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Water/marine Zero Pollution Outlook

A forward-looking, model-based analysis of water pollution in the EU

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Contents

Abstract

Forward looking analyses are needed in order to anticipate which policy/management options can deliver the very ambitious objectives of the Zero Pollution (ZP) action plan. Integrated and sophisticated numerical modelling tools are useful to generate future scenarios and '*what if*' analysis as they allow the virtual manipulation of the anthropogenic pressures on ecological systems.

JRC has been developing an integrated modelling framework covering the inland and marine waters of the EU, the Marine Modelling Framework (MMF) that follows the principle of the Digital Twins (DT) and that allow to assess the impacts of diverse management strategies on the status of freshwater and marine ecosystems through the EU. In the present report, the JRC-DT for water and marine ecosystems is used to test how different policy options can help achieve some of the ZP objectives.

From the six top ambitions of the ZP action plan, two are particularly relevant for the water/marine environments. First, the ZP action plan states that it aims at '*improving water quality by reducing waste*' and in particular, it mentions the (reduction of) '*plastic litter at sea (by 50%)*'. The second relevant ambition refers to '*improving soil quality by reducing nutrient losses and chemical pesticides' use by 50%'*, which does not only impact soil quality but also the receiving waters (rivers, lakes and seas).

Given these overarching priorities of the ZP ambition, the water/marine outlook, thus, focuses on three particular pollution pressures in aquatic environments: inorganic nutrients, chemical pollutants and plastics. The JRC-DT, has been used to explore how current and future policy implementations could help delivering the particular ambitions of the ZP for these environmental pressures by 2030 including the background impacts of climatic changes.

The analysis indicates that it is possible to substantially reduce the leakage of nutrients from soils and freshwater into marine ecosystems (between 17% and 32%) but to reach the 50% reduction ZP objective a much more ambitious policy scenario is needed. The JRC-DT also indicates the need to pay particular attention at the reduction measures as creating an imbalance between nitrogen and phosphorous in marine environments can provoke unexpected problems, such as Harmful Algae Blooms.

Modelling work also indicates that reducing the inputs of chemical substances into marine basins can significantly reduce their concentration in the sea water, a reduction that depends on the physico-chemical characteristics of the substances (e.g., half-life). However, model results indicate also that the latency of natural systems makes the diminution of chemicals' concentration at sea to be slow and that hydrographical changes associated to climate change (e.g., alteration of currents) can provoke the accumulation of certain substances in some areas even if their mean presence in the whole basin has decreased.

Finally, the JRC-DT has been proven a useful tool to analyse the distribution of plastic litter at sea and how EU and non-EU countries can cross-pollute each other in shared basins. This high connectivity of marine areas calls for international collaboration in the fight against this pervasive form of pollution as the EU, by itself, will not be able to reach the ZP objectives of litter reduction.

The scenarios modelling described in this report for European freshwater and marine ecosystems and in Grizzetti et al. (2022) for European freshwater provide the details of the analysis presented in the Zero Pollution Outlook report (Joint Research Centre, 2022).

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1 Introduction

The Zero Pollution (ZP) ambition of the European Commission (EC) is a transversal policy initiative aiming at fostering the fight against environmental pollution and consequent ecosystems' deterioration already included in EU legislations. It does so by establishing some ambitious pollution reduction targets on many different components of the natural system including water, soil, air, marine and even human health.

Aquatic ecosystems are the final recipients of pollutants originating from many different anthropogenic activities (e.g., agriculture, industrial activities, urban processes) originally delivered into different components of the Earth System (such as soil or air). At EU level, a number of legislative acts aim to fight against pollution of aquatic systems such as the Water Framework Directive (WFD) together with the Environmental Quality Standards and Groundwater Directives under which the Commission plans has proposed an update of the lists of surface and groundwater pollutants, the Marine Strategy Framework Directive (MSFD), the Nitrate Directive (ND), the Plastics Directive or the Urban Waste Water Treatment Directive (UWWTD) for which a revisions has been proposed. The ZP ambition is highly relevant in this complex legislative context as it can foster synergies between the different legislative acts. The ZP Action Plan should help to strengthen and streamline the connections among the water/marine directives and increase, thus, the added benefit of EU intervention in the fight against aquatic pollution.

This report constitutes a forward looking exercise (outlook) that explores potential policy and management options that can help achieving some of the ZP targets for the water/marine ecosystems, at EU scale and beyond. Obviously, policy measure analyses should ideally be carried out in an integrated and holistic manner, considering the sources of pollution to aquatic systems, their distribution and dispersion in freshwater and marine ecosystems and their impact on the ecological status of the receiving basins. This was done for this outlook exercise by making use of an integrated modelling framework developed at JRC (the JRC-Digital Twin for water/marine environments here JRC-DT).

From the different environmental targets or ambitions established by the ZP, this report concentrates on three particular pollution pressures: nutrients, chemical pollution and plastics. This choice is basically determined by the primary importance of these types of pollution for aquatic environments and the technical capabilities of the JRC-Digital Twin (JRC-DT).

This report starts with a brief review of the current legislation linked to the ZP ambition and the European Green Deal (EGD) objectives, in section 2. It then describes briefly the models that compose, all together, the JRC-DT in section 3. Section 4 deals with the human pressures included in the report, explaining the reasons behind their selection and their connections to the ZP overarching goals. The different scenarios tested with the JRC-DT are explained in detail in section 5, including the policy options behind them. The main results of the models simulations are then presented in section 6. Section 7 describes future perspectives on how to further strengthen the ZP ambition while the main conclusions of the report are summarized in section 8.

2 Legislations, strategies and actions

The EGD adopted by the European Commission in July 2020, sets out a new growth strategy that aims to transform the EU into a fair and prosperous society, with a modern, resource-efficient and competitive economy characterized by an absence of net emissions of greenhouse gases in 2050 and where economic growth is decoupled from resource use. It also aims to protect, preserve and enhance the EU's natural capital, and protect the health and well-being of citizens from environment-related risks and impacts. At the same time, this transition must be just and inclusive (COM(2019) 640 final).

The EGD sets out several long-term strategic objectives transforming the EU's economy for a sustainable future:

- increasing the EU's climate ambition for 2030 and 2050;
- supplying clean, affordable and secure energy;
- mobilising industry for a clean and circular economy;
- building and renovating in an energy and resource efficient way;
- a **zero-pollution ambition** (ZPA) for a toxic-free environment;
- preserving and restoring ecosystems and biodiversity;
- from Farm to Fork: a fair, healthy and environmentally friendly food system;
- accelerating the shift to sustainable and smart mobility;
- a renewed sustainable finance strategy.

As of 2020, new measures in support of the EGD were rapidly introduced, one of which was the Eighth Environment Action Programme (8th EAP). On 1 December 2021, EU ambassadors approved a provisional political agreement reached between the Council presidency and the European Parliament's negotiators on 1 December, regarding the 8th EAP.

The 8th EAP translates the political commitments of the EGD into law and will help guide the EU's environmental and climate policymaking and implementation until 2030, with a long-term vision to 2050. The 8th EAP aims to accelerate the transition to a climate-neutral, resource-efficient and regenerative economy, which gives back to the planet more than it takes.

The 8th EAP includes the following six priority objectives:

- Achieving the 2030 greenhouse gas emission reduction target and climate neutrality by 2050;
- Enhancing adaptive capacity, strengthening resilience and reducing vulnerability to climate change;
- Advancing towards a regenerative growth model, decoupling economic growth from resource use and environmental degradation, and accelerating the transition to a circular economy;
- Pursuing a **zero-pollution ambition** (ZPA), including for air, water and soil and protecting the health and well-being of Europeans;
- Protecting, preserving and restoring biodiversity, and enhancing natural capital (notably air, soil, and forest, freshwater, wetland and marine ecosystems); and
- Reducing environmental and climate pressures related to production and consumption (particularly in the areas of energy, industrial development, buildings and infrastructure, mobility and the food system).

The ZP vision is, hence, one of the major initiatives within the EGD and the 8th EAP. It is thought to be a facilitator and a strengthening of the current EU legislations fighting environmental pollution. The overall aim of the ZP for 2050 is for air, water and soil pollution to be reduced to levels no longer considered harmful to health and natural ecosystems, which respect the boundaries with which our planet can cope, thereby creating a toxic-free environment. In order to achieve this overarching ambition, the ZP sets out a series of targets for <u>2030</u> to speed up reducing pollution at source. These targets include:

— improving air quality to reduce the number of premature deaths caused by air pollution by 55%;
— improving water quality by reducing waste, plastic litter at sea (by 50%) and micro-plastics released into the environment (by 30%);
— improving soil quality by reducing nutrient losses and chemical pesticides' use by 50%;
— reducing by 25% the EU ecosystems where air pollution threatens biodiversity;
— reducing the share of people chronically disturbed by transport noise by 30%, and
— significantly reducing waste generation and by 50% residual municipal waste.

The ZP action plan aims to strengthen the EU green, digital and economic leadership, whilst creating a healthier, socially fairer Europe and planet. It provides a compass to mainstream pollution prevention in all relevant EU policies, to step up implementation of the relevant EU legislation and to identify possible gaps. More information can be found on the updated web page: https://ec.europa.eu/environment/strategy/zero-pollution-action-plan_en.

To support the ZP vision and ambitions, a holistic approach to environmental issues is mandatory. This approach makes use of forward-looking analyses and planning tools. This type of integrated and outlook analysis is achieved thanks to the use of advanced numerical modelling tools (or Digital Twins) of natural systems.

The ZP Action Plan will support and strengthen existing EU policy instruments, such as Common Agricultural Policy (CAP), Nitrate Directives (ND), Urban Waste Water Treatment Directive (UWWTD), Plastic Directive, Single Use Plastic Directive (SUPD), Water Framework Directive (WFD), Marine Strategy Framework Directive (MSFD) and Biodiversity Strategy (BDS). Some measures already planned under these sectorial directives and strategies are incorporated in the ZP outlook simulations analysed in this study.

3 Integrated modelling chain form the land to the sea

In order to address the needs for integrated and holistic assessments of human impacts on the status of the EU environment, employing innovative and multidisciplinary tools is mandatory. An example of such tools are Integrated modelling frameworks, as they allow to evaluate the impacts of different management options in a digital copy of the real ecosystems. This approach is being pursued by the EC in the context of the digital and green transition and, specifically, in the Destination Earth initiative and the associated Digital Twins (DTs) (see the DestinE digital strategy at https://digital-strategy.ec.europa.eu/en/policies/destination-earth).

The JRC, as the scientific branch of the EC, has been working on implementing and using this type of modelling tools to help assessing policy options since a number of years. Different aspects of the JRC integrated freshwater/marine modelling framework (or JRC-Digital Twin) have been developed by different groups in diverse units, at differential speeds, during the last decade. More recently, collaborations between these groups have been reinforced with the objective to build a more holistic description of the natural systems in response to the policy needs (i.e., the ZP and EGD requests).

3.1 Freshwater models

3.1.1 Freshwater quantity

The water resources calculations are done with the distributed water resources model LISFLOOD (De Roo et al., 2000; Van der Knijff et al., 2010; Burek et al., 2013; Bisselink et al., 2018a) coupled with crop growth processes from the EPIC model (Williams et al., 1989; Williams, 1995; Sharpley and Williams, 1990) and a newly developed irrigation module. Driven by meteorological forcing data, the integrated LISFLOOD-EPIC model simulates dynamically hydrology, crop growth and irrigation, accounting for water abstractions for household, livestock, industry and energy sectors at a daily time step and every grid-cell defined in the model domain (5x5 km for Europe).

Processes simulated for each grid cell include snowmelt, soil freezing, surface runoff, infiltration into the soil, preferential flow, redistribution of soil moisture within the three-layer soil profile, drainage of water to the groundwater system, groundwater storage, and groundwater base flow. Runoff produced for every grid cell is routed through the river network, using a double kinematic wave approach, one for the main channel, and one for the floodplain. Lakes, reservoirs and retention areas or polders are simulated by giving their location, size and in- and outflow boundary conditions and steering parameters. Discharges are calibrated and validated from approximately 1500 gauging stations

3.1.2 Freshwater quality

Annual nutrient (total nitrogen (TN) and total phosphorus (TP)) loads from land to sea were assessed with the conceptual model GREEN (Grizzetti et al., 2012; 2021), as implemented in the R open source package GREENeR (Udias et al. 2022). Briefly, the model builds on the spatial architecture of the CCM2 hydrological network (Vogt et al. 2007; 2008), which identifies catchments of about 7 km2 area, each having one main reach with an upstream node and a downstream node to form the network that connects land from headwaters to the seas or internal endorheic lakes. The total land extent considered in the GREEN model application amounts to 6.27M km2, encompassing all river basins draining in European seas, covering in part or completely 44 countries, namely 27 EU countries and 17 non-EU countries.

The model assesses annual loads of nutrients (t/y) considering the upstream-downstream accumulation of sources, loess retention in land of diffuse emissions, and retentions in rivers and lakes. Land retention is an inverse function of total annual rainfall, and thus changes annually. Conversely, river retention is a function of reach length, whereas lake retention is a function of lake depth and hydraulic residence time. Retention in land and rivers is regulated by two parameters that are calibrated against loads estimated at monitoring stations. From nutrient loads, the mean concentration of nutrient in rivers is calculated dividing loads by the mean annual discharge in the reach (full details available at Grizzetti et al., 2021).

3.2 Marine models

3.2.1 Hydrodynamic model

The model used to simulate the 3D physical structure and the hydrodynamics of the different EU basins is the General Estuarine Transport Model (GETM, http://www.getm.eu/) combined with the General Ocean Turbulence Model (GOTM). The meteorological forcing from the European Centre for Medium Range Weather Forecast (ECMWF,) available from http://www.ecmwf.int, based on 3-hourly records of ERA5 dataset has been used in all the implementations presented. The Black Sea implementation is described extensively in Miladinova et al. (2017 and 2018), the Mediterranean Sea setup is described by Macias et al. (2014), for the Baltic Sea information could be found in (Parn et al., 2019) while the North Sea configuration is described in Friedland et al. (2020).

The meteorological forcing from the Coordinated Regional Climate Downscaling Experiment (CORDEX) has been employed as it permits to perform hindcast as well as future “climate change” (RCP4.5: intermediate emission scenario) experiments in a consistent way (see details in section 5.1 below).

3.2.2 Biogeochemical models

Different biogeochemical models (which are detailed below) have been used to reproduce the particular features of each EU basin. Each of these models is coupled on-line with the GETM-GOTM system through the Framework for Aquatic Biogeochemical Models (FABM) (Bruggeman and Bolding, 2014).

The Black Sea biogeochemical model (BSEM) is tailor-made for its particular N-limited ecosystem (Miladinova et al., 2016). It represents the classical omnivorous food web, including several phytoplankton and zooplankton groups, as well as gelatinous zooplankton species *Mnemiopsis* and *Beroe Ovata*. For the Mediterranean Sea, the biogeochemical model MedERGOM (Macias et al, 2019) is implemented. This model has been developed to simulate the strong P-limitation observed in this basin by allowing variable N:P ratio in phytoplankton nutrient uptake (Line of Frugality concept) (Galbraith & Martiny 2015). In the North Sea, the European Regional Seas Ecosystem Model (ERSEM, Butenschon et al., 2016) is applied (Friedland et al., 2020 & 2021). This complex model, explicitly accounting for benthic functional types (meiofauna, deposit- and suspended-feeders), is particularly suitable to describe the strong benthic-pelagic coupling, typical of this area. Finally, for the Baltic Sea, the ERGOM model (Neumann, 2000) has been used. ERGOM has been shown to adequately simulate the ecosystem of this basin in previous publications (e.g., Lessin et al., 2014; Parn et al., 2020).

3.2.3 Eulerian (chemical dispersion) models

To allow tracking substances in the Black Sea basin, a tracer model with a time dependent decay was coupled on-line (i.e., the tracer concentration is simulated after each hydrodynamic time step), via FABM, with the hydrodynamic model. Tracer simulations were carried out by solving equations of transport, diffusion, and degradation of a synthetic tracer. The chemical substance is transferred into the sea by the rivers whose freshwater input has been estimated using the values from the LISFLOOD model (see section 3.1).

3.2.4 Lagrangian (plastic) models

Lagrangian models simulate the pathway of individual particles (usually several thousands) transported by a flow field (see Van Sebille et al., 2018 for a review). In this case the flow field is derived from the 3D current structure of the Mediterranean Sea provided by the hydrodynamic model described in section 3.2.1. These currents are coupled off-line (i.e., after the model experiments has been performed) with the particle-tracking, Lagrangian model LTRANS-ZLev (Laurent et al., 2020) to simulate the dispersion, accumulation and beaching of floating macro-plastic items entering the Mediterranean Sea through the rivers’ discharges (see more details about the input estimates in section 5.3). Similar approaches have been followed in previous applications of the JRC-DT in both the Mediterranean Sea (Macias et al., 2019 and 2022) and in the Black Sea (Miladinova et al., 2020b).

4 Human pressures considered in the present analysis

Form the specific targets of the ZP ambition presented above, **nutrients**, **plastics** and **chemical** pollution were selected as key pressures to be analysed in the present report. The details of these pressures are provided in the section below.

4.1 Nutrients

Inorganic nutrient pollution of surface and ground waters is a major cause of eutrophication and environmental degradation throughout the world, being one of the most long-standing environmental issues for freshwater and marine environments. The Water Framework Directive (WFD) includes nutrient pollution in the pressures affecting the status of waters; similarly, the Marine Strategy Framework Directive (MSFD) considers nutrient pollution and associated eutrophication risk as one of the descriptors for Good Environmental Status (GES). However, the ZP ambition recognizes the need to make more efforts to fight this recursive anthropogenic impact (i.e. eutrophication) by aiming to '*reduce nutrients losses into the environment by 50% by 2030*'.

Sources of inorganic nutrient pollution are diverse (diffusive, point sources, atmospheric, etc.) and involve many different anthropogenic activities, from agriculture (e.g. fertilization), to industries and households. Given the widespread nature of the impacts associated to nutrients leakage into the environment, the application of integrated modelling is particularly relevant to evaluate alternative management scenarios.

The model GREEN (see section 3.2) considers major diffuse nutrient sources to land, which undergo reduction through soil filtering and plant uptake before reaching the riverine network, and point sources that are directly emitted to the stream network. Diffuse sources comprise mineral and organic fertilization on agricultural land, and domestic emissions from houses and individual systems that are disconnected from sewerage systems (termed herein as scattered dwellings), and discharged into the soil. Additional diffuse sources of nitrogen are from soil and plant fixation, and from atmospheric deposition. An additional diffuse source of phosphorus is represented by emissions from non-agricultural land (background emissions). Point sources include industrial and domestic emissions collected in sewerage systems and discharged directly in the stream network or in coastal areas.

River-induced fertilization of marine coastal waters is a natural process, but when exacerbated due to human activities (as worldwide observed in recent times Diaz, 2001) may lead to eutrophication. A major manifestation of eutrophication is the development of persistent micro- or macro- algal blooms (Heisler et al., 2008) producing large quantities of organic matter (OM) in the river plume and adjacent shelf water (Lohrenz et al., 1997; Dagg and Breed, 2003). A fraction of this OM, along with terrestrial OM inputs, settle to the seafloor where it is decomposed by benthic fauna and bacteria (Rabouille et al., 2008). OM decomposition, consuming oxygen, increases stress and mortality of benthic organisms (Diaz and Rosenberg, 1995). A disproportional increase in nitrogen and phosphorus, especially if associated with changes in the nitrogen to phosphorus (N:P) ratio, may also alter the phytoplankton community structure leading to the development of harmful algal blooms. Eutrophication and its consequences are, therefore, a serious threat to coastal ecosystems (Cloern, 2001).

The marine biogeochemical models of the JRC-DT (see section 3.2.2) are specifically designed to describe the dynamics of inorganic nutrients and their effect on the low-tropic levels of the marine food web (i.e. phytoplankton and zooplankton) and key biogeochemical variables (*i.e.*, oxygen). Model variables have been post-processed so they are consistent with the criteria described within Descriptor 5 (D5) of the MSFD (see details in section 6.1.2)

4.2 Chemicals from land-based sources

The diversity of chemical substances released by human activities in the environment is huge (United Nations Environment Programme (UNEP), 2012). For this reason, it is impossible to design and implement accurate biophysical models to simulate all of them. Here we focus on two types of substances that rank amongst the most abundant chemical pollutants in natural waters, pesticides, and pharmaceuticals. Pesticides are typically released from agricultural practices and can cause freshwater, groundwater, and marine water pollution through drainage, leaching, runoff and surface deposition (Radović et al., 2015; Chapman et al., 2016).

Pharmaceutical substances are introduced into aquatic systems through industrial waste, farming practices and domestic wastewater (Patel et al., 2019). The widespread occurrence of pharmaceuticals in the marine waters (Mezzelani et al., 2018; Almeida et al., 2021) indicates the lack of efficient removal during wastewater treatment. In complement to these two broad groups, caffeine, a largely consumed stimulant, is also found in noticeable quantity in waste waters. The ZP aims to '*reducing chemical pesticides' use by 50%*' and also to '*significantly reducing residual municipal waste by 50%*', So, for the purposes of this report the following pesticides, pharmaceuticals and stimulants (used as typical tracers for the cleaning power of the wastewater treatment plants) have been selected for investigating their actual relevance causing pollution, their physical-chemical behaviour in the environment and their potential to be managed within the ZP remit (Table 1).

All substances listed in this table 1 are water-soluble and with low bioaccumulation potential. Additionally, they are highly persistent since their degradation half-life (DT50) in marine water is higher than 60 days (ECHA European Substances Agency, 2011). Environmental Quality Standards (EQS) or the Predicted No-Effect Concentration (PNEC) values are widely used to protect the environment and human health from substances deriving from human activity. They are usually associated with specific chemical concentrations, below which undesirable effects are not expected. In Table 1 are given the annual average EQS values of atrazine, diuron and simazine for marine waters and the lower bound PNEC values of the rest of the substances for marine waters.

Table 1. Type, persistency (DT50), relevance to EU policies and EQS (Environmental Quality Standard) or PNEC (Predicted non-Effect Concentration) values of selected substances

Substance	Type	**Persistency DT50 (day)**	**Relevance to EU policies**	**PNEC (ng l^{-1}) or EQS (*) (ng l^{-1})**
Atrazine	Pesticide	578 (US EPA, 2006)	Priority list (WFD, 2000)	600* (EC, 2016)
Simazine	Pesticide	50 -700 (EPA US, 2006)	Priority list (WFD, 2000)	1000* (EC, 2016)
Terbuthylazine	Pesticide	High (EPA US, 1995)	Emerging (Tornero and Hanke, 2018)	10 (Slobodník et al., 2017)
Diuron	Pesticide	80	Priority list	200* (EC, 2016)
Sulfamethoxazole	Pharmaceutical	High (Benotti and Brownawell, 2009)	Watch list (WL, 2015)	600
Carbamazepine	Pharmaceutical	1278 (Björlenius et al., 2018) 328 (Wenzel and Shemotyuk, 2014)	Emerging (Tornero and Hanke, 2018)	50 (Moermond, 2014)
Diclofenac	Pharmaceutical	50 (Bonnefille et al., 2018)	Emerging (Tornero and Hanke, 2018)	5 (Diclofenac-HELCOM-pre-core-indicator-2018)
Caffeine	Stimulant	70	Emerging (Tornero and Hanke, 2018)	1200

4.3 Plastic pollution

Plastic litter presence in our seas and oceans is a growing global concern (Law, 2017; Lebreton et al., 2018) that has prompted multiple initiatives aiming to fight against this pervasive form of pollution. These initiatives include awareness raising, waste management plans, removing the items from the environment (Burt et al., 2020; Rodríguez et al., 2020) or reducing the production and consumption of certain plastic articles (e.g., Directive on the reduction of the impact of certain plastic products on the environment, (EU) 2019/904). In this context, the ZP aims to '*reducing plastic litter at sea by 50%*' while it also targets the reduction of '*micro-plastics released into the environment by 30%*'. For the present report, only the first target will be considered as sources of micro-plastics and measures to control them are still not well-enough known to be incorporated in the numerical modelling tools.

Using the Mediterranean Sea as a test basin for this pressure (see section 3.2.4) is a choice based on the characteristics of this marine basin that make the Mediterranean Sea particularly vulnerable to plastic pollution. Its semi-enclosed nature, large coastal population (Jambeck et al., 2015), intense touristic and maritime activities and anti-estuarine general circulation (Macías et al., 2016) turns the Mediterranean into an accumulation basin from which floating litter hardly escape (Cozar et al., 2015). It has been estimated that the Mediterranean Sea host 5% - 10% of global plastic mass (Van Sebille et al., 2015) while it represents less than 1% of the overall ocean surface.

Coastal countries of the Mediterranean Sea present vastly different socio-economic realities and the approaches (e.g., regulatory frameworks) used for waste-management resources are often quite diverse. They also have distinct cultural attitudes towards plastic use, disposal, and recycling. However, cross-country litter pollution in the Mediterranean can be quite significant (on average 30% of coastal litter is originated beyond the borders of any given country) with the basin being described as a 'melting pot' for plastic pollution (Macias et al., 2022). All these characteristics make the Mediterranean Sea a very appropriate showcase to highlight the relevance of the international dimension of EU environmental initiatives and that the overarching ambition of the EGD needs to go beyond EU's borders.

5 Description of scenarios

Considering the forward-looking nature of this report and the capacities of the models used here to provide '*what if*' analyses, a set of specific scenarios have been carefully designed for each pressure considered, keeping in mind the overarching goals of the ZPA as described further above.

Two main timeframes are considered, one reflecting the recent conditions of EU rivers and sea (i.e., a reference (REF) scenario) and another projecting measures into the future. Both future scenarios considers the climatic conditions under the representative common pathway (RCP) 4.5 scenario as simulated by the MPI-ESM-LR model downscaled with the COSMO-CLM regional model. Details on the measures included in the different scenarios for each considered pressure are detailed in the following sub-sections.

It is important to consider that, due to computational resources limitations, only one RCP simulated by one general circulation model (GCM) has been considered. This produces only one future climate scenario without providing an idea of the uncertainty associated with climate projections. In future work, more than one RCP and (hopefully) more than one GCM should be considered when addressing outlook scenarios.

5.1 Nutrient scenarios

Given the ambitious reduction objective set in the ZP action plan regarding nutrients (i.e., 50% reduction) and the widespread origin of this type of pollution, the JRC D.2 modelling groups have designed and implemented a series of future projections model scenarios aiming at quantifying how an ambitious management strategy could improve freshwater quality (in terms of nutrient pollution) and quantity (full details of the different alternative scenarios considered could be found in Grizzetti et al., 2022 and Grizzetti et al., submitted).

For the purposes of this report, the present situation in the EU regarding nutrients leakage into the environment and actual status of marine ecosystems with respect to eutrophication (REF scenario) will be compared with a future scenario in which ambitious nutrient-reduction measures are implemented (HAS scenario).

5.1.1 REF scenario

The REF scenario constitutes the best approximation to the actual state of EU freshwater and marine ecosystem for the past few decades. It uses the downscaled hindcast climate by COSMO-CLM from 2005 and for comparison with the future (HAS) scenario, we averaged conditions in the period 2014 – 2018. Freshwater quantity (determined by the LISFLOOD model) and quality (determined by the GREEN model) provides annual loads to the different marine regions that are, then, considered by the respective hydrodynamic-biogeochemical models to simulate the low-trophic level status of EU Marine ecosystems. Even if this simulation approximates the 'real' ecosystem status, it provides a holistic, integrated view of all the EU and it is included in this report with the only purpose to provide a reference to which compare the impact of measures included in the HAS scenario.

5.1.2 HAS scenario

It incorporates measures aiming to decrease the extraction of freshwater from natural streams and others targeting reducing nutrient leakage into the environment. The measures that may reduce water abstraction and net water consumption (abstraction minus return flows) contemplated in the HAS are:

— Increasing irrigation efficiency in agriculture

— Increasing urban water efficiency by reducing leakage

— Re-using treated urban wastewater for irrigated agriculture

— Water use efficiency in the energy sector by cooling water requirements

— Use of desalination of sea water for public water use

On the other hand, the models quantified the possible reduction of nitrogen and phosphorus input from the major sources of point and diffuse nutrient pollution in the river basins, namely domestic wastewater discharges, agriculture, and atmospheric deposition (for nitrogen), corresponding to nutrient reduction measures under different EU policies, including:

— Reduction of nutrient discharges from domestic wastewaters. From different scenarios explored for the Impact Assessment of the revision of the UWWT Directive (Pistocchi et al., submitted) we selected the more stringent one that includes full compliance with the measures established in the UWWTD and a combination of additional measures for extending the efficiency of the level of treatment and the extent of the Sensitive Areas.

— Reduction of nutrient emissions from agricultural sources. We consider the implementation of the new CAP legislative proposal plus measures to achieve the EGD targets also using New Generation EU Funds, according to the CAPRI model scenarios of Barreiro Hurle et al. (2021).

— Reduction of nitrogen input from atmospheric deposition. A scenario of N atmospheric deposition reduction was developed by the EMEP model considering the measures adopted by the Commission to reduce atmospheric emissions by 2030 in the Fit for 55 package (Pisoni et al. submitted).

This HAS scenario includes the projection under emission scenario rcp4.5 (IPCC) of the above described measures until 2030. The forward projection is, thus, based on an ambitious future, where EU goes beyond current legislation obligations to reduce as much as possible nutrient leaks into natural waters.

The resulting freshwater loads and nutrient concentrations are provided to the marine models to quantify the impacts of the proposed measures (+ climate change) on the environmental status of EU marine ecosystems. The comparison with the REF scenario is made averaging the conditions for the period 2026 – 2030. Furthermore, from this HAS simulation, it is possible to isolate and assess the impacts of climate change on the status of EU marine ecosystems as detailed in section 6.1.2.2 below

5.2 Chemicals scenarios

Chemical contamination of natural waters (freshwater, groundwater and marine) occurs almost continuously and from many different sources. We have concentrated the modelling efforts of this report in the handful of substances indicated in Table 1. The Black Sea is the first EU basin to which the contaminant model has been implemented due to data availability on the presence of chemical substances and on the maturity level of the involved hydrodynamic models.

The first set of scenarios was related to the ZP ambition of '*reducing chemical pesticides' use by 50%*'. In this case, the most used pesticides and important pharmaceutical products were simulated to decrease their actual concentration in riverine waters by 50% in 2030 assuming a progressive, linear decrease. An evaluation of the relative impact in the different areas of the studied sea basin allows identifying the more sensible regions to this management option. The selected scenario period is 2019 – 2030. The impact of climate change is therefore simultaneously considered together with the management measures as it includes the atmospheric conditions provided by the same COSMO-CLM RCP4.5 used in the 'nutrients scenarios'.

Some of the studied substances (such as simazine and atrazine) have been already banned in the EU but are still found in certain environmental reservoirs. For this reason, the second set of scenarios addresses the effect of past policy measures. Simazine and atrazine concentrations in the rivers is assumed to be 100% from 1995 to 2004, then decreases linearly from 100% to 0 in 2004-2008 and remains 0 from 2009 to 2019. This will allow identifying regions within the Black Sea where such chemical compounds would be concentrated and stored in the long term. This second set of scenarios were performed using the hindcast (observed) climate on the Black Sea region as atmospheric forcing.

5.3 Plastics scenarios

Plastic pollution of water and marine ecosystems is one of the major environmental concerns of our time justifying the ZP goal of '*reducing plastic litter at sea by 50%*'. There is, however, a fundamental lack of information about the presence, inputs, and distribution of plastics in our environment. The number of available datasets is increasing (see EMODnet and ZP monitoring report (EEA)). This effort is however hampered by methodological and sampling issues that still prevent to obtain a comprehensive view of plastic pollution in the EU.

For the purposes of this report, we link a plastic input model/estimate (provided by a modification of González-Fernández et al., 2021) that combine estimations of mismanaged plastic per inhabitant in the different countries with the total population living on a particular river basin district. By multiplying the annual mismanaged litter production by the total population, we obtain an approximation of the number of plastic items entering the Mediterranean Sea with a spatial pattern that should mimic the actual macrolitter pollution sources to this marine basin (Figure 17). The combination of this inputs estimates and the Lagrangian model described above forced by the hindcast (observed) atmospheric conditions for the Mediterranean basin (for years 2016 to 2018) provides the baseline distribution of litter in the basin, both floating and deposited at the beaches. Against this baseline, we assess three different scenarios, two related with management options and the third aiming at addressing the impacts of climate change (see below).

The first scenario (SUP ban) simulates the impact of a total ban of Single Use Plastic (SUP) items in EU Member States (MS). It has been reported that SUP constitute up to 60% of beach litter throughout European coasts (Hanke et al., 2021) so, in this simulation the total inputs to the Mediterranean Sea will be reduced by 60%. The time-period for this simulation will be same as for the baseline (2016 – 2018) so to have a direct comparison.

The second scenario (EU ban) is designed to quantify how much of the actual litter pollution of the Mediterranean Sea could be attributed to EU MS. In shared basins, as the Mediterranean Sea, the amount of cross-country, trans-boundary pollution could be very high (*e.g.*, Macias et al., 2022), so by removing all plastic litter from EU MS, a proper quantification of the real impact of EU pollution could be evaluated/achieved. As above, this scenario is run for the same present-time period (2016 – 2018) to allow a one-to-one comparison with the baseline results.

The third and final scenario is designed to identify and quantify the impact of climate change on litter distribution and accumulation. It is well known that the atmospheric conditions over the Mediterranean Sea will change in the future (e.g., Somot et al., 2020), and that includes a warmer and dryer climate but also a substantial change in wind intensity and direction (e.g., Macias et al, 2015). All these changes will have an impact on the position and strength of oceanic surface currents (e.g., Macias et al, 2018) with obvious consequences for the dispersion patterns and accumulation areas of floating litter (e.g., Macias et al., 2020). This simulation is made with the total input as in the baseline but using the ocean currents predicted for the 2030 time horizon (2028 – 2030) when forcing the ocean model with the same COSMO-CLM RCP4.5 used in the 'nutrients' and 'chemicals' scenarios.

6 Analysis of scenarios

This section is separated in the three pressures considered in the present report (nutrients, chemical contaminants, and plastic litter). For each of the pressures, the analysis of the impacts of the different scenarios are evaluated (when possible) against the ZP ambition targets, also making use of the spatially explicit nature of the applied modelling tools.

6.1 Nutrients

6.1.1 Freshwater environment

The main aim of the present study with regards to nutrients scenario is to evaluate up to which extent a very ambitious set of nutrient-leakage reduction measure could help deliver the ambition of the ZP action to achieve the 50% reduction.

The REF scenario (section 5.1) provides a description of the nutrient pressures and water saving measures in the EU in the present time. So, the relative change between REF and HAS (computed as (HAS-REF)*100/REF) provides a quantification of the possible impact of the different extra measures on the amount of nutrients delivered to EU aquatic ecosystems.

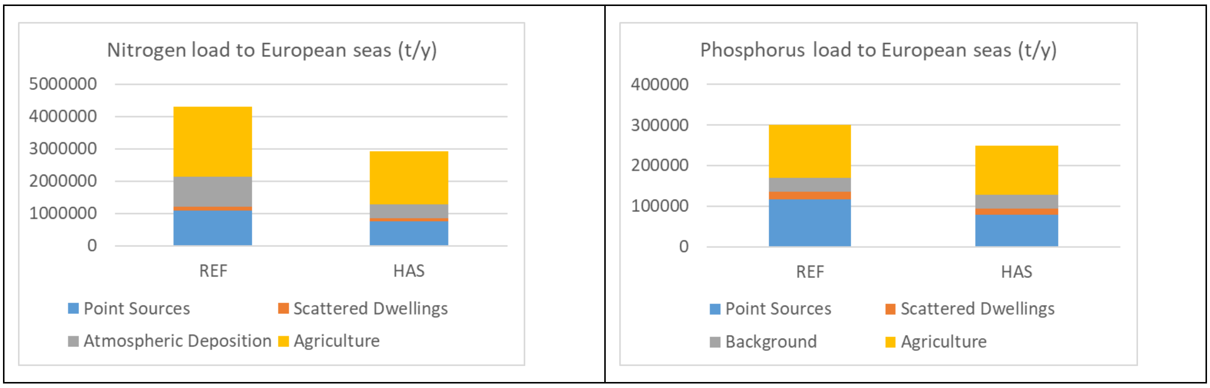
Table 2. Total loads of nutrients to European Seas in both scenarios (REF and HAS) and the relative reduction computed as (HAS-REF)*100/REF

Nutrient	Total load REF (t/y)	Total load HAS (t/y)	% Reduction
Nitrogen	4302251	2937885	-32
Phosphorus	299652	249974	-17

As shown in table 2, the scenario analysis suggests that the very ambitious measures (HAS scenario) will achieve considerable reductions of nutrients, but not the 50% reduction in nutrient delivery to the sea, aimed by the ZP action plan. The modelling framework allows, furthermore, to identify which set of measures are more effective in reducing nutrients leakage (Fig. 1). For nitrogen, the measures aiming at cleaning up the air are the most effective together with agriculture practices considered in the new CAP and Farm to Fork and Biodiversity Strategy. For phosphorus, on the contrary, measures targeting point sources reduction are the largest contributor to the reduction.

It is important to highlight that modelling assessments have uncertainties. The scenarios analysis cannot completely take into consideration the nutrient pollution legacy in soils and groundwater, which delays the response of the natural system, as well as the time necessary to fully implement the measures

Figure 1 Loads of nitrogen and phosphorus to European seas (t/y) for the two evaluated scenarios (REF and HAS) and by activity type



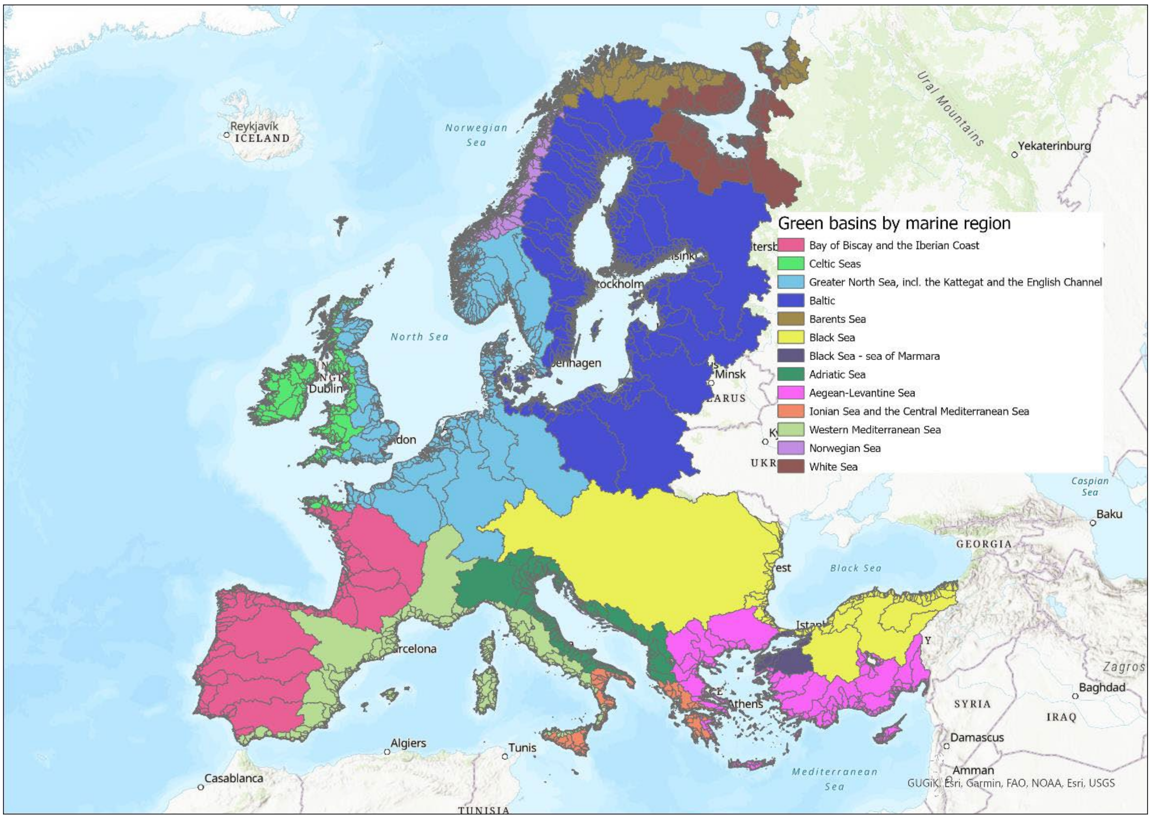
6.1.2 Marine environment

The consequences of the nutrients scenarios on the marine environment can be divided in two categories, 1) changes in the pressure (i.e., on the amount of nutrients each marine region receive) and 2) changes on the environmental status (i.e., how other biogeochemical variables in marine water change as consequence of the reduced amount of nutrients from rivers).

6.1.2.1 Impacts on pressures

The impacts of measures can be analysed regionally, aggregating nutrient load reduction by river basins draining into the different marine regions shown at the map in Fig. 2.

Figure 2 Basins of marine regions used for the impact of the scenario on the pressures (nutrients loads) into the marine ecosystems



As mentioned above, for nitrogen, the application of the measures contemplated in the HAS scenario implies an overall reduction of 32% of the annual loads into European seas (Table 3). This percentage of reduction is regionally dependent (Table 3) and it ranges from 51% to 4% only being close to the ZP target (50% reduction) in the Western Mediterranean Sea region (MWE).

Table 3. Total nitrogen loads to sea foreseen under the REF and HAS scenarios per MSFD marine region (annual average of 5-year period 2026-2030). ABI=Bay of Biscay and Iberian Coast; ACS=Celtic Seas; ANS=Greater North Sea; BAL=Baltic Sea; BLK=Black Sea; BLM=Bosporus and Sea of Marmara; MAD=Adriatic Sea; MAL=Aegean Levantine Mediterranean Sea; MIC=Ionian Sea and Central Mediterranean Sea; MWE=Western Mediterranean Sea.

Marine Regions	Tot Load REF	Tot Load HAS	Total load	Load from Point sources & scattered dwellings	Load from Atmospheric deposition	Load from Agriculture
	(tN/y)	(tN/y)	Change (%) (HAS-REF)*100/REF			
ABI	606760	348605	-43	-52	-76	-30
ACS	309696	240860	-22	-12	-15	-26
ANS	1259738	906985	-28	-22	-53	-23
BAL	482182	325863	-32	-28	-47	-20
BLK	604724	428263	-29	-25	-46	-21
BLM	59759	57077	-4	0	-27	0
MAD	277329	172357	-38	-41	-62	-24
MAL	288982	249972	-13	-10	-33	-7
MIC	80456	44184	-45	-55	-61	-35
MWE	332626	163721	-51	-60	-71	-24
ALL regions	**4302251**	**2937885**	**-32**	**-30**	**-53**	**-24**

All measures, i.e. reducing atmospheric deposition, domestic emissions (point sources and scattered dwelling) and agricultural losses, contribute to decrease nitrogen loads into EU marine ecosystems. Their effectiveness depends on the specific regional characteristics and the relative share of the different sources (atmospheric, point and diffuse) (Table 3).

For phosphorous, the overall reduction for all marine regions with the measures under the HAS scenario is estimated to be 17% (Table 4) with a variability ranging from 53% and 5% reduction depending on the specific regions. It only exceed the ZP target in the Western Mediterranean Sea (MWE) as in the case of nitrogen.

Table 4. Total phosphorous loads to sea foreseen under the REF and HAS scenarios per MSFD marine region (annual average of 5-year period 2026-2030). ABI=Bay of Biscay and Iberian Coast; ACS=Celtic Seas; ANS=Greater North Sea; BAL=Baltic Sea; BLK=Black Sea; BLM=Bosporus and Sea of Marmara; MAD=Adriatic Sea; MAL=Aegean Levantine Mediterranean Sea; MIC=Ionian Sea and Central Mediterranean Sea; MWE=Western Mediterranean Sea.

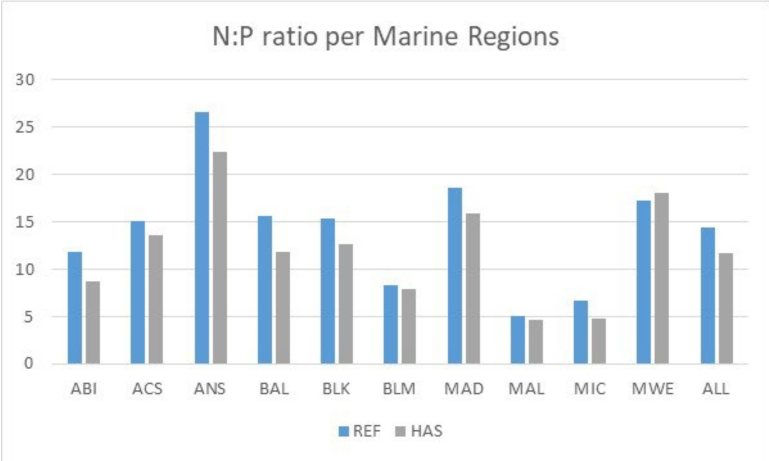
Marine Regions	Tot Load REF	Tot Load HAS	Total load	Point sources & scattered dwellings	Background	Agriculture
	(tP/y)	(tP/y)	Change (%) (HAS-REF)*100/REF			
ABI	51441	40037	-22	-60	0	-11
ACS	20467	17660	-14	-13	0	-15
ANS	47249	40373	-15	-20	0	-8
BAL	30725	27539	-10	-22	0	-3
BLK	39476	33793	-14	-22	0	2
BLM	7170	7170	0	0	0	0
MAD	14853	10813	-27	-49	0	-3
MAL	57006	54232	-5	-15	0	-2
MIC	11940	9300	-22	-79	0	-5
MWE	19325	9058	-53	-73	0	-1
ALL regions	**299652**	**249974**	**-17**	**-31**	**0**	**-6**

Overall, the most effective measures in reducing phosphorous loads to the sea are those related with domestic emissions, with agriculture measures sitting in second (Table 4).

A crucial aspect derived from these results is that measures considered in the HAS are more effective in reducing nitrogen than phosphorus leakages into EU marine ecosystems. This can be explained by the key role of measures to reduce nitrogen atmospheric deposition, but also by the fact that nitrogen is more studied than phosphorus in agricultural scenarios (such as in the model CAPRI).

The N:P ratio is a fundamental parameter for marine primary producers and any imbalance in this quantity can lead to deleterious impacts on the ecosystem, such as proliferation of Harmful Algal Blooms (HAB) (e.g., Billen and Garnier 2007). It is then, very relevant to analyse how this N:P ratio changes from the REF to HAS scenario.

Figure 3 N:P ratio of the riverine loads to each marine region in the two considered scenarios. ABI=Bay of Biscay and Iberian Coast; ACS=Celtic Seas; ANS=Greater North Sea; BAL=Baltic Sea; BLK=Black Sea; BLM=Bosporus and Sea of Marmara; MAD=Adriatic Sea; MAL=Aegean Levantine Mediterranean Sea; MIC=Ionian Sea and Central Mediterranean Sea; MWE=Western Mediterranean Sea.



As the resulting reduction of nitrogen is larger than the one for phosphorus (see Tables 3 and 4), the overall N:P ratio decreases in almost all marine regions around the EU (Figure 3 and Table 5).

Table 5. N:P ratio and change in the riverine load the sea per Marine Regions under the REF and HAS scenarios (average annual values 2026-2030). ABI=Bay of Biscay and Iberian Coast; ACS=Celtic Seas; ANS=Greater North Sea; BAL=Baltic Sea; BLK=Black Sea; BLM=Bosporus and Sea of Marmara; MAD=Adriatic Sea; MAL=Aegean Levantine Mediterranean Sea; MIC=Ionian Sea and Central Mediterranean Sea; MWE=Western Mediterranean Sea.

Marine Regions	N:P ratio	N:P ratio	Change (%)
	REF	HAS	(HAS-REF)*100/REF
ABI	12	9	-26
ACS	15	14	-10
ANS	27	22	-16
BAL	16	12	-25
BLK	15	13	-17
BLM	8	8	-4
MAD	19	16	-15
MAL	5	5	-9
MIC	7	5	-29
MWE	17	18	5
ALL regions	**14**	**12**	**-18**

The mean reduction of the N:P ratio is 14% and the range of variability spread from -29% to +5%. In general, this change indicates that there is more available phosphorus (with respect to nitrogen) in European marine ecosystems. This could create conditions for the proliferation of nitrogen-fixing phytoplankton types (e.g., HAB species) as also suggested by the results of the marine biogeochemical models shown further below.

6.1.2.2 Impacts on status

The biogeochemical models of the JRC-Digital twin were run into the future (2019 – 2030) by imposing the freshwater conditions (quantity and quality) of the HAS scenario, as described above. The comparison of key environmental variables at the end of this simulation (2027 – 2030) with those simulated on the REF simulation allows to evaluate the impact of the measures plus those derived from the climate (e.g., changes in atmospheric forcing to the ocean).

In this case, key environmental indicators are selected considered the criteria defined in Descriptor 5 (D5) of the MSFD (eutrophication) and includes:

— D5C1: Surface concentration of inorganic nutrients (in this case both phosphate and nitrate and their relative ratio)

— D5C2: Surface chlorophyll-a concentration

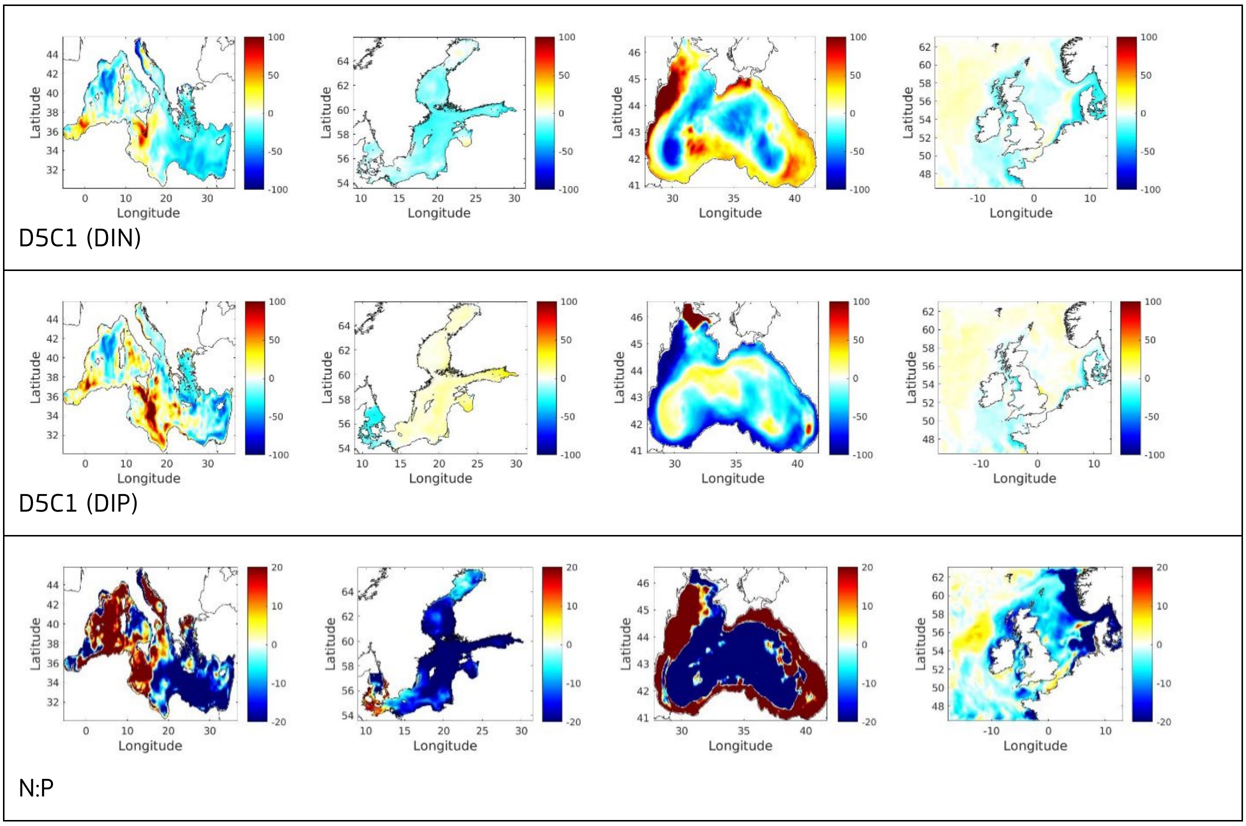
— D5C5: Bottom oxygen concentration

6.1.2.2.1 Impacts of climate plus measures on the different environmental variables

a) D5C1: surface concentration of dissolved inorganic nutrients

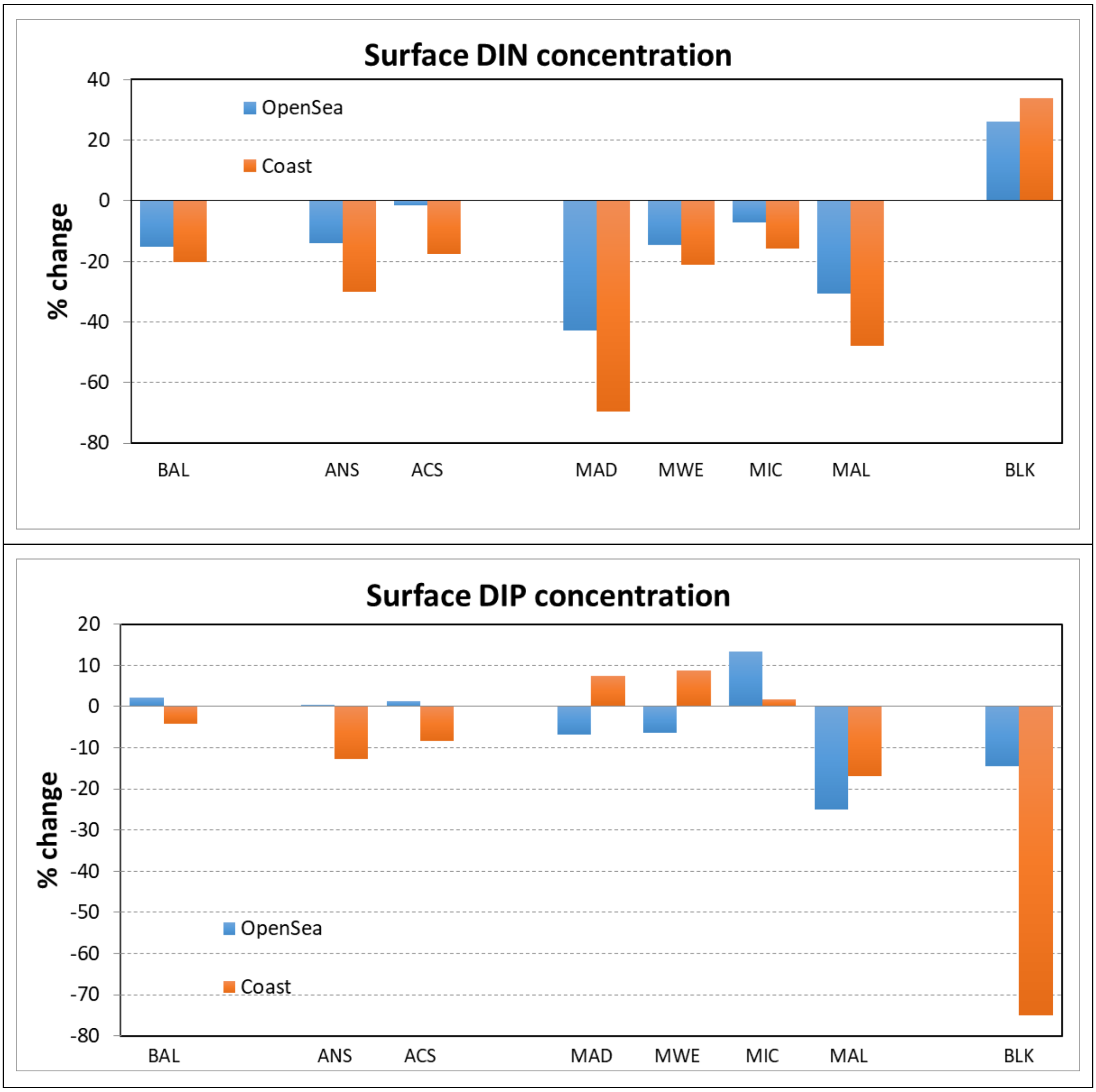
The implementation of the measures contemplated in the HAS scenario provokes a general decrease in the concentration of free dissolved inorganic nitrogen (DIN) in three of the four marine basins included in the present study (upper row in Fig. 4). The only exception was observed in the Black Sea that showed a significant increase everywhere excluding its central region. In the case of dissolved inorganic phosphorous (DIP, mid row Fig. 4), the application of the HAS scenario induces a generalized decrease in the Black Sea, except in the northern tip of the basin, in the Baltic and Mediterranean seas (although certain areas of the central Mediterranean show a significant increase) and almost no change in the North Western Shelf. This happened despite the significant reduction (-17 %) of total load of phosphorus into European marine basins (see Table 4 in the previous section). Consequently, the N:P ratio (a supporting indicator of D5C1) strongly decreases, under the HAS scenario, in the Baltic and North Western European Shelf, strongly increases in the shelf regions of the Black Sea and has contrasting responses in the western and eastern basins of the Mediterranean Sea (Fig. 4, lower row). As it will be discussed below, a relative excess of DIP in many marine regions (i.e., a lower N:P ratio) could foster the blooming of N-fixing species, including those forming harmful algal bloom (HAB), hence potentially deteriorating environmental marine conditions in the EU.

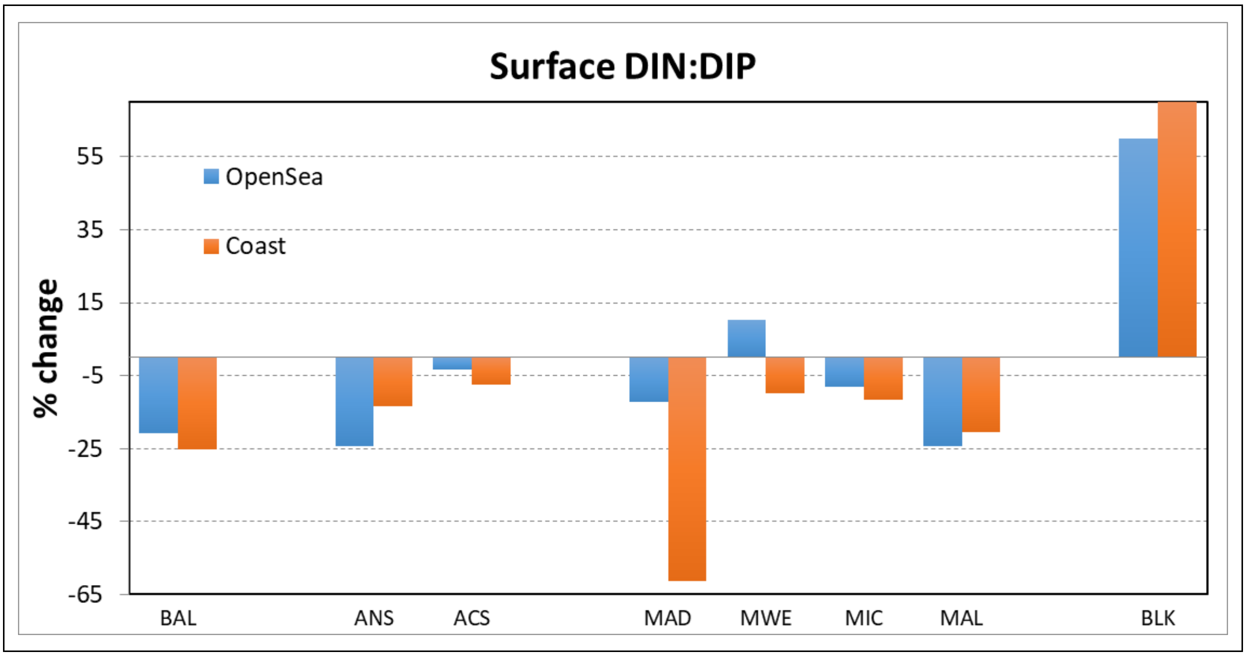
Figure 4 Relative changes (HAS-REF)*100/REF for D5C1 (surface concentration of dissolved inorganic nutrients) indicators in the four investigated EU marine basins



If the spatially explicit data shown in figure 4 is integrated considering the MSFD regions and sub-regions, it is possible to calculate (for coastal/open sea areas) the mean change in the different D5C1 indicators (Fig. 5) per MSFD area. The application of the HAS measures implies a significant reduction (between 2 and 15%) of DIN for all MSFD regions except for the Black Sea, where DIN increases by 20% (Fig. 5, upper panel). DIP, on the other hand, varies among the MSFD regions, showing a strong reduction (up to 25%), for example, in the Black Sea and Eastern Mediterranean, and increase (up to 12%), e.g., in the Central Mediterranean Sea (Fig. 5, central panel). Despite these diverse individual changes, the N:P ratio tends to decrease in all MSFD regions except for the Black Sea (Fig. 5, lower panel).

Figure 5 Barplot of relative change of the different D5 indicators on the different MSFD regions (coastal/open sea).

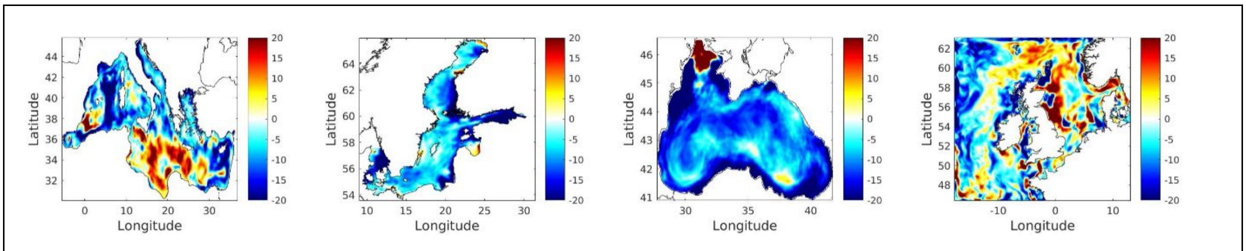




b) D5C2: surface Chlorophyll-a (Chl-a) concentration

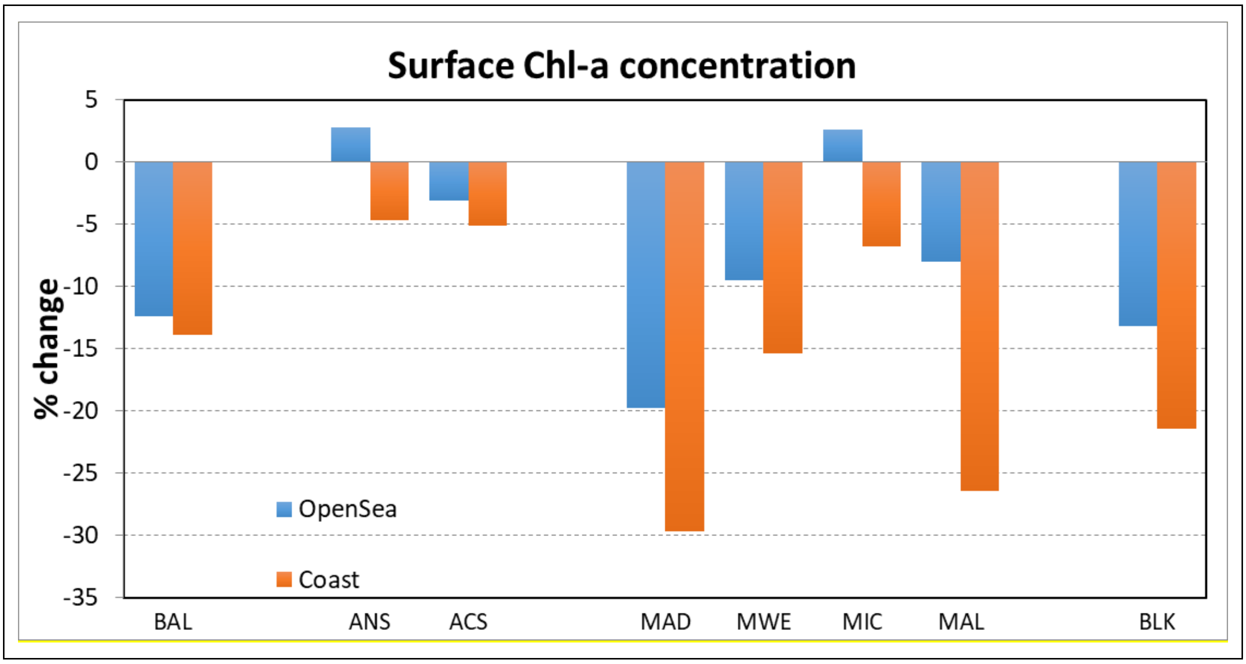
The HAS scenario creates mixed responses also when Chlorophyll-a (Chl-a) concentration is considered (Fig. 6). In the Baltic Sea, the majority of the regions shows a decrease in Chl-a levels. In the case of the Black Sea, there is a very distinctive pattern of change (Fig. 6) with Chl-a values strongly increasing in the North Western Shelf region and substantially decreasing elsewhere. In the Mediterranean Sea, there are regions where Chl-a decreases, such as the Gulf of Lion or the northern Adriatic Sea and others where Chl-a increases such as the central and eastern Mediterranean. We need to consider, however, that in those two last regions the average Chl-a concentration is very low (strongly oligotrophic environments) so a minor change between HAS and REF will appear as relatively large percent change. The North Western European Shelf also shows a diversity of responses with the Celtic Sea predominantly showing a decrease while the North Sea presenting substantial increases.

Figure 6 Relative changes (HAS-REF)*100/REF for D5C2 indicator (surface Chlorophyll-a concentration) in the four investigated EU marine basins



When Chl-a data are integrated spatially into the MSFD regions (Fig. 7), it is easier to disentangle the complex patterns shown above. In most regions the application of the HAS scenario implies a reduction of mean Chl-a values ranging from 28% (in the Adriatic Sea) to 5%. Certain open sea regions (such as the North Sea and Central Mediterranean) show small relative increases of less than 5%. In the Black Sea, even if in most of the basin Chl-a decreases (Fig. 6), the substantial increase simulated for the north-western continental shelf makes the mean change to be positive, yet with different amplitude, both in the open-sea and in the coastal regions.

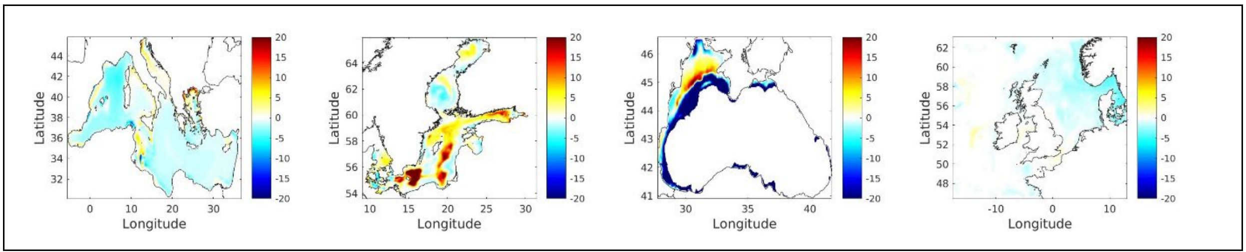
Figure 7 Barplots of relative change of the Chl-a on the different MSFD regions (coastal/open sea).



c) D5C5: Bottom oxygen concentration

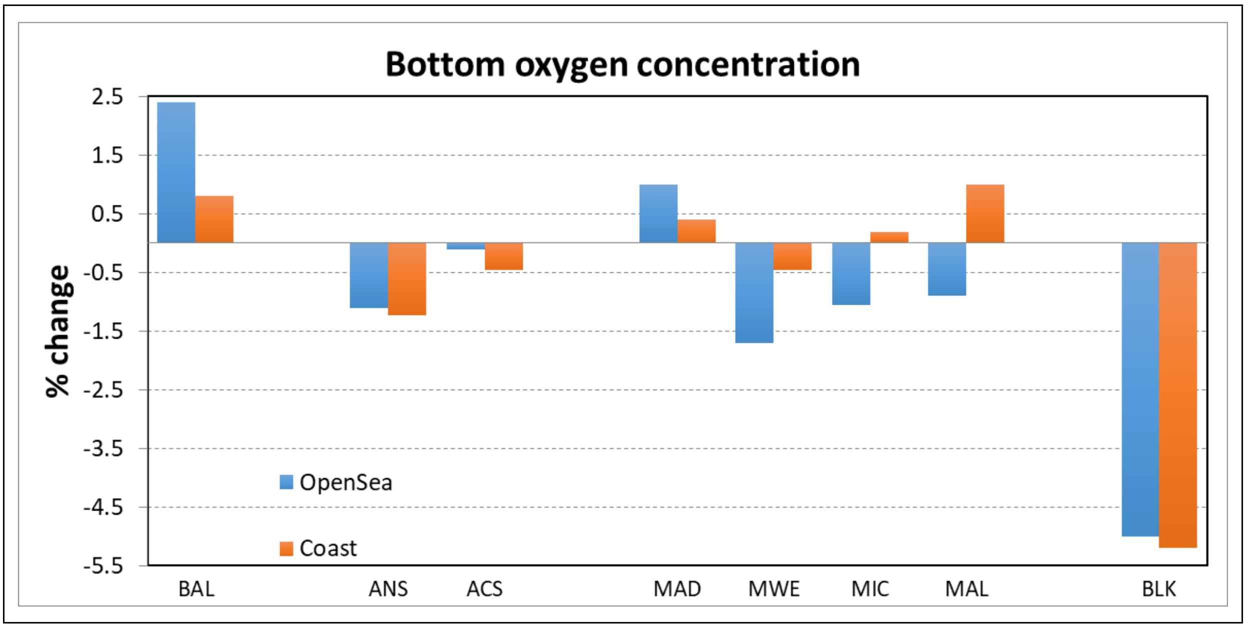
Bottom oxygen levels are governed both by water temperature (dissolution and vertical stratification) and by export production (consumption), mostly reflected by phytoplankton biomass (i.e., Chl-a). The correlation between changes in Chl-a and changes in bottom oxygen is clearly apparent in Fig 8. The positive impact of HAS on this indicator is clearer in the Baltic Sea and north-western continental shelf of the Black Sea. For the Mediterranean and North Sea basins the percent changes are smaller, and the sign of change is region-dependent.

Figure 8 Relative changes (HAS-REF)*100/REF for D5C5 indicator (bottom oxygen concentration) in the four investigated EU marine basins



.Bottom oxygen increases in the Baltic Sea and most coastal areas of the Mediterranean. It, however, decreases in most open sea regions of the Mediterranean Sea, in the North Sea and in the Black Sea (Fig. 9).

Figure 9 Barplots of relative change of the bottom oxygen on the different MSFD regions (coastal/open sea).



Another way to visualize the impacts on the ecological status of EU marine waters, under the HAS scenario, is shown in Table 6 below. Overall, all D5 related indicators show an improvement in the HAS scenario with respect to the REF (lower row of Table 6), with only bottom oxygen showing a marginal degradation. The largest improvements are simulated for DIN followed by DIP and Chl-a. Regionally, most of the descriptor 5 criteria show improvement, except for the Black Sea (as already commented), where only DIP decreases significantly.

Table 6. Percentage of change of the different D5 criteria for the diverse MSFD marine regions/sub-regions due to the application of ambitious measures and climate change (HAS scenario). Green indicate substantial improvement (over 5%), red substantial deterioration (over 5%) and yellow small relative changes (less than 5%).

	DIN		DIP		Chl-a		Bottom oxygen	
Region	Open	Coast	Open	Coast	Open	Coast	Open	Coast
BAL	-15.2	-20.2	2.07	-4.13	-12.4	-13.9	2.4	0.8
ANS	-14.1	-30.1	0.51	-12.7	2.8	-4.7	-1.1	-1.23
ACS	-1.6	-17.5	1.36	-8.35	-3.1	-5.1	-0.1	-0.45
MAD	-42.8	-69.7	-6.9	7.5	-19.8	-29.7	1	0.41
MWE	-14.6	-21.1	-6.35	8.82	-9.5	-15.4	-1.7	-0.45
MIC	-7.1	-15.9	13.4	1.78	2.6	-6.8	-1.05	0.2
MAL	-30.7	-47.9	-25	-16.9	-8	-26.46	-0.9	1
BLK	26	33.7	-14.4	-75	-13.2	-21.4	-5	-5.2
ALL	-11.6	-28.4	-0.03	-8.7	-10.3	-17.2	-0.1	-0.25

This analysis highlights that it is very important to consider the impact of nutrient reduction measures on the N:P ratio. Indeed, depending on the limiting nutrient of the receiving marine regions, the same reduction in nutrient load may lead to diametrical responses in the local N:P ratio, with potential consequences on the functioning of the ecosystem (i.e., shift in phytoplankton community, development of HABs, reduced grazing and trophic efficiency). This is exemplified by the contrasting changes of N:P ratio in the western Mediterranean and central Black Sea (N-limited basins) and the eastern Mediterranean Sea, shelf regions of the Black Sea and partially the Baltic Sea (P-limited basins).

A decrease of the N:P ratio means that there is an excess of P in the planktonic ecosystem with respect to N. After N and P are used at a fixed proportion of roughly 16 (Redfield, 1934, 1958) by spring bloom-forming phytoplankton groups (e.g., diatoms) this excess of P can be used by phytoplankton species able to fix atmospheric nitrogen, forming potentially harmful blooms.

Indeed, large cyanobacteria blooms (a N-fixing group) have already been observed in the Baltic Sea after a strong reduction of nitrogen loads in the mid 90's. After later increases in the nitrogen loads a few years later, the cyanobacteria bloom in the Baltic Sea strongly reduced (Kahru et al., 2000). It has been postulated that this very favourable period for cyanobacteria blooms was created by the excess phosphate compared to nitrogen that inhibited the growth of most phytoplankton groups that typically out-compete and maintain under control cyanobacteria population (Stal et al, 2003; Vuorio et al., 2005)

Another recent example of the impact of N:P alterations in phytoplankton community is provided by Liu et al. (2022). These authors followed the biogeochemical conditions of the Yangtze estuary for over 30 years and observed an increase of dinoflagellates abundance and blooming coincident with a decrease in the N:P ratio of riverine inputs to the estuary. Also in this case, alteration of the nutrients ratio was linked to management actions such as reservoirs and wastewater treatments plans constructions.

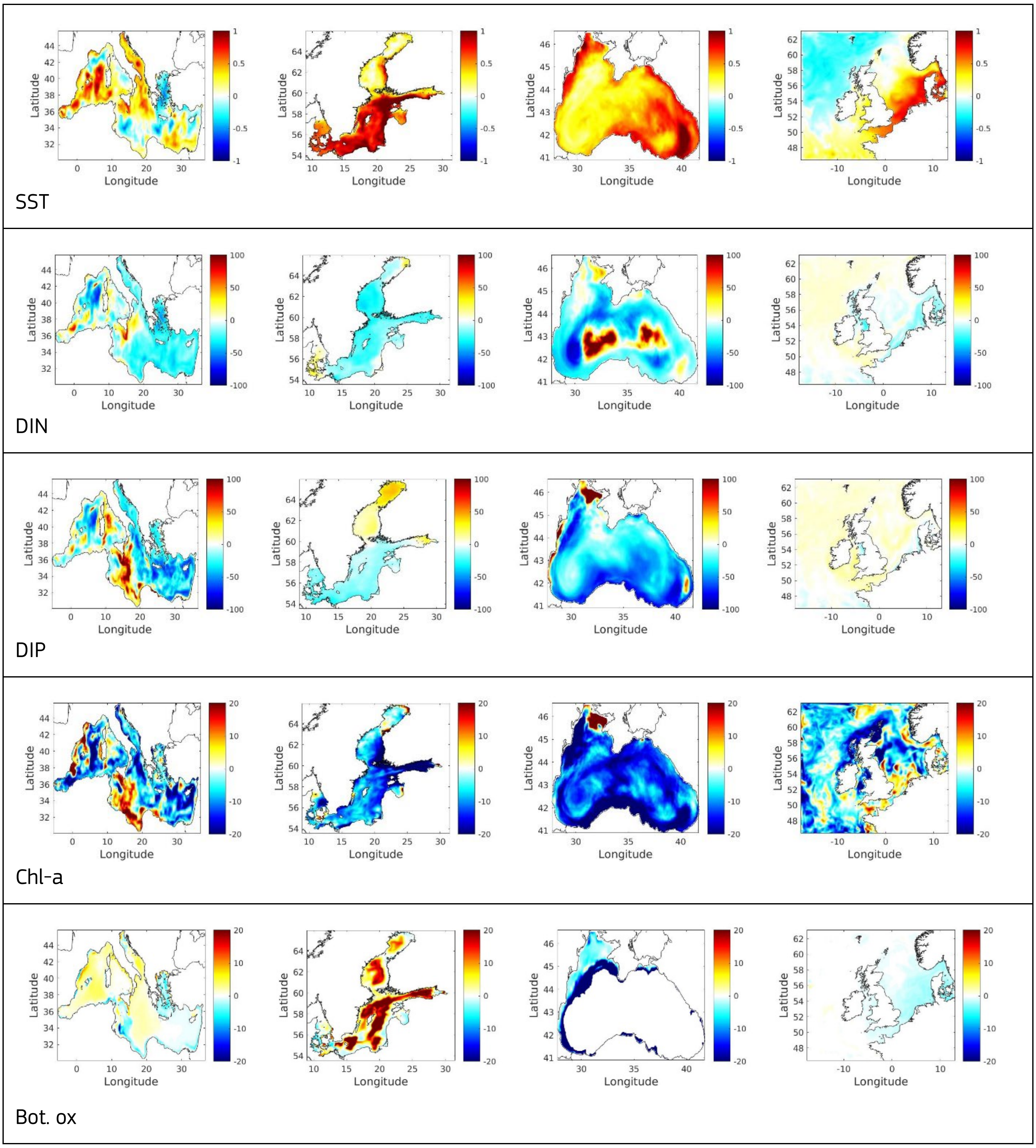
The biogeochemical models at the JRC-DT are not fully suited to represent the presence of many potentially harmful phytoplankton groups (e.g. toxic dinoflagellates) so we cannot expect to be able to simulate their blooming with the current tools. However, all used biogeochemical models includes different phytoplankton functional types making a distinction between 'large' (i.e. diatom-like) and 'small' (i.e. cyanobacteria-like) groups, so it is possible to compute if the relative contribution of small versus large phytoplankton changes in the HAS versus the REF scenarios. Indeed, for all EU basins but for the Baltic the application of the HAS scenario provokes an increase of the smaller phytoplankton ranging from +32% in the Mediterranean Sea, to +6% in the Black Sea and +3% in the North West European Shelf. In the Baltic Sea the difference is very small (-0.8%) and likely not significant. All these results are indications of the potential negative impacts on the eutrophication status of EU marine ecosystems if their N:P ratio decreases as results of the applied measures.

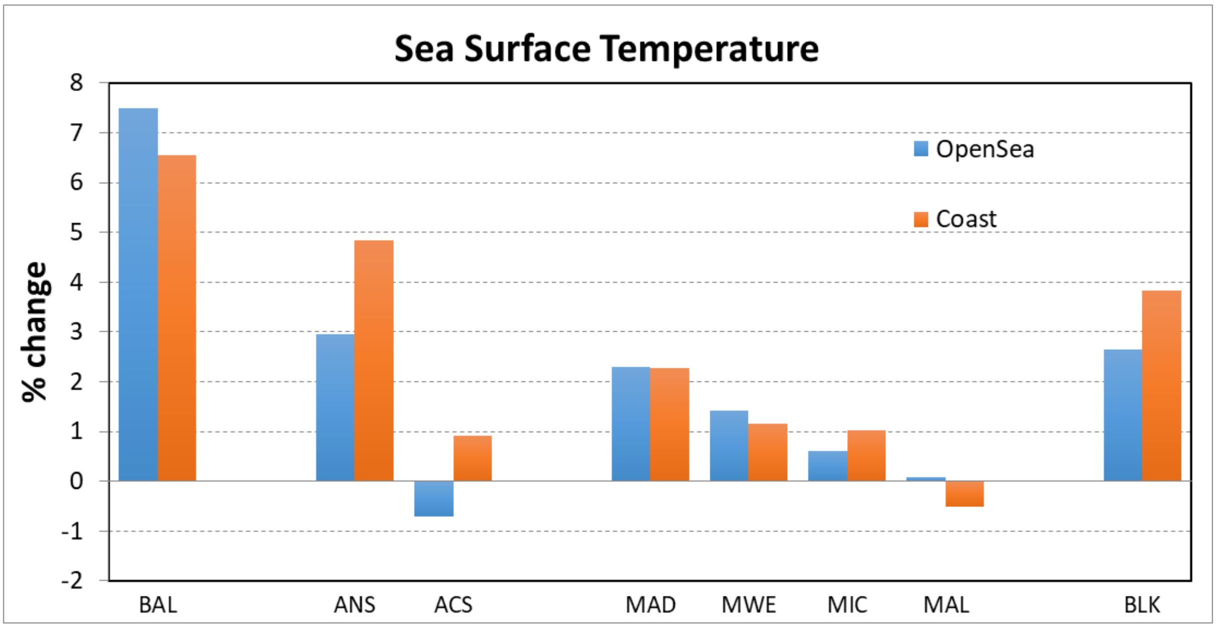
6.1.2.2.2 Impacts of climate on the different indicators:

Given that the HAS scenario is projected into the future covering the time-period 2019 – 2030 (see details in section 5.1), it is possible to isolate the impacts of the changing climate on the different ecosystem indicators by making the comparison of the status of marine basins at the beginning of the simulation (2019-2021) and at the end (2028 – 2030). Climate-change associated impacts are not explicitly within the remit of the ZP ambition, but it is something necessary to consider (and quantify if possible) when performing outlooks into the future. A word of caution is needed here as we only tested the impacts simulated by a single global circulation model and within a unique emission scenario. This is not a wide-enough analysis to properly assess climate change impacts (needing multiple models and emission scenarios, i.e. an ENSEMBLE approach) but we were limited by time and computational resources in our analysis. As a result, conclusions in this sub-section should be treated with due caution.

As it could be expected, all EU marine basins show an increasing trend in surface temperature (SST, upper panel Figs. 10 and 11), although certain regions within the Mediterranean and at the North Sea boundaries show a weak cooling trend. Changes in SST will provoke alterations of the water-masses vertical structure and stability, which will, in turn, induce changes in vertical mixing and upper water column fertilization (particularly in open-sea regions). In general, for all EU basins and for both DIN and DIP there is a decreasing trend, which corresponds with a generalized increase in vertical stratification (Figs. 10 and 11). This is not true for DIP in some regions of the northern Black Sea and the central Mediterranean where an increase in DIP is simulated (see Fig. 11). Accordingly, the generalized change of Chl-a due to climate change is a decrease with punctual increases in scattered regions in all basins.

Figure 10 Absolute changes in SST (°C) and relative changes (%) in D5 criteria due to climate change (2019ish to 2030ish under rcp4.5 scenario).




Sea Surface Temperature
% change
8
7
6
5
4
3
2
1
0
-1
-2
OpenSea
Coast
BAL
ANS
ACS
MAD
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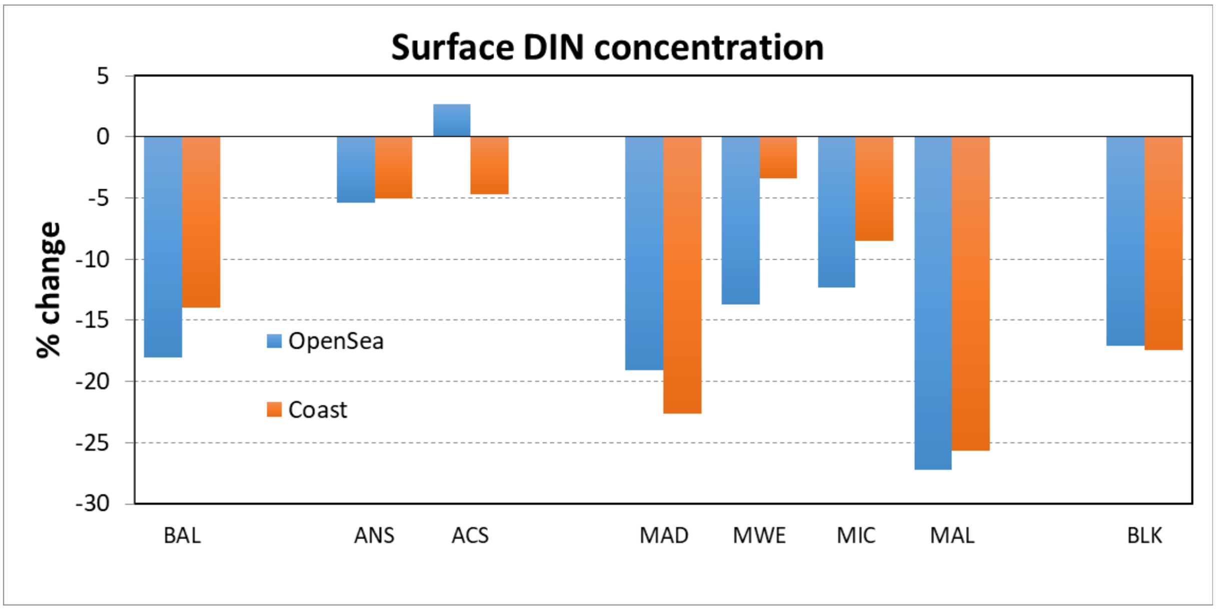
Surface DIN concentration
% change
5
0
-5
-10
-15
-20
-25
-30
OpenSea
Coast
BAL
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MIC
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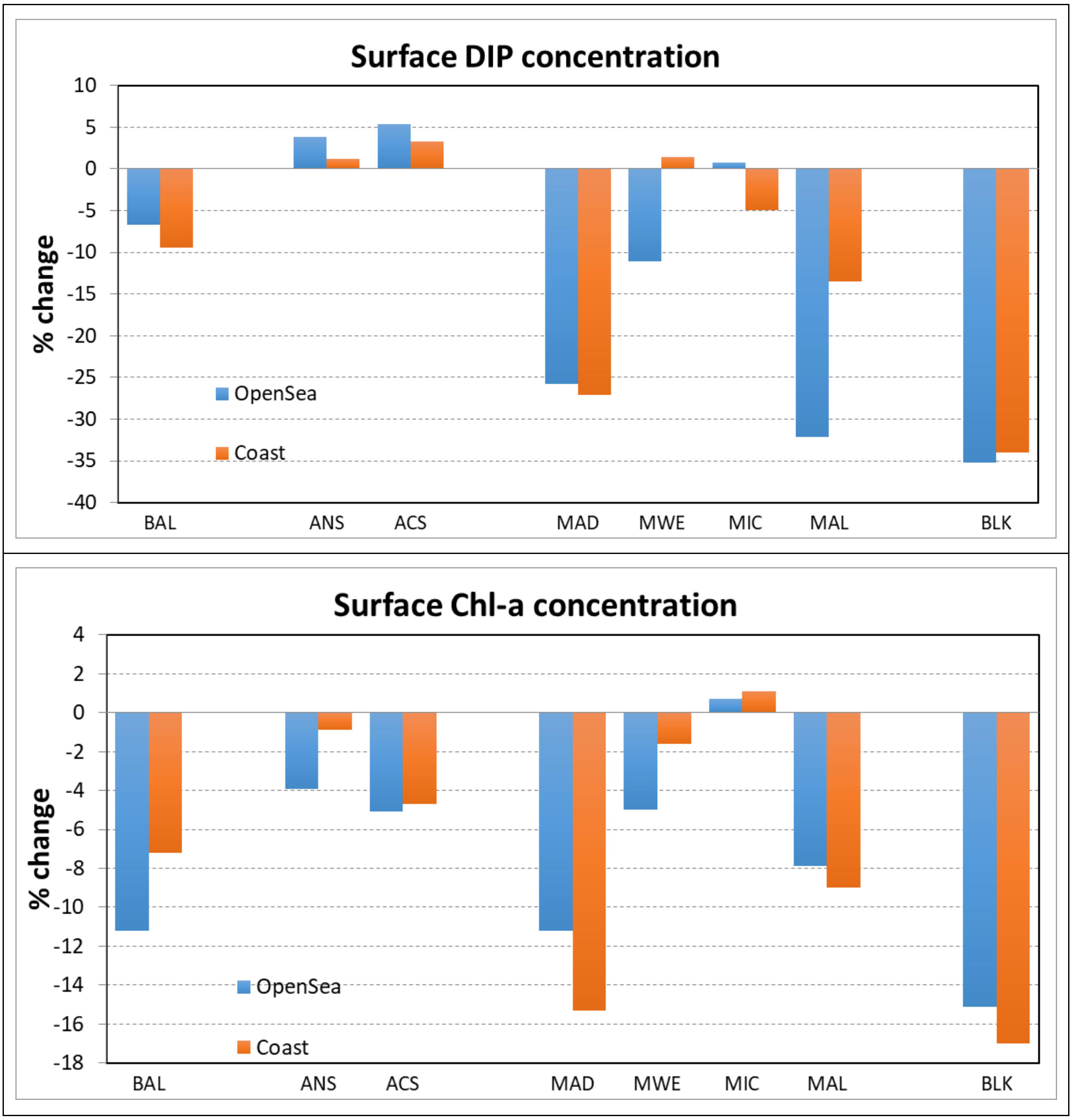
Surface DIP concentration
% change
10
5
0
-5
-10
-15
-20
-25
-30
-35
-40
OpenSea
Coast
BAL
ANS
ACS
MAD
MWE
MIC
MAL
BLK
Surface Chl-a concentration
% change
4
2
0
-2
-4
-6
-8
-10
-12
-14
-16
-18
OpenSea
Coast
BAL
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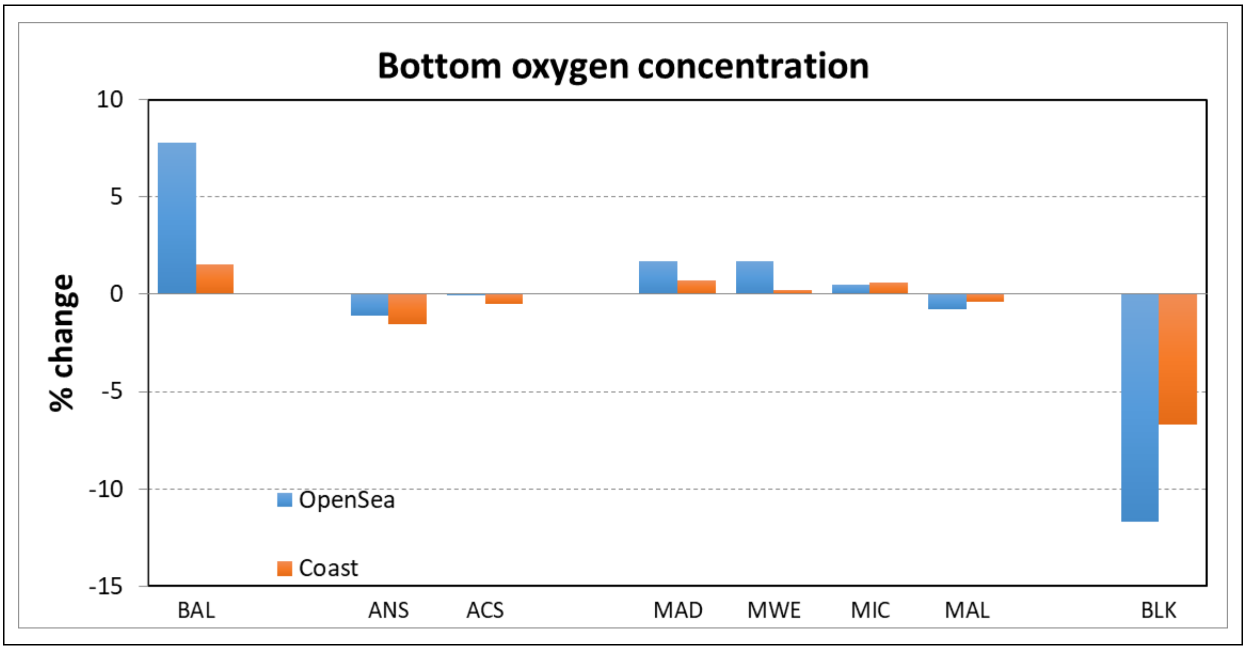


Figure 11 Barplots of relative change (%) of SST and the different D5 criteria per MSFD region/sub-region (coastal/open sea) due to climate change (2019ish to 2030ish under rcp4.5 scenario).

Table 7. Percentage of change of the different D5 criteria for the different marine basins due to the climate impacts alone. Green indicates substantial improvement (over 5%), red substantial deterioration (over 5%) and yellow small relative changes (less than 5%).

	DIN		DIP		Chl-a		Bottom oxygen	
Region	Open	Coast	Open	Coast	Open	Coast	Open	Coast
BAL	-18	-14	-6.7	-9.4	-11.2	-7.2	7.8	1.53
ANS	-5.4	-5	3.8	1.2	-3.9	-0.9	-1.1	-1.53
ACS	2.7	-4.7	5.4	3.3	-5.1	-4.7	-0.05	-0.5
MAD	-19.1	-22.6	-25.8	-27.1	-11.2	-15.3	1.7	0.7
MWE	-13.7	-3.4	-11.1	1.4	-5	-1.6	1.7	0.2
MIC	-12.3	-8.5	0.72	-5	0.7	1.1	0.5	0.6
MAL	-27.2	-25.7	-32.1	-13.5	-7.9	-9	-0.8	-0.4
BLK	-17.1	-17.4	-35.2	-34	-15.1	-17	-11.7	-6.7
ALL	-6.4	-6	-5.3	-4.2	-9.8	-8.6	1.4	-0.25

If the mean relative changes due to climate change (Table 7) are compared with those presented above due to measures + climate change (Table 6), it is possible to observe that the number of criteria showing 'not significant changes' have substantially increased (yellow cells in Table 7) and that relative bigger changes are usually simulated for the open sea regions of the different marine basins. This is indicating that climate change impacts on the eutrophication status of marine ecosystems are mostly mediated by the alteration of the physical structure of the water column and, particularly, the alteration of the vertical mixing and horizontal ocean currents (e.g., Holt et al., 2016). Changes in mixing intensity are, typically, more relevant to open sea regions than in coastal areas, which are constrained by a wider range of physical processes (e.g., tides, coastal currents, etc.).

6.1.2.2.3 Impacts of measures on the different descriptors:

Making use of the calculations for the different marine regions derived from the two simulations shown in Table 6 and Table 7 ('measures + climate change' and 'climate change'), it should be possible to estimate the isolated impact of the measures alone, without considering the accumulated impacts of climate change. However, this calculation is just an approximation, and it is not suitable for precise, region-by-region analysis. Henceforth, it is only shown below for the aggregated eutrophication criteria (in the whole EU) and only discriminating between open sea and coastal areas (Fig. 12)

Figure 12 Relative changes in the different D5 descriptors for all the marine regions and separated by impacts of climate change (CC) + measures (blue bars), only by climate change (orange bars) of only by measures (grey bar).

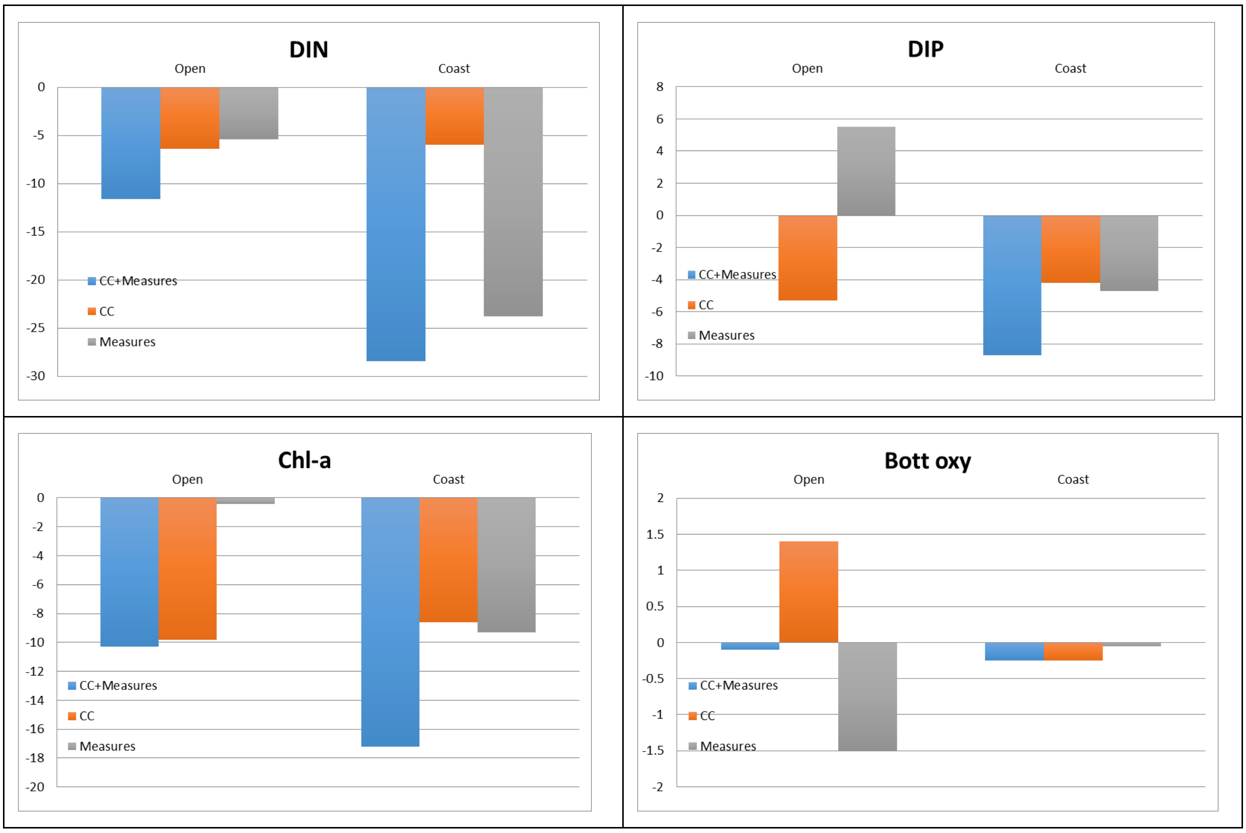


Figure 12 indicates that the impacts of the measures alone (with the cautious due the calculation methodology applied) are much more relevant in coastal regions than in open sea (a logical result). It is also worth mentioning that for DIP, the isolated impact of the measures results in an increase in open sea regions. This could be an artefact of the calculation method (see above) or a likely result of the strong reduction in DIN induced by the application of the measures (see discussion on the N:P ratio above). It could, then, be another clear indication of the need for smart-reduction measures when fighting nutrient contamination of EU aquatic ecosystems.

6.2 Chemical pollutants

The presence of water-soluble persistent substances in the aquatic environment has raised environmental concern due to their potential toxicity, sometimes even at low concentrations. Because of their hydrophilic character, many of them are very mobile in the aqueous phase and consequently can be transported by the river waters to the marine environment, which is their final recipient. Some of these so-called micro pollutants have been detected in all water environments and their removal in conventional wastewater treatment plants is known to be incomplete (Castaño-Trias et al., 2020). Recently, it has become increasingly clear that marine and coastal ecosystems face threats from multiple anthropogenic activities (United Nations Environment Programme (UNEP), 2012). Persistent substances can act as additional stressors for marine ecosystems already affected by climate change, eutrophication, and overfishing (Hylland et al., 2017). The release of substances, such as pesticides, due to agricultural activities, can cause freshwater, groundwater, and marine water pollution through drainage, leaching, runoff and surface deposition (Radović et al., 2015; Chapman et al., 2016). Pharmaceutical substances can be introduced into aqueous systems through industrial waste, farming practices and domestic wastewater (Patel et al., 2019).

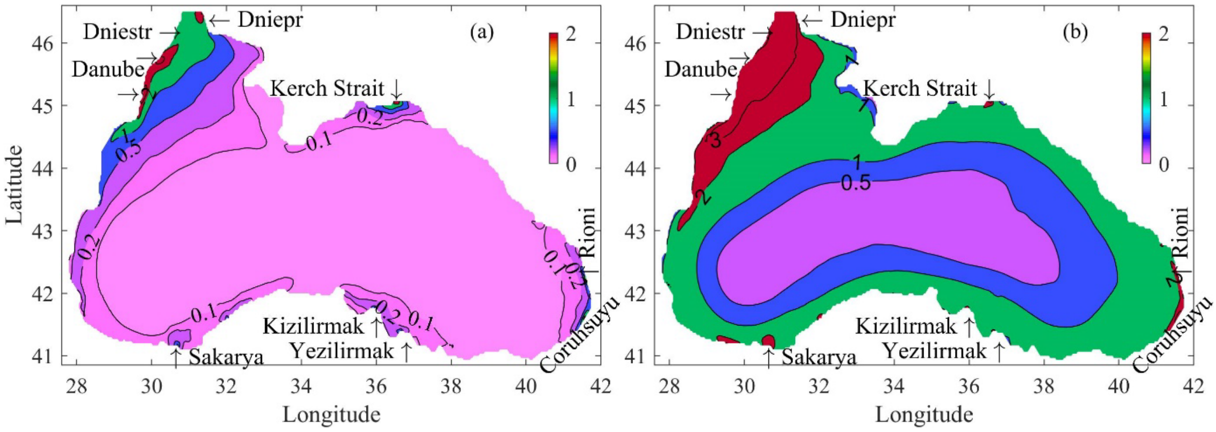
Despite the difficulty in assessing distribution, dispersion and accumulation of the myriad compounds present in the marine environment, the JRC-DT can be used to evaluate the dispersion and potential accumulation of any given chemical substance if they are relatively inert and water soluble and given that their concentration in the rivers and half-life in the natural environment are known (for full details of the methodology the reader is referred to Miladinova et al., 2022). The substances listed in Table 1 are, then, grouped into compound classes with comparable properties and one representative member of each group is selected to perform the simulations presented below.

6.2.1 50% reduction of chemical pollution

One main aim of the ZP ambition is to reduce the use of chemical pesticides by 50%. Hence, we designed this first set of simulations to explore how a 50% reduction from 2020 to 2030 of the present-day river concentration of the selected substances (see Table 1) will influence and determine their distribution patterns in the Black Sea.

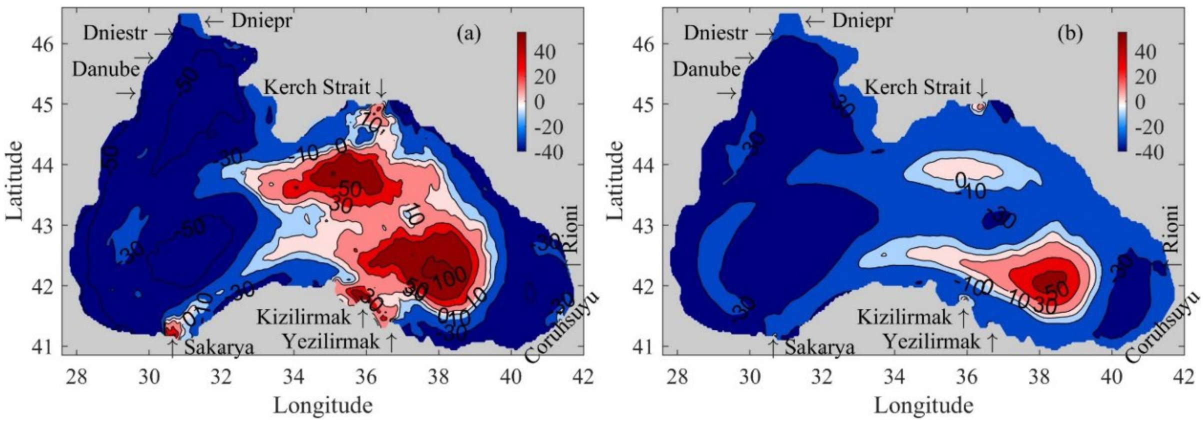
This modelling exercise is focused on water-soluble persistent substances that do not tend to bioaccumulate. The six substances (pharmaceuticals and pesticides) considered here are consolidated into two cases according to their persistence in seawater, which is measured by their half-life (DT50). Diuron, diclofenac and caffeine are supposed to have DT50=50 days, while for terbuthylazine, sulfamethoxazole and carbamazepine DT=360 days. The reference (REF) runs started in 2010 and finished in 2018. The concentration of each substance in the rivers is assumed to be 1 ng m-3 for the reference run. In the Danube, in 2 small Romanian and in 2 small Bulgarian rivers, this concentration decreases linearly to 0.5 from 2019 to 2030 in the future HAS scenario (highly ambitious scenario) for which the same climate simulation used before (i.e., MPI-rcp4.5) is applied. In other rivers, the concentration of the substance is maintained at 1 ng m-3 for HAS.

Figure 13 Vertically integrated substance concentrations (ng m^{-2}) over 2016 – 2018 (REF) if the concentration in the rivers is 1 (ng m^{-3}). (a) DT50=50 days and (b) DT50=360 days. To facilitate reading, the tracer color bar is set in the range 0 – 2 ng m^{-2}. The locations and names of the biggest Black rivers and the Kerch Strait are also shown.



Substances with low DT50 are usually only found near the mouth of a river (Fig. 13a). The zone of influence is not visible for the smaller rivers even for substances with high DT50. The spreading of substances coming from the northern shelf rivers and Azov Sea can be seen. Rivers in the northern shelf have very high discharges and the spreading of the river-borne materials depends on complex hydrodynamic conditions (Miladinova et al., 2020a). The major factors that control the northern shelf river plumes include winds direction and frequency, the magnitude of river discharge and the strength of the currents. For the run with DT50 = 50 days, only the north western part of the shelf and the area near the Kerch Strait accumulate more than 1 ng m^{-2} of the substance coming with the rivers (Fig.13a). The substance concentration in the deep part of the basin is below 0.5 ng m^{-2}. Increasing the substance persistence DT50 about 7 times leads to about a tenfold increase in the open sea concentration (Fig. 13b).

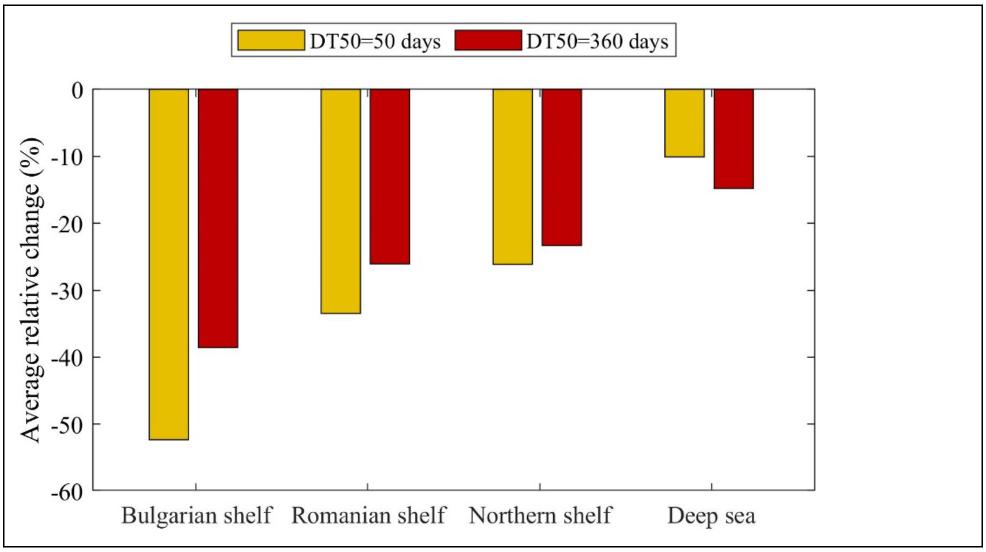
Figure 14 Relative difference (%) between the vertically integrated concentrations between the period 2016 – 2018 (REF) and the period 2028 - 2030 (HAS). (a) DT50=50 (day) and (b) DT=360 (day).



The relative difference between HAS and REF is calculated as a difference between the vertically integrated concentrations in the period 2028 - 2030 (HAS) and in the period 2016 - 2018 (REF) divided by the REF concentrations. In the future simulations with 50% reduction of the chemical substance in the rivers (Fig. 14), a strong reduction of concentration is expected for both cases (high and low persistence) in the shelf regions of the sea. This reduction could be as high as 52.3% on the Bulgarian shelf for DT50=50 days (Fig. 14a). As the reduction of the substance concentration in the EU rivers reaches 50% at the end of 2030, the greater reduction of the substance in the Bulgarian shelf is also supported by climate change. Climatic changes affect the river-born substance distribution mainly by changing the river plume pathway and basin general circulation (Miladinova et al., 2020a). In both cases, there are regions of the central and eastern Black Sea where there is a relative increase of the substance concentration (Fig. 14). This increase is related with changes in the general circulation of the basin induced by the different atmospheric forcing in the future scenario. The relative increase of the difference for DT50=50 days does not affect the quality of the marine water due to the very low substance concentration in the deep sea (less than 0.1 ng m^{-2}, see Fig. 13a).

Obviously, the reduction in diuron, diclofenac and caffeine is more pronounced for the shelf areas (Fig. 15), as their occurrence in the deep sea is less likely due to their lower degradation rate. We predicted a higher decrease in the average concentration in the deep sea for a substance with DT50 = 360 days, although there are areas with a positive change in the eastern part of the sea. Again, the Bulgarian shelf reaches the highest reduction of the substance among all considered areas. In summary, terbuthylazine, sulfamethoxazole and carbamazepine are forecasted to decrease from 14.8% in the deep sea to 23-38 % on the shelf areas.

Figure 15 Average relative change (%) in the HAS scenario compared to the REF scenario in different areas for DT50 = 50 and DT50 = 360 days. The change is calculated as the mean value of relative DIFF (see Fig. 14) in the particular area. The shelf areas are considered to cover zones with depth less than 50m.

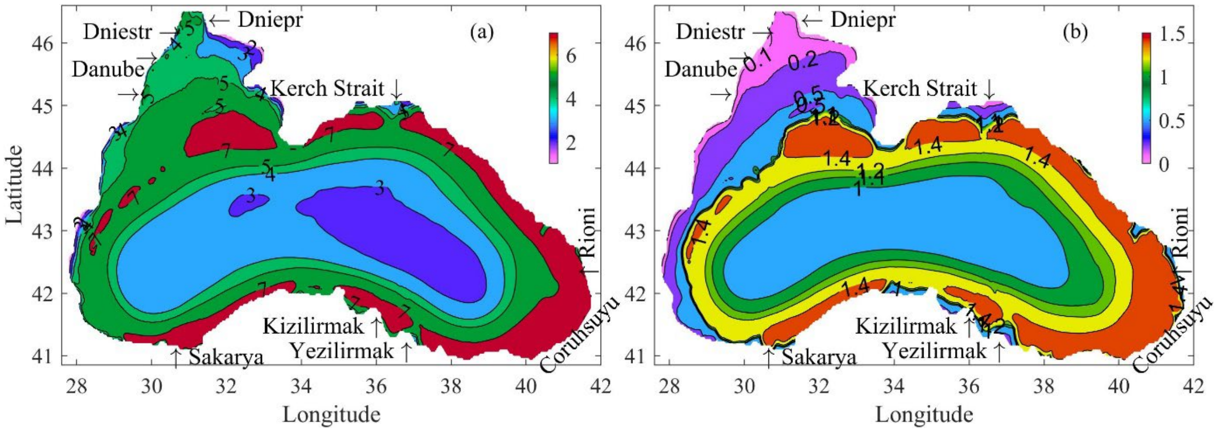


6.2.2 Long-term impacts of persistent substances (legacy or natural inertia)

When dealing with natural systems, it is always important to consider the 'reaction time' or 'inertia' of the involved ecosystem. This is clearly the case for many eutrophic lakes when a drastic nutrient-reduction does not lead to an obvious environmental improvement or, in the present example, the fact that long-time banned substances such as some pesticides (e.g., simazine and atrazine) are still being found in samples from all EU seas.

To explore the long-term accumulation of very persistent substances after their complete ban and subsequent disappearance from the rivers, an additional scenario simulation was performed with the JRC-DT. This simulation addresses the three stages of a substance from river loads that is subjected to a regulation: full load, phasing-out period after ban and finally zero load. The fact that after decades of banning, atrazine and simazine are still detectable highlights the need to assess the environmental behaviour of these pesticides on longer time scales. A degradation time, based on short-term laboratory experiments, is frequently used to characterise substance persistence despite questions about the applicability of the methods. Simazine and atrazine were widely used and their degradation times in seawaters appear to be much longer than found in lab-based experiments. The Black Sea basin is semi-enclosed, so after years of low use, the two substances are still found in the basin due to their high persistence. Based on the evidence of atrazine and simazine concentrations in the shelf and deep areas over the years (EMBLAS project; https://emblasproject.org), we estimated the approximate degradation time of these substances (DT50~2000 days). A simulation with DT50 equal to 2100 days is performed. The concentration of a substance in all rivers is set to 1 from 1995 to 2004; then, is decreased linearly from 1 to 0 in 2004-2008 and finally is set to 0 from 2009 to 2019. We suppose that these assumptions are appropriate and close to the physical-chemical properties and banning conditions of atrazine and simazine.

Figure 16 Vertically integrated substance concentrations (ng m-2) for DT50=2100 days: (a) average over 2000 – 2004; (b) average over 2019. The substance concentration in all rivers is 1 ng m^{-3} in 1995 – 2004, then decreases to 0 until the end of 2008 and is 0 until the end of simulation period in 2019.



Due to the low degradation rate (DT50=2100 days), the simulated substance achieves very high mean concentrations of about 5-7 ng m^{-2} (Fig. 16a) in the convergent and shelf areas (with depth<1500m). After 15 years of banning, the substance concentration is still the highest in the convergent area, while it decreases significantly in the north-western coastal and shelf areas (Fig. 16b). Note that in the deep sea, the concentration is still about 1 ng m^{-2}, which is higher than the deep sea concentration for DT50=360 days and without any banning (Fig. 13b). During a Black Sea survey in November 2013 (Orlikowska et al., 2015) estimated the average value of 40.5(±3.1) (ng m^{-3}) for atrazine and 9.4(±0.6) (ng m^{-3}) for simazine. Both values are close to the deep sea concentrations reported by the EMBLAS projects used to develop our model. Namely, the average concentration over 2016-2019 for atrazine is 31.55 (±16.45) (ng m^{-3}) and for simazine 13.51 (±7.86) (ng m^{-3}). Moreover, both pesticides are rather uniformly distributed over the shelf and offshore surface waters in 2013 and 2016-2019 like in Fig. 16b.

In summary our model can represent the fate and distribution of atrazine and simazine in the Black Sea over long period of time. The main conclusion includes the estimation of their degradation time, which is found to be 4 times longer than expected (US EPA, 2006). Their extremely high persistency is the main reason that these pesticides are still detected in the sea. The leaks from the non-EU rivers do not contribute significantly for their accumulation in the deep sea. The observed concentrations are below the EQS values and continue to decrease slowly, which is why we can conclude that there is a strong positive impact of the ban measures. However, it should be borne in mind that the potential environmental risks posed by pesticides could only be assessed once their fate in the environment has been understood, including the potential degradation rates under site-relevant conditions.

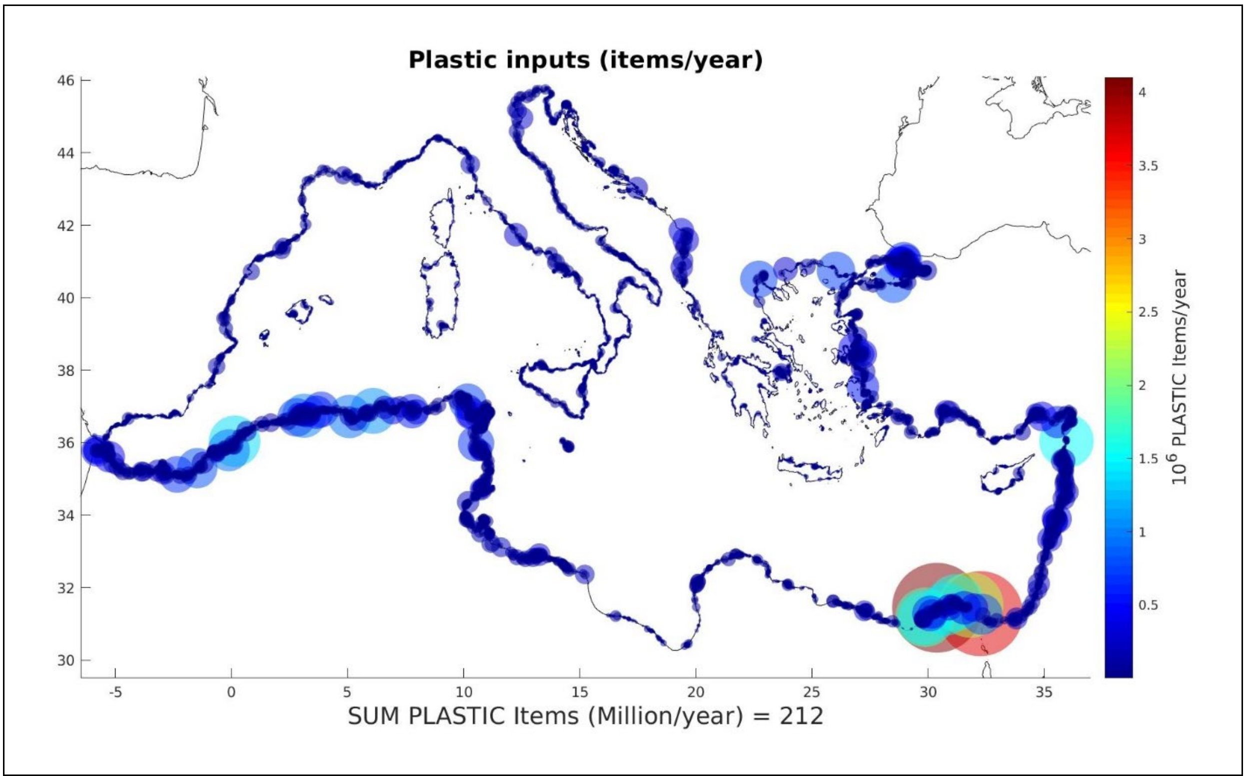
6.3 Plastic pollution

6.3.1 Uncertainties in plastic modelling, baseline scenario

The first main issue to perform any type of assessment about this particular type of pollution traces back to the lack of reliable data, both on the pressure (i.e., the amount of plastic entering our seas) and on the status (i.e., the quantity of plastics at sea) of the environment. Slowly, datasets are being created, mostly in the frame of the MSFD reporting and monitoring system (e.g., Addamo and Hanke, 2019) although the only consistent and reliable information available so far refers to the presence and abundance of beached litter (Hanke et al., 2021). However, even if datasets on beached litter includes valuable information, it is sampled with irregular frequency and on a limited number of locations. Henceforth, it is extremely challenging to setup, calibrate and validate any model aiming at addressing this particular type of pollution.

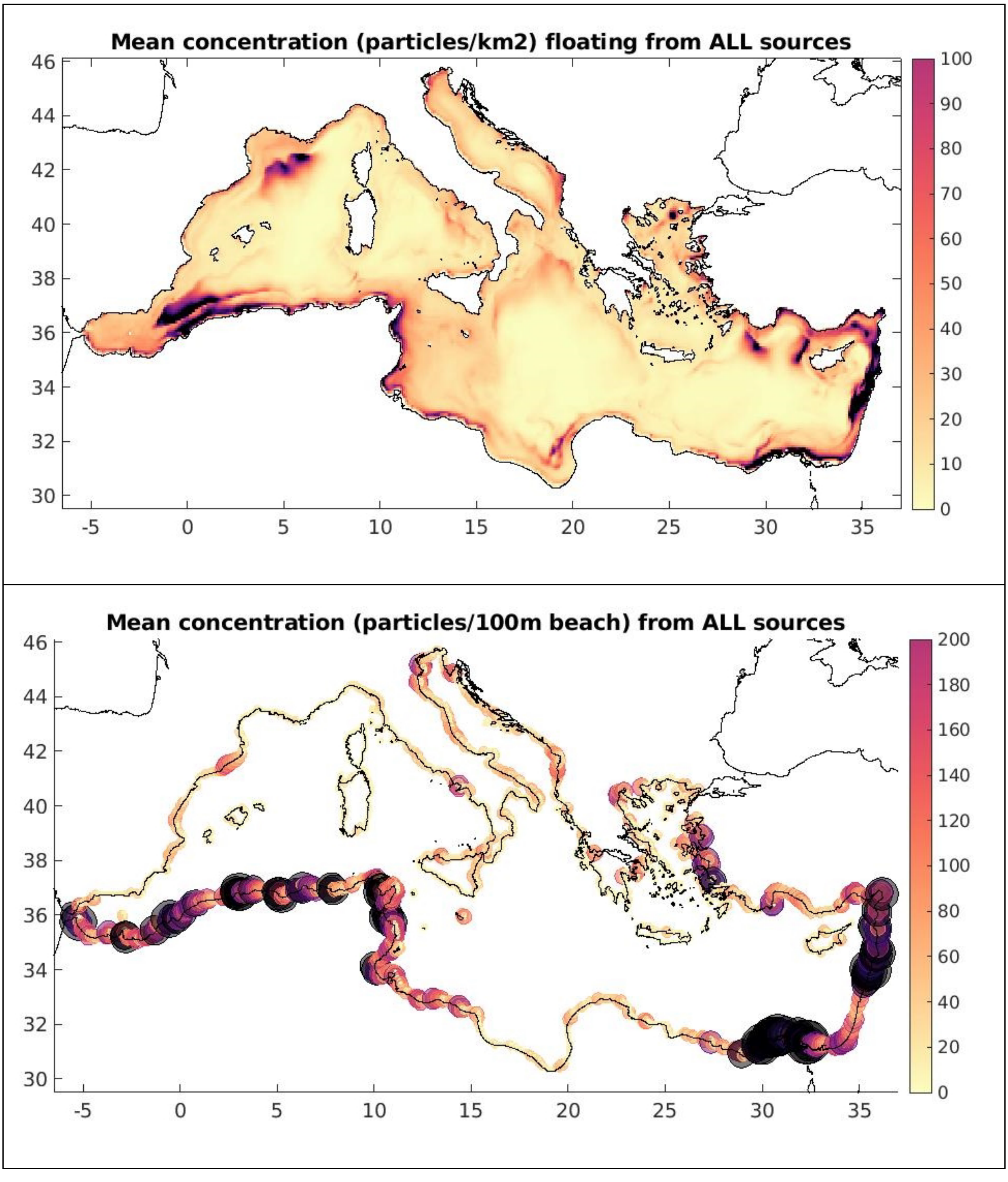
Many different attempts have been done at JRC (Macias et al, 2019 and 2022; Miladinova et al., 2020b) and beyond (Lebreton et al., 2012; Iwasaki et al., 2017; Liubartseva et al., 2018, Tsiaras et al, 2021) to provide modelling assessments of plastic impacts on marine ecosystems but the uncertainty associated to these works remain high. In the present contribution we aim to apply the best available estimates of macro-plastics (> 2.5 cms) inputs in the Mediterranean Sea by using a modification of the modelling approach by González-Fernández et al. (2021) which provides an input estimate as shown in Fig. 17.

Figure 17 Plastics inputs estimates for the baseline simulation around the Mediterranean coast



Using these 'best' estimates as input to the Lagrangian module of the JRC-DT it is possible to get a description of the distribution of floating and beached litter in the Mediterranean Sea (Fig. 18). The mean annual amount of plastic items in the region (beached + floating) in the baseline scenario is 212 million, with 15,081 individual entry points along the entire Mediterranean coastline (Fig. 17). From this amount, around 90% is simulated to be beached and 10% to be floating plastic in the 150 days of model integration (Table 8). These estimates are not far from previous calculations (Tsiaras et al., 2021) which estimated 82% of beached litter and 18% of floating litter for this same marine region.

Figure 18 Mean concentration of floating (upper figure) and beached (lower figure) plastic litter for the 'baseline' simulation.



It is very difficult to assess how precise the used models are in representing the actual distribution of litter (both floating and beached) in the Mediterranean Sea mostly because the lack of adequate/comparable measurements. Although large improvements on monitoring and reporting methodologies have recently taken place (e.g., Hanke et al., 2021), there is still very little meaningful information on litter abundance that can be directly compared with the JRC-DT results.

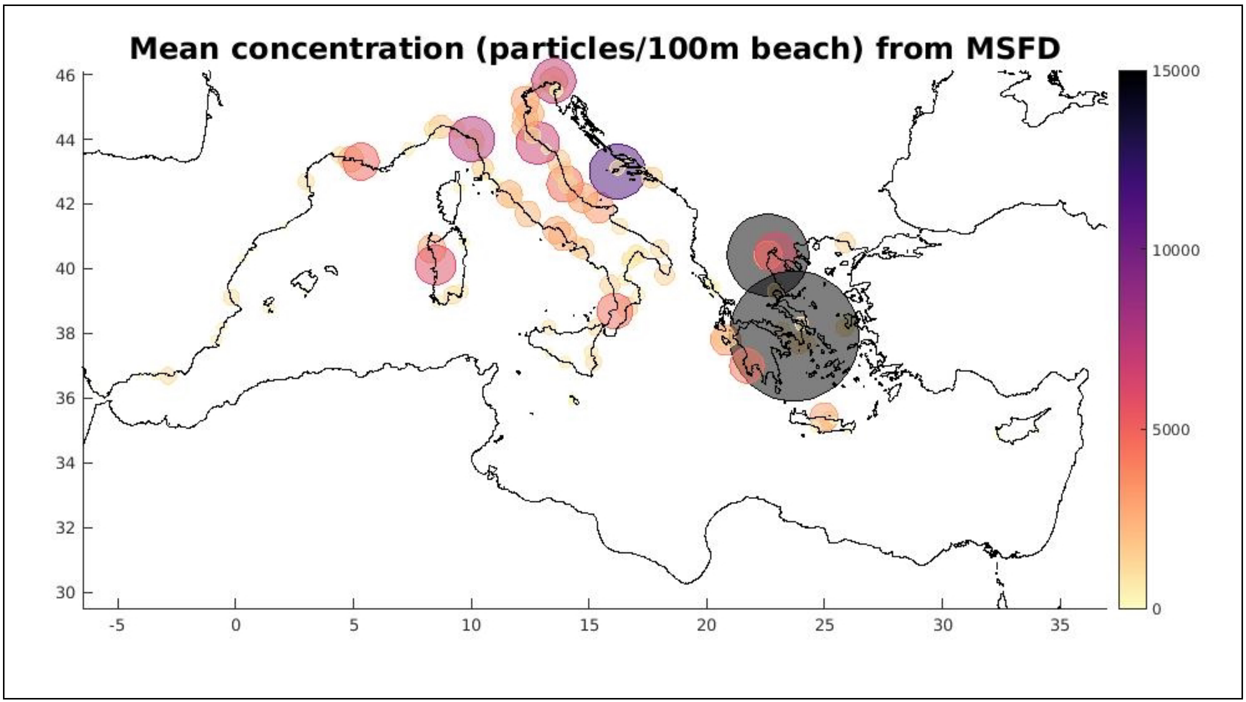
This is particularly true for floating litter, as there is still a long way to go before comprehensive observational datasets are available for the different EU basins. Thus, the main source of independent data that can be used for comparison/validation are previous modelling works. The surface distribution of floating litter simulated by the JRC-DT (upper panel, Fig. 18) compares relatively well with Tsiaras et al. (2021) surface distribution of macro-litter larger than 20 cm (their Figure 7). In both cases there are coastal accumulation of litter in the far-east Mediterranean coasts and along the Adriatic eastern and western coasts. Also, the Algerian current region shows significant accumulations in both models as well as the Catalan Sea. It is also remarkable that the overall amount of floating litter seems to be comparable in both cases. The maximum concentration of macro-litter is around 100 items/km^2 in both works while average concentration is 18 items/km^2 in Tsiaras et al. (2021) (although only computed in certain locations where field data was available) and it is 9 items/km^2 in the JRC-DT simulations including all the Mediterranean Sea. This value is somewhat lower than other estimates at around 16 to 25 items/km^2 (Arcangeli et al., 2020; Campana et al., 2018 and Suaria and Aliani, 2014) depending on the area and season. However, all estimates (including the JRC-DT) are on the same order of magnitude and they are quite difficult to compare as different spatial and temporal resolutions are involved in the different analyses.

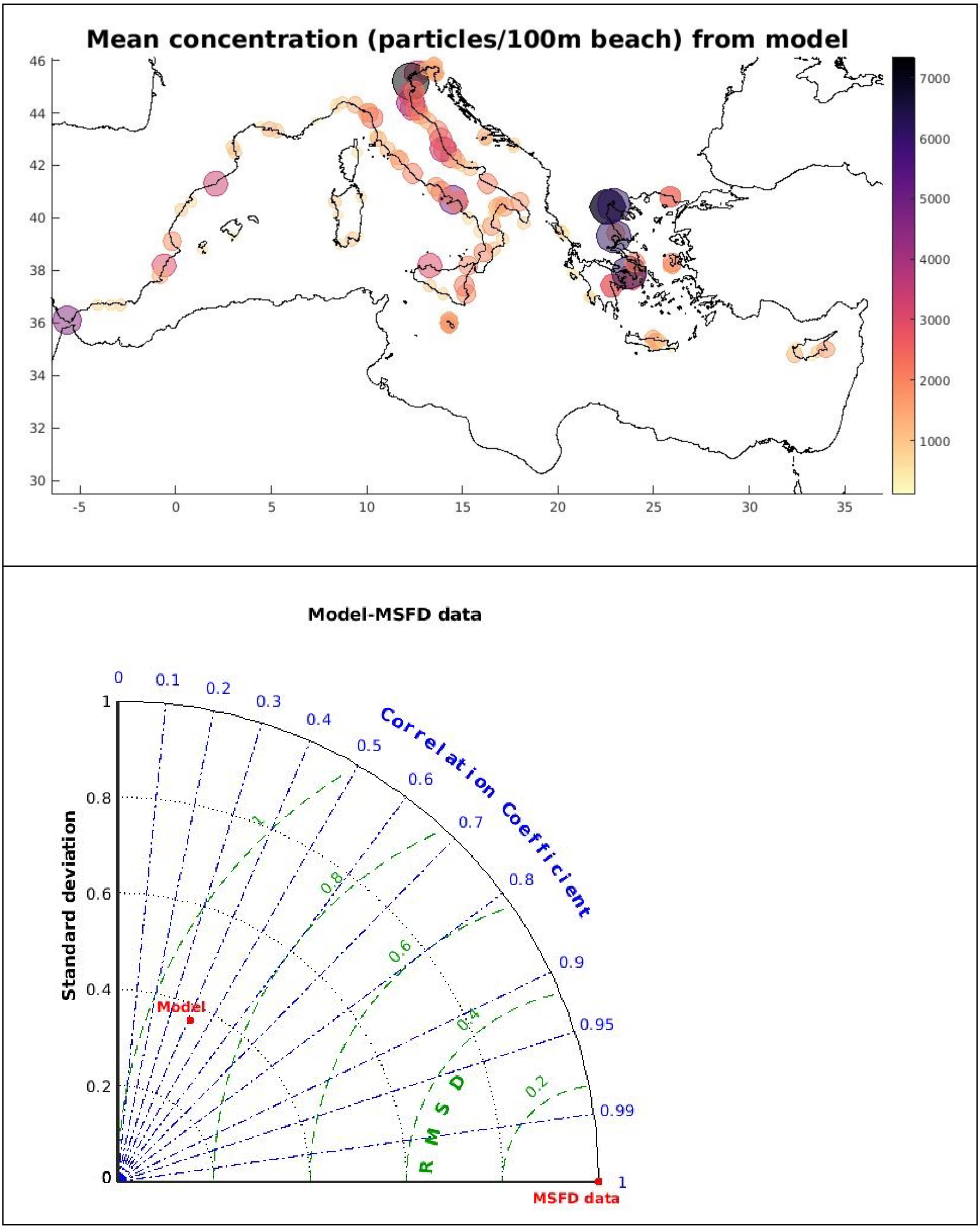
However, for beached litter the monitoring and reporting conditions are much more mature and there are available datasets than can be used to (partially) compare with simulated patterns as EU MS are requested, within the MSFD, to report on the presence of litter items on their beaches following a standardized protocol (Hanke et al., 2019). The mean concentration (items/100m beach) contained in such dataset (covering the period 2015 – 2018) is shown in Fig. 19 (upper panel) below.

However, we need to consider that the total number of monitored beaches within the Mediterranean basin is limited (150), the sampling frequency is irregular for most of them and they are only located within the EU borders (Fig. 19, upper panel). Henceforth, we used an additional source of beached litter information, the Marine Litter Watch dataset (https://www.eea.europa.eu/themes/water/europes-seas-and-coasts/assessments/marine-litterwatch) of the EEA as it contains more observation points within the Mediterranean (950) including some non-EU beaches (see Fig. 20, upper panel).

In both cases, we need to consider that the mean concentrations in these datasets include both litter coming from the sea, together with litter that may come directly from land. For example, litter pushed down to the coastline by direct wind transport or litter left behind by beachgoers. Conversely, the JRC-DT estimates only include the litter that arrives to the beaches from the sea, so only a fraction of the total observed litter found in the coastlines. However, this are the current best available datasets for comparison.

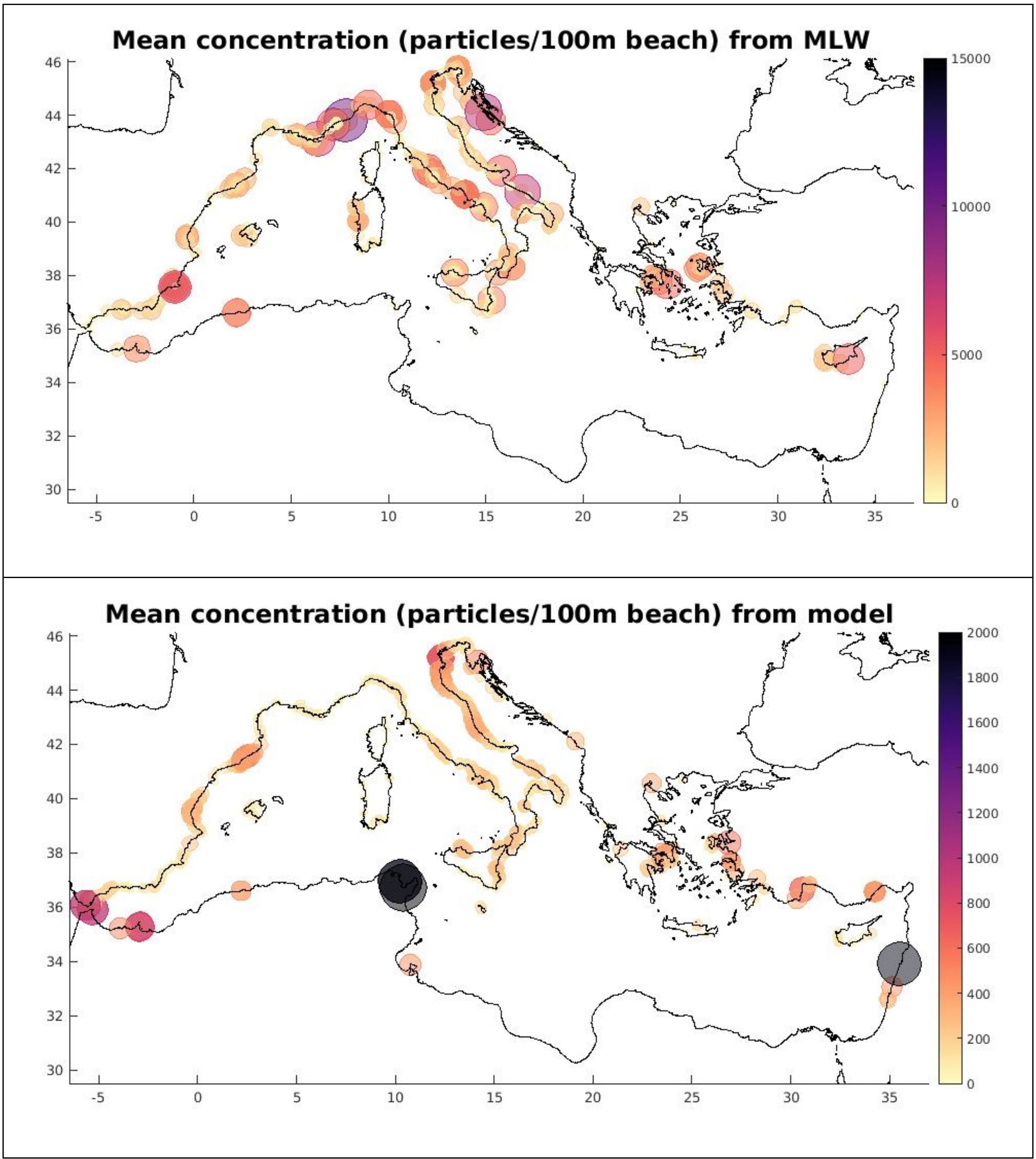
Figure 19 Upper panel, mean beached litter in EU Mediterranean beaches from MSFD reporting. Middle panel the corresponding simulated beached litter from the model for those locations. Lower panel, Taylor Diagram of model versus MSFD reported data

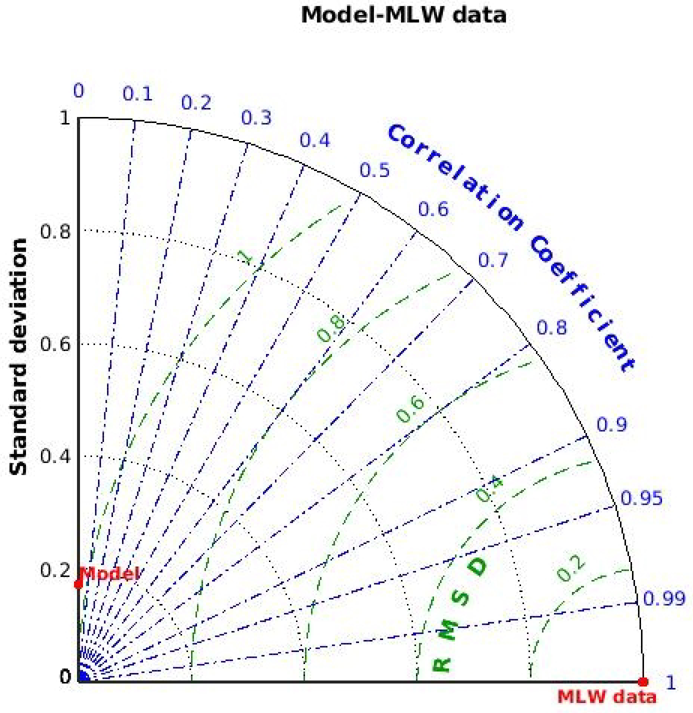




From the simulated beached litter abundance in the baseline run (Fig. 18, lower panel) we extracted the values on the positions (beaches), corresponding to the monitoring sites for both the MSFD reporting (Fig. 19, upper panel) and the MLW dataset (Fig. 20, upper panel). The comparison of model estimates vs. observed abundance indicates a weak (R=0.41) positive correlation yet strongly significant (p<0.01) for the MSFD (Fig. 19, lower panel) while a non-significant correlation (p>0.1) is obtained with MLW data (Fig. 20, lower panel). These comparisons also indicate that model simulation strongly underestimates the observed beached litter. This might be related with an underestimation of the inputs to the sea (see description further up) or with the fact that monitored litter at the beaches do not only include items coming from the sea (the ones the model simulates) but also those directly deposited on the beaches by e.g., beachgoers or wind.

Figure 20 Upper panel, mean beached litter in EU Mediterranean beaches from MLW add data. Middle panel the corresponding simulated beached litter from the model for those locations. Lower panel, Taylor Diagram of model versus MLW reported data





In any case, the general pattern of beached litter abundance is quite similar in both datasets (model and MSFD observations), and that provides a certain confidence on the performance of the Lagrangian model and on the validity of the inputs estimates (at least with regard the spatial distribution of those inputs).

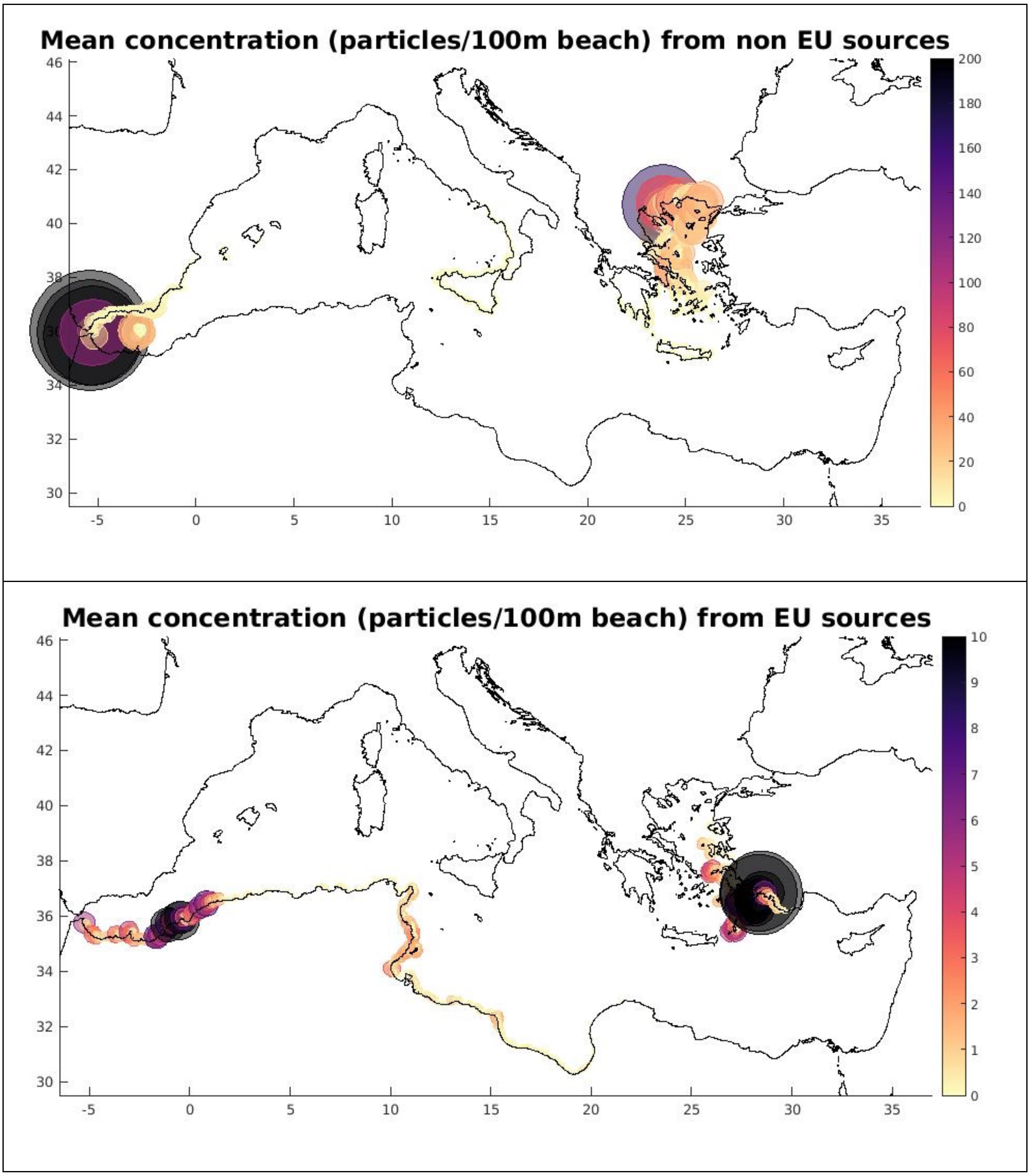
For the input estimates (Fig. 17), it could be derived that EU countries generates 24% of the total litter entering the Mediterranean Sea. If we look at how this macro plastic litter is divided between floating and beached (Fig. 18, baseline simulation), it could be observed that a larger percentage of the EU litter remains floating (during the 150 days of simulations) while the non-EU litter tend to quickly arrive to the beach and less of it remains free floating (Table 8).

Table 8. Relative contribution of EU/non-EU countries to litter pollution on the Mediterranean Sea

	EU	Non - EU	ALL
Litter inputs	24.3%	75.6%	
Beached litter	21.7%	78.3%	90%
Floating litter	30.9%	69.1%	10%

Given the spatially explicit nature of the Lagrangian simulations, it is possible to explore the cross-boundary littering between EU and non-EU countries in the Mediterranean Sea (Fig. 21). First, we can look at how no-EU generated litter might end up on EU beaches (Fig. 21, upper panel). As expected, the western and eastern boundaries of the EU are the regions most affected by this ‘cross-continental’ pollution, this include the southern Spanish coasts and the east Greek coasts. There is also litter pollution simulated in beaches around Sicily and even mainland Italy. In numbers, this non-EU generated litter represents around 10% of the total litter found in EU beaches (baseline scenario). However, this pollution affects almost 36.6% of all EU beaches giving a clear idea of the importance of this cross-boundary pollution (full details in Table 9 at the end of this section).

Figure 21 Beached litter on EU coasts originated from non-EU countries (upper map). Beached litter on non-EU coasts originated on EU countries (lower map).



The same analysis could be done looking at the opposite direction of the cross-continental pollution flow, e.g. how much EU-produced litter contributes to non-EU beached litter (Fig. 21, lower panel). In this case affected areas stretch from the Strait of Gibraltar eastwards, including Tunisia and Libyan coasts with another hotspot of pollution on Turkey beaches. In numbers, this EU-generated pollution represents only about 1% of the overall beach litter in non-EU coasts but it spans over 40% of all non-EU beaches (Table 9).

As already indicated by previous analysis (e.g., Macias et al., 2022), cross-boundary litter pollution in the Mediterranean Sea is very important even if we look at supranational level as in the case of the present work. This particularity should be kept in mind when analysing the different scenarios presented here below.

6.3.2 Impacts of measures

As described in section 5.3, two measures scenarios are tested in relation with this pressure, the full ban of SUP items on EU MS and a total elimination of plastic litter from the EU.

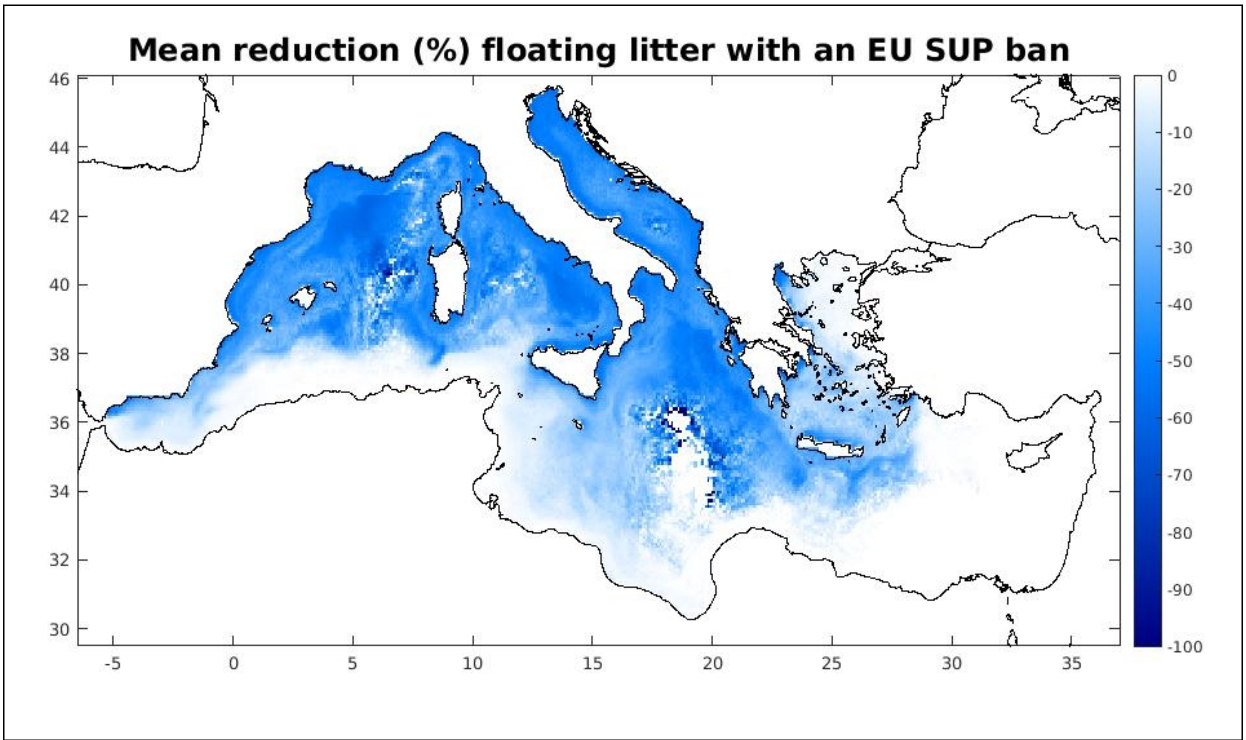
a) SUP ban on EU MS

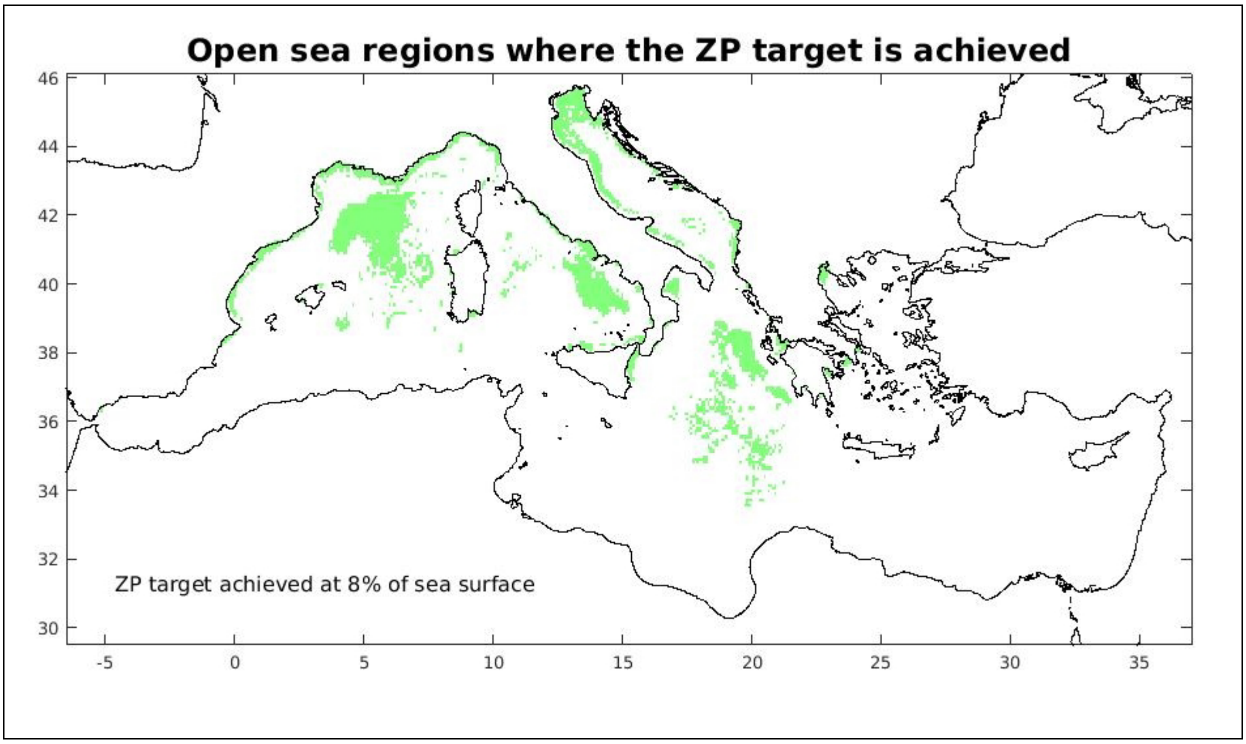
A very quick calculation using the percentages shown above in Table 9 indicates that EU-generated SUP items represents approx. 14.5% of all plastic inputs to the Mediterranean Sea (60% of the 24% pollution generated on the EU). When analysing the results from the SUP-ban simulation, this percentage is corroborated with a reduction of 14.7% of all floating particles and of 13.1 % of beached litter.

However, the Lagrangian models included in the JRC-Digital Twin allow assessing how this reduction explicitly impacts the different compartments (floating/beached) and in the diverse regions around the Mediterranean basin (see Figs. 22 and 23 below).

Regarding floating litter, the distribution of the reduction is very heterogeneous, with much larger percentage reductions in the northern part of the basin (Fig. 22, upper panel). The same analysis allows calculating that the ZP target (i.e., a reduction of 50% litter at sea) is achieved in only 8.2 % of the overall surface of the Mediterranean Sea (Fig. 22, lower panel) including many EU coastal regions and some of the northernmost accumulation areas described above for the baseline simulation.

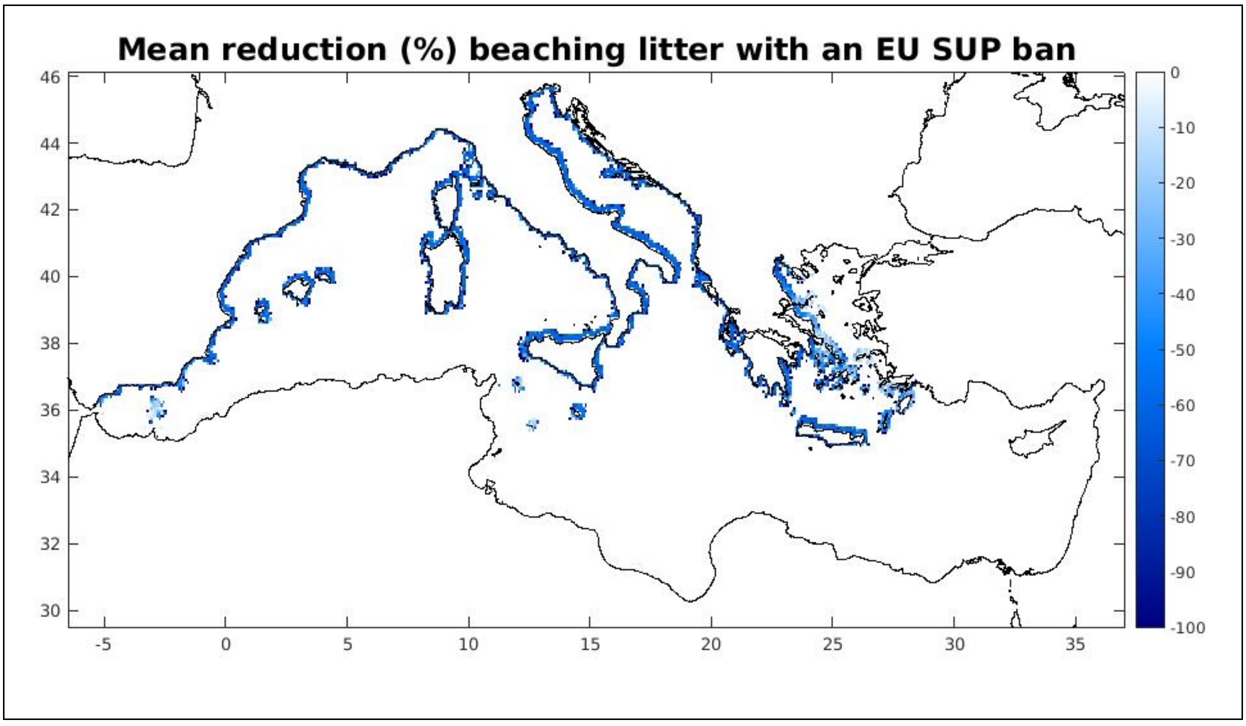
Figure 22 . Upper panel, mean percentage reduction ((SUP-baseline)*100/baseline) of the number of floating plastic items following the full stop of EU inputs. Lower panel, open sea regions where the ZP target (reduction > 50%) is achieved after the full ban of SUP in the EU.

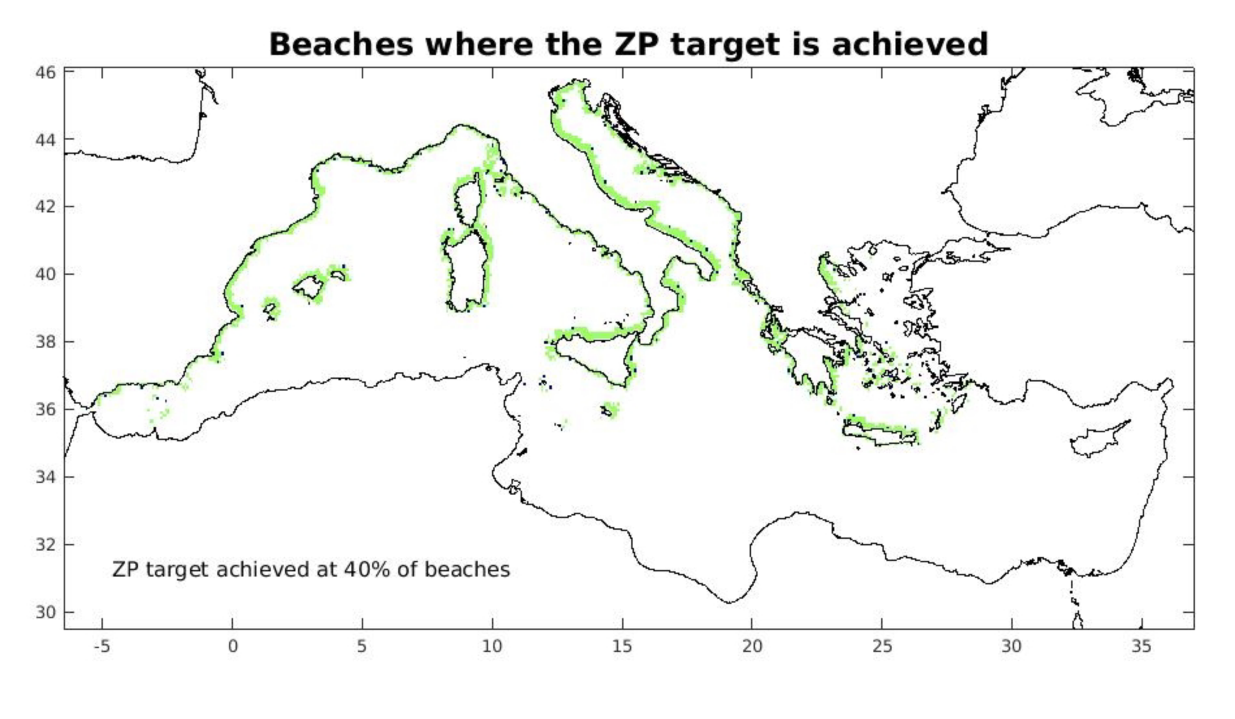




The same analysis can be done for beached litter. Fig. 23 shows that the reduction percentage in beached litter can be very high in certain regions of the EU coastlines (reaching almost 90% at certain locations). The spatial analysis (lower panel, Fig. 23) indicates that the ZP target (a reduction over 50%) is achieved in 44 % of all the Mediterranean beaches, mostly within EU borders. For those EU beaches, this SUP-ban scenario means a mean decrease of 54.1 % of the number of beached litter items.

Figure 23 . Upper panel, mean percentage reduction ((SUP-baseline)*100/baseline) of the number of beached plastic items following the full stop of EU inputs. Lower panel, beaches where the ZP target (reduction > 50%) is achieved after the SUP ban at the EU.





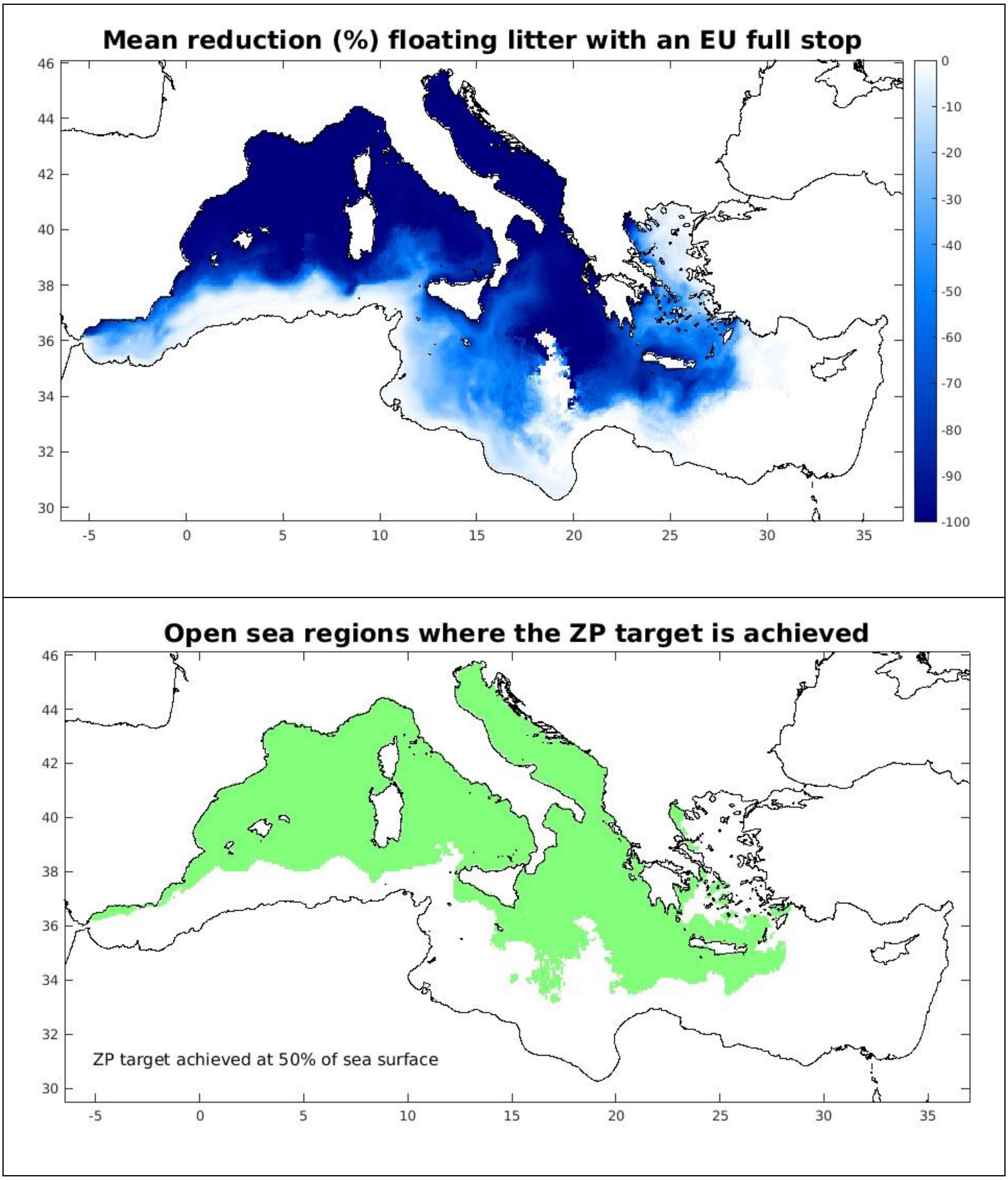
Finally, for this EU SUP-ban scenario it is worth investigating how the transboundary litter (EU - non-EU) is affected. As shown in Table 9 below, the percentage of litter generated outside the EU that ends up in EU beaches increases with respect to the baseline (from 10% to 22%). This relative increase is simply due to the fact that the total litter beached at the EU decreases significantly in this scenario (in around 51% as indicated above). The percentage of EU beaches affected by non-EU litter remains, however, unchanged at 37%. For the opposite case, the percentage of EU generated litter polluting non-EU coastlines decreases (from 1% to 0.3 %) and also decreases the percentage of non-EU beaches affected by EU litter (from 40 % to 32 %).

b) Full stop of plastic litter pollution from EU MS

This is a not very realistic scenario from the management point of view as it is not reasonable to expect all plastic pollution from the EU to cease (even with the most sophisticated waste management options ever applied). However, it is a useful exercise to understand the full extent of the impacts from the EU with regard to plastic pollution in the basin, helping to identify potential problematic areas where detailed attention might be needed.

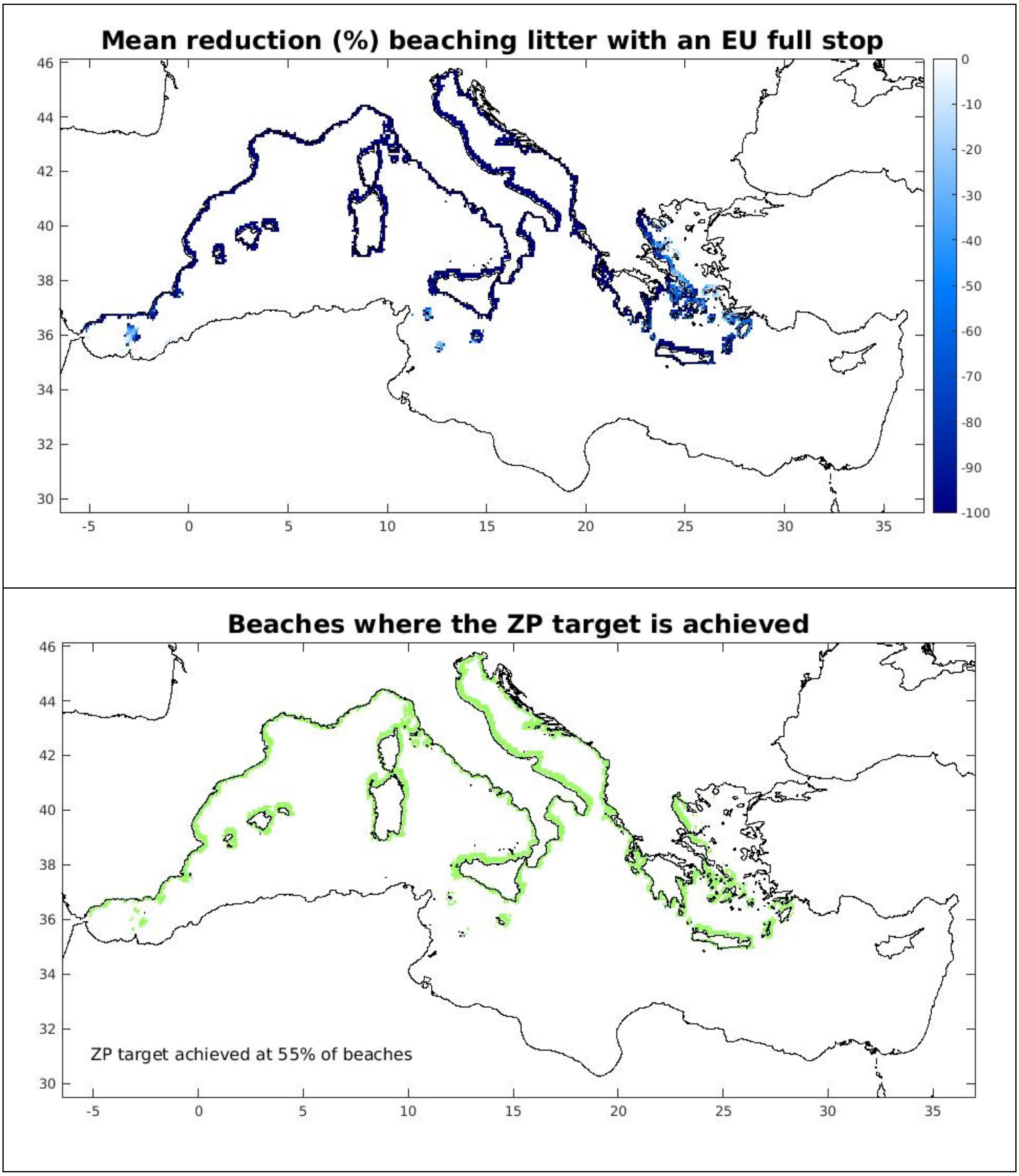
When no floating litter is entering from EU countries, a strong diminution of floating plastic is simulated on the northern half of the Mediterranean basin (Fig. 24, upper panel). In numbers of floating items, this scenario represents a reduction of almost 31% with respect the baseline (mean reduction in the whole basin) and it implies reaching the ZP objective (reduction of litter > 50%) in almost 51% of the whole basin (mostly in regions close to the EU).

Figure 24 Upper panel, mean percentage reduction ((BAN-baseline)*100/baseline) of the number of floating plastic items following the full stop of EU inputs. Lower panel, open sea regions where the ZP target (reduction > 50%) is achieved after the full stop of plastic inputs from the EU.



The same analysis can be conducted for the beached litter (Fig. 25). The overall mean reduction of beached litter in the Mediterranean amounts to almost 22% (i.e., slightly smaller than for the floating litter) being much more evident in the coasts of EU countries (Fig. 25, upper panel). As for the ZP ambition, the 50% reduction is achieved in almost 54% of all Mediterranean beaches (Fig. 25, lower panel). For EU coasts, the average litter reduction is 89.9% with respect to the baseline, the 10% beached litter remaining in EU coastline being generated in non-EU as also indicated in section 6.3.1.

Figure 25 Upper panel, mean percentage reduction ((BAN-baseline)*100/baseline) of the number of beached plastic items following the full stop of EU inputs. Lower panel, beaches where the ZP target (reduction > 50%) is achieved after the full stop of plastic inputs from the EU.



The analysis of the two management scenarios presented in this section 6.3.2 (SUP ban and full stop of EU littering) have clearly indicated that the EU (by itself) will not be able to achieve the ZP targets for plastic pollution in the Mediterranean Sea. Even with a very ambitious (albeit potentially achievable) policy measure such as the full ban of single use plastics in the EU, it will not be possible to reduce plastic pollution by 50% in Mediterranean waters and coasts. Not even a full stop of plastic littering from EU sources (a highly implausible scenario) will deliver the achievement of the ZP targets for the whole basin.

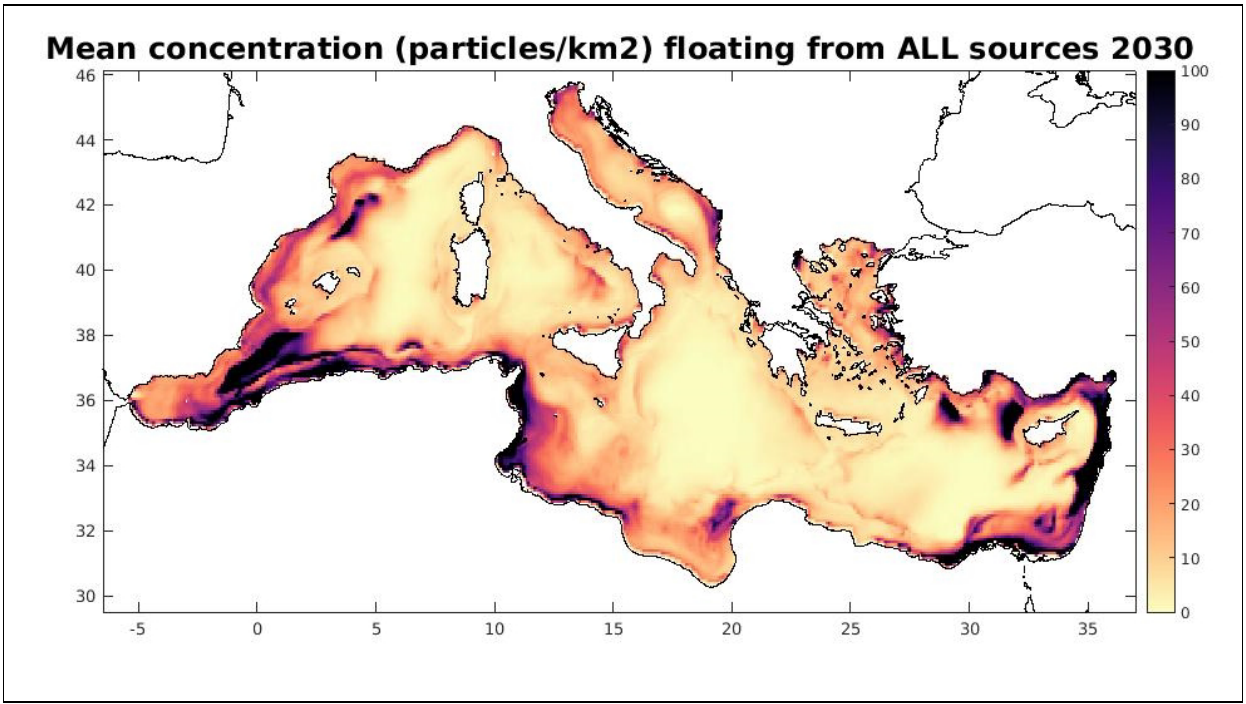
Given the relatively large amount of litter generated from non-EU countries (Gonzalez-Fernandez et al., 2021) and the very high connectivity of the oceanographic system in the Mediterranean (Macias et al., 2022), international cooperation and engagement is essntial to achieve the ZP objectives for the whole basin. The European Green Deal (EGD) already aims to increase the EU world leadership in environmental protection by fostering international engagement of neighbouring countries. The case of plastic pollution in the Mediterranean Sea (and other shared basins around the EU) is a perfect example where this approach could be first tested and tried.

6.3.3 Impacts of climate change

It has been clearly determined that atmospheric (e.g., air temperature and wind intensity) and oceanographic (e.g., vertical stratification and surface currents) features in the Mediterranean Sea could be altered in future climatic conditions (e.g., Somot et al., 2016; Macias et al., 2018). Such environmental changes will, undoubtedly, impact the distribution, accumulation and beaching patterns of floating litter in the basin modifying, hence, the baseline litter simulation presented above (Fig. 18). Climate is already changing and, in spite of any measures taken now, will continue to change for the next decade. Hence, and in order to fully evaluate the potential impact of management measures in the future we need to understand how climatic variations will impact plastic pollution in the Mediterranean Sea.

As presented in section 5.3, to isolate the impacts of climate change, litter inputs to the Mediterranean Sea are kept equal to the baseline simulation, while oceanographic conditions (currents) are derived from a simulation forced with a climate model (MPI) under the IPCC emission scenario RCP4.5 (middle emission) for the years 2028, 2029 and 2030. The comparison below is made with the distributions obtained by the Lagrangian model for the baseline simulation (i.e., those shown in Fig. 18 above) and those realized with the 2030 atmospheric conditions (Fig. 26).

Figure 26 Mean concentration of floating (upper figure) and beached (lower figure) plastic litter for the 'climate change' simulation.



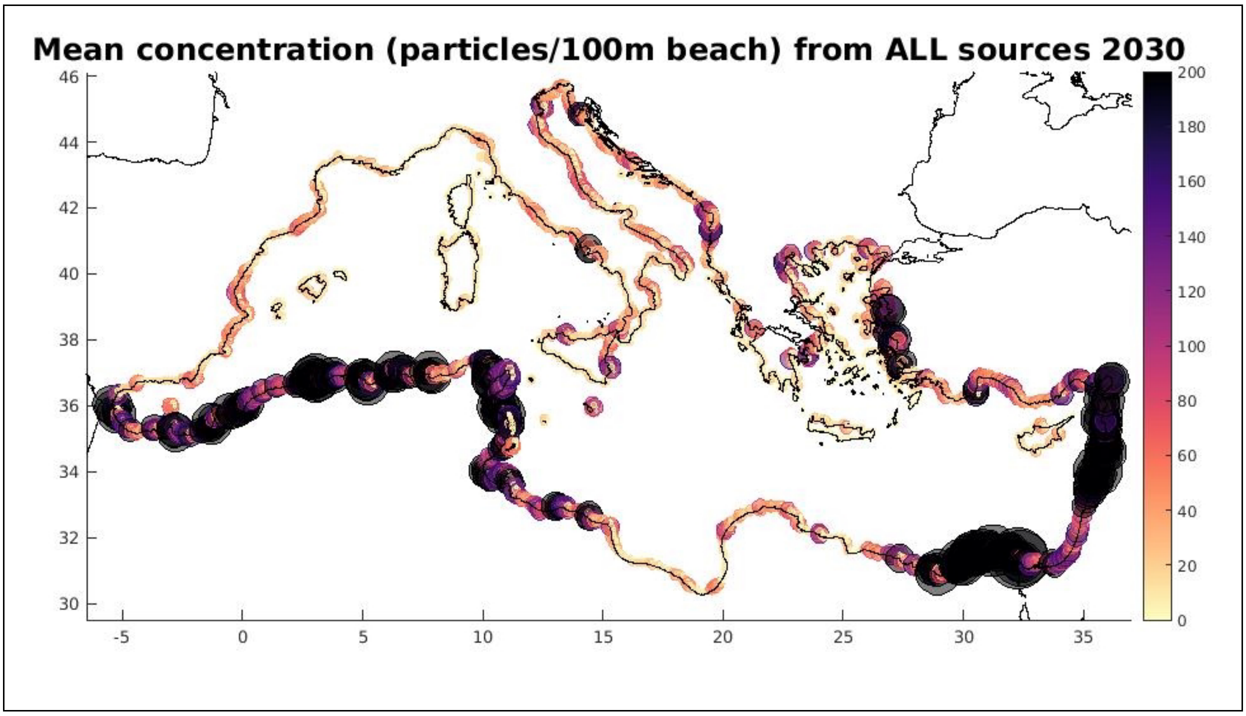
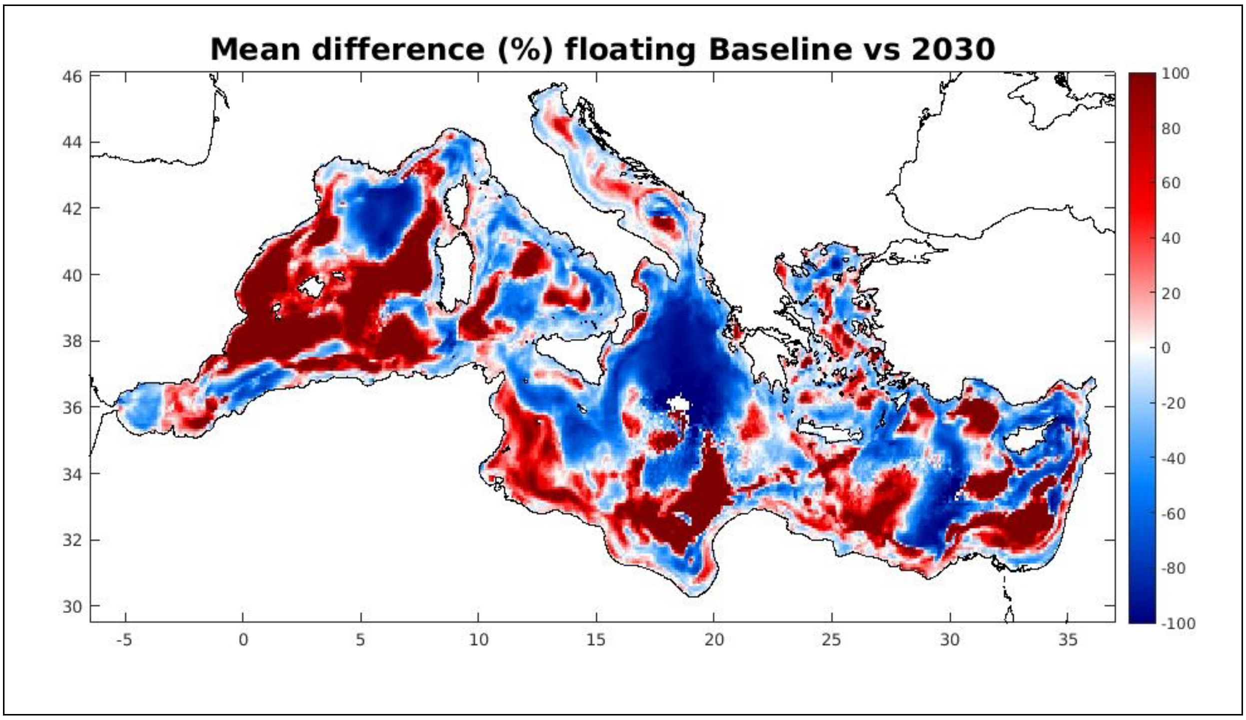
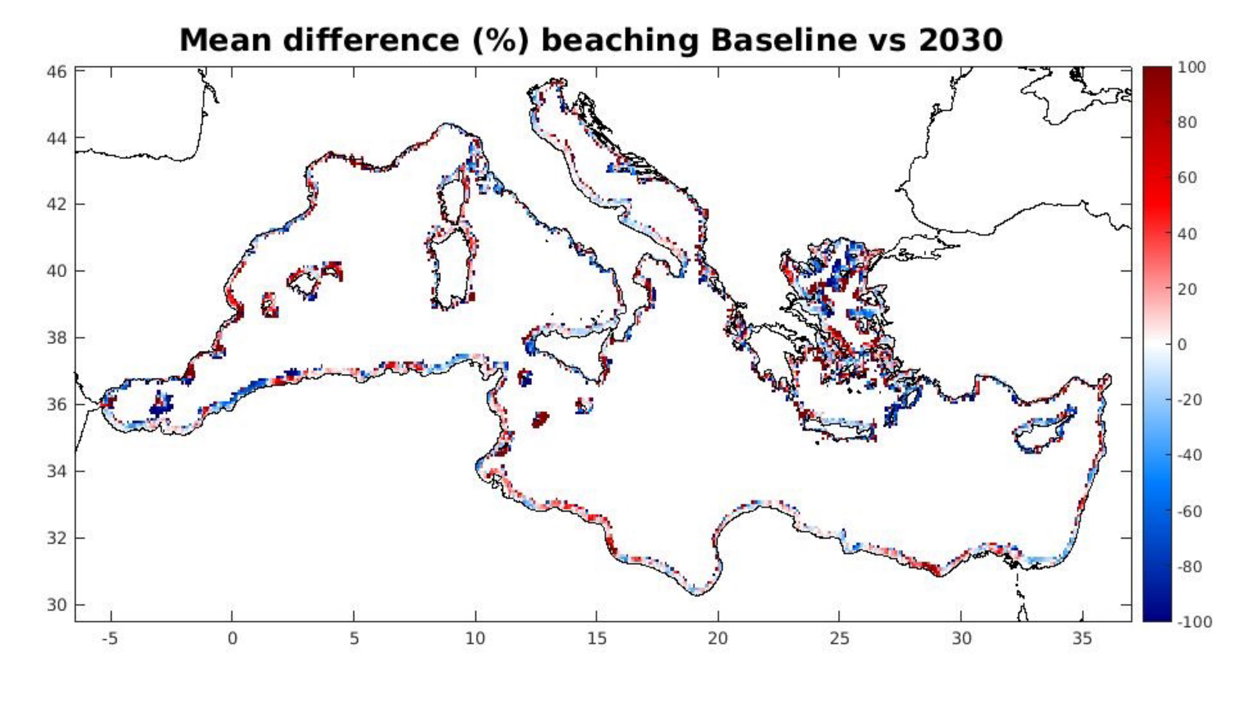


Figure 27 . Upper panel, mean percentage reduction ((2030-baseline)*100/baseline) of the number of floating plastic items in the RCP4.5 scenario. Lower panel, Upper panel, mean percentage reduction ((2030-baseline)*100/baseline) of the number of beached plastic items in the RCP4.5 scenario





The patterns shown in this comparison (Fig. 27) are less clear and patchier than in the management scenarios analysed above. Changes in floating litter concentration are both positive and negative and spread all over the entire Mediterranean basin. Overall there are no significant changes in the amount of free-floating items (difference - 2.3 %) or in the beached litter (difference 0.09 %) although a slight increase in the amount of beached litter at EU coastline (around 3%) is simulated.

There are no significant differences in the percentage of cross-boundary pollution (see also Table 9 below), with non-EU litter amounting to 10% of EU beached litter and EU litter contributing around 1% to non-EU beached litter. However, in this climate change scenario, the model predicts a decrease of the number of EU beaches affected by non-EU litter (from 36.6 % to 29.1 %) and an increase in the number of non-EU beaches affected by EU litter (from 40% to 43.2 %).

Table 9. Summary of the cross-boundary (EU – non EU) pollution in the different scenarios tested

Scenario	Baseline	SUP-ban	Climate change
Non-EU litter present at EU beaches	10.2 %	21.9 %	10.6 %
EU beaches affected by non-EU litter	36.6 %	36.6 %	29.1 %
EU litter present at non-EU beaches	0.98 %	0.3 %	0.94 %
Non-EU beaches affected by EU litter	39.8 %	32.1 %	43.2 %

7 Conclusions

With the currently maximal achievable nutrient leakage reductions it is possible to reduce nitrogen inputs to marine ecosystems by 32% with respect its actual value and by 17% in the case of phosphorous. Our analysis of nutrients pollution indicates that it is of paramount importance to consider not only their total reduction but also the change of their relative ratio (N:P) in the receiving waters. Marine ecosystems have a natural N:P ratio that allows a diversity of phytoplankton types to co-exist and maintain a healthy biodiversity. If this ratio is altered (for example by interventions at the sources) that will tilt the delicate natural balance allowing opportunistic, highly specialized species to thrive and dominate the ecosystem. The appropriate reduction targets for both macro-nutrients (N and P) might be different for the different EU marine basins as their biogeochemical structure is diverse. Thus, further regionally focused analyses are needed to set nutrient reduction targets for each marine region.

The simulations of chemical contaminants show that many substances, even very persistent will be almost totally disappearing from the coastal/riverine regions after a ban on their use. However, a crucial element emerging from the simulations on chemical pollutants is the need to account for the natural inertia of the water/marine ecosystems. This element is common also to the nutrients' case, as chemical substances can accumulate in different environmental compartments (such as soil, sediment or even the biota). For this reason, there is a delay (or inertia) from the moment a measure is applied (e.g., a certain percentage of reduction of the pressure) until substantial or measurable improvements could be registered in the targeted environment. On top of this, impacts of climate change should be also considered. For certain pressures and regions climate-induced changes can counter-act the effects of the measures (as happen for the tested chemicals in the centre of the Black Sea) while in some other cases it might have a synergistic effect. To better assess the consistency of these conclusions, it will be necessary to include more chemical compounds in the JRC model system (also applying them to different basins) and to consider alternative climate models and scenarios to obtain an estimate of the uncertainty in the model simulations.

Current simulations on plastic dispersion and accumulation is limited by data availability on both floating and beached litter and on the actual loads to the sea. There are ongoing efforts to fill this knowledge gap but further harmonisation and coordination work is still needed from the monitoring point of view.

Despite these limitations, two messages are emerging from the analysis done on plastic pollution. First, planned measures result in significant reductions of the amount of plastics in the Mediterranean Sea. And, second, international collaboration and coordination are key elements to achieve the ambitions ZP targets in shared basins, the EU should guide with the example and become a leader and inspiration for our neighbouring regions.

The outlook exercise presented in this report indicates that it is possible to advance in the fight of water and marine pollution in the EU with planned and expected future policies development. However, a clear result of the modelling simulations is that to achieve the highly ambitious objectives of the ZPAP we need to step up current approaches and planned interventions into more firm commitments for coming years.

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List of abbreviations and definitions

ABI	Bay of Biscay and Iberian Coastline
ACS	Celtic Sea
ANS	Greater North Sea
BAL	Baltic Sea
BLK	Black Sea
BLM	Bosphorous and Marmara Sea
BSD	Biodiversity Directive
CAP	Common Agricultural Policy
Chl-a	Chlorophyll-a
CORDEX	Coordinated Regional Climate Downscaling Experiment
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphorous
DT	Digitial Twin
DT50	Degradation half time
EAP	Environmental Action Plan
ECMWF	European Centre for Medium Weather Forecast
EGD	European Green Deal
EQS	Environmental Quality Standards
ERSEM	European Regional Seas Ecosystem Model
FABM	Framework of Aquatic Biogeochemical Models
GCM	Global Circulation Model
GES	Good Environmental Status
GETM	General Estuarine Transport Model
GOTM	General Ocean Turbulence Model
HAS	High Ambitious Simulation
MAD	Adriatic Sea
MAL	Aegean-Levantine Mediterranean Sea
MIC	Ionian Sea and Central Mediterranean
MMF	Marine Modelling Framework
MS	Member States
MSFD	Marine Strategy Framework Directive
MWE	Western Mediterranean Sea
N	Nitrogen
ND	Nitrates Directive
OM	Organic Matter
P	Phosphorous
PNEC	Predicted Non Effect Concentration
RCP	Representative Common Pathway

REF	Reference simulation
SST	Sea Surface Temperature
SUP	Single Use Plastics
SUPD	Single Use Plastic Directive
TN	Total Nitrogen
TP	Total Phosphorous
UWWTD	Urban Waste Water Treatment Directive
WFD	Water Framework Directive
ZP	Zero Pollution

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