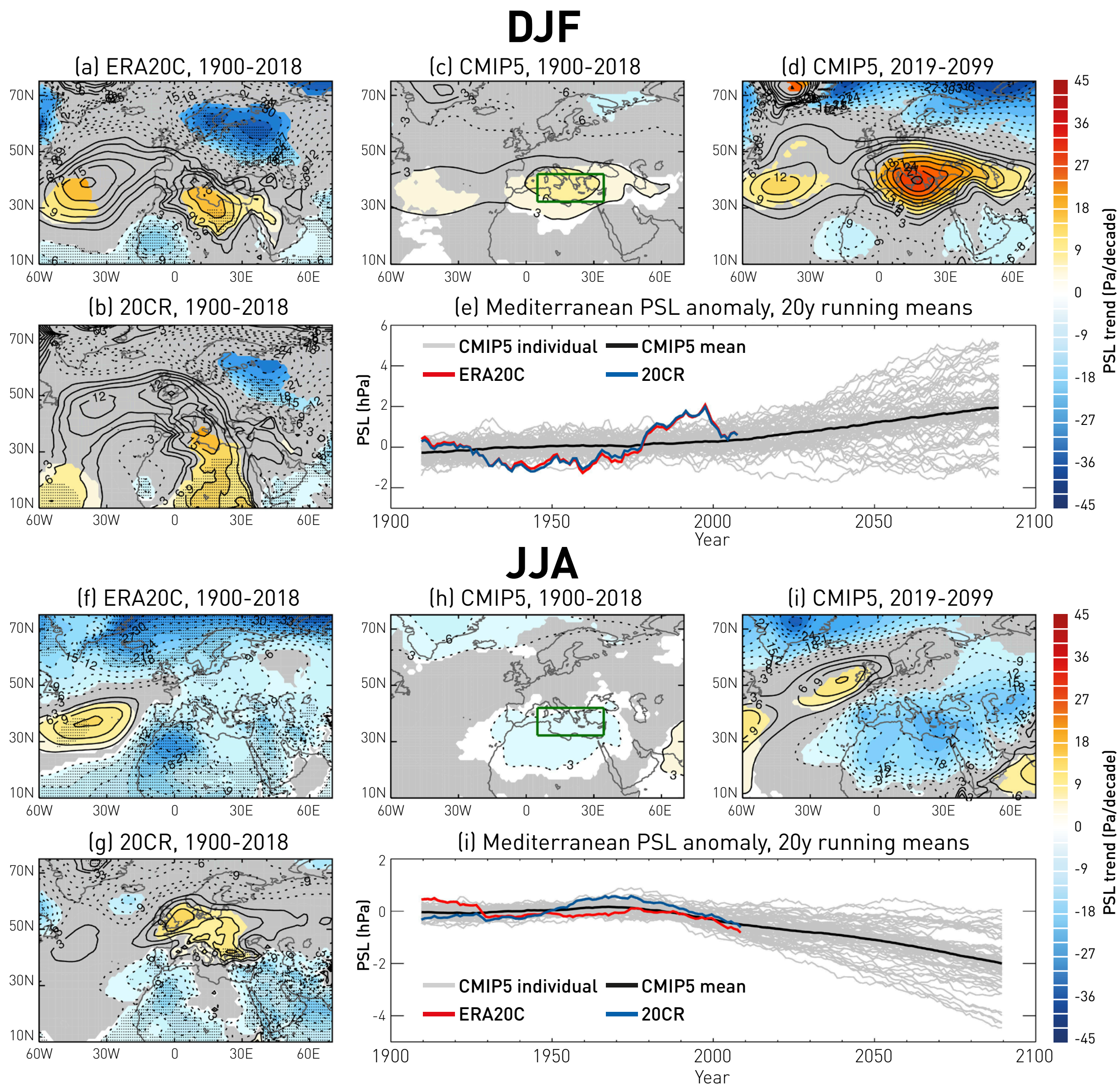


Figure 2.2 | Trends in sea level pressure (SLP). (a)-(e) show the DJF (December-January-February) season. (a) and (b) show 1900-2018 trends in SLP from ERA20C and 20CR reanalyses, respectively. Grey = not significantly different from zero at the 95% level. Significance is calculated by resampling, with replacement, the residuals of the linear trend, adding the resampled residuals to the linear trend and re-calculating the linear trend. This is repeated 1,000 times to obtain the probability (p-value) at each grid point that the trend is significantly different from zero. Spatial autocorrelation is accounted for using the False Discovery Rate method of Wilks (2016) with control value = 0.1. Stippling depicts grid points where the magnitude of the trend is larger than in any of the individual CMIP5 ensemble members. (c) shows the CMIP5 multi-model mean trend for 1900-2018. The ensemble mean for each model is calculated, then the linear trend is obtained before calculating the average trend across all models. Grey depict regions where less than 3/4 of the models agree on the sign of the change. (d) is as (c) but for future trends from 2019-2099. (e) shows time series of 20 year running mean SLP averaged over the Mediterranean (green box in c) for the two re-analyses, all individual ensemble members from all models and the CMIP5 multi-model mean. (f)-(j) are as (a)-(e) but for JJA (June-July-August).

over the Mediterranean will continue at an accelerated pace *(Fig. 2.2d)* (Giorgi and Coppola 2007). This is accompanied by a strengthening of the westerly winds and increased storminess over northern Europe, reduced westerlies over north Africa and decreasing storminess over the Mediterranean (Woollings and Blackburn 2012; Rojas et al. 2013; Zappa et al. 2013). Climate models predict a summertime poleward shift of the North Atlantic jet (Simpson et al. 2015) and a summertime decrease/wintertime increase in sea-level pressure centered over the Mediterranean *(Fig. 2.2i)* (Giorgi and Coppola 2007; Bladé et al. 2012), with the reduction in Mediterranean sea-level pressure continuing at an accelerated pace over the coming decades *(Fig. 2.2i)*, albeit with a large inter-model spread *(Fig. 2.2j)*. This was argued to be dominated by a heat-low response to drier soils in the Mediterranean region (Haarsma et al. 2009).

2.2.2.2 Teleconnection patterns

The influence of teleconnection patterns (linkages between weather changes occurring in widely separated regions of the globe) on Mediterranean climate has been extensively studied (Corte-Real et al. 1995; Hurrell and Van Loon 1997; Wibig 1999; Pozo-Vázquez et al. 2001; Quadrelli et al. 2001; Xoplaki et al. 2003a, 2003b, 2004; Hatzaki et al. 2009; Toreti et al. 2010; Ulbrich et al. 2012; Tyrlis and Lelieveld 2013; Rousi et al. 2015; Sen et al. 2019). Particularly during winter, the region is prone to the impacts of the westerly flow and the teleconnection patterns of North Atlantic Oscillation (NAO), Eastern Atlantic/Western Russia (EA/WR) and Scandinavian (SCAN) (Barnston and Livezey 1987).

The NAO is, for parts of the region, one of the most important modes of internal climate variability. It affects especially the storm-tracks and cyclogenesis over parts of the basin (Trigo et al. 2000, 2004; Reale and Lionello 2013) and mainly precipitation over the western-central Mediterranean in winter (Lamb and Peppler 1987; Rodríguez-Fonseca and de Castro 2002; Xoplaki et al. 2004). NAO is also found to have some influence on winter precipitation in parts of the eastern Mediterranean, however this influence is smaller (Dünkeloh and Jacobeit 2003; Xoplaki et al. 2004; Feliks et al. 2010; Felis and Rimbu 2010; Nissen et al. 2010). A lesser but distinct influence is detected between NAO and the Mediterranean surface air temperature (Hurrell 1995; Cullen and DeMenocal 2000; Ben-Gai et al. 2001; Pozo-Vázquez et al. 2001; Sáenz et al. 2001; Castro-Díez et al. 2002; Trigo et al. 2002b; Türkeș and Erlat 2003; Xoplaki et al. 2003b; Toreti et al. 2010).

Observed trends of NAO are not monotonic and are difficult to assess since decadal oscillations are too large to reach a conclusion with an acceptable level of confidence. Nevertheless, mostly negative trends prevail since the early 1990s following a positive trend starting in the 1960s (Ulbrich and Christoph 1999; Mariotti et al. 2002b; Türkeș and Erlat 2003; Trigo et al. 2004, 2006; Xoplaki et al. 2004; Pinto and Raible 2012; Saffioti et al. 2016; Iles and Hegerl 2017).

Climate projections mostly suggest a weak positive NAO trend in a warmer future climate *(low/medium confidence)*, accompanied by a small northeastward displacement of its centers-of-action by the end of the 21st century (Ulbrich and Christoph 1999; Gillett et al. 2003, 2013; Hu and Wu 2004; Stephenson et al. 2006; Bacer et al. 2016; Deser et al. 2017; Barcikowska et al. 2020). Some studies indicate no significant trends (Fyfe et al. 1999; Dorn et al. 2003; Rauthe et al. 2004; Fischer-Bruns et al. 2009), or even decreasing trends (Osborn et al. 1999).

Studies on the effect of El Niño Southern Oscillation (ENSO) phenomenon on Mediterranean precipitation have shown that links exist, particularly during autumn and spring in the western Mediterranean and during winter in the Eastern Mediterranean. However, results are not conclusive concerning their evolution and robustness. In fact, the ENSO signal is difficult to be isolated, because of the dominating mid-latitude dynamics, the sign of its correlation with total precipitation depends on season and it is not stationary (Rodó et al. 1997; Rodó 2001; Mariotti et al. 2002b, 2005; Knippertz et al. 2003; Hasanean 2004; Alpert et al. 2006; López-Parages and Rodríguez-Fonseca 2012; Kalimeris et al. 2017).

2.2.2.3 Extratropical cyclones and medicanes

The Mediterranean is one of the main cyclogenetic areas of the world (Petterssen 1956; Hoskins and Hodges 2002; Wernli and Schwierz 2006), with much of the high-impact weather (e.g., strong winds and heavy precipitation) associated with cyclonic structures. Cyclogenesis areas such as the north-western Mediterranean, North Africa, the north shore of the Levantine Basin, the seasonality (Alpert et al. 1990a, 1990b; Trigo et al. 1999, 2002a; Lionello et al. 2006, 2016; Campins et al. 2011), as well as the occurrence of explosive cyclogenesis (Kouroutzoglou et al. 2011; Reale et al. 2019) are well documented in the literature. Within Mediterranean cyclones, there is a sub-group of hybrid depressions of extratropical cyclogenesis, the so-called 'medicanes' (Mediterranean hurricanes) or tropical-like cyclones (Rasmussen and Zick 1987; Reale and Atlas 2001; Emanuel 2005). These are mesoscale maritime extratropical cyclones that can physically emulate tropical characteristics at a certain point of their life cycle (Emanuel 2005; Miglietta 2019). Such features can include a cloud-free area at the center (the "eye"), spiral bands with deep convection around it, intense surface winds and a warm-core and symmetric structure (Miglietta et al. 2015). These events can pose serious societal and ecological threat to the affected coastal regions (Nastos et al. 2018).

During the recent past there is an absence of strong trends in cyclone numbers affecting the Mediterranean (Lionello et al. 2016), however when trends are detected these are mostly negative *(low/medium confidence)* (Trigo et al. 2000; Maheras et al. 2001; Flocas et al. 2010; Nissen et al. 2010). Similarly, the number of explosive Mediterranean cyclones has likely decreased, but this reduction is not statistically significant (Kouroutzoglou et al. 2010). The statistical record of medicanes has limited reliability and sample size, given their maritime characteristics, small size and infrequent occurrence. Thus, it has not been possible to derive an objective climatology. Observational studies cannot be used to identify trends because the identification is commonly subjective (Miglietta et al. 2013; Tous and Romero 2013; Nastos et al. 2018). Dynamical downscaling methods have been used to build a medicane climatology, but only negligible trends were obtained (Cavicchia et al. 2014).

For the future, climate models project a reduction in the number of cyclones *(medium/high confidence)* especially in winter (Lionello et al. 2002; Geng and Sugi 2003; Bengtsson et al. 2006; Leckebusch et al. 2006; Lionello and Giorgi 2007; Pinto et al. 2007; Löptien et al. 2008; Ulbrich et al. 2009; Raible et al. 2010; Zappa et al. 2013; Nissen et al. 2014). There is some uncertainty, as the spread in the model responses appears to be quite large (Ulbrich et al. 2008, 2009; Harvey et al. 2012). With respect to cyclone intensity, climate projections are more controversial, as some models suggest a decrease in the frequency of the most intense systems (Pinto et al. 2007; Raible et al. 2010), while other models show more extreme events or increases in the intensity of extreme cyclones (Lionello et al. 2002; Gaertner et al. 2007). For medicanes, climate projections indicate a decreasing response in frequency but increasing intensity (Gaertner et al. 2007; Romero and Emanuel 2013, 2017; Cavicchia et al. 2014; Walsh et al. 2014; Tous et al. 2016; Romera et al. 2017; González-Alemán et al. 2019).

2.2.2.4 Regional winds

Surface wind speed and its changes on different temporal and spatial scales are governed by driving and drag forces, where all relevant contributions are difficult to estimate and disentangle (Wu et al. 2018a). Surface wind climate studies are less common than air-temperature and precipitation studies for example, and more work is needed to explain historical wind speed evolution and precisely estimate different sources of uncertainty in the future projections. This variable is now becoming more important, since parts of the region, both inland and offshore, have high potential for the production of wind energy (Balog et al. 2016; Onea et al. 2016) *(Chapter 3.3)*.

Observation-based studies of winds over the Mediterranean are rare, and depend on the availability of homogenized and long time series. In most regions, wind trends were found non-monotonic over the past decades and, concrete conclusions are difficult to be established (Pirazzoli and Tomasin 2003; Vautard et al. 2010; Azorin-Molina et al. 2014). An additional source of information are reanalysis datasets, but robust trends have been identified over only a few regions in the Mediterranean (Nissen et al. 2010; Donat et al. 2011; Bett et al. 2013). Climate model simulations over historical periods can also be used in assessing and understanding past trends (Knippertz et al. 2000).

Despite the uncertainties in future projections (Shepherd 2014; Belušić Vozila et al. 2019), there is a general agreement for a limited wind speed reduction over most of the Mediterranean, with the exception of the Aegean Sea and north eastern land areas *(Fig. 2.3, Section 2.2.8) (medium confidence)* (Somot et al. 2006; Mcinnes et al. 2011; Dobrynin et al. 2012; Planton et al. 2012; Belušić Vozila et al. 2019). Changes in the local winds (such as Bora, Mistral, Tramontane, Sirocco and Etesians) may have more complex responses involved, depending on the changes in their underlying feedbacks (Grisogono and Belušić 2009; Ulbrich et al. 2012). Regional projections over the Adriatic reveal strong sensitivity in the climate change signal of the local Bora and Sirocco winds (Belušić Vozila et al. 2019). In particular, the frequency of winter Bora events is projected to increase while the frequency of Sirocco events is expected to decrease. Overall, the mean wind speed during Bora and Sirocco events is expected to be reduced, with the exception of Bora in northern Adriatic. RCM projections of Mistral and Tramontane show small changes in the former and significant decrease in the frequencies of the latter (Obermann-Hellhund et al. 2018). Etesian winds over the Aegean Sea is one of the few exceptions since increases in the wind speed are expected for the future (Ezber 2018; Dafka et al. 2019). In general, RCM projections have the

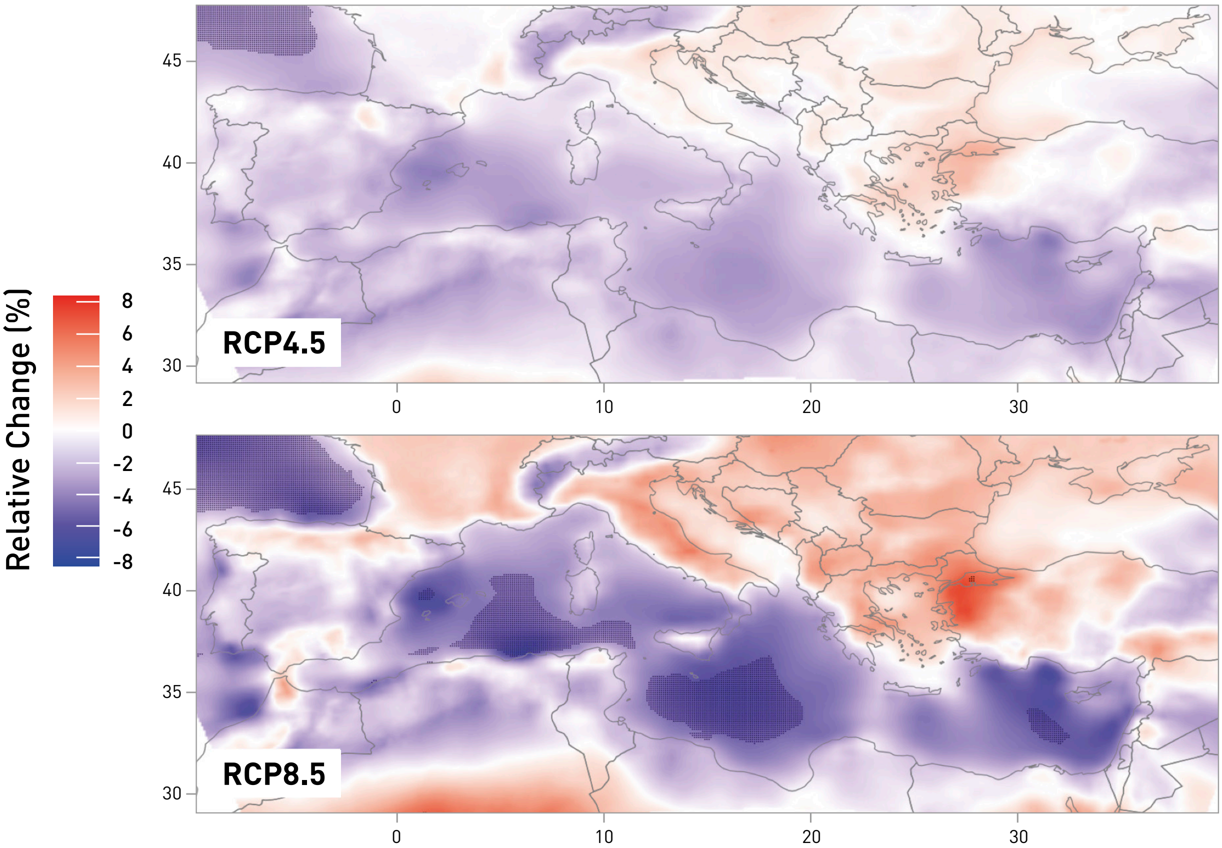


Figure 2.3 | Projected changes (%) in surface wind speed based on Med-CORDEX simulations (Ruti et al. 2016) for the end of the 21st century (2071-2100) relative to the period 1961-1990, for pathways RCP4.5 (top panel) and RCP8.5 (bottom panel). Seven RCMs were used for RCP4.5 and eight RCMs were used for RCP8.5. Dotted areas show differences that passed a 95% significance test.

tendency to simulate decrease of the wind energy density over the Mediterranean, with the exception of the Aegean Sea (Hueging et al. 2013; Tobin et al. 2015; Moemken et al. 2018).

Regional climate simulations indicate changes in wind speed over land regions as well *(Fig. 2.3)*. Most pronounced changes, consistent for both RCP4.5 and RCP8.5 pathways, are an increase of wind speeds over the Balkans and a decrease over the Alps.

2.2.3 Radiation, clouds and aerosols

The amount of solar radiation reaching the Earth's surface is a key determinant of the spatio-temporal variations of climate on our planet and is the ultimate energy source for all processes relevant for climate and life. The main factors explaining the variability of surface solar radiation over the Mediterranean region for different time scales (daily, seasonal, interannual, past and future trends) are clouds (Pyrina et al. 2015) and aerosols (Nabat et al. 2015a, 2015b). The daily variability in clouds and aerosols is strongly influenced by weather regimes (Rojas et al. 2013; Nabat et al. 2020) and climate oscillations such as the North Atlantic Oscillation, NAO (Chiacchio and Wild 2010).

Aerosols in the Mediterranean come from various and numerous sources (Lelieveld 2002), both natural, notably dust and sea-salt, and anthropogenic, notably sulfates, nitrates and black carbon *(Section 2.3)*. Their interactions with radiation and clouds are essential in understanding climate in this region.

2.2.3.1 Observed change in surface radiation

The long-term solar radiation records taken at widespread locations around the globe underwent substantial multidecadal variations, characterized by a reduction of surface solar radiation from the 1950s to the 1980s, known as "global dimming" and a partial recovery, thereafter, referred to as "brightening" (Wild 2009, 2012; Wild et al. 2017). This dimming/brightening pattern is also observed

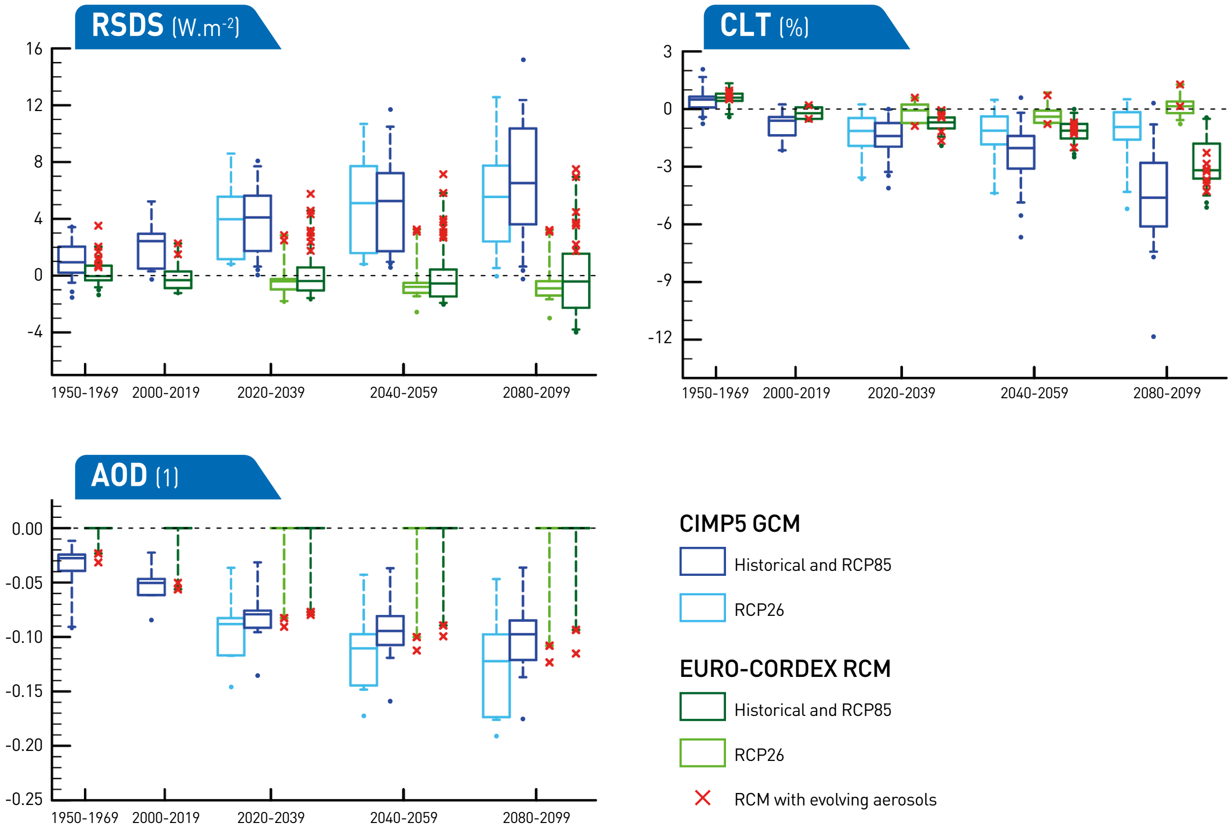


Figure 2.4 | Past and future evolution of Surface Downwelling Shortwave Radiation (RSDS in W m^{-2}), Total Cloud cover (CLT in %) and Aerosol Optical Depth (AOD) from 1950 to 2100 averaged over the Mediterranean area in CMIP5 Global Climate Models (GCM) and 12-km Euro-CORDEX Regional Climate Models (RCM) ensembles. Figures based on Bartok et al. (2017), redrawn and extended using published datasets.

in the Mediterranean area *(very high confidence)*, both in all-sky and clear-sky conditions, documented with many ground-based and satellite observations after careful data quality assessment and homogenization (Sánchez-Lorenzo et al. 2007, 2013, 2017; Zerefos et al. 2009; Kambezidis et al. 2016; Manara et al. 2016; Alexandri et al. 2017; Pfeifroth et al. 2018), as well as climate simulations (Folini and Wild 2011; Zubler et al. 2011; Nabat et al. 2014). The surface solar radiation trends averaged over the Mediterranean have been estimated in climate model simulations between -3.5 and -5.2 W m^{-2} per decade for the dimming period (1953-1968), against for the brightening period between +0.9 and +4.6 W m^{-2} per decade in 1989-2004 (Folini and Wild 2011) and 2.3 W m^{-2} per decade in 1980-2012 (Nabat et al. 2014). The surface solar radiation anomalies calculated by global climate models for the periods 1950-1969 and 2000-2019 against the reference period of 1980-1999 are positive, showing respectively the dimming and the brightening effects, since the reference period refers to the period where surface solar radiation was the lowest *(Fig. 2.4)*.

In parallel to the brightening period, a decrease in aerosol loads has been observed since 1980 both in ground-based stations (Li et al. 2014) and in satellite data (Floutsi et al. 2016). This decrease, corresponding to a trend in aerosol optical depth of -0.03 decade^{-1}, is mainly due to reductions in anthropogenic emissions, leading to a decrease in anthropogenic aerosol concentrations such as sulfate (Nabat et al. 2013). These aerosol trends have been shown to be the main explanation of the dimming-brightening phenomenon in the Mediterranean area *(high confidence)* through attribution model studies (Folini and Wild 2011; Zubler et al. 2011; Nabat et al. 2014), and with the direct aerosol effect responsible for about 80% of the simulated brightening. This phenomenon is qualitatively reproduced by most GCMs, but only by a few regional climate models, due to different treatments of aerosols in models *(Fig. 2.4)*. The evolution of natural aerosols is more uncertain over the same period.

Concerning clouds, a decrease in cloud cover of 0.63% per decade since the 1970s has also been detected from different observations datasets over the Mediterranean area (Sánchez-Lorenzo et al. 2017). This trend mainly concerns low and mid cloud layers (Kambezidis et al. 2016) *(medium confidence)*. The spread between models that capture these trends is high, because of their difficulties to capture cloud characteristics *(Fig. 2.4)*. Clouds may also have played a significant role in the past trend of surface solar radiation, at least locally *(low confidence)*. Stronger positive trends in surface solar radiation are detected in spring over western Mediterranean Basin, explained by a decrease in cloud optical depth for this season over this basin (Kambezidis et al. 2016), and despite an averaged positive trend of surface solar radiation observed over the eastern basin, more uncertainty exists due to the lack of observations of both clouds and aerosols (Alexandri et al. 2017).

2.2.3.2 Projected change in surface radiation

In future climate projections, anthropogenic aerosol loads over the Mediterranean are expected to keep on decreasing *(high confidence)* because of decreases in anthropogenic emissions in Europe (Shindell et al. 2013). The decrease is expected to be more pronounced in the near future with an aerosol optical depth anomaly between -0.12 and -0.03 (5-95% uncertainty range) for the 2040-2059 period with respect to the reference period 1980-1999 *(Fig. 2.4)*, and will slow down in the far future (between -0.18 and -0.04, 2080-2099 vs 1980-1999). The evolution of natural aerosols is more uncertain, due to current unknown future evolution of the desert dust *(Section 2.3.2)*.

Total cloud cover is also expected to decrease during the 21st century over the Mediterranean *(medium confidence)* (Boé and Terray 2014; Enriquez-Alonso et al. 2016; Bartók et al. 2017; Hentgen et al. 2019). This is consistent with the northward expansion of the Hadley cell (Sánchez-Lorenzo et al. 2017; D'Agostino et al. 2020) *(Section 2.2.2)* and with enhanced lower tropospheric drying (Hentgen et al. 2019). The expected anomaly in cloud cover for the mid-21st century ranges from -4.9 to -0.2% in the RCP8.5 (5-95% uncertainty range, *Fig. 2.4*), because of the difficulty of models to capture the spatial variability of the cloudiness evolution (Bartók et al. 2017).

As projected by GCMs, surface solar radiation is expected to continue increasing in the 21st century, especially in the near future *(medium confidence)*. The anomaly is between 0.6 and 7.7 W m^{-2} for the period 2020-2039 in the RCP8.5 (5-95% uncertainty range), in line with the decrease in anthropogenic aerosols (Boé et al. 2020; Gutiérrez et al. 2020) *(Fig. 2.4)*. However, this evolution is not shared by all RCMs (Bartók et al. 2017; Gutiérrez et al. 2020). Only regional climate models which consider aerosol dynamics simulate the increase in surface solar radiation as the global models (Boé et al. 2020; Gutiérrez et al. 2020). This increase in future surface solar radiation is reinforced by an expected decrease in cloud cover (Enriquez-Alonso et al. 2016), despite a compensational effect of increased absorption in clear sky conditions due to higher water vapor content in the atmosphere (Haywood et al. 2011) *(Fig. 2.4)*.

2.2.4 Temperature and related extremes

2.2.4.1 Observed temperature changes

Climate reconstructions, ground-based observations, reanalysis and remote-sensing datasets all corroborate the transition to warmer conditions during the 20th century and that warming has accelerated during the last decades *(high confidence)*. Basin-wide, annual mean temperatures are now 1.5°C above late 19th century levels *(Box 2.1, Fig. 2.33)*. Particularly after the 1980s, regional warming has accelerated and increases at a higher rate than the global average (Lelieveld et al. 2012; Lionello et al. 2012a; Zittis and Hadjinicolaou 2017; Cramer et al. 2018; Lionello and Scarascia 2018; Zittis et al. 2019). These studies present a strong consensus that the recent observed warming is robust throughout the region analysis, though magnitude and level of significance of the observed temperature trends in the Mediterranean varies depending (a) on the region, country or station under consideration, (b) on the type of data set investigated and (c) on the season and period of analysis.

Solar forcing and large volcanic eruptions are found to have a strong influence on the Mediterranean temperature variability over the last centuries (Trouet 2014). A combination of climate reconstructions, documentary sources and observed data suggests that looking at the long-term timescale (e.g., over the last 500 years), warm periods are not exceptional for the Mediterranean, which is characterized by a sequence of warming–cooling cycles (Luterbacher and Xoplaki 2003; Camuffo et al. 2010; Lelieveld et al. 2012). A study of summer temperature since Roman times shows that although the mean 20th century European (including the northern part of the Mediterranean Basin) was not significantly warmer than some

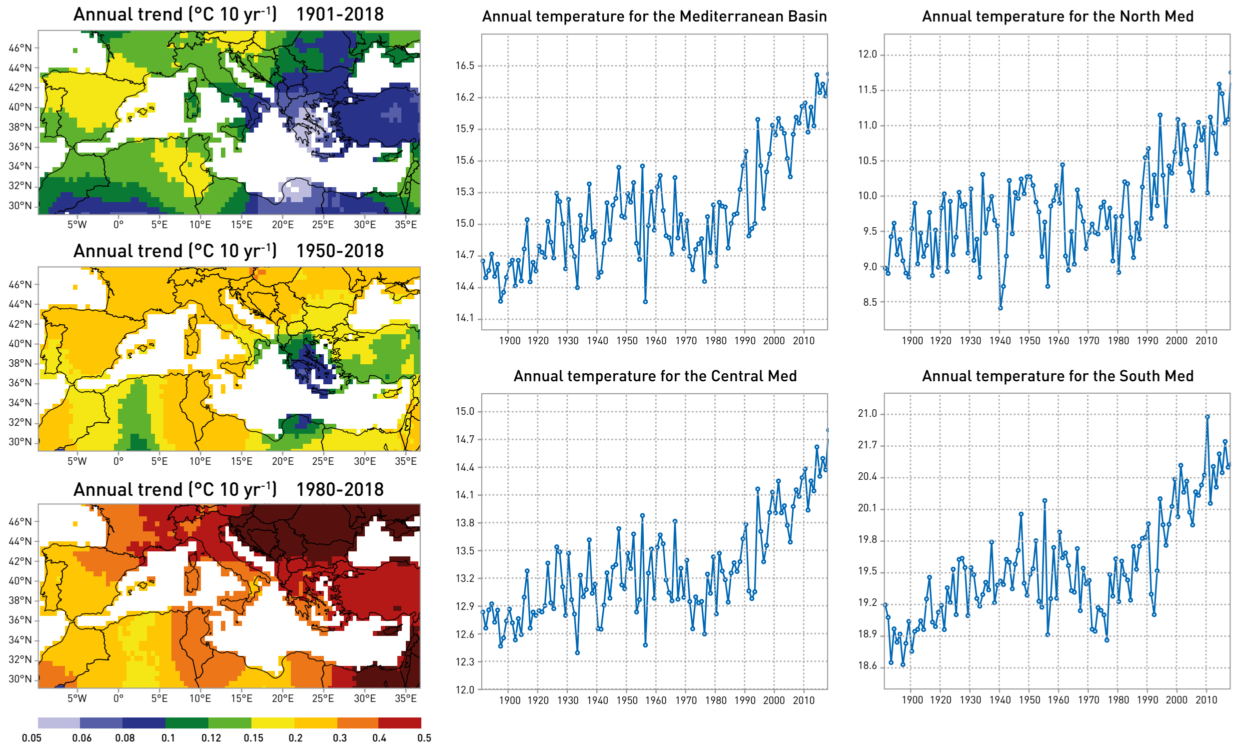


Figure 2.5 | Observed temperature trends (left panels) and time-series of temperature over land for the Mediterranean based on the Climatic Research Unit (CRU) (Harris et al. 2020) gridded observations. Time-series refer to the whole Mediterranean as defined in the left panels and for three Mediterranean sub-regions *(Fig. 2.1)*: North (FR, ALBA), Central (IB, CM, AN) and South Mediterranean (WM, EM, LE).

earlier centuries, there are no earlier 30-year periods found to be warmer than the most recent 3 decades (Luterbacher et al. 2016).

Recent climatic trends are clear, particularly after the 1980s. Over these last decades, according to different type of observations, significant positive trends of the order of 0.1-0.5°C per decade have been identified *(Fig. 2.5)* (Nasrallah and Balling 1993; Saaroni et al. 2003; Feidas et al. 2004; Brunetti et al. 2006; El Kenawy et al. 2009; Tanarhte et al. 2012; Lelieveld et al. 2012; Lionello 2012; Donat et al. 2014; Mariotti et al. 2015; Lionello and Scarascia 2018; Bilbao et al. 2019). In addition, for parts of the basin, there is some evidence that the diurnal temperature range has also changed (Price et al. 1999; Bilbao et al. 2019; Sun et al. 2019).

Besides mean values, hot and cold extremes have also become warmer, while in particular there is strong evidence and consensus that heat waves have become more frequent and severe. Various climatic indicators show significant increasing trends of extreme heat events characteristics (e.g., duration, frequency and intensity). The number of warm and tropical nights has also increased over most Mediterranean locations including Iberia, north Africa, Italy, Malta, Greece, Anatolia and the Levant (Kostopoulou and Jones 2007; Bartolini et al. 2008; Kuglitsch et al. 2010; El Kenawy et al. 2011; Galdies 2012; Donat et al. 2014; Filahi et al. 2015; Lelieveld et al. 2016; Ceccherini et al. 2017; Nashwan et al. 2018; Tolika 2019). Parts of the region were impacted by some of the most severe record-breaking weather events of the last decade, mainly related with summer heat extremes (Coumou and Rahmstorf 2012). Considering only winter, some studies that suggest a different behavior of hot and cold extremes between the eastern and western parts of the Mediterranean, with negative temperature trends in the former and positive trends in the latter (Hertig et al. 2010; Efthymiadis et al. 2011), but these studies do not include the most recent warm decades.

2.2.4.2 Future temperatures

According to future projections, the greater Mediterranean Basin is among the most responsive regions to global warming. Previous studies have identified the region as one of the most prominent climate change hot spots (Giorgi 2006; Lionello et al. 2006; Giorgi and Lionello 2008; Diffenbaugh and Giorgi 2012; Lionello 2012; Lionello and Scarascia

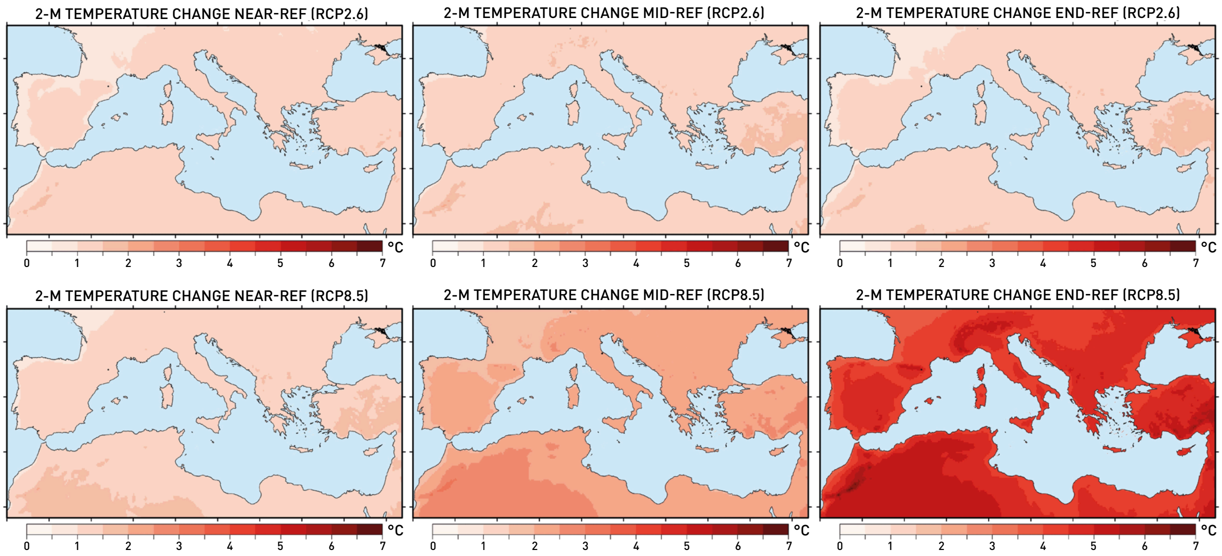


Figure 2.6 | Projected changes in annual temperature between the recent past reference period (REF: 1980-1999) and three future sub-periods (NEAR: 2020-2039, MID: 2040-2059, END: 2080-2099), based on the ensemble mean of EURO-CORDEX 0.11° simulations for pathways RCP2.6 (top panels) and RCP8.5 (bottom panels).

2018). Multi-model ensembles of climate simulations indicate that widespread warming will almost certainly occur in the Mediterranean in the 21st century *(high confidence)*, though, climate models might overestimate actual values in warm and dry conditions (Boberg and Christensen 2012).

The warming level of warming strongly depends on the reference period definition, the future time horizon and the level of greenhouse gas forcing (Christensen et al. 2007; Giorgi and Lionello 2008; Collins et al. 2013; Dubrovský et al. 2014; Jacobeit et al. 2014; Mariotti et al. 2015; Ozturk et al. 2015; Lionello and Scarascia 2018; Zittis et al. 2019). A quantitative estimation based on state-of-the-art EURO-CORDEX regional simulations is presented in *Table 2.1* and *Fig. 2.6-2.7*. Over land regions, a robust and significant warming of the range of 0.9-5.6°C (with respect to the reference period 1980-1999) is suggested for the future. The robustness and significance of the climate signal is much higher for air temperature rather than other variables such as precipitation (Knutti and Sedláček 2012; Lelieveld et al. 2016). There are strong indications and a general consensus that regional warming will continue faster than the global average and will exceed the global mean value by 20% on an annual basin and 50% in summer *(high confidence)* (Vautard et al. 2014; Dosio and Fischer 2018; Lionello and Scarascia 2018; Nikulin et al. 2018). Daytime temperatures are expected to increase more than nighttime temperatures, indicating an increase of the amplitude of the diurnal temperature range (Lionello and Scarascia 2018).

Changes in the occurrence of extreme events closely follow changes in inter-annual variability. Therefore, such changes can be also considered as a proxy measure of seasonal extremes (Schär et al. 2004; Giorgi 2006). The intensity of extreme temperature is projected to increase more rapidly than the intensity of more moderate temperatures over the continental interior due such increases in temperature variability (Beniston et al. 2007).

Projected changes in extreme temperature indicators suggest that the frequency and severity of heat waves will increase *(high confidence)* (Diffenbaugh et al. 2007; Goubanova and Li 2007; Giorgi and Lionello 2008; Fischer and Schär 2010; Diffenbaugh and Giorgi 2012; Sillmann et al. 2013; Russo et al. 2014; Jacob et al. 2014; Kostopoulou et al. 2014; Zittis et al. 2016; Lelieveld et al. 2016; Ouzeau et al. 2016; Lionello and Scarascia 2020). According to projections for a business-as-usual scenario, summer daily maximum temperature is expected to increase up to 7°C by the end of the 21st century in comparison with the recent past (Sillmann et al. 2013; Lelieveld et al. 2016). Besides warmer daytime temperature maxima, parts of the Mediterranean will likely face an increase of more than 60% in the number of tropical nights. Increase of warm temperature extremes will be dramatic particularly in summer and with a 4°C global warming almost all nights will be warm and there will be no cold days (Sillmann et al. 2013; Dosio and Fischer 2018; Lionello and Scarascia 2020).

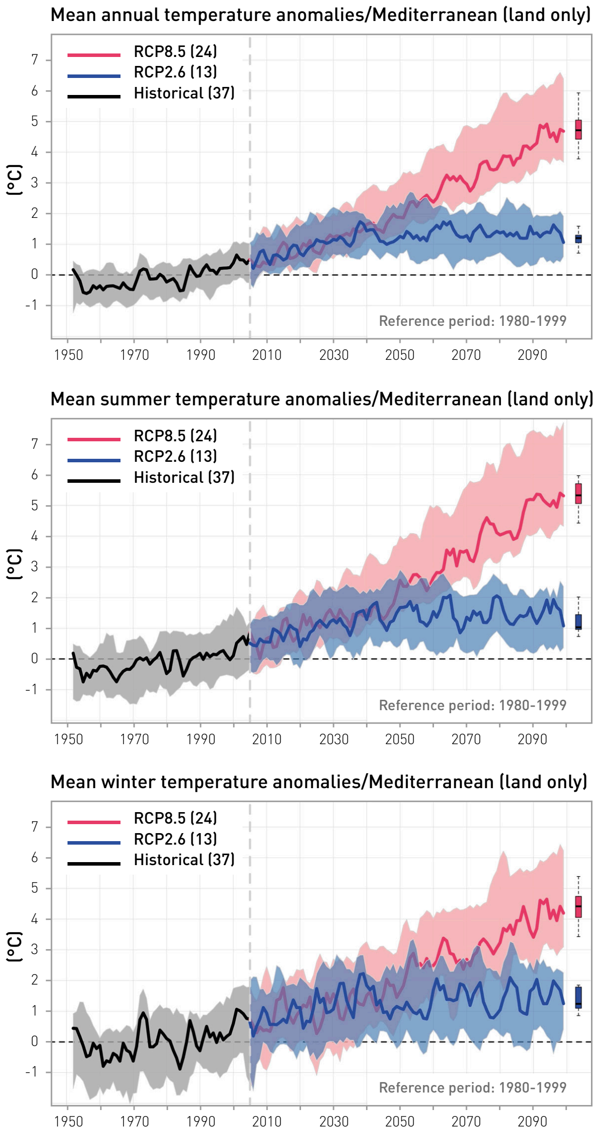


Figure 2.7 | Time-series of simulated mean annual (top panel), summer (middle panel) and winter (bottom panel) temperature averaged over the Mediterranean based on EURO-CORDEX 0.11° simulations for historical times (black curve) and future pathways RCP2.6 (blue curve) and RCP8.5 (red curve). Solid lines indicate the ensemble means and shaded areas the spread of the simulations. Box-plots represent the averages over the decade 2091-2100 in terms of model spread.

Warming is projected to be milder in winters and much stronger during summers. This is mainly attributed to land-atmosphere interactions and the transition to drier conditions (Seneviratne et al. 2006; Jaeger and Seneviratne 2011; Quesada et al. 2012; Zittis et al. 2014). Another important feedback, particularly for southern Mediterranean, is the coupling of longwave radiation between the desert soil surface and lower atmosphere which amplifies warming and intensifies the summertime heat low over the Sahara (Cook and Vizy 2015; Evan et al. 2015). The exceptional summertime warming over parts of the region is also likely associated with a thermal low, which is explained by the widening of the Persian trough that extends from South Asia to the eastern Mediterranean, and is projected to expand westward and combine with the intensifying thermal low over the Sahara (Lelieveld et al. 2016).

2.2.5 Precipitation, related extremes and the water cycle

2.2.5.1 Observed trends in precipitation

Observed precipitation trends during the full or portions of the 20th century up to present day, covering the full or portions of the Mediterranean Basin, are available in gridded format from various sources, such as CRU, UDEL, E-OBS, EURO4M. Annual, DJF (December-January-February) and JJA (June-July-August) precipitation trends from the CRU dataset for different time periods, 1950-2018 and 1980-2018 are shown in *Fig. 2.8*. *Fig. 2.9* shows the temporal evolution of land precipitation (1901-2018) averaged over the full Mediterranean area and its northern, central and southern portions (defined as the sum of the three northern, central and southern regions of *Fig. 2.8*, respectively).

The sign of the observed precipitation trend exhibits pronounced spatial variability and depends on the time period and season considered *(Fig. 2.8)*. For example, the period 1950-2018 shows a prevailing decreasing trend over most of the Mediterranean Basin of annual and winter precipitation, which is reversed over large portions of the basin if we only consider the period 1980-2018. This is because of the marked multidecadal variability of precipitation in the Mediterranean, which may actually mask trends induced by greenhouse gas emissions. The prominent role of multidecadal variability is also evident when precipitation is regionally averaged *(Fig. 2.9)*. In this case the most evident trend is a decrease of winter precipitation over the central and southern portions of the basin since the second half of the 20th century. Overall, because of the marked multidecadal variability of precipitation and the small magnitude of trends, the confidence in the detection of trends from greenhouse gas emissions for the historical past is low (Lelieveld et al. 2012; Lionello et al. 2012a; Peña-Angulo et al. 2020; Vicente-Serrano et al. 2020).

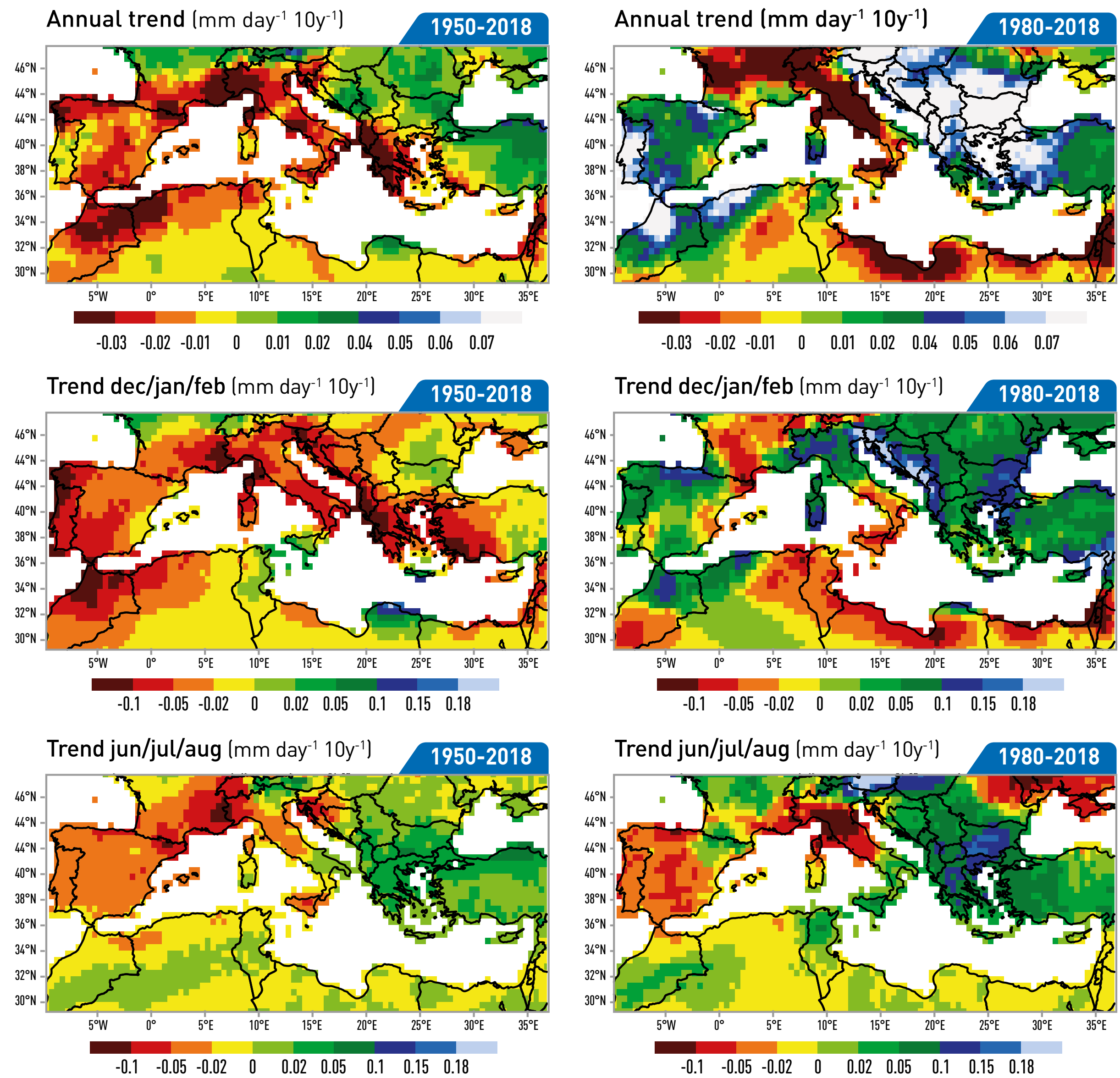


Figure 2.8 | Observed annual, DJF, JJA precipitation trends from the CRU dataset. Left and right panels consider the 1950-2018 and 1980-2018 periods, respectively.

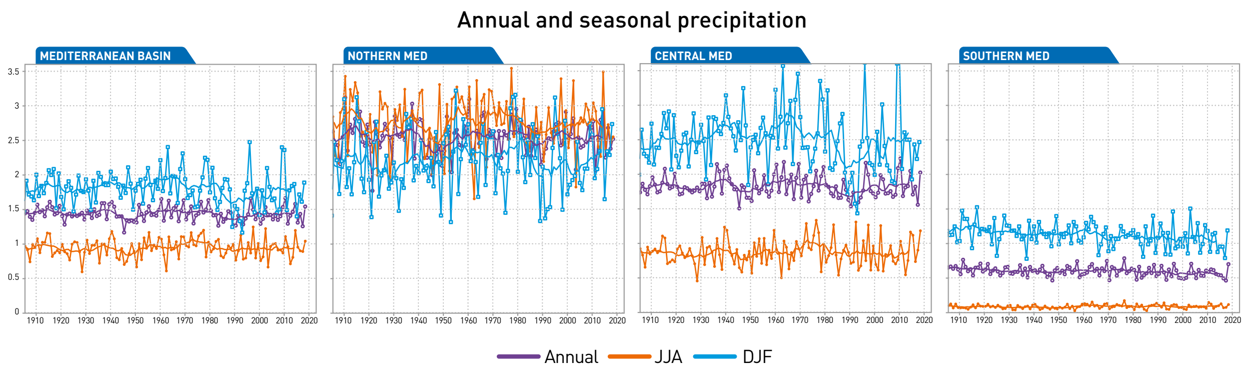


Figure 2.9 | Time series of annual, DJF, and JJA precipitation over land from the CRU dataset. Time-series refer to the whole Mediterranean as defined in the left panels and three Mediterranean sub-regions *(Fig. 2.1)*: North (FR, ALBA), Central (IB, CM, AN) and South Mediterranean (WM, EM, LE).

2.2.5.2 Future precipitation

Mean precipitation changes

21st century precipitation projections for the Mediterranean region have been produced based on GCM and RCM ensembles of experiments. Analyses of GCM projections over the Mediterranean region have been conducted for CMIP3 (Giorgi and Coppola 2007; Giorgi and Lionello 2008; Mariotti et al. 2008) and CMIP5 (Mariotti et al. 2015; Lionello and Scarascia 2018). Several generations of RCM-based projections for the EURO-Mediterranean region are also available from projects such as PRUDENCE (Déqué 2007), ENSEMBLES (Déqué et al. 2012) and EURO-CORDEX (Jacob et al. 2014). In addition, projections based on coupled regional atmosphere-ocean models have been conducted as part of the CIRCE (Gualdi et al. 2013) and MED-CORDEX (Ruti et al. 2016) projects.

A consistent dominant signal emerges from these projections, consisting of a predominant drying through the entire Mediterranean Basin in the warm seasons (April through September, with largest magnitude in JJA), drying in most central and southern areas along with wetting in the northernmost regions (e.g., the Alps) in the winter season *(medium confidence)*. This large-scale pattern of change is illustrated for the Euro-CORDEX dataset, at the annually averaged scale, in *Fig. 2.10* for the RCP2.6 and RCP8.5 scenarios and different future time slices. *Table 2.1* provides quantitative values of precipitation change for different scenarios and model ensembles over the Mediterranean region.

In general, the patterns of change intensify in magnitude from the near future to the far future time slices and from the low to high greenhouse gas emission scenarios, i.e., they intensify with the anthropogenic forcing and resulting global warming. As a result, for example, at the Mediterranean scale, the CMIP5 ensemble yields a decrease of annual precipitation over the Mediterranean area of about 4% per degree of global warming (Lionello and Scarascia 2018).

The magnitude and pattern of precipitation decrease vary widely across models. For example, the summer precipitation reduction in the CMIP3 and CMIP5 datasets for the high-end greenhouse gas emission scenarios (roughly equivalent to the RCP8.5) varied from less than 10% to over 40% across models (IPCC 2007; Giorgi and Lionello 2008; Lionello and Scarascia 2018). Although qualitatively consistent, different ensembles show different sensitivities over the Mediterranean. The CMIP5 GCM ensemble produced a less pronounced summer drying than the CMIP3 one, when expressed in terms of change per degree of global warming (IPCC 2013b). RCM-based projections, e.g., as part of Euro-CORDEX yield a lower drying that GCM-based ones, with reduced areas

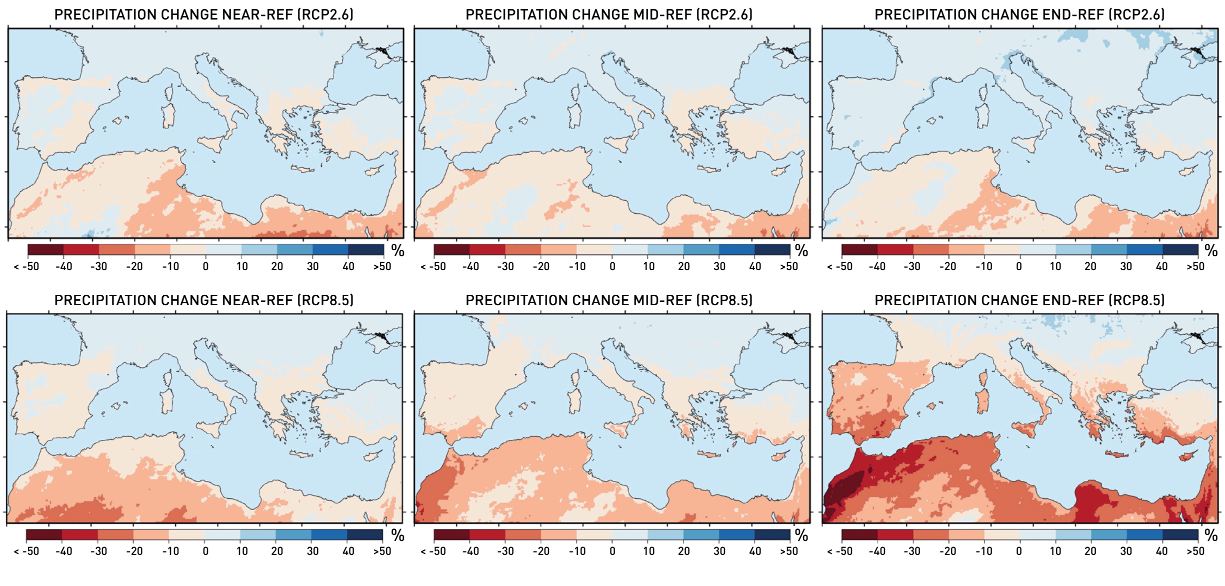


Figure 2.10 | Maps showing EURO-CORDEX-based change in annual, winter and summer precipitation change, for the RCP2.6 and RCP8.5 scenarios and the near-future, mid-term and far-future with respect to the reference period.

of precipitation decrease in the future compared to the GCMs. Considering a range of global warming of 0.9-5.6°C *(Section 2.2.4.2)* at the end of the 21st century (with respect to the reference period 1980-1999) and a decrease of 4% per degree of global warming, this gives a reduction between 4 and 22% for the Mediterranean annual precipitation (on land).

Fig. 2.11 shows the temporal evolution of Mediterranean scale precipitation (RCP2.6 and RCP8.5) from 1950 to 2100 in the EURO-CORDEX dataset, and presents both mean and inter-model range of data. The large inter-model spread includes some positive values and two scenarios start to separate, at least in an ensemble mean sense, only after the mid of the 21st century. Therefore, while it is possible to assess that precipitation will likely decrease over the Mediterranean Basin, at least under the higher end scenarios, it is difficult to assign robust quantitative values, especially at sub regional scale.

The uncertainty in projections is even larger as we move to the local scale because of the effects of local forcings, e.g., topography and coastlines. For example, focusing on the high elevations of the Alpine chain, Giorgi et al. (2016) found that the RCM projections at 12 km resolution exhibit an increase of summer precipitation in areas where the GCMs project a decrease. This is due to the occurrence of increased convection related to high elevation warming and heating. As another example, increases of cold season precipitation in the upwind side of mountain chains and decreases in the lee side have been found by the high-resolution simulations of Gao et al. (2006) in response to the topographically-forced precipitation shadowing effect. These results were confirmed by the analysis of the EURO-CORDEX ensemble projections (Kotlarski et al. 2019). In other words, high resolution RCM projections suggest that care needs to be taken when extending large scale patterns to the local scale, since local scale changes can be heavily affected by topography and coast lines.

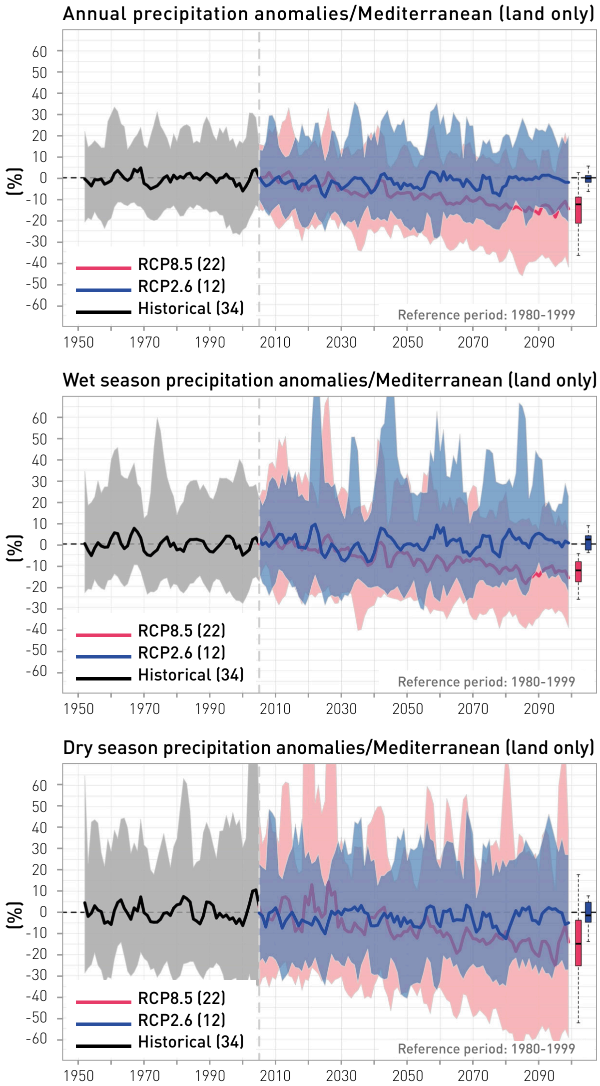


Figure 2.11 | Time-series of simulated mean annual (top panel), wet (middle panel) and dry (bottom panel) season over the Mediterranean land areas based on EURO-CORDEX 0.11° simulations for historical times (black curve) and future pathways RCP2.6 (blue curve) and RCP8.5 (red curve). Solid lines indicate the ensemble means and shaded areas the spread of the simulations. Box-plots represent the averages over the decade 2091-2100 in terms of model spread.

Variability and extremes

Several studies have assessed changes in interannual variability of precipitation over the Mediterranean region in GCM-based projections, mostly using as a measure of variability the coefficient of variation (i.e., the interannual standard deviation divided by the mean), which removes the strong dependence of precipitation standard deviation from the mean (Räisänen 2002; Giorgi and Bi 2005; Giorgi and Coppola 2009; Giorgi et al. 2019). They all found a prevailing increase in precipitation variability over the Mediterranean, especially over areas showing strong precipitation decreases, which thus appears to be a robust response in global climate projections. However, this result may not apply if the metrics used to measure variability is the standard deviation.

Giorgi et al. (2011, 2014) carried out an analysis of changes in different hydroclimatic indices from an ensemble of CMIP5 projections and, on an annual basis, consistently found an increase in mean daily precipitation intensity and 95th percentile of daily precipitation, a decrease in precipitation frequency and wet spell length and an increase in the number of dry days and dry spell length *(medium confidence)*. The hydroclimatic intensity index introduced by Giorgi et al. (2011), which is essentially the product of precipitation intensity and mean dry spell length, shows a consistent increase throughout the Mediterranean Basin. These conclusions based on GCM projections, were essentially confirmed by an analysis of high-resolution EURO-CORDEX projections by Jacob et al. (2014), except for a slight decrease of 95th percentile daily precipitation over some areas of the Iberian, Italian and Hellenic peninsulas and southern France in summer. A recent study (Lionello and Scarascia 2020), based on CMIP5, shows that global warming will further increase the existing difference in intensity of precipitation and hydrological extremes between North and South Mediterranean areas. Both the daily precipitation intensity the total precipitation during extreme events are already larger in the North than in the South Mediterranean areas, and differences will increase with global warming. The projected increase of dry spell length is larger in the south than in the North Mediterranean *(medium confidence)*.

In conclusion, *(high confidence)* both global and regional climate projections indicate a predominant shift towards a precipitation regime of higher interannual variability (when measure by the coefficient of variation), higher intensity of precipitation and greater extremes (especially in winter, spring and fall, but not in the southern areas), decreased precipitation frequency and longer dry spells (especially in summer). This hydroclimatic response to global warming is greater for the RCP8.5 than the RCP2.6 scenario and for the far future vs. the near future time slices *(high confidence)*.

2.2.5.3 Changes in evaporation, net water losses over sea and over land

Evaporation in the Mediterranean not only provides moisture locally, but also results in a net export of water to neighboring areas, primarily to the South and East (Mariotti et al. 2002a; Nieto et al. 2006). The Mediterranean Sea is the dominant regional evaporation source, and changes in Mediterranean Sea evaporation impact the sea's water, salt and heat budgets. Large-scale internal climate variability and greenhouse gas forced global change have been primary drivers of Mediterranean evaporation changes during the 20th century and into the 21st century, while local processes have acted to modulate those effects (Mariotti and Dell'Aquila 2012). Overall, the net surface water loss (evaporation minus precipitation over the sea) has increased over most of the Mediterranean surface, mainly due to a decrease of precipitation during the period 1960-1990 and a strong evaporation increase since the mid-seventies due to local warming (Mariotti 2010; Sevault et al. 2014; Mariotti et al. 2015; Skliris et al. 2018). The freshwater discharge due to the river runoff has also decreased (Ludwig et al. 2009). Projected regional warming trends point to continuing increases in Mediterranean Sea evaporation, land drying in southern areas during summer and a net regional water loss.

Observations-based estimates of Mediterranean Sea evaporation from the OAFlux Programme starting in 1958 point to decadal variations with a minimum around 1965-1975, and an overall positive trend of about 10% decade^{-1} (0.06 mm day^{-1} decade^{-1}) (Mariotti 2010). Since the mid-1970s, there is a substantial evaporation increase (0.1-0.2 mm day^{-1} decade^{-1}) with a tendency toward higher rates of increase during the 1990s. Much of the evaporation increase since the mid-1970s has been in early winter, especially in the Ligurian Sea, Adriatic Sea, and southeastern Mediterranean. The evaporation increase has resulted in a rate of increase in freshwater fluxes during 1979-2006 estimated in the range of 0.1-0.3 mm day^{-1} decade^{-1}. Increases in sea surface temperatures have primarily driven these evaporation changes via changes in the surface humidity gradient. Based on OAFlux data, the estimated Mediterranean mean rate of evaporation change in relation to the warming is about 0.7 mm day^{-1} K^{-1} (or 25% K^{-1}) over the period of 1958-2006. An increase in net Gibraltar water flux to compensate for the overall increase in fresh water loss has been derived (Fenoglio-Marc et al. 2013).

For the land surrounding the Mediterranean Sea, past evapotranspiration changes are regionally and seasonally dependent and largely follow precipitation trends, since soil moisture availability is a primary limiting factor. Increasing soil-moisture limitations seem to have driven recent global evapotranspiration decline and increased drought tendencies over the Mediterranean region (Sheffield and Wood 2008; Vicente-Serrano et al. 2014; Samaniego et al. 2018). Evapotranspiration estimates from French National Centre for Meteorological Research (CNRM) (Douville et al. 2013) display a tendency for evapotranspiration to increase during winter since the 1970. For the summer there is a progressive decrease (Mariotti et al. 2015).

The future evolution of the Mediterranean Sea physical characteristics is strongly related to the evolution of the air-sea and land-sea exchanges of water and heat. For the Mediterranean Sea, the net surface water loss by the sea is constituted by the combination of the evaporation over the sea, the precipitation over the sea, the river runoff and the Bosphorus Strait net transport. Increase in the net surface water loss by the sea is expected in the future due to a decrease in precipitation and in river runoff and an increase in evaporation (Mariotti et al. 2008, 2015; Sánchez-Gomez et al. 2009; Elguindi et al. 2011; Dubois et al. 2012; Planton et al. 2012; Adloff et al. 2015). Relative to the 20th century, this increase ranges from +8 to +35% for the mid-21st century (2020-2049) and from +20 to +60% at the end of the 21st century (2070-2099) in the medium-range A1B socio-economic scenarios (Planton et al. 2012).

To a first order, CMIP5 projections are largely similar to those based on CMIP3 (Mariotti et al. 2008) and consistent with those based on regional model downscaling (Sánchez-Gomez et al. 2009; Dell'Aquila et al. 2018). By 2071-2098, the Mediterranean Sea evaporation is projected to increase during all seasons and especially in winter (projected annual-mean increase is 0.25 ±0.08 mm day^{-1}) (Mariotti et al. 2015). Note that future change in the Nile freshwater inflow remains unknown due to the impossibility so far to accurately model the influence of regional water- and land-use anthropogenic activities on its past and future evolution (Somot et al. 2006; Dubois et al. 2012).

Over land, evapotranspiration projections present mixed changes, with a precipitation-driven increase in winter over Northern areas, and a decrease in summer over many land areas, especially over Spain, western Northern Africa and Turkey. Evapotranspiration increase will be also driven by increase of atmospheric evaporative demand (Vicente-Serrano et al. 2015). These evapotranspiration changes have been linked to a projected northward expansion of the Mediterranean land type (Alessandri et al. 2015) and regional surface vegetation changes (Anav and Mariotti 2011).

Changes in precipitation and evaporation over the Mediterranean Basin will lead to changes in drought occurrence. Drought can be of different types, such as meteorological, hydrological and agricultural drought, which can often be difficult to separate. Here we focus on meteorological drought, essentially measured by indices of monthly, seasonal up to annual precipitation deficits, such as the precipitation index (PI) or the standardized precipitation index (SPI). The Mediterranean Basin, is impacted by frequent drought episodes due to the strong inter-annual variability of rainfall in this region, and a trend towards drier conditions and increased meteorological drought occurrence after the 1970s over the Mediterranean Basin was found based on analyses of observations (Vicente-Serrano et al. 2011; Hoerling et al. 2012; Spinoni et al. 2015; Caloiero et al. 2018). Due to the pronounced interannual and decadal variability of Mediterranean precipitation, the robustness of this result needs to be confirmed, and may differ for different areas of the Mediterranean.

Concerning projections, since most model simulations indicate a trend towards drier conditions over the Mediterranean, especially in the warm season and over the southern areas, it is expected that the frequency and intensity of meteorological drought will increase under warmer climates. This has been confirmed *(high confidence)* by extensive analyses of precipitation projections with both global and regional climate models (Giorgi and Lionello 2008; Mariotti et al. 2008; Dai 2013; Dubrovský et al. 2014; Spinoni et al. 2015, 2018; Stagge et al. 2015; Quintana-Seguí et al. 2016; Naumann et al. 2018; Lionello and Scarascia 2020).

2.2.6 The cryosphere

2.2.6.1 Observed trends in the cryosphere

After the peak of the "little ice age" (~1,400-1,860 AD, Ivy-Ochs et al. 2009) increasing summer and mean annual air temperature led to a dramatic reduction in the area and volume of glaciers across high mountains of the Mediterranean (Hughes 2018). Short glacier readvances were observed in the 1890s, 1920s, 1970s and 1980s (Zemp et al. 2008). Deglaciation rate generally accelerated in recent decades (Rabatel et al. 2013), although the patterns of glacier retreat were complicated by the sensitivities of glaciers to different climatic regimes (Hughes 2018). As glacier retreats to cirque headwalls, it becomes more dominated by local topoclimatic controls, especially avalanching snow. Nevertheless, a complete loss of glaciers in some low-latitude mountain ranges has already occurred (Rabatel et al. 2013), accompanied by a shorter duration of seasonal snow cover (Brown and Mote 2009). Several small cirque glaciers existing in the southern Dinaric Alps, Balkan Peninsula, Turkey, Pyrenees, Sierra Nevada and the Apennines disappeared across the 20th century and in the last decades (Hughes 2018). In the Alps, glaciers covered 4,470 km^2 in 1850, 2,909 km^2 in the 1970s and 2,270 km^2 in 2000, meaning a 50% loss from 1850 to 2000 (Zemp et al. 2008). Few very small glaciers still exist in mountains of Montenegro and Albania. Elsewhere, perennial ice and snow patches still survive and attest to how close some Mediterranean mountains are to supporting small glaciers even where the equilibrium line altitude (ELA) is located above the highest peaks (Hughes 2018). ELA raised by about 170 m in the western Alps over the period 1984-2010 (Rabatel et al. 2013) while in the southeastern Alps change in the ELA was in the order of about +250 m between the 1980s and 2010 (Colucci and Žebre 2016).

Temperature increase led to a shift of periglacial processes to higher elevations as well as degradation of mountain permafrost in high mountain environments (Oliva et al. 2018). In the western and central Mediterranean, permanently frozen ground is now rarely found below 2,500 m. Alpine permafrost belt is detected above 2,630 m in northern aspects and 2,800 m in southern ones and in the Pyrenees, above 2,400 m in the Southern Alps, above ~2,350 m on Rila Mountain and ~2,700 m on Mount Olympus and above 2,800-3,400 m in northeastern Turkey and central Anatolia (Oliva et al. 2018). No permafrost belt is found in the highest mountains in southern Europe (Sierra Nevada) and northern Africa (Atlas) where permanent frozen conditions are only found in the form of isolated patches at the highest elevations at 3,000-3,100 m (Oliva et al. 2016) and 3,800 m (Vieira et al. 2017), respectively. Certain climate conditions (i.e., reduced snow cover) can favour the presence of permafrost patches at relatively low elevations in the Central Apennines or by lithological conditions (i.e., volcanic sediments, karst lithology), as detected in the highest active European volcano (Mt. Etna) at elevations above 2,900 m (Maggi et al. 2018) or in limestone dominated mountains across the Mediterranean (Colucci and Guglielmin 2019).

2.2.6.2 Future conditions in the cryosphere

Mountain glaciers in the Mediterranean region are projected to continue losing mass in the 21st century until complete disappearance of most mountain glaciers by the end of the century *(very high confidence).* A recent multi-model projection exercise (GlacierMIP, Hock et al. 2019) indicates that relative volume losses by 2100 (average of model runs ±1 standard deviation) are of the order of 69 ±19% for RCP2.6 and 93 ±10% for RCP8.5. This indicates that, even under scenarios with strong reduction in greenhouse gas emissions, only glaciers at the highest elevation will persist at the end of the 21st century. For mid-century, changes depend far less on the climate scenario, with reductions of the order of 50 ±20% for RCP2.6 and 60 ±20% for RCP8.5.

Projected changes of the mountain snow cover are studied based on climate model experiments, either directly from GCM or RCM output, or following downscaling and the use of snowpack models. Future changes in snow conditions are mostly driven by changes in meteorological drivers. The projections generally do not specifically account for future changes in the deposition rate of light absorbing particles on snow and associated changes in snow albedo. At lower elevation, under the current multi-annual mean rain/snow transition elevation, the water mass of the snow cover is projected to decline by 25% likely range: (10-40%, between the recent past period (1986-2005) and the near future (2031-2050), regardless of the climate scenario). By the end of the 21st century (2081-2100), reductions of up to 80% (likely range 50-90%) are expected under RCP8.5, 50% (likely range from 30-70%) under RCP4.5 and 30% (likely range 10-40%) under RCP2.6 (Beniston et al. 2018; Hanzer et al. 2018; Verfaillie et al. 2018). At higher elevations, projected reductions are smaller *(high confidence),* as temperature increases at higher elevations affect the ablation component of snow mass evolution (in particular, melt and sublimation), rather than the onset and accumulation component. The strong interannual variability of snow conditions is projected to remain a key feature of this cryospheric component throughout the 21st century *(high confidence).*

In the Mediterranean domain, permafrost is only located in the mountains, often patchy and confined to areas of rugged topography, including cliffs. In contrast to glacier and snow cover, climate projections of the ground thermal regime have not been performed in a comprehensive manner, using a cascade of climate models and impact models. Evidence stems from small-scale studies, but all studies points towards increased permafrost thaw in mountain environments, following surface air temperature changes (Marmy et al. 2016; Beniston et al. 2018). Future changes in mountain permafrost have major implication for natural hazards (slope instabilities).

2.2.7 Ocean hydrology

The Mediterranean Sea can be considered as a laboratory of the global ocean (Lacombe 1990; Béthoux et al. 1999) as it shows many key and interesting oceanic physical processes, such the open-sea deep convection occurring in some areas (Gulf of Lion, South Adriatic, and the Cretan Sea) leading to the formation of cold and salty deep-water masses (Tsimplis et al. 2006; Schroeder et al. 2012). The dominance of evaporation in the Mediterranean Sea and intermediate water formation (in the Rhodes Gyre) leads to an anti-estuarine thermohaline circulation with a surface layer with the Atlantic Water (AW, comparatively fresh and warm) and a layer with the Levantine Intermediate Water (LIW, very salty and comparatively cold) entering and exiting across the Gibraltar Strait, simultaneously.

In addition, the Mediterranean Sea is surrounded by various and complex topography channeling regional winds (Mistral, Tramontane, Bora, Meltem, Sirocco, Etesians) that define local circulations. The presence of complex coastlines, islands, narrow and shallow straits require adapted observation strategies and high-resolution modeling tools. Further, the Mediterranean Sea is also known to impact the Atlantic Ocean through the Mediter-

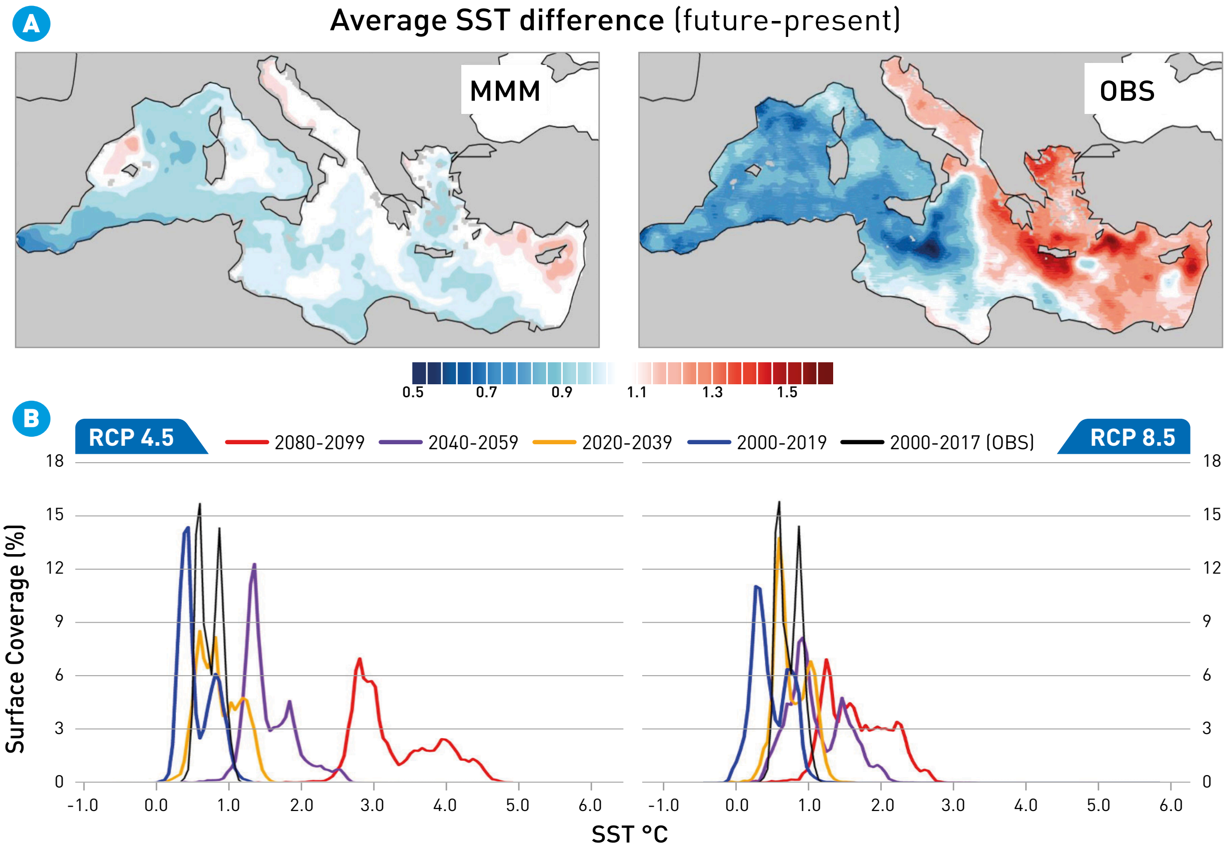


Figure 2.12 | A) Local amplification factor of the Mediterranean sea surface warming using 1980-1999 as reference period. Local sea surface warming values are divided by the basin-averaged warming value. Reddish (resp. blueish) colors mean that the area is warming more (resp. less) that the basin-average. The top panel is the ensemble mean of five Med-CORDEX coupled regional climate system models, further averaging warming rates of four 20-year long time periods (2000-2019, 2020-2039, 2040-2059, 2080-2099) and 2 scenarios (RCP4.5, RCP8.5). The bottom is based on the CMEMS observations (2000-2017). **B) Fraction of the Mediterranean Sea surface (in %) experiencing a given sea surface temperature change value (in °C),** compared to the reference period (1980-1999) for various periods (2000-2019 in blue, 2020-2039 in orange, 2040-2059 in purple, 2080-2099 in red) and for the scenarios RCP4.5 and RCP8.5 using the envelope of the results of 5 Med-CORDEX coupled regional climate system models. The change in the CMEMS observations is added for the period 2000-2017 (in black). Information is first aggregated at the yearly scale.

ranean Outflow Waters that flow into the Atlantic at about 1,000 m depth and are considered as a source of salt and heat for the Atlantic Ocean (Artale et al. 2006).

2.2.7.1 Observed change in marine waters

There is increasing evidence that Mediterranean water masses are becoming warmer and deep water masses saltier. This assertion is supported both by direct measurements (Béthoux and Gentili 1999; Rixen et al. 2005; Vargas-Yáñez et al. 2010, 2017) and by numerical simulations (Beuvier et al. 2010; Harzallah et al. 2018; Somot et al. 2018).

Since the 1980s upper layer temperature has increased (Rivetti et al. 2017; Vargas-Yáñez et al. 2017) as well as sea surface temperature (Marullo et al. 2010; Pastor et al. 2018), with acceleration since the 1990s (Macías et al. 2013). Since the beginning of the 1980s, the sea surface warming rate ranges between +0.29 and +0.44°C decade^{-1} on average over the whole Mediterranean Sea, depending on the studied period and on the reference data sets (Nabat et al. 2014; CEAM 2019; Darmaraki et al. 2019a). In the period 2000-2017 with respect to 1980-1999, all Mediterranean Sea areas show a positive yearly-mean sea surface temperature anomaly of at least +0.2°C. The sea surface warming has not been uniform, but mostly bimodal *(Fig. 2.12b)* with stronger trends in the eastern basin (Adriatic, Aegean, Levantine and North-East Ionian Seas, *Fig. 2.12a*), where some areas warmed by +1.2°C. Very local places in the Levantine Basin have warmed 50% more rapidly than the Mediterranean Sea average whereas a spot in the Ionian Sea has warmed 50% less than the basin average *(Fig. 2.12a)*. Note that the climate models currently underestimate the observed sea surface warming (blue and black lines in *Fig. 2.12b*) (Nabat et al. 2014; Sevault et al. 2014; Dell'Aquila et al. 2018).

In the Mediterranean Sea, periods of abnormally warm sea surface, also called "marine heat waves" have become more frequent, more intense, spatially more extended and more severe over the last decades (Oliver et al. 2018; Darmaraki et al. 2019a). To illustrate this trend, the most severe marine heat waves detected since 1982 are 2003, 2012, 2015, and 2017 (Bensoussan et al. 2019). In addition, 14 marine heat waves occurred during the 2008-2017 10-year period whereas only 2 occurred during the 1982-1991 period (Darmaraki et al. 2019a). Contrary to sea surface temperature, a corresponding sea surface salinity evolution has not been reported (Rixen et al. 2005; Sevault et al. 2014), except for specific locations (Ozer et al. 2017; Vargas-Yáñez et al. 2017).

Long-term trends in the Mediterranean Sea intermediate and deep hydrology have been detected, though they are affected by marked variability on decadal time scales, the Eastern Mediterranean Transient (EMT, Roether et al. 2007), Western Mediterranean Transition (WMT, Schroeder et al. 2016) and BiOS (Gačić et al. 2010) being probably the most known manifestations of this.

Since the mid 1990s the temperature and the salinity of the Levantine Intermediate Water (LIW) have increased by 0.53°C (*Fig. 2.13,* "Sicily Channel" panel) and 0.13 psu, i.e., with rates of 0.024°C yr^{-1} and 0.006 psu yr^{-1}, respectively (Schroeder et al. 2017). Such trends are at least one order of magnitude greater than those reported for the global ocean intermediate layer (Schroeder et al. 2017). The western basin deep waters are shown to have gradually increased their temperature and salinity since the 1950s (Rohling and Bryden 1992; Béthoux et al. 1998; Rixen et al. 2005; Marty and Chiavérini 2010), with an acceleration after the mid 1980s and an even stronger rate since 2005 due to an abrupt WMT (Marty and Chiavérini 2010; Borghini et al. 2014; Schroeder et al. 2016). Deep-water trends of 0.04°C ±0.001°C decade^{-1} and 0.015±0.003 psu decade^{-1} (since 1961) have been reported by comparing time series of deep CTD (conductivity-temperature-depth) casts (Borghini et al. 2014) (*Fig. 2.13*, bottom panel).

Changes in the Mediterranean water mass characteristics have a signature also in the water outflowing from the Mediterranean Sea through the Strait of Gibraltar (Millot et al. 2006; Naranjo et al. 2017). Mooring observations collected since 2004 (*Fig. 2.13*, "Gibraltar Strait" panel) show a positive trend in temperature and salinity of 7.7x10^{-3} °C yr^{-1} and 0.63x10^{-3} psu yr^{-1}, respectively (von Schuckmann et al. 2018). Since 2012 a noticeable increase of these trends is interpreted as the signal of the WMT (Naranjo et al. 2017). In addition, no significant changes in the strait transports (net exchange: Fenoglio-Marc et al. 2013; Boutov et al. 2014; Soto-Navarro et al. 2015) and surface circulation (Pascual et al. 2014) have been detected.

No significant trends in frequency of dense water formation events have been detected (Beuvier et al. 2010; Houpert et al. 2016; Somot et al. 2018; Dunić et al. 2019), although a strong interannual

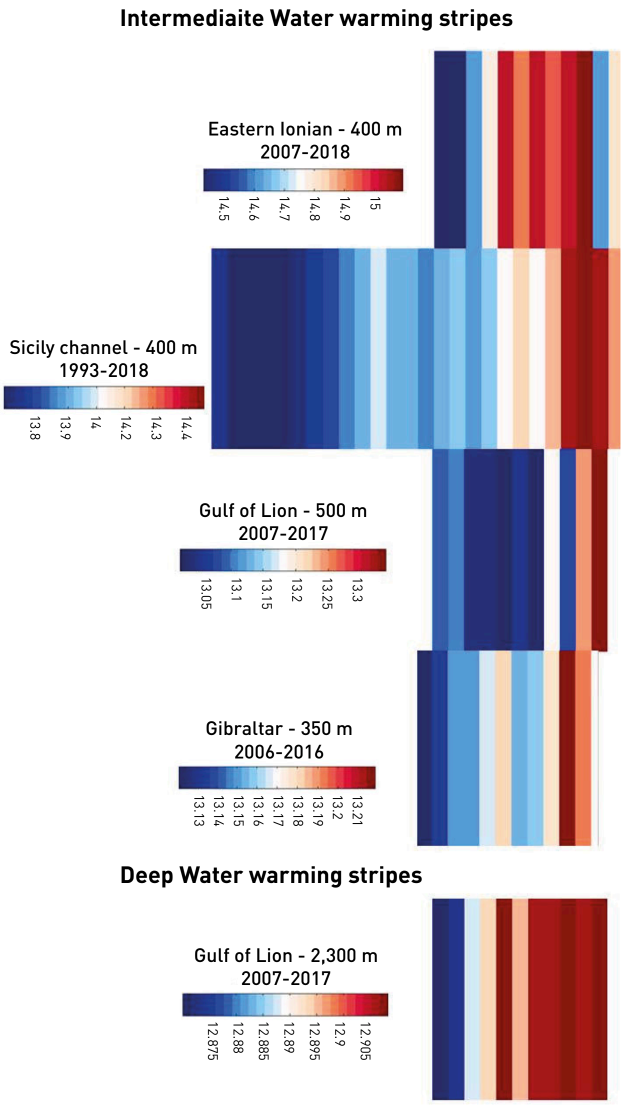


Figure 2.13 | Warming stripes in the Intermediate Water (from east to west) and the Deep Water (in the Gulf of Lion). Each stripe refers to a single year and covered periods differ depending on variable. Values have been computed using yearly potential temperature averages in different locations where long-term mooring data are available at different depths. Data from the eastern Ionian come from the HCMR Pylos deep Observatory (Velaoras et al. 2013), and have been downloaded from CMEMS. Data from the Sicily Channel come from the CNR-ISMAR mooring (Schroeder et al. 2017) and can be downloaded from CMEMS. Data from the Gulf of Lion (500 m and 2,300 m) come from the LION Observatory of the MOOSE Network (Houpert et al. 2016) and can be downloaded from SEANOE (Testor et al. 2019). Data from the Gibraltar Strait come from the IEO (Spanish Oceanographic Institute) mooring (von Schuckmann et al. 2018) and can be downloaded from the Copernicus Marine Environment Monitoring Service (CMEMS). Note that colour scales are different for each panel.

variability is reported for each of the dense water formation sites, the northwestern Mediterranean (Marty and Chiavérini 2010; Houpert et al. 2016; Somot et al. 2018; Waldman et al. 2018), the Adriatic Sea (Dunić et al. 2019), and the Aegean Sea (Roether et al. 2007; Beuvier et al. 2010).

2.2.7.2 Future change in marine waters

Air-sea and land-sea exchanges

The future evolution of the Mediterranean Sea physical characteristics is strongly related to the evolution of the air-sea and land-sea exchanges of water and heat. For the Mediterranean Sea, the net surface water loss by the sea is constituted by the combination of the evaporation over the sea, the precipitation over the sea, the river runoff and the Bosphorus Strait net transport. The net surface heat loss by the sea consists of the shortwave radiation, the longwave radiation, the latent heat and the sensible heat fluxes (these variables are assessed in more detail in *Sections 2.2.3, 2.2.4* and *2.2.5*, as well as in *Section 3.1.3.2*).

In addition to the changes in radiation, the future evolution of the Mediterranean Sea physical characteristics is strongly related to the evolution of the other air-sea heat fluxes. Under present-climate conditions, the net heat loss by the Mediterranean Sea surface (namely the sum of shortwave radiation, longwave radiation, latent heat and sensible heat fluxes) is positive meaning that sea is losing heat by its surface over a long period of time. In the future, the net heat loss by the sea surface is expected to decrease (Somot et al. 2006, 2008; Dubois et al. 2012; Gualdi et al. 2013; Adloff et al. 2015; Soto-Navarro et al. 2020), because the increase in shortwave, net longwave and sensible heat fluxes will dominate the increase in latent heat loss (Dubois et al. 2012). With respect to the end of the 20th century and based on coupled regional climate models, the decrease in the net heat loss could reach between -1.8 and -5.5 W m^{-2} by 2050 following the medium-range A1B scenario (Dubois et al. 2012) and between -2.1 and -6.4 W m^{-2} (resp. -1.0 and -3.7 W m^{-2}) at the end of the 21st century following high-range RCP8.5 (resp. medium-range RCP4.5) scenarios (Soto-Navarro et al. 2020). This implies that the atmosphere could even start to warm the Mediterranean Sea from the mid-21st century instead of cooling it in the present-day climate according to some models *(medium confidence)*. Changes in the Mediterranean Sea surface heat budget depend to a great extent on the socio-economic scenario chosen: the higher the greenhouse gas emissions, the greater the response of the budget.

To summarize, an increase in the net surface water loss by the sea is expected in the future due to a decrease in precipitation and in river runoff and an increase in evaporation (*Sections 2.2.5* and *3.1.1*). In addition, a decrease in the net surface heat loss by the sea is expected in the future (*Sections 2.2.3* and *2.2.5*) because the increase in shortwave, net longwave and sensible heat will dominate the increase in latent heat loss. In particular, this means that, from the middle of the 21st century, some models predict that the atmosphere could, in average, warm the Mediterranean Sea instead of cooling it in the present-day climate.

Future changes in the wind strength over the sea will likely remain low even at the end of the 21st century in pessimistic scenarios *(Section 2.2.2)*.

Sea surface temperature

In future climate change scenarios based on both GCMs and RCMs including the Mediterranean Sea representation, a significant warming of the Mediterranean Sea surface temperature is projected *(very high confidence)* (Somot et al. 2006; Planton et al. 2012; Shaltout and Omstedt 2014; Adloff et al. 2015; Mariotti et al. 2015; Alexander et al. 2018; Darmaraki et al. 2019b). The warming rate depends at the first order on both the temporal horizon and the greenhouse gas emission scenario *(very high confidence)* (Adloff et al. 2015; Mariotti et al. 2015; Darmaraki et al. 2019b). The sea warming will generally remain below that of the air over surrounding land *(high confidence)* due to ocean thermal inertia, probably leading to an increase in land-sea temperature contrast (Somot et al. 2008). With respect to the end of the 20th century, the annual-mean and basin-mean sea surface temperature is expected to increase by 0.6-1.3°C before the mid-21st century and by 2.7-3.8°C (resp. 1.1-2.1°C) at the end of the 21st century period under the pessimistic RCP8.5 (resp. medium RCP4.5) scenario (Darmaraki et al. 2019b). The upper values of those warming ranges are possibly underestimated as higher warming are obtained in CMIP5 GCMs (Mariotti et al. 2015; Darmaraki et al. 2019b).

Future warming will be roughly homogeneous in space *(medium confidence)* with the Balearic Sea, the North Ionian Sea, the Northeast Levantine Sea and the Adriatic Sea identified as potential hotspots of maximum warming *(low confidence) (Fig. 2.12a)* (Adloff et al. 2015; Darmaraki et al. 2019b; Soto-Navarro et al. 2020). This hotspot pattern however does not match well with the observed warming pattern *(Fig. 2.12a)*, illustrating that climate change related evolution is likely still hidden by natural variability. Spatially, the future warming of the sea surface is bimodal as it has been in the past *(Fig. 2.12b)*. For the near-future (2020-2039) with respect to the end of the 20th century, local annual-mean sea surface temperature change is everywhere positive and can reach locally +1.6°C at maximum whatever the scenarios whereas at the end of the 21st century, the local annual-mean warming spreads from 2 to 5°C for scenario RCP8.5 (resp. from 0.5-3.0°C for RCP4.5).

Warming is not projected to be constant all year round. Stronger warming is expected in summer and weaker warming in winter *(medium confidence)*, resulting in substantial increase in warm extremes and a decrease in cold extremes (Alexander et al. 2018). As an illustration, under RCP8.5, maximum monthly-mean sea surface temperature anomalies could reach +3°C over 2040-2059 and more than +5°C over 2080-2099 (median of the CMIP5 models) averaged over the Mediterranean Sea (Alexander et al. 2018). In addition, from the period 2040-2069, the 30-year mean sea surface temperature will always be warmer than the warmest year during the period 1976-2005 *(medium confidence)*. This will already be the case in about 50% of the years for the 2010-2039 period (Alexander et al. 2018).

Marine heat waves will very likely increase in spatial coverage, become longer, more intense and more severe than today *(medium confidence)*. The intensity of this evolution strongly depends on the temporal horizon and on the socio-economic scenario (Frölicher et al. 2018; Darmaraki et al. 2019b). By 2021–2050, it is expected that marine heat wave frequency increases by a factor 1.5, duration by 2.4–2.7, mean intensity by 1.5 and severity by 5-7 with values largely independent from the socio-economic scenarios (Darmaraki et al. 2019b). By 2100, models project at least one long-lasting marine heat wave occurring every year under RCP8.5 up to 3 months longer, and about 4 times more intense and 42 times more severe than today's events. Their occurrence is expected between June and October, affecting at peak, the entire Mediterranean Basin (Darmaraki et al. 2019b). Under a RCP8.5 scenario, the 2003 marine heatwaves may become a normal event for the period 2021-2050 and a weak event at the end of the 21st century *(medium confidence)* (Darmaraki et al. 2019b).

The warm extreme sea surface temperature changes at the end of the 21st century is likely due to a combination of three factors: a mean sea surface warming, an amplification of the seasonal cycle and an increase in the interannual and day-to-day variability (Alexander et al. 2018; Darmaraki et al. 2019b).

Sea surface salinity

The future evolution of sea surface salinity of the Mediterranean Sea remains largely uncertain as its sign of change (Adloff et al. 2015; Soto-Navarro et al. 2020). Any change will likely be spatially and temporally inhomogeneous *(medium confidence)* due to the primary role of the river and near-Atlantic freshwater inputs (Adloff et al. 2015; Soto-Navarro et al. 2020). For the end of the 21st century, basin-scale surface salinity anomalies range from -0.18 to +0.16 psu (resp. -0.25 to 0.25 psu) for the pessimistic RCP8.5 (resp. RCP4.5) scenario (Soto-Navarro et al. 2020). However, a surface salinity increase in the eastern Mediterranean Basin is more likely than not whereas the western basin may see an increase or a decrease in its surface salinity (Adloff et al. 2015; Soto-Navarro et al. 2020).

Surface circulation and exchanges across straits

Change in sea surface circulation has not been deeply assessed yet in the literature (Adloff et al. 2015; Macías et al. 2018), despite their strong capacity to locally modulate the future sea surface temperature and salinity anomalies. In particular, it is likely that the surface circulation changes affect the local sea surface warming hotspots listed above. Noticeable surface circulation changes have been reported for the end of the 21st century in the Balearic Sea and in the North Ionian Sea independently from the scenario choice (Adloff et al. 2015).

At the strait of Gibraltar, the net heat transport towards the Mediterranean Sea will likely increase due to near-Atlantic warming as well as the net mass transport due to increased sea surface water deficit (Somot et al. 2006; Marcos and Tsimplis 2008; Carillo et al. 2012; Adloff et al. 2015; Soto-Navarro et al. 2020). The future evolution of the net salt transport at the strait is unclear, because it depends on the salinity change in the near-Atlantic Ocean surface layer entering the Mediterranean Sea (Marcos and Tsimplis 2008; Adloff et al. 2015; Soto-Navarro et al. 2020). This means that it is currently unclear if the salt transport from the Atlantic will increase or decrease in the future leading to large uncertainty for the salinity change in the Mediterranean Sea.

Deep water characteristics

Due to the contrasting effects of increase in sea surface temperature and salinity, the future evolution of the sea surface density is uncertain. Generally, scenarios with strong greenhouse gas concentration increase project a decrease in surface density associated to an increase in vertical stratification of the water column. Increase in density (thus a decrease in stratification) is still possible in scenarios with low level of warming (Adloff et al. 2015). Due to its active thermohaline circulation, the surface climate change signal may be propagated efficiently towards the deeper layers of the Mediterranean Sea (Somot et al. 2006; Carillo et al. 2012) and lead to larger deep warming rates than in other oceans in the world.

The warming and saltening rates of the deep layers is very uncertain as it depends on various factors such as the surface signal, the intensity of the present and future Mediterranean thermohaline circulation (MTHC). This means in particular that the socioeconomic scenario is not the main source of uncertainty in future changes of the deep layers (Adloff et al. 2015). At the end of the 21st century, water masses deeper than 600 m may warm between +0.03 and +1.38°C, and their salinity may increase or decrease with a large uncertainty range, depending on the model (-0.05; +0.51) psu (Adloff et al. 2015; Soto-Navarro et al. 2020).

All published studies agree on a long-term weakening of the open-sea deep convection, the winter deep water formation and the related branch of the thermohaline circulation in the western Mediterranean Sea in high emission scenarios (Thorpe and Bigg 2000; Somot et al. 2006; Adloff et al. 2015; Soto-Navarro et al. 2020). However, natural variability may lead to increase in deep water formation with respect to today's situation during short periods in the future (Macías et al. 2018). For the end of the 21st century and the A2 scenario, decrease of the maximum mixed layer depth reached in the northwestern Mediterranean Sea reach between -17% and -82% depending on the model choice (Somot et al. 2006; Adloff et al. 2015). The picture in the eastern Mediterranean Sea is more contrasted with weakening in some simulations but enhanced convection and thermohaline circulation in others (Somot et al. 2006; Adloff et al. 2015; Soto-Navarro et al. 2020). Some simulations (but not all) project that EMT-like situation may become the new normal situation for the eastern basin (Adloff et al. 2015).

2.2.8 Sea level, storm surges and wave heights

A particular characteristic of the Mediterranean Sea in terms of sea level variability is that it is a semi-enclosed domain linked to the global ocean through the Strait of Gibraltar. This implies that changes in the nearby Atlantic are quickly transferred into the Mediterranean as a basin-wide barotropic signal. At the same time, basin-wide sea level anomalies caused by local forcing (e.g., thermal expansion, evaporation) tend to be transferred to the global ocean in a way that the Mediterranean is in balance with the nearby Atlantic. As a consequence, the Mediterranean basin-wide variations, especially at low frequencies, closely follow the variations in the nearby Atlantic (Calafat et al. 2012; Adloff et al. 2018). The exception to this is the part of the variability related to changes in the atmospheric mechanical forcing (i.e., wind and atmospheric pressure), which can produce Mediterranean-Atlantic differences of few cm even at multidecadal time scales (Menemenlis et al. 2007; Jordà et al. 2012).

The Mediterranean Sea is a microtidal region, with tidal range mostly below 15 cm and relatively low sea levels with 50-year return values below 60 cm over most of the basin (Marcos et al. 2009). These values are small compared to other European Seas. The two exceptions are the North Adriatic and the Gulf of Gabes, where storm surge levels are estimated to be several times per year above 50 cm, with 5-year return values around 90 cm and 70 cm, respectively (Conte and Lionello 2013).
The wave climate in the region is milder than in the Atlantic with smaller mean wave heights (1-1.5 m) and shorter periods (5-6 s) and presents an important spatial variability due to the complex orography and coastline surrounding the basin (Menéndez et al., 2014). Its variability is connected to NAO and the Indian Monsoon index (Lionello and Sanna 2005) and other northern hemisphere teleconnection indices, particularly the East Atlantic Pattern (Lionello and Galati 2008). Annual maxima along the coastlines are largest (above 5 meters) at the northwestern coast of Africa, but high values well above 4 meters occur in several parts (Lionello et al. 2017).

2.2.8.1 Observed change in sea-level

During the 20th century, coastal tide gauges around the Mediterranean have recorded a rise in the mean sea level. Once tide gauge data have been corrected for the vertical land motion, the sea level trend is very consistent among sites being ~1.4 mm yr^{-1} (Wöppelmann and Marcos 2012). This trend is superimposed on interannual and decadal variability that can temporarily mask the sea level rise. The clearest example is the period 1960-1980 during which Mediterranean sea level showed a decreasing trend because a higher than usual atmospheric pressure (Tsimplis et al. 2005). After that period, the atmospheric pressure returned to the typical values and sea level continued to follow the global evolution. For the more recent period, in which sea level has been monitored by satellite altimetry (1993-2018), Mediterranean sea level trend has increased up to 2.8 ±0.1 mm yr^{-1}, consistent with global sea level trend (3.1 ±0.4 mm yr^{-1}) (Cazenave and WCRP Global Sea Level Budget Group 2018). The rise at global scale is mainly the result of a combination of water thermal expansion and land-based ice melting. During the 20th century both factors contributed equally, although during the last decades, glacier melt is dominating (Cazenave and WCRP Global Sea Level Budget Group 2018).

Analyses of tide gauge data have revealed an increase in the magnitude and duration of the extreme sea level events in the region during the last decades, caused by the rise in the relative mean sea level (for the northern Adriatic Sea: Lionello et al. 2012b; Marcos et al. 2015). In general, wave observational records are too short for assessing multidecadal trends, with the exception of the Northern Adriatic Sea, where one among the worldwide longest instrumental time series (1979 to present) shows an increase in the number of storms, but a decrease of the extreme wave heights (Pomaro et al. 2017).

2.2.8.2 Future sea-levels

The modeling of Mediterranean mean sea level future variations is not straightforward. With their coarse spatial resolution, present-day global climate models (GCMs) are not able to reproduce the regional processes in the basin, although, they are better suited to represent the connection to the global ocean (Calafat et al. 2012). Conversely, regional climate models (RCMs) can capture part of the regional variability but are usually not designed to reproduce the connection with the global ocean, and thus missing a key part of the variability (Adloff et al. 2018). Therefore, sea level rise projections solely based on RCMs have missed that component and only should be considered for the regional patterns, which can cause local spatial deviations from the basin average by up to +10 cm (Carillo et al. 2012; Adloff et al. 2015, 2018). In conclusion, accounting for all components shows that the Mediterranean sea level rise will be close to the northeastern Atlantic, where future sea level will be similar (difference lower than 5%) to the global mean sea level because regional differences produced by changes in the circulation and mass redistribution almost compensate each other (Slangen et al. 2017). This leads to estimate that the basin mean sea level will likely be 37-90 cm higher than at the end of the 20th century, with a small probability to be above 110 cm. Main contributions to basin-average sea level changes are coming from terrestrial ice melting and the northeastern Atlantic dynamics (Jordà et al. 2020, *Fig. 2.14*). A different computation based on the sea level projections in the SROCC (Oppenheimer et al. 2019) and accounting for the uncertainty calculation method of the AR5 (Church et al. 2013), confirms that the likely range of the Mediterranean Sea level will be approximately in the range from a function of time and of the emission scenario, reaching a value in the range from -5% and -10% at the end of the 21st century in the RCP8.5 scenario (Lionello et al. 2017). In any case future sea level rise will become the dominant factor and it will lead to an increase frequency and intensity level of coastal floods (Lionello et al. 2017; Vousdoukas et al. 2017).

Regarding future changes in waves, they will be determined by changes in the wind field over the Mediterranean Sea *(Section 2.2.2).* Published studies point towards a generalized reduction of the mean significant wave height field over a large fraction of the Mediterranean Sea, especially in winter (Lionello et al. 2008, 2017; Perez et al. 2015). Similarly, the wave extremes are expected to decrease in number and intensity, although there is no consensus whether very large extreme events, associated with very strong winds, would also decrease (Gaertner et al. 2007; Romera et al. 2017; Romero and Emanuel 2017).

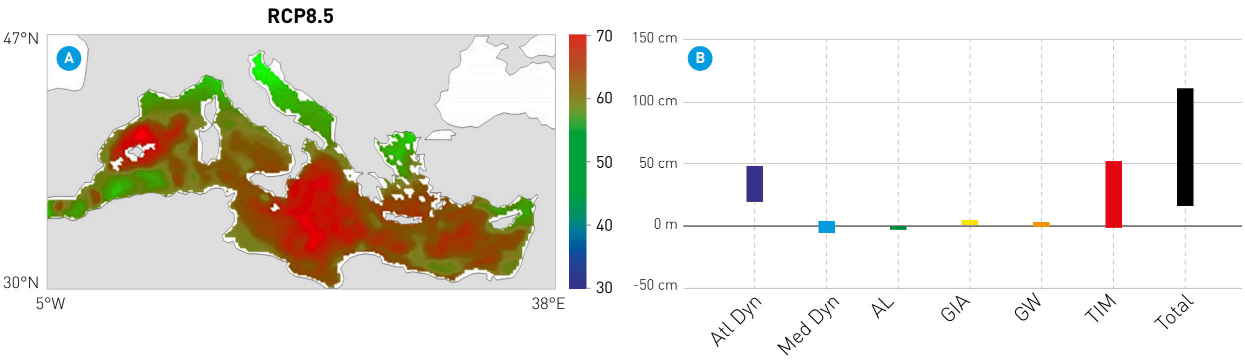


Figure 2.14 | Projected Mediterranean sea level rise averaged in (2080-2099) with respect to present climate (1980-1999) under scenario RCP8.5. Results based on CMIP5 and Med-CORDEX outputs for the dynamical components and Slangen et al. (2017) for other components. (a) Sum of all contributors (b) Range of projected values for the different contributors: NE Atlantic dynamics (Atl Dyn), Mediterranean dynamics (Med Dyn), Atmospheric Loading (AL), Glacial Isostatic Adjustment (GIA), Ground Water (GW) and Terrestrial Ice Melting (TIM) and Total.

20 to 110 cm higher (depending on scenario) at the end of the 21st century than at the end of the 20th century (Le Cozannet et al. 2019; Thiéblemont et al. 2019).

RCMs and GCMs do not model extreme sea level events and specific 2D simulations forced by high frequency atmospheric forcing are needed. Published studies point towards a reduction on the average number of positive surges throughout the 21st century (Marcos et al. 2011; Conte and Lionello 2013; Lionello et al. 2017). Overall, the results indicate small progressive reduction in comparison with their present-day magnitude as

2.2.9 Acidification of the Mediterranean Sea

Human activities are responsible for an increase in atmospheric CO_2 since the beginning of the industrial era. The input of anthropogenic carbon in the Mediterranean is caused by the flux at the air-sea interface, but also results for the Mediterranean Sea, from exchange with the Atlantic Ocean across the Strait of Gibraltar. Approximately 30% of anthropogenic carbon is absorbed by the oceans (Sabine et al. 2004) and leads to decrease of pH in ocean water masses. The Mediterranean Sea is able to absorb relatively more anthropogenic CO_2 per unit area than the

	RECENT CHANGE			PROJECTED CHANGES RCP2.6									PROJECTED CHANGES RCP8.5								
				20-YEAR PERIODS									20-YEAR PERIODS								
	2000-2019			NEAR-FUTURE (2020-2039)			MID-CENTURY (2040-2059)			END OF THE 21ST CENTURY (2080-2099)			NEAR-FUTURE (2020-2039)			MID-CENTURY (2040-2059)			END OF THE 21ST CENTURY (2080-2099)		
CHANGE IN SURFACE TEMPERATURE (°C, MEAN VALUE, LAND-ONLY)																					
	BASED ON CRU			BASED ON EURO-CORDEX 12 KM ENSEMBLE																	
	ANN	DJF	JJA	ANN	DJF	JJA	ANN	DJF	JJA	ANN	DJF	JJA	ANN	DJF	JJA	ANN	DJF	JJA	ANN	DJF	JJA
MED	0.8	0.41	1	1.1	1.1	1.2	1.3	1.2	1.5	1.3	1.4	1.3	1.3	1.3	1.5	2.2	2	2.5	4.6	4.2	5.3
WEST MAGHREB	0.5	0.1	0.8	1.3	1.1	1.3	1.4	1.1	1.6	1.3	1.3	1.4	1.4	1.2	1.6	2.4	2.1	2.7	5	4.2	4.6
EAST MAGHREB	0.6	0.3	0.6	1.2	1	1.2	1.4	1.1	1.6	1.4	1.3	1.4	1.4	1.2	1.6	2.3	2	2.6	4.6	4	5.3
LEVANT	0.9	0.7	1	1.2	1	1.2	1.3	1.3	1.4	1.3	1.3	1.3	1.3	1.3	1.4	2.1	1.9	2.4	4.3	3.9	4.8
ANATOLIA	0.9	0.4	1.2	1.2	1.2	1.3	1.4	1.4	1.6	1.5	1.6	1.5	1.4	1.4	1.6	2.3	2.1	2.6	4.6	4.3	5.3
CENTRAL MED	0.8	0.3	1	1.1	1	1.3	1.3	1.2	1.4	1.2	1.4	1.2	1.3	1.3	1.5	2.1	1.9	2.5	4.3	3.9	5.1
IBERIA	0.5	0.2	0.8	1	0.9	1.2	1.1	0.9	1.3	1	1	1.1	1.1	1	1.3	2	1.7	2.4	4.3	3.6	5.3
FRANCE	0.7	0.4	0.7	0.9	0.9	1	1.1	0.9	1.1	0.9	1.1	0.8	1	1	1.1	1.8	1.7	2	3.9	3.6	4.7
ALPS	0.8	0.5	1	1.1	1.1	1.1	1.3	1.2	1.3	1.1	1.4	1	1.2	1.3	1.3	2.1	2.1	2.2	4.5	4.5	5
BALKANS	1.1	0.8	1.4	1.1	1.1	1.1	1.3	1.4	1.3	1.2	1.7	1	1.3	1.4	1.4	2.1	2.3	2.2	4.4	4.6	4.8
CHANGE IN PRECIPITATION (%, MEAN VALUE, LAND-ONLY)																					
	BASED ON CRU			BASED ON EURO-CORDEX 12 KM ENSEMBLE																	
	ANN	WET	DRY	ANN	WET	DRY	ANN	WET	DRY	ANN	WET	DRY	ANN	WET	DRY	ANN	WET	DRY	ANN	WET	DRY
MED	1.3	2.5	-0.4	0.8	2.3	-1	1	2.3	-0.6*	3.9	6	1.1*	0	1.5	-2.1*	-1.9	0	-4.5	-6.8	-2	-13.3
WEST MAGHREB	2.7	1.3	6.2	-6.7	-7.8	-5.2	-6.2	-5.7	-6.9	-1.7*	0.8	-4.9	-11.2	-9.4	-13.7	-16.8	-20.3	-11.8	-31.2	-33	-28.7
EAST MAGHREB	-7.6	-11.1	5.9	-10.1	-11	-8.7	-5.4	-3.4*	-8.4	-7.8	-8.2	-7.2	-9.7	-8.6	-11.4*	-11.8	-11.7	-11.9	-23.6	-24.1	-22.7
LEVANT	-1.6	-3.5	12.2	-5.8	-5.1	-8.4	-5.8	-4.7	-10.1	-5.8	-4.3	-12	-4.6	-5.1	-2.4*	-10.7	-10	-13.5	-23.5	-23.6	-23.1
ANATOLIA	5.7	3.7	10	0.5*	3.1*	-4.1	-0.1*	2*	-3.9*	2.6	5.3	-2.4	0.2*	2.2	-4.1	-2.4	-0.2	-6.5	-8.7	-4.4	-16.4
CENTRAL MED	2.8	1.3	6.3	0.1*	1.8	-2.4	0.8*	2.1*	-1.2*	3.8	6.3	-0.8	-0.5*	1.5*	-4*	-3.3	-0.7	-7.6	-9.7	-4.7	-18.1
IBERIA	4.9	10	-4.1	-0.1	2.3*	-3.6	0.3*	2.6	-3.2*	5	10.1	-2.8*	-1.5*	0.5*	-4.9*	-6	-2.2	-12.4	-15.1	-7.5	-27.7
FRANCE	-1.8	0.7	-4.4	2.5	4.3	0.3*	2.4	4.2	0.2*	4.1	5.2	2.8	1.6	3.1	-0.3*	0.1	4.4	-5.2	-2.5*	6.7	-13.9
ALPS	-0.5	2.2	-2.8	3.5	6.3	0.7*	4.6	6	3.2	6.2	7.1	5.3	2.9	5	0.9*	3.1	6.6	-0.4*	1.5	9.2	-6.3*
BALKANS	2.5	6.7	-0.4	5.3	8.8	2.4	4.5	6	3.3	8.3	11.8	5.4	3.9	6.6	1.5	4.7	8.1	1.6*	4.9	14.4	-3.7*
CHANGE IN SURFACE SOLAR RADIATION (W/M², 90% INTERVAL BASED ON CMIP5 SIMULATIONS, LAND+SEA)																					
	ANN			ANN			ANN			ANN			ANN			ANN			ANN		
MED	0.3 ; 5.2			0.8 ; 8.6			0.8 ; 10.7			0.5 ; 12.6			0.6 ; 7.7			1.0 ; 10.5			0.6 ; 12.4		
CHANGE IN CLOUD COVER (%, 90% INTERVAL BASED ON CMIP5 SIMULATIONS, LAND+SEA)																					
MED	-2.1 ; 0.2			-3.6 ; 0.2			-4.4 ; 0.5			-4.3 ; 0.5			-3.3 ; 0.0			-4.9 ; -0.2			-7.4 ; -0.8		
CHANGE IN AOD (-, 90% INTERVAL BASED ON CMIP5 SIMULATIONS, LAND+SEA)																					
MED	-0.06 ; -0.02			-0.12 ; -0.04			-0.15 ; -0.04			-0.18 ; -0.05			-0.10 ; -0.03			-0.12 ; -0.04			-0.14 ; -0.04		
CHANGE IN SURFACE TEMPERATURE (°C)																					
	CMEMS (2000-2017)			TOTAL RANGE BASED ON MED-CORDEX RCSM																	
MEDSEA	0.74 (+0.23 ; +0.87)			X			X			X			0.59 ; 1.18			1.30 ; 2.07			2.86 ; 4.10		
CHANGE IN SEA LEVEL (CM, BASED ON BLENDED MULTIPLE DATABASE, SEE THE TEXT, CM, VERY LIKELY RANGE)																					
MEDSEA										(+42, +82)									(+70, +110)		

Table 2.1 | Climate change as a function of time period and Representative Concentration Pathway for the land sub-regions in *Fig. 2.1* and the whole Mediterranean Sea area. All changes are with respect to the (1980-1999) reference period. For temperature and precipitation, recent changes are based on the Climate Research Unit CRU-TS, future changes are based on the EURO-CORDEX regional model simulations. When values are annotated with "*" less than 2/3 of the models agree on the sign of projected changes. For surface solar radiation, cloud cover and aerosol optical depth values are based on CMIP5 global simulations. For sea surface temperature, recent changes are based on CMEMS observations (2000-2017), future changes on the Med-CORDEX regional simulations. For sea level rise, future changes are based on blended multiple databases *(see text)*.

global ocean for two reasons. It is more alkaline, thus giving it greater chemical capacity to take up anthropogenic CO_2, and deep waters are ventilated on shorter timescales (Schneider et al. 2010), thus allowing rapid penetration of CO_2 in its interior.

2.2.9.1 Observed change in acidity

Concerning the past trends of anthropogenic carbon absorption by the Mediterranean Sea, the presence of natural CO_2 prevents to determine it from direct measurements in the water column. Estimations of anthropogenic CO_2 from data-based approaches are limited and with large uncertainties that provides concentrations that disagree by more than a factor of two in the Mediterranean Sea (Schneider et al. 2010; Touratier and Goyet 2011). These large differences further result in even opposing estimates for the net CO_2 transport across the Strait of Gibraltar.

In this context, the modeling approach using high-resolution regional model provided some insights on the information resulting from the data based-estimates and quantification of processes responsible of anthropogenic CO_2 storage and acidification of the Mediterranean Sea (Palmiéri et al. 2015). 25% of total anthropogenic carbon inventory in the Mediterranean Sea is due to net exchange at the Strait of Gibraltar, while the remaining 75% is from the air-sea flux. It confirms that the 10% higher mean total alkalinity of the Mediterranean Sea is responsible for a 10% increase in anthropogenic carbon inventory. The higher alkalinity acts to neutralize acidification and simulated average surface pH change is thus similar for the Mediterranean Sea and the global ocean (-0.08 units), with deep waters exhibiting a larger anthropogenic change in pH than typical global ocean deep waters because ventilation times are faster (between -0.005 and -0.06 units) (Palmiéri et al. 2015).

2.2.9.2 Future change in acidity

The published literature concerning specifically the future acidification of the Mediterranean Sea is quite limited leading to low confidence in the assessment. Based on thermodynamic equations of the CO_2/carbonate system chemical equilibrium in seawater, Goyet et al. (2016) calculated the variation of pH (ΔpH) as a function of theoretical anthropogenic CO_2 concentrations. Under the most optimistic SRES scenario, the results indicate that in 2100, pH may decrease down to 0.245 in the western basin and down to 0.242 in the eastern basin (compared to the pre-industrial pH). Whereas for the most pessimistic SRES results for 2100 project a pH decrease down to 0.462 and 0.457, for the western and for the eastern basins, respectively (Goyet et al. 2016). However, these estimates do not consider that the warming of seawater will decrease exchanges across ocean-atmosphere interface and penetration of anthropogenic Jungcarbon, thus they tend to overestimate future acidification.

2.3 Pollution

2.3.1 Introduction

The 2030 Agenda for Sustainable Development pledges to "ensure that all human beings can enjoy prosperous and fulfilling lives and that economic, social and technological progress occurs in harmony with nature" (UN 2015). Pollution puts at risk the possibility of achieving these outcomes and hence health and well-being. Pollution touches all parts of the planet. It is affecting our health through the food we eat, the water we drink and the air we breathe. Approximately 19 million premature deaths are estimated to occur annually as a result of the way we use natural resources and impact the environment to support global production and consumption. By definition, "pollutant" shall mean any substance that is introduced into the environment that has undesired effects, or adversely affects the usefulness of a natural resource (air, soil, water and ecosystems). Pollutants can take many forms: (i) physical, substances that are not necessarily involved in chemical or biological reactions, e.g., dust; (ii) chemical, substances that are involved in chemical reactions, e.g., pesticides; and (iii) biological, e.g., bacteria.

2.3.2 Physical pollutants

Particulate matter (PM) levels have been monitored during the past decades, mainly because of their effects on health and climate. Aerosols into the atmosphere arise from a variety of anthropogenic activities (transport, industry, biomass burning, etc.) as well as natural sources (volcanic eruptions, sea salt, soil dust suspension, natural forest fires, etc.) (Seinfeld and Pandis 2006). Both sources result in direct emission of PM (primary PM) and emission of gaseous aerosol precursors (leading to secondary PM). A number of epidemiological studies have examined the impact of PM on human health, expressed as increased mortality and morbidity varying according to the physical (size, shape, etc.) and chemical (composition) characteristics of PM (Van Dingenen et al. 2004).

The PM impact on climate is primarily a cooling effect due to increased scattering to space as the atmospheric aerosol burden increases. The overall cooling by aerosols might be equivalent to a radiative forcing of up to 2.5 W m^{-2}, counterbalancing global warming by greenhouse gases (Gillett et al. 2013; Knutson et al. 2013). More important than this direct effect may be the indirect effect that aerosols have on climate, acting as cloud condensation nuclei (CCN) (Levin et al. 2003; Gerasopoulos et al. 2006). Moreover, the particles have a large effect in reducing visibility as well as play a significant role in the deterioration of monuments and buildings (Gerasopoulos et al. 2006). Several studies conducted over the Mediterranean Basin revealed a distinct spatial inhomogeneity (Gerasopoulos et al. 2006), with PM levels increasing from north to south and west to east of the basin (Querol et al. 2009), and distinct sources. PM analysis through the years allowed the identification and classification of PM episodes as follows: (i) local urban PM pollution events (mostly in the cold season), (ii) regional PM pollution episodes (warm season) and (iii) dust outbreaks (Rodríguez et al. 2003).

2.3.2.1 Particulate matter (PM) levels and sources

Several factors favor the occurrence of high PM concentrations in the Mediterranean Basin. First, the abrupt topography, coupled with the characteristic synoptic scale patterns, results in low mean wind speeds that hinder the air mass renovations and favor the accumulation of PM in the surrounds of emission regions – leading to the so-called Atlantic/Northern clean air advections events (Rodríguez et al. 2007). Second, the low precipitation in the Mediterranean Basin favors the long residence time of PM in the atmosphere, leading to higher background PM levels (Rodríguez et al. 2007; Querol et al. 2009). The joint influence of low precipitation rates and traffic-forced resuspension of road (which is strongly enhanced by the dust accumulation in streets and roads), construction and demolition dust promoted the local urban episodes in the Mediterranean Basin (Rodríguez et al. 2007; Talbi et al. 2018). This factor, combined with the high percentage of water coverage of the area, especially in the East region of the basin, has a great contribution of the sea-salt aerosols to the PM levels and composition (Im 2013).

PM observations from monitoring networks, in the period 2007-2009, were analyzed in order to characterize particulate pollution and its health effects across Mediterranean countries (Karanasiou et al. 2014). It was concluded that the average concentrations for PM across the Mediterranean Basin are within the range of annual means typical of European sites and according to the monitoring site characteristics (traffic and urban background sites) (Querol et al. 2004; Putaud et al. 2010). The regional patterns mentioned in previous studies were highlighted, with higher PM concentrations in Italian and Greek cities, and lower levels in the Western Mediterranean (Barcelona, Marseille, Madrid, Huelva). PM10 levels at the traffic sites showed a quite similar variation. In Turin, as in the other cities of the Po valley (Bologna, Milan, Parma, Modena and Reggio Emilia), the combination of stagnant air conditions with high emissions and high population density is the main cause of very strong pollution episodes (Cyrys et al. 2012). Similarly, the air pollution problems in Athens and Thessaloniki are the result of the high population density and the accumulation of air pollutants over the city, due to topography (basin surrounded by mountains), narrow and deep street canyons and adverse meteorological conditions (Karanasiou et al. 2009; Kassomenos et al. 2011). Thermal inversions, followed by accumulation of air pollutants in the lower layers of the atmosphere are also very common in different locations like Athens (Karanasiou et al. 2014) or Beirut (Saliba et al. 2006), increasing the evening concentrations of ambient PM10.

In most countries of the southern Mediterranean, air pollution is not sufficiently monitored (Naidja et al. 2018). Emission inventories are less precise than that available in the northern Mediterranean since they are generally based on surveys and questionnaires. Because of that, local scientific articles were relatively scarce and hard to find. However, most of the available studies show that PM concentrations in this Mediterranean region are much higher than the limit values given in WHO guidelines (Naidja et al. 2018). Emissions from road traffic, resuspension of road dust, especially on unpaved roads, and natural contributions have been found to be an important source of fine particles and play a key role on the observed levels and exceedances (Mahmoud et al. 2008; Abderrahim et al. 2016; Naidja et al. 2018). Cairo (Egypt) is an example of a city where road traffic emissions are hugely important in PM10 concentration, and, ac-cording to WHO is ranked in the 33rd position on the list of the most polluted cities by PM10 (Mahmoud et al. 2008; Lowenthal et al. 2014; Naidja et al. 2018).

Since it is expected that the majority of the Mediterranean population will continue to live in cities, especially in the eastern and southern part of the basin, with a tendency to growth, higher anthropogenic pressure in a context of climate change will occur (Rafael et al. 2015; Naidja et al. 2018). Most PM exceedances were registered in regional background sites (Escudero et al. 2007), with more than 70% of them being attributed to dust outbreaks (Escudero et al. 2007; Mitsakou et al. 2008). Compared with the central and northern Europe, the occurrence of higher PM concentrations associated with dust outbreaks is higher in the Mediterranean Basin (Rodríguez et al. 2007), and are more frequent and more intense in the central and eastern than those in the western Mediterranean Basin. These episodes have been studied on an 11-yr period (2001–2011) (Pey et al. 2013). Dust outbreaks are very frequent in the southern Mediterranean, where they occur more than 30% of the days, while in northern Mediterranean this value is below 20%. The central Mediterranean appears as a transitional area, with a decreasing south to north gradient of dust outbreaks, with slightly higher frequency of dust episodes in its south, when compared to west and east sides of the basin, for similar latitudinal positions (Pey et al. 2013).

Regarding intensity characteristics and seasonality patterns, significantly high contributions are common in autumn-spring in the eastern Mediterranean, with occurrence of many severe episodes (daily dust averages over 100 µg m^{-3} in PM10) throughout the year. However, in the western Mediterranean a clear summer prevalence is noticed, with low occurrence of severe episodes; and no seasonal trend is detected in the central region, with moderate-intensity episodes (Pey et al. 2013). The contribution of dust outbreaks to PM concentrations reveals a downward trend in the period between 2006 and 2011, a period in which there was also a decrease of the NAO index for the summer period. Therefore, it can be concluded that a sharp change in the atmospheric circulation have affected the number of dust episodes and, consequently, the annual dust inflows to PM10 *(Fig. 2.15)* observed in the Mediterranean Basin (Pey et al. 2013).

The low PM2.5/PM10 ratio (approximately 0.25) in the eastern Mediterranean region also indicates that the particle size distribution has a large contribution of coarse particles which are either affected by a background level of naturally occurring dust (dust outbreaks from the Saharan Desert and sea salt particles from the Mediterranean Sea itself) or that the region is characterized by high levels of primary coarse PM emissions (Koçak et al. 2007b, 2007a). Even though the PM2.5/PM10 ratio showed seasonal variations, the values remained lower than 0.5 in most cases (Koçak et al. 2007a; Asaf et al. 2008), a value that is least two times lower than those of the western Mediterranean (Saliba and Massoud 2010).

2.3.2.2 Particulate matter (PM) chemical profiles

Regarding the chemical composition of PM, different species can be found such as carbonaceous compounds, inorganic ions and metals (Galindo et al. 2018). Although they are present at extremely low levels, some components such as trace metals are relevant in air quality studies because of

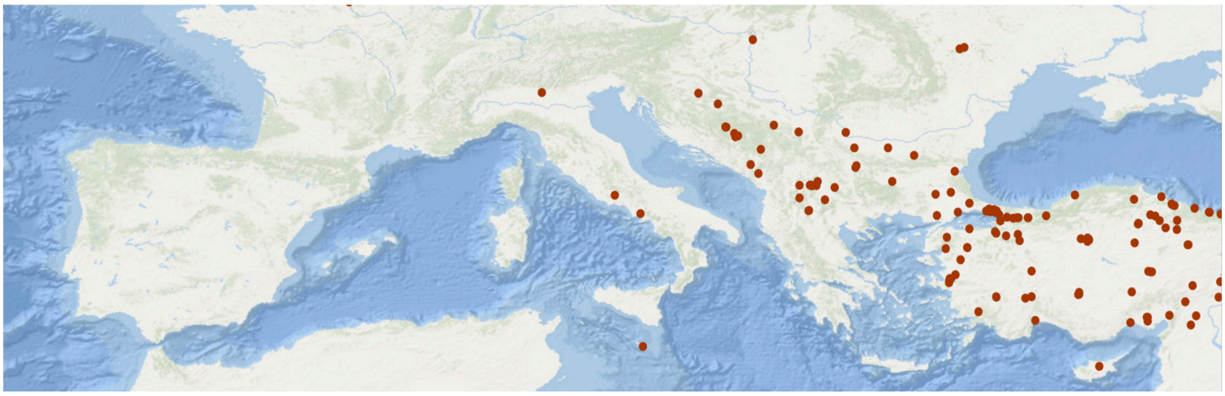


Figure 2.15 | PM10 concentration above the annual limit value of 40 µg·m^{-3} (based on EU Directive 2008/50/CE).

their toxicity and environmental persistence (Roig et al. 2013). Recent clinical and toxicological studies demonstrate the link between exposure to airborne metal through inhalation and pulmonary and cardiovascular effects, genotoxic and carcinogenic outcomes and increased daily mortality (Gottipolu et al. 2008; Lippmann and Chen 2009; Tchounwou et al. 2012).

Cooling metal concentrations are considered as good tracers of specific pollution sources, both natural and anthropogenic (Arhami et al. 2017; Diapouli et al. 2017). The main natural sources include wind-blown dust and sea-spray (Chen et al. 2008; Engelbrecht and Jayanty 2013) including elements such as calcium, aluminum, iron, potassium, sodium and magnesium. Desert dust contribute to PM composition and have a high influence on climate in the North as well as in the South of the Mediterranean (Kchih et al. 2015; Kaskaoutis et al. 2019). Specific meteorological circulations and natural sources like the Mediterranean Sea and the proximity of Sahara create specific patterns of aerosol concentrations that could influence not only the particulate concentrations through Europe but also the global climate due to the transport of dust from the Sahara (Ganor et al. 2010).

Regarding anthropogenic activities, exhaust and non-exhaust vehicle emissions, coal combustion and a variety of industrial processes, like metal works and smelters, are the major sources of heavy metals such as zinc, copper, nickel or chromium (Thorpe and Harrison 2008; Pant and Harrison 2013). In the last decades, emissions of some heavy metals in Mediterranean Basin have dropped significantly, in particular from industrial facilities due to improvement of abatement techniques (Dayan et al. 2017). In the case of lead, a drastic reduction in ambient concentrations has been observed since the introduction of unleaded gasoline (Cho et al. 2011; Salvador et al. 2012). The influence of traffic and dust outbreak intrusions on PM levels and metal content have been studied (Galindo et al. 2018), showing that the PM coarse fraction was affected more by variations in traffic intensity than the submicron fraction: the highest decreases during the weekends due to the reduction in traffic induced resuspension. That dust outbreaks had a greater impact on the levels of other metals such as titanium and lead, significantly affecting their seasonal variability. High concentrations of vanadium and nickel compared with the values found at larger urban areas were observed. This could be attributed to a significant contribution from soils, dust outbreaks (Galindo et al. 2018) and even ship emissions (Monteiro et al. 2018a; Russo et al. 2018).

Another issue related to PM composition is its radionuclide content. Radionuclides in the atmosphere rapidly attach on submicron-sized aerosols, and their variability in ground-level air is driven by the behavior of aerosols (Povinec et al. 2012; Hirose and Povinec 2015). Atmospheric radionuclides are deposited from the air onto the land and sea surface by wet and dry deposition. In this way, the terrestrial and marine environments are labeled by natural and anthropogenic radionuclides that can be used as tracers of environmental processes (Pham et al. 2017). Radionuclide content can pose a health hazard following an accident involving nuclear material (Baeza et al. 2016). However, the occurrence of anthropogenic radionuclides in aerosols is also due to erosion and resuspension processes, as well as the emission and transport of particulate matter due to biomass burning as consequence of wild fires (Strode et al. 2012; Evangeliou et al. 2014), and dust transport due to dust outbreaks (Hernández et al. 2005). Due to these processes, the anthropogenic radionuclide concentration in near surface atmosphere is variable. Naturally occurring radionuclides are also present in airborne particles as they are also present in soil particles able to be eroded, re-suspended or transported by the processes previously described, and also due to the radon exhalation from soil, which is especially significant to lead-210 and polonium-210 (Baeza et al. 2016).

2.3.2.3 Plastics (macro/micro/nano)

We live in the plastic age, since synthetic polymers are present in most aspects of human life both in developing and industrialized countries. The worldwide production for plastics increased annually by 10% since the 1950s, reaching 300 Mt in 2015 (Geyer et al. 2017). As of 2015 approx. 6,300 Mt of plastic waste had been generated, around 9% of which had been recycled, 12% was incinerated, and 79% was accumulated in landfills or the natural environment (Geyer et al. 2017). Synthetic thermoplastics constitute the most abundant and still growing component of anthropogenic debris entering the Earth's oceans (Ivar do Sul and Costa 2014). Up to 80%, or sometimes more, of the waste that accumulates on land, shorelines, the ocean surface or seabed is plastic (Barnes et al. 2009). The smallest form of plastic litter is called micro-plastic (<5 mm) and can represent up to 335,000 items km^{-2} or 5 kg km^{-2} in marine waters, and up to 25 kg km^{-2} in coastal sediments (Koutsodendris et al. 2008; Ryan et al. 2009). Plastic debris, their dissolved derivatives, as well as, the adsorbed organic pollutants (Hirai et al. 2011) pose a direct risk to human and marine ecosystem health (Galloway 2015; Koelmans et al. 2017). As a rule, widely used plastics do not rapidly degrade naturally when released into the environment, it can take 50 or more years for plastic to fully decompose (Müller et al. 2001).

In the Mediterranean Sea, the average density of plastic (1 item per 4 m²), as well as its frequency of occurrence (100% of the sites sampled), are comparable to the accumulation zones described for the five subtropical ocean gyres (Cincinelli et al. 2019), increasing the impact for marine biota with hotspots for the risk of plastic ingestion across multiple taxa especially in the coastal zone (Compa et al. 2019). Plastic debris in the Mediterranean surface waters *(Fig. 2.16)* was dominated by millimeter-sized fragments, but showed a higher proportion of large plastic objects than that present in oceanic gyres, reflecting the closer connection with pollution sources (Cózar et al. 2015). Multi-annual simulations of advected surface passive debris depict the Tyrrhenian Sea, the northwestern Mediterranean sub-basin and the Gulf of Sirte as possible retention areas (Mansui et al. 2015). No permanent structure able to retain floating items in the long-term were found, as the basin circulation variability brings sufficient anomalies to alter the distribution (Mansui et al. 2015).

Beyond the concern with "traditional" PM effects, an emergent research issue worldwide has been focused in the occurrence of microplastics in the atmospheric compartment (MP; plastic particles with a longest dimension < 5 mm). MP may undergo photo-oxidative degradation in the environment, along with wind shear and/or abrasion against other ambient particulates, eventually fragmenting into fine particles (Gasperi et al. 2018). The risk of inhaling fibrous MP following widespread contamination within different environmental compartments deserves special attention owing to both the scale of their worldwide production and their potential to fragment into smaller, more bioavailable fibers. Human exposure to MP could also occur through ingestion, for example fibrous MP can settle on the floor; children, owing to crawling and frequent hand-to-mouth contact, ingest daily settled dust (Gasperi et al. 2018). Two studies have demonstrated the presence of MP in the atmosphere (Dris et al. 2016, 2017), thereby suggesting potential human exposure (none of these studies has been conducted in the Mediterranean Basin).

2.3.3 Chemical pollutants

2.3.3.1 Nutrients

Nutrients (mainly nitrogen, N, and phosphorous, P) constitute an important factor controlling marine

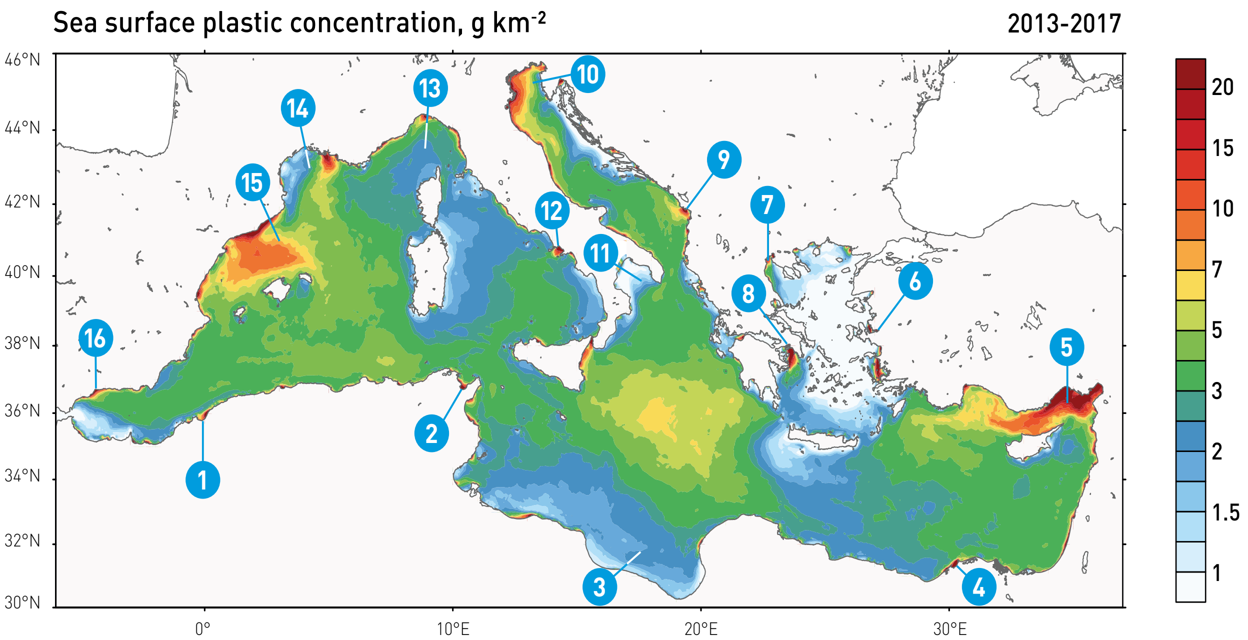


Figure 2.16 | Averaged 2013–2017 map of plastic debris concentration (g km^{-2}) at the sea surface. Some geographical names used in the text are given: the (1) Gulf of Arzew, (2) Gulf of Tunis, (3) Gulf of Sidra, (4) Abu Qir Bay, (5) Cilician Sea, (6) Izmir, the (7) Thermaic Gulf, (8) Saronic Gulf, (9) Buna-Bojana, (10) NW Adriatic, (11) Taranto Gulf, (12) Gulf of Naples, (13) Gulf of Genoa, (14) Gulf of Lion, (15) Catalan Sea, and (16) Malaga Bay (Liubartseva et al. 2018).

primary producers, as they control phytoplankton growth, biomass and species composition (Sakka Hlaili et al. 2006). According to the nutrient concentrations, marine waters are characterized as oligotrophic (low nutrient concentrations), mesotrophic (nutrient enriched water), or eutrophic (nutrient rich water). The Mediterranean Sea is characterized by oligotrophic off-shore waters, with decreasing levels of nutrients eastwards from Gibraltar to the Levantine Sea (Ignatiades et al. 2009; Tanhua et al. 2013). The Eastern Mediterranean Sea is the most oligotrophic region, with very low nitrate concentrations (< 0.5 μm) and phosphorous (< 0.2 μM) (Pujo-Pay et al. 2011). Pronounced phosphorous limitation, with N/P ratio > 30, is observed for the south of Levantine Sea and Ionian Sea (Kress et al. 2003; Pujo-Pay et al. 2011).

However, Mediterranean coastal areas, which are highly populated, are experiencing increasing N and P loading from anthropogenic activities, such as urban effluents, industrial discharges, agricultural runoffs, aquaculture activities and riverine inputs from a drainage area of 1.5×10^6 km^2 (UNEP/MAP 2017). The overall inputs of N and P in these areas are about 1.5-4.5 and 0.1-0.4 Mt yr^{-1}, respectively. The main sources of N in Mediterranean are urban wastewater treatment (45%) and livestock farming (24%). Fertilizer use can also bring nitrogen and inputs can exceed 10^6 kg yr^{-1} *(Fig. 2.17)*. Aquaculture contributed also to the emission on N (10%). For P, the main emitters are manufacture of fertilizers (40%), farming of animals (39%) and urban wastewater treatment (13%) (UNEP/MAP 2012b).

Some coastal regions are known as hotspots of nutrient inputs. In the North of Mediterranean Sea, the Lagoon of Venice and the Gulf of Lion sustained high nitrate levels, 18 μM and 9 μM, respectively (Acri et al. 2004; Severin et al. 2014). Nitrate rich waters characterize also the Eastern Adriatic Sea (4 μM) (Skejic et al. 2017) and the Western Tyrrhenian Sea (6.5 μM) (Astraldi et al. 2002). In the southern Mediterranean, The Gulf of Gabès is a main region known for P enrichment (1–11.2 μM,), since Tunisia is an important producer country of P fertilizers. Nitrate (6-6.5 μM) and ammonia (~4 μM) showed also pronounced levels in this Gulf (Drira et al. 2016). High nitrate concentrations were often measured in other Tunisian coastal systems, such as the Lagoon of Bizerte (NO_3^-: 1-6.3, NH_4^+: 20–30 μM) (Sahraoui et al. 2012) and the North Lake of Tunis (NO_3^-: 7.5–198 μM) (Armi et al. 2011). In the Algerian-Provençal Basin and the Gibraltar Strait, enrichment of water with nitrate has been reported (9.5-10 μM) (Béthoux et al. 1998; Gómez et al. 2000).

Nutrient enrichment of Mediterranean Sea may result in a high increase in phytoplankton growth and biomass, leading to the eutrophication. The impacts of eutrophication include hypoxia or anoxia and may provoke harmful algal blooms, some of them toxic. Harmful algal blooms (HABs) cause human illness and mortality and have socio-economic impacts related to toxicity of harvested fish and shellfish, loss of aesthetic value of coastal ecosystems, and reduced water quality impacting tourism *(Section 2.3.4)*.

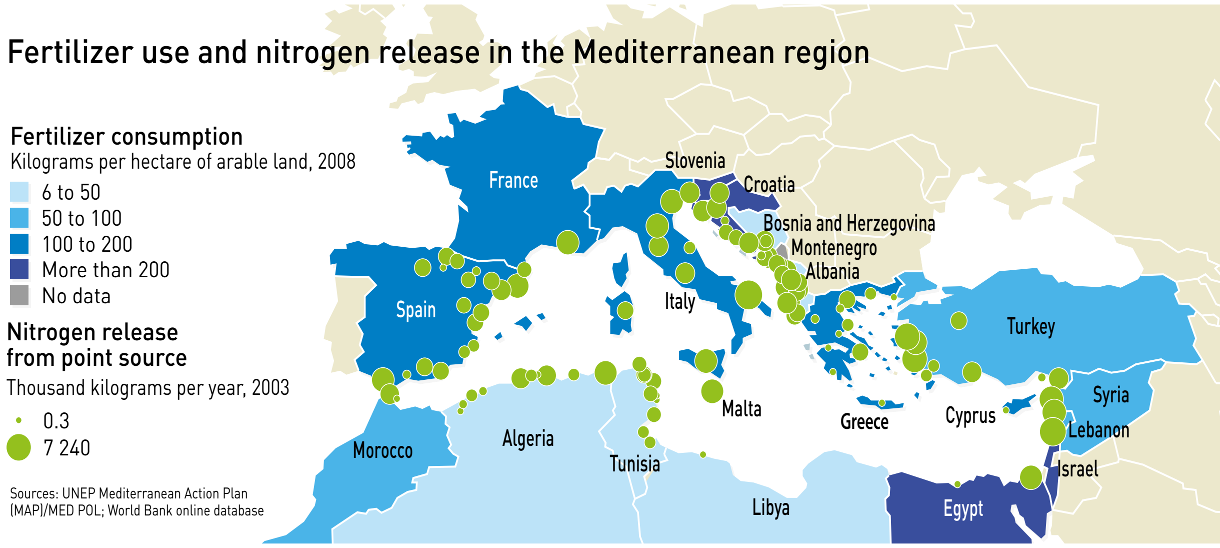


Figure 2.17 | Fertilizer use and nitrogen release in Mediterranean Sea (UNEP-GRID Arendal 2013).

2.3.3.2 Gaseous pollutants: nitrogen dioxide, sulphur dioxide, ozone

The Mediterranean Basin is one of the regions in the world where high concentrations of gaseous air pollutants (nitrogen dioxide – NO_2, sulphur dioxide – SO_2, and ozone – O_3) have been reported frequently (Dayan et al. 2017). The elevated concentrations observed are attributed to the combination of diverse emission sources affecting the Mediterranean Basin including industry, traffic and shipping emissions (Karanasiou et al. 2014). The Mediterranean climate is characterized by arid conditions as well as many hours of sunshine and specific atmospheric recirculation patterns that significantly enhance air pollution levels (Karanasiou et al. 2014; Querol et al. 2018).

Commonly, NO_2 concentrations in North Mediterranean countries are higher than those observed in northern Europe (Cyrys et al. 2012). This fact was attributed to the transport sector, and to the higher conversion of nitric oxide (NO) to NO_2 caused by high temperatures and O_3 concentrations (Schembari et al. 2012; Karanasiou et al. 2014). The spatial pattern of N deposition varies across the Mediterranean Basin *(Fig. 2.18)*. In Iberia, dry deposition is an important component of the total atmospheric N input to natural habitats (García-Gómez et al. 2018; Oliveira et al. 2020).

Ships are among the major emitters of air pollutants such as SO_2 and NO_x, their contribution to the emissions from the transport sector (Schembari et al. 2012) and to the air pollution in the Mediterranean Basin (Monteiro et al. 2018b; Russo et al. 2018) is growing. Several studies have also shown that NO_2 exceedances (yearly and hourly) in cities of the Mediterranean Basin are caused by road traffic emissions (Borrego et al. 2012; Belhout et al. 2018) *(Fig. 2.19)*.

The large variety of Volatile Organic Compounds (VOCs), NO_x emissions and the climate conditions of Mediterranean Basin influences O_3 formation and destruction (Sahu and Saxena 2015; Sahu et al. 2016). These factors result in higher O_3 concentrations and frequent tropospheric O_3 episodes recorded across the Mediterranean Basin, with different frequencies in the East and West (Sicard et al. 2013).

The western Mediterranean Basin is characterized by frequent sea breezes, driven by sea-land thermal contrast. These sea-land breezes play an important role for the O_3 concentrations since they transport air masses, including O_3 precursor gases, from urban agglomerations located in coastal areas, towards inland suburban and rural areas (Millán et al. 2000).

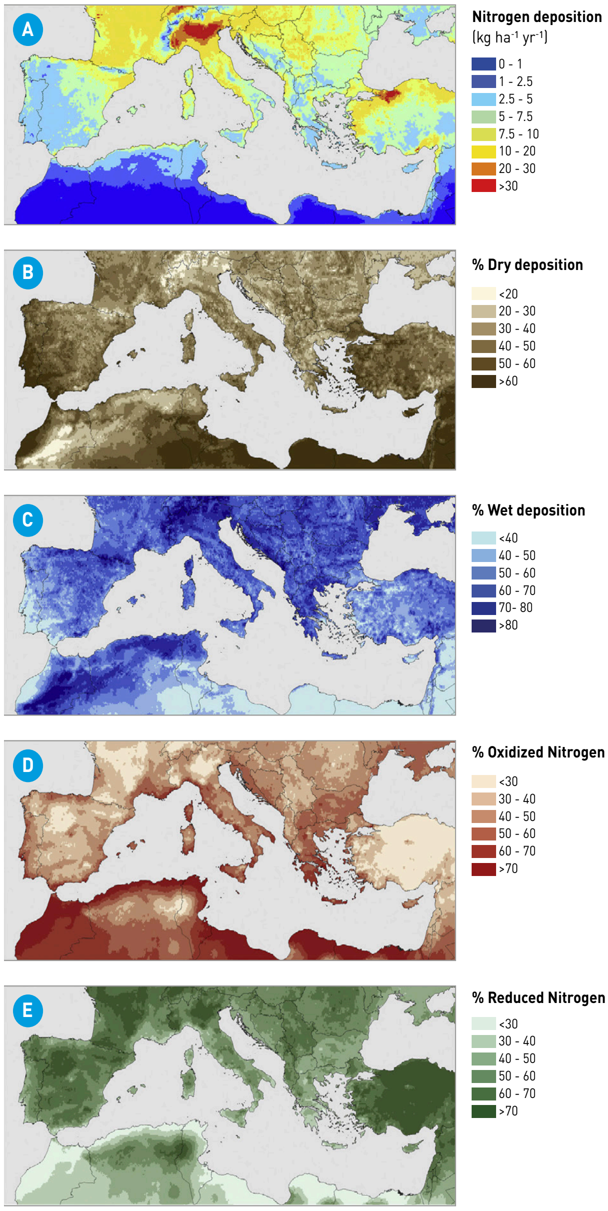


Figure 2.18 | Modelled nitrogen deposition for the Mediterranean region based on the European Monitoring and Evaluation Programme (EMEP) model at 0.1x0.1° longitude-latitude resolution (EMEP MSC-W chemical transport model version rv4.7)[6]. Modelled N deposition is based on 2013 emissions data. (A) Total N deposition (oxidized+reduced; dry+wet), (B) percentage of dry deposition, (C) percentage of wet deposition, (D) percentage of oxidized deposition and (e) percentage of reduced deposition (Ochoa-Hueso et al. 2017).

High O_3 episodes in this region are linked to the combination of one or several of these mechanisms: (i) local/regional photochemical production and surface transport from coastal to inland regions;

[6] www.emep.int

Figure 2.19 | Nitrogen dioxide (NO_2) concentrations above the annual limit value of 40 µg·m^{-3} (based on EU Directive 2008/50/CE).

(ii) O_3 transport from higher-altitude atmospheric layers, due to air mass re-circulation in the previous days; and (iii) long-range transport of O_3 and its precursor gases (Querol et al. 2018). In the eastern Mediterranean Basin, the O_3 episodes depends on the relative strength of the high-pressure system covering the eastern Mediterranean and Balkan area: (i) strong pressure gradient with northerly winds, creating good ventilation in the Athens Basin (Kallos et al. 2014); (ii) weak pressure gradient with local/regional O_3 events prevail; and (iii) stratospheric O_3 contributions to increase surface O_3 concentrations during specific meteorological scenarios (Zanis et al. 2014; Kalabokas et al. 2015). Tropospheric O_3 concentrations observed in the summer over this region *(Fig. 2.20)* are among the highest over the Northern Hemisphere (Dayan et al. 2017).

2.3.3.3 Trace metallic elements

Metal trace elements (MTE, or heavy metals in the old designation) whose main ones are cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), zinc (Zn) and mercury (Hg) are common elements in the earth's crust (Navarro-Pedreño et al. 2008). They are also generated by human activities (Hassanien and Abdel-Latif 2008; Tovar-Sánchez et al. 2016; Merhaby et al. 2018). Urban and industrial wastewaters, atmospheric deposition and run-off from metal contaminated sites constitute the major sources of toxic metals (UNEP/MAP 2012a).

High MTE levels have been found in various soils (vegetated soils, dikes, waste rock and slag) at mining sites in Morocco. These mining activities in addition to agricultural and pastoral practices constitute a way of entry of the MTEs into the food chain and thus increase the risk of contamination of the population. Several plant species are adapted to these high levels of MTE and thus represent an important potential for the development of mining site rehabilitation strategies (Smouni et al. 2010). In agricultural soils of the Argolida Basin (Peloponnese, Greece), the MTE concentrations are high, following a decreasing order: Fe > Mn > Ni > Zn ~ Cr > Cu > Co ~ Pb > As > Cd (Kelepertzis 2014).

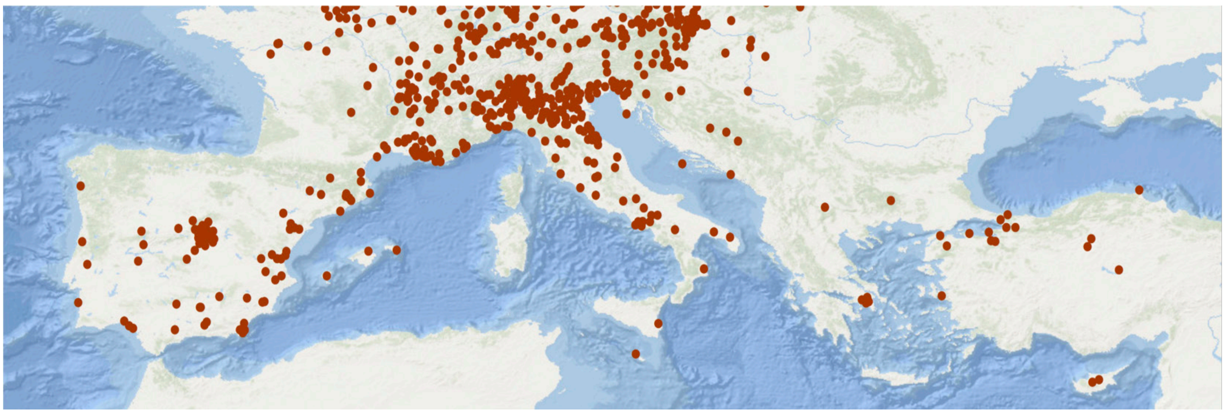


Figure 2.20 | Number of days (more than 25) above ozone (O_3) limit value of 120 µg·m^{-3} (based on EU Directive 2008/50/CE).

	Al	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Ti	Zn
Min	36,866	75.2	42.4	26,313	0.04	552	47.9	44.7	2,343	86.8
Max	72,020	102.8	52.3	36,098	0.41	2,826	60.7	74.8	3,876	129.0
Mean	58,564	85.9	47.4	31,566	0.12	893	53.3	57.8	3,065	102.3

Table 2.2 | Metal concentrations (µg g^{-1} dry weight) in marine sediment of Taranto Gulf (Ionian Sea, southern Italy) (Buccolieri et al. 2006).

In marine ecosystems, the hotspots of lead, mercury and cadmium were essentially located on the north-central and southeastern shores of the Mediterranean Basin *(Fig. 2.21)* (UNEP/MAP 2012b). Polluted surface samples on the Barcelona city continental shelf taken in 1987 reached enrichment factors of up to 490 for Hg, about 40 for Pb and Cd, and about 17 for Zn, Cr and Cu. In 2008, the data showed a decline with enrichment factors between 20 and 30 for Hg and Cd and between 5 and 12 for Zn, Cr, Pb and Cu (Palanques et al. 2017). In the Taranto Gulf (Ionian Sea, Southern Italy) *(Table 2.2)*, MTE distribution is principally influenced by industrial and urban wastes. River discharges and prevailing anticlockwise marine currents are further factors influencing metal accumulation in sediments (Buccolieri et al. 2006). In surface sediments in Lebanon (eastern Mediterranean Sea), trace metals (Cd, Pb, Zn, and Cr) contamination at Beirut Port was classified as "the most highly polluted" and as "moderately polluted" at Tripoli Port (Merhaby et al. 2018).

Freshwater ecosystems are also affected by MTE pollution. In the Ichkeul Lake Basin (northeastern Tunisia) MTE showed concentrations in the sediment samples following the order: Fe > Mn > Zn > Pb > Ni > Cr > Cu > Cd (Touaylia et al. 2016). The concentrations of Fe, Cd, Ni and Cr in the bottom sediments of the Lower Litani River Basin (Lebanon) were higher in the dry season (Nehme et al. 2014). In river sediments from a semi-arid Mediterranean Basin (Algeria), MTEs were grouped by their level of contamination: high (Pb, Cd, Zn, Cu) and low (Al, Fe, Cr, Co, Ni). Sources of this contamination were essentially industrial, agricultural and domestic waste, as well as very specific ones (gasoline station) and diffuse pollution from atmospheric deposition (gasoline, ores, aerosols) (Benabdelkader et al. 2018).

MTE are known for their toxicity, persistence, and bioaccumulation in human and animal tissues, and biomagnify (concentrate at successively higher levels in tissues) in food chains (UNEP/MAP 2012b). In the northwestern Mediterranean Sea, the concentrations of 21 trace elements showed great variability in three species; fish (sea bass, *Dicentrarchus labrax*), mussels (*Mytilus galloprovincialis*) and oysters (*Crassostrea gigas*). The essential elements (Cu, Mn and Zn) were highest in oysters, but Fe, Cr, Ni, Se, Co and Mo levels were highest in mussels. Fish had the lowest concentrations for all trace elements, which were at least one order of magnitude lower than in

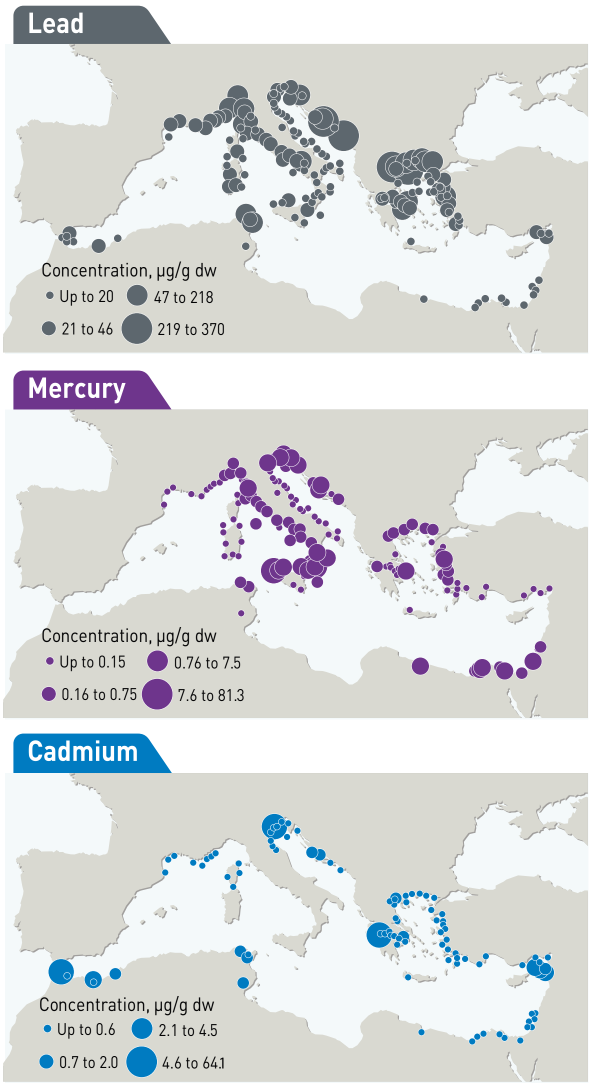


Figure 2.21 | Mean concentrations of principal trace metals in coastal sediments of the Mediterranean Basin (UNEP/MAP 2012b).

bivalves. The maximum values set by European regulations for Hg, Cd and Pb were never exceeded (Squadrone et al. 2016). The levels of As, Hg and Pb measured in some commercially key species from Sicilian coasts suggest relatively reduced pollution levels for fish resources in this part of the Mediterranean region (Traina et al. 2019).

2.3.3.4 Organic pollutants

Polycyclic Aromatic Hydrocarbons (PAHs)

The contamination of the marine environment by polycyclic aromatic hydrocarbons (PAHs) has received considerable attention since the early 1980s after occurrence of tragic marine oil spills like the Amoco Cadiz oil spill (March 1979, Brest, France; $223x10^3$ t crude oil). PAHs consist of two or more fused benzene rings. PAH distribution is controlled by multiple and inter-dependent parameters. Some of these parameters are linked to intrinsic physicochemical properties of these compounds, mainly hydrophobicity which controls their partition between dissolved and particulate phases. Others are related to the hydrological and biogeochemical characteristics of the environment including water agitation and turbidity, sediment granulometry and particulate or dissolved organic matter content. They are issued from unburned petroleum or oil-derived products (petrogenic PAHs) or from the incomplete combustion of fossil fuels and biomass (pyrogenic PAHs). Petrogenic PAHs are characterized by lighter compounds (phenanthrene, naphthalene, fluorene) and their alkylated derivatives (methyl, dimethyl etc.) and have high affinity for the dissolved phase. Pyrogenic PAHs are characterized by unsubstituted higher molecular weight compounds that exhibit more affinity for suspended particles. Distribution of petrogenic-like and pyrogenic-like compounds help identification of PAH sources in the marine environment. Eight PAHs belong to the list of priority substances: Anthracene, Benzo (a) Pyrene, Benzo (b) Fluoranthene, Benzo (ghi) Perylene, Benzo (k) Fluoranthene, Indeno (123cd) Pyrene, Fluoranthene and Naphthalene. The top 5 are listed as priority hazardous substances due to their potential toxic, mutagenic and carcinogenic effects on organisms and human health (Hussein et al. 2016).

The diagnosis of the Mediterranean with regard to these substances is governed by international and national environmental agencies (i.e., the US Environmental Protection Agency (US-EPA), the European Environmental Commission, the United Nations Environment Program (UNEP), The National Observatory of the Environment and Sustainable Development, ONEDD (Algeria), National Environmental Protection Agency, ANPE (Tunisia), Egyptian Environmental Affairs Agency, EEAA (Egypt) that recommend the monitoring of 16-19 PAHs with special emphasis on PAHs micropollutants in marine matrices.

The overall data set suggests that both petrogenic and pyrogenic PAHs contribute to the PAH pool in Mediterranean coastal waters. Pyrogenic inputs increase in urbanized areas due to large atmospheric inputs and deposition in winter (Guigue et al. 2014; Barhoumi et al. 2018). Total PAH inputs from the Rhône river represent 50 t yr^{-1} whereas inputs from sewage treatment plant are much lower (<1 t yr^{-1}) (Witkowski et al. 2017). Due to their hydrophobicity, PAHs are considered to be preferentially associated with particles in coastal marine waters and sediments (Adhikari et al. 2015). Dissolved PAHs concentrations in the water column may be 4-fold higher than in suspended particles (Guigue et al. 2011). Dissolved PAH concentrations are in the range 0.158-3.655 ng L^{-1} (∑18 PAHs) (Berrojalbiz et al. 2011) in Mediterranean open sea waters whereas higher concentrations characterize coastal urbanized areas, up to 560 ng L^{-1} (∑32 PAHs) in Marseille-Gulf of Fos (France) (Guigue et al. 2011), 12–267 ng L^{-1} (∑17 PAHs) in Venice lagoon (Italy) (Manodori et al. 2006), 13–120 ng L^{-1} (∑7 PAHs) in Alexandria coastal waters (Egypt) (El Nemr and Abd-Allah 2003). In coastal sediments, concentrations range from 10-200 ng g^{-1} d.w. (dry weight) (Asia et al. 2009) in Marseille Bay, France (Gogou et al. 2000), Cretan Sea, Greece (Zaghden et al. 2005), Sfax, Tunisia (Cannarsa et al. 2014), Liguria, Italy (Merhaby et al. 2015), Tripoli harbor, Lebanon (Emara et al. 2008) and Eastern harbor, Egypt (Emara et al. 2008).

PAH concentration gradients are pronounced along coast-open sea transects. PAH concentrations rapidly decrease from the vicinity of rivers, estuaries and small effluents towards coastal and offshore waters. The contamination of estuaries of large rivers and that of harbors in the vicinity of big industrial and urban centers indicates a poor quality environment where a potential risk to the local population may occur (Barhoumi et al. 2018). Otherwise in most Mediterranean coastal waters, data reflect contamination levels from slightly polluted to polluted. Industrial areas near the cities of Sfax and Gabès (Tunisia) were reported moderate-to-highly impacted by hydrocarbons compared to other Mediterranean coastal environments (Fourati et al. 2018b, 2018a). They remain globally higher than those reported in the northern Gulf of Mexico and much lower than those recorded in Chinese coastal environments (Zhou and Maskaoui 2003; Fourati et al. 2018b).

In some cases, PAH concentrations are influenced by physical circulation processes that can lead to deposits of contaminants an order of magnitude higher offshore that those near the source of pollutants. Episodic processes of pollutants redistribution may also significantly affect the pollution status of marine areas. For example, the physical accumulation of PAH at air/sea interface (x 200-1,000) during microlayer formation in absence of wind (Wurl and Obbard 2004), followed by PAH scattering during microlayer disruption by wind blow recovery, can locally enhance PAHs concentrations and impact the biota. Similarly, PAH sediment remobilization during resuspension events may greatly modify their potential harmful effects on marine biota (Guigue et al. 2017). In Toulon bay (France), the resuspension of highly contaminated surface sediments (concentration of Σ34 PAHs = 38.2×10^3 ng g^{-1}) led to a 10-fold increase of dissolved Σ34 PAH concentrations in the water above. The remobilization in seawater was higher for 4-6 ring PAHs, especially benzo(g,h,i) perylene, whose concentration exceeded the authorized limit values of the European Water Framework Directive (Guigue et al. 2017). It is important to monitor pollutants not only at active industrial facility sites but also in disused industrial areas close to the sea border where remnant pollution can produce chronic adverse effects on marine biota.

Species feeding on particles and phytoplankton may bioaccumulate and/or bioamplify PAH concentrations in their body tissues. Measuring accumulation of PAHs in mussel bivalves from the *Mytilus edulis* complex has become a European Commission control strategy of marine waters quality in the Mediterranean (Olenycz et al. 2015; Sire and Amouroux 2016). Different metabolites may be measured in fish and shells and considered as markers of exposure to PAH. Research on the relationships with emergent contaminants is at its beginning and cocktail effects have not been much studied yet. Microplastics have a high potential to adsorb these hydrophobic contaminants and to transfer them throughout the food web to the deep ocean for longer sequestration time.

Pesticides

Pesticide Active Ingredients (PAI) can be considered as a contaminant as well as a pollutant, in the compartments where they are detected. Mainly originated by agricultural activities, water pollution by PAI is a concern for continental water resources (rivers, lakes and aquifers) and coastal and marine environment of the Mediterranean Sea. Studies carried out in southern Europe showed the high leaching of herbicides in Mediterranean weather conditions (Louchart et al. 2001), allowing the contamination of groundwater resources. In the other side, by runoff process, surface waters would be contaminated by PAI and their metabolites or degradation products. A large number of pesticide active ingredients (PAI) (over 1,300) are presently used or were used until non-approval or non-renewal in Europe (European Food Safety Authority 2011) and even modern screening methods limit the number of PAI analyzed in one sample (< 450) (Rousis et al. 2017). In water bodies, the maximum allowable concentrations are 2 µg L^{-1} for each PAI and 5 µg L^{-1} for all quantified PAIs. For drinkable water these limits are 0.1 µg L^{-1} and 0.5 µg L^{-1} respectively.

Concentrations of these molecules in water bodies (surface and underground waters) were qualified and quantified in nearly all the countries around the Mediterranean Sea. But a recent review at world scale (Stehle and Schulz 2015) stated the difference of availability of referenced data sets for insecticide concentrations in water: notably in the North and the North-East Africa poor information was available.

In European countries, particularly in France (Dubois et al. 2010; Lopez et al. 2015), Italy (Onorati et al. 2006; Meffe and de Bustamante 2014), Spain (Balaguer et al. 2018) and Greece (Lekkas et al. 2004; Konstantinou et al. 2006) implementation of the EU Water Framework Directive (WFD) produced large public data sets for pesticide concentrations in surface waters and aquifers by state administrations. According to the statistical office of the European Union, Spain and Italy are the countries with most use of pesticide. As a result, in those countries, pesticides are one of the most frequently detected classes of micro-pollutants in water[7]. It is not possible to give in this report an exhaustive account of all PAI mentioned in the literature as encountered in Mediterranean waters. Thus, we decided to focus on the most frequently detected PAI and give some examples of maximum concentrations measured.

In water bodies most frequently mentioned PAI-insecticides already in use are chlorpyrifos (18.8 µg L^{-1}) (Ccanccapa et al. 2015), dimethoate (0.640 µg L^{-1}) (Campo et al. 2013), malathion (0.048 µg L^{-1}) (Yurtkuran and Saygı 2013), imidacloprid (0.350+/-0.433 µg L^{-1}) (Herrero-Hernández et al. 2013) and diazinon (14.5 µg L^{-1}) (Youssef et al. 2015). Prohibited PAI-insecticides mentioned are DDT (dichlorodiphenyltrichloroethane) and its metabolites (0.40 to 3.22 µg L^{-1}) (Dahshan et al. 2016), HCB (hexa-

[7] https://www.eea.europa.eu/airs/2018/environment-and-health/pesticides-sales

chlorobenzene) (1.1 µg L^{-1}) (Youssef et al. 2015) and endosulfan (0.247 µg L^{-1}) (El Bakouri et al. 2008).

The most frequently mentioned PAI-herbicides in use are simazin (3.18 µg L^{-1}) (Konstantinou et al. 2006), terbuthylazin (0.0219 µg L^{-1}) (Ricart et al. 2010), linuron (13.13 µg L^{-1}), 2,4-D (20 µg L^{-1}) and glyphosate with its metabolite AMPA (167 µg L^{-1}) (Meffe and de Bustamante 2014). Most prohibited PAI-herbicides mentioned in studies are atrazine-desethyl (0.158 µg L^{-1}) (Campo et al. 2013), metolachlor (1.120 µg L^{-1}) (Konstantinou et al. 2006), DEA (diethyl-atrazine) (1.98 µg L^{-1}) (Hildebrandt et al. 2008), diuron (0.0169 µg L^{-1}) (Robles-Molina et al. 2014), DIA (deisopropyl-atrazine) (8 µg L^{-1}) (Shomar et al. 2006), alachlor (0.213 µg L^{-1}) (Stamatis et al. 2013), isoproturon (7 µg L^{-1}) (Ricart et al. 2010) and molinate (0.026 µg L^{-1}) (Gómez-Gutiérrez et al. 2006).

Concentrations of PAI-fungicides are less mentioned than other PAI. Metalaxyl (0.49 µg L^{-1}) (Hildebrandt et al. 2008) and carbendazim (1.81 µg L^{-1}) (Licciardello et al. 2011) have been reported.

For the future, in most southern Mediterranean countries, prohibitions of particular PAIs for agricultural uses are applied within a few years delay from European Commission decisions. Many countries developed programs to reduce pesticide use that have uncertain effects like in France (Guichard et al. 2017). New proposed PAI are characterized by shorter standard half-lives and lower dose requirements, nevertheless there is no assurance that these "new" PAI once used widely in many different contexts will have virtuous environmental spreading behaviors.

Climate changes may have conversational effects on PAI water contamination. Firstly, some authors predict an increase use of pesticides to compensate increases of abundance and seasonal activity of bioaggressors (Boxall et al. 2009), although temperature enhancement has been declared to decrease PAI efficiency by conditional resistance of bioaggressors towards herbicides, insecticides and fungicides (Matzrafi 2019). Under these conditions, doses applied should be increased to guarantee the same protection with unseen consequences in environment spreading. Daily temperature fluctuation may increase PAI toxicity like for chlorpyrifos (Verheyen and Stoks 2019), but temperature enhancement should boost degradation of most known PAI probably shortening their half-life.

Change in precipitation patterns with increased occurrence of extreme precipitation events should also modify agricultural patterns. For instance, rainfed barley Mediterranean production yields would decrease (Verheyen and Stoks 2019) fueling some changes in the crop systems. Implementation of more intensive systems (Malek and Verburg 2018) can imply an increase of treatments. Increasing weight of extreme events would increase erosion (Raclot et al. 2018) and then promote displacement from land to sea for the PAIs easily adsorbed on soil particles like clay and organic matter.

2.3.3.5 Emerging contaminants

In the Mediterranean Basin, 63% of coastal settlements with more than 2,000 inhabitants operate a wastewater treatment plant, while 37% do not. Secondary treatment is mostly used (67%) in Mediterranean treatment plants, while 18% of the plants have only primary treatment (Chatha et al. 2017). As a consequence of this technical, social and environmental issue, different types of chemical substances are released into the environment (Gros et al. 2010; Ratola et al. 2012; Moreno-González et al. 2015; Paluselli et al. 2018b, 2018a). Among these substances, emerging contaminants (ECs) are a category that has received special attention over the last 25 years. ECs are defined as "contaminants of emerging concern that are naturally occurring, manufactured or man-made chemicals or materials which have now been discovered or are suspected present in various environmental compartments and whose toxicity or persistence are likely to significantly alter the metabolism of a living being" (Sauvé and Desrosiers 2014).

There are different classifications of these contaminants due to their usage or origin and effects. For example, (1) antibiotics, (2) antimicrobials, (3) detergent metabolites, (4) disinfectants, (5) disinfection byproducts, (6) estrogenic compounds, (7) fire or flame retardants, (8) fragrances, (9) insect repellants, (10) PAHs (polyaromatic hydrocarbons), (11) personal care products, and (12) pesticides or insecticides (13) pharmaceuticals, (14) plasticizers, (15) reproductive hormones, (16) solvents, (17) steroids and (18) surfactants (Singh and Kumar 2017).

ECs that are very soluble in water (tetracycline, sulfamethoxazole, carbamazepine, and erythromycin, etc.) receive more attention than others because of their impact on the environment (Klaper and Welch 2011). Potential ECs sources and pathways of ground and surface water pollution are shown in *Fig. 2.22.* Typically, the route of these compounds towards a water body begins with the excretion of the metabolites and parent compounds and their disposal to the wastewater treatment plants (Barrios-Estrada et al. 2018).

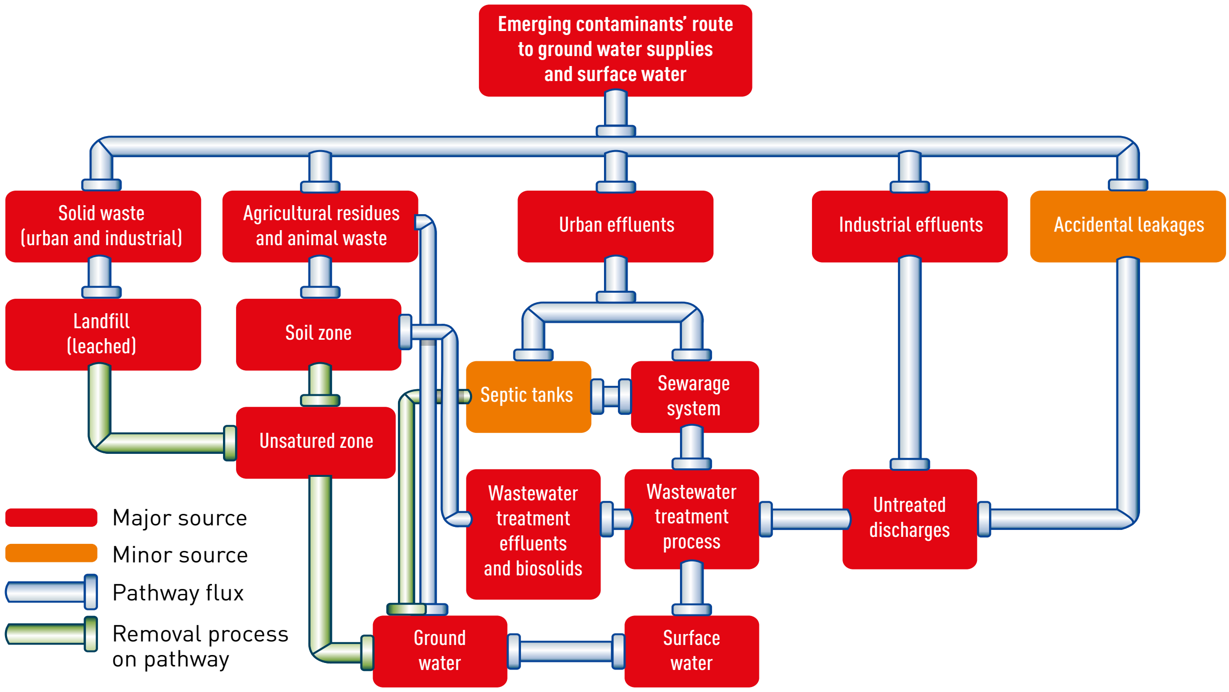


Figure 2.22 | Potential sources and pathways for grounds and surface water pollution (Barrios-Estrada et al. 2018).

The endocrine disruptors (EDs) are a subdivision of ECs with chemicals that interfere with the body's endocrine system and produce adverse developmental, reproductive, neurological, and immune effects in humans, abnormal growth patterns and neurodevelopmental delays in children (Moreno-González et al. 2015). The majority of the EDs come from products used to fight undesirable wildlife and agricultural threats (Moreno-González et al. 2015), for example, pesticides, fungicides and rodenticides, synthetic products used in plastic industry (bisphenols or phthalates) and a variety of buildings materials, isolation materials (polychlorinated biphenyl and metals). A list of common sources of EDs is shown in *Fig. 2.23.*

The presence of ECs in Mediterranean Basin is well documented. Phthalic Acid Esters (PAEs), including dimethyl phthalate (DMP), diethyl phthalate (DEP), di-isobutyl phthalate (DiBP), di-n-butyl phthalate (DnBP), benzylbutyl phthalate (BzBP) and diethylhexyl phthalate (DEHP), with total concentrations ranging from 130 to 1,330 ng L^{-1} were found in Marseille Bay (northwestern Mediterranean Sea) (Paluselli et al. 2018b). High concentrations of PAEs were also observed in deep waters offshore (310.2 ng L^{-1}) as well as in the Rhône River (615.1 ng L^{-1}) (Paluselli et al. 2018a).

A total of 20 pharmaceuticals in sea water and 14 in sediments were found at concentrations from low ng L^{-1} up to 168 ng L^{-1} (azithromycin) in sea water and from low ng g^{-1} up to 50.3 ng g^{-1} (xylazine) in sediments of Mar Menor lagoon located in the South East of Spain (Moreno-González et al. 2015). Pharmaceutically active compounds (PhACs) were detected in the Evrotas River (Southern Greece) waters. The diuretics and the analgesics/anti-inflammatory class were the most abundant, followed by antihypertensives, psychiatric drugs, ß-blocking agents and antibiotics and the concentration levels ranged from 0.31 ng L^{-1} up to 51 ng L^{-1} (Mandaric et al. 2019). Antibiotics were detected in more than 90% of the water samples collected from a Mediterranean river (Llobregat, Spain) and the concentration levels ranging from 0.3 ng L^{-1} (flumequine) to 907.6 ng L^{-1} (sulfamethoxazole) (Proia et al. 2013). Triclosan (an antimicrobial) was reported to be a contaminant of the Llobregat and Ebro rivers (Spain) and the concentrations in some samples were higher than 150 ng L^{-1}. These concentrations should be considered significant considering the toxicity of these compounds and their expected ability to be a precursor of other highly toxic compounds such as dioxins (Kantiani et al. 2008). Azithromycin (antibiotic) was measured at 16,633 ng L^{-1} in a tributary of El Albujòn (Spain) (Moreno-González et al. 2015) and acetaminophen was detected at 3,000 ng L^{-1} off Thessaloniki, Greece (Nödler et al. 2014). Some drug classes, such as analgesics, antibiotics and betablockers, were still quantified at levels between 0.3 (metoprolol) and hundreds of ng L^{-1} (azithromycin) in seawater. In

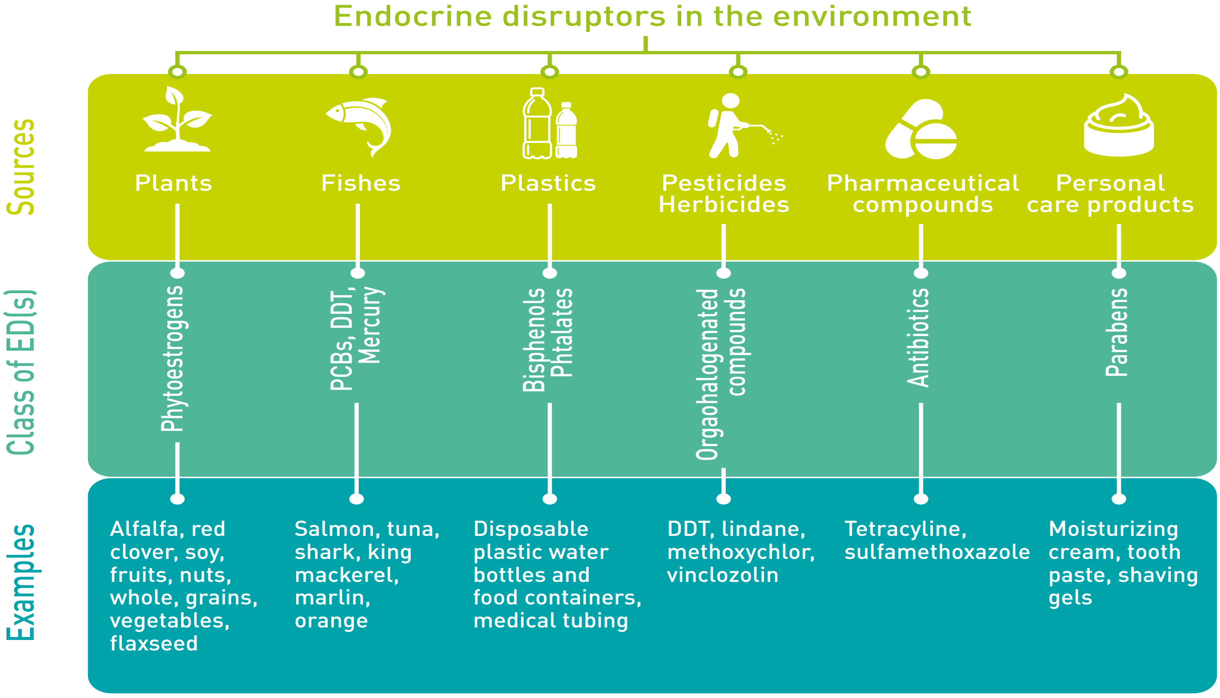


Figure 2.23 | Common sources of endocrine disruptors in the environment (Barrios-Estrada et al. 2018).

particular, six substances (e.g., azithromycin, amoxicillin, venlafaxine, salicylic acid, acetaminophen and ibuprofen) were measured at median concentrations higher than 20 ng L^{-1}. Pharmaceuticals were also found in coastal and oceanic waters adjacent to the Strait of Gibraltar (Biel-Maeso et al. 2018).

The impacts of exposure to some ECs have caused concern for both people and wildlife. Many of these substances may cause disorders of the nervous, hormonal and reproductive system, thus posing adverse health outcomes (Rezg et al. 2014; Bilal et al. 2018, 2019; Hernandez-Vargas et al. 2018; Ullah et al. 2018; Rasheed et al. 2019). These synthetic chemicals exhibit geno- or cytotoxic activity and can cause obesity, diabetes, cardiovascular and reproductive disorder or even leads to cancer (Tiwari et al. 2012). Etteieb et al. (2016) have shown that the components responsible of cytotoxicity in water samples from Medjerda river in Tunisia were mainly cyclopentasiloxane, decamethyl (D5), cyclohexasiloxane, dodecamethyl (D6), D-limonene, and ergoline-8-methanol, 8,9-didehydro-6-methyl. Some ECs, such as 17-α-estradiol, bisphenol A and phtalates were reported to alter marine community structure (Essid et al. 2013; M'Rabet et al. 2019).

2.3.4 Biological pollutants

Numerous viruses or bacteria of human or animal origin can spread in the environment and infect people via water, air and food, mostly through ingestion and occasionally through skin contact. These viruses and bacteria are released into the environment by various routes including water run-offs and aerosols. Furthermore, they can infect humans exposed to contaminated surface waters and ground water used for agriculture irrigation with severe consequences for human health (de Giglio et al. 2017). In most semi-arid areas, groundwater and surface waters constitutes an important and strategic resource, particularly as water stress increases and water resources of good quality become scarce (El Ayni et al. 2013). Molecular epidemiology and regular surveillance are necessary to elucidate the public health hazards associated with exposure to environmental viruses and bacteria (Cabral 2010; Rodriguez-Lazaro et al. 2012) especially in the South region where the water contamination combined to water scarcity strongly have large socio-economic impacts (UN-Water 2014). Climate projections for the Mediterranean climate areas estimate general warming and changes in precipitation distribution. Mediterranean coastal rivers are subject to flash floods during extreme events that transport the majority of the annual loads of bacteria and other contaminants (Chu et al. 2011). The frequency of extreme summer precipitation events increased over large regions of the Mediterranean (Giorgi and Lionello 2008) (*Sections 2.2.5* and *3.1.3.3*), increasing so the supply of fecal bacteria and viruses to the coastal zone. In a global context, wastewater management will be the key to preventing environmental dispersion of human

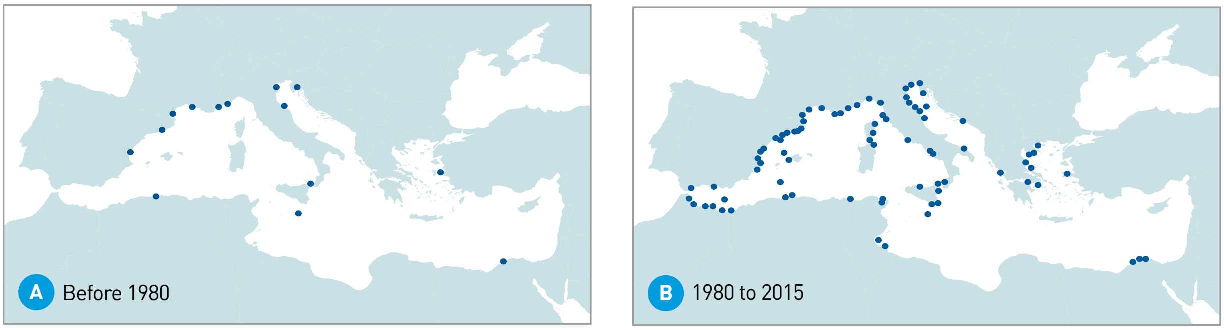


Figure 2.24 | Harmful algal blooms recorded before 1980 (A) and between 1980 and 2015 (B) along the Mediterranean coast (Cecchi et al. 2016).

fecal pathogens in future climate change scenarios (Rusiñol et al. 2015).

Harmful Algal Blooms (HABs) are sporadic phenomena triggered by massive proliferations of phytoplankton species reaching high cell concentrations (Sellner et al. 2003) that are suspected to have greater occurrences due to global warming (Hallegraeff 2010). HABs have environmental impacts (red-tide, mucilage production, anoxia) (Rodger et al. 2011; D'Silva et al. 2012) and represent serious economic threat for fisheries, aquaculture and tourism (Hoagland et al. 2002). They may also harm human health, since 40% of blooming microalgae are able to produce toxins responsible of different human intoxications (Santi Delia et al. 2015). HABs can occur in both freshwater and marine environments. In the Mediterranean Basin, marine ecosystems have received more attention although occurrence of HAB in freshwater lakes are reported (Cook et al. 2004; Romo et al. 2013) and are suspected to increase with climate change in the region (Romo et al. 2013) with consequences for potable water supply (Lévesque et al. 2014). Before 1980, HABs were rarely documented in the Mediterranean Sea. Since then, adverse events and several toxic episodes have been reported in different coastal regions *(Fig. 2.24)* (Cecchi et al. 2016; Garrido et al. 2016). Nowadays, harmful and toxic phytoplankton species become dominant in many coastal environments of the Mediterranean Sea.

Most toxic events in the Mediterranean Sea are mainly related to the dinoflagellates *Alexandrium* (known to induce Paralytic Shellfish Poisoning - PSP) and *Dinophysis* (producers of toxins causing the Diarrheic Shellfish Poisoning - DSP) (Vila et al. 2001; Penna et al. 2007). *A. minutum* is the most observed dinoflagellate, with recurrent blooms (10^3-10^8 cells l^{-1}) in the Ebro Delta and the Gulf of Gabès, inducing significant fish mortality (Abdennadher et al. 2012; Garcés and Camp 2012). Blooms of *A. catenella* (10^4 cells l^{-1}) have been observed both in the northern (Thau Lagoon) and the southern Mediterranean Sea (Bizerte Lagoon, Tunisia), causing sometimes shellfish contamination (Laabir et al. 2013). The epi-benthic dinoflagellate *Prorocentrum lima* (producer of DSP toxins) has been detected on most Mediterranean coasts (Garcés and Camp 2012), sometimes with high densities (>10^7 cells l^{-1}) (Aissaoui et al. 2014; Moncer et al. 2017). Blooms of *Karenia selliformis* (10^3–10^5 cells l^{-1}), associated with intensive fish mortality, were reported for many years in the Gulf of Gabès (Feki et al. 2013). Recently, human health problems are caused by blooms of *Ostreopsis cf. ovata* (10^3-10^4 cells l^{-1}) in Italy, Spain, Algeria and France (Ciminiello et al. 2014). New records of *Gambierdiscus* and *Fukuyoa* in the West and East of the Mediterranean Sea increase the risk of Ciguatera intoxication (Laza-Martinez et al. 2016).

Toxic diatoms *Pseudo-nitzschia* (producers of toxins causing the Amnesic Shellfish Poisoning - ASP) showed also blooms (10^3-10^7 cells l^{-1}) in Mediterranean coasts of Spain, France, Greece, Italy and Tunisia, and contamination of mussels by ASP were reported (Sakka Hlaili et al. 2016). Mussel contamination by ASP toxin were reported in Spanish and Tunisian waters and have been linked to the blooms of *Pseudo-nitzschia* spp. and *Nitzschia bizertensis*, respectively (Giménez Papiol et al. 2013; Bouchouicha-Smida et al. 2015).

Toxic and harmful algal blooms continue to increase in magnitude, frequency and geographical distribution around the world and over the Mediterranean Sea (Hallegraeff 2010). The chronic eutrophication and climatic change including global warming are reported as significant factors involved in this global increase of HABs and toxic events (Anderson et al. 2012; Sakka Hlaili et al. 2016).

2.4 Land and sea use changes

2.4.1 Land use changes

2.4.1.1 Past trends and recent dynamics

Mediterranean landscapes

The climatic conditions of Mediterranean Basin have played a key role for drawing its landscapes, which result from the interaction between human activity, a complex topography, and an extreme varied soil and climate (Pinto-Correia and Vos 2004). Altitudinal gradients and the distance from the sea are the main factors differentiating the landscapes of the Mediterranean Basin (Pinto-Correia and Vos 2004), and the main ecosystems contributing to outline these landscapes include the Mediterranean shores with a high number of habitats such as areas of dunes, rocks and wet areas. Moving away from the coast, Mediterranean maquis (a typical dense and closed shrub vegetation, mainly constituted by sclerophyllous species) and forests evolve, which includes evergreen shrubs or trees fully adapted to xeric conditions. In many cases, the maquis derives from the evergreen Mediterranean forest that develops in less water limiting growing conditions. Mediterranean steppic prairies represent the typical degradation of maquis that advances desertification. Finally, the Mediterranean deciduous forest extends where climates gradually shifts from typical Mediterranean to inland.

The modifications occurred in the Mediterranean landscapes are the result of land use practices that increased in intensity since the Neolithic, when in the Middle east domestication of plant and animals took the place of hunting and gathering (Blondel 2006). Human pressure on the natural ecosystem greatly varied depending on societal evolution developed over the area. In any case, wood exploitation has been key in localized periods of empires expansion. Forest destruction was the first step consequence of human pressure on the natural habitat as the effect of increasing agricultural activity and livestock and wood exploitation in the Mediterranean Sea trade cycle in order to cover the needs of the maritime empire (Barkaoui 2003). There is a long history of urbanization, reflected in countless archeological and historical settlements over the region (Diappi 2015).

As a result, the Mediterranean landscape is a complex mosaic of alternating semi-natural habitats. Grasslands and pastures represent one of the most spread land use in European Mediterranean areas in the plain and low hilly areas (Cosentino et al. 2014) and they can provide forage for grazing animals or hay for conservation. The use of terraces in hilly areas permitted the cultivation on slopes of olive groves, vineyard and sowing crops while at the same their cultivation represented a means to reduce soil erosion, prevent run-off and increase water saving (Blondel 2006). The Dehesa-Montado System (typical of Spain and Portugal) is characterized by low density trees (represented by evergreen Mediterranean oaks) combined with crop production or pastoral activities, mainly represented by animal grazing. This system integrates the three main rural activities (forest/cork product harvesting, livestock husbandry, and agriculture) within a single landscape that consists of grass and trees. This system that combines extensive grazing of natural pastures, cereal cultivation and harvest of wood products, has shown remarkable stability, biodiversity, and sustained productivity over 800 years or longer as the result of the maintenance of botanically rich mosaic-like herbaceous plant layers (Joffre and Rambal 1993). In addition to agricultural production, these complex systems contributed, at the same time, to several ecosystem services such as preservation of the environment and its natural resources securing the sustainability of the system (Blondel 2006; Hao et al. 2017).

Despite small-scale traditional farming systems has been practiced for long time and it is still adopted in many parts of the Mediterranean region, huge changes in agricultural practices have taken place during the last 50-100 years across several Mediterranean areas as driven by increasing profitability of new agricultural systems (Debolini et al. 2018). The greatest changes involved uprooting of ancient and small-scale vineyards, orchards and olive groves, which turned into industrial scale fruit or olive plantations. Similarly, mixed rotational farming systems were simplified and replaced usually by intensive monocultures, requiring high inputs (i.e., high fertilization rate and water requirement) (Debolini et al. 2018). Intensive and large-scale farming systems also required the creation of new infrastructure, such as basins to cope with water shortage, which contributed to change the natural landscape. All these changes caused unsustainable pressure on the surrounding environment, resulting in either loss of wildlife-rich habitats and socio-economic viability over large parts of the region due to land-abandonment of small-scale farmers.

Recent changes in Mediterranean landscapes

The rate of change of landscapes in the Mediterranean Basin has increased since the second half of the 20th century. Many regions in Mediterranean Europe experienced the abandonment of marginal lands, especially in arid and mountain areas (Lasanta et al. 2017) *(Section 3.2.3.1)*, and the following development by shrubs and tree species. Many studies show the abandonment of typical features like farming terraces, olive orchards, and upland grasslands leading to non-managed reforestation; for example, in 29% of the Iberian Peninsula for the 1989-2004 period (Hill et al. 2008); all-over Italy during 1990-2013, small forest patches cover increased a 27.4% (Sallustio et al. 2018), while 24% of pastures were turned into forests in areas of Tuscany from 1954 to 2005 (Amici et al. 2017); in Italian mountains, 16.3% of agricultural areas disappeared during the 1990's (Conti and Fagarazzi 2015); similarly in pre-alpine France during 1956-1991 (Taillefumier and Piégay 2003); in areas from the Eastern Mediterranean region of France, 14.2% of crops and 78.2% of pastures were converted into forests from 1958 to 2010 (Abadie et al. 2018); a 35.6% of vineyard areas in Serbia was transformed into meadows and pastures during 1985-2013 (Perović et al. 2018); Mediterranean islands as Elba, where up to 52% of agricultural areas were abandoned during the 1954-2000 period (Carta et al. 2018), and Lesvos, showing recent (2001-2011) slight decreases in agricultural land (Van der Sluis et al. 2016).

The remaining agricultural systems have generally become more intensive, with a shift towards livestock production and an increase of industrial inputs (fertilizers and pesticides), as it has been reported for Spain, especially since the 1960's to 2008 (Guzmán et al. 2018), and punctual areas of Greece and Italy, from 2001 to 2011 (Van der Sluis et al. 2016) *(Section 3.1.2.1)*.

Conversely, scarce and mainly pre-21st century studies from North Africa show extreme land degradation due to overexploitation (Le Houérou 1995), principally by grazing pressure but also by forest conversion to agriculture and fuel-wood recollection. For the total North Africa and Middle East regions, the rate of deforestation increased from the 1980's to the 1990's by 160%, the fastest increase worldwide (Hansen and DeFries 2004). For example, in Morocco, it has been registered forest regression and degradation from 1962 to 1992 (Rejdali 2004), while in Rogassa (Algeria) a long-term experiment (1975-1993) demonstrated the main role of overgrazing in such degradation (Slimani and Aidoud 2004). Recent studies based in new observations and remote sensing show similar desertification trends in most parts of the Maghreb (Hirche et al. 2018). Grazing has been also the principal factor of forest degradation in northern Mediterranean islands as Crete during the 1977-1996 period (Hostert et al. 2003). Moreover, increases over 10% in livestock density have been observed between 2001 and 2011 in Portofino (Italy) and Lesvos (Greece) (Van der Sluis et al. 2016), and up to 40.1% in Nisyros (Greece) from 1991 to 2001 (Petanidou et al. 2008), showing that land abandonment does not always result in forest encroachment.

Changes in Mediterranean landscapes have been particularly intense in metropolitan areas and their surroundings. These landscapes are growing very rapidly all-over the Mediterranean (17% between 1990 and 2000) (Underwood et al. 2009) but especially in coastal areas. Urbanization mainly occurred at the cost of arable land (e.g., in Barcelona) (Basnou et al. 2013) and forested areas, as generally reported in the Mediterranean (Gerard et al. 2010). However, in some periurban areas, agricultural land has also increased following the growth of cities, as in Murcia (Spain) between 1995 and 2007 (García-Ayllón 2018). Examples could be found also in eastern Mediterranean, such as in Erdemli (Turkey), where the total length of the roads increased 23.6% between 2004 and 2015 following the growth of periurban agriculture (Alphan 2018).

2.4.1.2 Principal impacts of land use changes

Forestry and other natural resources

New forests after land abandonment could alter biodiversity patterns in the Mediterranean (Fabbio et al. 2003), as has been demonstrated for the range of certain bird species (Gil-Tena et al. 2010). New forests are established by tree species whose dynamics have been favoured by recent land use changes, even more than climate (Améztegui et al. 2010). Meanwhile, other species as oaks are less benefited by such changes (Acácio et al. 2017), although general patterns show in fact successional dynamics towards Quercus dominance at the expense of Pinaceae (Alfaro-Reyna et al. 2018). Fire regimes are also altered, as fuel continuity is increased facilitating fire spread, which in turn could result in more landscape homogeneity (Loepfe et al. 2010). Also, forest continuity could increase forest insect pest spread (Hódar and Zamora 2004). The effect of increased forest cover in the diminution of water resources is more conflictive although the pattern is clearer in catchments with records of large and rapid forest expansion (Gallart et al. 2011) *(Section 3.1.1.3)*. During the first stages of land abandonment there is also a great risk of land degradation due to soil and nutrient loss (Thornes 2009). Nevertheless, the contribution of these new forests to carbon storage is certain (Vilà-Cabrera et al. 2018).

Despite the natural forest recovery after land abandonment, forests in the Mediterranean Basin are still interested by many different threatening factors, such as wildfires, overgrazing, incorrect management and extreme climate and meteorological events (drought, windstorms) and this in turn could lead eventually to desertification (Vilà-Cabrera et al. 2018). Wildfires become more and more frequent due to drier summers coupled with wood expansion due to land abandonment (Pausas and Fernández-Muñoz 2012). Although evidence indicates that fires are decreasing due to increased efforts in fire suppression all-over Mediterranean Europe in more recent periods (Turco et al. 2016), there is large potential risk of mega-fires in the near future (Loepfe et al. 2011). Changes of traditional silvicultural schemes able to produce high productive goals to a more extensive management or forest abandonment has led to deep modifications of composition and structure.

Another major factor affecting degradation of Mediterranean forests is grazing by domestic animals that utilize understory especially in period of reduced forage availability in pastures (such as summer or, in certain cases, winter). The traditional grazing in forest formation should not be banned as when the stocking rate is adequate, this result in a proper sustainable forest management that can reduce potential fuel biomass and can preserve this tradition landscape (Kairis et al. 2015) while, on the contrary, overgrazing can produce erosion, reduction in soil cover, losses of nutrients and in this way is it can be considered one of the most important factors of desertification in Mediterranean areas (Papanastasis and Kazaklis 1998). Droughts, heat waves or windstorms can have a negative effect on many forests in the Mediterranean Basin producing reductions in forest growth and of forest declines (Vayreda et al. 2012) that prelude to land degradation and, in turn, desertification. All these factors produce stresses to forests that can be exacerbated in the next decades by climate change (Valladares et al. 2014).

Land degradation is the principal consequence of plant cover loss (by uncontrolled forestry, overgrazing, fires, etc.), which could lead to desertification in combination with increasing aridity and extreme climatic events (Thornes 2009). In mountain environments, forest and understory cover loss, principally by overgrazing, have been clearly associated to higher erosion rates (Cheggour et al. 2012), as well as recent increases in forest cover were linked to the opposite (Barreiro-Lostres et al. 2017). Soil erosion is a serious problem throughout the Tunisian Dorsal (the easternmost part of the Atlas mountain range). It has been estimated that 7% of the area is badly damaged by erosion and 70% of the area is moderately damaged (DG/ACTA 1993). This degradation is an accumulated effect of agricultural strategies adopted over the Tunisian semi-arid areas during the last three millennia (Jebari 2009). If we only consider short-term effects, the degradation can be partly attributed to the building of large dams during the 1960s and 1970s. This was done without giving sufficient attention to proper management of upper catchment areas (Jebari et al. 2010). However, the contribution of the specific bioclimatic conditions of the Mediterranean climate in this degradation should not be underestimated. The soils are better characterized by the degradation of rock material rather than their organic matter content (Cerdan et al. 2004; Cudennec et al. 2007). Consequently, they are not well developed and often shallow. The human influence is crucial on catchment scale in terms of landscape degradation that affects the hydrological regime (Jebari et al. 2010). In the last hundred years, continuous changes were undertaken by the introduction of new crop cover (in fact, the most important change leading to better water management in the southern part of the Mediterranean may come from improving water efficiency in agricultural irrigation) (Berndtsson et al. 2016), deforestation, urbanization, river network modification, dam buildings and embankment. Fortunately, better water resource planning, reservoir maintenance, shortage, flood management, and hydro-agricultural infrastructure design are currently promoted (Verkerk et al. 2017) *(Section 4.5.1)*.

Urbanization is considered a major driving force of biodiversity loss and biological homogenization (Grimm et al. 2008), causing landscape fragmentation, dramatic loss of open habitats and of the land use gradient, replacing adjacent land uses such as agricultural and more natural vegetation. Urbanization is also one of the main drivers of introduction of non-indigenous species, generating high propagule pressure, and frequent and intense disturbance with complex consequences for biodiversity (Basnou et al. 2015; Clotet et al. 2016). The strong human pressure has also contributed to increase water shortages, pollution, forest fires, and the abandonment of ancient pastoral regimes. Recent studies also demonstrate the negative consequences of new artificial areas in coastal dunes habitats, both affecting carbon stocks (Carranza et al. 2018), and generating more complex effects when stabilizing the natural changing dynamics of dune systems (Manzano et al. 2019).

Crops and livestock

Food production in the Mediterranean region is changing rapidly, due to multiple local and global social and environmental changes. The increased number of urban and displaced people increase demand for food in urban areas, with limited agricultural production and with great water restrictions (FAO 2017a). Scarce resources, such as fisheries, are being exhausted (see next section). Evidence for the limited capacity to cope with these challenges can be documented in recent history. For example, water reserves were not able to cope with extensive droughts in the last two decades in Spain, Morocco and Tunisia, causing many irrigation dependent agricultural systems to cease production (Faurès et al. 2002; Garrido et al. 2006; FAO 2015).

Livestock production, mainly located in semi-arid and arid lands has shifted from extensive modes to systems heavily dependent on feed grain (32% of total food imports), inducing high poverty rates and rural exodus and rendering production sensitive to climatic shifts elsewhere (*Sections 3.1.2.1* and *4.5.1*).

Besides soil erosion, the major land degradation processes in the Mediterranean Basin are soil sealing, compaction mainly due to agricultural intensification, salinization, and contamination due to industrial activities. Soil organic carbon stocks tend to decrease when transforming grasslands, forest or other native ecosystems to croplands and to increase when restoring native vegetation on former croplands or by restoring organic soils to their native condition. Permanent and traditional woody cultivation such olive tree and grapevine may compensate this trend due to their positive contribution of their carbon uptake.

Human society will have to rely on an increase of output per unit area in agriculture and forestry. Intensively used agricultural systems are often N-saturated and the augmented use of fertilizer increases the leaching of N into aquifers and aquatic ecosystems and thus carries costs to environmental services such as water quality. In summer irrigated crops conditions are propitious for high N_2O losses. Emission factors for N_2O, distinguishing the effects of water management, crop type, and fertilizer management. Mediterranean agricultural soils produce large CH_4 emissions in flooded crops (e.g., rice) through methanogenesis, representing 6% of all CH_4 production from agricultural sources *(Section 3.2.3.2).*

During the past two decades, rural areas were reshaped by technological improvements in resources exploitation, the accelerating abandonment of traditional rural life and an increase in the mobility of individuals (Pinilla et al. 2008; Domon 2011). This pathway has led in many developed countries to a particular land-cover change pattern that consists in low plains and coastal areas that are being increasingly utilised for human activities due to their higher potential for agricultural productivity, while mountain or marginal areas are being abandoned because no more economically viable for production (Statuto et al. 2016; Nori 2018). Considering the increasing demand of food for an ever-growing population, leading to an increased productivity and intensification efforts in producing areas (Phelps and Kaplan 2017), this pattern has noticeable effects on the patchwork alternating semi-natural habitats that characterized the Mediterranean environment and the relevant natural ecosystem services. Management practices of grassland in in hilly Mediterranean areas were progressively reduced because of reduction of animal grazing and abandonment. This produced remarkable effects on floristic simplification, loss of biodiversity, reduction of habitat for wildlife or to the survivor itself of the resource due to shrubs encroaching (Papanastasis 2004; Argenti et al. 2011). Perennial cultivation, such as olive tree grove, originally planted in marginal areas on terraces and representing a quite stable ecosystem managed with few chemical inputs, were replaced by intensive modern plantations managed under an intensive and highly mechanized system. This intensification resulted into a progressive abandonment of marginal areas because of their low economic viability leading to a shift of land use to pasture for sheep and goats, since the land is not suitable for any other kind of cultivation (Loumou and Giourga 2003).

Conversely, the intensification process towards highly mechanized and high-density plantation is boosting farmers' income but it is also causing some environmental drawbacks. The combined effect of a more intensive management and cultivation extension is rising the issue of contamination by excess in the use of synthetic fertilizers and other agrochemicals to increase land productivity (Beaufoy and Pienkowski 2000; Beaufoy 2001). Intensification results into a degradation of habitats and landscapes and the exploitation of scarce water resources, thus putting the naturally scarce resources of olive growing areas to an edge. Drip irrigation, associated to an improved efficiency of irrigation, often has in fact no effect on efficiency but increases global water consumption. The use of this technique decreases the need for human power, allowing the increase of cultivation area and fostering multiple cropping (Kuper et al. 2017; Molle and Tanouti 2017). Soil erosion is also a growing issue due to the widening of cultivated area, as in tree crops a significant fraction of the soil is vulnerable to the action of rainfall and runoff. The intensification of agricultural land use has therefore raised the question of the long-term sustainability of agroecosystems (Liebig et al. 2004).

Different land uses as a single driver of change

Demography, technology, socio-economic factors and climate change have gradually transformed Mediterranean landscapes. Land uses are in fact the combination between the use of natural and food resources, constructions, road networks, etc., and land use changes result in the balance of their consequences. Natural habitats as affected by human activities are consequently reduced in size and continuity finally resulting into a loss of connectivity, i.e., the capability of the landscape to help or to prevent movements of organisms across habitat components (Taylor et al. 1993). A reduced or lack of connectivity may have consequences on biodiversity through losses of ecological fluxes between habitat patches and therefore trends in connectivity across different landscapes should be evaluated to consider proper actions to counteract potentially negative impact of human pressure on animal or vegetal biodiversity (Hernández et al. 2015). Over the Mediterranean Basin, two main trends, reforestation following the abandonment of agricultural areas in hilly and mountainous areas and the relevant expansion of agricultural and artificial areas in the coastal areas, plains and valleys, had different impacts on connectivity. In particular, reforestation after agricultural abandonment was correlated to a slight improvement in the connectivity of the European forest in the period 1990-2000 (Saura et al. 2011). At the same time, the spread of monoculture in the plain resulted in a de-fragmentation process. Both these trends produced a simplification and homogenization of the landscape, in terms of number, dimensions and typology of the patches that shape the agro-ecological territory.

2.4.1.3 Future projections

Land use change is expected to have different consequences on the productivity of several ecosystems and the carbon balance. The expected warmer and dryer conditions on southern Mediterranean Basin will likely shift crop cultivation to North, where water deficit is projected to be less harsh (Ceglar et al. 2019). Whilst the less adaptable crop systems will likely suffer changed pedological and climatic conditions, the most resilient crops, such as olive tree and grapevine may have the potential to resist to this trend due to their high adaptability to cope with high temperatures and water scarcity. Moriondo et al. (2013) and Hannah et al. (2013) predicted a gradual northward shift of this cultivation in the medium term (Tanasijevic et al. 2014). Increasing temperature joint with the expected reduce rainfall rate may lead detrimental consequences for those cultivations such as corn, rice and spring wheat requiring wetter conditions. Some crops may be replaced with more resilient crops such as barley, sorghum and hay, which, however, may not well fit with the market demand and the production chain of the Mediterranean area. For most of the main Mediterranean vegetable and cereals, a decrease in production is expected in the absence of specific high input agronomic strategies such as fertilization and irrigation (Bregaglio et al. 2017; Ruiz-Ramos et al. 2018; Brilli et al. 2019) whose requirement is expected to increase (Tanasijevic et al. 2014; Saadi et al. 2015).

Climate change impacts are expected to also affect managed forests, leading to shift in typical forest communities to higher altitudes (Gitay et al. 2001). These impacts will likely affect the whole woody spinneret, influencing timber extraction and plantations, and management practices. These latter practices (e.g., fuelwood collection, forest grazing, and road expansion) can degrade the forest ecosystem conditions, particularly when applied over new forest area. Accordingly, depredated soil and forestry systems may indirectly favor the introduction of pests and pathogens, changing fire-fuel loads, changing patterns and frequency of ignition sources, and changing local meteorological conditions (Nepstad et al. 1999). Grassland and pastures will likely experience a further decrease in extension due to a progressive rural abandonment and emigration to urban areas, often associated with low-income level in mountain areas and lack of job opportunity (Sturaro et al. 2013), and the impacts of climate change. This latter will particularly impact natural pastures, which are acknowledged to be very sensitive and vulnerable to climate conditions.

The predicted climate warming is also expected to lead changes of grasslands structure and composition, increasing the soil-water competition with trees that will be found to place at higher altitude as effect of higher temperatures. In temperate climate, warming may length the forage growing season but decrease forage quality, with important variations due to rainfall changes (Craine et al. 2010; Hatfield et al. 2011; Izaurralde et al. 2011), whilst Mediterranean pastures will likely show production decrease due to prolonged drought conditions. Modern agricultural practice can partly overcome expected production due to changed cultivation areas, but plant adaptation would require more time to be able to adapt to the new climate. Also, higher input needed to cope with changed agronomic conditions can lead to wrong perspective of the crop, forest and grassland production trend as well as extensive environmental damage. More specifically, a larger use of fertilizer or other high inputs may result in short-term increases in food production for long-term losses in ecosystem services, such as water quality degradation (Zalidis et al. 2002; Malagó et al. 2019), soil erosion, reduced fertility, or overgrazing (Wood et al. 2000). All these changes are expected to affect the carbon balance.

Land use is an important control of carbon storage; therefore, ecosystem shifts and harsher climatic conditions may lead to different forms of stress (i.e., water and nutrient stress, pedological stress, climatic stress, abiotic, etc.) which, in turn, reduce the potentiality of the different ecosystems in terms of carbon storage (i.e., biomass reduction, less growth and development). All these changes can have negative consequences in the perspective of CO_2 mitigation capacity (Foley et al. 2005), since is expected a decrease of total carbon sequestration capacity from agro and forestry systems, increased carbon fluxes from soils due to quicker decomposition process and lower carbon mineralized in soil.

2.4.2 Sea use changes

Fisheries (over) exploitation is the main driver of marine population decline and has led to the bad state of most highly commercial stocks and the low abundance of top predators. Climate change and variability may be responsible for catch fluctuations of some stocks (especially the small pelagic fishes), for distribution shifts but also for altering catch composition in favour of warm-water species. Recent theory predicts fish size decreases in response to increased sea surface temperature and low oxygen supply. Excessive exploitation will certainly lead to even lower stock biomasses, especially for top predators. Further sea warming will very likely lead to a higher percentage of warm-water species in the catch and smaller fish sizes.

2.4.2.1 Trends in fisheries exploitation

In the Mediterranean Sea, which together with the Black Sea, constitutes FAO Major Fishing Area 37, fishing has been practiced since antiquity. Today, Mediterranean fisheries are diverse among areas and the fishing vessels and techniques vary geographically as a result of different environmental, oceanographic, biological, climatic, cultural and socio-economic conditions prevailing in each area (Papaconstantinou and Farrugio 2000), with a strong contrast between the northern and southern coastlines. The high number of islands, ports and shelters across the Mediterranean and the contrast between north and south renders the enforcement of fisheries regulations and management very difficult. Mediterranean fisheries are highly multispecies in nature targeting over 200 fish and invertebrate species (Dimarchopoulou et al. 2017) and are operated through a large number of small sized and low tonnage fishing vessels with no large industrial fleets (Stergiou et al. 2016). The number of small-scale coastal vessels operating in Mediterranean EU waters is about 86% of the total (around 72600 vessels) with the remaining 9% being trawlers and 5% being purse seiners (Colloca et al. 2017). Although the number of all types of Mediterranean EU fishing vessels declined since 1991, the actual fishing effort has been increasing due to new technologies and higher capacity vessels (Colloca et al. 2017).

Small pelagic fisheries operate all year round but in many Mediterranean Sea subareas they show a strong seasonality that is reflected upon their catches and is derived from fishing regulations and consumer habits. According to the monthly distribution of landings and fishing effort, the main fishing season in most areas is concentrated in spring and summer months (Lloret et al. 2004b).

2.4.2.2 Current status of marine fisheries resources

In the Mediterranean Sea, recent publications based on scientific surveys, stock assessments and catch data, generally agree that the majority of Mediterranean fisheries stocks are declining in biomass as a result of their overexploitation (Colloca et al. 2013, 2017; Vasilakopoulos et al. 2014; Tsikliras et al. 2015). Local reports also confirm the bad status of Mediterranean fisheries, e.g., in Greek seas (Tsikliras et al. 2013b) and in the Ligurian Sea (Abella et al. 2010), often attributed to inadequate management practices (Tsikliras 2014; Cardinale et al. 2017). The long-lasting overexploitation of the Mediterranean Sea has been driving the decline in biomass of most commercial fish and invertebrate stocks across the basin and the near depletion of several of them (Froese and Kesner-Reyes 2002; Vasilakopoulos et al. 2014; Osio et al. 2015; Tsikliras et al. 2015; Stergiou et al. 2016; Colloca et al. 2017; Froese et al. 2018). The overall stock status is rather uniform across the Mediterranean ecoregions with low stock biomass being the common characteristic. However, the stock specific biomass levels vary among ecoregions (Froese et al. 2018).

The catch history of Mediterranean Sea stocks unmasked the overexploitation of many stocks since the 1950s, when about 40% of them were declining in biomass (Froese and Kesner-Reyes 2002). Recent literature reveals that fisheries overexploitation occurs across the entire area (Tsikliras et al. 2013a) and locally, e.g., in Greek Seas (Tsikliras et al. 2013b). Several Mediterranean stocks have been reported overfished based on data from landings (Tsikliras et al. 2013b, 2013a), scientific surveys (Stergiou and Tsikliras 2011), or stock assessments (Colloca et al. 2013). Other studies confirm that almost all species targeted by the fishing fleets are being overexploited in the Mediterranean Sea (Cardinale et al. 2017; Fernandes et al. 2017).

Based on the catch-based method, the cumulative percentage of collapsed and overexploited stocks appeared to exceed 60% across the Mediterranean Sea in 2010 (Tsikliras et al. 2013a) with the exploitation pattern differing among the Mediterranean subareas (Tsikliras et al. 2015). The western Mediterranean has been reported to be in a better state with less overexploited and collapsed stocks and more developing ones compared to the central and eastern parts of the sea (Stergiou et al. 2016). Similarly, based on various fisheries indicators, the western and central Mediterranean are in better condition compared to the eastern part of the sea (Tsikliras et al. 2015). According to the official stocks assessments that were then available, the percentage of overexploited stocks exceeds 90% in most areas (Colloca et al. 2013) and even reaching 95% in some (Osio et al. 2015). The model approach of Osio et al. (2015) estimates that 98% of the unassessed demersal fish species are potentially overexploited in most areas. Cardinale et al. (2017) reported that the stocks of all target species that have been assessed are overexploited with the average ratio of F/F_{MSY} (actual fishing mortality to the level that would provide maximum sustainable yield) ranging from 1.7 (giant red shrimp *Aristaeomorpha foliacea*) to 8.1 (hake *Merluccius merluccius*). Steadily increasing exploitation rates and deteriorating gear selectivity have been recently reported as two conditions that lead to shrinking fish stocks (Vasilakopoulos et al. 2014). The most recent assessment of 169 Mediterranean stocks showed that 126 of them (75%) were subject to ongoing overfishing (Froese et al. 2018).

2.4.2.3 The future of marine resources

The Gill-Oxygen Limitation Theory (GOLT) predicts a reduction in the size of fish due to their inability to compensate, via their gill surface, for the increased metabolic rate that results from higher temperatures. Fish individuals that survive are expected to shrink in size (Cheung et al. 2013a). The Mediterranean Sea is among the semi-enclosed areas where local species extinctions and range shifts were predicted to be most common (Cheung et al. 2009). The GOLT theory may also explain the poleward shift of marine organisms (Cheung et al. 2013b) and their expansion to deeper waters (Perry et al. 2005) both of which occur in the Mediterranean Sea (Tsikliras and Stergiou 2014) and may have an impact on Mediterranean fisheries in terms of catch and revenue (Cheung et al. 2010).

Besides fish distribution shifts and declines in local fish stocks, scientific projections suggest that marine resources and biodiversity will suffer increasing stress if temperatures are not held below 2°C above preindustrial levels (Gattuso et al. 2015). Sea warming and deoxygenation combined with fishing pressure and other stresses could affect growth, and distribution of fish populations, resulting in changes in the potential yield of exploited marine species and economic losses (Sumaila et al. 2011) as fisheries are expected to decline (Cheung et al. 2010). Reaching the goals of the UNFCCC Paris Agreement would benefit ocean life and economies by protecting millions of metric tons of high valued catch with 75% of maritime countries benefiting from this protection (Sumaila et al. 2019).

Fisheries in the Mediterranean will not be sustainable in the future unless the marine exploited populations are fished less, i.e., if they are allowed to recover and rebuild their biomass (Pauly and Zeller 2016) through the reduction of the fishing pressure that is applied upon them (Froese et al. 2018). Ecosystem-based approach has also an important role that will ensure that both higher and the lower trophic levels are rebuilding (Pikitch et al. 2004) and fully marine protected areas are a key management tool to accomplish rebuilding of the biomass of marine populations, ensure ecosystem health and resilience against sea warming (Roberts et al. 2017).

2.5 Non-indigenous species

The human-aided introduction of non-indigenous species into new biogeographic regions – has been one of the main increasing global drivers of ecological change for over a millennium (Elton 1958). Non-indigenous species homogenize biodiversity across the globe, resulting in shifted and sometimes more simple ecological communities (Mooney and Hobbs 2000; Rilov and Crooks 2009). They can displace native species out of their natural habitats through competition, consumption, or parasitism. This displacement results in affected communities with reduced native species diversity, altered species composition as well as in major changes in ecosystem functioning (Vilà et al. 2011; Cameron et al. 2016). Moreover, these changes alter supporting, provisioning, regulating ecosystem services that people depend upon, impacting human well-being (Katsanevakis et al. 2014b; Vilà and Hulme 2017).

Rates of introduction and impacts have accelerated dramatically in the past few decades due to various human activities and related pathways of non-indigenous species introduction (Carlton 1989; Crooks and Suarez 2006). In the marine environment, the growth of seaborne trade with its huge fleets facilitates the dispersal of organisms attached to the hulls of ships and inside ballast water (Crooks and Suarez 2006; Katsanevakis et al. 2013). Aquaculture, live marine seafood and bait, and aquarium trade have also become important vectors for the introduction of marine non-indigenous species, as well as the artificial connections of water bodies with very different biotas (Rilov and Crooks 2009), and major catastrophes like tsunamis (Carlton et al. 2017). In terrestrial ecosystems, intentional introductions prevail over unintentional. For non-indigenous plants, ornamental and horticultural introductions escaped from cultivation account for the highest number (Lambdon et al. 2008) and are increasing steadily (Van Kleunen et al. 2018). Terrestrial non-indigenous vertebrates follow similar patterns as plants. However, most terrestrial non-indigenous invertebrates have been introduced accidentally; many are major pests in forestry and agriculture (Roques et al. 2010).

2.5.1 Non-indigenous species in the Mediterranean Sea

The first introduction of marine species into the Mediterranean Sea dates back to the late 18th century (Poli 1791). Since then, due to maritime shipping expansion and after three centuries of accumulating human pressures, this basin has become a hotspot of introduction of non-indigenous species (Rilov and Galil 2009; Coll et al. 2010). Non-indigenous species in the Mediterranean Sea mostly arrive from the Indo-Pacific region either directly (by swimming or drifting) or indirectly as foulers or as hitchhikers inside ballast water in a process called Lessepsian migration (Por 1978). But there are many other vectors that deliver non-indigenous species into the Mediterranean and have varying importance depending on the region (Rilov and Galil 2009). Today, the total

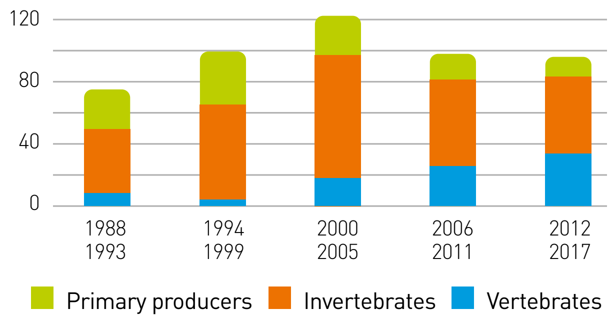


Figure 2.25 | Number of new non-indigenous species per 6 years in the Mediterranean since 1988 (Zenetos 2019).

known number of non-indigenous species is still debated, but the last count puts it close to a thousand species (Zenetos et al. 2017). For some of the species, the impact on local biodiversity, and possibly also ecosystem functions and services, seems obvious, but in most cases, the impact is unknown because of lack of research (Katsanevakis et al. 2014b). The Mediterranean Sea is warming rapidly (Nykjaer 2009; Sisma-Ventura et al. 2014) *(Section 2.2.4)*. Therefore, it is quite possible that the establishment of thermophilic non-indigenous species is strongly facilitated by climate change (Stachowicz et al. 2002).

2.5.1.1 Spatiotemporal trends, sources and vectors of introduction

The most recent assessment of marine non-indigenous species in the Mediterranean (November 2018) counts 957 species, including Foraminifera (Zenetos 2019). Among the introduced species during the last 30 years (491 taxa), invertebrates dominate with >58% (287 species) represented mostly by molluscs and decapods. Primary producers follow with approximately 114 species among which macroalgae, especially rhodophytes, prevail. Vertebrates (mostly fishes) follow with 90 species.

The trend in introduction of species in the Mediterranean *(Fig. 2.25)*, which culminated in the 2000-2005 period with more than 20 new species per year (122 in total), appears to be overall decreasing after 2005 *(Fig. 2.26)*. As opposed to invertebrates and primary producers, vertebrates continue increasing, with 34 species detected in the 2012-2017 period vs 26 species in the period 2006-2011. The overall decreasing rate in non-indigenous species is evident across the Mediterranean Marine Strategy Framework Directive (MSFD) areas, except for the Central Mediterranean where an increase is observed, attributed to vertebrates *(Fig. 2.26)*. 17 new fish species were detected in the central Mediterranean in the period 2012-2017 vs. 8 fish species in the period 2006-2011. These species are either spreading from the eastern Mediterranean to the central region or are newly introduced species in the area. Vertebrates (fish only) are dominated by Lessepsian immigrants but over the last decade the number of fish species related to aquarium trade which have been intentionally released to the wild (classified as escapees from confinement) is increasing (Zenetos et al. 2016; Marcelli et al. 2017; Deidun 2018).

Regarding the spatial distribution of non-indigenous species, the number of Lessepsian species is very high on the eastern Mediterranean coastline, reaching 129 species per 100 km², and declines toward the north and west (Katsanevakis et al. 2014a). The distribution of species introduced by shipping is strikingly different, with several hotspot areas occurring throughout the Mediterranean Basin. Two main hotspots for aquaculture-introduced species have been identified (the Thau and Venice lagoons). Certain taxonomic groups were mostly introduced through specific pathways-fish through sea corridors, macrophytes by aquaculture, and invertebrates through sea corridors and by shipping (Katsanevakis et al. 2014a). Hence, the local taxonomic identity of the non-indigenous species is greatly dependent on the dominant maritime activities/interventions and the related pathways of introduction. The composition of non-indigenous species assemblages differs among Mediterranean ecoregions; such differences are greater for Lessepsian and aquaculture-introduced species.

2.5.1.2 Non-indigenous species as drivers of biodiversity and ecosystem change

The introduction of non-indigenous species in the Mediterranean Sea have caused modifications in biodiversity patterns. One of the best documented and most profound impacts of the introduction of non-indigenous species on native Mediterranean ecosystems is the deforestation of algal forests and the creation of extent barrens (i.e., areas with bare rock and encrusting calcified algae; *Fig. 2.27*) by the overgrazing activity of two non-indigenous herbivore rabbitfishes: *Siganus luridus* and *S. rivulatus*. These species have become dominant in the ichthyofauna of shallow rocky habitats in the eastern Mediterranean, and have caused ecosystem-wide changes by creating and maintaining areas denuded of canopy algae (an important habitat for many coastal fishes) (Cheminée et al. 2013). This form of "deforestation" is associated with a dramatic reduction in biodiversity, biomass, and algal growth, and effects that move up the food chain to the local fisheries (Sala et al. 2011; Vergés et al. 2014b).

In the Levantine Sea, the catch of commercial fisheries is now dominated by non-indigenous species (Edelist et al. 2013; Katsanevakis et al. 2018), reflecting the decline of native biota and its replacement by thermophilic non-indigenous species (Arndt et al. 2018). In Turkish coastal waters in the Levantine Sea, non-indigenous

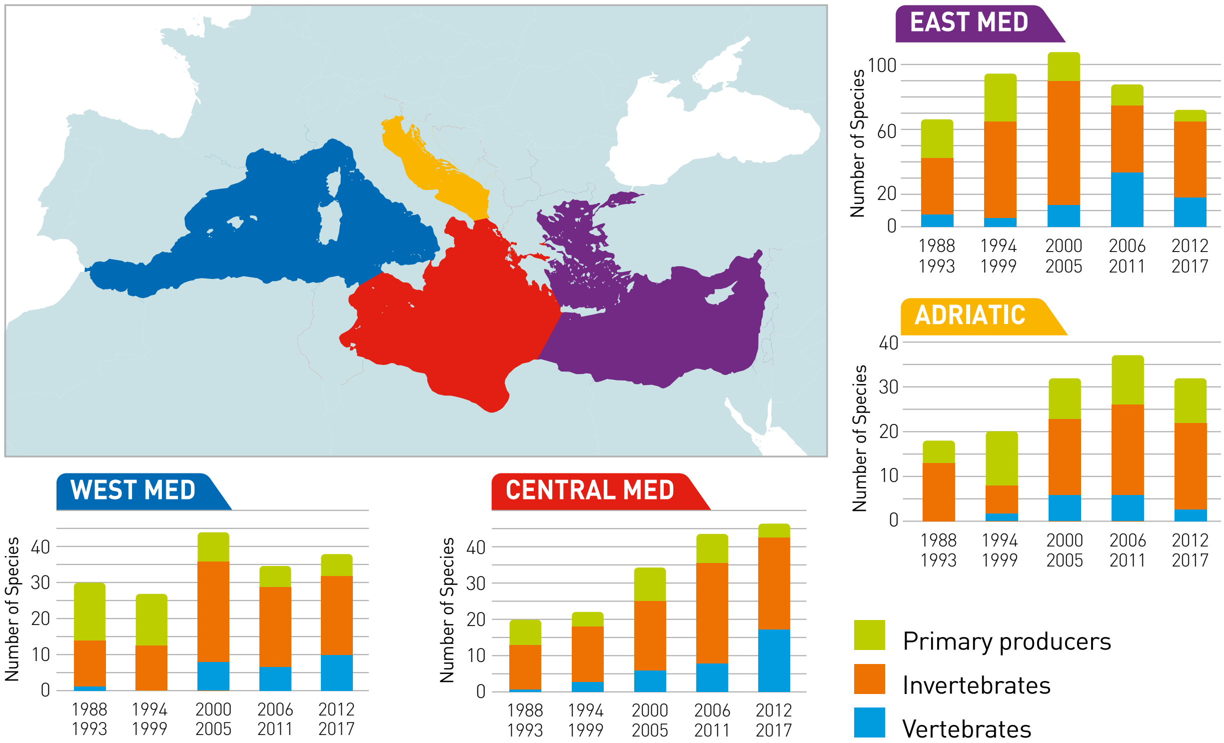


Figure 2.26 | Change in the number of new non-indigenous species per 6 years in the Mediterranean MSFD regions (Adriatic, Western Mediterranean, Central Mediterranean, Eastern Mediterranean Seas) since 1988. Notice that the scale is different for the eastern Mediterranean (Zenetos 2019).

fish species biomass exceeds 50% of the total fish biomass and 90% of the total herbivore fish biomass (Giakoumi et al. 2019). Fishers in the same area (Kaş, Turkey) perceived the introduction of non-indigenous species as the most important reason for the current fish stock depletion.

Yet, it is uncertain whether the decline of native biota in the Levantine Sea is driven mainly by biotic interactions with non-indigenous species or climate change, or both, as the Lessepsian species are thermophilic and their establishment is assisted by climate change (see discussion below). A recent meta-analysis suggests that the decline of native fish can be mostly attributed to ocean warming and not to negative interactions with non-indigenous species (Givan et al. 2017a). Multi-species collapses of native species are also at least partially attributed to climate change (Rilov 2016), with the collapse of the sea urchin *Paracentrotus lividus* experimentally demonstrated to be related to the fast ocean warming (Yeruham et al. 2015).

Analysis focused on shallow reef fish shows that non-indigenous species are very diverse ecologically, and they considerably increase the total community trait diversity of the Mediterranean (Givan et al. 2017b). Furthermore, trait similarity between non-indigenous and indigenous Mediterranean species was lower than expected, indicating that non-indigenous fish tend to occupy relatively vacant niches within the Mediterranean. Temporally, non-indigenous fish species display increased trait similarity to native Mediterranean species, suggesting that forecasting future establishment may be challenging. Givan et al. (2017b) conclude that the Mediterranean, at least in fish, is transforming into an extension of the Red Sea in terms of trait and species composition. Such biological trait analysis is also required for other taxonomic groups.

There is a serious research gap in assessing and quantifying the impacts of non-indigenous species on marine ecosystems in the Mediterranean Sea. Impact assessment is mostly based on expert judgment or correlational studies, while manipulative or natural experiments are largely lacking for assessing the impacts of most non-indigenous species in the region (Katsanevakis et al. 2014b). Disentangling the role of the introduction of non-indigenous species and climate change or other local or global stressors to derive cause-

Figure 2.27 | Ecosystem shift from algal forests to barrens due to the overgrazing activity of non-indigenous herbivore rabbitfish. Algal forests host high fish, invertebrate, and algae biodiversity whereas barrens are associated with low levels of biodiversity across all taxonomic groups. *Rabbit fish photo: Murat Draman.*

effect pathways is inherently difficult and would probably necessitate a combination of experimental and modelling approaches.

Efforts are made to assess non-indigenous species impacts with existing knowledge. Based on a conservative additive model, which downgrades reported impacts of low inferential strength, an index of the Cumulative IMPacts of invasive ALien species (CIMPAL) on marine habitats in the Mediterranean has been developed and estimated (Katsanevakis et al. 2016). The estimation of CIMPAL was based on assessments of impacts for every combination of 60 non-indigenous species and 13 habitats, and their distributions in the Mediterranean (i.e., presence in 10x10 km cells). It showed strong spatial heterogeneity in impacts. Spatial patterns varied depending on the pathway of introduction of the non-indigenous species in the Mediterranean Sea. Species introduced by shipping gave the highest impact scores and impacted a much larger area than those introduced by aquaculture and through sea corridors. Overall, non-indigenous macroalgae had the highest impact among all taxonomic groups, when estimated as the sum of impact scores across the entire Mediterranean Sea, i.e., accounting not only for the severity of the impacts but also their spatial extent. The most impactful non-indigenous species was *Caulerpa cylindracea*, which has become dominant over large areas of shallow waters in the western Mediterranean and compete with native species (Piazzi et al. 2016). Negative impacts of *C. cylindracea* have been documented on algal, sea grass, and sponge communities (Ceccherelli et al. 2002; Piazzi et al. 2005; Piazzi and Ceccherelli 2006; Baldacconi and Corriero 2009).

2.5.1.3 Further introductions, monitoring and managing non-indigenous species

The introduction of species in the Mediterranean Sea is a continuous process and it is very likely that it will continue for years to come. In most of the cases, these species fail to thrive but, evidently, some become numerically and ecologically dominant in their new environment, generating new and sometimes severe impacts of the introductions. As a consequence, in the past two decades research interests in non-indigenous species have increased, mostly stimulated by evidence on 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).

One of the most recent and potentially damaging non-indigenous species for this basin is the common lionfish (*Pterois miles*), which increasingly appears in many parts of the eastern Mediterranean in the last few years (Bariche et al. 2013). Due to its rapid increase in abundance (Kletou et al. 2016) and fast geographical expansion (Azzurro et al. 2017), this harmful species has become emblematic for raising concern on Mediterranean non-indigenous species introductions but it also well illustrates a process of developing monitoring capabilities and management strategies within the Mediterranean region.

Documenting the spread of this non-indigenous species can greatly benefit by the participation of resource users, a partnership which can support monitoring objectives (Azzurro and Bariche 2017) and be used to reduce, at least locally, the abundance of non-indigenous species (Kleitou et al. 2019). More generally, participatory approaches benefit the scientific consensus, which is a key element for both documenting and responding to these introductions (Scyphers et al. 2015). This is also reflected in some of the guiding principles on the management of non-indigenous species adopted by key regional and international bodies/legislative frameworks, concerning non-indigenous species, such as those provided by the EU (Regulation 1143/2014); UNEP-MAP and by FAO-GFCM, which are converging towards finding common strategies to face Mediterranean species introductions. In this regard, the Integrated Monitoring and Assessment Programme and related Assessment Criteria (IMAP) adopted through Decision IG.22/7 by the 19th Ordinary Meeting (COP 19, Athens, Greece, 9-12 February 2016) of the Contracting Parties to the Barcelona Convention, stress the need of comprehensive monitoring and coordinated transnational actions to face the common issue of Mediterranean non-indigenous species introductions. A recent expert assessment of management options of marine non-indigenous species prioritized 11 management actions for controlling 12 model species according to their dispersion capacity, distribution, and taxonomic identity (Giakoumi et al. 2019). The actions were assessed using five criteria (effectiveness, feasibility, acceptability, impacts on native communities, and cost), combined in an "applicability" metric. Raising public awareness and encouraging the commercial use of non-indigenous species gained the highest priority, and biological control was considered the least applicable (Giakoumi et al. 2019).

To predict future change in the distribution of native species and the spread of non-indigenous species, species distribution models (SDM) are regularly used. In these models, climate matching is calculated between the area of origin (donor) and the area of potential spread (recipient). However, a recent study that matched the native range of Red Sea fish and their new range in the Mediterranean Sea showed poor matching, and thus indicated that SDMs may underestimate the potential spread of non-indigenous species (Parravicini et al. 2015). The authors call for caution in employing such models for forecasting the introduction of non-indigenous species and their response to environmental change, as uncertainty is large. Better knowledge of the fundamental niche of species (their physiological performance under different environmental conditions) can potentially improve the prediction of spread of non-indigenous species in the Mediterranean. Furthermore, an analysis combining ship movements with port environmental conditions and biogeography can be used to quantify the probability of new primary introductions through ballast water (Seebens et al. 2013).

2.5.2 Terrestrial non-indigenous species and pests

2.5.2.1 Spatial patterns and temporal trends

Degree of introduction across Mediterranean-type ecosystems and geographical areas

The information available on terrestrial non-native species in the Mediterranean Basin countries is not comprehensive, and the number of non-indigenous species is underestimated due to the incompleteness of collected data and the monitoring bias towards some taxonomic groups. For example, Abellán et al. (2016) reported more than 370 non-indigenous birds for Spain and Portugal, more than twice the number listed by DAISIE (2009) for the whole Europe.

Most non-indigenous species in the Mediterranean Basin are plants, followed by invertebrates (Fig. 2.28). Natural habitats in the Mediterranean Basin host more than 400 non-indigenous plant species (Arianoutsou et al. 2013). The taxonomic similarity of the non-native flora among Mediterranean Basin countries is very low. For example, less than 30 species are common across 4 Mediterranean European countries (Arianoutsou et al. 2010), and only 10 species are common in 8 major Mediterranean islands (Lloret et al. 2004a). However, non-native plants share similar traits, as shown by the high proportion of perennial herbs with a long flowering period, which are pollinated and dispersed by the wind (Lloret et al. 2004a, 2005).

The major part of non-indigenous invertebrate species are arthropods, especially insects (Roques 2010), with a low representation of nematodes and flatworms (Naves et al. 2016; Justine et al. 2018). Phytophagous pest species are largely dominating among non-indigenous species all

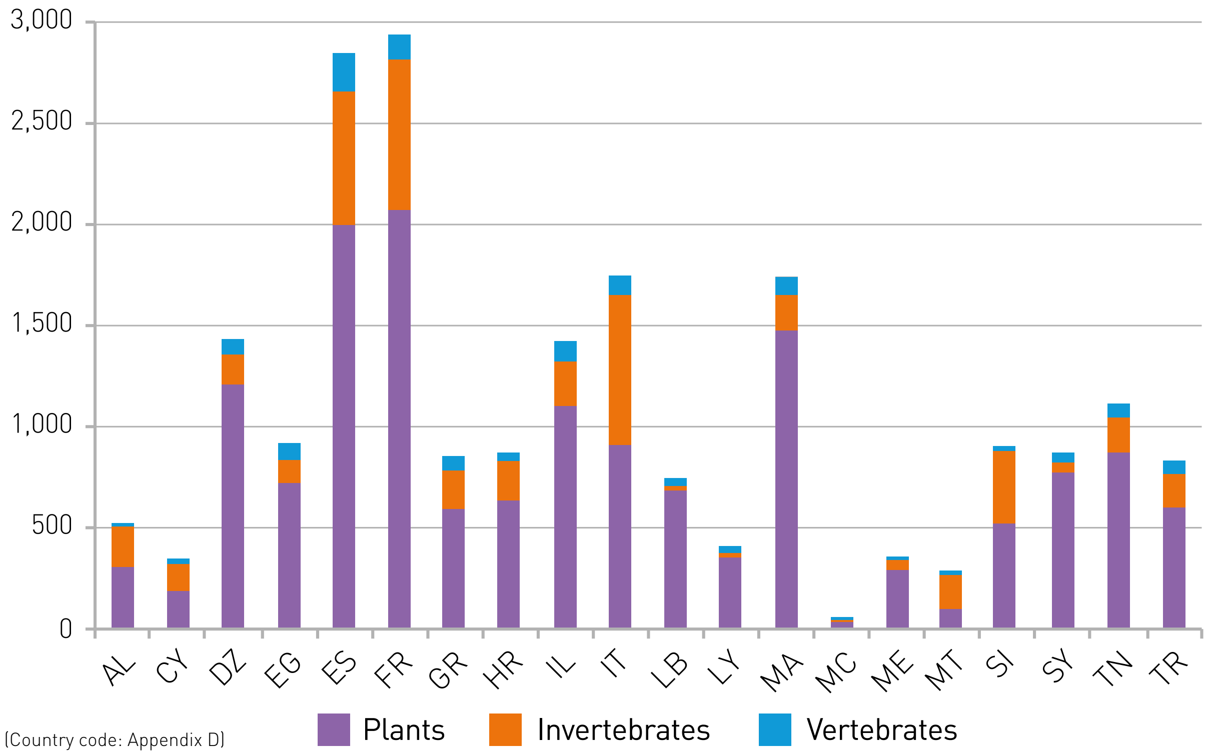


Figure 2.28 | Number and proportion of terrestrial non-indigenous plant, invertebrate and vertebrate species per country (EASIN)[8].

over the Mediterranean Basin, accounting for more than a half of the invertebrate species and about one-third are associated with woody plants (Matošević and Pajač Živković 2013; Roques 2015; Avtzis et al. 2017). Among them, Hemipterans, mostly scales and aphids, constitute the dominant group, accounting for 40-75% of the non-indigenous species in any part of the Mediterranean Basin (Roll et al. 2007; Inghilesi et al. 2013; Matošević and Pajač Živković 2013; Seljak 2013; Avtzis et al. 2017). This over-representation of Hemiptera in non-indigenous species seems related to their small size and easier of transportation with infested imported plants.

A complete plant pathogens database is not available. However, a European database of non-indigenous forest and woody pathogens indicate a list of 123 plant pathogens. In Mediterranean countries, Ascomycota is the most numerous group, while Oomycota and Basidiomycota represent 21% and 9% of the total, respectively (Santini et al. 2013). With regard to vertebrates, most non-indigenous species are birds, followed by mammals, while the number of reptiles and amphibians is low.

The impact of non-indigenous species varies largely among countries. Countries with higher Human Developmental Indexes and imports host a large density of non-native plants (Vilà and Pujadas 2001). Between countries and within countries, the density of non-native plants is related to the length of terrestrial transport networks. Areas with extensive road and rail networks, high anthropogenic disturbance, low altitude, short distance to the coastline and dry, hot climate show higher richness of non-indigenous plant species (Gassó et al. 2012). Most affected landscapes are those highly urbanized and with high population densities (Sobrino et al. 2002; González-Moreno et al. 2013). Accordingly, the most affected ecosystems by non-indigenous plants are human-modified such as ruderal, waysides or agricultural fields (Vilà et al. 2007; Hulme et al. 2008; Arianoutsou et al. 2010).

As for other non-indigenous taxa, most pests are introduced in managed habitats, such as agricultural lands and parks and gardens, forests being less affected (Matošević and Pajač Živković 2013). Among the infested plant species, non-native ornamental plants (palms, legume trees),

[8] https://easin.jrc.ec.europa.eu

Citrus and *Eucalyptus* are slightly more colonized than native species (Roques 2015). Vertebrates also tend to occupy anthropic habitats and, to a lower extent, woodlands (at least for birds) (Kark et al. 2009). The degree of introduction of non-indigenous invertebrate species also varies among countries, Italy and France showing much more established non-indigenous species than any other European country (Roques 2010). The same trend is reported also for plant pathogens (Santini et al. 2013).

Mediterranean islands and islets host a large number of non-indigenous species, mainly plants (Brundu 2013). For example, an analysis of 37 small Italian islands showed that they are affected by 203 non-native plants, with a remarkable increase of acacias and succulents in the last decades (Celesti-Grapow et al. 2016). The main determinants of non-indigenous plant species richness in small islands are tourist development and the percentage of artificial land-cover. However, at the local scale, para-oceanic island ecosystems such as the Balearic Islands have a relatively lower number of non-native plants than their mainland counterparts (Vilà et al. 2010). Yet, some species can be introduced into more ecosystem types in islands than in the mainland (Gimeno et al. 2006). In general, there are large differences in the taxonomic composition of non-indigenous insect assemblages between islands and continental countries, e.g., France and Corsica, Italy and Sicily (Liebhold et al. 2016), and Greece and Crete (Avtzis et al. 2017).

Temporal trends of non-indigenous species and pests

The rate at which humans have moved species beyond their native ranges has tremendously increased over the last 150–200 years (di Castri 1991; Reichard and Hamilton 1997), and more so in the last decades (Genovesi et al. 2009; Cardador et al. 2019). Although all taxonomic groups have shown a general rise during this period consistent with the exponential increase in trade and travel (Jeanmonod et al. 2011; Seebens et al. 2017; Cardador et al. 2019), little is known about how temporal dynamics of non-indigenous species varies among taxa. Where time series are available, the number of non-indigenous species established in Europe has increased exponentially in terrestrial ecosystems (Jeschke and Strayer 2005; Hulme et al. 2008; Lambdon et al. 2008; Santini et al. 2013). Abellán et al. (2016) analyzed data on bird introductions in Spain and Portugal since 1912 and found that most of them (99.9%) were recorded from 1955 onwards, with a sharp increase after the 1980s that mirrors the number of non-native birds imported into these countries. Cage birds (mainly Passeriformes and Psittaciformes) constitute the bulk of the species introduced during the last 40 years through escapes of individuals kept in captivity as pets. Although the information is less detailed, and reptiles and amphibians have smaller numbers of recorded non-indigenous species than birds, both groups have also increased their numbers during the 20th century in parallel with the rise in human immigration into Europe (Jeschke and Strayer 2006) and the international trade (Jenkins 1999).

The rate of establishment of non-indigenous insect species has also increased during the last decades (Roques 2010, 2015; Matošević and Pajač Živković 2013; Avtzis et al. 2017). A fast and quite linear increase, with about 10 new species per year, was noticed in Italy since World War II (Inghilesi et al. 2013), and an even higher rate of increase was noted in Croatia since 2007 (Matošević and Pajač Živković 2013). The species newly established during the last three decades tend to spread all over the Mediterranean Basin significantly faster than those that arrived between 1900-1990s (Roques et al. 2016). Such a rapid spread was especially impressive in some species, often relying on multiple introductions in different countries being used as bridgeheads (Rugman-Jones et al. 2013; Kerdelhué et al. 2014; Garnas et al. 2016; Roques et al. 2016; Bras et al. 2019; Lesieur et al. 2019).

Non-indigenous plant pathogenic species have increased exponentially in the last four decades (Santini et al. 2013). Since then, new non-indigenous plant pathogenic species have been introduced mainly from North America, and recently from Asia. Hybrid pathogens also appeared. Countries with a wider range of environments, higher human disturbances or international trade host more non-indigenous species. Rainfall influences the diffusion rates. Environmental conditions of the new and original ranges and systematic and ecological attributes affect pathogen success (Santini et al. 2013).

For plants, the success of introduction in terms of their area of occupancy is larger in species introduced a few centuries ago than species introduced in the 20th century (Lambdon and Hulme 2006), while for birds, establishment success is positively related to time since first introduction (Abellán et al. 2017).

Pathways of introduction (intentional and accidental) of non-indigenous species and pests

The majority of non-indigenous plants have been introduced into the Mediterranean Basin intentionally, as ornamentals that have escaped from gardens associated with anthropic developments and housing (e.g., touristic urbanizations) but also to embellish infrastructures (Hulme et al. 2008). Furthermore, many non-native trees (e.g., *Acacia, Pinus, Eucalyptus*) have been planted at large scales as forestry species and also in restoration programs for dune-stabilization, riverine water flux control, soil fertilization or afforestation of agricultural abandoned land. Many plant species have also been introduced unintentionally (accidentally) as "hitchhikers" or seed contaminants.

The main pathway of introduction for vertebrates, for example birds, are accidental escapes from private collections (Abellán et al. 2016). International wildlife trade is one of the main (if not the main) sources of current vertebrate non-indigenous species. When the EU banned the imports of wild birds, there was a rapid trade shift from wild-caught birds to captive-bred birds (which have lower potential to establish populations than wild-caught birds) (Carrete and Tella 2008, 2015; Cabezas et al. 2013) and a sharp decrease in the number of new introduced avian species in the wild (Cardador et al. 2019). However, this positive effect of the EU ban on wild-caught birds coincides with a significant increase in the trade in reptiles (Cardador et al. 2019).

For invertebrates, the vast majority of species introductions have been accidental (Hulme 2009; Roques 2015). A few introductions have been intentional, mostly for biological control between 1950 and 1999 (Rasplus et al. 2010), but such species always represent less than 15% of the total number of non-indigenous species per Mediterranean country, except in Israel (17.4%) (Roll et al. 2007). Since the majority of invertebrates established in the Mediterranean Basin are phytophagous, the major pathway of unintentional introductions appears to be via international trade in live plants (Rabitsch 2010; Inghilesi et al. 2013; Eschen et al. 2015; Roques 2015). Seed trade has also provided a few species and pests (Auger-Rozenberg and Boivin 2016) as well as firewood, logs and fallen timber (Meurisse et al. 2019). The trade of vegetable and fruit commodities also constitute an important pathway for non-indigenous pests (Desneux et al. 2010; Abbes et al. 2012; Cini et al. 2014). Hitchhiking is another significant pathway of pests as stowaways using wood packaging material (Rassati et al. 2015; Javal et al. 2019; Lesieur et al. 2019), transport infrastructure and vehicles (Javal et al. 2019; Kirichenko et al. 2019) or used tires such as for mosquitoes (Rabitsch 2010). Several other examples are associated with beekeeping (Mutinelli et al. 2014).

For plant pathogens, all the introductions occurred unintentionally. The exact pathway of introduction is almost unknown for most species. However, the most probable is the trade of living plants (57%) or wood (10%). Less than 10% of the introductions occurred through any of the other pathways (Santini et al. 2013). Introductions of even harmless fungi in a new environment give them the opportunity of mating with local or introduced related species giving rise to hybrid progenies. The hybridization process may result in an increase of pathogenicity in one of the species or in the emergence of a completely new plant disease, both of which may threaten the original host plant, and new and naïve host species (Ghelardini et al. 2017).

2.5.2.2 Non-indigenous species as drivers of biodiversity and ecosystem change

The introduction of non-native plants can decrease local flora and fauna diversity and change the community composition and functional structure of affected ecosystems (Vilà et al. 2006; Zahn et al. 2009; Rascher et al. 2011). Native plants that are most vulnerable to such introductions are those with small population sizes (Lapiedra et al. 2015). At least 12 endemic or critically endangered plant species from the "Top 50 Mediterranean Island Plant" list are threatened by non-native plants (de Montmollin and Strahm 2005). For birds, the impact of non-indigenous species is higher on island species, and in those species with small distribution ranges (Clavero et al. 2009). Destabilized ecosystems, including systems used for food and agricultural production, tend to be more vulnerable to the spread of non-indigenous species (e.g., Marvier et al. 2004; Chytrý et al. 2008). However, there is little evidence to support the hypothesis that highly diverse ecosystems are inherently more resistant to non-indigenous species than less-diverse systems (e.g., Keller et al. 2011).

Changes in ecosystem functioning after the introduction of non-indigenous plants are highly context-dependent and include alterations in decomposition rates, light and water soil availability, and changes in soil carbon and nitrogen pools (Vilà et al. 2006; Castro-Díez et al. 2009; Rascher et al.

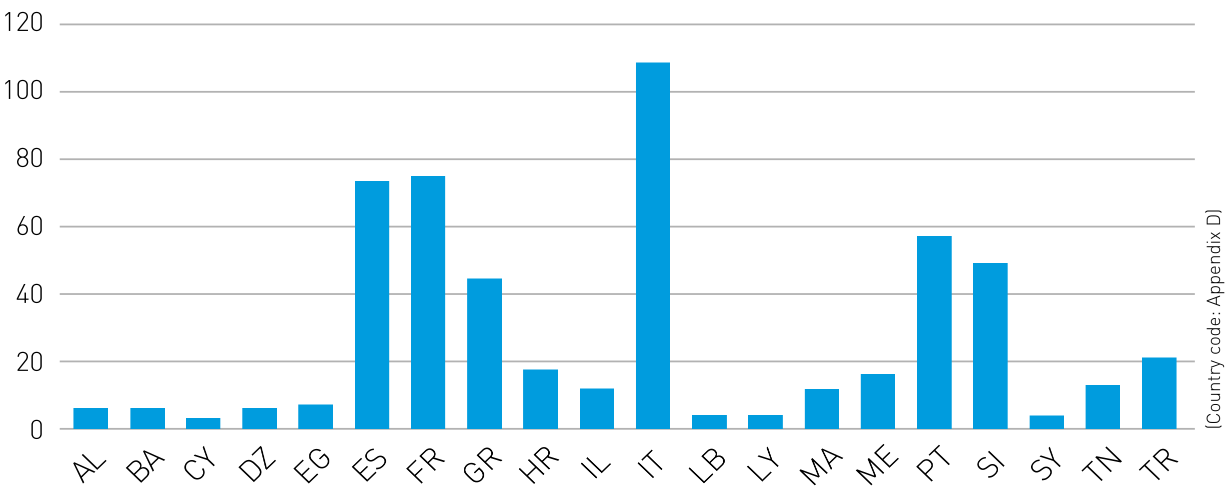


Figure 2.29 | Number of non-indigenous woody plant pathogens per country (Santini et al. 2013).

2011). Plant introductions can disrupt the positive relationship between native species diversity and multifunctionality (Constán-Nava et al. 2015) and may trigger regime shifts by changing plant succession (Stinca et al. 2015).

The impacts of vertebrates can be dramatic through competition for resources, predation and as vectors of diseases (Genovesi et al. 2009). For example, the introduction of non-native ungulates is a major threat to endangered plants, especially in islands (Pisanu et al. 2012). Some native amphibians have collapsed after the introduction of non-native anurans (Lillo et al. 2011). Non-indigenous parakeets can also interact with native species and have an impact on native populations and communities, largely in the form of harassment, displacement from nest sites and food competition (Hernández-Brito et al. 2014, 2018; Menchetti and Mori 2014; Menchetti et al. 2016; Covas et al. 2017). NIS arthropods can also negatively impact native biodiversity and ecosystem processes by destroying host plant populations, causing disturbances in native genetic resources or, indirectly, affecting the affected communities because of new species assemblages (Kenis et al. 2009; Kenis and Branco 2010; Auger-Rozenberg and Boivin 2016).

Some non-indigenous species can damage productive sectors, most notably agriculture and forestry, with important economic consequences. Non-native insects impact major crops in many countries of the Mediterranean region (Abbes et al. 2012; Abdallah et al. 2012; Mutke et al. 2016). Some non-indigenous bird species such as Monk and ring-necked parakeets can also cause important damages to crops (Senar et al. 2016; Turbé et al. 2017).

Forests as well as urban trees in the Mediterranean Basin can be severely economically impacted by non-indigenous pathogens *(Fig. 2.29)* (Santini et al. 2013; Ghelardini et al. 2017) and arthropods (Kenis and Branco 2010) some of which can be considered pests (Santini et al. 2013; Rassati et al. 2015; Auger-Rozenberg and Boivin 2016; Branco et al. 2016; Mendel et al. 2016) and can also transmit pathogenic fungi (Montecchio et al. 2014). For example, after World War II, chestnut blight epidemic in the mountains of southern Europe aggravated food shortages for local human populations and increased migration to urban areas (Adua 1999). Besides, canker stain disease of the plane tree was considered a nasty non-indigenous species of urban trees, until it was introduced into Greece, where the Oriental plane is endemic. The disease is presently destroying natural river wood ecosystems in Greece (Ghelardini et al. 2017; Tsopelas et al. 2017) and it is also spreading in neighbouring countries such as Albania (Tsopelas et al. 2017) and Turkey (Lehtijärvi et al. 2018). Currently, the pinewood nematode is massively killing pines, changing the landscape in Portugal (Naves et al. 2016).

Human health can also be affected by non-indigenous species. Of major concern are non-native plants that are allergenic; their advanced flowering phenology enlarges the period of airborne prevalence of allergens (Belmonte and Vilà 2004). Non-indigenous birds of the order Psittaciformes (parrots) are potential reservoirs of *Chlamydophila psittaci*, the etiological agent of human psittacosis, and can transmit other diseases to humans and wildlife (Menchetti and Mori 2014; Turbé et al. 2017). Some non-native invertebrates, mainly insects, can cause distress, and allergic reactions (e.g., the Asian hornet, *Vespa velutina*) or be vectors of infectious diseases, (e.g., the tiger mosquito, *Aedes albopictus*) (Lounibos 2002; Jucker and Lupi 2011; Monceau et al. 2014; Goubert et al. 2016; Roques et al. 2018; Lioy et al. 2019).

Some impacts of introduced non-indigenous species have attracted worldwide attention, e.g., on cultural heritage in Palermo (Manachini et al. 2013), on the survival of the endangered date palm, *Phoenix theophrasti* in Crete (Avtzis et al. 2017), and the severe attacks of cypress canker disease in southern Tuscany (Italy). Since researchers and policymakers rarely address the connection between non-indigenous species and damage to cultural heritage directly, the cost of these losses is often neglected or underestimated. The Mediterranean Basin has a long history of civilization and it is rich in cultural heritage that can be threatened by non-indigenous species. For example, in southern Tuscany (Italy), severe attacks of the cypress canker disease (caused by the North American fungal pathogen Seiridium cardinale) are threatening the survival of trees flanking a monumental avenue (Danti and Della Rocca 2017).

2.5.2.3 Further introductions, spread and impacts of non-indigenous species and pests

Existing tools for predicting the risk of introduction and research needs

Horizon scanning, prioritization and Pest Risk Analysis (PRA) are essential tools for focusing limited resources to predict the species which can have a high rate of spread, inflict high impacts, and can be cost-effectively managed. PRA are defined by the International Plant Protection Convention as "the process of evaluating biological or other scientific and economic evidence to determine whether a pest should be regulated and the strength of any phytosanitary measures to be taken against it" (FAO 2017b). An important step in the PRA scheme is the "Pest management section" which assesses phytosanitary measures for relevant pathways and their effectiveness in preventing the entry, establishment and spread of non-indigenous species.

Since 2006, European and Mediterranean Plant Protection Organization (EPPO) has formed expert working groups (EWG) to conduct PRA comprised of experts on the pest and cropping systems, mapping and modelling experts, along with experts on EPPO's PRA scheme, risk managers and EPPO PRA Core Members, all which acts to ensure consistency. EPPO is an international organization responsible for cooperation and harmonization in plant protection within the European and Mediterranean region. One of EPPO's main aims is to provide assistance and guidance to member governments on the administrative, legislative and operational measures necessary to prevent the introduction and spread of non-native plant pests (Smith 1979; Roy et al. 2011). Since 1999, EPPO has maintained an Alert List of plant pests and non-indigenous plants which acts as an early warning for pests, which can present a risk to the EPPO region. EPPO has also developed a prioritization tool for non-indigenous plants that classifies species into one of three lists: minor concern, observation list or list of non-indigenous plant species (EPPO 2012a, 2012b). Those species included in the list of non-indigenous plants are assessed for a PRA, where a higher priority is given to those species with a limited distribution in the EPPO region (EPPO 2012b).

In addition to the EPPO PRA tools, there are also a number of other PRA protocols (Roy et al. 2018) that can be applied to the Mediterranean Basin. To better improve PRA and the risk assessment process, a greater level of transparency and consistency between protocols would be beneficial (Vanderhoeven et al. 2017; González-Moreno et al. 2019). EU Mediterranean member states need to follow the EU Regulation that includes restrictions on keeping, importing, selling, breeding and growing non-indigenous species (European Union 2014).

A quite novel empirical approach to identify potential pests is the use of sentinel plantations of Mediterranean trees, e.g., cork oak, evergreen oak and cypress, in other continents as a priori identification of non-indigenous insect and pathogens capable of colonizing such plants. For example, such plantations in China provided a list of 39 potential non-indigenous insects of which five could be highly damaging (Roques 2015) and several pathogens (Vettraino et al. 2015). The development at potential ports of entry of trapping programs using lures presenting a generic attractiveness for some insect groups is expected to allow early detection of emerging non-indigenous species, even when not listed on quarantine lists (Rassati et al. 2014, 2015; Fan et al. 2019).

Non-indigenous species likely to be introduced into the Mediterranean in the next 20-50 years

As temperature increases, current major non-indigenous species are predicted to shift northwards at an average pace of 37-55 km decade^{-1}, leaving a window of opportunity for new non-indigenous species better adapted to xeric conditions (Gallardo et al. 2017). Regarding non-native plants, gardening practices and ornamental trade will have a major impact on the selection of these future non-indigenous species. The use of non-native drought-tolerant species for gardening and landscaping (i.e., xeriscape) is at its earliest stages in the Mediterranean, but it has already raised concerns in California because of its potential risk as a source of new non-indigenous species (Bradley et al. 2012). Global species niche modelling indicates that xeric shrublands in Mediterranean areas are among the most susceptible ecosystems to introduction by Cactaceae plant species from arid American areas (Novoa et al. 2015). Although some Cactaceae are already largely distributed across the Mediterranean (e.g., *Opuntia ficus-indica*), it is very likely that close relative species with currently restricted distribution or absence in the Mediterranean, such as *Cylindropuntia* spp., would thrive in the next decades aided by new gardening practices (Essl and Kobler 2009). Besides gardening, other relevant terrestrial plant species could be easily introduced and established as contaminants in soil, seeds or containers. For instance, *Parthenium hysterophorus* is a species not currently in the Mediterranean that has been highlighted as of high risk for the region because of its large potential negative impact on agriculture and human health (Kriticos et al. 2015).

While plants have been proportionally the main new non-indigenous species in Europe up until the 19th century, the trend has shifted towards an increasing number of introduced invertebrates and vertebrates in the 20th century (Hulme 2009). This is a pattern that is very likely to continue in the near future by increasing air and maritime cargo, where these taxa can be easily transported as stowaways. The establishment of non-indigenous invertebrates of tropical origin affecting woody ornamental plants has increased (Eschen et al. 2015), meaning that many ornamental plants, especially palms, fig trees and exotic legumes, are at risk for further introduction as well as *Citrus* and *Eucalyptus* trees (Floris et al. 2018). The recent establishment of ambrosia beetles of tropical origin directly threatening plants of the Mediterranean maquis (Faccoli et al. 2016; Francardi et al. 2017) suggests that such process is going to be amplified with global warming. A list of fruit flies likely to be introduced has been recently proposed (Suffert et al. 2018). Special attention should be paid to major agricultural pests currently not present in the Mediterranean but with the potential to be introduced and cause a major impact. The EPPO A1 quarantine list considers up to 128 species of insects, mites, nematodes and gastropods, currently absent from the EPPO region, recommended for quarantine measurements. For instance, Lepidoptera species such as *Spodoptera* or *Helicoverpa* spp. are polyphagous species that could easily thrive in the Mediterranean if they become established. *Spodoptera frugiperda,* a pest native to the Americas, has quickly spread in Africa causing large yield loss. A recent modelling exercise has identified small pockets of suitable habitats in the Mediterranean area but the potential for permanent populations is still uncertain (Early et al. 2018).

Regarding terrestrial vertebrates, several species have been recently highlighted in a horizon scanning exercise for European non-indigenous species, including the Mediterranean (Roy et al. 2019). Of special relevance is the common myna, *Acridotheres tristis,* a non-indigenous species with very restricted populations in the region, and *Lampropeltis* spp., a family of snakes mainly native to North America and adapted to arid conditions. These species are traded as pets and can easily be introduced due to accidental escapes.

2.6 Interaction among drivers

2.6.1 Drivers impacting other drivers

The potential for interactions among drivers is a key issue for analyzing their impacts on environment and human societies, and for developing effective conservation policies (Brook et al. 2008). Climate change, pollution, land and sea use change, and non-indigenous species are often studied and managed in isolation, although it is becoming increasingly clear that a single driver perspective is inadequate when ecosystems are threatened by multiple, co-occurring drivers (Halpern et al. 2008a, 2008b). Conceptually, there are three broad categories of interaction types describing the outcome of multiple stressors, the effects can be additive/cumulative (all the different stresses derived from the implied drivers show up), synergistic (increased stress) or antagonistic (decreased stress) (Folt et al. 1999; Crain et al. 2008). Also, and particularly for the Mediterranean, how different drivers interact could result in alteration, intensification, and even in generation of new impacts (Doblas-Miranda et al. 2017).

In order to facilitate the multi-stressor approach, as a key recognized concept, this section offers two different approximations. First, we describe potential pair interactions within the individual driver classes described before, and second, we provide a few examples of characteristic disturbances of Mediterranean ecosystem that are the result of the combination among multiple drivers.

2.6.2 Pairs of interacting drivers

2.6.2.1 Climate change effects on pollution

Generally, increases in temperature enhance the toxicity of contaminants and increase concentrations of tropospheric O_3 regionally, but will also likely increase rates of chemical degradation (Lelieveld et al. 2014). In general, climate change coupled with air pollutant exposures may have potentially serious adverse consequences for human health in urban and polluted regions (Noyes et al. 2009).

The increase in the intensity and frequency of storm events linked to climate change can lead to more severe episodes of chemical contamination of water bodies and surrounding watersheds (Noyes et al. 2009). Climate change may also increase the occurrence and the global expansion of harmful algal blooms (Paerl and Paul 2012) *(Sections 2.3.3 and 2.3.4)*.

2.6.2.2 Pollution effects on climate change

Many air pollutants that are harmful to human health and ecosystems also contribute to climate change by affecting the amount of incoming sunlight that is reflected or absorbed by the atmosphere, with some pollutants warming and others cooling the Earth. These so-called short-lived climate-forcing pollutants include methane, black carbon, ground-level O_3, and sulfate aerosols. They have significant impacts on the climate; black carbon and methane in particular are among the top contributors to global warming after CO_2 (Shindell et al. 2009; Stohl et al. 2015). Over the Mediterranean Basin the increase and decrease of anthropogenic aerosols during the second half of the 20th century have had an important role in the dimming-brightening phases, because of their direct action on the incoming solar radiation *(Section 2.2.3.1)*.

Pollution by heavy metals or organic compounds can also affect ecosystem functioning by inhibiting CO_2 fixation performed by photosynthetic organisms, thereby increasing global warming (Rochelle-Newall et al. 2008; Magnusson et al. 2010; Ben Othman et al. 2012) (*Sections 2.2.3* and *2.3.2*).

2.6.2.3 Impact of climate on land and sea use

Effects of climate change on land use

Recent accelerated climate change has exacerbated existing environmental problems in the Mediterranean Basin caused by the combination of changes in land use, increasing pollution and biodiversity decline (Cramer et al. 2018). Sea-level rise, combined with land subsidence, may significantly reduce the area available for agriculture. The effects of sea level rise in North Africa, especially

Impacting (column) – Impacted (row)	Climate change	Pollution	Land and sea use changes	Non-indigenous species
Climate change		2.6.2.2	2.6.2.4	?
Pollution	2.6.2.1		2.6.2.7	?
Land and water use changes	2.6.2.3	2.6.2.6		2.6.2.10
Non-indigenous species	2.6.2.5	2.6.2.8	2.6.2.9	

Table 2.3 | Main interactions among drivers

on the coast of the Delta region of Egypt, would impose additional constraints to the agricultural land *(Section 3.2.2.1)*, and also the salinization of coastal aquifers *(Section 3.1.2.2).*

Similarly, with 42% of the population living in coastal areas (Mediterranean Wetlands Observatory 2018), important direct effects of climate change on coastal settlements include dry-land loss due to erosion and submergence, damage of extreme events (such as wind storms, storm surges, floods, heat extremes, and droughts) on built environments, effects on health (food- and water-borne disease), effects on energy use, effects on water availability and resources, and loss of cultural heritage (Hunt and Watkiss 2011) *(Section 2.2.8.2).* Coastal industries, their supporting infrastructure including transport (ports, roads, rail, airports), power and water supply, storm water, and sewerage are highly sensitive to a range of extreme weather and climate events including temporary and permanent flooding arising from extreme precipitation, high winds, storm surges, and sea level rise (Horton et al. 2010; Handmer et al. 2012; Hanson and Nicholls 2012; Aerts et al. 2013). The tourism development experienced a comparable pattern, requiring host facilities and corresponding services. In Algeria, for example, construction projects have been carried out among the coastal paleo-dunes despite the existing Littoral Law 02-2002 (coastal protection) and the Law 01-3-2003 related to the Impact Expertise (Senouci and Taibi 2019). A similar situation exists in the industrial sector (e.g., desalinisation plant and electricity power station built on the beach).

Increases in temperature and decreases in precipitation could alter fire regimes affecting forest cover and could increase the intensity and frequency of drought resulting, in combination with other factors, in desertification (*Sections 2.4.1.2* and *2.6.3*). Future changes in climate could decrease food production *(Section 2.4.1.2)* and may alter the use of land all over the Mediterranean *(Section 2.4.1.3).* Future changes in the quantity and intensity of rain could affect the water cycles and increase the risk of floods (*Sections 2.2.6* and *2.6.3*).

Climate change and variability drives dynamics of marine species

Climate change and variability has led to concomitant changes in Mediterranean marine ecosystems and resources, with various implications on species diversity and composition, where species with limited locomotive capacity or confined in fragmented habitats seem more likely to be affected (Lejeusne and Chevaldonné 2006; Ledoux et al. 2015). Examples of this changing environment, among others, are the mass mortality events of gorgonians and other sessile metazoans in northwestern Mediterranean (Garrabou et al. 2009; Rivetti et al. 2014) and the continuous decline of *Posidonia* meadows (Marba and Duarte 2010), the increase in the frequency of red tides and of gelatinous carnivore outbreaks (Conversi et al. 2010), the "tropicalization" of marine fauna in favour of the more thermophilic ones (Bianchi 2007), and the increase spread of microbial pathogens associated with water temperature rise (Danovaro et al. 2009) *(Section 4.1.1.1).*

At the end of 1980s and especially during the mid-1990s the Mediterranean Sea underwent regime shifts (Conversi et al. 2010; Alheit et al. 2019) that inflicted major atmospheric, hydrological and ecosystem changes, also affecting marine resources, mainly fisheries. There have been various studies linking ocean-atmospheric processes such as the Atlantic Multidecadal Oscillation (AMO), the North Atlantic Oscillation (NAO) and the Western Mediterranean Oscillation (WeMO) indices to alterations on the distribution and biomass of pelagic fish, as well as their catch composition (Alheit et al. 2019). Pelagic fish populations, more than other fish species, act as sentinels of these environmental changes. For example, during mid-1990s in the Mediterranean the highly correlated sea surface temperature and AMO index show a sharp increase (Marullo et al. 2011; Macías et al. 2013), whereas the dynamics of many fish species - mainly pelagic - show a conspicuous change around that time.

It is not yet clear how these changes impact pelagic fish population dynamics, combined with the pressures imposed by anthropogenic activity. Fifty-nine taxonomic groups (species or groups of species) showed an abrupt change in their landings in the mid-late 1990s (Tzanatos et al. 2014) with approximately 64% of these changes being correlated with sea surface temperature, mostly inversely correlated. The landings of some species (European sardine *Sardina pilchardus,* squids *Loligo* spp., Norway lobster *Nephrops norvegicus,* and hake *Merluccius merluccius*) decreased conspicuously in the mid-1990s, whereas those of other species (European anchovy *Engraulis encrasicolus* and greater amberjack *Seriola dumerili*) increased (Tzanatos et al. 2014). A study of the fisheries landings of 30 fish and invertebrate taxonomic groups revealed regime shifts at the mid-1990s, concurrent with the sea temperature increase in the eastern and western basins (Vasilakopoulos et al. 2017). The late 1990s was determined as the turning point for the northward expansion of warm-water species in the Mediterranean (Azzurro et al. 2011). This was confirmed by Raitsos et al. (2010) who showed a clear increase of non-indigenous species entering into the eastern Mediterranean Sea in 1998. Earlier, Pinnegar et al. (2003) reported that the diversity of the western Mediterranean finfish landings increased dramatically after 1995 as a result of new species entering the catch.

Round sardinella *(Sardinella aurita),* a warm-water small pelagic fish species distributed along the southern Mediterranean coastline has been reported to have expanded its distribution to the northern Aegean (Tsikliras 2008), the northern Adriatic (Sinovčić et al. 2004), the Gulf of Lions (Francour et al. 1994), and the northwestern Mediterranean (Sabatés et al. 2006, 2009). A significant positive relationship between round sardinella landings and sea surface temperature anomalies has been reported for the western (Sabatés et al. 2006) and eastern Mediterranean (Tsikliras 2008). The northward distributional shift coincides with the beginning of positive temperature anomalies in the mid-1990s (Tsikliras 2008; Sabatés et al. 2009; Stergiou et al. 2016). Similarly, concomitant with the sea surface temperature change in the western Mediterranean, the landings of bluefish *(Pomatomus saltatrix)* quadrupled due to a northward expansion of the species (Sabatés et al. 2012) and, at the same time, anchovies returned to high biomass, as a result of increasing sea surface temperatures in the Adriatic Sea (Vilibić et al. 2016).

The effect of the AMO and NAO signals across the Mediterranean Sea sub-regions (western, central and eastern) on the small (European sardine *Sardina pilchardus,* European anchovy *Engraulis encrasicolus,* round sardinella *Sardinella aurita* and European sprat *Sprattus sprattus*) and medium (Atlantic mackerel *Scomber scombrus,* Atlantic chub mackerel *Scomber japonicus,* Atlantic horse mackerel *Trachurus trachurus,* Mediterranean horse mackerel *Trachurus mediterraneus*) pelagic fishes have been recently studied in the western, central and eastern Mediterranean Sea (Tsikliras et al. 2019). The pelagic fishes of the central and eastern Mediterranean respond most strongly to AMO variability and those of the central and western Mediterranean also respond to the NAO, while the effect of the NAO on pelagic fishes of the eastern Mediterranean was not significant (Tsikliras et al. 2019). Generally, various indicators revealed that the time of the pelagic fish response to the AMO and NAO signals varied among the Mediterranean sub-regions (Alheit et al. 2014; Tsikliras et al. 2019).

Finally, the mean temperature of the catch, an indicator that assesses the effect of global warming on the exploited marine communities (Cheung et al. 2013b), has been increasing across the Mediterranean showing that the ratio of thermophilous (warm-water) to psychrophilous (cold-water) marine species has been changing in favour of the former. This is indicative of either an increase in the relative proportion of thermophilous species in the catches or a decrease in the relative proportion of the psychrophilous ones (Tsikliras and Stergiou 2014).

2.6.2.4 Effects of land use on climate change

Changes in crop use (Tribouillois et al. 2018), especially in forest cover, affect the balance between sink and release of CO_2 and the emissions of biogenic volatile organic compounds (BVOCs) in the atmosphere (Doblas-Miranda et al. 2017) (*Sections 2.4.1.2* and *3.1.2.1).*

Modification of surface albedo by land use changes also entail a highly potential impact on climate change (Benas and Chrysoulakis 2015). Changes in forest or dehesa/montado cover due to reforestation could reduce albedo (Rotenberg and Yakir 2011; Godinho et al. 2016), while fires increase radiations returns to the atmosphere (Sánchez et al. 2015), with contrasting effects on local climate. Agricultural cover may decrease or increase albedo (Giannakopoulou and Toumi 2012; Carrer et al. 2018), while urban sprawl definitely increases the radiation absorption and therefore local temperature (Salvati et al. 2019) *(Section 3.1.3.1).*

Change of land use and irrigation practices increase evapotranspiration and have a net cooling effect in some areas of the Mediterranean region (Zampieri and Lionello 2011; Thiery et al. 2017; Gormley-Gallagher et al. 2020).

2.6.2.5 Links between trends in non-indigenous species and climate change

Impact of climate change on marine non-indigenous species

The introduction of non-indigenous species and global warming interact in complex ways (Stachowicz et al. 2002), and are linked also in the Mediterranean Sea (Occhipinti-Ambrogi 2007).

This connection strongly depends on the species and the mode of its introduction, establishment and colonization. Overall, there is a strong trend of "tropicalization" of temperate areas through the movement of warm-loving (thermophilic) species toward the poles in areas of rapid ocean warming, and with increasingly strong impacts on local communities (Vergés et al. 2014a, 2016). These are not considered as introductions of non-indigenous species per se. But ocean warming may facilitate the establishment and spread of thermophilic non-indigenous species. The success of establishment of an introduced species depends on how suitable the ocean climatic parameters are in the region of introduction. Because successful non-indigenous species are typically generalists with broader climatic tolerances, they are usually considered able to cope better with climate change than native ones (Walther et al. 2009).

There is limited evidence for effects of climate change on the introduction of non-indigenous species. Theoretically, at the trailing "warm" edge of species distributions, the populations of sensitive cold-affinity species should reduce (and eventually extirpate) and that of warm affinity species (including thermophilic species) should increase (Bates et al. 2014). In the Mediterranean Sea, tropicalization evidently occurs (Vergés et al. 2014a), mainly in the Levant, by Lessepsian introductions, and ocean warming was suggested to facilitate the successful establishment of non-indigenous species (Raitsos et al. 2010). Ocean warming probably also helps to spread both native thermophilic species and successful Lessepsian species westward along the basin's temperature gradient, and also northward into the Aegean and Adriatic seas, or even the Ligurian Sea, but there very few direct empirical studies to demonstrate that. Recent analysis of fish trawl data from the southeast Mediterranean (Israel) does strongly suggest that non-indigenous species are indeed promoted by warming while natives are declining (Givan et al. 2017a).

Some studies suggest that habitats degraded by global warming are more likely affected by non-indigenous species than nearly-pristine habitats, envisaging explicitly or not, a cause and effect link between climate warming and the success of introductions (Stachowicz et al. 2002; Bianchi 2007; Galil 2007; Occhipinti-Ambrogi 2007). However, field observations do not support this idea, but reveal instead conflicting results that have provoked intense debate (Boudouresque and Verlaque 2010). For example, in the Mediterranean Sea, well-structured and conserved habitats (such as coralligenous or *Cystoseira* forests) are able to mitigate and delay the proliferation and spread of the non-indigenous alga *Caulerpa cylindracea,* probably because the complexity of substrata (enhanced by gorgonians or canopy algae presence) is a key factor limiting its colonization and spread (Ceccherelli et al. 2002; Bulleri and Benedetti-Cecchi 2008; Verdura et al. 2019). In contrast, mass mortality of structural native species and subsequent increase of turf-forming species due to an extreme climatic event indirectly promoted the introduction of *C. cylindracea* in a coralligenous habitat (Verdura et al. 2019). However, in other non-indigenous algae such is *Lophocladia lallemandii,* introduction is favoured by more complex and rich communities (Cebrián et al. 2018), and thus simplification derived from climate change effects is expected not to enhance the capacity of *Lophoclaria* to establish itself, but prevents its spread.

Using natural laboratories to test the thermal performance curves and sensitivity to acidification of key native and non-indigenous species, as well as the impact of climate change related environmental alteration on species interactions and communities and their ecosystem functions, are critical for better understanding and forecasting of the interactions between climate change and the introduction of non-indigenous species (Rilov et al. 2019a). For example, heat polluted areas and CO_2 vents (Hall-Spencer et al. 2008), as well as laboratory experiments in near-natural mesocosm systems (Wahl et al. 2015). Such recent measurements and experiments in the southeastern Levant have shown that some non-indigenous species (foraminifera) are tolerant to extreme thermal stress (Titelboim et al. 2017), that under warming and acidification conditions most Lessepsian species perform better than native species (Guy-Haim et al. 2016; Guy-Haim 2017), and demonstrated that different thermal performance of two Red Sea foraminifera explain why one species was introduced and the other did not (Titelboim et al. 2019). Furthermore, mesocosm work showed that a *Cystoseira* community becomes more heterotrophic and more dominated by non-indigenous species (but species richness does not change), demonstrating the profound impact of the combination of climate change and the introduction of non-indigenous species on ecosystem function (Guy-Haim et al. 2016; Rilov et al. 2019b).

Impacts of climate change on terrestrial non-indigenous species

There are five non-exclusive consequences of climate change on non-indigenous species: (1) altered transport and introduction mechanisms, (2) establishment of new species, (3) altered impact of existing non-indigenous species, (4) altered distribution of existing non-indigenous species, and (5) altered effectiveness of control strategies (Hellmann et al. 2008) *(Section 2.5.1.3).*

The influence of climate change on terrestrial non-indigenous species highly depends on species physiological strategy and reproductive adaptations (Bale and Hayward 2010; Antunes et al. 2018). Generalized ecosystem models of plant functional groups applied to Mediterranean islands indicate that climate change might promote the introduction of broadleaved trees (e.g., *Ailanthus altissima*) more than C_4 tropical grasses (e.g., *Amaranthus retroflexus*) (Gritti et al. 2006). Many non-indigenous species from temperate and cold climates might only be able to shift their ranges northward or to expand in altitude because they will be limited by drought and high temperatures (Storkey et al. 2014; Gallardo et al. 2017). While non-indigenous species whose native ranges are drier and warmer than their introduced ranges can be at an advantage to occupy niches at southern latitudes (Gallardo et al. 2017). Therefore, some species might loose and some gain suitable areas for introduction. Regions which will get drier are predicted to lose the highest number of potential non-indigenous species.

For introduced gardening plants, the climatically suitable areas with future climate change are unequally distributed across Europe with more suitable areas in the East than in the West of the Mediterranean Basin (Dullinger et al. 2017). This will be the case for *Cortaderia* which suitable area can increase 69-116% for 2060 (Tarabon et al. 2018) or for *Nassela* that can increase up to 47% for 2018 (Watt et al. 2011).

Similarly, weeds in crops can experience range shifts, niche shifts and trait shifts with climate change that will influence the agronomic practices to reduce their interference to crop production (Peters et al. 2014). Weeds in cereals crops will also advance towards northeastern Europe and remain or contract their distribution in warm areas of the Mediterranean region (Castellanos-Frías et al. 2014).

Climate change can advance the phenology of non-indigenous plant species including their fecundity (Chuine et al. 2012), pollen production and seed maturation (Leiblein-Wild et al. 2016). Changes in pollen production can exacerbate the problem caused by allergenic non-indigenous plants such as the American *Ambrosia* because the allergenic risk is predicted to increase under all climate scenarios tested (Rasmussen et al. 2017).

Besides the influence of climate change on the establishment and spread of non-indigenous species, a remaining question is whether their impacts on native species increase in combination with climate change. A few greenhouse experiments have explored the interaction between competition of non-indigenous species and drought on the performance of native species (García-Serrano et al. 2007; Matesanz et al. 2008; Werner et al. 2010) and have found a non-synergistic effect. The interaction of climate change and introduction of non-indigenous species is a research area that requires further experimentation for productive systems such as the effect of weeds, pests and pathogens on crops and forestry (Ramesh et al. 2017).

2.6.2.6 Impacts of pollution on land and sea use

One of the major drivers relative to greenhouse gases pollution may be the CO_2 fertilization affecting forests. The balance between faster growth due to the fertilization effect and hydric stress due to most likely warmer and drier conditions have generated a considerable debate in the Mediterranean area (Keenan et al. 2011; Peñuelas et al. 2011). However, the most recent studies mainly corroborate that the effects of CO_2 fertilization will be negligible under the predicted climate conditions for the region (Camarero et al. 2015; Nunes et al. 2015; Gea-Izquierdo et al. 2017), despite some exceptions (Koutavas 2013; Barbeta and Peñuelas 2017). The potential effects of nitrogen deposition on Mediterranean forest growth also seem to be low (Ochoa-Hueso et al. 2014).

2.6.2.7 Impacts of land and sea use change on pollution

Intensive farming increases releases of nutrients and pesticides in aquifers while higher releases of methane in the air. The effects of the increase of livestock production on greenhouse gas emissions are assessed in *Section 3.2.3.2.*

Urban sprawl is associated to higher traffic related emissions *(Sections 2.3.3* and *2.4.1.2).*

2.6.2.8 Pollution effects on non-indigenous species

Pollution can make environmental conditions less tolerable for native species, and provide space and nutrients for opportunists, including non-indigenous species (Crooks et al. 2011).

2.6.2.9 Effects of land and water use on non-indigenous species

Habitat destruction causes disturbance, which opens space for non-indigenous species (Hobbs and Huenneke 1992).

2.6.2.10 Effects of non-indigenous species on land and sea use

Outbreaks of forest non-indigenous insects could alter forest cover *(Section 2.5.2.2).*

2.6.3 More complex interactions among drivers

2.6.3.1 Floods

Floods are an illustrative example of the combination of different drivers such as climate change (extreme precipitation events), land use change (catchment changes on river forests, forest cover, etc.) and even indirect drivers (among them and principally, urban sprawl in risk areas) (*Sections 3.1.3.3* and *3.1.4.1*).

2.6.3.2 Desertification

Puigdefábregas and Mendizabal (1998) analyzed FAO data from Morocco, Algeria and Tunisia during the period 1950-1993, associating desertification to socio-economic boundary conditions and over-exploitation by showing clear increases in population (pressure) and in the use of unsustainable land use practices in the Mediterranean, principally irrigation *(Section 6.6).*

Desertification is in fact the result of two different factors in origin operating in combination, prolonged drought of climatic origin and land exploitation of human origin (Le Houérou 1996). In Mediterranean arid lands, mainly during the 20th century, short-term planning of agricultural policies and overexploitation, mainly in the form of overgrazing but also fuelwood collection and ground water exploitation, contributed to soil quality decline and massive erosion. Deteriorating conditions have a great impact on the lives of inhabitants of Mediterranean drylands and force most of them to migrate (Mohamed and Squires 2018) (*Sections 3.2.1.4, 4.3.1* and *6.6*).

2.6.3.3 Wildfires

One relevant consequence of the Mediterranean Climate characterized by dry summers are forest fires. Those can be exacerbated by drought conditions (Turco et al. 2018) but in turn they can affect drastically the flood generation both due to the erosion and the loss of forest mass. Although some forest fires can be provoked or as a result of recklessness, they mostly depend on the state of the vegetation and the climatic and meteorological situation. Consequently, fire regimes will be affected by climate change, if not already affected (Sarris et al. 2014). The Mediterranean is a high fire-risk region, where fires are the cause of severe agricultural, economic and environmental losses and even human casualties (Moreira et al. 2011; Keeley et al. 2012; San-Miguel-Ayanz et al. 2013; Bowman et al. 2017). For instance, the fire seasons in 2017 and 2018 was severe in many regions of Southern Europe, with large wildfires associated with unusually intense droughts and heat-waves (Sánchez-Benítez et al. 2018). In Portugal, the year of 2017 was particularly tragic. An extended and extraordinarily intense fire season yielded a record total burned area of about 500,000 hectares and more than 120 fatalities (Turco et al. 2019). Instead, the summer of 2018 will be remembered by the deadliest fires ever recorded affecting Greece, when a series of wildfires close to Athens killed 99 people, the deadliest in Greece history (AghaKouchak et al. 2018).

However, although several reports, ranging from popular media through to peer-reviewed scientific literature, have led to a shared perception that fires have increased or aggravated in recent years, the quantitative evidence available indicated that fires are decreasing on recent decades in this area (Turco et al. 2016). The increased efforts in fire suppression have probably played an important role in driving the general downward trends described for most of the Mediterranean area (Moreno et al. 2014; Ruffault et al. 2015). In recent decades fire management strategies have improved thanks to new technologies and experience while climate drivers have led to an opposite trend (Amatulli et al. 2013; Batllori et al. 2013; Bedía et al. 2013; Turco et al. 2014; Dupire et al. 2017; Fréjaville and Curt 2017) *(Section 4.3.2.1).*

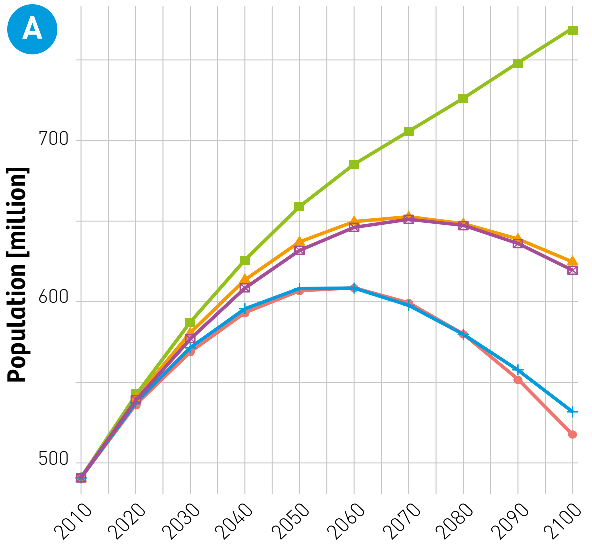
FULL NAME	SHORT NAME	THEMATIC FOCUS	NO. OF SCENARIOS	TIME HORIZON	SPECIFIC REGIONAL FOCUS	REFERENCE(S)
GLOBAL SCALE						
Special Report on Emissions Scenarios	SRES	Emission of greenhouse gases	4	2100	-	Nakićenović, 2000
Shared Socioeconomic Pathways	SSPS	Multidisciplinary with a focus on challenges to climate change adaptation and mitigation	5	2100	-	(O'Neill et al. 2014, 2017)
MEDITERRANEAN SCALE						
A sustainable future for the Mediterranean	-	Multidisciplinary and cross-sectoral with an emphasis on sustainable development	2	2025	-	Benoit and Comeau, 2005
Mediterranean scenarios (MedAction project)	-	Multidisciplinary and cross-sectoral with an emphasis on desertification	3	2030	Northern Mediterranean case studies in ES, GR, IT, PT	Kok et al., 2006
EuroMed-2030	-	Multidisciplinary and cross-sectoral with a focus on the Euro-Mediterranean relationship	4	2030	-	EC/DG for Research and Innovation, 2011
Tomorrow, the Mediterranean	-	Multidisciplinary, cross-sectoral, with emphasis on economic development	3	2030	-	IPEMED, 2011
Scenarios for the Mediterranean Region	-	Evolution of regional dynamics and role of the private sector in shaping business and political environments	3	2030	Exclusion of AL, BA, HR, IL, ME, PS, TR	World Economic Forum, 2011
Mediterranean Coastal SSPs	-	Regional and sectoral extension of the global SSPs for Mediterranean coastal regions	5	2100	-	Reimann et al., 2018
EUROPEAN SCALE						
Integrated Visions for a Sustainable Europe	VISIONS	Sustainable development	3	2020, 2050	-	Rotmans et al., 2000
Demographic and Migratory Flows Affecting European Regions and Cities	DEMIFER	Demography and European policies	5	2050	-	Rees et al., 2012
Climate Change Integrated Assessment Methodology for Cross-Sectoral Adaptation and Vulnerability in Europe	CLIMSAVE	Multidisciplinary and cross-sectoral, with emphasis on ecosystem services and provisions	4	2050	-	Gramberger et al., 2013
Territorial Scenarios and Visions for Europe	ET2050	Territorial development and cohesion	4	2050	-	MCRIT, 2015
Demographic Scenarios for the EU	-	Demographic development with a focus on aging, migration and education	3-4	2060	EU	Lutz et al., 2019
European Shared Socioeconomic Pathways	Eur-SSPs	Regional extension of the global SSPs for the European context	4	2040, 2070, 2100	Additional case study in Iberia	Kok et al., 2019

Table 2.4 | Overview of selected socioeconomic scenarios that cover Mediterranean countries, partly based on Rohat et al. (2018) and Sanna and Le Tellier (2013). ISO country codes: AL: Albania, BA: Bosnia and Herzegovina, ES: Spain, GR: Greece, HR: Croatia, IL: Israel, IT: Italy, ME: Montenegro, PT: Portugal, PS: State of Palestine, TR: Turkey.

2.7 Mediterranean socioeconomic scenarios

Environmental-change-related impacts will be driven not only by changes in climatic conditions, but also by changes in socioeconomic conditions. Prevailing socioeconomic conditions, in particular, determine a society's resilience to climatic hazards. When assessing future risks due to climate change, it is therefore crucial to account for plausible changes in socioeconomic conditions using a range of socioeconomic scenarios (González-Moreno et al. 2013).

A large number of socioeconomic scenarios have been developed in the past decades, focusing on a multitude of disciplines, sectors, and regions. Few of these scenarios were developed specifically for the Mediterranean region and even those usually only cover some of the Mediterranean countries, with a strong bias toward northern Mediterranean countries that are members of the European Union. *Table 2.4* provides an overview of a range of socioeconomic scenarios developed in the last two decades that cover socioeconomic developments



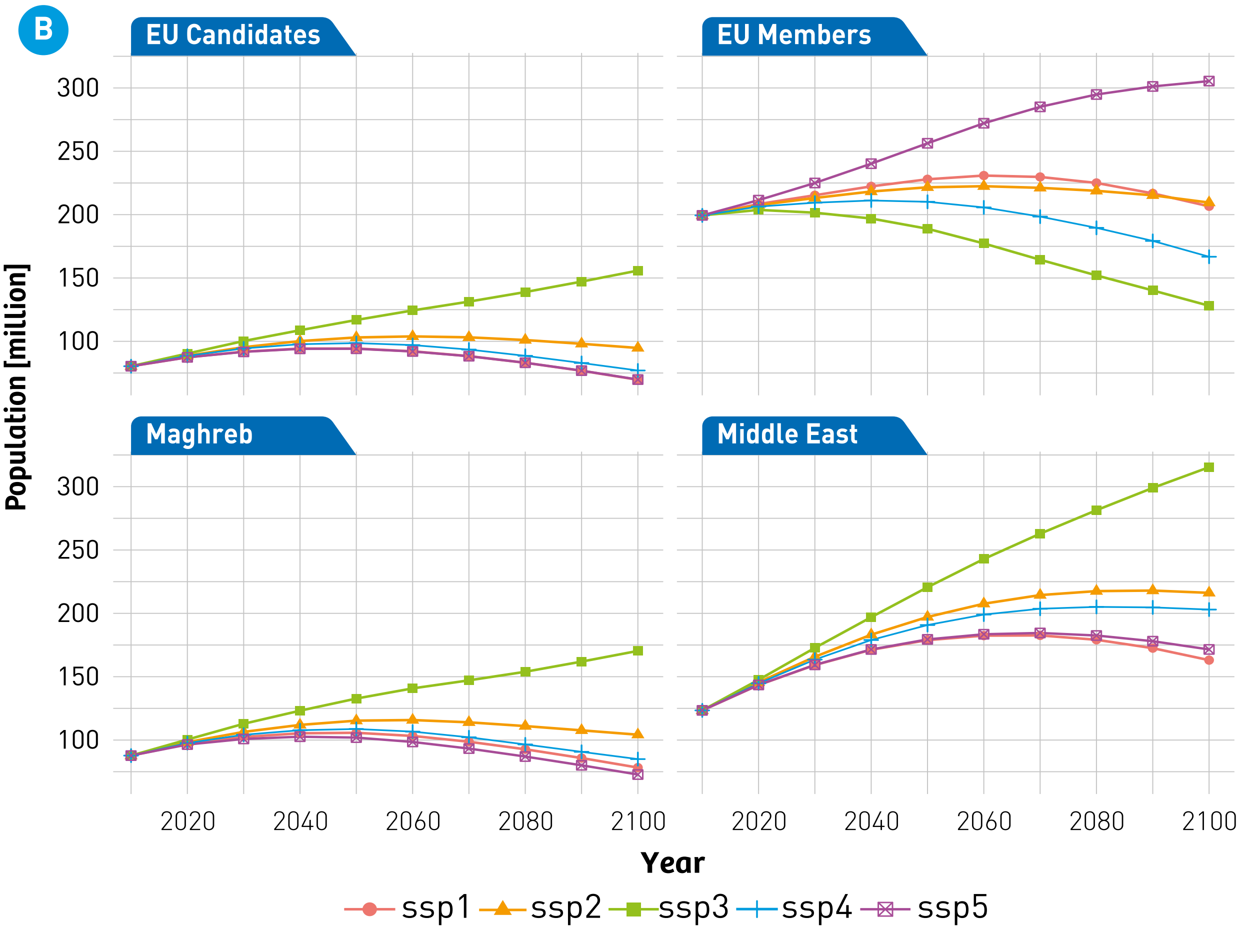
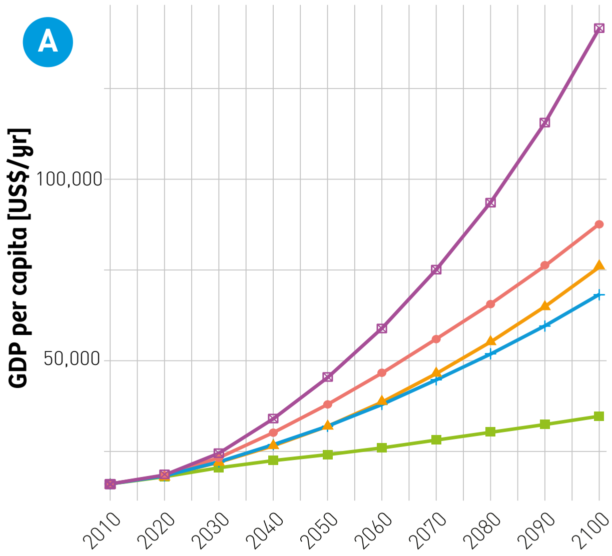


Figure 2.30 | Mediterranean population projections under the different Shared Socioeconomic Pathways (SSPs), A) total Mediterranean population, B) population by geographical region (Kc and Lutz 2017).

in the Mediterranean either fully or partially in terms of geographic coverage.

The most recent socioeconomic scenarios that account for socioeconomic developments in the entire Mediterranean region (as defined in this report) are the state-of-the-art global-scale Shared Socioeconomic Pathways (SSPs). The SSPs explore broad-scale societal trends in the course of the 21st century both qualitatively, in the form of scenario storylines, and quantitatively, in the form of national-level projections of key variables such as population (Kc and Lutz 2017), urbanization (Jiang and O'Neill 2017), and Gross Domestic Product (GDP) (Cuaresma 2017; Dellink et al. 2017; Leimbach et al. 2017). The Mediterranean population is projected to range from 607 million (SSP1) to 659 million (SSP3) in 2050 and from 518 million to 770 million in 2100 *(Fig. 2.30a)*, with considerable differences across regions *(Fig. 2.30b)*. The largest share of the population is projected to live in Egypt under all SSPs, except SSP5 where France is the most populous country in 2100 due to very high work migration into northern Mediterranean countries (O'Neill et al. 2017).



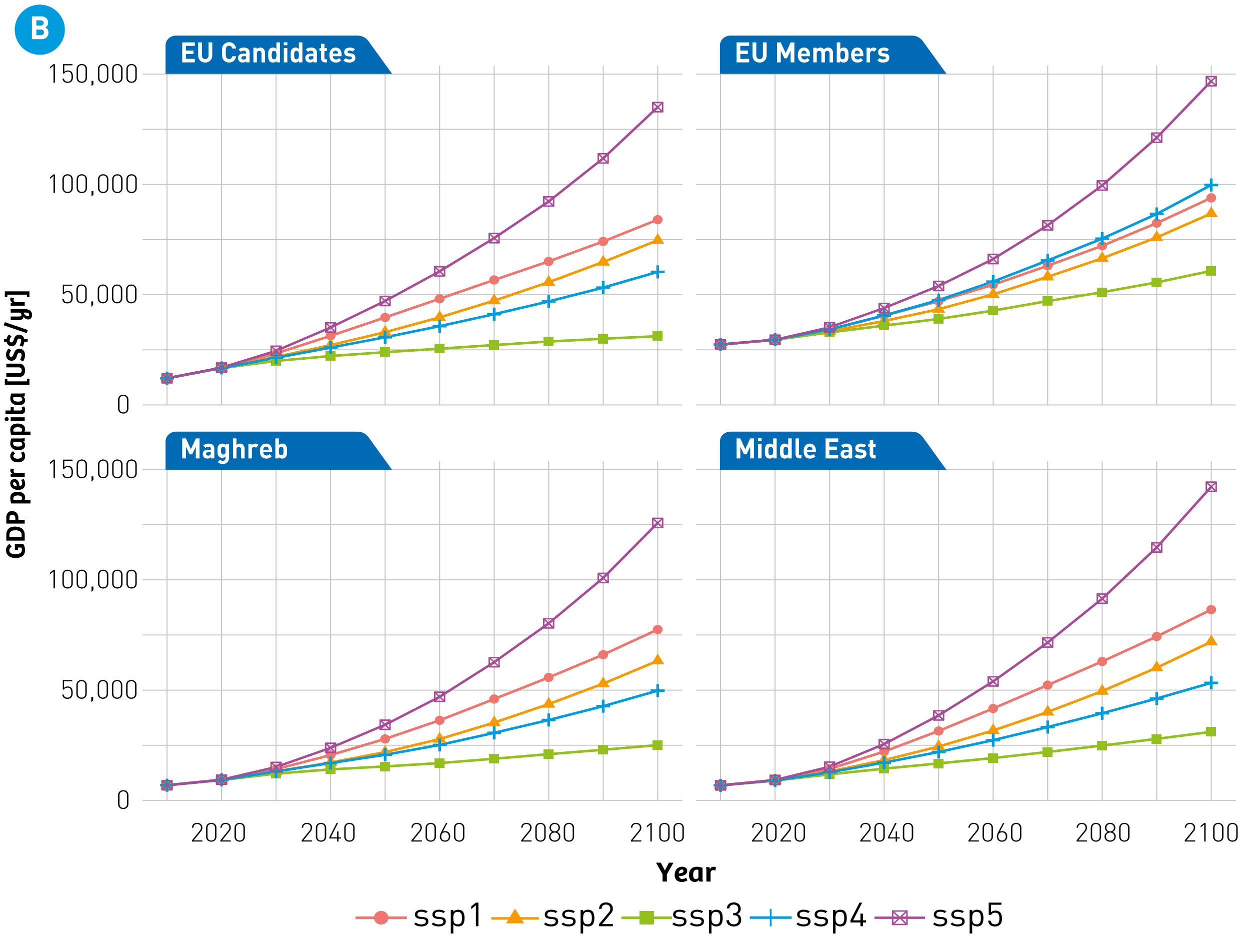


Figure 2.31 | Mediterranean Gross Domestic Product (GDP) projections under the different Shared Socioeconomic Pathways (SSPs), A) Mediterranean average GDP per capita, B) average GDP per capita by geographical region (Dellink et al. 2017).

The regional average GDP per capita is projected to grow from 16,000 US$ yr^{-1} in 2010 to between 24,000 US$ yr^{-1} (SSP3) and 45,000 US$ yr^{-1} (SSP5) in 2050 and to roughly 35,000 US$ yr^{-1} (SSP3) to 142,000 US$ yr^{-1} (SSP5) in 2100 *(Fig. 2.31a)*. The differences in average GDP per capita are small between (potential) EU candidate countries, countries from the Middle East and the Maghreb region. EU member states have the highest average GDP per capita under all SSPs *(Fig. 2.31b)*.

In order to increase the usefulness of SSPs for impact, adaptation, and vulnerability assessments (van Ruijven et al. 2014), spatially explicit population projections that account for spatial changes in population distribution in the course of the 21st century have been produced, using the national totals as input data. These are available for all Mediterranean countries at a horizontal resolution of 7.5 arc minutes (Jones and O'Neill 2016) and 30 arc seconds (Merkens et al. 2016; Gao 2017). Further, downscaled GDP projections are available for SSPs 1-3 at a resolution of 30 arc minutes (Murakami and Yamagata 2019). All of these projections are based on the underlying global SSP assumptions.

As the global assumptions do not necessarily reflect the socioeconomic developments at the regional scale, extensions of the global-SSPs for the Mediterranean coastal zone have been developed (Reimann et al. 2018). These Mediterranean coastal SSPs account for region-specific developments as well as for changing attractiveness of coastal regions for human settlement across the SSPs, while at the same time ensuring consistency with the global SSPs (Zurek and Henrichs 2007). The Mediterranean coastal SSPs consist of qualitative narratives for each coastal SSP – SSP1 "Green Coast", SSP2 "No Wind of Change", SSP3 "Troubled Waters", SSP4 "Fragmented Coast", and SSP5 'Coast Rush' – differentiating between regional socioeconomic developments in northern versus southern and eastern parts of the region; and of spatially explicit population projections for all Mediterranean riparian countries at a resolution of 30 arc seconds *(Fig. 2.32)*.

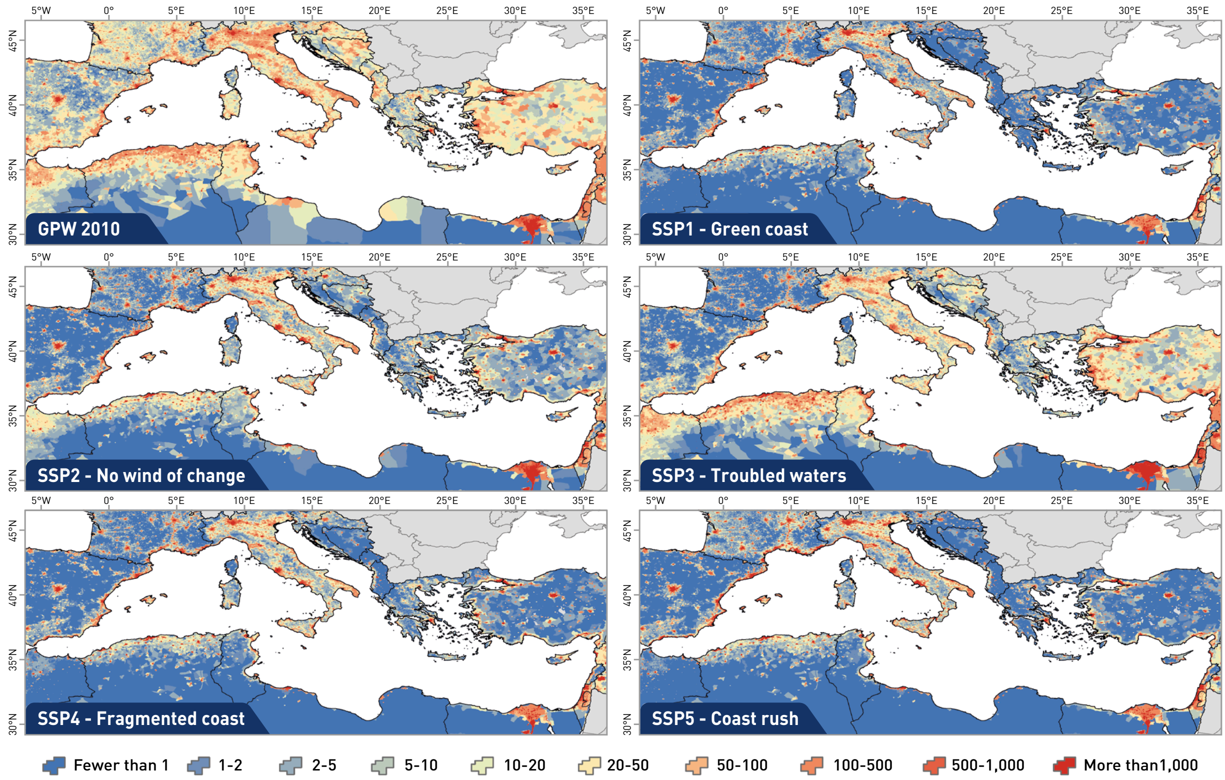


Figure 2.32 | Spatially explicit population projections produced for the Mediterranean Coastal SSPs. Selected population grids for the base year 2010 and each SSP in 2100 (Reimann et al. 2018), GPW = Gridded Population of the World (Center for International Earth Science Information Network - CIESIN - Columbia University 2016).

BOX 2.1

How much has the Mediterranean Basin warmed since the pre-industrial period?

The UNFCCC Paris Agreement of 2015 strengthens the initial goal of the Article 2 in the United Nations Framework Convention on Climate Change (UNFCCC), "to achieve … stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system", by "holding the increase in the global average temperature to well below 2°C above pre-industrial levels and pursuing efforts to limit the temperature increase to 1.5°C above pre-industrial levels". While these temperature goals refer to the global average, it is a natural question to ask, for any region, how much warming has been observed "above pre-industrial levels".

This is only apparently a simple question. To provide such an estimate it is necessary to clarify the meaning of pre-industrial, the information available on its average temperature in the region, a definition of present average temperature and the method used for estimating it. These issues were at the core of the IPCC special report on global warming of 1.5°C (SR15, IPCC 2018) and specifically considered in its *Chapter 1* (Allen et al. 2018).

The period 1850-1900 has been identified (Allen et al. 2018) as a suitable approximation for the estimate of the pre-industrial average temperature, because it combines typical pre-industrial solar and volcanic forcing, low anthropogenic greenhouse gas concentrations, and sufficient coverage of instrumental temperature observations. The choice of the period 1850-1900 is not completely free of problems, because it is indeed already affected by increasing greenhouse gas concentrations, with a partial compensation caused by aerosols. Further, strong volcanic eruptions occurred in the period 1880-1900. In this box, we follow SR15 but also the indications of the World Meteorological Organization (WMO 2017) and compute average pre-industrial temperature for a 30-years period, the central 1861-1890 period of the 1850-1900 "pre-industrial" period.

The number and distributions of instrumental observation have changed significantly over time, and station density could become critically low in African and Asian areas of the Mediterranean Basin. Further, the collection of observations over sea is systematically more problematic than over land, notably prior to the 20th century. Rather than analyzing station data, we seek consistency with global estimates and we use the HadCRUT4 dataset (Morice et al. 2012) and the CRUTEM4 (Jones et al. 2012) data sets for the land+sea and land only analysis, respectively. These are two widely used gridded global data sets with a resolution of 5 degrees longitude/latitude since 1850 until present. Other data sets at higher spatial resolution are available (such as the recently updated version of CRU TS) (Harris et al. 2020), but they do not reach back far enough into the 19th century for the estimation of pre-industrial conditions. For this analysis, the Mediterranean Basin is defined as the domain from 10°W to 40°E of longitude and from 30°N to 47.5°N of latitude. For the averaging, an interpolation (based on the closest neighbours) to 1° spatial resolution was undertaken.

Obviously observed temperature values are not yet available for the period 2020-2034. This prevents computing the level of warming in 2020 using a simple 30-year average. Here, we make the conservative assumption that warming in the future will continue at the same rate of the last 50 years and compute the 2020 temperature by extrapolating the linear trend of the 1970-2019 period *Fig. 2.33*). In 2020 the Mediterranean Basin is 1.5°C warmer than in the preindustrial with a likely uncertainty range of +0.11°C. Land areas have warmed more than the sea. If only land areas are considered, the 2020 temperature is 1.8°C warmer than pre-industrial with a likely uncertainty range of +0.12°C.

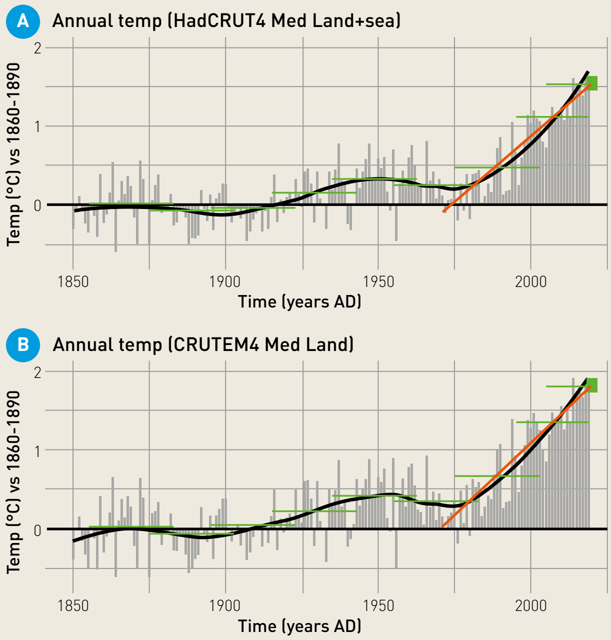


Figure 2.33 | Time series (grey bars) of the annual average temperature of the Mediterranean Basin (30°N to 47.5°N and 10°W to 40°E) considering the whole region (panel A) or only its land areas (panel B). The analysis is based on the HadCRUT4 data set for the land+sea analysis (Morice et al. 2012) and on the CRUTEM4 data set for the land analysis (Jones et al. 2012)(Jones et al, 2012). To avoid edge effects, before calculating the averages, the data have been interpolated to 1° spatial resolution. The values are expressed as anomalies from the pre-industrial period (1860-1890). The blue horizontal lines represent the 30-yr averages by steps of 20 years. The red curve is the linear trend linear trend calculated for the 1970-2019 period, extrapolated up to 2020. The black line is the smoother time-series. The blue square represents the likely interval (probability >0.66) for the present warming.

BOX 2.2

Representative Concentration Pathways (RCPs)

Representative Concentration Pathways (RCPs) are greenhouse has concentration pathways, developed by the IPCC in order to explore the physical outcomes of different climate policies, notably regarding the mitigation of greenhouse gas emissions.

RCP2.6: The RCP2.6 was developed by the IMAGE modeling team of the Netherlands Environmental Assessment Agency (van Vuuren et al. 2011). The emission pathway is representative for scenarios in the literature leading to very low greenhouse gas concentration levels. It is a so-called "peak" scenario: its radiative forcing level first reaches a value around 3.1 W m^{-2} mid-century, returning to 2.6 W m^{-2} by 2100. In order to reach such radiative forcing levels, greenhouse gas emissions (and indirectly emissions of air pollutants) are reduced substantially over time.

RCP4.5: It was developed by the MiniCAM modeling team at the Pacific Northwest National Laboratory's Joint Global Change Research Institute (Clarke et al. 2014). It is a stabilization scenario where total radiative forcing is stabilized before 2100 by employment of a range of technologies and strategies for reducing greenhouse gas emissions. It is often considered as an intermediate scenario.

RCP8.5: The RCP8.5 was developed by the MESSAGE modeling team and the IIASA Integrated Assessment Framework from the International Institute for Applies Systems Analysis (IIASA), Austria (Riahi et al. 2011). The RCP8.5 is characterized by increasing greenhouse gas emissions over time representative for scenarios in the literature leading to high greenhouse gas concentration levels, reaching +8.5 W m^{-2} additional surface radiative forcing in 2100. It is often considered as a "business-as-usual" scenario.

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Information about authors

Coordinating Lead Authors

Semia Cherif:
Tunis El Manar University, Higher Institute for Applied Biological Sciences of Tunis (UTM-ISSBAT), Tunis, Tunisia

Enrique Doblas-Miranda:
CREAF, Bellaterra (Barcelona), Spain & Universitat Autònoma de Barcelona, Bellaterra (Barcelona), Spain

Piero Lionello:
University of Salento, Lecce, Italy

Lead Authors

Carlos Borrego:
Department of Environment and Planning (DAO), Centre for Environmental and Marine Studies (CESAM), University of Aveiro, Aveiro, Portugal

Filippo Giorgi:
Abdus Salam International Centre for Theoretical Physics (ICTP), Trieste, Italy

Ana Iglesias:
Universidad Politécnica de Madrid (UPM), Madrid, Spain

Sihem Jebari:
Tunisian National Research Institute for Rural Engineering, Water and Forestry (INRGREF), Tunis, Tunisia

Ezzeddine Mahmoudi:
University of Carthage, Bizerte, Tunisia

Marco Moriondo:
Institute of BioEconomy, Italian National Research Council (CNR-IBE), Florence, Italy

Olivier Pringault:
Mediterranean Institute of Oceanography (MIO), Institute of Research for Development (IRD), Marseille, France

Gil Rilov:
National Institute of Oceanography, Israel Oceanographic and Limnological Research (IOLR), Haifa, Israel

Samuel Somot:
Centre National de Recherches Météorologiques (CNRM, Université de Toulouse, Météo-France, CNRS), Toulouse, France

Athanossios Tsikliras:
Aristotle University of Thessaloniki, Thessaloniki, Greece

Monsterrat Vila:
Spanish National Research Council, Doñana Biological Station (EBD-CSIC), Sevilla, Spain

Georgios Zittis:
The Cyprus Institute, Nicosia, Cyprus

Contributing Authors

Giovanni Argenti:
University of Florence, Florence, Italy

Marie-Anne Auger-Rozenberg:
Orléans Forest Zoology research unit, French National Research Institute for Agriculture, Food and Environment (INRAE), Orléans, France

Ernesto Azzurro:
Institute for Marine Biological Resources and Biotechnology of the National Research Council (CNR-IRBIM), Ancona, Italy

Corina Basnou:
CREAF, Bellaterra (Barcelona), Spain

Sophie Bastin:
Laboratoire ATmospheres Milieux Observations Spatiales/Institut Pierre Simon Laplace (LATMOS/IPSL), UVSQ Université Paris-Saclay, SU, CNRS, Guyancourt, France

Mustapha Béjaoui:
University of Carthage, Bizerte, Tunisia

Lorenzo Brilli:
Institute of BioEconomy, Italian National Research Council (CNR-IBE), Florence, Italy

Martina Carrete:
University Pablo de Olavide, Sevilla, Spain

Emma Cebria:
Blanes Centre for Advanced Studies (CEAB), Spanish National Research Council (CSIC), Blanes, Girona, Spain

Hanene Chaabane :
Laboratory of Pest control and Integrated Protection in Agriculture / LMI NAILA, National Institute of Agronomy of Tunisia (INAT), University of Carthage, Tunis, Tunisia

Sílvia Coelho:
Centre for Environmental and Marine Studies (CESAM), University of Aveiro, Aveiro, Portugal

Renato Colucci:
Institute of Polar Sciences, Marine Science Institute, National Research Council (CNR-ISP), Trieste, Italy

Styliani Dafka:
Justus-Liebig University, Giessen, Germany

Sofia Darmaraki:
Dalhousie University, Canada

Camilla Dibari:
University of Florence, Florence, Italy

Donna Dimarchopoulou:
Aristotle University of Thessaloniki, Thessaloniki, Greece

Jean-Claude Dutay:
Laboratory of Climate and Environmental Sciences (LSCE), Pierre Simon Laplace Institute (IPSL), Gif-sur-Yvette, France

Monia El Bour:
National Institute of Marine Sciences and Technology (INSTM), Carthage - Salammbô, Tunisia

Antonietta Elia:
University of Santiago de Compostela, Santiago de Compostela, Spain

Elena Georgopoulou:
National Observatory of Athens, Athens, Greece

Sylvaine Giakoumi:
Zoological Station Anton Dohrn, Naples, Italy & School of Biological Sciences, University of Queensland, Australia

Juan Jesus Gonzalez Aleman:
Complutense Universiy of Madrid, Madrid, Spain

Pablo González-Moreno:
CABI, Egham, United Kingdom

Madeleine Goutx:
Mediterranean Institute of Oceanography (MIO), CNRS, Marseille, France

Olivier Grunberger:
National Agricultural Institute of Tunisia (INAT), French National Research Institute for Sustainable Development (IRD), Tunis, Tunisia
Ivan Güttler:
Croatian Meteorological and Hydrological Service, Zagreb, Croatia
Nathalie Hilmi:
Monaco Scientific Centre (MSC), Monaco
Gabriel Jorda:
Spanish Institute of Oceanography, Balearic Islands Oceanographic Centre (COB-IEO), Palma, Spain
Stelios Katsanevakis:
University of the Aegean, Mitilini, Greece
Mehdi Lahlou:
National Institute of Statistics and Applied Economics (INSEA), Rabat, Morocco
Manfred A. Lange:
The Cyprus Institute, Nicosia, Cyprus
Luisa Leolini:
University of Florence, Florence, Italy
Myriam Lopes:
Centre for Environmental and Marine Studies (CESAM), University of Aveiro, Aveiro, Portugal
Annarita Mariotti:
National Oceanic and Atmospheric Administration (NOAA), Silver Spring, United States of America
Ana Isabel Miranda:
Centre for Environmental and Marine Studies (CESAM), Department of Environment and Planning, University of Aveiro, Aveiro, Portugal
Meryem Mojtahid:
University of Angers, Angers, France
Alexandra Monteiro:
Centre for Environmental and Marine Studies (CESAM), University of Aveiro, Aveiro, Portugal
Samuel Morin:
Université Grenoble Alpes, Université de Toulouse, Météo-France, CNRS, CNRM, Centre d'Études de la Neige, Grenoble, France
Pierre Nabat:
Centre National de Recherches Météorologiques (CNRM, Université de Toulouse, Météo-France, CNRS), Toulouse, France
Anika Obermann-Hellhund:
Goethe University, Frankfurt am Main, Germany
Tuğba Öztürk:
Işık University, Istanbul, Turkey
Androniki Pardalou:
Aristotle University of Thessaloniki, Thessaloniki, Greece
Sandra Rafael:
Centre for Environmental and Marine Studies (CESAM), University of Aveiro, Aveiro, Portugal
Francesca Raffaele:
Abdus Salam International Centre for Theoretical Physics (ICTP), Trieste, Italy
Lena Reimann:
Christian-Albrechts University, Kiel, Germany
Alain Roques:
Orléans Forest Zoology research unit, French National Research Institute for Agriculture, Food and Environment (INRAE), Orléans, France
Asma Sakka Hlaili:
University of Carthage, Bizerte, Tunisia
Alberto Santini:
Institute for Sustainable Plant Protection, Italian National Research Council (CNR), Sesto Fiorentino, Italy
Giuseppe Scarcella:
National Research Council, Institute of Marine Biological Resources and Biotechnologies (CNR-IRBIM), Ancona, Italy
Katrin Schroeder:
Italian National Research Council, Institute of Marine Science (CNR-ISMAR), Venice, Italy
Isla Simpson:
Climate and Global Dynamics Laboratory of the National Center for Atmospheric Research in Boulder, Colorado, United States of America
Nicolina Staglianò:
University of Florence, Florence, Italy
Meryem Tanarhte:
Hassan II Mohammedia University, Casablanca, Morocco
Rob Tanner :
European and Mediterranean Plant Protection Organisation (EPPO), Paris, France
Rémi Thiéblemont:
French Geological Survey, Orléans, France
Yves Tramblay:
HydroSciences Montpellier (University Montpellier, CNRS, IRD), Montpellier, France
Marco Turco:
University of Murcia, Murcia, Spain
Athanasios Vafeidis:
Christian-Albrechts University, Kiel, Germany
Martin Wild:
Institute of Atmospheric and Climate Science, ETH Zürich, Zürich, Switzerland
Elena Xoplaki:
Department of Geography & Center for international Development and Environmental Research, Justus Liebig University, Giessen, Germany
Argyro Zenetos:
Institute of Marine Biological Resources and Inland Waters, Hellenic Centre for Marine Research (HCMR), Argiroupoli, Greece