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Ecotoxicological effects of sunscreen derived organic and inorganic UV filters on marine organisms: A critical review

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ABSTRACT

Sunscreens are topical personal care products that provide protection against the sun's ultraviolet A (UVA) and ultraviolet B (UVB) radiation. Ultraviolet (UV) filters are compounds added to sunscreens to block, absorb, or reflect the sun's UV rays, but are of major emerging concern due to their widespread use and global distribution. They pose a significant risk to marine organisms owing to their chemical properties, including high lipophilicity which increases their bioavailability. The present review identifies and summarises the factors that contribute to UV filter pollution, their sources, pathways, and effects on marine organisms. We identify and evaluate the current knowledge base and gaps pertaining to their effects. Here, we retrieved 111 peer-reviewed articles from four academic search engines between January and October 2024 with the topic search relating to UV filters, sunscreen and ecotoxicology. Most publications (60 %) focused on the biological effects of organic UV filters, with oxybenzone (benzophenone-3) being the most studied (57 %). Fewer publications assessed the biological effects of inorganic UV filters (40 %). Throughout all search results, the most commonly tested species were in the class of bivalvia (24 %) and oxidative stress based assays were the most popular (organic studies 40 %, inorganic studies, 39 %). To enhance understanding, future research should explore a broader range of organisms and life stages, considering dietary uptake and realistic environmental conditions, including the use of UV lighting in laboratory settings.

1. Introduction

UV (ultraviolet) filters are used in sunscreens, personal care products (e.g., shampoos, lipsticks, shower gels) and various other commercial products including plastics, rubber, paint, and cement to enhance light resistance and prevent photodegradation (Beiras, 2021; Giraldo et al., 2017). UV filters are compounds that absorb, reflect or scatter UV radiation in the 280 to 400 nm range of solar spectrum (Parades et al., 2014; Sánchez-Quiles and Tovar-Sánchez, 2015). In recent decades, production has surged alongside the rising popularity of sunscreen (Schneider and Lim, 2019); driven by market shifts, sunscreens are now marketed and designed to appeal for everyday use, not just for holiday periods. The generation of sunscreen sprays, mousses, sticks, oils and powders, as well as skin-specific formulations suitable for those with allergies and acne has revolutionised the market to appeal to the wider consumer. In 2022, the global market for sunscreen products was estimated to have generated a revenue of 10.3 billion USD, which is expected to increase to 13.64 billion USD by 2026 (Marcin and Aleksander,

Abbreviations: AVO, Butyl Methoxydibenzoylmethane (Avobenzone); A-ZnO, 3-Aminopropyl- Trimethoxysilane coated Zinc Oxide; BEMT, Bis-Ethylhexyloxyphenol Methoxyphenyl triazine (Ethyl Triazine); BP, Benzophenone; BP-1, 2,4-Dihydroxybenzophenone (Benzophenone-1); BP-2, 2,2',4,4'-Tetrahydroxybenzophenone (Benzophenone-2); BP-3, 2-hydroxy-4-methoxybenzophenone (Benzophenone-3, Oxybenzone); DBT, Diethylhexyl Butamido Triazone (Iscotrizinol); DHHB, Diethylaminohydroxybenzoyl Hexyl Benzoate; D-ZnO, Dodecyltrichlorosilane coated Zinc Oxide; ED-PABA, Ethylhexyl Dimethyl P-Aminobenzoic Acid; EHMC, Ethylexyl Methoxy Cinnamate (Octinoxate); ENS, 2-phenylbenzimidazole-5-sulfonic acid (Ensulizole); ERA, Environmental Risk Assessment; ES, 2-Ethylhexyl Salicylate (Octisalate); ET, Ethylhexyl triazone; FDA, Food and Drug Admintsration (USA); HS, 3,3,5-Trimethylcyclohexyl salicylate (Homosalate); 4-MBC, 4-Methylbenzylidene Camphor; MBBT, Methylene Bis-Benzotriazolyl Tetramethylbutylphenol (Bisoctrizole); NP, Nanoparticles/nanoparticulates; nZnO, Nanoparticulate Zinc Oxide; $nTiO_2$, Nanoparticulate Titanium Dioxide; OC, 2-ethylhexyl 2-cyano-3,3-diphenyl-2-propenoate (Octocrylene); ROS, Reactive Oxygen Species; TiO_2, Titanium Dioxide; UV, Ultraviolet; ZnO, Zinc Oxide.

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2023; Statista Search Department, 2022). A recent estimation suggests that approximately 10 million tonnes of UV filters are produced annually for the global market, of which, an estimated 6000–14,000 t are released into coral reef zones annually (Marcin and Aleksander, 2023; Downs et al., 2016; Prakash and Anbumani, 2021). With the growing concern of photocarcinogenesis and photoaging linked to sun exposure, this has led to an increase in the use of photoprotective personal care products and consequently, an increase in the production of sunscreens (Schneider and Lim, 2019; Bosch et al., 2015).

UV filters used in sunscreens can either be organic (chemical-based) or inorganic (mineral-based) and each filter type can have different properties, structures and solubilities (Gonzalez et al., 2022). Organic UV filters, which consist of up to 55 different compounds registered for use globally (examples shown in Fig. 1), act by absorbing incoming radiation (Shaath, 2010; Daly et al., 2016; Giokas et al., 2007). In contrast, inorganic UV filters, which include titanium dioxide (TiO_2) and zinc oxide (ZnO), reflect or scatter incoming radiation (Caloni et al., 2021). Inorganic filters can also come in nanoparticulate versions, which are classified by at least one dimension of 100 nm or less, for example, nanoparticulate titanium dioxide ($nTiO_2$) (Chen et al., 2012). Typically, between three to eight different UV filters are used in a single organic based sunscreen formulation, which can make up to 15 % of the overall product mass (Schreurs et al., 2002; Sánchez-Quiles and Tovar-Sánchez, 2015). A single UV filter on its own has a limited absorption wavelength against UV radiation, and therefore, combinations are used to protect against the whole UV spectrum (Grabicova et al., 2013; Dong et al., 2022).

UV filters are considered emerging pollutants due to the rapid increase in their production and widespread use in recent years (Schneider and Lim, 2018; Marcin and Aleksander, 2023; Tsui et al., 2014; Richardson, 2010). Many of these filters are also dubbed 'pseudo-persistent pollutants' due to their continuous introduction via primary and secondary sources (Vione et al., 2013; Mottaleb et al., 2009). These compounds can enter the marine environment either directly or indirectly (Fig. 2). Direct pathways include via swimming or other recreational activities, and indirect pathways include sources such as effluent from wastewater treatment plants and domestic water discharges (Giokas et al., 2007; Tian et al., 2021). For instance, washing towels that have been used to dry sunscreen-coated skin, washing off residue during showering, and even via urine (Li et al., 2007).

Traditional sewage and water treatment technologies, such as ozonation, do not have the capacity to remove most UV filter compounds from effluent, due to their general low water solubility, high lipophilicity and high organic carbon-water coefficient (Fivenson et al., 2021; Hopkins et al., 2017; Lopes et al., 2022; Tian et al., 2021; Schneider and Lim, 2018). UV filters have been detected in the marine environment at concentrations in the ng L^{-1} to mg L^{-1} range (Sun et al., 2024; Cadena-Aizaga et al., 2020; Labille et al., 2020; Prakash and Anbumani, 2021; Tovar-Sánchez et al., 2013) in suspended matter (Sun et al., 2024), beach sediments (Downs et al., 2022; Astel et al., 2020; Tarazona et al., 2014), marine sediments (Apel et al., 2018, Mitchelmore et al., 2019; Tsui et al., 2015; Tsui et al., 2017), surface waters in busy, and touristy locations (Labille et al., 2020; Sánchez Rodriguez et al., 2015; Landeweer et al., 2024), nearshore and offshore coastal areas (Bargar et al., 2015; Tsui et al., 2014), as well as being recorded in remote locations such as Antarctica (Balakrishna et al., 2023; D'Amico et al., 2022; Emnet et al., 2014), and the Arctic (Tsui et al., 2014).

Given their ubiquity and lack of biodegradation due to high octanol water-partition coefficients (log $K_{ow} > 4$), UV filters have been detected in marine biota (Sang and Leung, 2016; Marcin and Aleksander, 2023; Wang et al., 2017). Exposure pathways of UV filters in the marine environment include aqueous/aquatic exposure from the water column, sediment exposure and trophic exposure through dietary uptake (Appendix 1). High octanol water-partition coefficients, or 'log K_{ow}', is a parameter that is used to assess the distribution of a substance in different environmental substrates and mediums. It is a measure of the

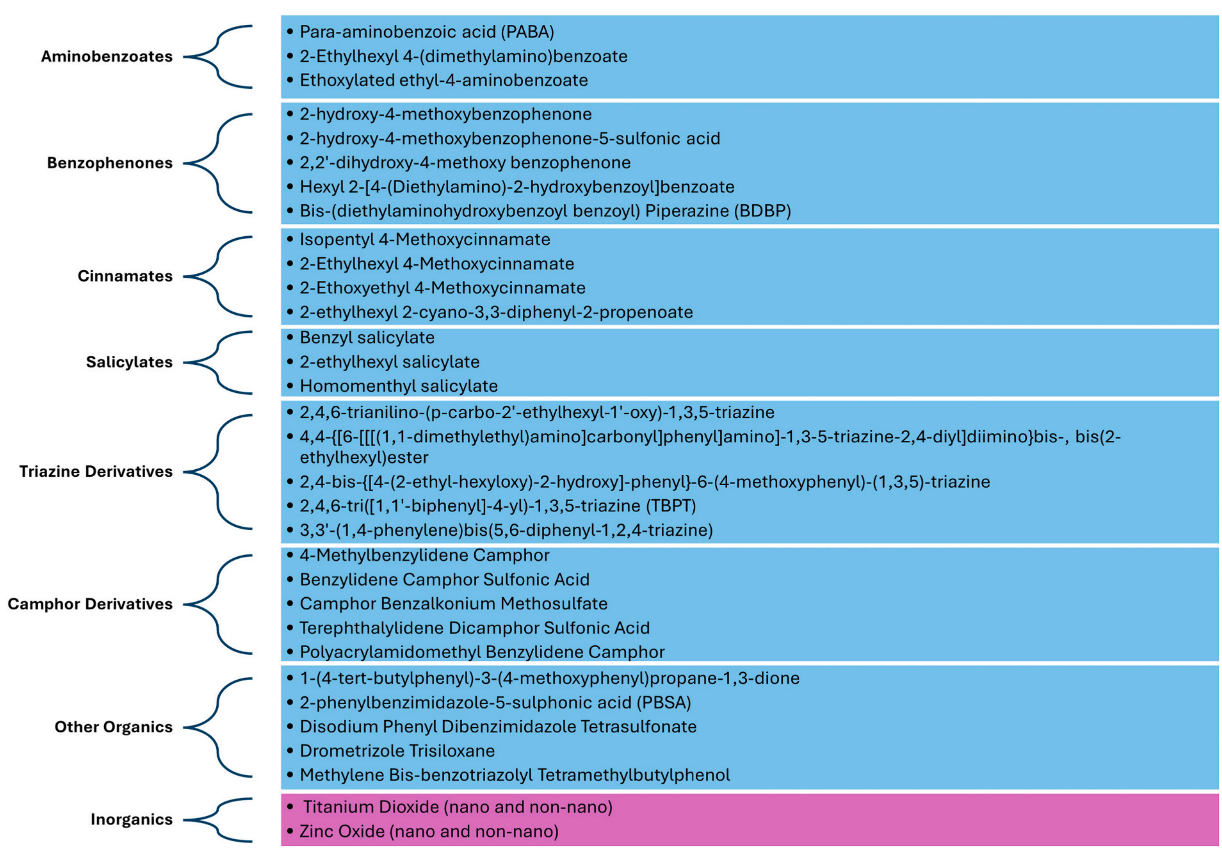

Fig. 1. Examples of chemicals used as UV filters in sunscreens and personal-care products globally.

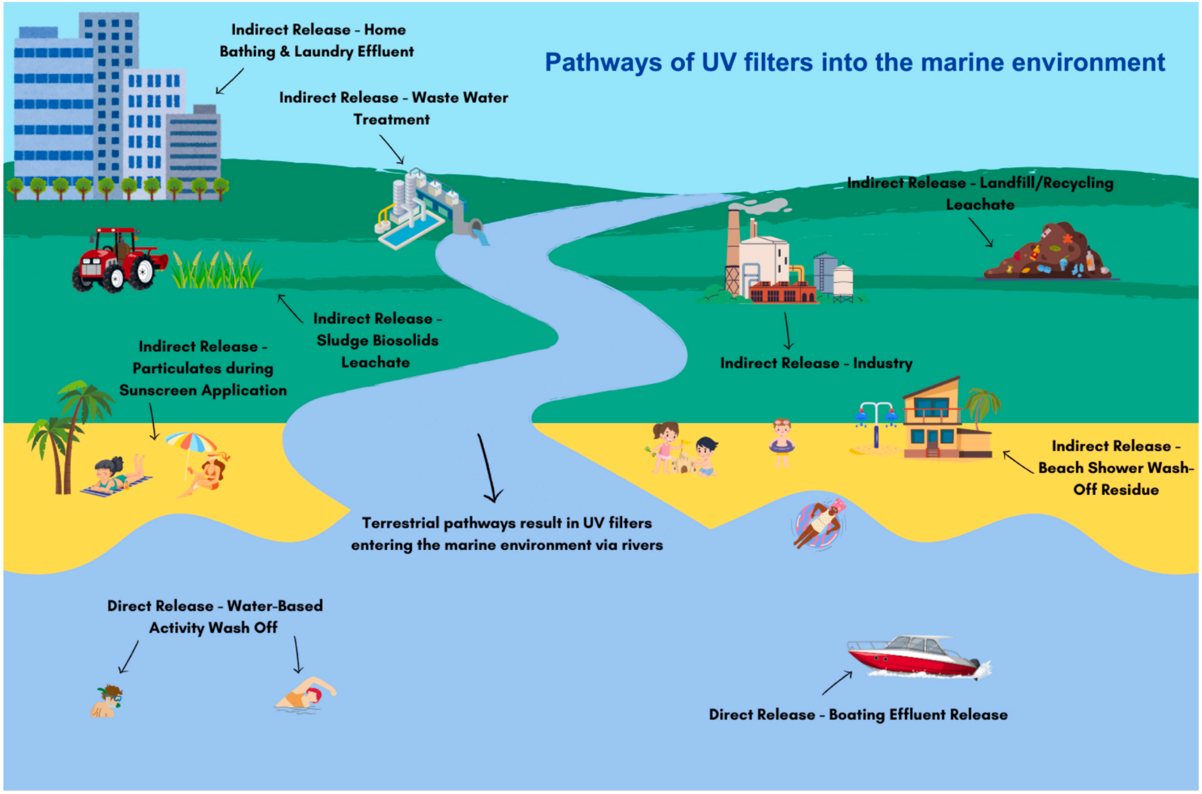

Fig. 2. Summary of the direct and indirect pathways of UV filters in the marine environment via terrestrial and aquatic routes.

substances' hydrophilicity and lipophilicity (Cumming and Rucker, 2017). Organic, lipophilic and substances with a high partition coefficient can easily cross cell membranes and are therefore more likely to alter physiological processes (Emnet et al., 2014). Generally, substances that present a high log K_{ow} value are absorbed more easily and are of great concern as they have the potential to bio-concentrate. Current bioaccumulation and effects research has focused mainly on a limited number of marine organisms of different phylogenetic groups with a main focus on bivalve molluscs (Table 1). A range of biological effects at different levels of biological organisations in phylogenetically different groups of organisms have been observed (Tables 1 and 3).

It is to be realised that UV filters pose not only a threat to the marine environment on their own, but they may also interact with other environmental contaminants and physical stressors (Wijgerde et al., 2020). For example, oxybenzone (BP-3) increased generation of reactive oxygen species and antioxidants in the digestive glands of the yellow clam (*Amarilladesma mactroides*), with lipid peroxidation when thermal stress was applied (Lopes et al., 2022). Other potential synergies include microplastics (O'Donovan et al., 2020) and increasing salinity (Fastelli and Renzi, 2019). For example, microplastics can act as a vector for the accumulation of other pollutants, including UV filters, which when ingested, desorb within the organism (O'Donovan et al., 2020).

This critical and timely review is required to highlight gaps of knowledge in relation to the effects of UV filters on marine organisms. To date, focus has largely been centered on effects to freshwater ecosystems and biota (O'Malley et al., 2021; Amankwah et al., 2024; Ramos et al., 2016; Carrao et al., 2021; Huang et al., 2016; Gago-Ferrero et al., 2015; Ekpeghere et al., 2016; Lucas et al., 2021; Gayathri et al., 2023; Zhao et al., 2023. Little research has been conducted on specifically the ecotoxicological effects of these omnipresent compounds and the long-term effects associated with their dispersal in marine environments. Production of UV filters and their release into the marine environment is expected to increase in the future due to growing coastal urbanisation and tourism (Sánchez-Quiles and Tovar-Sánchez, 2015), thus it is essential to understand the effects of these compounds to marine biota, as well as estimate the differential sensitivities towards them.

Therefore, in this review, we aim to (1) identify and summarise the factors that contribute to UV filter pollution, and their sources and pathways, (2) classify the current knowledge base on the ecotoxicological effects of UV filters on marine organisms at different levels of biological organisations and trophic levels, and (3) highlight the current research gaps and provide recommendations for assessing these critical knowledge areas.

2. Methodology

Between January and October 2024, Scopus, Web of Science, Google Scholar and Lens, were used to search for peer-reviewed journal articles using the keywords associated with UV filters and their marine ecotoxicological effects. Lens compiled articles from Microsoft Academic, PubMed and Crossref with the assistance of OpenAlex. The keywords were UV filter* OR sunscreen*, AND effect* OR ecotox* AND marine*, followed by another analysis with all known sunscreen UV filter names (both common and chemical) certified for use in the EU, UK, or US (Appendix 2) with the keywords AND effect* OR ecotox* AND marine*. The searches were limited to publications written in English. During the screening process, the following papers were excluded: (a) papers not including UV filters associated with sunscreens, (b) papers not pertaining to the topic of the effects of UV filters derived from sunscreens on marine organisms, (c) any duplicates defined as repetition of the same result. All remaining papers were included if they adhered to the following criteria:

(1) The article contained primary research and was not a review-based article.
(2) The article directly assessed the ecotoxicological effects on marine organisms.

Table 1
Studies investigating ecotoxicological effects of organic UV filters on marine organisms.

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
Acartia tonsa	Copepoda	Benzophenone-1 (BP-1)	0.51-2 mg L^{-1}	• Individuals most resistant towards BP-1 at 20 °C (EC50 = 1.1 mg L^{-1}). EC50 values significantly lower at 15 °C (0.49 mg L^{-1}). EC50 at 25 °C (0.77 mg L^{-1}).	Kusk et al. (2011)
Acropora tenuis	Anthozoa	Benzophenone-3 (BP-3)	50 μg L^{-1} & 500 μg L^{-1}	• Several differentially expressed genes were detected after exposure at both concentrations.	Ishibashi et al. (2024)
Adudefduf saxatilis	Actinopterygii	BP-3, Ethylexyl Methoxy Cinnamate (Octinoxate) (EHMC), 2-Ethylhexyl Salicylate (Octisalate) (ES)	5, 25 & 50 μg L^{-1}	• No statistically significant effects were observed.	Soto and Rodríguez-Fuentes (2014)
Arenicola marina	Polychaete	Butylmethoxydibenzoylmethane (Avobenzone) (AVO), nano Zinc Oxide (nZnO – Inorganic).	3 mg nZnO kg^{-1} of dry sediment, 50 μg L^{-1} of AVO	• Increase in tissue antioxidant capacity and oxidative stress.	Bruhns et al. (2024)
Artemia franciscna	Gammaproteobacteria	BP-1, EHMC, AVO, Benzophenone-2 (BP-2), BP-3, 4-Methylbenzylidene Camphor (4-MBC), 2-ethylhexyl 2-cyano-3,3-diphenyl-2-propenoate (Octocrylene) (OC), ENS – 2-phenylbenzimidazole-5-sulfonic acid (Ensulizole) (ENS), 3,3,5-Trimethylcyclo-hexyl salicylate (Homosalate) (HS),	100 mg L^{-1} (BP-1, BP-2), 12.5 mg L^{-1} (BP-3) and 10 mg L^{-1} (OC, EHMC, EHS, HMS, BM).	• EHMC (EC50 = 1.38–2.16 mg L^{-1}) most toxic. 4-MBC least toxic (EC50 = 12.97–15.44 mg L^{1}). BP-1, BP-2, BP-3, EHMC and BM caused 50 % inhibition in luminescence.	Marcin and Aleksander (2023)
Amarilladesma mactroides	Bivalvia	BP-3	1 μg L^{-1}	• BP-3 altered glutathione-S-transferase (GST) and glutathione cysteine ligase (GCL) activity. Increased level of glutathione (GSH). Carbonic anhydrase activity (CA) increased in the digestive gland and decreased in the gills.	Lopes et al. (2020)
Amarilladesma mactroides	Bivalvia	BP-3	1 μg L^{-1}	• Increase in reactive oxygen species (ROS) at 20 °C. No significant effect on lipid peroxidation. 24 °C, antioxidant defence suppressed, lipid peroxidation observed.	Lopes et al. (2022)
Amnia clausi & meroplanktonic larva of the Paracenuatus lividus	Copepoda, Echinoidea	4-MBC	1, 10 mg L^{-1}	• Microplastics (MPs) caused no increase in bioaccumulation or toxicity of 4-MBC.	Beiras et al. (2019)
Amphiprion ocellaris	Actinopterygii	BP-3	1, 10 μg L^{-1}	• BP-3 caused abnormal body weight, alterations to insulin content and immune and digestive related enzymatic activity.	Zhang et al. (2023b)
Amphiprion ocellaris	Actinopterygii	BP-3	1000 ng BP-3/g food	• Survival, growth and social ranking not affected. Body weight of dominant fish higher in BP-3 group.	Chen et al. (2018a)
Aurelia coerulea	Scyphozoa	HS	1 uM	• HS caused significant delay in metamorphosis development.	Chen et al. (2024)
Brachionus plicatilis	Rotifera	EHMC, nano Titanium Dioxide ($nTiO_2$ – Inorganic)	0.2, 0.5, 1 & 2 mg L^{-1} (EHMC), 1 & 10 mg L^{-1} ($nTiO_2$)	• Increased EHMC dose delayed age of first reproduction and reduced total offspring.	Yu et al. (2024)
Chaetoceros neogracilis	Coscinodiscophyceae	BP-1, BP-3, 2-hydroxy-4-methoxyphenyl; Dioxybenzone (BP-8)	0.05–6.4 mg L^{-1} (BP-3), 0.0625–8 mg L^{-1} (BP-8), 0.25 - 32 mg L^{-1} (BP-1)	• Reduced growth, reduced photosynthesis, membrane damage	Yang et al. (2024a)
Chlorella sp.	Chlorellales	EHMC	0.228, 2.28 & 11.4 mg L^{-1}	• Reduction in ribulose-1,5-bisphosphate carboxylase/oxygenase activity in the dark. No effect on photosynthetic electron transport capacity. Excess excitation energy and ROS generation causing decreased growth and pigment bleaching in the light.	Tian et al. (2021)
Chlorella sp. & *Arthrospira* sp.	Chlorellales, Cyanophyceae	BP-3	22.8 ng L^{-1} - 11.4 mg L^{-1}	• Photobleaching, reduction in growth and chlorophyll inhibition observed at >2.28 mg L^{-1}. Inhibition of photosynthetic and respiration electron transport mechanisms	Zhong et al. (2019)

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Table 1 (*continued*)

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
				causing overproduction of ROS at 0.228 mg L^{-1}.	
Crossostrea gigas	Bivalvia	BP-3	1 & 100 μg L^{-1}	• Rapid absorption of BP-3, reaching 34.6 μg BP-3 per g of tissue within 7 days.	Pesenato Magrin et al. (2024)
Effrenium voratum & Cladocopium goreaui	Dinophyceae	BP-3	5, 50, 500, 5000 μg L^{-1}	• Reduced cell division and chlorophyll-a content, increased cell structure damage and changes in bacterial composition between 500 and 5000 μg L^{-1}.	Yang et al. (2023)
Epibacterium mobile	Alphaproteobacteria	BP-3	350 μg L^{-1}	• Alterations to xenobiotic export, detoxification, oxidative stress, motility, fatty acid, iron and amino acid metabolism proteins.	Lozano et al. (2021)
Ficopomatus enigmaticus	Polychaete	EHMC, nZnO	0.01, 0.1 & 0.5 mg L^{-1}	• EHMC caused lipid peroxidation and concentration dependent activation of Glutathione S-Transferase (GST).	Cuccaro et al. (2022c)
Galexea fascicularis	Anthozoa	BP-3	0.31–10 mg L^{-1}	• BP-3 did not cause bleaching. • Polyp retraction sub-lethal behavioural response.	Conway et al. (2021)
Gladioferens pectinatus	Copepoda	Benzophenone (BP)	0.5 mg L^{-1}	• Reduced egg hatching success and viability.	Guyon et al. (2018)
Gracilaria vermiculophylla	Florideophyceae	BP-3	2.28, 22.8, 228 μg L^{-1}	• Significant decrease in growth rate, pigment content and photosynthetic rate observed at 228 μg L^{-1-1}.	Xing et al. (2022)
Heterostegina depressa	Foraminifera	ENS	10, 50, 200 mg L^{-1}	• Significant reduction in areal fluorescence signal.	Lintner et al. (2022)
Isochrysis galbana, Mytilus galloprovincialis, Paracentrotus lividus	Coccolithophyceae, Bivalvia, Echinoidea	Ethylhexyl Dimethyl P-Aminobenzoic Acid (ED-PABA), OC	5–150 μg L^{-1}	• Cell division of *I. galbana* most sensitive endpoint. EC10 between 26.5 and 127 μg L^{-1} for ED-PABA and 103–511 μg L^{-1} for OC.	Giraldo et al. (2017)
Isochrysis galbana, Mytilus galloprovincialis, Paracentrotus lividus, and Siriella armata	Coccolithophyceae, Bivalvia, Echinoidea, Mysida	4-MBC, BP-3, Benzophenone-4 (BP-4), EHMC	< 1–100 mg L^{-1}	• EHMC and 4-MBC most toxic to test species. Most affected species *I. galbana* (toxicity threshold in the range of μg L^{-1}).	Parades et al. (2014)
Leptastrea purpurea, Tubastraea faulkneri, Acropora digitifera & A. millepora	Anthozoa	BP-3	1.3 μg L^{-1} - 5.3 mg L^{-1}	• Most sensitive species = *A. digitifera* (LC_{50} = 0.75 μg L^{-1}).	Miller et al. (2022)
Lithothamnion sp.	Corallinales	BP-3	50 mg L^{-1}	• No evidence of negative effects of on *Lithothamnion* sp.	MacVicar et al. (2022)
Mytilus edulis	Bivalvia	ENS	10 ng L^{-1}, 10^2 ng L^{-1}, 10^3 ng L^{-1}, 10^4 ng L^{-1}, & 10^5 ng L^{-1}	• Antioxidant response, energy storage and cell death-related processes in mussel tissues affected. Low concentrations caused shorter air survival time than the control.	Pham et al. (2022)
Mytilus edulis	Bivalvia	ENS, OC	10–100 μg L^{-1}	• Both UV filters caused sublethal effects. Induction of oxidative stress, genotoxicity, upregulation of apoptosis and inflammation and dysregulation of the xenobiotic biotransformation system.	Falfushynska et al. (2021)
Mytilus galloprovincialis	Bivalvia	4-MBC	100, 300 & 600 μg L^{-1}	• Functional and structural impairments, hyperactivation and DNA damage in sperms. Physiological, metabolic/energetic dysfunctions, DNA damage and activation of oxidative and biotransformation enzymes in adults.	Cuccaro et al. (2022a)
Mytilus galloprovincialis	Bivalvia	4-MBC, BP-3	1–100 μg L^{-1}	• 4-MBC cause oxidative stress, sperm structural impairments, motility and kinetic alterations. BP-3 caused DNA damage,	Cuccaro et al. (2022b)

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Table 1 (*continued*)

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
				compromised mitochondrial activity and hyperactivation.	
Mytilus galloprovincialis	Bivalvia	4-MBC	1.0, 10.0 & 100.0 μg L^{-1}	• Decreased filtration rates, metabolic capacity and increase in deployment of energy reserves, cellular and DNA damage and increase in AChE activity.	Cuccaro et al. (2023)
Mytilus galloprovincialis	Bivalvia	BP-3	10, 100, & 1000 ng L^{-1}	• Significantly higher activity of electron transport system and energy reserves (100 ng L^{-1}). Mussels unable to increase metabolic activity, cellular damage (1000 ng L^{-1})	Bordalo et al. (2020)
Mytilus galloprovincialis	Bivalvia	AVO	0.1, 1.0 & 10.0 μg L^{-1}	• Significant overproduction of superoxide anions and DNA damages in sperms. Decreased sperm viability at 1 and 10 μg L^{-1}. Complete inhibition of motility at 10 μg L^{-1}.	Bordalo et al. (2022)
Mytilus galloprovincialis	Bivalvia	AVO	5 μg L^{-1}	• Genotoxicity and increased respiration rate in sperm. Increased biotransformation enzyme activity in adults. AVO combined with warming caused oxidative stress, cellular damage, genotoxicity, and decreased motility in sperm. Only antioxidant enzyme activity was enhanced in adults.	Bordalo et al. (2023)
Octopus maya	Cephalopoda	BP-3	5, 25, 50, 500 μg L^{-1}	• BP-3 did not induce oxidative stress at this life stage.	Moreno-Ortiz et al. (2023)
Perinereis aibuhitensis	Polychaete	EHMC	2, 20, 200 μg L^{-1}	• Exposure inhibited burrowing behaviour. • Activation of antioxidant response and reduction in lipid peroxidation.	He et al. (2024)
Perna perna	Bivalvia	BP-3	0.1 & 3μg L^{-1}	• Significant reduction in G6PDH and glutathione peroxidase (GPx) activity at 0.1 μg L^{-1}. No significant differences in glutathione S-transferase (GST) and lipoperoxidation (MDA).	Cruz et al. (2023)
Phaeodactylum tricornutum	Bacillariophyceae	BP-4	1–125 mg L^{-1}	• Impairment to the cell structure. • Excessive levels of ROS	Yang et al. (2024b)
Pocillopora damicornis	Anthozoa	AVO	5, 50, 300, & 1000 μg L^{-1}	• Derivatives of AVO absorbed in coral tissue after 7 days.	Clergeaud et al. (2023)
Pocillopora damicornis	Anthozoa	OC	5, 50, 300, & 1000 μg L^{-1}	• OC caused abnormal metabolism related to mitochondrial dysfunction.	Stein et al. (2019)
Pocillopora damicornis	Anthozoa	OC, ES, Methylene Bis-Benzotriazolyl Tetramethylbutylphenol (Bisoctrizole) (MBBT), BP-3, AVO, HS, Diethylaminohydroxybenzoyl Hexyl Benzoate (DHHB), Bis-Ethylhexyloxyphenol Methoxyphenyl triazine (Ethyl Triazine) (BEMT), Diethylhexyl butamido triazone (Iscotrizinol) (DBT), Ethylhexyl triazone (ET).	5, 50, 300, & 1000 μg L^{-1}	• OC caused mitochondrial dysfunction. • ES caused an inflammatory and stress response at 300 μg L^{-1}	Stein et al. (2020)
Pocillopora damicornis Seriatopora caliendrum,	Anthozoa	EHMC, OC	0.1 to 1000 μg L^{-1}	• Death of corals at 1000 μg L^{-1-1}. Total polyp retraction in both species at 10–1000 μg L^{-1} under both UV filters.	He et al. (2019a)
Pocillopora damicornis, Seriatopora caliendrum	Anthozoa	BP-1, BP-3, BP-4, BP-8	0.1–1000 μg L^{-1}	• BP-1 and BP-8 caused significant settlement failure, bleaching and mortality of *S. caliendrum* larvae. No compounds affected *P. damicornis* larvae. Nubbins more sensitive to BP-3, BP-1 and BP-8 than larvae.	He et al. (2019b)
Pocillopora acuta Seriatopora caliendrum, Montipora aequituberculata	Anthozoa	Isoamyl-p-methoxicinnamate (Amiloxate) (IMC), 4-MBC, BP-1, BP-3, 5-Benzoyl-4-hydroxy-2-methoxybenzenesulfonic acid (Sulisobenzone) (BP-4), BP-8, EHMC, AVO, ENS, HS, OC, ED-PABA	200 ng L^{-1}	• UV filter mixture caused bleaching to 37.5 % and mortality in 12.5 % of *S. caliendrum.* Under elevated temperatures, 100 %	He et al. (2023)

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Table 1 (*continued*)

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
				S. caliendrum and *P. acuta* bleached causing 100 % and 50 % mortality, respectively. Significant increase of catalase activities and oxidative stress in *P. acuta* and *M. aequituberculata* nubbins.	
Posidonia oceanica	Monocotyledoneae	BP-3	53.6–1115 ng L^{-1}	• Increase in oxidative stress, reduced chlorophyll-a concentration and restriction of nitrogen fixation.	García-Márquez et al. (2024)
Ophryotrocha diadema	Polychaete	BP-3	0.025, 0.050, 0.100, & 0.200 mg L^{-1}	• Negative effects on body growth and reduction of egg production between 0.1 and 0.2 mg L^{-1}.	Santovito et al. (2023)
Ruditapes philippinarum	Bivalvia	BP-3, 4-MBC, EHMC	1, 10 & 100 μg L^{-1},	• Induction of immune responses, immune system restrained after prolonged exposure.	Dong et al. (2022)
Ruditapes philippinarum	Bivalvia	4-MBC	10 & 100 μg L^{-1},	• Significant number of differentially expressed genes. Energy depletion, oxidative stress and carbohydrate and lipid metabolism affected.	Colás-Ruiz et al. (2024)
Ruditapes philippinarum	Bivalvia	4-MBC	1, 10 & 100 μg L^{-1}	• Accumulation in the clams. Mortality and concentration positively correlated. Antioxidant and cellular stress enzymes increase with dose.	Santonocito et al. (2020)
Scrobicularia plana	Bivalvia	BP-3	20 μg L^{-1}	• MPs with BP-3 caused oxidative stress, neurotoxic and genotoxic effects. Gills were more affected than the digestive gland.	O'Donovan et al. (2020)
Solea senegalensis	Actinopterygii	4-MBC	0.025–0.935 mg L^{-1}	• Mortality, malformations, growth and swimming times affected in dose-response manner. Did not induce oxidative stress.	Araújo et al. (2018)
Solea senegalensis	Actinopterygii	4-MBC	0.2–2.0 mg L^{-1}	• Acceleration of metamorphosis progression after 2 days exposure at all concentrations. Reduced length, inhibition of catalase activity and induction of oxidative stress.	Araújo et al. (2021)
Sparus aurata	Actinopterygii	BP-4	10 μg L^{-1}	• Differentially expressed genes in the liver. Energy metabolism pathways of amino acids, carbohydrates, and lipids affected and enzymes involved in steroid and thyroid hormone biosynthesis and DNA and RNA synthesis.	Colás-Ruiz et al. (2022)
Sparus aurata	Actinopterygii	BP-3	50 μg L^{-1}	• No mortality or physiological alterations. Liver and plasma metabolome perturbed and pathways of lipid metabolism significant altered.	Ziarrusta et al. (2018a)
Sparus aurata	Actinopterygii	BP-3	0.83 mg L^{-1}	• BP-3 and by-products observed in all tissue/biofluids analysed. Highest concentrations found in bile (1.8–17 μg mL^{-1}).	Ziarrusta et al. (2018b)
Stylophora pistillata	Anthozoa	OC	300 & 1000 μg L^{-1}	• Accumulation of OCT triggers increase in concentration of metabolites and leads to mitochondrial dysfunction.	Thorel et al., 2022
Stylophora pistillata	Anthozoa	BP-2	0.00001-1 mM	• Planulae transformed from motile state to deformed sessile state. Dose response to bleaching observed. BP-2 proven to be a genotoxicant, causing DNA AP lesions, necrosis, and autophagic cell death.	Downs et al. (2014)
Stylophora pistillata	Anthozoa	BP-3	0.00001–1 mM	• Planulae transformed from motile to deformed, sessile state. Dose response to bleaching observed. BP-3 proven to be a genotoxicant,	Downs et al. (2016)

(*continued on next page*)

Table 1 (*continued*)

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
				and skeletal endocrine disruptor, causing DNA-AP lesions.	
Stylophora pistillata & *Acropora tenuis*	Anthozoa	BP-3	1 μg L^{-1}	• BP-3 affected function of photosystem II of coral photosymbionts and altered microbiome.	Wijgerde et al. (2020)
Scylla parmamassain	Malacostraca	BP, BP-1, BP-2, BP-3	50, 500 & 1000 μg L^{-1}	• Antioxidant system damage observed. • BP-3 and BP-1 have greater effects on the organism, with severe damage to organelles and ribosomes.	Zhou et al. (2023)
Symbiodiniaceae Cladocopium goreaui	Dinophyceae	BP-3	2–20 mg L^{-1}	• Increased growth at 2 mg L^{-1} and rapid death at 20 mg L^{-1}. Cells observed to uptake BP-3 showed elevated photosynthetic efficiency and decreased cellular carbon and nitrogen content.	Zhang et al. (2023a)
Tetraselmis sp.	Chlorodendrophyceae	BEMT, MBBT, DHHB, Diethylhexyl Butamido Triazone (Iscotrizinol) (DBT), Ethylhexyl Triazone (ET), BP-3, AVO, ENS, OC	10 μg L^{-1} & 1 mg L^{-1}	• Growth declined significantly when exposed to HS, BP-3 and ES. Esterase activity declined in algae exposed to ES and HS at 10 μg L^{-1-1}. Effect of BP3, DHHB and OC was significant at 100 μg L^{-1} and above.	Thorel et al. (2020)
Tetraselmis suecica	Chlorodendrophyceae	BP-3	0.25–2 mg L^{-1}	• Significant increase in growth rate, metabolic activity and chlorophyll-a fluorescence.	Seoane et al. (2017)
Tigriopus japonicus	Copepoda	4-MBC	0.5–10 μg L^{-1}	• Mortality of *T. japonicus* in the F0 generation. Stimulated developmental rate, and ROS responsive genes and reduced reproduction.	Chen et al. (2018b)
Tigriopus japonicus	Copepoda	4-MBC	0 and 5 μg L^{-1}	• Increasing salinity increased toxicity, uptake and bioconcentration of 4-MBC and increased oxidative stress.	Hong et al. (2021)
Vibrio sp., (Please see Appendix 4 for the full list of species).	Gammaproteobacteria	BP-3, OC, EHMC, 4-MBC, HS	100–4000 μg L^{-1}	• OC most toxic UV filter. 7 out of 27 species showed sensitivity to one or more UV filters at 1000 μg L^{-1}.	Lozano et al. (2020)

(3) The research was conducted on either organic or inorganic UV filters found in sunscreens.

Of the 1507 search results, 843 were unrelated or did not meet the criteria, 553 were duplicates and 111 were included in this review (67 on organic UV filters, 44 on inorganic UV filters).

The search retrieval was followed by a critical appraisal step to validate the trustworthiness of each article. A JBI critical appraisal tool was adapted to suit the requirements of this review. Each article was examined to ensure it met the appraisal criteria. All 111 articles were included after examination, no articles were excluded.

3. Studies with organic UV filters

Of the 67 search results which focus specifically on the ecotoxicological effects of organic UV filters on marine biota (Table 1), most studies focus on the effects of the compounds BP-3 (57 %) and 4-MBC (22 %) (Fig. 3), two widely used UV filters with differing properties. Articles published on organic UV filters were largely conducted on organisms from Southern Europe, China and the USA, with no publications based on organisms retrieved from Africa, South Asia including India, the Pacific Islands or the Arctic and Antarctic regions (Fig. 6A). This lack of geographical organism representation limits our understanding of the differential sensitivities among marine organisms from different latitudes. Publications were largely produced by individuals associated with institutions in China, Spain and Portugal, with little or no publications from South, West and Eastern Asia, Africa and South and Central America (Fig. 7A).

There has been a steady increase in the number of published articles focusing on organic UV filters, which peaked in 2022 after the COVID-19 pandemic (Fig. 8A). The keywords used by the authors of the articles are shown in Appendix 3 through a network visualization model. This model demonstrates the common usage of 'oxidative stress', 'oxybenzone', 'UV filters', 'personal care products', 'sunscreen', and 'toxicity' across the articles examined.

Studies largely investigated the effect of these compounds on the classes bivalvia (27 %) and anthozoa (27 %), with few studies on phytoplankton, copepods and mysids (Fig. 5), which are common ecotoxicological reference organisms. Bivalves are often used in ecotoxicology, particularly in studies assessing effects of nano, or micro compounds such as UV filters, plastics and metals. Due to their filter feeding nature, bivalves can accumulate toxicants in their tissues, making them particularly susceptible to aquatic pollution. A summary of the parameters used in the studies collated is shown in Table 2. However, limiting organism analysis to mainly bivalves and anthozoa restricts our understanding of the true potential of UV filters to cause effects to different phylogenetic classes and trophic levels, which is essential to develop environmental risk assessments (ERA's).

Studies largely used individuals that were collected from the field (49 %) (e.g. Lopes et al., 2022) and DMSO was the most widely used solvent (48 %) (Table 2), which is consistent with ecotoxicological studies on other pollutants (Barmshuri et al., 2023; Barranger et al., 2019), due to its strong capability of acting as a compound carrier (Okumura et al., 2001), low level of toxicity (Di Mauro et al., 2023) and

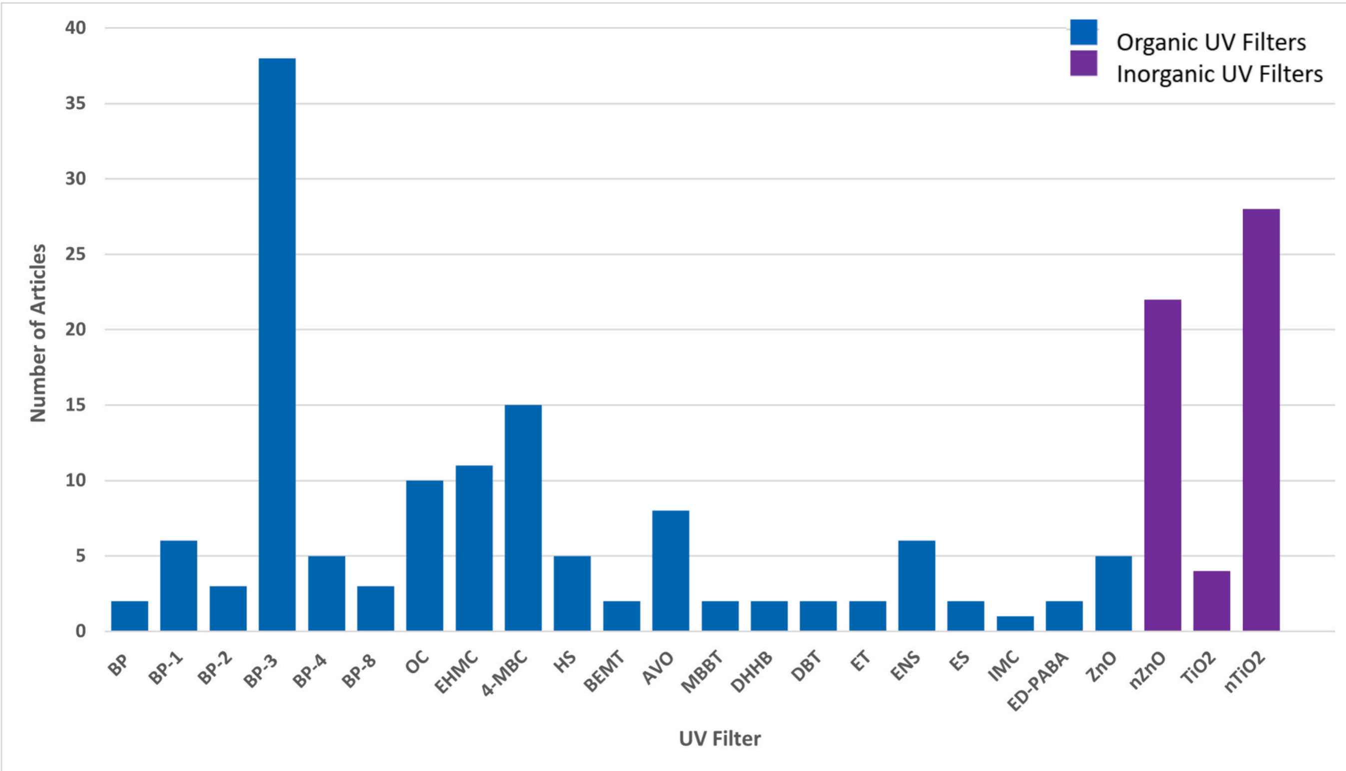


Fig. 3. Distribution of the article results in relation to the UV filters assessed. Abbreviations for each compound are as follows: $nTiO_2$: Nano Titanium Dioxide; TiO_2: Titanium Dioxide; nZnO: Nano Zinc Oxide; ZnO: Zinc Oxide; ED-PABA: Ethylhexyl Dimethyl P-Aminobenzoic Acid; IMC: Isoamyl-p-methoxicinnamate (Amiloxate); ES: ethylhexyl salicylate; ENS: 2-phenylbenzimidazole-5-sulfonic acid (Ensulizole); ET: Ethylhexyl Triazone; DBT: Diethylhexyl Butamido Triazone (Iscotrizinol); DHHB: Diethylaminohydroxybenzoyl Hexyl Benzoate; ES: 2-Ethylhexyl Salicylate (Octisalate); MBBT: Methylene Bis-Benzotriazolyl Tetramethylbutylphenol (Bisoctrizole); AVO: Butyl Methoxydibenzoylmethane (Avobenzone); BEMT: Bis-Ethylhexyloxyphenol Methoxyphenyl triazine (Ethyl Triazine); HS: 3,3,5-Trimethyl-cyclohexyl salicylate (Homosalate); 4-MBC: 4-Methylbenzylidene Camphor; EHMC: Ethylexyl Methoxy Cinnamate (Octinoxate); OC: 2-ethylhexyl 2-cyano-3,3-diphenyl-2-propenoate (Octocrylene); BP-8: 2-hydroxy-4-methoxyphenyl) (Dioxybenzone); BP-4: 5-Benzoyl-4-hydroxy-2-methoxybenzenesulfonic acid (Sulisobenzone); BP-3: 2-hydroxy-4-methoxybenzophenone (Benzophenone-3, Oxybenzone); BP-2: 2,2′,4,4'-Tetrahydroxybenzophenone (Benzophenone-2); BP-1: 2,4-Dihydroxybenzophenone (Benzophenone-1); BP: Benzophenone. NB: When a study investigated more than one UV filter, each UV filter has been included in the number of articles.

widespread use in OECD and ISO guideline-based experiments. However, some studies (22 %), used methanol as their chosen solvent due to the confounding impacts associated with DMSO use (e.g. antioxidant properties and enhanced uptake of chemicals [Mitchelmore et al., 2021], particularly in coral based studies that performed antioxidant response assays (Conway et al., 2021). Most studies only reported the use of a solvent carrier control, with few reports of a solvent validation assessment to elucidate the effect of a series of solvents on the test organism(s). This would be beneficial to ensure the chosen solvent was the least harmful towards the test organism, and did not enhance the toxicity of the compound.

Concentrations of UV filters to which test organisms were exposed under laboratory conditions were generally stated by the authors as environmentally relevant, with some higher concentrations used to determine EC_{50} (half-maximal effective concentration) or LC_{50} (half-maximal lethal concentration) values. Nominal concentrations ranged from being environmentally relevant (ng L^{-1}), to worst case scenario (mg L^{-1}) values, both of which are important to apply to ecotoxicological experiments due to the possibility of bioaccumulation and biomagnification, as well as to understand mechanisms within sensitive organisms (Carve et al., 2021). Most papers stated use of a manual aqueous exposure method, and opted for a concentration (dose) – response approach (Table 2), which is often common in ecotoxicology to enable the generation of ERA's and to elucidate the full degree of the organisms' response to the contaminant. Sample sizes were generally large, with at least 3 replicates per treatment, and the exposure period was more often chronic (>96 h) [70 %], than acute (<96 h) [33 %]. There were many studies (49 %) that did not assess the concentration of the UV filters within the water column of the exposure tanks. Therefore, only the concentration within the organism was assessed in most papers, meaning the bioavailability of the substance at any given time after initial exposure was not recorded. Assessing the concentration of the compound in the exposure medium is essential to help predict uptake rates and bioavailability, which can aid in developing an understanding of how the substance interacts with the organism and its surroundings. 52 % of studies focused on organisms that are found in only temperate waters (defined as organisms inhabiting oceans found north of the Tropic of Cancer or south of the Tropic of Capricorn) and 30 % investigated effects on organisms inhabiting only tropical waters (defined as from oceans positioned within the Tropic of Cancer and the Tropic of Capricorn). 18 % of papers involved organisms found in both temperate and tropical waters, for example, migratory species.

12 % of papers used UV or solar simulation lighting during experiments, and of those that did not, 1.5 % irradiated the exposure solutions to undergo similar conditions to the natural environment. Lack of UV or solar simulation lighting during experiments eliminates the photocatalytic properties of the UV filters, in turn voiding the environmental relevance of the study. Mortality, along with oxidative damage were the most common biological assays performed, with 40 % of studies performing either lipid peroxidation, protein carbonylation and hydrogen peroxide presence assays. Other popular biological assays included enzymatic activity and antioxidant response (Table 2).

Table 2
Summary of the studied parameters collated from included literature. On occasions where a paper used more than one type of parameter (i.e. obtained organisms from both the field and aquaculture), both occurrences are included. Temperate waters defined as organisms inhabiting Oceans found north of the Tropic of Cancer, or south of the Tropic of Capricorn. Tropical waters defined as positioned within the Tropic of Cancer and the Tropic of Capricorn. NB – Some studies did not list all parameters, so have not been included in the below percentages.

Parameter	Type	% of organic papers	% of inorganic papers
Organism Collection/Origin	Field	49	52
	Aquaculture	31	36
	Pure Stock (bred-onsite)	12	2
Solvent	DMSO	48	2
	Methanol	22	0
	Ethanol	13	0
	Ethyl alcohol	3	0
	Acetone	1.5	0
	No Solvent	7	98
Dose	Concentration Dose Response	84	64
	Single Concentration	37	36
Dosing System	Flow-through	15	7
	Manual	87	93
Exposure Route	Aqueous	96	82
	Dietary	3	9
	Sediment	3	2
	Intraperitoneal Injection	1.5	7
Biological Assays Performed	Oxidative Damage	40	39
	Enzymatic Activity	21	17
	Mortality	34	7
	Antioxidant Activity	30	7
	Photosynthetic Parameter	30	18
	Compound Accumulation	18	27
Tropical/ Temperate	Tropical	30	18
	Temperate	52	57
	Tropical & Temperate	18	25

4. Properties of organic UV filters

Organic UV filters can be categorised by structural groups: aminobenzoates, benzophenones, cinnamates, salicylates, triazine derivatives and camphor derivatives (Figs. 1 and 4) (Nunes et al., 2018; Baillo and Lima, 2012; Cantrell et al., 2001; Cahova et al., 2021; Ramos et al., 2015; Santos et al., 2012). Some chemical UV filters, such as octocrylene, are formed via the condensation of both salicylates and benzophenones (Downs et al., 2021a). Organic UV filters mainly absorb UV-B radiation. Some, such as those in the benzophenone family, can also partly absorb UV-A (Gao et al., 2013). Solubility of organic UV filters can range from low (bioxybenzone 0.013 mg L^{-1}) to high (sulisobenzone 300,000 mg L^{-1}), volatility is generally low at ambient temperature and log K_{ow} is generally high (National Academies of Sciences, 2022). Organic UV filters concentrate on the surface-microlayer of the ocean as a slick, as well as accumulate in sediments and biota due to their high lipophilicity (Log K_{ow} 4–8) [Caloni et al., 2021; Gao et al., 2013]. Due to their photocatalytic capacity, organic UV filters are contributors to the overproduction of reactive oxygen species (ROS), such as hydrogen peroxide. In photosynthetic organisms, the production of ROS caused by the presence of organic UV filters leads to the oxidation of macromolecules, and subsequently results in damage to the structural and functional integrity of the chloroplast (Li and Kim, 2022). In seawater, sunscreen forms stable colloidal residues, including macroscopic aggregates, agglomerates and submicronic fractions (Botta et al., 2011) These compounds can be released into the aqueous phase as dissolved chemicals via various physical-chemical processes (Rodriguez-Romero et al., 2019). Several studies have highlighted the ability of UV filters to degrade via photolysis (Diaz-Cruz et al., 2008a; Rodil et al., 2009; Richardson, 2010), although little is known about the toxicity and environmental fate of associated break-down products (Parades et al., 2014).

4.1. Benzophenones

Benzophenones (BPs) are the most common and extensively used UV filters in sunscreens, with a total of fourteen BP derivatives being used in commercial personal care products (Downs et al., 2021b; Li and Kannan, 2022). BPs have two benzene rings, joined with a carbonyl group and all contain aromatic methoxy moieties on a benzophenone backbone (National Academies of Sciences, 2022). Oxybenzone (CAS number 131-57-7), commonly called BP-3 is the most used compound thanks to its low cost, and excellent absorption properties (Kim and Choi, 2014; Ma et al., 2022; Kinnberg et al., 2015). BPs possess aromatic structures that absorb and stabilise solar UV radiation, meaning many BPs are photostable, as well as lipophilic (Gasparro et al., 1998; Kim and Choi, 2014). BP-3 has a relatively high partition coefficient (log-K_{ow}) of 4.0, which means it has a slow rate of degradation, a tendency to sorb to suspended solids and sediments and has a low volatilisation potential from water surfaces (Kim and Choi, 2014). Benzophenones have been classified as persistent, bio-accumulative and toxic (PBT) substances, and benzophenone-3 is currently on the European Chemical Agency's watch-list as 'under investigation as endocrine disrupting' (Rascon et al., 2023; European Chemicals Agency, 2023a).

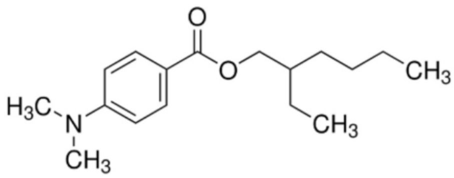
4.2. Cinnamates

Cinnamates are heavily used in sunscreens due to their high molar absorption coefficients and their photostability (Cantrell et al., 2001). 2-ethylhexyl-2-cyano-3,3-diphenyl-2-propanoate, also known as octocrylene (CAS number 6197-30-4) is a hydrophobic cinnamate ester substance that is highly effective at absorbing both UVA and UVB radiation (Pawlowski et al., 2019; Shaath, 2010). Octocrylene is liquid at ambient temperature, highly lipophilic and has a high absorption potential to organic material (Pawlowski et al., 2019). Octocrylene has a high log octanol-water partition coefficient at 6.9 (He et al., 2019a). Since the banning of BP-3 in certain regions (e.g. Hawaii, Key West Florida, Bonaire, Aruba, Palau and the U.S. Virgin Islands) (Zhong et al., 2020; Suh et al., 2020), sunscreen manufacturers have begun adapting formulations to use octocrylene as an alternative (Downs et al., 2021b). However, octocrylene is contaminated with benzophenone and it 'cannot be removed entirely when processed' (Rodan and Fields, 2016; Superior Court of California, 2015). Downs et al. (2021a) discovered that octocrylene undergoes a slow retro-aldol condensation reaction which produces benzophenone. All octocrylene containing sunscreen products tested in their experiment showed a concomitant increase in benzophenone creation upon ageing (Downs et al., 2021a).

Ethylhexyl methoxycinnamate (EHMC), often called octinoxate (CAS number: 5466-77-3), is another widely used organic UV filter. It is a derivate of cinnamic acid and is lipophilic with low water solubility and high octanol-water partition coefficient (5.8) (Cahova et al., 2021). EHMC absorbs UVB radiation only and therefore is often added to products alongside other UV filter compounds (Carve et al., 2021). EHMC is listed on the 'watch list' of contaminants of emerging concern (CECs) as published by the European Union (2018) and is also being investigated by the European Chemical Agency as a potential endocrine disruptor (European Chemicals Agency, 2023b). Additionally, it is registered under the High Production Volume Chemicals (HPVC) list, suggesting the chemical is produced, or imported into the European Union (EU), at a rate of >1000 t per year (Vione et al., 2015).

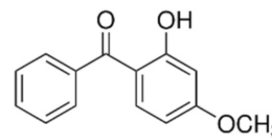
4.3. Aminobenzoates

Aminobenzoates are aromatic amino acids that occur as components



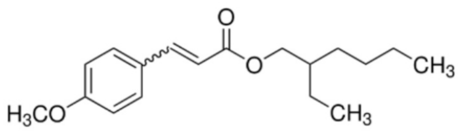
Chemical Group: Aminobenzoates

Example: 2-Ethylhexyl 4-(dimethylamino)benzoate



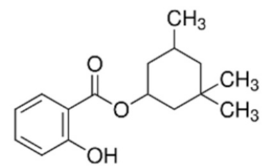
Chemical Group: Benzophenones

Example: Benzophenone-3



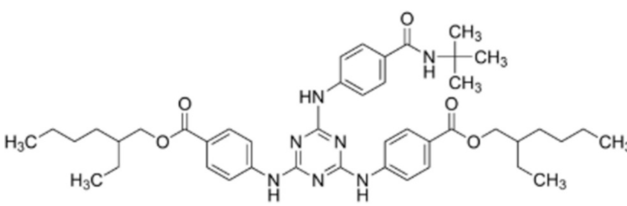
Chemical Group: Cinnamates

Example: Ethylhexyl Methoxy Cinnamate



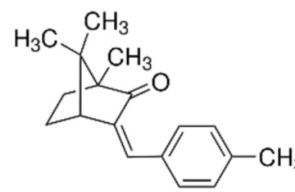
Chemical Group: Salicylates

Example: 3,3,5-Trimethylcyclohexyl salicylate



Chemical Group: Triazine Derivatives

Example: Diethylhexyl Butamido Triazone



Chemical Group: Camphor Derivatives

Example: 4-methylbenzylidene camphor

Fig. 4. Examples of chemical structures and groups of organic UV filters.

of microbial metabolism, formed via nitration of toluene and hydrogenation (Walsh et al., 2011; Gadgil et al., 2023). These stable, white powders, include p-aminobenzoate (PABA) [CAS number: 150-13-0] and its derivatives, such as Padimate O PABA (CAS number: 21245-02-3). Once widely used in sunscreens, their use has declined due to concerns about skin sensitivity, with derivatives now predominating (Gabros et al., 2023; Gadgil et al., 2023). PABA and its derivatives produce photoproducts derived from free radicals, as well as producing singlet oxygens, and binding to DNA and free bases in aqueous solution, resulting in formation of DNA photoproducts (Shaw et al., 1992; Chignell et al., 1980; Cantrell et al., 2001). Aminobenzoates absorb only UVB radiation, with peak absorption at 283 nm (Guan et al., 2021). Due to sensitivity concerns, cinnamates have largely replaced aminobenzoates in sunscreen formulations (Gabros et al., 2023).

4.4. Triazine derivatives

Triazine derivatives such as ethylhezyl triazone (EHT) [CAS number: 88122-99-0] and anisotriazine (BEMT) [CAS number: 187393–00-6] contain a heterocyclic 1,3,5-triazine group with three aromatic amine groups separated with carbon atoms. BEMT is the first new sunscreen active UV filter to be evaluated by the FDA for inclusion in formulations of over-the-counter sunscreen products in the USA (D'Ruiz et al., 2023; Food and Drug Administration (FDA), 2021). Generally, triazine derivatives are highly photostable, have a high molecular weight, and are oil soluble (D'Ruiz et al., 2023).

4.5. Camphor derivatives

Camphor is a transparent bicyclic monoterpenoid solid that is optically active (Anjaneyulul et al., 2021). The most common camphor-based UV filter is 3-(4-methylbenzylidene) camphor (CAS number: 36861-47-9), also known as 4-MBC which easily undergoes photoisomerisation (Jesus et al., 2022). It absorbs most efficiently at 300 nm, making it suitable for protection against UVB radiation (Shaath, 2010). Despite 4-MBC being one of the most extensively researched UV filters on marine biota (Fig. 3), occurrence is declining in UK sunscreen products (Scientific Committee on Consumer Safety, 2022), largely due to safety concerns (European Commission, 2022). A survey conducted on 337 sunscreen products found that 4-MBC was used in 25 % of sunscreen products in 2005, but only 1.2 % of products in 2010 (Kerr, 2011).

4.6. Salicylates

Salicylates are very weak UVB absorbers and thus are used in high concentrations in sunscreen formulations, or alongside other UV filter combinations (Gabros et al., 2023). They are salts or esters derived from salicylic acid (Holt et al., 2020). Homosalate (CAS number: 118-56-9) and octisalate (CAS number: 118-60-5) are examples of salicylate-based UV filters, and they are included as ingredients to prevent the photodegradation of other compounds (Gabros et al., 2023). Some salicylates such as butyloctyl salicylate (CAS number: 190085–41-7) and tridecyl salicylate (CAS number: 19666–16-1), which are non-FDA approved UV filters, are used by manufacturers as a way of 'doping' sunscreen formulations, added under the term of 'stabiliser' and not 'UV filter' (Moradi Tuchayi et al., 2023). These 'doping' components are added to many inorganic based sunscreens to reduce the white cast appearance that is prevalent in mineral-based formulations (Moradi Tuchayi et al., 2023). Butyloctyl salicylate is approved for use in personal care products in the European Union (EU) (European Chemicals Agency, 2024).

5. Transformation products

Organic UV filters can undergo complete transformation via biotic and abiotic processes. These processes can include photolysis and hydrolysis, as well as biodegradation and metabolic transformation (National Academies of Sciences, 2022). Both direct and indirect photodegradation can take place. Direct photolysis occurs when the UV filter is transformed during light absorption, whilst indirect photolysis occurs when they react with ROS (National Academies of Sciences, 2022; Diaz-Cruz et al., 2008a; Rodil et al., 2009; Richardson, 2010). Indirect photodegradation requires the presence of photosensitisers such as nitrate or nitrite, which can react with the sunlight and generate free radicals (al Housari et al., 2010). Photoisomerisation can also take place, whereby the three-dimensional structure of the UV filter is changed (e.g. octinoxate) (MacManus Spencer et al., 2011). Little is known about the effects and environmental fate of UV filter breakdown products; however, some studies have begun to highlight the potential risk factors towards marine biota (Parades et al., 2014). For example, a study conducted by Ziarrusta et al. (2018b) found >20 degradation products of BP-3 in gilt-head bream. Additionally, organic UV filters have been observed to produce chlorinated and brominated byproducts that are known to be more toxic than the original compound (Diaz-Cruz et al., 2008b; Manasfi et al., 2017). Transformation products of these compounds have been observed to take place in vivo, including in freshwater algae (Lee et al., 2020), rats (Suzuki et al., 2005) and bacteria (Zhao et al., 2013), but with little information on biotransformation in marine organisms.

6. Ecotoxicological effects of organic UV filters

Although the effect of UV filters on marine biota is limited, a number of studies report ecotoxicological effects at molecular, cellular, individual and assemblage levels of organisation (Table 1), primarily on marine bivalves and corals (Fig. 5).

6.1. Cellular & molecular level effects

Molecular effects within marine biota can include DNA damage, endocrine disruption, differentially expressed genes, cytotoxicity and neurotoxicity. For example, in fish, BP-3 can modulate the oestrogen receptor signalling pathways, thus reducing reproductive potential (Bluthgen et al., 2012; Cosnefroy et al., 2011). BP-3 can also result in fish gender shifts, where-by male species develop female traits and females display reduced egg production (Kunz and Fent, 2009). Genotoxic effects have also been observed in the anthozoa *S. pistillata* (Table 1), where BP-2 increased DNA-apurinic/apyrimidinic (AP) site lesions as a function of concentrations (Downs et al., 2014). Both ensulizole and octocrylene induce genotoxicity (via the upregulation of DNA sensing and repair markers) and promote the upregulation of apoptosis and induce inflammation and dysregulation of the xenobiotic biotransformation system in *Mytilus edulis* (Falfushynska et al., 2021). Furthermore, exposure to just 10 μg L^{-1} of avobenzone resulted in superoxide anions and DNA damage to the bivalve *M. galloprovincialis* (Bordalo et al., 2023). Zhang et al., (2023b) conducted a comprehensive study on the effects of BP-3 on one-month old clown anemonefish *Amphiprion ocellaris.* BP-3 interfered with the obesity and feeding expression-related genes, whilst RNA sequencing revealed that BP-3 influences pathways specifically related to growth, learning behaviour, Mitogen-Activated Protein Kinase (MAPK) signalling pathways and insulin excretion (Kyoto Encyclopaedia of Genes and Genome, KEGG), (Zhang et al., 2023b). These studies highlight the extensive and diverse molecular level effects UV filters can have on marine organisms.

Cellular level responses can include effects on growth, enzymatic antioxidants, reactive oxygen species production (ROS) and its subsequent associations, as well as lipid peroxidation and protein

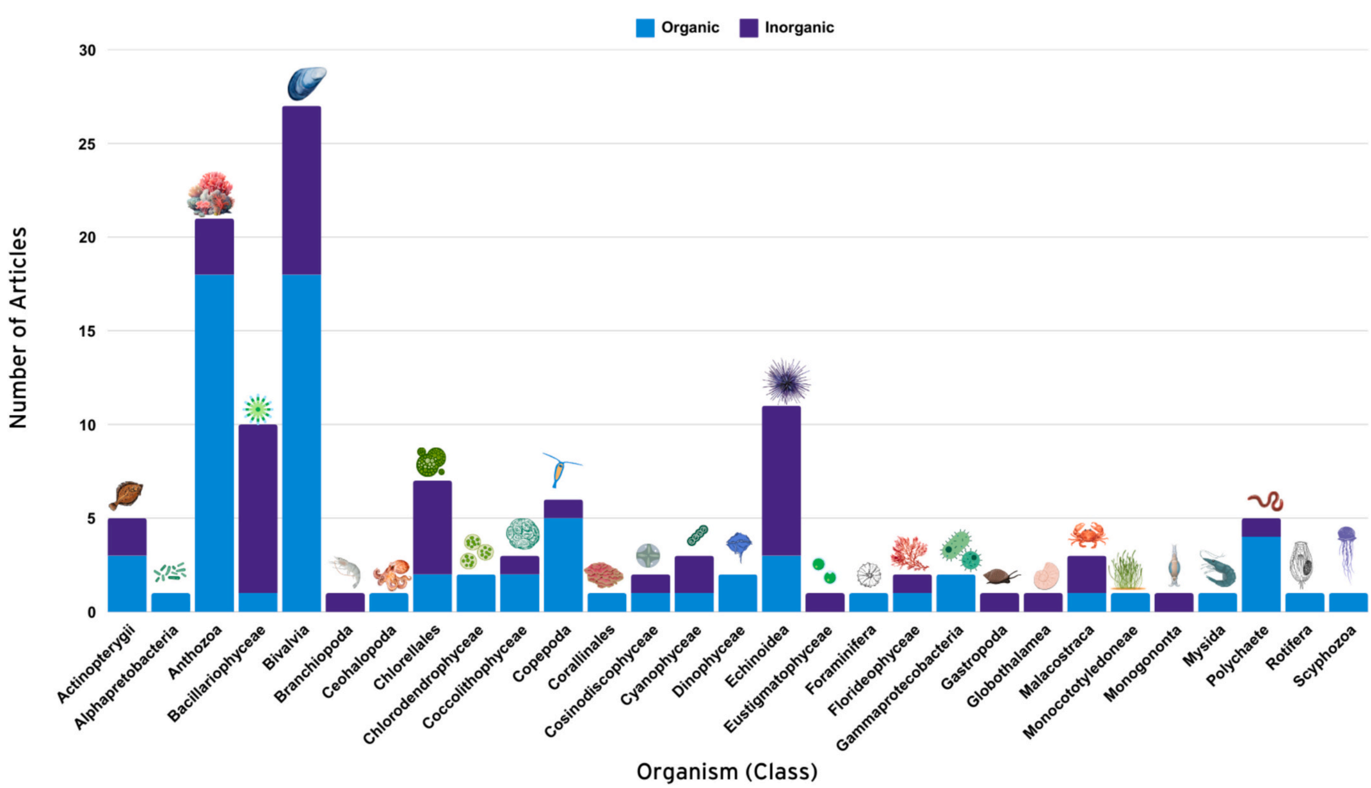


Fig. 5. Distribution of the classes of organisms used in the assessed literature. NB: When a paper investigated more than one class, each class has been included in the number of articles.

carbonylation. Many organic UV filters produce ROS (Tian et al., 2021). ROS have been associated with damaging both the structural and functional integrity of chloroplasts within marine algae (Li and Kim, 2022), as well as inducing oxidative stress which can impact lipid and protein metabolism in the Manila clam *Ruditapes philippinarum* (Colás-Ruiz et al., 2022, 2024) (Table 1). Additionally, BP-3 has been linked to photosynthesis inhibition and mitochondrial oxidative phosphorylation due to the compound acting as a multi-functional herbicide (Mao et al., 2017). Furthermore, BP-3 has potential to induce photo-oxidative stress to molecular structures within the thylakoid membrane of chloroplasts in coral planulae (Downs et al., 2016). Other UV filters including octisalate, ensulizole and homosalate led to increased lipid peroxidation in loggerhead sea turtles (Cocci et al., 2022), whilst 4-MBC has been observed to cause the activation of oxidative and biotransformative enzymes in the Mediterranean mussel (Cuccaro et al., 2022a). Some studies have begun to investigate the combined effect of other marine stressors (such as warming scenarios or acidification) with UV filter contamination.

6.2. Individual and assemblage level effects

Several studies have focussed on the effects of UV filters on early-life stages of organisms (Cuccaro et al., 2022a, 2022b; Bordalo et al., 2022; Chen et al., 2018b), with most reporting effects at an individual level. For example, exposure of the copepod *Tigriopus japonicus* to 4-MBC resulted in significant mortality in the F0 generation at 5 and 10 μg L^{-1}, but had no impact on survival rate in the F1-F3 generations (Chen et al., 2018b) (Table 1). However, the hatching rate and developmental duration between nauplii to the copepodite decreased significantly in F1-F3 generations, even at the lowest exposure of 0.5 μg L^{-1}. The exposure of the sperms of marine bivalve *Mytilus galloprovincialis* to 4-MBC resulted in functional and structural impairments (Cuccaro et al., 2022a). In the copepod *G. pectinatus,* 0.5 mg L^{-1} of BP resulted in decreased egg hatching success and egg viability, with implications for reproductive success (Guyon et al., 2018). In jellyfish, exposure to BP-3 has been found to negatively affect the swimming patterns of *C. xamachana* and *C. frondosa,* as well as reduce metamorphosis and increase mortality rate (Fitt and Hoffmann, 2020). In adult Senegalenis sole, 4-MBC induced mortality and resulted in a lower swimming time in a dose-response manner (Araújo et al., 2018). Assemblage level effects are associated with an entire community or population of organisms. For example, coral bleaching due to exposure of UV filters is becoming increasingly more documented in scientific literature (Tsui et al., 2017; Downs et al., 2014, 2016, 2021b; Danovaro et al., 2008). Reports from laboratory studies show rapid and complete bleaching events of all individuals even at low concentrations (Cadena-Aizaga et al., 2020) including coral cell mortality at 62 ng L^{-1} of BP-3 (Downs et al., 2016). EHMC has been observed to cause 24 % mortality and 96 % bleaching in *Acropora* sp. in 91 h (Danovaro et al., 2008), and 33.3 % mortality and 83.3 % bleaching in *S. caliendrum* at 1000 μg L^{-1} (He et al., 2019a). Bleaching has also been recorded in anthozoa exposed to BP-2, where-by necrosis in the epidermis and gastrodermis resulted in autophagic cell death (Downs et al., 2014).

7. Inorganic UV filters

Of the 44 search results that focussed specifically on the ecotoxicological effects of inorganic UV filters on marine organisms (Table 3), 48 % of the articles investigated zinc oxide (or its nanoparticulate version), and 68 % investigated titanium dioxide (or its nanoparticulate version) (Fig. 3). Articles published on inorganic UV filters were largely conducted on organisms from Central Europe and the United States, with no publications on organisms originating from Australasia, Central America, Eastern or Southern Africa and the Pacific Islands (Fig. 6B). The most frequent primary institution origins were Italy, China and the USA (Fig. 7B).

The publication year of articles focusing on inorganic UV filters has been quite sporadic (Fig. 8B), with peaks in 2017 and 2023. Studies were dominated by research on the classes bivalvia (20 %), bacillariophyceae (20 %) and echinoidea (18 %) [Fig. 5] and largely investigated the effect of these compounds on organisms that were collected from the field (Table 2) [52 %]. 70 % of papers assessed effects on the test organisms after and during acute exposure (i.e. any period <96 h), and only 32 % investigated the effects from chronic/long-term exposure (any period of >96 h). 57 % of the organisms investigated inhabit temperate waters only and 18 % of organisms investigated inhabit tropical waters only, with another 25 % of studies using organisms from both temperate and tropical waters (Table 2).

During the experiments, organisms were mainly exposed to the inorganic UV filters via the aqueous route (82 %) [Table 2]. All studies exposed organisms to environmentally relevant concentrations, of which each were largely exposed to the compounds in a dose-response manner (64 %). However, similar to the papers investigating the ecotoxicological effects of organic UV filters, studies very rarely used UV lighting during the experimental periods (14 %), and of those that did not, only four papers irradiated the solutions prior to exposure. Additionally, 70 % of the studies assessed in this review did not take water column samples during, or at the end of the experiment to assess for compound concentration within the exposure tank. Furthermore, the most common biological assays conducted focussed on oxidative stress (39 %), with antioxidant response, enzymatic activity and transcriptomics being largely under reported. A total of 9 % of the papers assessed investigated the effect of different coatings of nanoparticles, for titanium dioxide these included alumina, silica and dimethicone. For zinc oxide these included 3-aminopropyl-trimethoxysiane and dodecyltrichlorosilane.

8. Properties of inorganic UV filters

Both zinc oxide (ZnO) and titanium dioxide (TiO_2) are used in sunscreens due to their photoactivity (Aitken et al., 2006). Inorganic UV filters can reflect or scatter UV radiation, but do not absorb it. Both nano titanium dioxide ($nTiO_2$) and nano zinc oxide (nZnO) are semiconductors with wide band gaps that can successfully shield UV light (Yuan et al., 2022). Unlike organic UV filters, they are not absorbed easily by the body as they sit on top of the skin in a layer, which can often result in an opaque and pasty appearance when applied to the skin (Cunningham et al., 2020; Lu et al., 2015; Steifel and Schwack, 2015). To avoid this appearance and to allay the growing concerns over organic UV filters passing through the skin barrier after topical application, nanoparticulate versions (nZnO and $nTiO_2$) of these mineral based UV-filters have grown in popularity for use in personal care products (Matta et al., 2019; Takasu et al., 2023; Yuan et al., 2023; Wong et al., 2009; Masala and Seshadri, 2004). These nanoparticles (NPs) possess a smaller particle, leading to a thinner screen when applied to the skin and increased UV blocking potential (Catalano et al., 2020). This is achieved as the nanoparticles are smaller than the optimal light scattering size, allowing visible light to be transmitted (Smijs and Pavel, 2011). Nanoparticulate inorganic UV filters have been observed to alter their physiochemical properties when in seawater compared to freshwater (Wong et al., 2009; Klaine et al., 2008; Kashiwada, 2006). For example, increasing the ionic strength and pH of NaCl suspensions results in increased $nTiO_2$ aggregation rate and size (French et al., 2009; Wong et al., 2009). Given that nanoparticulate UV filters have a greater surface area: volume ratio than their bulk forms, it would be expected that their toxicity would also be greater (Wong et al., 2009). However, due to their aggregation characteristics, in which they can reach or exceed the size of bulk materials, this is not necessarily the case (Adams et al., 2006).

8.1. Titanium dioxide (TiO_2)

TiO_2 can exist in either rutile, anatase or brookite crystalline phases,

Table 3
Studies investigating ecotoxicological effects of inorganic UV filters on marine organisms.

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
Acropora sp.	Anthozoa	Nano zinc oxide (nZnO) & titanium dioxide (TiO_2)	6.3 mg L^{-1}	• Uncoated nZnO induced severe and fast coral bleaching and stimulated microbial enrichment. TiO_2 caused minimal alterations and did not cause bleaching.	Corinaldesi et al. (2018)
Ammonia veneta	Globothalamea	Nano titanium dioxide ($nTiO_2$)	1 mg L^{-1}	• 1 h exposure caused production of ROS in acidic endosomes and mitochondria.	Ishitani et al. (2023)
Artemia salina	Branchiopoda	nZno, $nTiO_2$	10–160 mg L^{-1}	• nZnO more toxic than nTiO2. • Increased catalase (CAT) activity indicating oxidative stress.	Bhuvaneshwari et al. (2017)
Arenicola marina	Polychaeta	$nTiO_2$, TiO_2	1–3 g kg $^{-1}$ sediment	• Cellular damage and a decline in casting rates when exposed to $nTiO_2$.	Galloway et al. (2010)
Brachionus plicatilis	Monogononta	nZnO	0.1–19 mg L^{-1}	• Uptake of nZnO observed, mortality at 19 mg L^{-1-1}	Mohammadi et al. (2021)
Coccoid cyanobacteria	Cyanophyceae	Zinc oxide (ZnO) & $nTiO_2$	1 & 10 mg L^{-1}	• Growth rate declined at 10 mg L^{-1} of TiO_2 and 1 and 10 mg L^{-1} ZnO. Size structure of assemblages was modified by presence of nanoparticles (NP).	Takasu et al. (2023)
Dunaliella salina & *Dunaliella tertiolecta*	Chlorophyceae	$nTiO_2$	50–200 ppm	• Growth inhibition, decrease in chlorophyll and photosynthesis.	Ghazaeu and Shariati (2020)
Dunaliella tertiolecta	Chlorophyceae	$nTiO_2$	1–100 mg L^{-1}	• Population growth rate significantly declined in dose-response manner.	Manzo et al. (2015)
Dunaliella tertiolecta	Chlorophyceae	nZnO & ZnO	0.1–10 mg L^{-1}	• Growth inhibition more prominent in nZnO treatments.	Manzo et al. (2013)
Gracilaria lemaneiformis	Florideophyceae	$nTiO_2$	5–40 mg L^{-1}	• Photosynthetic pigments decreased and antioxidant defence was disrupted after $nTiO_2$ exposure.	Liu et al. (2018)
Haliotis diversicolor supertexta	Gastropoda	$nTiO_2$	0.1–10 mg L^{-1}	• No acute effects observed • Increase in lipid peroxidation and oxidative stress.	Zhu et al. (2011)
Lytechinus variegatus	Echinoidea	$nTiO_2$	0.005–5 μg mL^{-1}	• Embryo and larval toxicity observed.	Palmeira-Pinto et al. (2023)
Lytechinus pictus	Echinoidea	nZnO & $nTiO_2$	0.5–10 mg L^{-1} for $nTiO_2$, 10–200 mg L^{-1} for nZnO	• $nTiO_2$ was not toxic to 10 mg L^{-1}. nZnO was highly toxic even at low concentrations (EC_{50} = 99.5 μg L^{-1}).	Fairbairn et al. (2011)
Mytilus edulis & *Crassostrea virginica*	Bivalvia	$nTiO_2$	1.0 mg L^{-1}	• Both species ingested approximately half of the $nTiO_2$ with >90 % of the NPs being ingested in the faeces in 12 h of exposure. Ingestion not dependant on exposure via diet or marine snow.	Doyle et al. (2015)
Mytilus edulis & *Crassostrea virginica*	Bivalvia	$nTiO_2$	4.4–4.7 mg L^{-1}	• Capture and ingestion not dependant on form of exposure. >90 % of the NPs were eliminated from tissues after 6 h, with trace amounts left after 72 h.	Doyle et al. (2016)
Mytilus galloprovincialis	Bivalvia	nZnO	0.1–2 mg L^{-1-1}	• Reduced growth and increased mortality with an increase in nZnO.	Hanna et al. (2013)
Mytilus galloprovincialis	Bivalvia	nZnO	1, 10, 100 μg L^{-1}	• Lipid peroxidation and significant changes to total protein content (TPC), total antioxidant capacity (TAC), catalase (CAT) and acetylcholinesterase (AChE) enzyme activity.	Roma et al. (2024)
Mytilus galloprovincialis	Bivalvia	$nTiO_2$	0.5–64 mg L^{-1}	• Light was shown to increase toxicity of $nTiO_2$ in the bivalve.	Libralato et al. (2013)
Mytilus galloprovincialis	Bivalvia	$nTiO_2$ & TiO_2	10 mg L^{-1}	• $nTiO_2$ accumulated at a greater rate than bulk TiO_2 and was toxic to transcription process and histopathology.	D'Agata et al. (2014)
Mytilus galloprovincialis	Bivalvia	TiO_2	5–100 μg L^{-1}	• Cellular damage and activated antioxidant defences.	Leite et al. (2020)
Paracentrotus lividus	Echinoidea	nZnO	6 μg L^{-1}	• DNA damage in spermatozoa after 30 min. Sperm fertilization capability not affected, but morphological alterations, including skeletal observed in offspring.	Oliviero et al. (2019)
Paracentrotus lividus	Echinoidea	$nTiO_2$	1 and 5 μg mL^{-1}	• nTiO2 influences signal transduction downstream targets of signalling pathways without inflammatory response.	Pinsino et al. (2015)
Paracentrotus lividus	Echinoidea	$nTiO_2$	0.001–500 mg L^{-1}	• Sillica-coated $nTiO_2$ caused metabolic plasticity on the biosynthesis of metabolites that mediate inflammation, phagocytosis and antioxidant response.	Catalano et al. (2020)
Paracentrotus lividus	Echinoidea	$nTiO_2$	10, 20, and 40 μg mL^{-1}	• Spermatozoa decreased in both vitality and motility.	Ignoto et al. (2023)
Paracentrotus lividus	Echinoidea	nZnO	1 mg and 10 mg Zn/ kg food	• 64 % of individuals had a damaged nucleus in immune cells and 84.7 % of larvae was malformed.	Manzo et al. (2017)
Paracentrotus lividus	Echinoidea	nZnO	6.3 mg L^{-1} and 12.5 mg L^{-1}	• nZnO was incorporated by the larvae and led to skeletal malformations.	Marcellini et al. (2024)

(*continued on next page*)

Table 3 (*continued*)

Species	Organism (Class)	UV Filters Tested	Concentration	Main Findings	Reference
Phaeodactylum tricornutum	Bacillariophyceae	$nTiO_2$	5–200 mg L^{-1}	• Growth inhibition greatest on first exposure day. • Oxidative stress and shading effect observed.	Wang et al. (2016)
Picochlorum sp.	Chlorophyceae	nZnO & $nTiO_2$	10 mg L^{-1}	• Modifications to chlorophyll-a concentrations during early growth stages, effects were reversed during late growth stages.	Hazeem et al. (2016)
Prochlorococcus MED4	Cyanophyceae	$nTiO_2$	1–100 μg L^{-1}	• Decline in growth with increasing $nTiO_2$ concentrations, which recovered with time.	Dedman et al. (2021)
Ruditapes philippinarum	Bivalvia	nZnO	1 and 10 μg L^{-1}	• Increase in catalase and superoxide dismutase activity. • No damage to lipids, proteins or DNA.	Marisa et al. (2018)
Scophthalmus maximus	Actinopterygii	$nTiO_2$ & BP-3	3 μg g^{-1} fish weight	• No oxidative stress observed, liver metabolism altered.	Carvalhais et al. (2021)
Seriatopora caliendrurn	Anthozoa	nZnO	50, 100 and 200 μg L^{-1}	• Corals increased glycerophosphocholine production. Lipid profiles varied under different nZnO exposures.	Tang et al. (2017)
Skeletonema costatum, Thalassiosia pseudonana, Tigriopus japonicus & Elasmopus rapax	Bacillariophyceae, Copepoda & Malacostraca	nZnO, ZnO	4, 40 mg L^{-1}	• nZnO was more toxic towards algae than ZnO, but less toxic towards crustaceans.	Wong et al. (2009)
Talitrus saltator	Malacostraca	$nTiO_2$	151.5 μg L^{-1} & 303 μg L^{-1}	• Significant increase in CAT compared to the control.	Sellami et al. (2024)
Tegillarca granosa, Meretrix meretrix & Cyclina sinensis	Bivalvia	$nTiO_2$	100 μg L^{-1}	• Acidification increases the uptake of $nTiO_2$ by the bivalves.	Shi et al. (2019)
Tetraselmis suecica & Phaeodactylum tricornutum	Eustigmatophyceae & Bacillariophyceae	ZnO & nZnO	0.1–10 mg L^{-1}	• Growth inhibition and reduced replication when exposed to nZnO.	Li et al. (2017)
Thalassiosira pseudonana, Skeletonema costatum, Dunaliella tertiolecta & Isochrysis galbana	Bacillariophyceae, Chlorophyceae	$nTiO_2$	1–7 mg L^{-1}	• Low levels of UV light cause an increase in the capability of $nTiO_2$ to cause oxidative stress.	Miller et al. (2012)
Thalassiosira pseudonana, Chaetoceros gracilis & Phaeodactylum tricornutum	Bacillariophyceae	nZnO	10–80 mg L^{-1}	• Growth inhibition observed in presence of nZnO	Peng et al. (2011)
Thalassiosira pseudonana	Bacillariophyceae	nZnO	10 & 50 mg L^{-1}	• Industrial type nZnO was more toxic than sunscreen nZnO at concentrations below 50 mg L^{-1}. Above this, extent of toxicity was similar.	Spisni et al. (2016)
Thalassiosira pseudonana	Bacillariophyceae	$nTiO_2$	5 mg L^{-1}	• Sunscreen derived $nTiO_2$ was more likely to inhibit growth than industrial or toothpaste derived $nTiO_2$.	Galletti et al. (2016)
Thalassiosira pseudonana, T. weissflogii & I. galbana	Bacillariophyceae, Cosinodiscophyceae & Coccolithophyceae,	nZnO (different coated)	0.1, 0.5, 1, 3, 5, 10, 30, 50, 80 or 100 mg L^{-1}	• A-ZnO (3-Aminopropyl- Trimethoxysilane coated Zinc Oxide) NPs inhibited growth of cells. *T. pseudonana* exposed to ZnO-NPs, A-ZnO-NPs or D-ZnO (Dodecyltrichlorosilane coated Zinc Oxide) NPs resulted in differential expressions of genes.	Yung et al. (2017a)
Thalassiosira pseudonana	Bacillariophyceae	nZnO	0.5–50 mg L^{-1}	• Toxicity of nZnO was greatest at 30 °C.	Yung et al. (2017b)
Trachinotus carolinus	Actinopterygii	$nTiO_2$	1.5–3 μg L^{-1}	• Genotoxicity was induced, but was dose and time dependant.	Vignardi et al. (2015)
Zoanthus sp.	Anthozoa	ZnO & $nTiO_2$	1 mg L^{-1}	• Rapid bleaching, growth inhibition and reduced moisture content in zooxanthellae after 24 h. $nTiO_2$ had little to no effect.	Yuan et al. (2023)

of which, pure rutile $nTiO_2$ coated with a surface passivation layer is the most common in sunscreen formulations (Ignoto et al., 2023; Markowska-Szczupak et al., 2011; Catalano et al., 2020; Gerloff et al., 2012; Botta et al., 2011). Rutile and anatase TiO_2 have a tetragonal structure, with anatase being particularly less dense and more photocatalytically active (Doyle et al., 2015; Dankovic and Kuempel, 2011; Serpone et al., 2007). However, TiO_2 can only reflect UVB radiation and it is weakly soluble in water (Theodore, 2006; Parades et al., 2014; Miller et al., 2010). It is estimated that the average quantity of $nTiO_2$ in sunscreens is approximately 4 % of the overall product mass (Yuan et al., 2022). Sunscreen grade $nTiO_2$ have particle ranges of between 10 and 60 nm (Popov et al., 2005), which then form aggregates of between 30 and 150 nm (Schiavo et al., 2018). When TiO_2 reaches an electronically excited state, an electron is promoted from the valence band to the conduction band, which thus generates a hole within the valence band (h^+) (Miller et al., 2012). The resulting electron-hole pair can recombine or migrate to the surface of the particle, allowing it to react with H_2O or OH^- to form $OH^•$ (Miller et al., 2012). The electrons may also react with the absorbed molecular oxygen to form $O_2^{•-}$ ions (Czili and Horvath, 2008). It has been suggested that toxic effects of TiO_2 on biota when exposed to UVA radiation is likely to take place via secondary products that are less reactive, such as carboxyl radicals (Dodd and Jha, 2009, 2011; Reeves et al., 2008).

8.2. *Zinc oxide*

Zinc exists in two crystalline forms - wurtzite and zinc-blende, with wurtzite most common to sunscreen formulations (Smijs and Pavel, 2011). Zinc has a strong absorption ability and chemical stability, but it is highly soluble in water (Osmond and McCall, 2010; Catalano et al., 2020). ZnO can reflect both UVA and UVB and nZnO is a very effective absorber of UV light (Theodore, 2006; Yung et al., 2017a, 2017b; Özgür

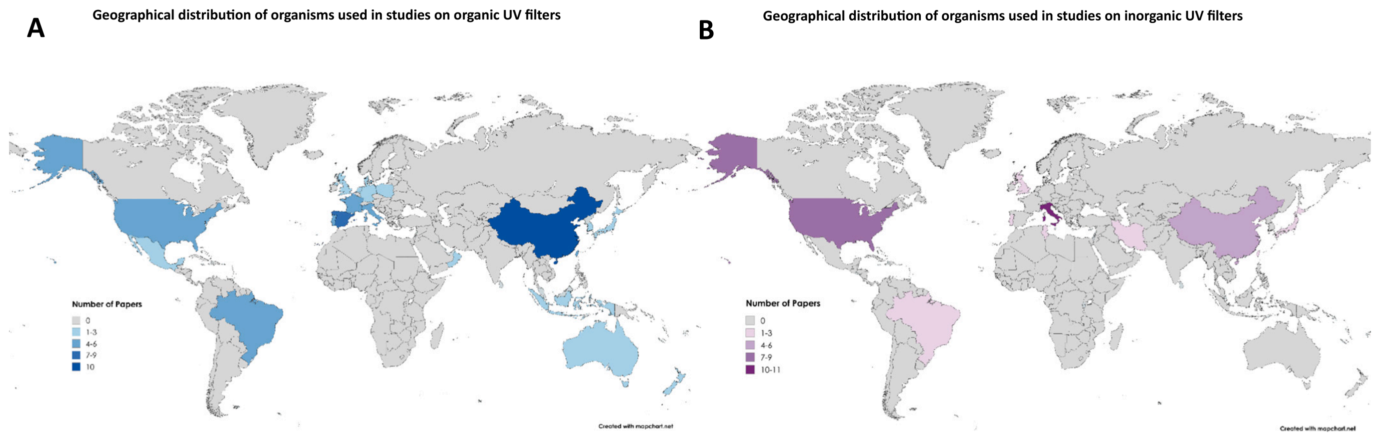


Fig. 6. A&B - Geographical distribution by country of the number of articles published on the ecotoxicological effects of organic (A) and inorganic (B) UV filters on marine organisms included in this literature review. Countries were assigned based on the location organisms used in the studies were obtained from. 4 of the organic and 5 of the inorganic UV filter studies did not list the origin of the organisms, and have not been included in these figures.

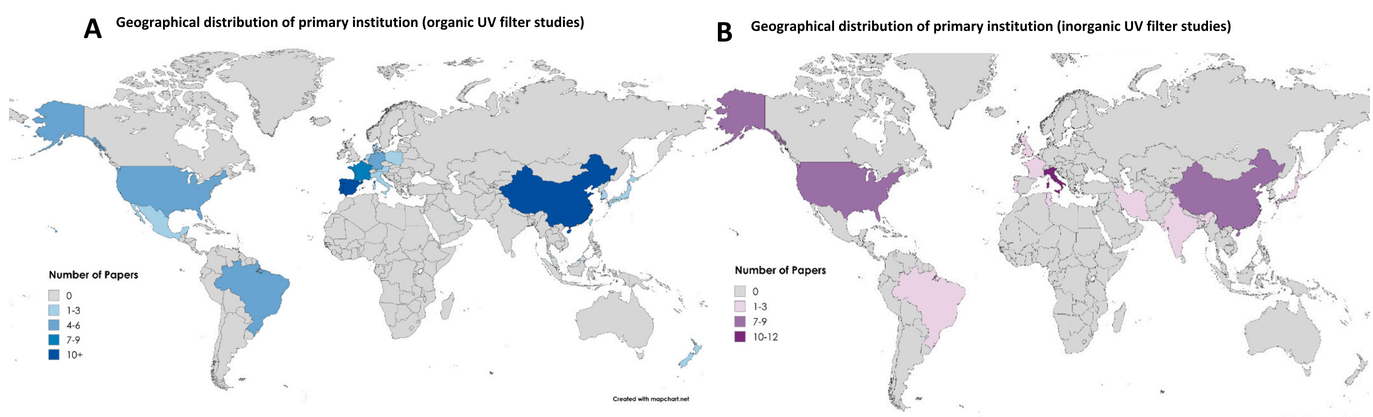


Fig. 7. A & B - Geographical distribution of the number of articles published from each country on the ecotoxicological effects of organic (A) and inorganic (B) UV filters on marine organisms, based on the primary author's institution address.

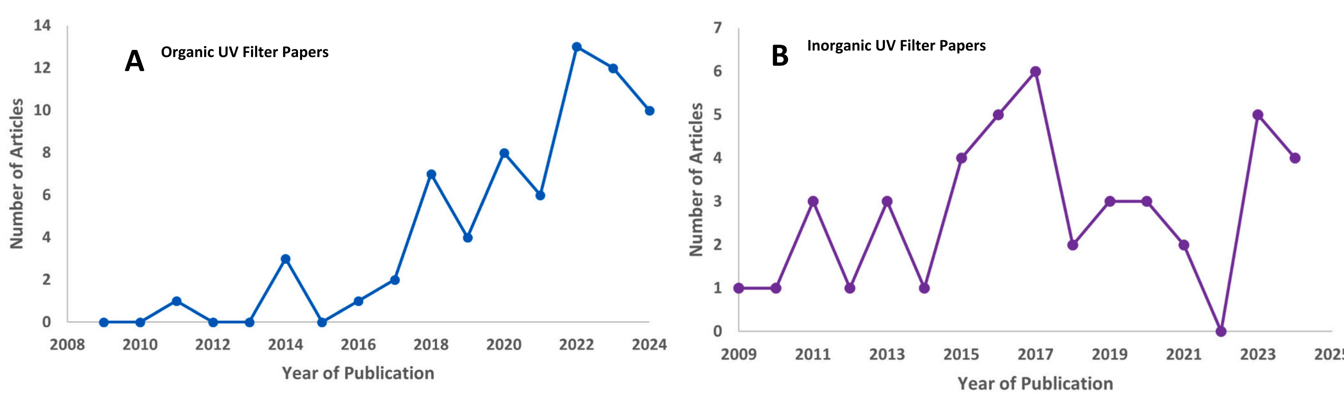


Fig. 8. A & B - Number of articles published each year on the ecotoxicological effects of organic (A) and inorganic (B) UV filters.

et al., 2005). ZnO nanoparticles release free metal ions in solution state when in an aqueous media, and the dissolution of the particles depends on particle size, surface area and rough degree (Suman et al., 2015).

9. Nanoparticle surface coatings

Nanoparticles in sunscreens are often coated with engineered polymers or inorganic and organic substances to improve the stability of the particle, but which can also inadvertently increase the toxicity of the sunscreen (Brayner et al., 2006; Yuan et al., 2022). These coatings modify the physiochemical properties of the raw nanoparticle, and therefore affect the particle stability and mobility (Wang and Fan, 2014). A study conducted in 2011 found that eight out of nine commercial sunscreen products were coated in non-volatile inorganic residues including aluminium oxide (Al_2O_3) and silicon dioxide (SiO_2) to minimise the photochemical activity of the TiO_2 (Lewicka et al., 2011).

Aluminium hydroxide ($Al(OH)_3$) can also be added to sunscreen matrices as an opacifying and viscosity controlling agent (Becker et al., 2016). Additional additives such as cobalt are often added to improve the performance of $nTiO_2$ and allow for optimal skin conditioning (De la Calle et al., 2017). However, this coating layer is rapidly dissolved in aqueous media, resulting in the potential ROS formation to remain once in aquatic environments (Brezová et al., 2005; Lewicka et al., 2013).

10. Transformation products

$nTiO_2$ can co-occur with other chemicals in the aquatic environment and increase their toxicity on organisms (Rodriguez-Romero et al., 2019). TiO_2 is degraded within aquatic environments to generate new products, the fate and impact of which is unknown (Fouqueray et al., 2012; Labille et al., 2010). Further to this, other pollutants such as hydrocarbons or pesticides present in the aquatic environment may act synergistically with nanoparticulate UV filters (Schlumpf et al., 2004). $nTiO_2$ can act as a pollutant absorber and can also facilitate the bioaccumulation of pollutants (Lu et al., 2018), as observed with phenanthrene in marine ark shell (Tian et al., 2014), and cadmium in marine zooplankton (Lu et al., 2018). There is considerable evidence for the formation of reactive oxygen species (ROS) when titanium dioxide is exposed to UV light (Uchino et al., 2002; Cai et al., 1992; Konaka et al., 2001). This could include hydroxyl radicals (OH), superoxide radical anions (O^{2-}), hydrogen peroxide (H_2O_2) and singlet oxygen ($^1O_2$) (Reeves et al., 2008). $nTiO_2$ nanotoxicity in marine environments has been reported to be related to increased ROS levels caused by the internalisation of $nTiO_2$ (Xia et al., 2015). ZnO nanoparticles (nZnO) can release free zinc ions, resulting in enhanced production of ROS (Suman et al., 2015). nZnO releases more zinc ions in seawater compared to freshwater due to its higher ionic strength (Zhang et al., 2016) and can disturb the balance between oxidation and anti-oxidation processes, which can result in oxygen stress responses (Hao and Chen, 2012). ROS can result in protein oxidation, and in turn, protein carbonylation and lipid peroxidation (Kera et al., 2024; Cuccaro et al., 2022c).

11. Ecotoxicological effects of inorganic UV filters

There are a smaller number of studies which report the ecotoxicological effects of inorganic compared to those focusing on organic UV filters (Fig. 3). Table 3 summarises the results retrieved, which include ecotoxicological effects of inorganic UV filters at a molecular, cellular, individual and assemblage level.

11.1. Molecular and cellular level effects

Marine mussels (*Mytillus galloprovincialis*) have been demonstrated to significantly accumulate $nTiO_2$ within the digestive gland, and haemocytes showed significantly enhanced DNA damage (D'Agata et al., 2014). Additionally, $nTiO_2$ caused chromosomal alteration in European sea bass at a concentration of 1 mg L^{-1} (Nigro et al., 2015), as well as histopathological alterations in gills and the digestive gland of marine scallops (Xia et al., 2017). Furthermore, nZnO induced DNA damage in the spermotazoa of *Paracentrotus lividus* (Oliviero et al., 2019) while $nTiO_2$ enhanced the production of ROS in acidic endosomes and mitochondria of *Ammonia veneta* (Ishitani et al., 2023). $nTiO_2$ particles are photocatalytic (Tsuang et al., 2008; Dodd and Jha, 2009, 2011) and UV radiation has the potential to increase the toxicity of $nTiO_2$ through the production of reactive oxygen species, (Nakagawa et al., 1997; Reeves et al., 2008) which induce oxidative stress and cellular damage within the organism (Labille et al., 2010; Dodd and Jha, 2009, 2011). When zinc oxide nanoparticles penetrate the cell barrier, they can undergo translocation into the intracellular environment via diffusion and endocytosis (Yuan et al., 2022; Zhu et al., 2010). Once inside the cell, the nanoparticles can interact with the DNA, or attach to organelles (Wang et al., 2016). Small nanoparticles (<10 nm) can reach the nucleus through pores, whilst larger nanoparticles have the potential to bind with DNA molecules during mitosis (Peng et al., 2017). A study investigating the effects of nZnO on oysters *(Crassostrea gigas*) found that the nanoparticles accumulated in the gills and the digestive gland, and subsequently induced mitochondrial disruption and triggered oxidative stress (Trevisan et al., 2014). In algae *(Chlorella vulgaris)*, nZnO between the concentrations of 200 and 300 mg L^{-1} increased the antioxidant defence capacity and resulted in cellular damage (Suman et al., 2015). In *Isochyrisis galbana* cells, $nTiO_2$ negatively impacted chlorophyll concentrations, resulting in a reduction in photosynthesis, despite no effect on cell size (Hu et al., 2018).

11.2. Individual and assemblage level effects

Very little research has investigated the biological effects of sunscreen derived nZnO UV filters. A study conducted by Wong et al. (2020) found that sunscreens containing zinc oxide nanoparticles can trigger oxidative stress and toxicity in the marine copepod *Tigriopus japonicus.* Furthermore, several studies have assessed the biological effects of nZnO on the marine mussel *Mytilus edulis* (George and Pirie, 1980; Akberali and Earnshaw, 1982; Anandraj et al., 2002; Burbidge et al., 1994), with more limited data available for *Mytilus galloprovincialis* (Hanna et al., 2013; Roma et al., 2024). From these studies, it has been concluded that nZnO can decrease enzymatic activity in mussel tissue (Akberali and Earnshaw, 1982), and ZnO can accumulate within the mantle and the gills (George and Pirie, 1980). In echinoderms, $nTiO_2$ decreased the vitality and motility of *Paracentrotus lividus* spermatozoa (Ignoto et al., 2023). ZnO exposure resulted in significant skeletal abnormalities, stage arrests and axis determination disruption in *Strongylocentrotus pupruratus* (Cunningham et al., 2020). Despite several studies outlining negative effects of nanoparticles on aquatic life, some studies have found little or no effect. For example, a study conducted on clownfish tested varying concentrations of both BP-3 and TiO_2 containing sunscreens and found TiO_2 to be significantly less toxic than the BP-3 treatments (Barone et al., 2019). The toxicity of $nTiO_2$ is heavily dependent on aggregate size and surface chemistry (Grassian, 2008), which plays a key factor in the determination of $nTiO_2$ bioavailability to plant roots, algae and fungi (Navarro et al., 2008). Aggregate size within a solution can be dependent upon pH, ionic strength and cation valence (French et al., 2009), as well as the presence of organic acid, which can decrease aggregation (Domingos et al., 2009). Sysoltseva et al. (2017) found that sunscreen SPF rating was negatively correlated with aggregate size, with the lowest SPF's containing the largest $nTiO_2$ particles at sizes up to 1.6 μm. There is evidence to suggest that nanoparticles can inhibit algal photosynthesis by accumulating on the surface of the cells and limiting light availability, producing a shading effect (Wang et al., 2016), as well as evidence that size structure of cyanobacteria assemblages can be modified by the presence of TiO_2 and ZnO nanoparticles (Takasu et al., 2023).

12. Pathways of UV filters into the marine environment

At least 25 % of photoprotective personal care products applied to the skin get washed off during sea bathing, meaning a significant quantity of sunscreen product enters the marine environment directly through water-based activities (Danovaro et al., 2008; Stokes and Diffey, 1999; Downs et al., 2021b; Caloni et al., 2021). However, shedding rates from swimmers are more likely to be closer to 50 % (Downs et al., 2022). A recent estimation has stated that for every 1000 visitors wearing sunscreen on a single beach, >35 kg/day of sunscreen is deposited into the marine environment (Downs et al., 2022; U.S. NOAA, 2022; Diffey, 2001).

Sunscreen pollutants can also enter the marine environment via indirect pathways including industrial and domestic wastewater discharges (Tian et al., 2021). For instance, via the washing of towels that have been used to dry sunscreen-coated skin, washing off residue during showering and even via urine (Li et al., 2007). Further to this, sunscreen pollution has also recently been associated with agricultural practices, where-by recycled water from wastewater treatment plants and sludge biosolids are used as soil fertilisers (Plagellat et al., 2006; Sunyer-Caldu and Diaz-Cruz, 2021; Cadena-Aizaga et al., 2022). This practice can result in the spread of UV filter contaminants not only onto crops, but in associated run-off and discharges into aquatic environments (Serra-Roig et al., 2016; Bigott et al., 2022).

Another source of UV filter contamination in the aquatic environment is through the use of beach showers. A study conducted by Downs et al. (2021b), found that sands around the beach showers of Hanauma Bay in Hawaii were highly contaminated with sunscreen residues (Downs et al., 2021b). A further study conducted by Downs et al. (2022) found that beach showers are a point-source of contamination to coastal surface waters and concluded that sunscreen contamination was directly correlated with high visitation rates (Downs et al., 2022).

Traditional sewage treatment technologies do not have the capacity to remove UV filter compounds from effluent (Tian et al., 2021; Schneider and Lim, 2018) due to their low water solubility, high lipophilicity and high organic carbon-water coefficient (Fivenson et al., 2021). Ozonation is often used at wastewater treatment plants to degrade organic pollutants, however, this method of disinfection is ineffective in reducing the toxicity of UV filters (Hopkins et al., 2017).

13. Recommendations for future research

This review has examined the current understanding and the key knowledge gaps pertaining to the effects of UV filters on marine organisms. Current research has largely focussed on the biological effects of organic UV filters, namely benzophenone-3 on bivalves and anthozoa. However, there are significant knowledge gaps in the literature relating to the effects of inorganic and new-age synthetic UV filters such as Diethylamino Hydroxybenzoyl Hexyl Benzoate and Diethylhexyl Butamido Triazone. Additionally, there is a significant lack of research in certain geographical locations, with little to no studies having been conducted on organisms from Northern Europe, Africa, South America, the Arctic and Antarctic regions, and many island nations. The literature examined in this review largely did not use UV lighting in their methodology, focussed on a very narrow range of marine organisms and life stages, did not have an emphasis on bioaccumulation or biomagnification within the food-chain, nor an emphasis on varied uptake routes (i.e. dietary, sediment, aqueous exposures), with both dietary and sediment exposure very underrepresented in both organic and inorganic studies. The chronic, long-term effects were largely neglected for inorganic studies; with more specific focus on acute biological effects that took place within 96 h.

In this final section, we explore areas that require further research, such as the ecotoxicological effects of UV filters with a food-chain emphasis, and we provide recommendations for future research efforts to focus on, including the implementation of ultraviolet lighting and emphasis on the biological effects after chronic exposure.

13.1. Use of UV lighting

Lighting used during experimental procedures should include the use of UV, to replicate the conditions the chemicals will be exposed to in the natural environment. The photocatalytic activity of TiO_2 has to be induced by UV light, and the exposure of UV filters to UV induces the chemicals' full toxicity potential (Hund-Rinke and Simon, 2006; Zhang et al., 2021). However, the majority of studies reviewed here reported the use of conventional light emitting diode (LED) lighting. The use of UV lighting during experimental procedures was reported for only 13 % of the papers reviewed, with 4.5 % reporting irradiating the UV solutions prior to organism exposure. The use of UV lighting or irradiation in experimental designs replicates pseudo-natural conditions, without which, some important effects could be masked.

13.2. Use of a wider range of organisms and life stages

Many studies have reported the use of early life stage marine organisms as bioindicators of UV filter contamination, particularly effects on the embryos and larval developments of the purple sea urchin, *Paracentrotus lividus* (Varella et al., 2022; Beiras et al., 2019; Giraldo et al., 2017; Parades et al., 2014). This species is often used in ecotoxicological studies due to its status as an 'ecosystem engineer' (Varella et al., 2022; Pages et al., 2012), as well as being a geographically established and widespread species (Pages et al., 2012). Early life stage organisms are particularly susceptible to environmental pollutants, often more-so than juvenile or adult organisms (Mohammed, 2013), as reported for other toxicants like metals (Ringwood, 1990; Green et al., 1986). Whilst the classes bivalvia, echinoidea and anthozoa are the most largely represented in existing studies, there are large gaps in our understanding of the effects on many other organisms (Fig. 5). In order to build a more comprehensive understanding of the effects of UV filters at species and ecosystem levels, a wider diversification of the tested species and life stages is required, along with an emphasis on organisms from a wider range of geographical regions.

13.3. Emphasis on a food-chain approach & dietary uptake

In general, UV filters are hydrophobic and lipophilic substances, with the potential to accumulate within biota (National Academies of Sciences, 2022). Only 5 % of studies reviewed here opted to expose organisms via dietary uptake. Currently, very little research has been conducted on food-chain analysis and potential of bioconcentration and biomagnification despite the European Legislation on the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) requiring additional ecotoxicity data on UV filters (European Commission, 2007). An understanding of a compounds' bioaccumulation and bioamplification capacity, potentially in the form of trophic magnification factors (TMF), is essential to understand long-term exposure risks to organisms at various trophic levels, as well as to understand their level of persistence, mode of transfer and potential chronic risks to provide benchmark data for safety and regulatory standards and ERAs. More work must be conducted on dietary exposure mechanisms to elucidate their effects via this uptake route (Carve et al., 2021), with separate analysis for both marine and freshwater food-chains, as compounds may act differently in saline matrices. Future studies should focus on long-term, chronic exposures to assess the extent of this potential, considering the potential for and effects of trophic transfer and their degradation products, as well as studies encompassing a range of model organisms from different trophic levels, different geographical locations (e.g. temperate and tropical) and varying life stages.

13.4. Streamlined focus of the UV filters researched

Most studies to date focus on BP-3 and 4-MBC, with minimal consideration of the diversity of compounds that are used in commercial sunscreen products. BP-3 absorbs UVB (280-315 nm) and UVA (315-355 nm) [American Chemicals Society, 2018], whilst 4-MBC absorbs UVB (280-320 nm) [Cosmile Europe, 2024]. Future studies should assess the effects of a range of UV filters with different properties and chemical profiles. It is essential to study the potential effects and relative toxicity of all known UV filters used in sunscreens and personal care products, with particular priority given to new-age synthetic UV filters, such as BEMT and DHHB, that are rapidly replacing the presence of compounds such as Padimate O PABA, as well as widely used UV filters registered for use in the EU, UK and USA. In addition, the ecotoxicity and environmental fate of transformation and breakdown products should also be assessed to aid the generation of comprehensive ERAs.

Far fewer studies have focussed on the ecotoxicological effects of inorganic filters compared to organics, or on the effects of their transformation and breakdown products, as well as the effects of different nanoparticle coatings. Further to this, little is known about the effects of aggregated nanoparticles – are they more toxic than their bulk counterparts? Rodríguez-Romero et al. (2022) state that many papers do not consider the chemical changes that take place to nanoparticles in seawater. For example, once ZnO is released in the aqueous phase, it reaches an equilibrium with Zn in stable colloidal residues and there-by influencing the ability of free Zn and its bioaccumulation (Rodríguez-Romero et al., 2022). Despite this, very little research has focussed on the uptake routes and mechanisms at play during the process of transmission throughout the food-chain, which is imperative to understand the true effects of UV filters on marine organisms.

13.5. Biological effects after chronic exposure

Elucidating biological effects in ecotoxicology after and during an acute exposure period (<96 h), is highly valuable to understand the immediate effects of the compound. However, exposing organisms for a chronic, or long-term period (i.e. over 96 h), allows for an assessment of its inherent toxicity, and therefore can allow for appropriate risk and hazard assessment. A major aim in ecotoxicology is to elucidate the mechanisms by which contaminants influence normal biological performance. Therefore, this cannot be comprehensively achieved without an investigation of the effects of a compound after chronic exposure. An organism may be subjected to low concentrations of a compound in the natural environment for several months, and thus, we cannot accurately predict the effects, without a fully comprehensive chronic exposure assessment. In the research examined in this literature review, most studies on inorganic UV filters adopted an acute exposure period, which is often more logistically feasible and financially viable. However, further research is required to fully elucidate their effects on marine organisms after chronic exposure.

14. Conclusion

It is evident that UV filters can have significant biological effects on marine organisms when exposed to environmentally relevant concentrations, including, but not limited to, reproductive and growth inhibition, DNA mutations and oxidative damage. However, chemical characterisation adopted by many of the studies do not provide an integrated approach to determine the true potential hazard of UV filters to biological organisms.

Current research has largely focussed on the ecotoxicological effects of only a selected UV filters (e.g. benzophenone-3, 4-methylbenzylidene camphor) with limited studies on inorganic and new-age synthetic filters. Future research should address these gaps by exploring a wider range of UV filters and by adopting a food-chain approach to assess bioconcentration, bioaccumulation and biomagnification potential, allowing for predictions with ERA. A wider range of organisms from varied geographical locations and life stages should be used to fully determine the risk at all trophic levels from different global latitudes. Additionally, as the effects of degradation and transformation products is largely unknown, we cannot accurately predict the ecotoxicological effects without an understanding of the mechanisms with which they breakdown, their stability, and their environmental fate. Improvements to sampling strategies and the chemical analysis of environmental samples, the use of more environmentally appropriate methods (i.e. UV lighting), and an integrated approach incorporating an array of exposure durations (i.e. acute vs chronic) and methods (i.e. dietary, sediment, aqueous), will all contribute to better constraining the environmental and toxicological effects of UV filters on marine organisms at different biological levels of organisation.

CRediT authorship contribution statement

Anneliese A. Hodge: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Frances E. Hopkins:** Writing – review & editing, Supervision. **Mahasweta Saha:** Writing – review & editing, Supervision. **Awadhesh N. Jha:** Writing – review & editing, Supervision.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2025.117627.

Data availability

All literature data used in this review has been provided in the review tables.

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