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Review

Biodiversity impacts by multiple anthropogenic stressors in Mediterranean coastal wetlands

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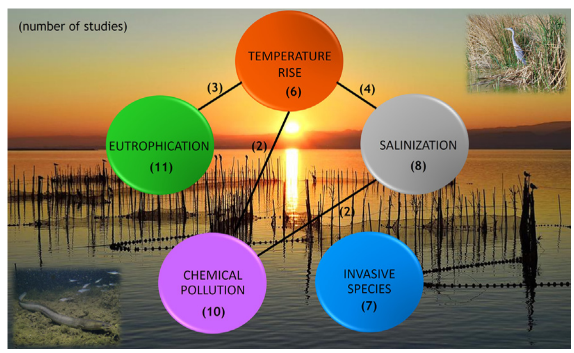
HIGHLIGHTS

- We reviewed 54 studies describing biodiversity impacts in Mediterranean coastal wetlands.
- Eutrophication and chemical contamination effects have been largely studied.
- Studies assessing interactions between more than one stressor are very limited.
- Non-additive responses to multiple stressors are frequently observed.
- A holistic approach is needed to define appropriate ecosystem management measures.

GRAPHICAL ABSTRACT



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ABSTRACT

Mediterranean coastal wetlands are considered biodiversity hot-spots and contain a high number of endemic species. The biodiversity of these ecosystems is endangered by several pressures resulting from agricultural and urban expansion, climate change, and the alteration of their hydrological cycle. In this study we assess the state-of-the-art regarding the impact of several stressor groups on the biodiversity of Mediterranean coastal wetlands (*i.e.*, lagoons, marshes, estuaries). Particularly, we describe the impacts of eutrophication, chemical pollution, invasive species, salinization, and temperature rise, and analyze the existing literature regarding the impact of multiple stressors on these ecosystems. Our study denotes a clear asymmetry both in terms of study areas and stressors evaluated. The majority of studies focus on lagoons and estuaries of the north-west parts of the Mediterranean basin, while the African and the Asian coast have been less represented. Eutrophication and chemical pollution were the most studied stressors compared to others like temperature rise or species invasions. Most studies evaluating these stressors individually show direct or indirect effects on the biodiversity of primary producers and invertebrate communities, and changes in species dominance patterns that contribute to a decline of endemic populations. The few available studies addressing stressor interactions have shown non-additive responses, which are important to define appropriate ecosystem management and restoration measures. Finally, we propose research needs to advance our understanding on the impacts of anthropogenic stressors on Mediterranean coastal wetlands and to guide future interventions to protect biodiversity.

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Contents

1. Introduction

Mediterranean coastal wetlands (lagoons, marshes and estuaries) are included in the category of transitional waters, and form part of a particular ecotone which shares features of freshwater and marine ecosystems (Pérez-Ruzafa et al., 2011). They are characterized by the occurrence of large gradients of abiotic variables (*e.g.* oxygen, pH, salinity) and are considered biodiversity hotspots, including a wide range of migratory birds and endemic species (Perennou et al., 2018; Sala et al., 2000). Moreover, they host a large primary productivity and have a high capacity for nutrient cycling and carbon sequestration (Herbert et al., 2015), thus having a significant influence on global climate regulation (Camacho et al., 2017; Morant et al., 2020).

During the last decades, Mediterranean coastal wetlands have been impacted by several anthropogenic stressors or stressor groups (hereafter stressors). As a result, the abundance of wetland-dependent species has declined (−15% between 1990 and 2013), and it is estimated that about 36% of the species that inhabit these ecosystems today are threatened with extinction (Perennou et al., 2018). The biodiversity decline in Mediterranean coastal wetlands is intrinsically related to the expansion of urban areas and agriculture (Newton et al., 2018; Perennou et al., 2018), which have resulted in the emission of high nutrient loads that contribute to eutrophication (Pérez-Ruzafa et al., 2019). Intensive agriculture and urbanization are also characterized by the emission of a wide range of contaminants, including for example pesticides, pharmaceuticals, persistent organic pollutants or heavy metals (Ccanccapa et al., 2016; Sadutto et al., 2021), which are potentially toxic to aquatic organisms.

Another environmental problem affecting coastal wetlands in the Mediterranean zone is water scarcity, which is a result of climate change and/or the re-direction of water flows for agriculture or urban supply (García de Jalón et al., 2014; Mekonnen and Hoekstra, 2016; Perennou et al., 2018). Water scarcity can lead to habitat alteration, increasing turbidity (González-Ortegón et al., 2015), and lowers the dilution potential of contaminants (Arenas-Sánchez et al., 2016). Furthermore, water scarcity has been considered as the main cause of salinity intrusion in coastal wetlands (Herbert et al., 2015), which can cause osmotic stress to aquatic organisms and habitat deterioration (Anton-Pardo and Armengol, 2012).

The proliferation of Invasive Alien Species (IAS) represents another major threat to the biodiversity of Mediterranean coastal wetlands (Taylor et al., 2021). The Mediterranean basin is one of the areas where the ecological consequences caused by this phenomenon are expected to be specially harmful due to its biogeographical characteristics, including periods of habitat fragmentation and isolation of aquatic ecosystems, and convergent evolutionary processes (Sala et al., 2000). The ecological impacts caused by IAS often go beyond a loss of local taxa, and can affect regional biodiversity as well as ecosystem services (Lastrucci et al., 2018).

The Mediterranean region has been identified as one of the most vulnerable regions to the impacts of global climate change. The models issued by the International Panel on Climate Change forecast an increase in average temperatures in the range of 2.2 to 5.1 °C for the period 2080–2100 (IPCC, 2014). Wetlands in this region are expected to be highly influenced by climate change as they are relatively shallow, and have limited capacity for stratification and for buffering the impact of increasing air temperatures (Vidussi et al., 2011). Moreover, climate change co-occurs with other anthropogenic disturbances or stressors (*e.g.* eutrophication, contamination, water scarcity and salinization; Ficke et al., 2007), which potentially exacerbate their ecological effects (Moe et al., 2013).

In general, research on the interactive effects of multiple stressors has been performed more frequently in lotic ecosystems than in lentic ones, and has focused on aquatic ecosystems of central and northern Europe (Karaouzas et al., 2018; Mondy et al., 2016; Piggott et al., 2012). Understanding and predicting the impact that stressor combinations may have on the structure and functioning of Mediterranean coastal wetlands is crucial to ensure their long-term sustainability as well as to meet the quality standards set by the international regulatory frameworks that protect them, *e.g.* the International Ramsar Convention (Ramsar Convention Secretariat, 2010) or the Water Framework Directive for European countries (European Commission, 2000).

Therefore, the aim of this study was to describe the state-of-the-art regarding the impact of different anthropogenic stressors on the biodiversity of Mediterranean coastal wetlands, and to characterize their potential interactive effects on relevant biological endpoints. Despite the number of stressors affecting freshwater biodiversity may be large, here we focused on five stressor groups (*i.e.*, eutrophication, chemical pollution, invasive species, salinization and temperature rise) that have been considered major drivers of ecological change during the last years (Pérez-Ruzafa et al., 2019; Taylor et al., 2021). To characterize the impacts of these stressors on biodiversity, we performed a review of field monitoring studies, micro- and mesocosm experiments, and laboratory tests performed with species characteristic of Mediterranean coastal wetlands. From these studies, we evaluated the different stressor's contribution to physiological changes in organisms, as well as structural changes in populations and communities. Finally, we propose research needs to advance our understanding on the impacts of single and multiple stressors on Mediterranean coastal wetlands, and to define management measures to protect biodiversity.

2. Methods

A literature search was performed including all available studies published up to the 8th of July of 2021 using the Web of Science (WoS) platform (with all major databases including in it). The literature search included a variety of search strings with the terms: chemical pollution, eutrophication (and algal blooms), invasive species, salinization, temperature increase, Mediterranean, coastal lagoons, wetlands, and estuary. The full list of search terms used in this study can be found in the Supplementary Material (Table S1). The term "review" was also added in combination with the above search terms to confirm that there were no previous reviews on this or similar topics. We also tested the inclusion of country names in the search but did not provide additional results.

Our article search yielded 557 publications, which were screened regarding their geographical scope, the biological assessment and the experimental design, following the ROSES approach described by Haddaway et al. (2018), see Fig. S1. Here we only included studies performed under conditions that represent Mediterranean wetlands, such as freshwater and brackishwater lagoons, marshes and estuaries, which were located near the coastline and have a slight sea influence. Our study, thus, excluded monitoring studies performed in lotic ecosystems and transition zones with very limited coastal influence, as well as marine studies. We only included studies in which there was a quantitative assessment of one or several of the evaluated stressors (*i.e.*, eutrophication, chemical pollution, invasive species, salinization, and temperature rise) on relevant biological endpoints, including those that resulted in stress reduction or biodiversity remediation. We differentiated the studies that had been performed on low levels of biological organization (*i.e.*, impacts on individuals or populations of single species) from those that followed an integrative approach and that assessed impacts on species assemblages or communities considering species interactions. Regarding the type of experimental design, we included field monitoring studies, which often included a gradient of stressors, and micro- or mesocosm experiments and laboratory experiments, which were often based on a limited number of stressor levels. We only considered impacts that were evaluated after the application of an environmentally realistic stressor level. Regarding multiple stressors, we classified all studies that evaluated the impact of several co-occurring stressors on biological endpoints, but we only discussed in detail those that used a factorial design and that allowed the classification of multiple stressor effects into additive, synergistic or antagonistic, following the approach described by Piggott et al. (2015).

3. Biodiversity impacts

We found 54 studies that matched the above-mentioned screening criteria and that were included in this review. The number of studies assessing the impact of anthropogenic stressors in Mediterranean coastal wetlands has almost doubled during the last five years (Fig. 1), particularly in some coastal areas of Europe (Fig. 2). The coast of France (specifically the Camargue region) and the Iberian Peninsula were the areas with the largest number of published studies (16 each), followed by Italy (13). We found only five studies performed in the coast of Africa (Tunisia and Egypt) and four in the coast of Asia (Turkey and Israel; Fig. 2). Most studies focused on chemical pollution (15) and eutrophication (14), followed by salinization (8), invasive species (7) and temperature increase (6). The impact of these stressors on Mediterranean coastal wetlands are described in the following sub-sections.

3.1. Eutrophication

Most studies addressing eutrophication are based on field measurements of phytoplankton dynamics as response to nutrient loads (Table S3). For instance, Acquavita et al. (2015) assessed the relationship between nutrient inputs (up to 368 μM of NO_3^-, 12.2 NO_2^- and 26.4 NH_4^+ coming from hydric regulation and fish farm discharges) and chlorophyll-a concentrations in the Marano and Grado coastal lagoons (Italy), as well as the occurrence of macroalgae blooms. Quintana and Moreno-Amich (2002) conducted a field survey in

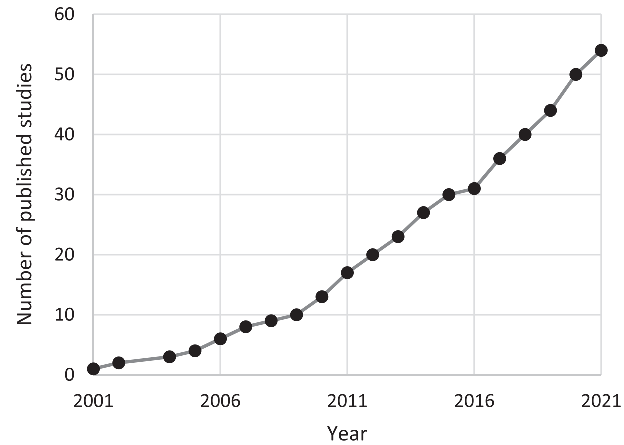

Fig. 1. Temporal dynamics of the number of studies that have been published on the impact of anthropogenic stressors on Mediterranean coastal wetlands.

temporary wetlands of Empordà (Spain) with different trophic status (based on different biomass levels of phytoplankton groups) and during periods of "flux regulation" (*i.e.* consisting in freshwater inputs from nearby croplands). They took samples of phytoplankton and invertebrates (*i.e.* mainly zooplankton followed by a lesser proportion of benthic amphipods and insects) and observed higher chlorophyl-a values, phytoplankton biovolume, and mixotrophic species at higher levels of eutrophication, in addition to a high phytoplankton/invertebrates ratio. Le Fur et al. (2019) performed a long-term evaluation of 21 lagoons of the Mediterranean coast of France. They show how the communities of primary producers (*i.e.* phytoplankton, macroalgae and seagrasses) varied after specific management measures (which focused on diminishing N and P inputs). After seven years of monitoring, the primary producers suffered a progressive shift in their species composition, which started from phytoplankton-dominated communities in the hypereutrophic state to a macrophyte dominance – including perennial macroalgae communities – under oligotrophic-like conditions. Other two monitoring studies were carried out in the Biguglia coastal lagoon (Italy). Garrido et al. (2016) highlighted the occurrence of micro-nanoplankton blooms dominated by the dinoflagellate *Prorocentrum minimum*, as a consequence of freshwater and sewage inputs with ammonia concentrations up to 26.9 μmol/L. On the other hand, Pasqualini et al. (2017) described phytoplankton blooms dominated by the cyanobacteria *Anabaenopsis circularis* as a result of a dystrophic crisis in the previous decade because of the high nutrient inputs and the decrease of salinity. These authors noted an inverse correlation between chlorophyll-a concentrations (as a proxy of phytoplankton biomass) and the biomass of macroalgae, while demonstrated little influence of phytoplankton on the biomass of the submerged macrophyte angiosperms.

Rodrigo et al. (2018) evaluated the capacity of constructed wetlands (dominated by the herbaceous plant *Iris pseudacorus* and the common reed *Phragmites australis*) as a nature-based solution to reduce the high nutrient concentrations of the Albufera lagoon (Spain). All plots with this vegetation resulted in a marked reduction of the microalgae biomass downstream, including cyanobacteria responsible for toxic algal blooms. In addition, the zooplankton community (metazooplankton, unless specified) showed a notable biodiversity increase (particularly Ostracoda and Cladocera). Another study carried out with aquatic vegetation is the one performed by Eid et al. (2021) in the Burullus lake (Egypt), who evaluated the influence of large nutrient inputs coming from agriculture and fish farming on physiological parameters (*i.e.* shoot height, stem diameter, number of leaves, etc.) of the common reed (*P. australis*).

The impacts of eutrophication on the biodiversity of aquatic animals has also been described. Boix et al. (2007) performed a field sampling of

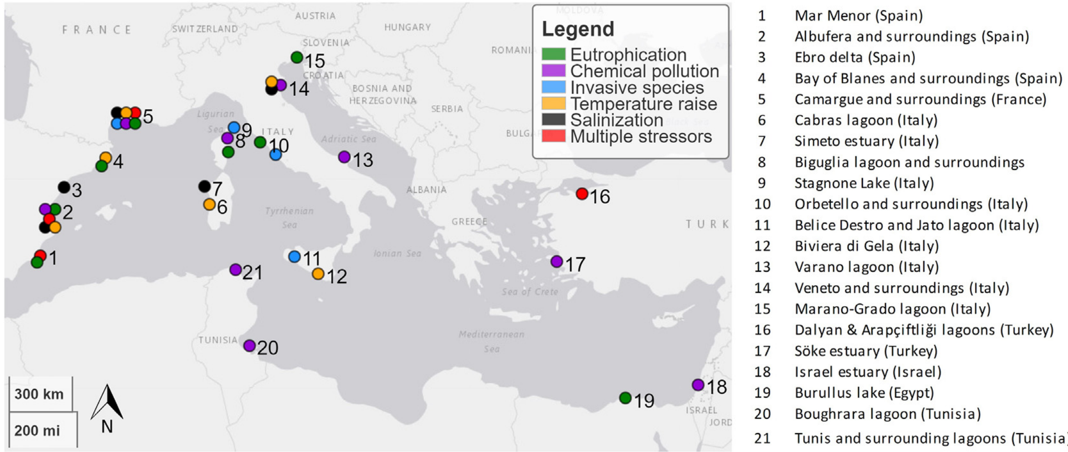

Fig. 2. Map showing the location of the reviewed studies. Nearest dots represent the same study area addressed in papers evaluating different focal stressors. Colors show the focal stressors addressed in each study area (see legend). The full list of references per location are provided in the Supplementary Material (Table S2). The map has been created following Esri (2021).

macroinvertebrates communities in wetlands of five different trophic categories. The results of their study showed that in highly eutrophic wetlands, crustacean populations exhibited an increase of Branchiopoda species in detriment of Copepoda. In the case of insects, it was found a lower presence of Diptera and especially Coleoptera at high levels of eutrophication (*i.e.* waterbodies with 110 μM NO_3^- and 6 μM PO_4^{3-}), which corresponded to an overall reduction of species richness (from 171 to 53). Lloret and Marín (2011) assessed the benthic fauna of the Mar Menor lagoon (Spain), a semi-enclosed coastal lagoon affected by large nutrient inputs mainly coming from surrounding agriculture. These authors noted that eutrophication resulted in a biomass increase of suspension feeders, which modulated the proliferation of phytoplankton. On the other hand, Lardicci et al., (2001) evaluated a macroinvertebrate data series (1976–1999) corresponding to the Orbetello lagoon (Italy) and observed that the communities during the years of a dystrophic crisis were characterized by a high abundance of opportunistic species on the detriment of polychaeta and gastropods. Finally, Mouillot et al. (2005) carried out a sampling of macrophytes, fish and benthic communities in three wetlands of southeastern France (Ingril, Prevost and Mauguio) affected by different levels of eutrophication (established on the basis of dissolved oxygen, and concentrations of N, P and chlorophyll-a). While zoobenthos richness seemed to be inversely correlated with increasing eutrophication, the fish community was not severely affected by this condition. On the other hand, macrophyte communities showed a higher number of green (Ulvaceae) and red (*Gracilaria* sp.) algae in the wetland with the highest eutrophication level, while these communities of primary producers in the less eutrophic waters showed a higher taxonomic divergence.

The influence of nutrient loads has also been investigated making use of model ecosystem experiments (*i.e.*, micro-/mesocosms) and factorial test designs. Ferriol et al. (2017) evaluated the effects of different nutrient levels of N (1–10 mg/L of $NaNO_3$) and P (0.1–1 mg/L of KH_2PO_4) on a phytoplankton community using outdoor mesocosms in the Marjal dels Borrons (Xeresa wetlands, Spain). These authors noted an increased growth of several phytoplankton genera (*e.g. Pseudanabaena*, *Planktolyngbya*, *Monoraphidium*, and *Chlorella*) in the mesocosms with highest nutrient inputs, and a decline of macrophytes (*Chara* sp.), corresponding with a high concentration of ammonia. In the same study area, Romo et al. (2004) assessed the effect of N and P addition (up to 10 mg/L for N and 1 mg/L for P) together with planktivorous fishes (0, 4, and 20 g/m^2) on plankton communities over two sampling campaigns. They described an increase in algal biomass in the mesocosms exposed to higher nutrient levels, but a lower biodiversity due to a dominance of the cyanobacterium *Chroococus* sp. Similarly, the zooplankton biomass increased in the systems with higher nutrients, but the dominance of some taxa like cyclopoids, Rotifera or Cladocera was negatively affected by the interaction between nutrients and fish due to the enhancement of top-down effects.

Leruste et al. (2019) carried out an experiment in which phytoplankton communities were incubated in the laboratory under N (20 μM) and P (0.8 μM) enrichment conditions. They found that high inputs of ammonia during summer favored the replacement of bigger phytoplankton species by others with lower cell size (picophytoplankton), which have a preference for these chemical forms (*i.e.*, higher N assimilation rates) in contrast to diatoms and dinoflagellates, which prefer N uptake in the nitrate form. They also found that dinoflagellate blooms occurred mainly during wet seasons (in moments of P limitation).

Fig. 3 shows a summary of the biodiversity impacts caused by eutrophication in the reviewed studies. In the case of phytoplankton, the results found by the different studies are rather consistent, indicating a general increase of phytoplankton abundance for most groups (except for Diatoms). The response of macrophytes shows contrasting results, which could probably be due to the initial species composition and the eutrophication level evaluated in the different studies. However, in systems impacted by very high nutrient levels, rooted macrophytes tend to be replaced by a phytoplankton dominated community. Regarding consumers, the literature shows a negative relationship between eutrophication and diversity. Under nutrient rich conditions, zooplankton generally exhibited an increase in the abundance of Branchiopoda, but some sensitive taxa within this group, such as Cladocera, showed a decrease. Regarding benthic invertebrates, the literature shows a trend towards the decrease of some predators (*e.g.* Coleoptera). Other groups like herbivore-detritivore ones (*e.g.* gastropods and polychaeta) show contrasting results due to differences in trophic status, showing a decline highly eutrophicated systems.

3.2. Chemical pollution

One of the main sources of chemical pollution in Mediterranean coastal wetlands is intensive agriculture, and particularly rice cultivation in the surroundings of coastal lagoons, marshes and estuaries. Barbieri et al. (2020) identified 35 different pesticides in water samples taken in the Ebro River Delta (Spain) during the rice cultivation period, with 17 of those being currently banned in Europe. In a similar study, Calvo et al. (2021) identified 21 different pesticides in water and sediment samples collected in the Albufera Natural Park (Spain). Both studies have shown environmental concentrations that are above safe limits for aquatic organisms (including algae, invertebrates, and fish),

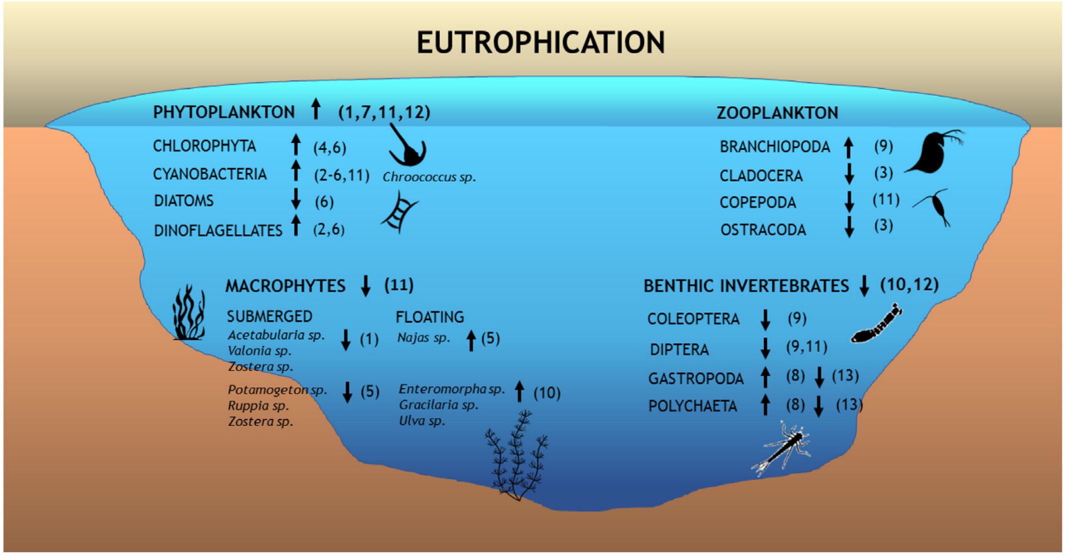

Fig. 3. Scheme showing the influence of eutrophication on different aquatic organisms of Mediterranean coastal wetlands. Arrows indicate an abundance increase or decrease caused by eutrophication. References: (1) Rhazi et al. (2009); (2) Leruste et al. (2019); (3) Rodrigo et al. (2018); (4) Ferriol et al. (2017); (5) Pasqualini et al. (2017); (6) Garrido et al. (2016); (7) Acquavita et al. (2015); (8) Lloret and Marín (2011); (9) Boix et al. (2007); (10) Mouillot et al. (2005); (11) Romo et al. (2004); (12) Quintana and Moreno-Amich (2002); and (13) Lardicci et al. (2001).

and have highlighted the potential synergistic effect of mixtures containing substances with different toxicological mode of action. Another important source of chemical contamination is the release of treated or untreated wastewater. Sadutto et al. (2021) assessed the risks of pharmaceuticals and wastewater-based contaminants in the Albufera Natural Park (Spain) and identified a high toxicity potential caused by caffeine for algae and the analgesic tramadol to all aquatic taxa (Table S4).

The afore-mentioned studies assessed the risks for aquatic biodiversity using risk quotients calculated with monitored exposure concentrations of contaminants and laboratory toxicity obtained for standard test species. Only few studies have evaluated the impacts of chemical pollution on local biodiversity, either under laboratory conditions or in the field. These studies have focused on assessing the ecological impacts of non-point chemical pollution such as Organochlorine Pesticides (OCPs), Polychlorinated Biphenyls (PCBs) and Polycyclic Aromatic Hydrocarbons (PAH) on fish. For instance, Barhoumi et al. (2014) assessed the effects of OCPs, PCBs, PAHs and metals (As, Cr, Cu, Mn, Pb, V and Zn) on the expression of some biomarkers in the grass goby (*Zosterisessor ophiocephalus*) by sampling them in a coastal lagoon from Tunisia affected by intensive agriculture, urban runoff and sewage and industrial effluents. The results showed a positive correlation between enzymes of liver damage and tissue levels of pesticides, as well as neurotoxic effects. In the same area, Mansour et al. (2020) show how clams (*Ruditapes decussatus*) from polluted sites exhibit a higher lysozyme activity and total hemocyte count, and a decrease in phenoloxidase activity. Bejaoui et al. (2020) also in Tunisia lagoons, assessed the response in gills and digestive tissues of the clam *Verenupis decussata* under the exposure of some trace metals (Al, Cd, Cu, Fe, Ni and Zn). The results showed an increment of antioxidative responses and malondialdehyde activities as well as necrosis and atrophy in tissues. Kocagöz et al. (2014) provides one of the few field sampling studies that investigates the potential bioaccumulation in liver and muscle and effects of OCPs, PCBs and Polybrominated Diphenyl Ethers (PBDEs) in waterbirds, and found high levels of antioxidant enzymes and mithocondrial superoxide dismutase (SOD, an oxidative stress enzyme) correlating to contaminant concentrations in liver tissue. Pitacco et al. (2020) performed a field sampling of sediment and zoobenthic communities at Sacca di Goro coastal lagoon (Italy), polluted with OCPs, PCBs as well as other persistent contaminants and trace elements. They detected a clear impact on the benthic invertebrate community, which was dominated by tolerant and opportunistic species (mainly annelids), and showed a decrease on sensitive taxa (*i.e.*, amphipods). Recently, Vilas-Boas et al. (2021) assessed the effects of an environmentally relevant concentration of the insecticide chlorpyrifos on zooplankton communities collected from the Albufera lagoon (Spain) using indoor microcosms, and observed a significant decline of Cladocera accompanied by an increase of Copepoda and some Rotifera taxa.

Other studies have assessed the impacts of contaminants on reproductive parameters of invertebrates and fish. For instance, Fabbrocini et al. (2017) conducted a bioassay consisting in a sperm motility-test (MOT-test) to assess the ecotoxicological effects of PAHs, PCBs and metals (As, Cr, Cu, Hg, Ni, Pb and Zn) occurring in the sediments of the Varano coastal lagoon (Italy) on the sea urchin *Paracentrotus lividus*. After 1 h of exposure to polluted water they observed a decrease in curvilinear velocity (VLC), which refers to sperm velocity. Zoller (2006) performed an exposure experiment with zebrafish *Danio rerio*, an exotic species used as model organism, in an estuary ecosystem to test the effect of Alkylphenol ethoxylates (APEOs) and showed up to 90% reduction in egg production after 20 days of exposure. On the other hand, Galgani et al. (2006) developed a field survey to test the effects of PCBs, PAHs, DDT and metals (As, Cd, Co, Cr, Cu, Hg, Ni, Pb and Zn) occurring in sediments from three coastal lagoons of Corsica in oyster (*Crassostrea gigas*) embryos. Their results showed abnormalities in up to 28% of the embryos exposed to environmental concentrations.

The application of insecticides to control mosquitoes in the surroundings of urban areas in the Mediterranean coast is a widespread practice. Some studies have evaluated the impact of these practices on wetland biodiversity. For example, Fayolle et al. (2015) assessed the effects of the insecticide Bti applied at the recommended doses (0.29–1.24 L/ha) in phytoplankton communities on temporary wetlands in the Rhone Delta (France). In this case, they did not identify insecticide-related effects. In a mesocosm study performed in the same area, Duchet et al. (2015) assessed the impacts of Bti and Spinosad, another insecticide commonly used in mosquito pest control. They could not demonstrate either an effect of the Bti treatment, but the number of emerging chironomid larvae were significantly reduced by Spinosad applied at concentrations that resemble recommended doses for mosquito control (8, 17 and 33 μg/L).

3.3. Invasive species

Several of the species listed within the IAS of concern for the European Union (UE, 2014) are well distributed within Mediterranean coastal wetlands, as well as others which have been less researched. IAS have been introduced by humans, either accidentally (escaping from farms, international travels, pets and ornamentals) or deliberately, to deal against other autochthonous species or for commercial interests. Several studies have focused on assessing the biological impacts of imported plant species (Table S5), mainly in Italy and along the Rhone delta. Lastrucci et al. (2018) showed that the spread of *Myriophyllum aquaticum*, a macrophyte imported from South-America, has resulted in the displacement of some native species (*i.e.*, Equisetales, Nymphaeales and Thyphaceae), and in a change of the arthropod species composition (decrease of Ephemeroptera and increase of insects like *Aedes albopictus* and *Culex* sp.). Rhazi et al. (2009) performed a microcosm experiment to evaluate the influence of the introduction of saltmarsh bulrush (*Bolboschoenus maritimus*) in wetlands of southern France. They found that *B. maritimus* resulted in a decrease of the sexual reproduction of the indigenous quillworts (*Isoetes setacea*) in those treatments where the hydrological characteristics of substrate were flooded or wet. A wider study was carried out by Foggi et al. (2011), who studied macrophyte succession in the Stagnone lake (Italy) by the analysis of an orthophoto series for a period of 18 years. During that period, the authors showed that the initial populations of invasive *Typha* sp. and *Phragmites australis* increased their distribution area, which inevitably led to the progressive regression of areas occupied by endemic species and, especially in the case of *P. australis*, there was shift towards an habitat that resembles marsh drylands or terrestrial ecosystems.

Several alien amphibian species have been introduced in Mediterranean coastal wetlands as a collateral effect of pet and nursery plants trade as well as by ships. Lillo et al. (2011) monitored the impact of the African clawed frog *Xenopus laevis* on native amphibians in coastal wetlands of Sicily (Italy) for four years. They found that, except in the case of the common toad, the rest of native amphibians suffered a progressive loss of density and reproduction rates with the establishment of this species, although neither niche overlapping nor predation by *X. laevis* could be identified.

Large crustaceans invasions such as the American crayfish (*Procambarus clarkii*) have been reported since the late 70s (Gherardi and Lazzara, 2006). Some studies show that this species has displaced native populations, and has reduced the abundance of other macroinvertebrates (*e.g. Chironomus riparius* and *Physa acuta*) (Correia et al., 2005). Furthermore, it is considered a vector for many pathogens (Geiger et al., 2005), which affect the native crayfish (by transmission of *Aphanomyces astaci*) and amphibians (by transmission of *Batrachochytrium dendrobatidis*) (Souty-Grosset et al., 2016). Poulin et al. (2007) analyzed the impact of the American crayfish (*P. clarkii)* on bittern populations (*Botaurus stellaris*) in the Rhone Delta wetlands and noted that male bitterns were more abundant in areas with a high abundance of American crayfish due to its use as a food resource. Also, Rodriguez-Perez et al. (2014) carried out a mesocosm experiment and a field survey to test the predation of *P. clarkii* over tadpoles and eggs of the European frog *Hyla meridionalis*, but did not identify a negative response.

Regarding invasive mammals, it stands out the presence of nutrias (*Myocastor coypus*) introduced due to their farming for fur trade and the subsequent escape or release. Marini et al. (2013) and Angelici et al. (2012) analyzed their impact on plant communities and waterbirds in a coastal lagoon of Italy alone and in combination with the invasive rat *Rattus norvegicus*, respectively. Marini et al. (2013) show that the occurrence of *M. coypus* led to physical alterations of the edaphic structure due to their digging behavior for feeding of plant roots, which promoted the decline of most of the characteristic species of the *Juncetalia maritimi* grasslands, a protected habitat type. As noted by Angelici et al. (2012), the predation of waterbird eggs by *R. norvegicus* favored the colonization of nests by *M. coypus*, thus causing damage in habitats besides affecting the reproductive success of waterbirds. Furthermore, it has been described that when more than one invasive species coexist, it is relatively common the appearance of facilitation between them (Angelici et al., 2012; Lastrucci et al., 2018).

3.4. Salinization

The majority of Mediterranean coastal wetlands are characterized by having moderate to high levels of salinity. However, alterations of the hydrological balance of these ecosystems by water regulation and droughts are expected to result in abrupt alterations of their salinity basal range. Here we refer to the ecological impacts created by such alterations, which are mainly of anthropogenic origin. The ecological effects of increasing salinity have mainly been studied on zooplankton and macroinvertebrates (Table S6), as they are highly sensitive to the variation in osmotic conditions. For instance, Waterkeyn et al. (2011) in a mesocosm experiment performed at Camargue wetlands showed that the conductivity of 0.5 and 1 mS/cm result in an increase of large branchiopods and copepods and lowers the density of other cladocerans (except for *Daphnia* sp.), while in the treatments with higher conductivity (2.5 and 5 mS/cm), these trends appeared to be reversed. The Cladocera responses observed in their study were notably different from those obtained by Anton-Pardo and Armengol (2012) and Waterkeyn et al. (2008), who describe a lower richness of Cladocera when salinity increases in brackishwater ponds and mesocosms, respectively. Vilas-Boas et al. (2021) tested the effect of salinity increase (5.5 mS/cm) on zooplankton communities from the Albufera lagoon (Spain) and observed an overall decrease in diversity and abundance. Regarding macroinvertebrates, Waterkeyn et al. (2008) observed that at 10 mS/cm there was a richness decline of about 50% in the Camargue wetlands. Muresan et al. (2020) sampled permanent coastal ponds with different degrees of salinity and noticed that communities in polyhaline regime had a lower taxonomic and functional diversity. Opposite patterns were observed by Prado et al. (2014) in three brackish lagoons which suffered continuous freshwater inputs from agricultural ditches, as well as a complete isolation of seawater for more than a decade. In this case, long-term changes in physiological requirements of the species led to a significant shift in the community structure from one containing mainly euryhaline and polyhaline taxa, to another resembling a community typical of freshwater environments.

Few studies have considered the effects of salinity on native plant species. Jaoudé et al. (2013) conducted a greenhouse experiment with two different populations and age ranges of the shrub *Tamarix africana*, collected from coastal lagoons and estuaries from Italy to assess how they responded to different flooding regimes, one of them enriched with salts. They observed that plants of the populations from the Baratz lake suffered in their twigs a decrease of almost 50% of their photosynthetic activity. The rest of the treatments did not show any response to saltwater, proving a specific adaptation of this species to salinized environments.

The studies describing the impacts of salinization in Mediterranean coastal wetlands generally show consistent effects (Fig. 3). Based on these results, increasing levels of salinity are expected to alter the habitat and the distribution of aquatic plants. Furthermore, it can contribute to decrease (at least temporarily) the diversity and biomass of macroinvertebrates, promoting an increase of r-strategist taxa and deposit-feeders. Regarding zooplankton, it is expected that increasing salinization reduces the abundance of Cladocera and gives opportunities for some copepod and rotifers to thrive (Fig. 4).

3.5. Temperature rise

Only 6 studies have tested the impacts of water temperature increase related to climate change in Mediterranean coastal wetlands

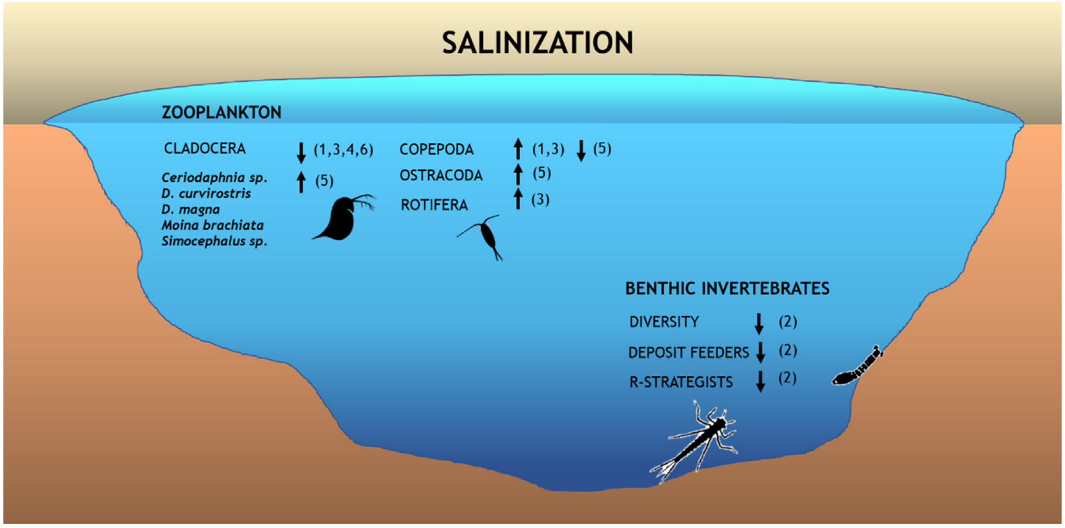

Fig. 4. Scheme showing the influence of salinization on different aquatic organism groups of Mediterranean coastal wetlands. Arrows indicate an abundance increase or decrease caused by salinization. References: (1) Vilas-Boas et al. (2021); (2) Muresan et al. (2020); (3) Anton-Pardo and Armengol (2012); (4) Waterkeyn et al. (2011); (5) (Waterkeyn et al., 2010); (6) Waterkeyn et al. (2008).

(Table S7). Some have focused on assessing the influence of temperature increase on the succession of planktonic communities using long-term monitoring data series. Barone et al. (2010) analyzed changes in temperature and precipitation patterns, and performed a monthly field sampling of phytoplankton during three years at Biviera di Gela lagoon (Sicily, Italy), which was compared with the phytoplankton communities monitored 20 years earlier. The results showed that in the last 40 years there was a mean temperature increase of 0.6 °C, and that communities suffered a shift in their species dominance, from Cryptophytes, diatoms and dinoflagellates to cyanobacteria species. Furthermore, it was observed a dominance of *Prymnesium parvum* in winter, which together with the cyanobacteria *Aphanizomenon ovalisporum*, probably led to the extinction of the copepod *Calanipeda aquaedulcis* (due to its sensitivity to the exotoxins emitted by the cyanobacteria). Moreover, the authors pointed at an increase of water temperature as the main cause for increasing the eutrophication status of the ecosystem.

The majority of available studies have assessed temperature variations using microcosm or mesocosm experimental set-ups. Vidussi et al. (2011) performed a mesocosm experiment combining both increasing water temperatures (+3 °C) and UV on the shore of the Thau lagoon (France). Warming resulted in a trophic cascade effect, triggered by the earlier development of copepods adult stages, which led to a decline of ciliates and an increase of the abundance of heterotrophic flagellates. Pulina et al. (2020) developed a microcosm experiment with three different temperature regimes (*in situ* temperature, +3 °C and +6 °C) to assess changes in plankton communities. At the end of the experiment, the cell density was higher at the warmest temperatures, in addition to the occurrence of changes in taxonomic composition of phytoplankton (Chlorophyceae dominance) and ciliates (*Cyclidium* sp. dominance). Overall, in all groups there was a reduction in cell size. It was unclear whether such an effect was a direct consequence of temperature regimes or an indirect effect caused by other factors like selective grazing over larger organisms. Vilas-Boas et al. (2021) compared adaptation and species succession patterns of zooplankton communities from the Albufera lagoon (Spain) at two different temperatures (20 °C and 30 °C). They found a higher individual abundance at high temperature, while some Rotifera (*Ascomorpha* sp.) and Cladocera (*Moina* sp.) showed a slight decline, probably by competition exclusion. Vázquez-Domínguez et al. (2012) conducted a microcosm experiment at Bay of Blanes (Spain) with two different temperature regimes (*in situ* and +2.7 °C) and focusing on the response of heterotrophic and microplankton communities. After one year, the warming treatment resulted in a 25% of increase of biomass and gross production of heterotrophic bacteria, but at the same time a 50% loss rate of heterotrophic bacteria due to grazing by nanoflagellates. Microplankton only showed a significant loss in their ciliate community. These increments in production solely at lower trophic levels is consistent with a top-down control by predators as well as the microbial food web functioning. Finally, Strain et al. (2017) developed a mesocosm experiment in saltmarshes with aquatic plant species (*Spartina* spp. and *Salicornia veneta*) subjected to ambient temperature and 2 °C increase. The tested species showed different responses to warming. While *Spartina* spp. suffered a biomass loss and a lower number of flowering stems, *S. veneta* revealed an ecological advantage with an increasing of 42% in biomass and plant height.

Similarly to eutrophication, the studies assessing temperature raise indicate an increase of primary productivity and significant changes in community structure among invertebrates. Temperature raise is expected to result in an increase of some phytoplankton taxa such as dinoflagellates, cyanobacteria and Chlorophyceae, while studies suggest an overall decrease of diatoms. On the other hand, zooplankton responses are expected to be taxon specific and to depend on competition patterns, while a clear pattern on the response of ciliates to this stressor has not been identified (Fig. 5).

4. Multiple stressors

Our review has identified 7 studies assessing multiple stressors effects on Mediterranean coastal wetlands (Table S8). However, only 5 of them included an experimental design capable of disentangling the individual effects caused by those and their interactions (Fig. 6), while the other two provide an overall description of biodiversity dynamics in the presence of eutrophication, temperature and salinity stress in two coastal lagoons of the Marmara Sea, Turkey (Akbulut and Tavşanoğlu, 2018), and in the Ghar el Mehl lagoon, Tunisia (Dhib et al., 2013).

Two studies assessed the interaction between stressors on populations and communities using microcosms. Duchet et al. (2010) assessed the effects of different insecticides (Bti and Spinosad) on the population density of microcrustacean *D. magna* using microcosms with different levels of salinization and water temperature in the Camargue (France). They found significant negative synergistic interactions between Spinosad and salinity, as well as between Spinosad and temperature. Vilas-Boas et al. (2021) assessed the combined effects of increasing temperature (20 °C and 30 °C), salinity (control and 2.5 g

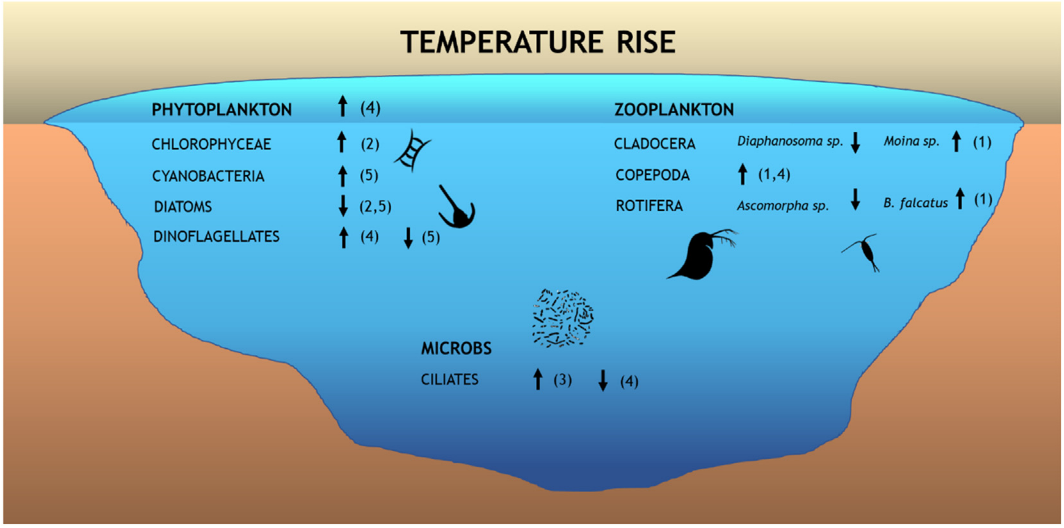

Fig. 5. Scheme showing the influence of temperature rise on different aquatic organism groups of Mediterranean coastal wetlands. Arrows indicate an abundance increase or decrease caused by temperature rise. References: (1) Vilas-Boas et al. (2021); (2) Pulina et al. (2020); (3) Vázquez-Domínguez et al. (2012); (4) Vidussi et al. (2011); (5) Barone et al. (2010).

NaCl) and the insecticide chlorpyrifos (control and 1 μg/L) on zooplankton from the Albufera lagoon (Spain). Their study showed non-additive interactions between the stress factors. For instance, the interaction between salinity and temperature reduced total abundance and was negative synergistic for Cladocera, suggesting that at higher temperatures the increase in metabolic activity exacerbates the impacts of salinity. They also identified positive antagonistic effects between salinity and chlorpyrifos on the most sensitive taxonomic group (Cladocera), potentially due a reduced bioavailability of the compound under salinity conditions. Finally, additive effects were found between temperature and the insecticide, although post-exposure recovery dynamics were slightly different between the different temperature scenarios.

Belando et al. (2019) studied laboratory cultures collected from the Mar Menor lagoon (Spain) to assess how different stochiometric

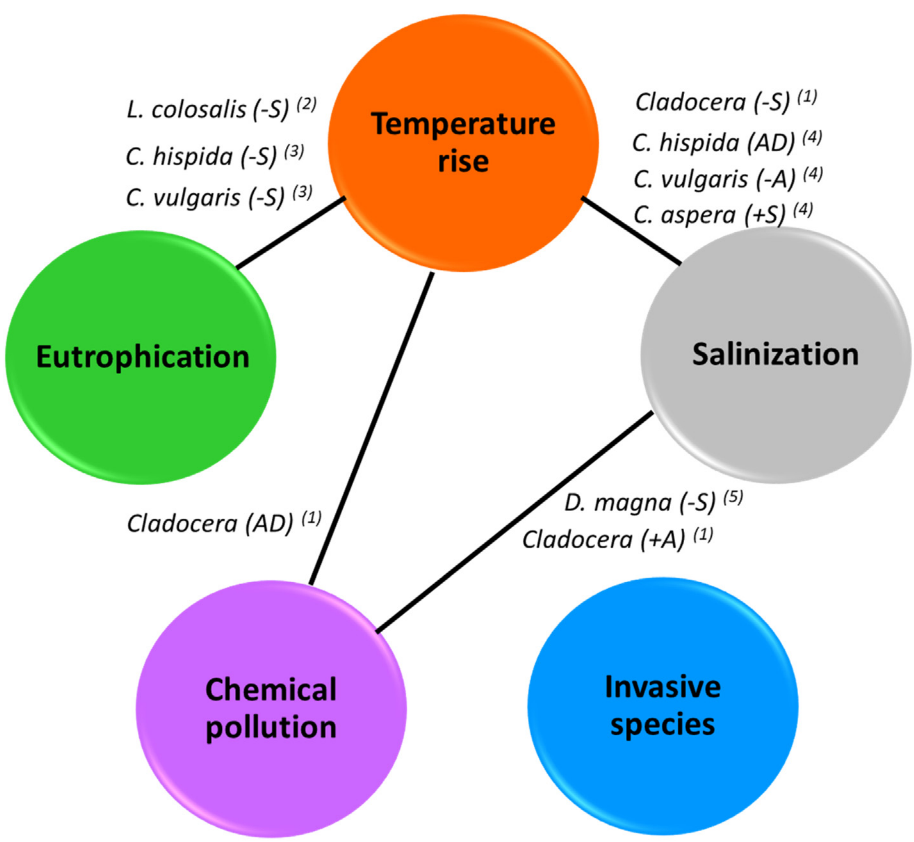

Fig. 6. Diagram representing the binary interactive effects between stressors found in the literature. Black lines indicate paired stressors interactions. Capital letters between brackets show the interaction type: A: antagonistic; S: synergistic; AD: additive; +: positive; −: negative. Stressor groups showing no lines mean that they have not been evaluated in combination with other stressors. The assessed endpoints were: physiological variables (growth-related) in *L. colosalis*, *C. aspera*, *C. hispida* and *C. vulgaris*; population density of *D. magna*; total abundance of Cladocera. References: (1) Vilas-Boas et al. (2021); (2) Belando et al. (2019); (3) Puche et al. (2018); (4) Rojo et al. (2017); (5) Duchet et al. (2010).

balances of nitrogen and phosphorous, in combination with three temperature regimes, affects some physiological variables of a native diatom species (*Licmophora colosalis*). Their results showed a negative synergistic interaction between warming and the stoichiometry N:P on most physiological endpoints. Also, the treatments with five degrees of increasing temperature (31 °C) and nutrient limitation (both N and P) resulted in a higher cell size. However, at high temperatures, the diatom mortality rates increased markedly. Another laboratory experiment was conducted by Puche et al. (2018) with charophyte species (*C. hispida* and *C. vulgaris*), and the combination of two nitrate concentrations (5 and 10 mg/L) and temperature regimes (20 and 24 °C). Their results suggest a negative synergistic interaction between temperature and nitrate increase on the uptake of N and the N:P ratio of the tested plants, but denoted a high phenotypic plasticity of charophytes from the coastal ecotype, which would benefit them in future scenarios of warming and nutrient enrichment. Rojo et al. (2017) also measured some morphological variables of charophytes (*C. aspera*, *C. hispida* and *C. vulgaris*) in a laboratory experiment at two temperature regimes (23 and 27 °C) and salinity (0.4 and 4 PSU). The interactions were additive for *C. hispida*, synergistic positive for *C. aspera*, and antagonistic negative for *C. vulgaris.* In addition, *C. aspera* increased their length, growth rate and dry weight under high temperature and salinity conditions, while these variables seemed to be diminished in *C. vulgaris* under high salinity conditions. *C. hispida* suffered a decreasing in its growth and development at all treatments, being the species most vulnerable to these scenarios. Overall, these studies show how mild variations of nutrient availability, temperature or salinity are expected to result in structural changes of primary producer communities, even for taxonomically similar species which had *a priori* similar environmental tolerance ranges.

5. Discussion and way-forward

Mediterranean coastal wetlands have an important ecological role and provide essential goods and services for humans. As shown by several monitoring studies, their ecological status may be affected by several pressures resulting from agricultural and urban expansion, climate change, and the alteration of their hydrological cycle. Eutrophication is considered to be one of the major threats affecting Mediterranean coastal lagoons and estuaries (Pérez-Ruzafa et al., 2019), and has been, together with chemical contamination, the most researched pressure. Most studies assessing the impacts of eutrophication denote species succession patterns occurring after heavy and sustained nutrient loads, which generally result in a reduction of biodiversity. Several studies assessing eutrophication effects have indicated the need to improve the sanitation system and the efficiency of WWTPs surrounding Mediterranean coastal wetlands, particularly in North Africa and Middle East (Perennou et al., 2018), and have proposed the use of natural and constructed wetlands as an efficient solution to reduce nutrient loads and as a remediation measure (Martín et al., 2020; Rodrigo et al., 2018). However, despite nutrient loads seem to be slightly reducing in some European wetlands (Perennou et al., 2018), further studies are needed to define nutrient carrying capacities and tipping points to revert the eutrophication status under the specific light intensity, temperature and seasonal biological dynamics of these ecosystems. This is particularly important, since there are indications that nutrient thresholds for maintaining the macrophyte-dominated status of ecosystems in the Mediterranean zone are different, and most likely lower, than those derived in temperate regions (Romo et al., 2004). Furthermore, research should be performed to distinguish the autochthonous plants that play a major role in nutrient cycling and that therefore should be the target of regional conservation policies (Rodrigo et al., 2013).

The current literature shows that Mediterranean coastal ecosystems are exposed to licit and illicit drugs (LIDs) (Vazquez-Roig et al., 2013), pharmaceuticals and personal care products (PPCPs) (Andreu et al., 2016; Vazquez-Roig et al., 2013; Carmona et al., 2014; Sadutto et al., 2021), trace metals (Andreu et al., 2016; Barhoumi et al., 2014; Galgani et al., 2006), microplastics (Simon-Sánchez et al., 2019), and a wide range of pesticides (Barhoumi et al., 2014; Belenguer et al., 2014; Duchet et al., 2015; Fayolle et al., 2015; Galgani et al., 2006; Kocagöz et al., 2014; Yalvaç and Taner, 2012). Moreover, due to their semi-confined nature, they can retain relatively high concentrations of these contaminants. The scientific literature has focused on describing the chemical pollution status of areas of high ecological value, such as the Ebro Delta or the Albufera lagoon (Spain), which contain complex mixtures of pesticides and pharmaceuticals in surface waters and sediments (Barbieri et al., 2020; Calvo et al., 2021; Sadutto et al., 2021; Vazquez-Roig et al., 2011). However, our knowledge on their net impacts on local and regional biodiversity is still relatively scarce. Although some studies have demonstrated that the enzymatic and morphological responses of fish and crustacean organisms can become suitable bioindicators of (re-mobilized) persistent pollutants (Authman, 2015; Bertrand et al., 2018), the number of studies assessing the resilience of populations and communities to emerging contaminants is low. The few available studies suggest that Mediterranean ecosystems have a different sensitivity and resilience to specific contaminant groups such as pesticides than the temperate ones (Daam et al., 2011; Rico et al., 2018). However, there are very few studies assessing the impacts of chemical pollution on autochthonous and endemic organisms under (semi-)natural conditions, so further investigations are needed to obtain more robust conclusions in this direction. Research should also be conducted to identify chemical mixtures that are potentially damaging ecosystems in different moments of the year and that should be the target of chemical monitoring and risk reduction programs. Chemical monitoring programs should include the evaluation of sediments, as they have been identified as major sinks for legacy and emerging contaminants and may constitute a potential contamination source in different moments of the year, *e.g.* after heavy storms or winds (Simon-Sánchez et al., 2019; Sadutto et al., 2021).

The introduction of invasive plants, amphibians, reptiles, crustaceans, and mammals is considered one of the key drivers of biodiversity loss and community structure modification in Mediterranean ecosystems. In some locations, the introduction of these species has been regarded as beneficial for fisheries, pest control, and tourism (Schüttler et al., 2011), neglecting the impacts that this may have for local and regional biodiversity (García-Llorente et al., 2008). The establishment of some invasive species has become so efficient that its eradication could yield a problem for native species that depend on them, complicating ecosystem management decisions. This is the case of the American crayfish (*P. clarkii*), which is widespread in most Mediterranean wetlands, and which constitutes an important food resource for top predators (*e.g.* birds), many of them in a condition of a declining population (Adrian and Delibes, 1987; Geiger et al., 2005; Poulin et al., 2007). Studies performed in other parts of the world suggest that alien species colonization and dispersion is notably influenced by habitat alteration produced by salinization (Herbert et al., 2015) or climate change (Carreira et al., 2017). However, this review shows that there have been very little efforts devoted to assessing how global climate change will influence the introduction and establishment of invasive species in coastal wetlands of the Mediterranean region.

Salinization is a problem that affects many Mediterranean coastal wetlands, and that is expected to be exacerbated due to altered precipitation patterns, the anthropogenic alterations of the hydrologic cycle and the sea level rise (Herbert et al., 2015; Schuerch et al., 2018; Taylor et al., 2021). Many endemic species of these ecosystems are euryhaline, however the available literature shows that abrupt salinity increases results in a loss of biodiversity and changes the structure of invertebrates and plant communities. Furthermore, the effects of salinization often show their greatest impacts at juvenile stages of individuals (Herbert et al., 2015), thus affecting population recruitment and reducing their competition advantage as regards to halophilic taxa. Further experimental research should consider biodiversity impacts in relation to salinity gradients, particularly on periods of water scarcity (*e.g.* summer), when salinity peaks are expected to occur. This research will guide sustainable hydrological management options such as the maintenance of minimum river flows and groundwater recharge for specific water bodies. Moreover, the impact that salinity intrusion may have on helophyte plant coverage should be further investigated, as may have consequences for carbon sequestration and the emission of greenhouse gasses in Mediterranean coastal wetlands (Morant et al., 2020).

Climate change contributes to an increase in the mean air and water temperature, but is also expected to increase the frequency and magnitude of heat waves and other extreme events in the Mediterranean region (Kuglitsch et al., 2010; Vautard et al., 2014). As shown by this review, the steady increase of mean water temperatures generally results in an increase of biomass of primary producers and a decrease in richness and density of consumers, simplifying trophic webs, and reducing overall biodiversity. However, no studies have evaluated the influence of sudden temperature increases on the biodiversity of Mediterranean coastal wetlands. Recurrent heat waves are expected to impact all trophic levels (Bertani et al., 2016), and may contribute to a 'tropicalization' of these ecosystems, favoring the occurrence of toxic algae blooms and the introduction of alien species (Carreira et al., 2017; Gobler, 2020). Further research should focus on describing the plasticity of Mediterranean wetland organisms and communities to extreme events such as heat waves or extreme precipitations to support prospective evaluations on biodiversity change.

As pointed out by several authors, Mediterranean coastal wetlands have been exposed to several co-occurring stressors during the last years, and the trend is expected to increase due to increasing demographic pressure and climate change impacts (Geijzendorffer et al., 2019; Navarro-Ortega et al., 2012; Sala et al., 2000; Vilas-Boas et al., 2021; Taylor et al., 2021). This review shows that there is a limited number of studies evaluating the potential interaction between stressor combinations in these ecosystems. Among them, temperature increase, in combination with nutrient availability, salinity or single pesticide contamination have been the most researched ones, while the interaction of invasive species with any other stressor is clearly underrepresented. Moreover, only one study has assessed the impacts of multiple stressors on high levels of biological organization following a factorial design (Vilas-Boas et al., 2021), so that their impact of on species succession patterns, ecosystem resilience, and biodiversity decline are yet to be further defined.

The few available studies on multiple stressors in Mediterranean coastal wetlands show that the interaction among them frequently results in non-additive responses in individuals and populations. Similar conclusions have been obtained from broader studies evaluating multiple stressor effects on other aquatic ecosystems (Jackson et al., 2016). Multiple stressors research supports our mechanistic understanding of biodiversity impacts and the ecologic processes and services that may be impaired given different levels of anthropogenic stress, and can be used to guide effective ecosystem management and restoration (Orr et al., 2020). In this regard, it is important to note that Mediterranean coastal wetlands have a high capacity to absorb multiple sources of stress without denoting adverse effects, so it is difficult to predict a change until it effectively happens (Pérez-Ruzafa et al., 2011). Our predictive capacity of multiple stressor effects on aquatic systems requires the characterization of the background levels of stress, their temporal and spatial dynamics, the species sensitivities, as well as how species interactions may yield correlated responses (Perujo et al., 2021). For instance, some studies show that pesticide pollution or increasing temperatures may significantly contribute to algae blooms and eutrophication (Gobler, 2020; Lu et al., 2019), so that measures that target only nutrient emissions of Mediterranean coastal wetlands will not be sufficient to re-store biodiversity in multi-stressed ecosystems. Similarly, reducing the impacts of invasive species requires a holistic approach that tackles the factors that affect the competition strength with native species such as chemical contamination or salinity increase.

Table 1
Summary of research needs identified for the different evaluated stressor groups.

Stressors	Research needs
Eutrophication	– Define tipping points to revert the eutrophication status of Mediterranean coastal wetlands – Identify local taxa that plays a major role in nutrient cycling and eutrophication prevention
Chemical pollution	– Assess the effect of chemical mixtures on populations and communities – Characterize the sensitivity and resilience of Mediterranean coastal ecosystems to chemical stress
Invasive species	– Evaluate the influence of climate change on invasive species distribution – Assess side-effects of invasive species on trophic webs and ecosystem functioning
Salinization	– Assess the seasonality and magnitude of salinity increase – Quantify the influence of salinity on species competition and succession
Temperature increase	– Explore Mediterranean species plasticity to temperature rise – Assess the impact of heatwaves on populations and communities
Multiple stressors	– Assess interactive effects of invasive species and other stressors – Assess interactive effects on species succession patterns, resilience, and overall biodiversity – Define relevant gradients of stress for different ecosystem types – Develop bioindicators for different stressors and combinations

Despite Mediterranean wetland ecosystems are subjected to similar environmental conditions, there is some heterogeneity regarding the type of water bodies, their connectivity, geographical location or distance to urban areas and other sources of anthropogenic stress (Pérez-Ruzafa et al., 2007). In this regard, it would be interesting to study their differences in stressor response patterns. The available literature prevents doing so, as there is too little information for some stressors and limited consistency in the methods and types of endpoints evaluated in the different studies. Further research should consider the implementation of systematic experimental approaches to evaluate stressor's distribution and ecosystem's response patterns within the Mediterranean region, as well as the identification of bioindicators that may relate to these.

In conclusion, after analyzing the studies that assessed the impact of anthropogenic stressors on the biodiversity of Mediterranean coastal wetlands, we have denoted an asymmetry in the nature of the available studies, both in terms of the frequency of stressors assessed and the areas where these studies have been performed. Overall, the south and east of the Mediterranean basin are underrepresented compared with the north and west part of the basin, which may be partly influenced by their lower publication rate in international scientific journals. On the other hand, biodiversity impacts by stressors like temperature raise or species invasions have been weakly discussed in contrast with others like eutrophication or chemical pollution. Here we have provided a summary of the currently knowledge regarding their ecological impacts and highlighted specific research needs to improve our understanding on the impacts of each of these stressors on biodiversity, which are summarized in Table 1. Our study also shows that the interaction between stressors requires further investigation in Mediterranean coastal wetlands. Biodiversity protection to achieve the goals set by the different regulatory frameworks requires a holistic approach that accounts for the interaction between stressors on individuals and populations, as well as their propagation to high levels of biological organization.

CRediT authorship contribution statement

Claudia Martínez-Megías: Conceptualization, Investigation, Methodology, Visualization, Writing – original draft, Writing – review &

editing. **Andreu Rico:** Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Supervision, Writing – original draft, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.151712.

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