Climate change driven effects on transport, fate and biogeochemistry of trace element contaminants in coastal marine ecosystems

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Human activities and climate change substantially threaten coastal areas, impacting ecosystem functions, services, and human-wellbeing. Trace elements, from both natural and anthropogenic sources, can contaminate coastal regions, and at high concentrations may become toxic to marine biota. Climate change is likely to affect the sources, sinks and cycling of trace elements in coastal systems: for example, riverine runoff is set to increase as precipitation in the Arctic intensifies, and more frequent extreme floods are expected to activate previously deeply buried trace elements. Furthermore, changes in human activity under a warming climate, such as increased Arctic shipping and potential geoengineering projects such as ocean alkalinity enhancement, will likely introduce more trace elements to coastal ecosystems. Advancing our understanding of trace element cycling is at present limited by factors including lack of data coverage in the Global South, challenges in studying multi-stressor effects and ecosystem responses, lack of long-term data, and the difficulty in parametrizing robust models in coastal environments.

In the hydrosphere, especially in marine waters, trace elements (TE) play an essential role as micronutrients and tracers of processes of fundamental importance[1]. Elements such as cadmium (Cd), cobalt (Co), copper (Cu), chromium (Cr), iron (Fe), nickel (Ni), selenium (Se), and zinc (Zn) are considered bio-essential elements since they are required in small quantities for plants and wildlife in natural waters to maintain metabolic processes such as carbon uptake, photosynthesis, and nitrogen fixation[2–4]. On the other hand, while some TE, such as aluminum (Al)[5], have no known biological function, others, like arsenic (As) and mercury (Hg) are highly toxic[6,7]. The principle that „the dose makes the poison" applies to all bio-essential elements, which can become toxic at high concentrations. This toxicity can result from natural sources, such as hydrothermal vents and groundwater seepage, as well as anthropogenic influences like industrial

effluents and emissions[7]. Additionally, there is often a narrow optimal concentration range for marine biota, as is the case for Cu[8].

Isotope ratios of TE, including barium (Ba), Cd, Cr, Cu, Fe, molybdenum (Mo), Ni, lead (Pb), uranium (U), and Zn serve as valuable tracers of human activities[9–11]. They provide insight into various geological, physical, biological, and chemical processes in the ocean, and elucidate the sources and sinks of these elements[12–14]. Furthermore, TE such as silver (Ag), gold (Au), Cd, Co, Cu, Ni, and Mn, alongside technology-critical elements (TCE), like rare earth elements (REE), platinum group elements (PGE), and others, including gallium (Ga), germanium (Ge), indium (In), Niobium (Nb), antimony (Sb), tantalum (Ta), tellurium (Te), and thallium (Tl), are expected to play increasingly significant roles in driving global economic growth[15,16].
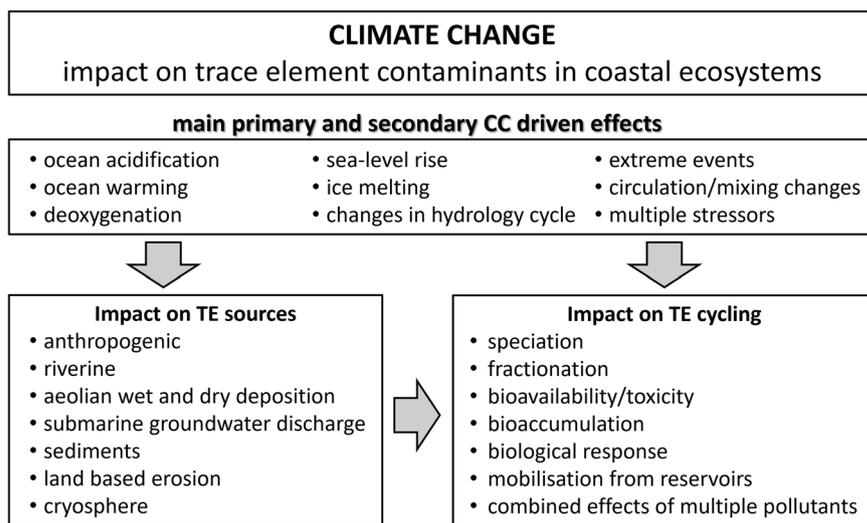
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Fig. 1 | Schematic representation of climate change-driven effects on trace element contaminants in coastal ecosystems. The arrows indicate interactions. CC Climate change, TE Trace elements.

CLIMATE CHANGE
impact on trace element contaminants in coastal ecosystems

main primary and secondary CC driven effects
- ocean acidification
- ocean warming
- deoxygenation
- sea-level rise
- ice melting
- changes in hydrology cycle
- extreme events
- circulation/mixing changes
- multiple stressors

Impact on TE sources
- anthropogenic
- riverine
- aeolian wet and dry deposition
- submarine groundwater discharge
- sediments
- land based erosion
- cryosphere

Impact on TE cycling
- speciation
- fractionation
- bioavailability/toxicity
- bioaccumulation
- biological response
- mobilisation from reservoirs
- combined effects of multiple pollutants

Notwithstanding their critical role in both environmental and economic contexts as nutrients, contaminants (defined as being present at above their natural background levels), tracers, and resources, our understanding of TE remains incomplete, despite decades of research (e.g., through the GEOTRACES Program[17,18]). There are notable knowledge gaps in the literature, particularly regarding the complex dynamics and biogeochemical cycles of TE contaminants, including their chemical speciation, physical fractionation, bioavailability, toxicity, and fate under global CC and multiple stressors[19]. These gaps are especially pronounced for the rapidly increasing TCE, where the sources, sinks, and geochemistry remain largely unknown, requiring in-depth and urgent investigations[15,16] to assess their fate and both short- and long-term impact on marine ecosystems.

Today's anthropogenic pressures are leading to changes happening at unprecedented rates, magnitudes, and variabilities, thereby greatly influencing the dynamics and fate of TE at various temporal and spatial scales, i.e., local, regional, and global[20]. Major CC drivers involve physical and chemical ocean- and coastal changes, including warming (leading to e.g., ice melt, sea-level rise, altered precipitation patterns, dust deposition, and river runoff rates), deoxygenation, acidification, frequency of extreme events (storms, floods, fires, droughts), (de)salinization, changes in near-shore ocean circulation and mixing, as well as the interaction of co-occurrent multiple stressors. Representative Concentration Pathways (RCP) are climate change scenarios projecting future greenhouse gas concentrations. Even under very stringent RCP 2.6 projection (UN IPCC)[21] it is estimated that by the end of the 21st century, atmospheric CO_2 concentrations could be as high as 478 µatm, global mean sea surface temperature could increase by 0.73 °C, the mean sea level could rise by 0.55 m, and surface pH and dissolved oxygen could decrease by 0.06 units and 0.6%[21], respectively. These perturbations, stemming directly or indirectly from human activities such as fossil fuel combustion, deforestation, fertilizer use, land-use change, and industrial activities[22], will inevitably affect the physicochemical nature (e.g., speciation, complexation, redox chemistry, sorption properties, and remobilization) of diverse TE contaminants and thereby impact their reactivity, bioavailability, and ultimate fate. Such changes will have follow-on effects on ecosystem functions and services, coastal and ocean health and sustainability, as well as human health and livelihoods (i.e., seafood safety and security).

Policy frameworks, legislation, and global agreements have been implemented to reduce specific TE contaminants in the marine environment. These include the London Convention/Protocol, and the Oslo Convention (Annex I and II), and, more recently, the Minamata Convention. However, despite these efforts, increasing TE contamination continues to negatively impact ocean health and sustainability, as well as human wellbeing, compromising the achievement of the Sustainable Development Goal Target 14.1 "*Reduce Marine Pollution*", and Goal 2 "*Zero Hunger*" (i.e., seafood safety and security), as well as the outcomes of the UN Decade of Ocean Science for Sustainable Development (2021–2030). Thus, increasing the quality, coverage, and availability of sound data, expertise, and research capabilities to understand marine TE contaminants and their cumulative, long-lasting effects on human health and ecosystem functioning is crucial[19].

To guide the scientific community towards future research priorities, this review aims to (i) summarize existing knowledge to provide a broad perspective on the impact of the major CC drivers on sources, sinks, cycling, distribution, transport, toxicity, and fate of TE contaminants, as well as their influence on the marine ecosystem (as shown schematically in Fig. 1), (ii) identify knowledge gaps to help streamline and focus future research efforts, and (iii) make recommendations for future research directions. While CC and human activities will affect the environmental distribution, fate, and toxicity of numerous chemical contaminants in marine ecosystems, this review will focus on TE in coastal ecosystems, for which, contamination is one of the most prominent pressures due to the proximity to sources (Fig. 1). However, where necessary, discussions of ocean waters will be included, given the intrinsic connections between coastal and ocean systems. Our effort will complement the work already undertaken by Working Group 45, titled '*Climate Change and Greenhouse Gas Related Impacts on Contaminants in the Ocean*' (WG45[19]), as approved by the Group of Experts on Scientific Aspects of Marine Environmental Protection (GESAMP), and the Vision 2030 White Paper of Challenge 1 '*Understand and beat marine pollution*'[23].

Sensitivity of trace element sources and sinks to climate change

There are numerous sources of TE for the marine environment, stemming from both natural processes and anthropogenic activities. Natural sources include point sources such as rivers, volcanic activity, hydrothermal vents, and forest fires. Diffuse sources include weathering of rocks, atmospheric deposition, sediment resuspension, sea-ice melt/retreat, and submarine groundwater discharge (SGD). Since the industrial revolution, anthropogenic emissions have substantially contributed to TE fluxes, sometimes surpassing natural fluxes along the land-ocean continuum[24,25]. Anthropogenic inputs can derive from activities like mining (both on land and at sea), smelting, combustion of fossil fuels, aquaculture, agriculture, ports, shipping, industrial and residential activities, waste disposal, as well as medical applications. For most TEs, their final sink in the marine ecosystems is their permanent burial in sediments and/or mineral deposits. Both CC and human activities affect these sources and sink at various spatial and temporal scales, thereby altering the TE partitioning between and across different environmental spheres (atmosphere, hydrosphere, geosphere, and

Fig. 2 | Conceptual diagram of the natural and anthropogenic sources, sinks, and transport pathways of trace element contaminants in coastal ecosystems, which may interact with climate change (CC) drivers. The red arrows indicate the expected directions of changes due to CC. SGD submarine groundwater discharge, WWTP waste-water treatment plant.
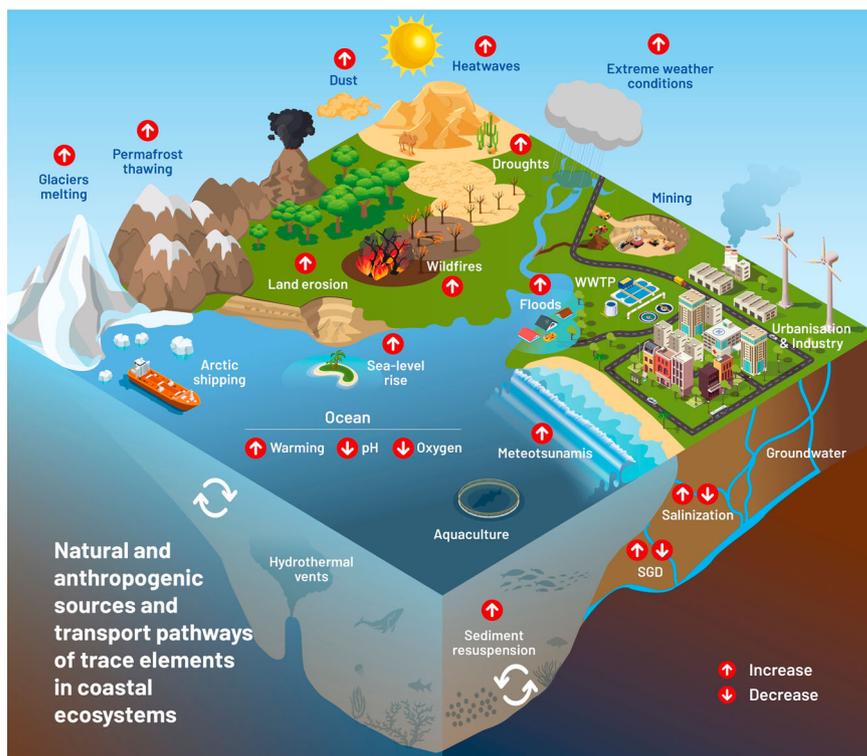
biosphere), their transport pathways, their internal cycling processes, and their physical, chemical, and biological drivers[26] as exemplified schematically in Fig. 2.

Natural sources and transport pathways

Hydrological and eolian transport pathways. Observations and climate model simulations indicate significant changes in the hydrological cycle in response to CC, resulting in alterations in the precipitation amount, intensity, and frequency[27]. Global average precipitation is projected to increase by ca. 2% per °C in response to global warming with the highest increase in annual precipitation expected in high-latitude regions[28]. Particularly, Arctic precipitation is expected to rise significantly, far exceeding the global average value, with a projected 4.5% increase per °C in this century[28]. This process is happening along with persistent permafrost thaw and snowmelt, leading to increased river discharge and allowing more and deeper groundwater flow to occur, thus, causing large-scale changes in hydrologic fluxes[28–30]. These changes impact the chemical composition and residence time of chemical exports[29,30]. Conversely, regions such as Central and South America, Australasia, the Mediterranean, and southern Africa are expected to experience worsening aridity, heightening the risk of droughts and fires[31].

Atmospheric dry and wet deposition. Atmospheric deposition has long been recognized as a major source of TE such as Al, Cu, Co, Fe, Mn, Pb titanium (Ti), vanadium (V), and Zn to the surface ocean[32,33]. While the anthropogenic emissions of TE into the atmosphere emitted through waste incineration, burning of fossil fuels, and traffic-related sources decreased since 1990 due to technological innovation[34], there has been a notable increase in global dust mass loading since pre-industrial times. This increase, estimated at 55 ± 30%, is largely attributed to natural and anthropogenic induced CC including desertification, droughts, and wildfire emissions, as well as changes in land use[35,36]. Climate model simulations, however, have difficulties in reproducing the observed historic increase[37] and future changes in dust concentrations diverge widely, that is, it is not clear whether dust concentrations will increase or decrease with CC[36,38]. Atmospheric dust concentrations over the North Atlantic, for instance, vary in response to drought cycles in Africa and large-scale atmospheric features such as the North Atlantic Oscillation[39]. The behavior and impact of atmospheric TE deposits, including their solubility and post-deposition processes, are influenced by various environmental factors (Box 1). These complex interactions, combined with diverse sources, sizes, chemical composition, and solubility of atmospheric aerosols, make it challenging to predict future trends in atmospheric dust loading and its impacts on ocean ecosystems[40].

Box 1 | Atmospheric input specifics of TE into the surface ocean

Atmospheric metal deposits into the ocean influence phytoplankton growth rates, modify plankton community structure and impact marine biogeochemistry, ecosystem productivity, and carbon sequestration on the timescale of decades[40]. The variability in Fe solubility arises from several factors, including the aerosol source, atmospheric processing, and the physical, chemical, and biological characteristics of the receiving seawater, particularly its acidic conditions[40,174].

Upon deposition on the seawater surface, the behavior and impact of TE contaminants are governed by their fractional solubility in seawater[122], as discussed in "Bioavailability, bioaccumulation, and biological

responses" section. Since aerosol dissolution is particularly furthered in acidic conditions[174], the lower pH of seawater resulting from CC significantly affects the post-deposition dissolution rates of particulate rain and dust inputs[35]. Further, increased thermal stratification due to surface ocean warming will cause the mixed layer depth to shoal, which could increase the concentrations of aerosol-derived TE in surface waters[40]. Additionally, continuous and intense UV radiation in the surface microlayer could enhance the dissolution processes and the diffusive transfer of TE between the surface microlayer and seawater[175].

Box 2 | Riverine input specifics of TE

Rivers in eastern Iceland experienced a substantial increase in nutrients and trace and major elements fluxes into the ocean between 1961 and 2004, driven by global warming and increased runoff[51]. The average annual air temperature increased by 0.06 °C, leading to a corresponding increase in the average river runoff from 0.2 to 2.0 cm/year. This change triggered alterations in fluxes of dissolved Al, Cd, Cu, and Pb of 8.0–18.0%, 7.5–16.0%, 6.0–16.5%, and 8.0–17.5% per °C,

respectively[176]. Conversely, predictions for the Keersop catchment in the south of the Netherlands indicate a potential reduction in TE contaminants leaching by the end of the 21st century[177]. This unexpected outcome, despite predicted higher winter precipitation, is due to the drying of the catchment as a result of lower precipitation rates in summer. In addition to CC, the magnitude and spatial extent of land cover and type of land use are also changing, further altering river hydrology and

Box 3 | SGD specifics of TE

It is well documented that TE is sensitive to changes in salinity, pH, and redox potential, which control the solubility and stability of their dominant species[55,178]. A study on the distribution and controls of TE in shallow groundwater at two sites on the Baltic coast of Sweden indicates that CC might impact TE dynamics in the area as a result of the transition to more reducing conditions caused by increased precipitation, which could decrease the soluble pool of some TE (Cd, Cu, Ni, and U) in groundwater systems. Conversely, areas experiencing substantial decreases in annual net precipitation may experience the opposite trend[179]. Additionally,

increasing temperatures owing to CC might decrease groundwater O_2 supply and interfere with microbial processes, thereby further affecting biogeochemical cycling and TE concentrations in SGD[52,178]. For instance, increased Hg methylation or TE reduction could occur under these conditions[52,178]. Groundwater can also accumulate high levels of particulate TE due to their adsorption on Fe and Mn oxyhydroxides, increasing the solubility of various TE contaminants such as As, Co, Pb, or Zn at lower redox potentials[180].

Riverine inputs. Simulations with global hydrological and climate models[41–43] showed a significant increase in mean annual river discharge across the high northern latitudes for the period 2071–2100 relative to 1971–2000, due to large increases in precipitation and snowmelt. Similar trends were also projected for large parts of the tropical regions, including the Amazon, Congo, and Indus catchments, due to increasing monsoon rainfall[43]. Increased seasonality of river discharge was also projected for ca. 30% of the global area, primarily affecting temperate and continental climate zones[43]. Conversely, decreases in mean annual river discharge are anticipated for regions in the mid-northern latitudes and southern latitudes, including the southern United States, Central America, Europe, the Middle East, eastern China, southern Africa, and southern Australia[42,43]. This altered hydrologic cycle resulting from CC is poised to affect weathering, denudation, and riverine runoff, consequently influencing the fluxes of TE contaminants into the coastal ocean across various spatial and temporal scales[44]. Furthermore, increased runoff is expected to accelerate leaching, resulting in enhanced TE loads[45,46] (Box 2).

Submarine groundwater discharge (SGD). The exchange of groundwater between land and sea, known as submarine groundwater discharge (SGD), is an essential hydrogeological feature that can be a major pathway for delivering contaminants to the coastal zone, influencing coastal ocean chemistry[47,48]. In addition to the land-to-sea flow via SGD, seawater intrusion into coastal aquifers also occurs. Although the magnitude of fresh SGD to the coastal ocean was reported to be approximately 1% of the annual river discharge[49], it represents a globally important source of nutrients, carbon, and TE contaminants to the ocean[49,50].

There are early signs indicating that both the quantity and composition of SGD are undergoing changes due to ongoing CC and urbanization. For example, projections for Northern Europe suggest a moderate (0.2 m) increase in average groundwater levels over the next two decades, largely driven by increased precipitation, potentially increasing the mobilization and flux of TE through groundwater by a factor of 2–10[51]. In addition, seasonal and spatial variations in groundwater, driven by higher evapotranspiration in spring and summer in warmer climates, as well as warmer winters in snow-dominated regions, should also be expected. A decrease in fresh groundwater recharge, coupled with sea-level rise, could lead to significant seawater intrusions into coastal aquifers and prolonged residence times[52,53]. The associated shift of the freshwater-seawater interface can

disturb the SGD chemistry and affect the biogeochemistry of aquifers that have not been in contact with seawater for centuries[47,54].

Furthermore, the projected increase in the magnitude and frequency of floods and droughts, attributed to CC, may impact groundwater quality[55]. During floods, extremely high groundwater levels can increase TE solubility or facilitate their transport by particles from the topsoil[51]. Contaminated groundwater is already impacting TE concentrations in stream water under both, extremely low and high river flows, potentially diminishing the efficacy of remediation efforts for TE contamination, such as those arising from mining activities[56]. Consequently, the inland expansion of subterranean estuaries, increasing temperatures, and extreme events becoming more frequent due to CC, could lead to greater mobility of hazardous TE and subsequent contamination of coastal ecosystems (Box 3).

Sediments and coastal erosion under extreme weather events and sea-level rise

Sediments as a potential source of contaminants. Sediments and soils in coastal ecosystems represent an important sink and reservoir for contaminants, linking the atmosphere, geosphere, and hydrosphere through natural and anthropogenic TE fluxes. However, sediments can also act as a secondary source of historically accumulated contaminant loads, which may be released during events that disrupt sediments, leading to the remobilization, transport, and redistribution of contaminants[57]. Mobilization and abrupt changes in the availability of TE may occur due to hydraulic perturbation, caused by erosion, dredging, trawling, turbulent mixing, tsunamis, biological processes (bioturbation), and modifications of the chemical milieu that disrupt the equilibrium partitioning between water and sediment[58–60]. The mobility of TE in sediments is governed by various environmental factors, such as redox potential, pH changes, and DOM interactions (Box 4). The lack of field observation on the biogeochemical responses of sediments to pressures driven directly or indirectly by CC, particularly in a multi-pressure context, poses considerable challenges for further investigation[46].

Extreme floods. Flood and extreme flood events, with extreme floods being high-magnitude events characterized by exceptionally high water levels significantly exceeding the historical flood levels for a given area, are amplified by CC[61]. Globally, the absolute flooding damage from flooding could increase by a factor of 20 by the end of the century[62]. By 2100, if no

Box 4 | Factors influencing TE mobility in sediments

Sediments in coastal ecosystems are naturally exposed to highly dynamic conditions. The mobility of TE will depend on a complex mosaic of various factors, including (i) the sediment redox potential, which directly alters the chemical and physical speciation, and hence the mobility of redox-sensitive elements (e.g., As, Cr, Fe, Mn), (ii) pH decrease, which results in the dissolution of carbonates in sediments and soils, usually increasing the mobility and solubility of metal cations (e.g., Cd, Cu, Ni, Pb, Zn)[181], (iii) dissolved organic matter (DOM) release from sediments, which can chelate and mobilize TE contaminants, (iv) reductive dissolution of Fe and Mn hydroxides, which releases adsorbed and co-precipitated TE, and (v) sulfate reduction, which immobilizes contaminants into sulfide precipitates[57,182,183], and/or produces large amounts of dissolved organic carbon (DOC), which increases TE bioavailability[184]. For instance, the re-oxidation of anoxic sediments in the Baltic Sea was found to remobilize TE into the water phase in the order of days, with the redox-sensitive metals (Mn, Mo, U, and As) being most affected[69]. While research on single CC stressors exists, understanding how multiple stressors will affect sediment biogeochemistry along the land-ocean continuum in both present and future scenarios remains underdeveloped[46]. Consequently, predicting the release, ultimate fate, and bioavailability of TE contaminants during disturbances is challenging, especially in the case of multi-stressor influences, associated with CC drivers.

Box 5 | Examples of TE during extreme floods

Studies conducted in the USA, following Hurricane Katrina (August 2005, Category 4, precipitation peak of ~417 mm) showed that extended flooding and reflooding of low-lying areas promoted contaminant redistribution, with levels of As and Pb in soils and sediments exceeding regulatory thresholds for human health by up to 70- and 5-fold, respectively[185,186]. In contrast, the record discharges caused by Hurricane Irene (August 2011, USA, Category 1, precipitation total of ~250–360 mm) and Tropical Storm Lee (September 2011, USA, precipitation total of ~250–500 mm) had a limited effect on the sediment chemistry of Cr, Cu, Pb, and Zn[187]. Conversely, Hurricanes Emily (July 2005, Category 4, precipitation total of ~208 mm) and Wilma (October 2005, Category 3, precipitation total of ~76–337 mm), which hit seagrass meadows in the Caribbean, had positive follow-on effects on marine organisms by flushing out toxic TE and the replenishing the meadows with essential TE, like Fe[188]. Additionally, a study on tsunamis revealed that the 2011 Japanese tsunami caused an increase in the concentration of platinum (Pt) in the water column by at least a factor of 2[189]. Similar impacts can be expected for other TE contaminants, particularly in shallow coastal regions that are strongly affected by meteorological tsunamis, which are tsunami waves generated by mesoscale atmospheric processes such as atmospheric gravity waves, tropical cyclones, pressure jumps, convective storms, frontal passages, squalls, and hurricanes[190–192]. Meteotsunamis climate modeling is in its early stage, but some models suggest that meteotsunamis could increase by one-third with higher spatial variability under extreme warming conditions (RCP 8.5 scenario)[192].

coastal protection or adaptation measures are implemented, 48% of the world's land area, 52% of the global population, and 46% of global assets will be at risk of flooding[63].

Extreme floods are more complex than typical events because of the differences in duration, intensity, and extent (river/estuary/bays versus watershed), and it is predicted that extreme floods will account for 68% of the global coastal area flood events, with regional sea-level rise contributing the remaining 32%[63]. Flooding alters the physicochemical properties of sediment and soils and influences biological processes[64] (Box 5). Extreme floods can also promote the transport and translocation of TE contaminants from deeper anoxic sediments, exposing organisms to high concentrations of TE[65,66], and causing ecological adverse effects and health risks through various exposure routes, including bioaccumulation[67].

During flooding, rapid depletion of oxygen due to the loss of contact with the oxygenated atmosphere leads to a decrease in redox potential. The associated material transport influences the environmental fate of TE, but the effect on mobility varies among contaminants[68]. Metal partitioning between particulate and dissolved phases can change during flood events, i.e., the particulate phase may be diluted by coarser, less contaminated particles eroded from bed sediments, while the dissolved phase may become enriched owing to mobilization[69–71]. Further, high-magnitude flood events can wash out bottom sediments, and lead to a decrease in fine-sediment and TE concentrations in labile fractions and, consequently, affect the downstream transfer of contaminants (e.g., Cr, Cu, Fe, Ni, and Zn)[72]. Moreover, turbulent flow conditions during these events may favor strong TE mobilization due to sediment resuspension, the desorption of TE, and the transformation of TE into more bioavailable forms[57,70].

Extreme droughts. Extreme droughts, characterized by prolonged periods of less than average precipitation that significantly exceed the typical or historical drought conditions for a given area, are projected to increase in magnitude and/or frequency in most dry subtropical regions[73]. Droughts favor water quality deterioration regarding TE contamination, temperature, and eutrophication[74] (Box 6). Further, the drying up of contaminated areas may expose TE-enriched sediments to the atmosphere that may be eroded and transported away from their original deposition site[75]. As global water scarcity increases and the prospect of more frequent or more intense droughts and heatwaves remains, managing drought-driven TE contaminant flux will become increasingly important and requires the understanding of the system's hydrology and sediment transport dynamics.

Box 6 | Examples of TE during extreme droughts

The extreme drought in Australia between 1995 and 2009 induced extensive oxidation of sulfidic sediments, causing widespread acidification and dissolution of TE contaminants in sediments[193]. The resulting mobilization of contaminants and REE was recorded in sediments from the River Murray estuary in Australia[194] showing that estuarine sediment can record drought events and convert them into a long-lasting geochemical signature.

Box 7 | Examples of TE during sea-level rise

A recent experimental study using highly contaminated coastal sediments showed that sea-level-induced flooding promotes a significant release of contaminants, with As levels increasing by up to 150% compared to pre-disturbance levels[66]. The study also observed a notable increase in the more mobile and toxic form of As (i.e., As(III)) after the disturbance. Additionally, another study showed that sea-level rise mobilizes As from contaminated coastal soils through the reductive dissolution of As-bearing mineral oxides with less As release in seawater scenarios compared to river water, likely due to the inhibition of oxide dissolution by seawater[195].

Sea-level rise. Predicted sea-level rise will increase the flood risk in coastal zones, exacerbate erosion, and lead to the salinization of groundwater and soils[76]. This change will have major implications to transport of sediments and associated TE contaminants and their residence times in coastal systems[77] (Box 7). For instance, sea-level rise accompanied by warming may produce more anoxic conditions in mangrove environments, not only due to increased flooding but also due to less bioturbation, causing enhanced As mobility and suppressing Sb mobility[78].

Coastal plains in many parts of the world contain acidic soils that are highly vulnerable to sea-level rise. These soils are characterized by high concentrations of exchangeable and hydrolyzable acidic metal cations such as Al and Fe, along with other toxic TE released by the weathering of aluminosilicates[79]. Flooding by seawater may cause rapid, short-term increases in acidity via the desorption and mobilization of acidic metal cations, creating a positive feedback loop that further increases acidity[80]. This process can lead to declining water quality due to the mobilization of high concentrations of Al, Ni, and Zn[80]. These examples highlight the importance of understanding the factors that influence the mobility of individual TE at a specific site to predict the impacts of increased flooding duration and magnitude on their environmental fate, bioavailability, toxicity, and risks to affect ecological integrity and human health.

Cryosphere. Polar regions are highly susceptible to a warming climate and, as such, are on the frontline of CC. The main effects of CC in these regions will be visible through changes in TE contaminant transport pathways, including atmospheric, riverine, sea-ice, ocean currents, coastal erosion, groundwater discharge, and biological transport driven by migrating animals (e.g., Hg biomagnified in birds and transported over large distances)[81]. Further, sea-ice reduction is expected to alter surface albedo, salinity gradients, TE input, and the exchange of heat, moisture, and trace gases between the atmosphere and ocean. These changes affect the stability of the water column, ocean-atmosphere circulation patterns, biological composition, and in-situ biogeochemical conditions, thereby affecting the entire ocean including coastal areas[81]. Thus, the polar seas and the cryosphere act as precursors for global changes in other ecosystems.

An important aspect of CC in polar regions is the change in the biogeochemical cycling and dynamics of TE since the cryosphere can be rich in nutrients, organic matter, and TE[82,83]. Consequently, permafrost thawing, glaciers/ice melt, and increased river runoff are expected to enhance the flux of nutrients and TE to the polar seas, potentially affecting the biological productivity in these ecosystems[84].

Ice and glaciers melting is a foreseeable CC factor with many concerning implications[76]. Sea ice serves as a dynamic interface between the atmosphere and the coastal ocean, playing a crucial role in the biogeochemical cycling of TE[85]. For example, concentrations of TE in the surface water of the cove (Antarctic coastal waters) have increased substantially due to melting ice, with Fe increasing 18-fold, Al and Mn increasing 8–10 fold, and Co, Cr, Ni, and REE increasing up to 4-fold[86]. Ice melting also allows the input of atmospheric dust, which has been observed in Cu and Cd profiles, showing surface maxima in the Ross Sea[87]. Glaciers also acquire a significant load of terrigenous material through glacial erosion processes, atmospheric deposition, and direct contact with shelf sediments. Thus, glacier fragments contain elevated levels of land-borne material, including TE, which are released directly into the surrounding water during melting and thaw cycles[88–90].

Enrichment with particulate TE has been observed in both Arctic and Antarctic Sea ice relative to underlying waters, with Al and Fe showing the highest enrichment[91,92]. Additionally, the colloidal phase (>80%) was enriched in cryosphere reservoirs for Cd, Fe, Mn, Ni, and Zn, whereas seawater was dominated by the truly dissolved phase[83]. Given the responses to CC, an important question is how changes in sea-ice coverage, thickness, and melt season length, including the increased dispersal of trapped particulate and colloidal TE and increased direct atmospheric deposition of TE to the surface ocean, will affect TE residence times and bioavailability in polar seas[83,93].

Overall, the Arctic is more vulnerable to environmental changes connected to CC than the Antarctic due to greater seasonal warming and freshening, as well as its lower alkalinity and nutrient limitation[94]. The Arctic Ocean, influenced by higher freshwater input, less biological uptake, and/or less scavenging removal under the sea ice, has exceptionally high concentrations of dissolved bioactive TE such as Co, Cu, Fe, Ni, and Zn[82], thereby differing from other oceans. In addition, the release of materials stored in permafrost poses significant risks of increased contaminant levels in Arctic environments (Box 8). Mercury is particularly significant

Box 8 | Permafrost melting threats

A noteworthy increase in contaminant levels in the Arctic environment is expected from enhanced transport due to melting permafrost[81]. With the Arctic warming, up to 65% of its near-surface permafrost (<15 m depth) could be lost by 2100, threatening to release materials that have been stored for thousands of years[196]. Progressive slow thawing of peat ice could double or triple the annual riverine export of Cs, DOC, phosphorous (P), and Zn over the next 100 years, and increase the export of Al, As, Ba, Cd, Co, Ga, Mo, and Rb by 20–40%, and for boron (B), Fe, Nb, Ni, Pb, and U by 10–20%[197]. Additionally, thousands of industrial sites are located on permafrost in the Arctic that have been neglected in CC impact analyses, even though they are posing serious environmental threats due to probable permafrost thawing before the end of this century[198]. Abrupt thawing processes are another concern since these short events rapidly expose the deeper permafrost layers that have inherently high levels of biological, chemical, and radioactive materials[199]. For example, concentrations of Al, Ca, Fe, Nd, P, Si, and U were 4–10 higher in surface waters at a site of abrupt permafrost collapse in the Western Siberia Lowlands, compared to steady thawing in the same region[200]. Increased frequency of freeze-thaw cycles in permafrost landscapes could also favor the transport of TE to the Arctic Ocean from soils in the more stable colloidal forms[201]. However, ongoing CC will likely decrease the export of colloidal and dissolved organic carbon, as well as various TE, from peat soil to lakes, and rivers in the Western Siberia Lowlands and the Arctic Ocean due to the northward shift of the permafrost zone boundaries[202–204].

Box 9 | Mercury in the Arctic region

Mercury (Hg) is of special concern in polar regions due to its tendency to accumulate in high latitudes. Glacier melting, permafrost thawing, and coastal erosion release sediments which increase the export of Hg from Arctic catchments[205], thereby impacting transport pathways, speciation, and cycling of Hg[206]. Thus, methylmercury (MeHg) and total Hg concentrations in lakes, wetlands, and basin tributaries show a strong temporal trend corresponding to snowmelt water inputs to the downstream Arctic[207]. In northeast Greenland, glacial lake outburst floods contribute 5–10% of the river discharge and 15–31% of the Hg export[208]. It is estimated that permafrost in the Northern Hemisphere stores 200–800 Gg Hg[209,210], which could represent nearly twice as much Hg as found in all other soils, oceans, and the atmosphere combined[210]. This storage capacity makes this environment a key potential source of Hg to rivers, estuaries, and the Arctic Ocean, where it may become bioavailable to

food webs, posing potential environmental risks. However, a recent study showed that most marine and terrestrial mammal species in the Arctic are currently at low risk of health effects from Hg exposure[211]. Nonetheless, human populations in the Faroe Islands, who rely on traditional fishing practices, have been found to exceed tolerable weekly intake levels of Hg[212]. Predicting future biotic Hg trends is complicated by the large legacy Hg inventory[213] and the complex processes controlling its recycling, speciation, bioavailability, methylation, and biological uptake, which are further exacerbated by CC[214]. For instance, a study conducted on Atlantic bluefin tuna (*Thunnus thynnus*) showed a 56% increase in tissue MeHg concentrations due to rising seawater temperature between 1969 and recent peak levels, despite a general decline in MeHg concentrations stemming from reductions in Hg loading[215].

concern (Box 9). Transport of these TE downward along the Transpolar Drift is facilitated by an increased concentration of organic ligands, originating mainly from Arctic rivers[82,95,96]. This process which will have implications for marine productivity not only in the Arctic region but also in the North Atlantic[82,95,96].

Anthropogenic sources

Past and present anthropogenic sources/activities. Rising industrial production, increasing demand for mineral resources, energy, and food, and the continuous development of high-technology products have, among other things, led to an unavoidable increase in TE contaminant inputs into the environment, particularly in coastal marine zones, via atmospheric deposition, surficial runoff, wastewaters, SGD, and riverine input[97]. It has been estimated that anthropogenic activities have increased the global fluxes of toxic metals such as Pb by more than tenfold[98] and of Hg by a factor of 3–7[99] relative to pre-anthropogenic levels[97]. A substantial fraction of these TEs eventually end up in the ocean[97]. Silver (Ag), an element highly toxic to marine biota, is increasingly found in coastal seawater of various regions[100] owing to atmospheric deposition from coal burning and riverine input from the increasing use of Ag nanoparticles in antibacterial consumer products[20]. Additionally, the shipping industry and associated port activities, such as the use of scrubbers in ship stack fumes[101], antifouling paints, and biocides, can result in the coastal and oceanic discharge of Cu, V, and Zn[102–104]. Plastics, a major concern in the 21st century, can also act as vectors for the transport of TE contaminants to the marine environment since plastics contain TE, including high concentrations of Cd and Zn, and can also adsorb TE, notably Cu, Pb[105,106], and REE[107]. In addition, decreasing pH has been shown to reduce the net adsorption of Cd, Co, Ni, and Pb to plastic pellets in estuaries[108], raising questions about how predicted ocean acidification might affect the interaction between TE and plastics.

Moreover, the decline of sea-ice cover is opening new Arctic shipping routes and allowing access to previously inaccessible areas. These changes lead to secondary follow-on effects, including economic growth, increased maritime activity which increased by 75% from 2013 to 2019[109], population growth, changing land use, and increasing land-based activities, all of which pose substantial risks to the sub-Arctic and Arctic regions by introducing new and more TE contaminants into the Arctic environment such as Cd, Cu, Hg, Pb, and Zn[109–111]. A similar situation is occurring in the Southern Hemisphere, where warming, population growth, and industrial development are increasingly introducing TE contaminants to the Antarctic environment, including Cu and Pb, owing to mining and smelting activities in Australia, Chile, Peru, Zaire, and Zambia[110]. Despite these observations, it remains largely unknown how anthropogenic sources and the biogeochemical behavior and cycling of associated TE (including their interaction) will be affected

by CC drivers, making it difficult to predict the environmental impacts of the emerging CC–TE interactions on coastal areas.

Future anthropogenic sources/activities. Anticipated future activities that will affect the TE contaminants input to coastal environments include the exploitation of natural resources (e.g., land and deep-sea mining), the establishment of coastal infrastructure (e.g., wind turbines, seawalls, coastal roads, land reclamation), the development of new and emerging technologies, the expansion of aquaculture, and the implementation of geoengineering activities for climate mitigation and adaptation[20]. The latter has found particular attention as it is now increasingly seen as a necessary step to achieve the Sustainable Development Goals of the Paris Agreement. Proposed marine geoengineering activities range from the manipulation of the carbon cycle (e.g., carbon removal activities such as Fe fertilization, artificial upwelling, CO_2 storage and capture, and alkalinity enhancement), to albedo modification, and hybrid technologies[20,112]. For example, increasing ocean alkalinity by adding Ca^{2+} or Mg^{2+} via minerals (e.g., olivine, calcite) or alkaline industrial residuals, theoretically leads to substantial CO_2 uptake from the atmosphere and a decrease in ocean acidification. However, these minerals also contain high amounts of TE (i.e., Cd, Cr, Cu, Fe, Ni, and Pb)[113] that could negatively impact the marine environment. Olivine-rich rocks, for example, can contain up to 30–50% Fe and a high abundance of Cr and Ni, with the latter being toxic to marine organisms above a certain threshold[114,115]. So, while initially beneficial, the knock-on effects of adding large amounts of geological material to marine systems have not yet been systematically studied, especially in the context of a (multi-stressor) changing ocean and CC drivers. The same is true for other anthropogenic activities, such as deep-sea mining. Secondary impacts of CC, such as the corrosion of coastal infrastructure (e.g., wind turbines, dikes, coastal roads, bridges, naval bases, harbors), however, have been fairly well studied[116,117].

Investigating the short- and long-term impacts of emerging CC–TE interactions remains critical to environmental risk assessments before any large-scale ocean solutions and new anthropogenic activities (e.g., deep-sea mining) are approved and implemented[20]. The societal challenge is to develop integrated and coordinated cross-sectoral approaches that balance the sustainable use of the ocean with its protection[23].

Sensitivity of trace elements cycling to climate change

In seawater, the reactivity and bioavailability of elements are determined by their chemical and physical speciation. The latter is often also referred to as fractionation, a term used from now onwards. While changes in the speciation of the carbonate system are key to ocean acidification, they also influence the fractionation and speciation of TE. These changes affect the biogeochemical cycles, transport (mobility and solubility), reactivity (residence times, sorption, and desorption), and biological availability (toxicity,
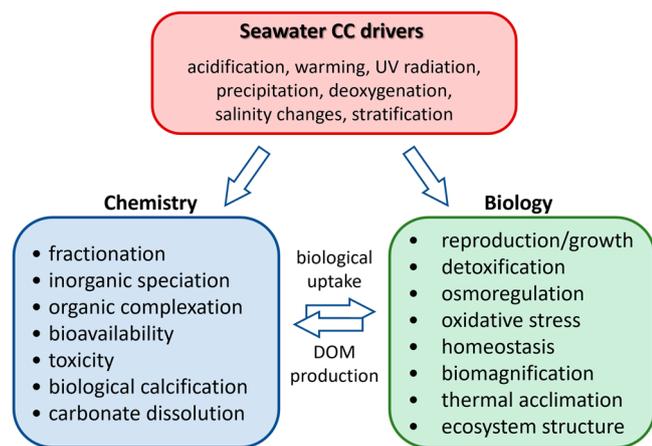
Fig. 3 | Schematic showing the simplified relationships between climate change drivers, trace element chemistry, and biology in seawater. The arrows indicate interactions. CC Climate change, DOM dissolved organic matter, UV Ultraviolet.

Table 1 | Current and future predictions of TE speciation in surface seawater using RCP 2.6 and RCP 8.5 predictions (IPCC, 2019)*

Surface seawater conditions	$t/°C$	20.2	20.9	22.8
	f_{CO2}[a]	425	478	1000
	pH_F[b]	8.105	8.063	7.786
	pH_T[c]	8.012	7.968	7.686
	$[OH^-]$[d]	1.91	1.84	1.13
	$[CO_3^{2-}]$[d]	75	70.4	42.4
	DIC[d]	2065	2081	2189
% free metal	Mn(II)	94	94	97
	Fe(II)	28	29	40
	Co(II)	90	90	94
	Ni(II)	78	79	84
	Cu(II)	7.5	8	13.2
	Zn(II)	88	89	93
	Cd(II)	100	100	100
	Pb(II)	5	5.2	6.8
log10 [total / free]	Al(III)	8.5	8.4	7.6
	Fe(III)	11.6	11.5	10.8

*Calculations were performed with MarChemSpec 1.10[172], assuming a practical salinity of 35, a seawater composition based on Millero et al. (2008)[173], and a fixed alkalinity of 2300 µmol kg^{-1}seawater.
[a]f_{CO2} in µatm.
[b]$pH_F = -log_{10}[H^+]$ with concentration in mol kg^{-1}seawater.
[c]$pH_T = -log_{10}([H^+] + [HSO_4^-])$ with concentrations in mol kg^{-1}seawater.
[d]concentrations in µmol kg^{-1}seawater.

bioaccumulation, and trophic transfer) of TE, thereby altering their fate and impact in the marine environment[7,118]. Figure 3 shows a simplified schematic of the anticipated effects of CC drivers on TE chemistry and the related biological effects in seawater.

Chemical speciation and fractionation

In the marine environment, TE exists in numerous forms (species). Physical species include particulate (>0.2 µm) and dissolved (<0.2 µm) phases, which can be subdivided into colloidal (1 nm–0.2 µm) and 'truly' dissolved species (≤1 nm). Along the entire physical size range, TE can also be distinguished by their chemical form, such as oxidation state, and complexation with inorganic (incl. Cl^-, CO_3^{2-}, OH^-, SO_4^{2-}) and organic ligands (incl. siderophores, thiols, phytochelatins, humic(-like) and exopolymeric substances).

Transformations among the various TE species and size fractions occur continuously at different temporal and spatial scales and are controlled by complex and interlinked biogeochemical processes that are dependent on environmental conditions[7,119]. This control makes TE speciation inherently susceptible to CC-related effects such as ocean acidification, increasing anoxia, and salinity changes[7].

In the context of CC, the interplay between TE and ocean acidification is the most studied. Ocean acidification will decrease the concentration of OH^- and CO_3^{2-} ions, which form strong complexes with divalent and trivalent metals impacting their speciation, solubility, adsorption to organic material, redox reactions, and potential toxicity. These anions are expected to decrease in surface waters by 4% and 6%, respectively, if atmospheric pCO_2 increases up to 478 µatm by the end of the century (Table 1). The TE most affected by this decrease will be those that form strong complexes with carbonates, such as Co^{2+}, Cu^{2+}, Ni^{2+}, Pb^{2+}, UO_2^{2+}, or Zn^{2+} and hydroxides, such as Al^{3+}, Cr^{3+}, or Fe^{3+}.

Most TEs become more soluble at lower pH and increasing temperature (Box 10), with those demonstrating a high degree of carbonate or hydroxide complexation being impacted the most. An exception to this trend is the oxyanion species of Cr (CrO_4^{2-}), Mo (MoO_4^{2-}), V (VO_4^{3-}), and

tungsten (W; WO_4^{2-}) for which solubility decreases (based on own calculations using an ion-pairing model written by CMG van den Berg, Univ. Liverpool)[120]. It should be noted that these estimates only pertain to the inorganic species and do not consider the effect of pH, temperature, and salinity on their organic complexation, for which little is known, although proton-binding characteristics of marine dissolved organic matter have recently been reported[121]. This complexation is specifically important for TE that are dominated by their organically complexed species, such as Cd, Co, Cu, Fe, Pb, or Zn[122]. The binding affinity of organic ligands with acidic functional groups (e.g., carboxylic) could be strongly affected by decreased pH. For Cu, the strength of organic-Cu interactions continuously decreases with the acidification of seawater[123]. However, the magnitude varies with DOC[124], suggesting that variability in DOC may have a greater impact on chemical speciation and bioavailability than the projected changes in inorganic TE speciation. Metals that form strong complexes with Cl^-, such as Cd or Hg, will be less affected by ocean acidification since Cl^- is insensitive to pH. Also, for TE that is predominantly in the free form (considering only inorganic species), such as Co^{2+} and Mn^{2+}, the pH decrease should lead to only relatively small changes in their free ion fraction[122]. Ocean acidification may also impact biological calcification and/or the dissolution of $CaCO_3$, which would increase Ca^{2+} concentration. The competition of H^+ and Ca^{2+} ions for binding sites with TE on organic ligands would then cause the release of TE from sediments and organic ligands[125,126].

Box 10 | Interactive effects of ocean acidification and temperature on TE

Considering not only the pH change and the change in CO_3^{2-} concentration but also the increase in sea surface temperature by 0.7 °C and 2.6 °C, as predicted by RCP 2.6 and RCP 8.5 (IPCC, 2019), respectively, the most bioavailable forms of TE, i.e., the concentration of free TE ions (e.g., $Cu^{2+}_{(aq)}$), would increase up to 7% (Cu) under RCP 2.6 and up to 77% (Cu) under RCP 8.5 scenarios. These predictions are based on the seawater speciation model MarChemSpec[172] (Table 1), which only pertains to estimates for inorganic TE species and does not consider the effect of abiotic parameters on organic TE complexation. The potential impacts of these changes on coastal ecosystems remain unknown and require further study.

Ocean acidification also affects photochemical processes such as the production of O_2^-, HO_2, and H_2O_2 that can change the oxidation state of TE (e.g., Cr(III)/Cr(VI), Cu(I)/Cu(II), or Fe(II)/Fe(III)[122]) and oxidizes dissolved organic material. Additionally, altered UV irradiation due to the depletion of ozone can break down the available binding sites for TE on organic ligands, thus increasing the proportion of free TE ions and their bioavailability and/or potential toxicity[127,128]. Other implications of increased UV irradiation include UV-induced oxidation of Hg^0 to Hg^{2+}, which can then be methylated and biomagnified along the food chain[129], and the mobilization of dissolved TE (i.e., Al, Cd, Cu, Co, Mn, Mo, Ni, Pb, Ti) from sunscreen products[130]. While most studies have focused on the effects of ocean acidification in the water column, TE speciation is also affected in contaminated sediments, increasing their toxicity[131].

Although empirical data showed that temperature increases have a negligible effect on metal speciation, particularly on free ion activity[132], warmer, more stratified, de-oxygenated, and acidified seawater will change microbial community structure and function in general[133]. This change may impact microbial detoxification abilities, which are closely related to the production of organic ligands[134]. How CC drivers affect microbial ligand production and associated speciation of TE in the future remains to be determined.

Bioavailability, bioaccumulation, and biological responses

Some TE (e.g., Hg) can be transferred through the marine food chain via bioaccumulation and subsequent biomagnification processes, affecting organisms at higher trophic levels and potentially impacting humans through seafood consumption[135]. For many TEs, the nature of the cellular

transport systems remains unknown, leading to some uncertainty on how climate- and acidification-related changes will affect bioavailability[136].

Climate change is expected to affect TE bioavailability and bioaccumulation primarily by altering seawater pH, temperature, oxygen levels, and salinity. Studies examining the combined impacts of CC and TE on marine biota, primarily focus on understanding ocean acidification's effect[126,137,138] (Box 11). Since divalent cations are predominantly transported via ATP-ases that bind to the free inorganic cations, acidification is expected to increase the bioavailability of Cr, Cu, Fe, and Pb but not of Cd, Co, Hg, Mn, Ni, and Zn. It has been shown that the high-affinity Fe uptake system of some phytoplankton cells depends on CO_3^{2-}, leading to a strong pH sensitivity[139]. Another point to consider is that the competition between H^+ ions and some TE, such as Cr and Cu, at the cell surface could diminish their bioavailability counteracting the increased free ion concentration[140–142]. However, the biotic uptake mechanisms are a bit more complex for TE whose speciation is dominated by complexation with organic ligands, such as Co, Cu, Fe, and Zn[143]. Traditionally, the strong organic complexes of Cd, Cu, and Zn were assumed to be non-bioavailable[144]. Contrary to these general assumptions, some Cu complexes with organic ligands have been shown to be available for uptake by phytoplankton and mussels[145,146]. Zinc organic complexes are relatively weak (e.g., compared to those of Co or Cu)[147] and a decrease in phytoplankton Cd and Zn uptake under decreased pH has been observed[148]. In contrast, in coastal seawater, an increase in bioavailable Zn was found under lower pH[149], demonstrating that the role of organic ligands in the interactions between acidification, speciation, and bioavailability is specific to the combination of TE and organic TE-binding ligands and far from being understood.

Box 11 | Interactive effects of ocean acidification and TE

Overall, the net effects of ocean acidification and TE contaminants on marine organisms depend on the specific TE, species, the life stage of the organism, and the degree of acidification[126,137,138]. For example, ocean acidification (pCO_2 1400 and 3000 µatm) increased Cu toxicity responses in the early life stages of the polychaete *Arenicola marina*[216] and inhibited antioxidant defenses against Cu toxicity in marine bivalves *M. petechialis*[217]. Ocean acidification (pCO_2 1500 µatm) also significantly alleviated the toxicity of Cd to coastal diatom *S. costatum*, although intercellular Cd accumulation increased[218]. It also increased Pb accumulation in the mussel *M. edulis* at pCO_2 conditions of 587–4154 µatm[219] and accumulation of TE such as As, Cr, Cu, Hg, Ni, Pb, Zn, but not Cd, from metal-contaminated sediments in juveniles of the benthic organism *R. philippinarum* (carpet clam) at pH conditions of 6.5–7.5[220]. In contrast,

the accumulation of Al, Fe, Mn, and Zn in tissues of adult *R. philippinarum* was independent of pH (pH conditions of 6.1–7.1), although significant associations between pH and biological effects were found[221]. A recent study using Hg stable isotopes ($Me^{202}Hg$ and $^{199}Hg(II)$), which allowed the quantification of Hg accumulation, Hg(II) methylation, and MeHg demethylation rates, suggested that pCO_2 (1600 µatm) may not significantly impact Hg bioaccumulation, metabolism, and microbiota diversity of the gut and digestive gland of cuttlefish[222]. This finding is supported by a meta-analysis[150], which showed that ocean acidification (−0.4 pH units) generally alleviates Hg bioaccumulation and toxicity in various marine organisms, possibly due to hydrogen ion competition and reduced membrane permeability.

Box 12 | Interactive effects of ocean warming and TE

Simulations of changes in Hg methylation rates with increased seawater temperatures project an average increase in MeHg concentrations and the subsequent uptake of MeHg by fish and mammals of 1.7% and 4.4% for predicted warming rates of 0.4 °C and 1.0 °C, respectively[212]. Similarly, a meta-analysis study by Wei et al.[150] found that warming of 4 °C increases Hg accumulation and its biotoxicity across all trophic levels of tested marine organisms to varying degrees (primary producers <herbivores <carnivores) by enhancing basal metabolic demand. However, long-exposure to ocean warming could decrease an organism's metabolism, which may prevent TE accumulation. This effect was observed in the mussel *M. galloprovincialis* in response to Hg pollution[223]. Higher temperatures were also found to increase Hg accumulation in the marine copepod *P. annandalei*, with subcellular fractionation revealing that the increased Hg concentration was stored largely in metal-sensitive

fractions directly related to metal toxicity[224]. Following an increasing thermal gradient, a tolerance decline to Cu was observed in the model organism *E. crassus*[225]. Elevated temperatures were also found to increase the sensitivity of oysters *C. virginica* to Cd at mitochondrial and whole-organism levels[226], whereas maximal Hg tissue concentrations in the surviving crustaceans *Daphnia magna* decreased with temperature[227]. Additionally, a study on REE demonstrated that the accumulation and toxicity of La were enhanced with increasing temperatures in the European eel (*Anguilla anguilla*)[228]. Blanckaert, et al.[229] showed that aerial dust enriches the water with essential nutrients and promotes coral photosynthesis at low temperatures or in bleached corals. Higher concentrations of Cd, Cu, Mn, and Mo in symbionts of dust-exposed corals were found as a consequence of the combined effect of temperature and bioactive TE concentrations.

Another possible effect of acidification on interactions between TE contaminants and biota, independent of changes in the bioavailable species in the medium, is its influence on the efficiency of the uptake and/or elimination systems. Reduced pH and elevated CO_2 concentrations can alter the physiological effects of TE contaminants and their associated bioaccumulation in marine organisms, which appears to be an even more important consequence of ocean acidification for marine biota than changes in bioavailability[126].

Another important driver of CC, impacting bioavailability and bioaccumulation, is ocean warming (Box 12). Studies have shown that TE uptake and accumulation increase with increasing temperature in numerous aquatic organisms[132]. In 80% of the cases, ectotherms were more affected. Since temperature changes have a minor effect on TE speciation, this can only be explained by the effects of temperature on biological factors[132]. Increased temperatures are expected to cause higher TE uptake by decreasing oxygen solubility, thus enhancing gill ventilation, however, it can also lead to higher rates of depuration and detoxification mechanisms[125,138].

Furthermore, increased precipitation and temperature due to CC will induce eutrophication and stratification, leading to the expansion of anoxic and hypoxic areas in aquatic ecosystems worldwide[125]. This additional stressor could exacerbate the effects of TE toxicity, with benthic species being particularly endangered as they are directly exposed to sediment TE contaminants and inhabit areas where hypoxia occurs more frequently than in the water column[125,150,151].

Moreover, while it has been found that moderate ocean acidification can mitigate TE toxicity for some marine species, ocean warming can counteract this positive effect, leading to increased TE toxicity under combined ocean warming and acidification conditions. This was shown for As, Cd, or/and Cu in anemones, copepods, corals, and various molluscs[138], as well as for Hg in numerous marine organisms[150]. It was observed that the effect of increasing CO_2 alone did not impact the bioaccumulation in mussels *M. galloprovincialis* but when combined with higher temperatures led to increased bioaccumulation of As, Cd, Cr, Ni, and Pb in mussels adapted to colder waters[152]. Likewise, the simultaneous exposure to elevated Cu levels, along with ocean acidification and warming exacerbated the detrimental effects on kelp's microscopic early life stages compared to the effects of the single stressors alone[153]. This synergistic effect was also observed in a study on REE (La and Gd), which revealed increased accumulation patterns in U. rigida[154]. Therefore, warming appears to pose a greater threat to estuarine and coastal organisms than acidification[150,155,156]. Given this context, it is crucial to consider the combined effects of acidification and warming to evaluate the risk posed by TE contaminants, especially under global change conditions.

Discussion

Despite the recognized importance and increasing interest in the effects of CC on TE sources, cycling, transport, and fate, research on these topics is still at an early stage. The knowledge gaps are large, and the depth to which they are addressed varies widely around the world and research topic. CC drivers such as ocean acidification and warming receive more attention in the context of TE contamination[157] than other equally important primary drivers such as deoxygenation, and secondary drivers such as sea-level rise, ice melting, changes in circulation/mixing, salinization, and extreme events. Additionally, the interactions and feedback mechanisms among multiple stressors are only poorly understood due to their intrinsic complexity and the lack of well-designed experiments that consider the diversity of potential simultaneous TE contaminants and CC stressors. Moreover, TE concentrations are usually determined in contaminant studies, while their speciation, which determines element reactivity and bioavailability, is largely overlooked. This lack of studies limits our understanding of current and future processes in the highly dynamic coastal marine environment. Addressing these knowledge gaps is crucial for making accurate predictions of coastal and ocean system changes, which are essential for effective human health and ecosystem protection measures. To build a comprehensive dataset, we suggest that scientists prioritize nine key research areas in the

future. This focus will help bridge current knowledge gaps and address science-policy requirements.

Adoption and development of best practices
We face ever-increasing methodological complexity and diversity owning to the interdependencies of CC drivers, the inherent interaction of multi-stressors in coastal environments, the emergence of new TE contaminants, as well as varying institutional capacities and capabilities. Additionally, the fragmentation of methods and data across regions, nations, and disciplines inhibits effective collaboration, global participation, data sharing, interoperability, and comparability[158]. A first and foremost requirement for TE contaminant research and monitoring is therefore the need to adopt and follow coherent, well-defined, and reproducible methodologies and quality assurance protocols[158–160].

Improved geographical and temporal research coverage
Coastal areas experience diverse environmental variations in space and time that affect TE biogeochemistry, organism responses, behavior, and trophic interactions, which are expected to worsen with CC. Comprehensive, long-term monitoring and research are needed in geographical regions that are currently poorly studied but highly affected by CC, such as polar regions, tropical coastal zones, various vulnerable environments such as coral reefs, and estuaries[52], and in natural archives of contaminants (e.g., estuarine and mangrove sediments). Ideally, the activities would be scaled to assess contamination impacts in all environmental compartments (i.e., atmosphere, sediments, water, and biota), and at a community and ecosystem level. To understand the global response of TE contaminants to CC, there is a need to better cover our understanding in the much-understudied Global South.

Research on so-far understudied sources and environments
The knowledge of the distribution of TE and their sources and cycling in different marine environments has substantially increased since the implementation of the GEOTRACES Program[17]. However, some sources, including eolian wet and dry deposition, and coastal shallow hydrothermal vents[161], and ecosystems like seagrass beds and tidal flats remain poorly studied[162] and should receive more attention given their high significance at regional scales.

Expanding biological research
The effects of key CC drivers on contaminants have been thoroughly explored in recent years, but mostly as single drivers using individual-based organism approaches[138,157]. Thus, current knowledge is limited to a relatively small set of model organisms. More holistic studies that assess community and ecosystem-level responses using a multi-stressor approach are still scarce[157] and limit the comprehension of the CC-contaminant interaction and their associated biological responses. Further, to account for organisms' ability to adapt to environmental changes, scientists should examine the whole life cycles of organisms. Therefore, to assess the long-term effects of CC and contaminants on marine biota, future studies should include multiple developmental stages, species from different geographic locations, and multiple taxa[126,137,138] to better address current management and policy needs (e.g., preservation of biodiversity and food safety[157]).

Research related to emerging contaminants
A number of new compounds and TE are introduced into the environment each year that have no natural use, termed as emerging contaminants. At present, TEs of emerging concern include TCE, REE, and PGE[16,97,163]. The introduction of emerging contaminants in the aquatic environment is gaining interest due to limited scientific knowledge about their distribution, cycling, transport, reactivity, toxicity, and ultimate fate, including bioaccumulation processes. Their impact on the coastal environment driven by CC remains unknown and requires urgent attention to avoid potentially irreversible damage.

Research on multiple stressors

While it is evident that CC is causing multiple stressor effects (see Figs. 1 and 2), most studies are concentrated on much simplified scenarios. Although this approach has greatly improved our understanding, it fails to reveal interactive and synergistic effects of multiple contaminants and CC stressors on single organisms, or entire ecosystems[19]. There is a clear need to increase the knowledge around the interactions and cumulative impacts of the growing mixture of TEs and other chemicals, starting with stressor pairs, to which marine biota are exposed under varying, interacting, and co-occurring CC conditions[157,164].

Development of in-situ sensors and low-cost methods and instrumentation

In-situ sensors or continuous samplers should be employed to capture the spatial and temporal variations of TE contaminants in dynamic coastal environments[165,166]. Although there has been some progress, additional research is needed to develop in-situ sensors and samplers capable of directly measuring TE concentrations and speciation in the aqueous phase[166–168]. However, in-situ sensors and samplers entail substantial costs from development and validation to production. To promote their widespread adoption, supporting the creation and operation of affordable versions is essential.

Development of better models

Gaps in the monitoring and understanding of processes are often filled by marine ecosystem models, which can also be a useful tool. However, our mechanistic understanding of the effects of CC drivers on TE contaminants, particularly their speciation and organic complexation, and their interaction with biology, is currently very limited, making model predictions and calculations non-trivial. Additionally, short-term fluctuations in environmental variables (e.g., salinity, temperature, and pH), common in coastal areas, are generally not accounted for in models. As we gather more high-quality data, with increased temporal resolution, including data on speciation and bioavailability, efforts to enhance modeling capabilities in the near future are underway. This effort is fueled by the current momentum to create a Digital Twin of the ocean (https://www.mercator-ocean.eu/en/digital-twin-ocean/) using machine learning tools.

Inclusion of trace element speciation in legislation and risk assessments

The implementation of appropriate legislation, regulation, and successful management strategies for contaminants is closely linked to the fundamental understanding of the key processes and factors controlling their speciation dynamics in marine ecosystems. However, contaminant speciation remains largely unapplied to hazard assessments, environmental risk assessments, and/or the regulations of contaminants. The biggest challenge to increase the use of speciation-based approaches for contaminants in regulations and assessments is providing practical approaches that are scientifically robust, easy, and fast to implement, and do not come with additional costs compared to the more traditional approaches for determining contaminant concentrations. Ongoing efforts aim to develop tools to meet these needs and thereby increase the ecological relevance of contaminant regulations[169]. A recent example in this direction is the MONITOOL project (https://www.monitoolproject.eu/), which aims to respond to European Directive demands for the assessment of the chemical status of transitional and coastal waters, by developing speciation-based environmental quality standards (EQS) using passive sampling technique for metals[170,171].

Addressing these nine research priorities related to the contaminant-CC nexus over the next years would facilitate the implementation of science-based policies and regulations that support effective mitigation and management strategies in vulnerable coastal areas and in so doing support the achievement of Sustainable Development Goal 14 ("*Life Below Water*").

Reporting summary

Further information on research design is available in Nature Portfolio Reporting Summary linked to this article.

Data availability

No new analytical data were generated or presented in this review. All sources of data, including databases where data was extracted from, are stated in-text.

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The authors declare no competing interests.

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