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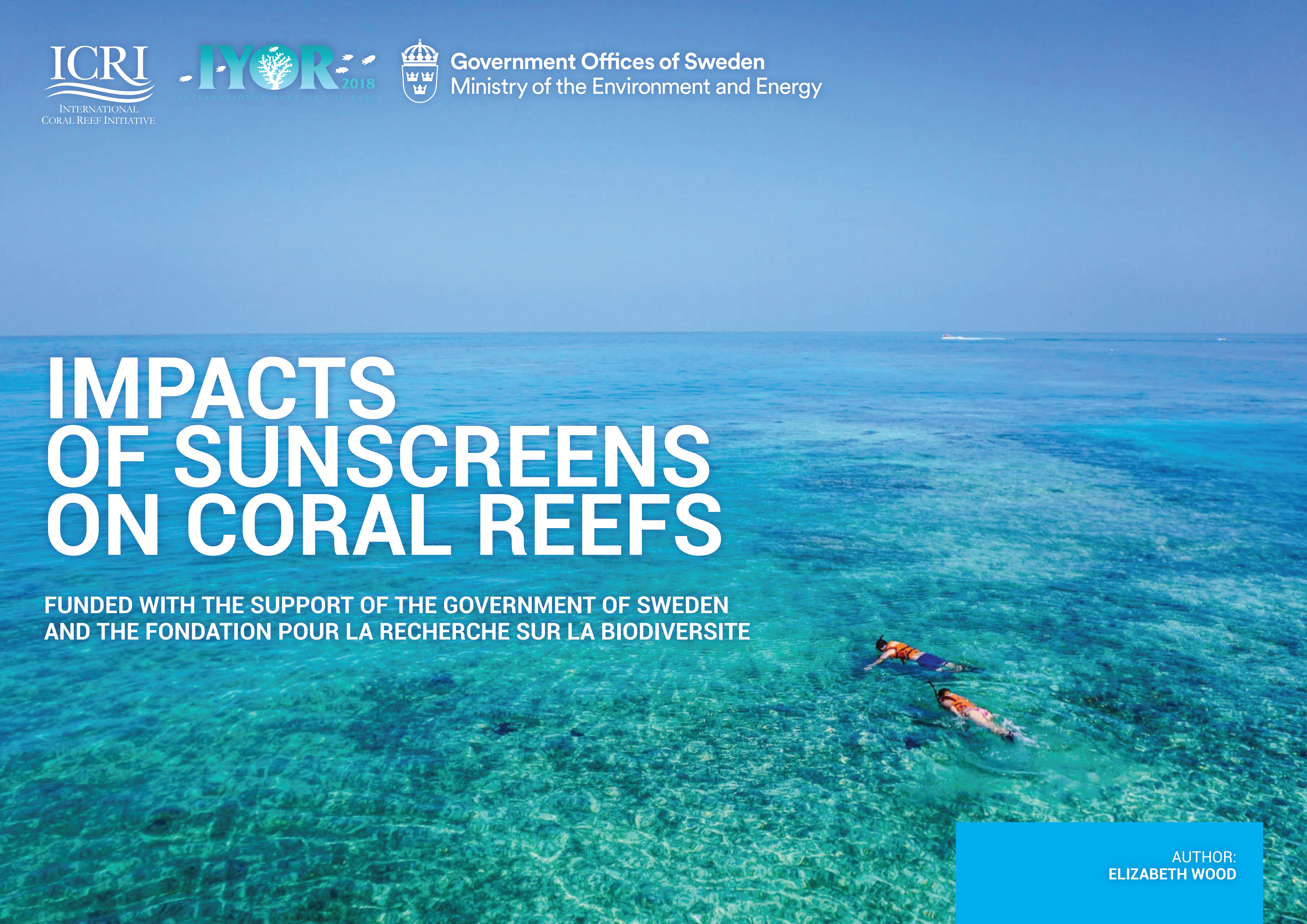

IMPACTS OF SUNSCREENS ON CORAL REEFS

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SUMMARY

This document responds to Goal 3(5) of the International Coral Reef Initiative (ICRI) Plan of Action 2016-2018, which seeks to review issues relating to the impact of sunscreens on coral reefs.

Sunscreens contain organic (chemical) and/or inorganic (mineral) UV filters that absorb, reflect or scatter UV light. They also contain inactive ingredients such as anti-microbial preservatives, moisturisers and anti-oxidants.

Sunscreen ingredients including chemical (benzophenone-3 and -4 (BP-3 or oxybenzone; BP-4), ethylhexyl methoxy cinnamate (EHMC), homosalate (HMS), 4-methylbenzylidene camphor (4-MBC), diethylamino hydroxybenzoyl hexyl benzoate (DHHB)) and mineral (titanium dioxide and zinc oxide) UV filters have been detected in coastal waters. UV filters reach coastal waters either directly as a consequence of washing off swimmers and/or indirectly from wastewater treatment plant effluents. Many of these components have also been found in marine biota including fish, molluscs and corals as well as in sediments.

Where sunscreen components have been detected, concentrations are very variable. They are generally found at barely detectable levels of a few parts per trillion but much higher concentrations of over 1 part per million (ppm) have been reported in a few locations (e.g. 1.395 ppm BP-3 reported by Downs et al (2015) in the US Virgin Islands).

A small number of studies have shown that sunscreen and certain individual components of sunscreen can have negative effects on corals and other marine organisms under certain circumstances. The chemical UV filter oxybenzone has been studied most intensively and the following effects have been described:

- Bleaching of coral fragments and coral cells from various species of hard coral. This effect is more pronounced at higher water temperatures.
- Induction of the lytic viral cycle in symbiotic zooxanthellae with latent infections.
- Damage and deformation of coral larvae (planulae).
- Damage to coral DNA and to their reproductive success.

To date, experiments have largely been undertaken ex-situ and there are concerns that they may not properly reflect conditions on the reef, where pollutants could be rapidly dispersed and diluted. In general, concentrations of UV filters used in experimental work have been higher than likely to be encountered in the reef environment. Most experiments have also been of relatively short duration (12 or 24 hours). On the reef, while UV filters may be at lower concentrations, they can accumulate in organisms and sediment and thus become persistent, with largely unknown consequences.

Research to date has also concentrated mainly on the effects of sunscreens and individual chemicals at subcellular, cellular and individual organism level, with very few studies of wider impacts.

Further research is needed to better understand which ingredients are safe and which pose a realistic threat to marine ecosystems, in particular:

- Determining concentrations of different sunscreen components in the water column, sediment and biota, comparing locations that are heavily visited and/or affected by coastal run-off with those that are more remote.
- Investigating the effects of sunscreen pollutants at reef community and ecosystem level.
- Exploring links between organismal or cellular/ subcellular-level biomarkers that more easily allow the study of pharmaceutical effects on biodiversity/ community composition and ecosystem functioning.
- Investigating UV filter toxicity in relation to predicted warming and ocean acidification conditions.
- Researching the extent and significance of bio-concentration and bio-accumulation of organic UV filters as well as the consequences of long-term, chronic exposure to sunscreen pollutants.
- Studying the effects of pharmaceutical and personal care products (PPCP) mixtures to provide a more realistic picture of ecological risks posed by the use of these products.
- Identifying those ingredients that are safe and those that pose a realistic threat to marine ecosystems.

Considering the many stresses already faced by reefs and current concerns about the toxicity of certain components of sunscreens to corals, a proactive and precautionary approach to dealing with this issue may be required. Reducing the amount of harmful sunscreen components that reach the reef environment is a high priority and will require the involvement of governments, reef managers, divers, snorkelers and swimmers, and the tourism and pharmaceutical industries. The following measures are recommended:

- Encouraging the manufacture of reef-friendly sunscreens.
- Promoting the use of reef-friendly sunscreens and other methods of UV protection
- Regulating the sale and use of sunscreens containing toxins
- Exerting consumer pressure to encourage development and use of eco-friendly sunscreens
- Introducing financial disincentives for manufacture and use of potentially damaging sunscreens

1/ INTRODUCTION

There is intense concern about the future health and ecological integrity of coral reefs in the face of global climate change. This is considered to be one of the greatest threats to reefs worldwide and is causing coral bleaching and ecosystem change at unprecedented levels. Coral reefs are also under considerable stress from overfishing, destructive fishing, coastal development and pollution. There is universal agreement that coral reefs face an unpredictable future and that action needs to be taken at all levels if their integrity and values are to be maintained.

In recent years, a number of studies have shown that sunscreens and other cosmetic products contain chemical substances that are adding to the pollution burden faced by coral reefs. Exposure to ultraviolet (UV) solar radiation poses a threat to public health, including the risk of sunburn, photo-aging and skin cancer (Pathak, 1987), and growing concern about these harmful effects has led to an increase in use of sunscreens. The world's coastal population and coastal tourism are expected to grow during this century and it is anticipated that the use of sunscreens and cosmetics containing UV-filters will rise further. Given the status of reefs, it is essential that the impact of sunscreens on corals is assessed and addressed.

2/ COMPOSITION OF SUNSCREENS

Early sunscreens worked by providing a physical barrier between the skin and the sun's rays but by the 1960s, the cosmetic industry had developed more complex sunscreens containing specific UV filters. These filters come in two forms, organic (chemical) and inorganic (mineral), which act by absorbing, reflecting or scattering UV light. Organic and inorganic filters are often used in combination to offer full protection against both UV-A and UV-B radiation. They are also present in a large number of other pharmaceutical and personal care products (PPCPs).

Organic filters include benzophenone-3 (BP-3; also known as oxybenzone), benzophenone-4 (BP-4), para-aminobenzoic acid (PABA) and PABA esters, cinnamates, salicylates, camphor derivatives, dibenzoylmethanes and anthranilates (Chisvert et al., 2001; Danovaro and Corinaldesi 2003). These are generally used in combination because no single one, at currently permitted concentrations, provides sufficient protection against UV radiation. Sunscreens typically comprise up to 20 or more chemical compounds (Danovaro et al., 2008).

The most widely used mineral filters are zinc oxide and titanium dioxide, often in the form of nanoparticles (NPs) to avoid the whitening effect on the skin caused by these compounds. The International Cooperation on Cosmetic Regulation defines a nanomaterial as an insoluble, intentionally manufactured ingredient with one or more dimensions ranging from 1 nm to 100 nm in the final formulation (Auffan et al., 2009; Ansell et al., 2010). Other minerals used in sunscreens include silicate-based substances such as talc and kaolin. Sunscreens also contain inactive ingredients such as anti-microbial preservatives, moisturisers and anti-oxidants which may make up between 30 and 70% of the product.

3/ OCCURRENCE OF SUNSCREEN COMPONENTS IN THE MARINE ENVIRONMENT

This section reviews studies have been carried out to investigate the occurrence and concentration of UV filters in seawater, marine sediments and biota. These are not confined to areas with coral reefs but provide a broad picture of the range of sunscreen components that reach the marine environment. Risk assessments have also been conducted to provide an indication of the environmental implications of this pollution.

Organic and inorganic UV-filters reach coastal waters either directly from people who have applied sunscreens before entering the water and/or indirectly from wastewater treatment plant effluents (e.g. Daughton and Ternes, 1999; Danovaro et al., 2008; Sánchez-Quiles and Tovar-Sánchez 2015). Sunscreen ingredients, including oxybenzone, BP-4, ethylhexyl methoxy cinnamate (EHMC), homosalate (HMS), 4-methylbenzylidene camphor (4-MBC), diethylamino hydroxybenzoyl hexyl benzoate (DHHB), titanium dioxide and zinc oxide, have been detected in coastal waters (Daughton and Ternes 1999; Giokas et al. 2007, Danovaro et al., 2008; Tovar-Sánchez et al., 2013; Sánches Rodríguez et al., 2015; Downs et al., 2015).

The overall quantity of sunscreens that enter the ocean is not known, although some estimates have been made. Some 20,000 tons of sunscreen were estimated by Corinaldesi (2001) to be released annually into the northern Mediterranean. Danovaro et al (2008) estimated that 4,000-6,000 tons of sunscreen wash off people into coral reef areas each year but Downs et al (2015), citing Shaath and Shaath (2005), UNWTO (2007), Danovaro et al. (2008) and Wilkinson (2008), concluded that 6,000-14,000 tons of sunscreen lotion are released into coral reef areas each year. Downs et al (2015) note that many of the lotions contain 1-10% of the UV filter oxybenzone and suggest that at least 10% of global reefs and 40% of coastal reefs are at risk of exposure to this chemical. However, to date, the level of exposure has only been quantified at a few coral reef sites.

Where sunscreen components have been detected, concentrations are very variable; most work has been undertaken on oxybenzone and information for this chemical is summarised in Table 1. Sunscreen filters are generally found at barely detectable levels of a few parts per trillion (ng/L), but concentrations of over 1 part per million (mg/L) have been reported (e.g. 1.395 ppm oxybenzone reported by Downs et al (2015) in the US Virgin Islands). It is however, important to note, that although the concentrations may be low, persistence of low concentrations may have additive effects if these chemicals are sequestered by marine organisms.

TABLE 1/ CONCENTRATIONS OF OXYBENZONE (BP-3) REPORTED IN SHALLOW WATERS AT VARIOUS SITES.

Further details for each of the countries/states where research has been undertaken are provided after the table.

SITE	SAMPLING LOCATION	OXYBENZONE CONCENTRATION
US VIRGIN ISLANDS: CARIBBEAN SEA		
Trunk Bay 2007 (Downs et al., 2015)	Shallow water adjacent to reef. 180 people in water prior to sampling	580µg L-1 - 1.395 mg L-1
Trunk Bay June 2013. (Bargar et al., 2015)	2 samples at shoreline, 1 sample 30m from offshore island at 1m depth	1,943 – 4,643ng L-1
Trunk Bay June 2014. (Bargar et al., 2015, Fig 3 data supplied by Bargar)	shoreline at <1m depth	6,073 ng L-1
Trunk Bay June 2014	60m from shore at <1m depth	1,416 ng L-1
Trunk Bay June 2014	120m from shore at <1m depth	363 ng L-1
Trunk Bay June 2014	220m from shore at <1m depth	116 ng L-1
Trunk Bay June 2014	220m from shore at c 3m depth	0 ng L-1
Hawksnest Bay 2007 (Downs et al., 2015)	Shallow water. 230 people in water prior to sampling	75 - 95 µg L-1
Caneel Bay 2007	Shallow water. 17 swimmers in 48hr prior to sampling	Not detectable
SOUTH CAROLINA: NORTH WESTERN ATLANTIC		
Folly Beach S Carolina 2010 (Bratkovics and Sapozhnikova, 2011)	4 sites. Shallow water 1.5-2m from water line	10 – 2,013 ng L-1
S Carolina local Beach Front Park (Bratkovics et al., 2015)	Shallow water	max 2,200 ng L-1
CANARY ISLANDS: EASTERN ATLANTIC (SÁNCHEZ RODRIGUEZ ET AL., 2015)		
Gran Canaria Island May to Oct 2011	3 'semi-enclosed' beaches at 1 - 1.5m depth	12.7 – 3,316 ng L-1
Gran Canaria Island May to Oct 2011	3 'open' beaches at 1 - 1.5m depth	<1.4 - 182.6 ng L-1

SITE	SAMPLING LOCATION	OXYBENZONE CONCENTRATION
CHINA		
Hong Kong (Tsui et al., 2017)	4 locations; wet and dry seasons	12.9 - 31.9 ng L-1
JAPAN		
Okinawa (Tashiro and Kameda, 2013)	Shallow water 300-600m from beach	0.4 - 3.8 ng L-1
HAWAII (DOWNS ET AL., 2015)		
Oahu Island; Maunalua Bay May 2011	4 sites; 35cm depth	Detectable >100 ng L-1; below quantitative range of 5 µg L-1
Maui Island: Kapalua Bay May 2011	1 site; 35cm depth Often >500 swimmers	1 site: 19.2 µg L-1 Detectable >100 ng L-1; below quantitative range of 5 µg L-1
PALAU (BELL ET AL., 2017)		
Palau: Jellyfish Lake water. Jan 2016	8 samples. Tourist location: visited by tens of thousands of people. Marine lake water	4.12 - 10.2 ng L-1 in 3 samples. Not detectable in 4 samples; present but not measurable in 1 sample.
Palau: Jellyfish Lake, from inlet by tourist dock: 8 samples Jan 2016.	8 samples. Marine lake water by tourist dock	4.99 - 5.36 ng L-1 in 2 samples. Not detectable in 6 samples
Palau: Outside Jellyfish Lake near outside dock: Jan 2016.	4 samples. Not a swimming area	Not detectable in any of 4 samples
Palau: Ngermeuangel Lake water. Jan 2016.	8 samples. Marine lake water. Intended as control site away from tourist areas	4.4 to 18.5 ng L-1 at 3 sites; not detected at 9 sites
Palau: lagoon outside entrance to Ngermeuangel Lake: Jan 2016.	4 samples. Not a swimming area	Not detectable in any sample
Palau: Ocean outside Lighthouse Reef	4 samples: control site	Not detectable in any sample

3.1. US VIRGIN ISLANDS (Caribbean)

Initial analysis of seawater at Trunk Bay on St John Island showed levels of oxybenzophenones in the water column between 1 ppm and 90 parts per billion (ppb) (Downs et al., 2011). Further sampling revealed oxybenzone levels of 1.395 ppm (mg L-1) at a site near the edge of the Trunk Island coral community (Downs et al., 2015). A sampling site 93 m east of this site contained 580 ppb (µg L-1) of oxybenzone. Samples were collected at 11:00–11:24 h with more than 180 swimmers in the water and 130 sunbathers on the beach within 100 m of the two sampling sites. Oxybenzone levels at Hawksnest Bay (230 swimmers) were lower (75-95 ppb) and were undetectable at Caneel Bay (17 swimmers over 48hr).

3.2. SOUTH CAROLINA (SE coast USA)

Simultaneous determination of seven of the most widely used organic UV filter compounds in surface waters of Folly Bay in summer 2010 found concentrations of individual components between 10 to 2013 ng L-1 (Bratkovics and Sapozhnikova, 2011). In general, the sites with the highest use were associated with higher concentrations of sunscreen compounds. Oxybenzone and octocrylene (OC) were found in the highest concentrations, up to 2013 ng L−1 and 1409 ng L−1, respectively. Concentrations for avobenzone, octinoxate, and padimate-O were 62–321 ng L−1, 30–264 ng L−1 and <1–111 ng L−1, respectively. Dioxybenzone and sulisobenzone were not detected in any samples from the four sites (method reporting limits 1 ng L−1 (i.e. one part per trillion) (Bratkovics and Sapozhnikova, 2011).
Monthly monitoring over a year at six coastal South Carolina sites found the highest concentrations measured were > 3700 ng OC L-1 and ~ 2200 ng oxybenzone L-1 at a local beach front park, where beach use was the greatest (Bratkovics et al., 2015).

3.3. MAJORCA ISLAND (Mediterranean Sea - Spain)

Chemical analysis of the surface nearshore waters of three areas around Majorca Island showed that four of the main chemicals used in commercial sunscreens were present in surface waters, with the highest concentrations measured in the unfiltered fraction of the surface microlayer, as follows: oxybenzone: 580 ng L-1; 4-MBC: 11.3 ng L -1, titanium: 38 µgL -1 and zinc: 10.8 µgL -1) (Tovar-Sánchez et al., 2013). Levels of these chemicals co-varied throughout the day reaching the highest concentrations between 14:00 and 18:00 h, a few hours after the maximum numbers of beachgoers occurring (around noon), and when sunlight radiation is maximum and sunscreen application is expected to be at its highest level (Tovar-Sánchez et al., 2013).

3.4. GRAN CANARIA (Eastern Atlantic - Spain)

Eight commonly used UV filters (benzophenone-3 (BP-3), (OC), octyl-dimethyl-PABA (OD-PABA), EHMC, HMS, butyl methoxydibenzoyl methane (BMDBM), 4-MBC and DHHB) were monitored in samples from six beaches around the island. With the exception of OD-PABA, all UV filters were detected in the samples collected, and 99% of the samples contained UV filters. Concentration levels varied and reached up to 3316 ng L-1 for benzophenone-3 (Sánchez Rodríguez et al., 2015).

3.5. HONG KONG (West Pacific)

Tsui et al. (2017) investigated the occurrence and distribution of seven commonly used organic UV filters in corals, seawater and sediment from the eastern Pearl River Estuary, South China Sea. Three of the sites had coral cover over 65% and were hotspots for snorkeling, scuba diving and swimming. The remaining site was dived less frequently due to strong currents and low (<10%) coral cover and diversity. Samples of five hard coral species (Platygyra acuta, Porites sp., Pavona decussata, Acropora valida and Favites abdita) and corresponding water column and sediment samples were collected from four locations in both the wet (August 2015) and dry (April 2015) seasons by scuba divers. Samples were stored on ice during transportation to the laboratory, where the tissues were separated from their skeletons. Both coral and sediment samples were freeze-dried and homogenized by mortar and pestle for chemical analysis to determine the exposure of the corals to different contaminants. The potential risk posed to the corals due to this exposure was evaluated using currently available published information on mortality (LC50) and deformity (EC50) of coral planulae (Downs et al., 2014, 2015), as well as bleaching in hard corals due to exposure to UV filters (Danovaro et al., 2008) (see Section 4).

Five of the seven filters were detected in the coral tissues: benzophenone-1 (BP-1), oxybenzone, benzophenone-8 (BP-8), OC, and OD-PADA. The highest detection frequencies (>65%) and concentrations (31.8 ± 8.6 and 24.7 ± 10.6 ng g-1 wet weight respectively) were found for oxybenzone and BP-8. Significantly higher concentrations of BP-3 were observed in coral tissues during the wet season, perhaps due to increased coastal recreational activities at this time leading to greater discharge of sunscreen agents. The risk assessment indicated that over 20% of coral samples from the study sites contained BP-3 concentrations exceeding the threshold values for causing larval deformities and mortality in the worst-case scenario. Higher probabilities of negative impacts of BP-3 on coral communities are predicted to occur in the wet season (Tsui et al., 2017).

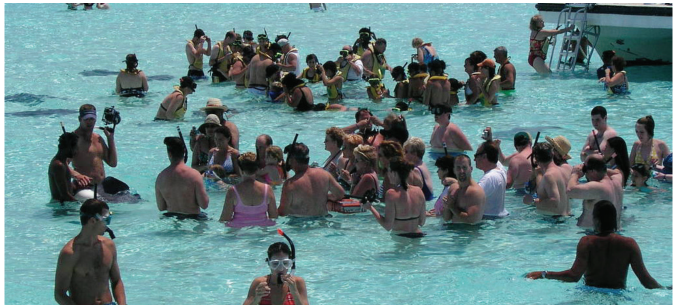
3.6. HAWAI'I (Pacific Ocean)

Seawater samples were collected in public swimming areas at five sites in Maunalua Bay (Downs et al., 2015). Four sites had detectable levels of oxybenzone ([100 pptrillion; ng L-1) but were below the quantitative range of measurement (5 ppb (µg L-1). The fifth site contained measurable levels of oxybenzone (19.2 ppb (µg L-1). Samples were also collected at two sites along the northwest coast of Maui Island. Kapalua Bay is a protected cove and has a public beach that can often receive 500 swimmers per day in the peak tourism season. The Kapalua sample, taken immediately above remnants of a coral reef, had detectable levels of oxybenzone but was below the quantitative range of measurement (5 ppb (µg L-1). Kahekili Beach Park is a public beach that also serves visitors from a number of nearby hotels and resorts (Downs et al., 2015). Unlike Kapalua, Kahekili is an exposed shoreline and not protected within a bay, and retention time of contaminants is thought to be minimal because of the prevailing currents. The Kahekili sample had detectable levels of oxybenzone but was below the quantitative range of measurement (5 ppb). Kahekili is a heavily visited beach and had 71 swimmers within 200 m of the sampling site at the time of sampling (11:45 h).

3.7. PALAU (Pacific Ocean)

Bell et al (2017) analysed 22 chemicals in water and 23 in sediment and jellyfish samples in Jellyfish Lake (a tourist site) and three comparison sites (non-tourist sites) in Palau. The chemicals were either found directly in sunscreen formulations or products (metabolites) that the body makes from one of the sunscreen chemicals. These were found to be widespread in Jellyfish Lake and also present in the sites presumed to be 'pristine' with little human use. In general, water samples had low levels of sunscreen compounds, while jellyfish tissues and sediment had relatively higher levels of these compounds and metabolites, a potential indication of bioaccumulation of these chemicals (Bell et al., 2017).
Jellyfish Lake had the highest concentrations of the chemicals, suggesting that these are entering the environment by washing off tourists and that jellyfish may then be absorbing and metabolizing oxybenzone (Bell et al., 2017). Compounds were also detected in Ngermeuangel Lake, where there are no visitors. Bell et al (2017) suggest that the sunscreen compounds here may have come from effluent/sewer leaking from the nearby population centre into the adjacent watersheds. Bell et al (2017) conclude that the presence of sunscreen chemicals in both the jellyfish and the sediments is cause for concern. The Golden Jellyfish medusa stage is relatively short lived, with a life span of about 6 months and this stage may not live long enough to be directly affected; however their benthic polyp stage, critical to their life cycle, could be affected (Bell et al., 2017).

Snorkeling in Jellyfish Lake - Palau. Photo: ©



Extreme tourism in the Caribbean

4/ IMPACT OF SUNSCREENS ON CORALS AND REEF BIOTA – EVIDENCE TO DATE

Given the growing concern that some sunscreen ingredients may be having a negative impact on corals and other reef biota, investigations are underway and this section summarises some of the key work. The experiments carried out so far have used concentrations of sunscreen and sunscreen components that are significantly higher than is generally found in the marine environment (Leonard pers. Comm. 19-12-2017). For example, Danovaro et al (2008) used sunscreen concentrations of 10, 33, 50, and 100 μL/L seawater. Assuming that the density of the compounds is close to 1, then μL/L equals mg/L and the experimental concentrations were approximately 10, 33, 50 and 100mg/L. This is markedly higher than most of the concentrations shown in Table 1, the exceptions being the swimming 'hotspots' where hundreds of people are in the water as in the US Virgin Islands (Downs et al., 2015). Further research is thus needed before firm conclusions can be reached.

4.1. Effect of sunscreens and selected sunscreen components on coral fragments

The first investigations into the possible harmful effects of sunscreen on corals (Danovaro et al., 2008) consisted of a series of experiments conducted in four coral reef areas: Siladen, Celebes Sea (Indonesia); Akumal, Caribbean Sea (Mexico); Phuket, Andaman Sea (Thailand), and Ras Mohammed, Red Sea (Egypt). Nubbins of Acropora were collected and washed with virus-free seawater prior to being transferred to polyethylene bags for in-situ incubation with commercially available sunscreens and also with individual sunscreen components. Concentrations used were 10, 33, 50, and 100 μl L-1 seawater (i.e. between 10 to 100 ppb). Additional experiments were performed with two other hard corals, Stylophora pistillata and Millepora complanata.

Corals were incubated at the same depth as the donor colonies at in situ temperatures. Samples of seawater were collected at 12-hr intervals and analysed for virus-like particles and zooxanthella released by the corals. At the end of the experiments, samples of coral tissue were preserved and stored for zooxanthellae counts. Levels of bleaching were quantified using colorimetric analysis of digital photographs taken at the beginning of the experiments and after various times of treatment (Danovaro et al 2008).

The experiments were carried out using aliquots of sunscreens at final quantities of 10, 33, 50, and 100 μL/L seawater. In all replicates and at all sampling sites, addition of commercial sunscreen resulted in the release of large amounts of coral mucous (composed of zooxanthellae and coral tissue) within 18–48 hr, and complete bleaching of hard corals within 96 hr. Bleaching was faster in corals subjected to higher temperature, suggesting synergistic effects with heat. The zooxanthellae released from treated corals had lost photosynthetic pigments and membrane integrity. In contrast, zooxanthellae membranes from untreated corals were intact (Danovaro et al. 2008).

Among the individual sunscreen organic compounds that were tested, butylparaben, ethylhexyl methoxycinnamate, oxybenzone and 4-MBC caused rapid and complete bleaching even at the lowest concentrations used in the experiments (10 μL L-1). Conversely, all other compounds tested (i.e., OC, ethylhexylsalicylate and 4-tert-butyl-4-methoxydibenzoylmethane (avobenzene)) and the solvent propylene glycol, which is also present in sunscreen formulations, had a minor effect or no effects when compared with controls (Danovaro et al., 2008).

After the addition of sunscreens, viral abundance in the surrounding seawater increased significantly, reaching values greater by a factor of 15 than in controls. As with the bleaching, this occurred even at the lowest concentration used in the experiments (10 μL L-1). The viruses were considered to have been released from the corals or their symbiotic zooxanthellae, and virus-like particles were found around and inside the zooxanthellae. In contrast, addition of organic nutrients without UV filters or preservatives (as controls) did not result in a significant increase in viral abundance. Viruses were not found inside or outside the zooxanthellae (Danovaro et al., 2008).

These findings correlate with previous research by Danovaro and Corinaldesi (2003) which investigated the effects of cosmetic sun products on viral abundance and bacterial activity in surface seawater samples from Ancona, northern Adriatic Sea. They tested sunscreen containing both chemical and physical sun-blocking agents (Ambre solaire Mexoryl sx 7, Laboratoires Garnier) and a solar oil without UV protection (Bilboa Sun Oil, Cadey) and found that sunscreen supplementation induced the lytic cycle in a large fraction of total bacterial abundance (13-24% of bacteria, at low and high concentrations, respectively), whereas solar oil had a lower impact (6-9%).

Danovaro et al (2008) concluded that sunscreens, by promoting viral infection, potentially play an important role in coral bleaching. However, Leonard (personal communication 19-12-2017) suggests that there is insufficient evidence at this stage to conclude that the increase in virus density causes bleaching. He points out that the coral microbiome is a complex ecosystem of bacteria, archea, fungi and viruses whose respective population growths are under subtle controls. When this ecosystem is disorganized by high concentrations of an external compound it is to be expected that certain organisms (here the virus) will take advantage of the stress situation and multiply when the other partners are weakened (Leonard; personal communication 19-12-2017).

In experiments designed to more closely mimic conditions on the reef, coral nubbins of Stylophora pistillata were exposed for 5 weeks to low concentrations of UV filters in 15 litre aquaria, using a closed-circuit system with weekly seawater renewal. PSII photosynthetic efficiency of the symbiotic micro-algae was monitored using PAM (Pulse Amplitude Modulation), to predict the sublethal endpoint of coral bleaching (Fel et al., 2017). This study found that the organic UV filters Mexoryl SX, Mexoryl XL, Ethylhexyltriazone and OC did not induce coral bleaching nor reduce the photosynthetic efficiency of the symbiotic micro-algae at nominal concentrations above those reported in natural seawaters (Giokas et al., 2007), and avobenzone showed an effect only at the highest nominal concentration (5,000 µg L-1). The mineral UV filter zinc oxide had no effect at 10 µg L-1 but altered the PSII at 100 µg L-1 and induced coral bleaching at 1,000 µg L-1 (nominal concentrations with analytical control).

4.2. Effects of benzophenones on coral planulae and in vitro coral cells

Downs et al. (2014; 2015) examined the effects of benzophenone-2 (BP-2) and oxybenzone on planula larvae of Stylophora pistillata at the Inter-University Institute of Marine Sciences (IUI) in Eilat, Israel. Planulae were collected using light traps and exposed to BP-2 and oxybenzone during four different time-period scenarios. Six concentrations were used for the experiments, ranging from 246 mg L-1 (ppm) to 2.46 µg L-1 (ppb) for BP-2 and 228 mg L-1 (ppm to 2.28 µg L-1 (ppb) for oxybenzone. Histopathology and cellular pathology, planula morphology, coral bleaching, DNA damage and planula mortality were then measured.

In some experiments, coral cells in culture (calicoblasts) were also used as a surrogate for coral planulae. Coral cells from Stylophora pistillata were used in the BP-2 experiments and this species together with Pocillopora damicornis, Montastraea annularis (valid name now Orbicella annularis), Montastraea. cavernosa, Porites astreoides and Acropora cervicornis were also used to investigate the effects of exposure to oxybenzone. Corals for the cell analysis were obtained from various sources and maintained in glass aquaria with custom LED lighting (Downs et al., 2015).

The results studies were consistent with the morphological observation by Danovaro et al. (2008). In the light, oxybenzone caused injury directly to the zooxanthellae. Under dark conditions, bleaching resulted from the symbiophagy of the zooxanthellae, a process whereby the coral gastrodermal cell 'digests' the zooxanthella (Downs et al. 2009). In addition, planulae exhibited an increasing rate of coral bleaching in response to increasing concentrations of both BP-2 and oxybenzone.

The Downs et al (2014; 2015) study showed that both BP-2 and oxybenzone are photo-toxicant, with adverse effects exacerbated in the light versus in darkness. The lethal concentration 50 (LC50: standard measure of toxicity: the concentration of the substance that causes the death of 50% of the test subjects) of planulae exposed to oxybenzone in the light for an 8- and 24-h exposure was 3.1 mg/L and 139 µg L-1, respectively. The LC50s for oxybenzone in darkness for the same time points were 16.8 mg L-1 and 779 µg L-1.

Whether in darkness or light, both benzophenones induced coral planulae to transform from a motile planktonic state to a deformed, sessile condition. Downs et al (2015) conclude from these experiments that oxybenzone is a skeletal endocrine disruptor, inducing 'ossification' of the planula and encasing the entire planula in its own skeleton.

It was also shown that both chemicals are genotoxic to corals, exhibiting a strong positive relationship between

DNA-AP lesions and increasing concentrations. BP-2 exposure in the light induced extensive necrosis in both the epidermis and gastrodermis, and in the dark, it induced autophagy and autophagic cell death (Downs et al., 2015). Monitoring at Trunk Bay (US Virgin Islands) showed minimal recruitment/survival of juvenile coral and lack of regeneration from experimentally induced lesions in established colonies of Porites astreoides over a 5-year period (Downs et al., 2011). Downs et al. (2011) found levels of oxybenzophenones in the water column at this site to be between 90 ppb and 1 ppm and suggested that the severe ecological degradation of the reefs in this area was caused by sunscreens from recreational swimmers, given that other forms of land-based or marine-based pollution are limited (Downs et al., 2015).

4.3. Effects of organic and mineral UV sunscreen filters on coral larvae

Sharp et al., (2017 unpublished; personal communication 09-01-2018) have provided information on the effects of sunscreens on coral larvae in a technical report. In these experiments, the effects of two sunscreens being developed as 'reef-friendly' products and one 'reef friendly' sunscreen ingredient (titanium dioxide) were compared with three leading commercially available sunscreen brands. The commercial sunscreens contained the active ingredients oxybenzone, avobenzone, octinoxate, octisalate, homosalate, and OC in varying concentrations as UV filters while the 'reef-friendly' sunscreens contained non-nano titanium dioxide as the UV filter.

Coral larvae released from adult colonies of Porites astreoides collected from reefs in the Florida Keys were exposed to each of the 5 sunscreen formulations and the one 'reef-friendly' ingredient (titanium dioxide). The three sunscreen/seawater concentrations tested were: 100µl L-1 (which corresponds to the maximum concentration tested by Danovaro et al. 2008), 1µl L-1 and 0.01µl L-1. The same procedures were followed for the controls, with sunscreen being replaced by the same volume of fresh seawater. Treatment order/placement was randomized and color-coded to enable blind scoring (Sharp et al., 2017).

In this study, none of the treatments (either 'reef friendly' or commercial sunscreens) resulted in significant larval mortality compared to the controls. The highest concentration (100µl L-1) of one of the commercial brands (Brand 1, containing 7.5% octinoxate, 5.0% octinosalate, 4.0% oxybenzone) significantly inhibited larval settlement but neither of the 'reef friendly' sunscreens significantly decreased larval settlement when compared to the controls.

Blum (personal communication 18-01-2018) has pointed out that testing finished products can be difficult. Some sunscreen formulas are polymer-based, with the result that the active ingredients become suspended at the surface or on the sides of test containers rather than dispersing in the water. They eventually disperse, releasing the chemical compounds from the polymer beads, but this is unlikely to happen during the initial 24-48 hour period. Water in oil emulsions or purely oil based formulas do not disperse well either and these factors need to be taken into consideration when conducting tests (Blum, personal communication 18-01-2018).

4.4. Direct and indirect effects exposing reef biota to sunscreens

McCoshum et al (2016) investigated the effects of sunscreens on lab-reared organisms commonly found in shallow coral reef ecosystems, including included flatworms (Convolutriloba macropyga) with symbiotic algae, soft corals (Xenia elongata), glass anemones (Aiptasia spp.) and diatoms (Nitzschia spp.). The organisms were grown together in a 208 litre, established multi-species aquarium. Experiments were conducted using artificial saltwater to which was added a commercial sunscreen ('Equate 50 SPF sunscreen' containing homosalate, oxybenzone, OC, octisalate, and avobenzone as active ingredients), following the methodology described by Danovaro et al., (2008).

All test organisms subjected to sunscreen-contaminated water showed reduced population growth compared to control groups, suggesting organisms near populated and common marine tourist destinations, where sunscreen contamination is expected, could be at risk of population and colony decline (McCoshum et al., 2016). The tested concentrations were 0.26 ml L-1, 0.026 ml L-1 which are equivalent to approximately 260 mg L-1 and 26 mg L-1 (ppm) and are very high compared to the levels of sunscreen generally found in natural waters and in Jellyfish Lake, Palau where Bell et al., (2017) levels of zero to 1205 ng L-1 (parts per trillion) of UV filters (see Table 1 for oxybenzone).

Nominal sunscreen contamination also affected the behaviour of the study organisms. Although motile flatworms did not appear to avoid sunscreen when given a choice of sunscreen-contaminated and uncontaminated areas, they did select light areas in uncontaminated water and dark areas in sunscreen-contaminated water.

The lack of avoidance of the sunscreen by flatworms suggests these organisms, and potentially other motile biota, will not avoid environments polluted by sunscreen.

Soft corals showed a reduction in polyp pulses per minute in sunscreen contaminated arenas. The observed reduction of pulses per polyp, and the change in flatworm light preference suggest that the sunscreen is causing some stress, but the mechanisms involved are unknown (McCoshum et al., 2016). Eyal (personal communication 16-01-2018) suggests that reduced pulsations and light avoidance might be stress responses due to oxygen radicals and/or malfunction of the photosynthetic apparatus of the symbionts.

4.5. Effects of sunscreen filters on sea urchin development

Corinaldesi et al (2017) compared the effects of different sunscreens on embryonic and larval development of the sea urchin Paracentrotus lividus, a key species of coastal ecosystems of the Mediterranean Sea and Eastern Atlantic Ocean and one of the most common model organisms for ecotoxicological studies. Whilst this sea urchin is found on coral reefs, the study is relevant because it compares the impacts of 'standard' and 'eco-friendly' products.

Two widely used commercial sunscreens in Europe and the USA were used, with one whose ingredients had been patented as eco-friendly (Danovaro et al., 2014). Both the Europe (Sunscreen A) and USA (Sunscreen B) products contained organic and inorganic filters (titanium dioxiste nanoparticles) already reported to affect marine organisms (Díaz-Cruz and Barceló, 2009; Manzo et al., 2013; Minetto et al., 2014). The eco-friendly sunscreen did not have these compounds or they were replaced by other ingredients (Corinaldesi et al., 2017). Sunscreen B, containing oxybenzone, homosalate and preservatives, had the strongest impact: it caused anomalies (mostly represented by development block or cell necrosis) in 100% of embryos after 24h of treatment at the maximum sunscreen concentration of 50 μL L−1 (equivalent to about 50mg L-1). The observed impact was dose dependent with alterations to development seen in half of the embryos treated at the lowest sunscreen concentrations. Sunscreen A had a lower impact, affecting the development of one third of the embryos of P. lividus at all concentrations (Corinaldesi et al., 2017). In contrast, the effects of the eco-friendly Sunscreen C were indistinguishable from the control after 24h of treatment.

An additional set of experiments revealed that the eco-friendly sunscreen had a similar efficacy to sunscreens A and B in protecting human fibroblasts from UVA radiation (Corinaldesi et al., 2017). These findings suggest that development of sunscreens that are both effective and environmentally friendly is potentially feasible.

4.6. Toxicity effects of inorganic UV filters

Most commercial sunscreens with inorganic filters employ materials with nanoscale dimensions so that the products are both transparent and smooth when applied to the skin (Lewicka et al., 2013). These are generally considered to be safer than chemical UV filters. However, certain types of titanium dioxide and zinc oxide nanoparticles produce reactive oxygen species (ROS) under UV illumination (Lewicka et al., 2013). ROS are a group of free radicals, reactive molecules and ions derived from oxygen, that are extremely harmful to organisms at high concentrations and that can induce 'oxidative stress' in cells that can damage health and ultimately cause death (Sharma et al., 2012).

Titanium dioxide nanoparticles used in sunscreen are coated with silica, magnesium, or aluminium to eliminate UV reactivity. It has been shown that, in this form, ROS production is low when the nanoparticles are exposed experimentally to ultraviolet light (Lewicka et al., 2013). In contrast, zinc oxide nanoparticles derived from the same sunscreens do not have coatings and produced substantial amounts of ROS under UVA illumination, and could thus potentially cause high levels of oxidative stress in marine organisms (Tagliati personal communication 22-12-2017). People are also less well protected from the adverse effects of sun exposure if photoactive nanomaterials are used in sunscreens. Lewicka et al (2013) suggest that sunscreens with zinc oxide should have nanoparticle surface coatings to limit ROS generation under ultraviolet illumination.
Yung et al., (2014) confirmed that exposure of marine algae and invertebrates to zinc oxide nanoparticles can induce a wide range of toxic effects, both acute lethal and chronic sublethal. They also noted that toxicity can occur in certain species when exposed to environmentally realistic concentrations. Wong et al., (2013) showed that in general, zine oxide nanoparticles ZnO were more toxic to algae (marine diatoms Skeletonema costatum and Thalassiosia pseudonana) than zinc oxide, but relatively less toxic to crustaceans (Tigriopus japonicus and Elasmopus rapa) and the medaka fish (Oryzias melastigma).

Whilst research shows that nanoparticles are potentially hazardous, Hanna et al (2013) point out that many of the studies used short-term laboratory exposure tests with relatively short-lived species, and may not be so relevant in the marine environment. They thus coupled laboratory-based studies of mussels (Mytilus galloprovincialis) with modelling in order to simulate the effects of nanoparticles on mussel populations using results from the studies on individuals. Mussels exposed to zinc oxide nanoparticles for 12 weeks had increased respiration rates and concentrations of zinc in their tissues. These impacts were related to decreases in growth and survival, and suggested that mussels were expending energy to combat the effects of excess environmental zinc but were unable to meet these demands at the highest exposure concentration of 2 mg L-1. Large mussels seemed to tolerate higher concentrations of zinc oxide nanoparticles for longer periods of time, as survival was higher for large than for small mussels after 6 and 12 weeks of exposure. Hanna et al (2013) noted that, whilst the results of exposure to 0.1–2 mg L-1 zinc oxide nanoparticles in seawater indicated that these are toxic to mussels, these levels were unlikely to be reached in natural marine waters. However, these chemicals may accumulate in the tissues of long-lived organisms following exposure.

Fel et al., (2017) exposed coral nubbins (Stylophora pistillata) for 5 weeks to the mineral UV filter zinc oxide at nominal concentrations above those reported in natural seawaters. They found that this treatment had no effect at 10 µg L-1 but altered the photosynthetic efficiency at 100 µg L-1 and induced coral bleaching at 1000 µg L-1.

Studies on the impacts of titanium dioxide nanoparticles on tropical corals are currently in progress by Tagliati et al. (unpublished, personal communication 19-12-2017), involving both nanoparticles alone and coated nanoparticles mixed with cosmetic ingredients to simulate a commercial-sunscreen composition. Results so far indicate that titanium dioxide nanoparticles used as UV filters in sunscreen do not necessarily have negative effects on growth and photosynthetic activity of coral symbiotic algae (genus Symbiodinium) at concentrations estimated to be present in sewage. Nevertheless, it is important to note that a full evaluation is still lacking (Tagliati et al. unpublished, personal communication 19-12-2017).

5/ KEY RESULTS AND CONCLUSIONS

It is known that significant amounts of sunscreen wash off into the sea and, in controlled experiments, some sunscreen chemical UV filters are toxic and have negative effects on corals and other marine life even at concentrations as low as 62 parts per trillion. Oxybenzone has been identified as the main substance of concern. Organic UV filters have been reported to induce acute toxicities, developmental toxicities and reproductive toxicities to different organisms. Some studies have shown that sunscreen ingredients promote viral infections in bacteria and zooxanthellae, causing coral bleaching.

Although estimates of the quantities of UV filters in sea water vary, concentrations have been found in sea water that are considered to be 'environmentally relevant' and pose a threat to reefs. Downs (2016) notes that "emerging research is showing that oxybenzone concentrations on nearshore reefs around the world are commonly between 100 parts per trillion and 100 ppb — well within the range of being a significant environmental threat". Furthermore, Downs et al. (2015) suggest that the threat of oxybenzone to corals and coral reefs from swimmers and point and non-point sources of waste-water could more extensive than just a few meters surrounding the release area. For example, in Okinawa, Tashiro and Kameda (2013) demonstrated that oxybenzone contamination from beaches can travel over 0.6 km from the pollution source.

The main impacts of chemical filters (the primary cause of concern), under certain circumstances, appear to be:

5.1. Bleaching of hard corals

Laboratory and controlled in-situ studies using fragments and cells from various species of hard coral have shown bleaching in response to sunscreens. This may be for a number of reasons:

- In darkness, bleaching results from the symbiophagy of the zooxanthellae; a process whereby the host cells 'eats' the zooxanthellae (Downs et al. 2009).
- In the light, BP-2 causes damage directly to the zooxanthellae, independent of any host-regulated degradation mechanism. Danovaro et al (2008) showed that bleaching occurred as a result of organic UV filters inducing the lytic viral cycle in symbiotic zooxanthellae with latent infections.
- Oxybenzone induces coral bleaching by lowering the temperature at which corals bleach when exposed to prolonged heat stress. This means that the impact of increasing sea surface temperatures may be exacerbated by the presence of sunscreen contaminants and the resilience of corals to climate change undermined.

5.2. Damage and deformation of coral larvae (planulae)

Some sunscreen chemicals, in certain situations, cause coral larvae to stop swimming, change shape and ultimately die. Oxybenzone has been shown to be an endocrine disruptor, causing the outer epidermal cells of coral larvae to turn into skeleton at the wrong stage in their development (Downs et al., 2015). Studies on other organisms have similarly shown that exposure to sunscreen UV filters such as benzophenones, camphor derivatives and cinnamate derivatives induce various endocrine disrupting effects (Kim et al, 2014; Wang et al., 2016). However, Wang et al (2016) note that the effective concentration (26 µg L-1) of oxybenzone, at which reproduction of the Japanese medaka fish was significantly decreased, was at least a couple of orders of magnitude greater than those detected in the ambient environment (e.g. 125 ng/ L-1 in a Swiss Lake).

5.3. Damage to coral DNA and reproductive success

Oxybenzone has been shown to be as genotoxic, meaning that it damages coral DNA, which can reduce a coral's lifespan and immunity to disease, as well as disrupting normal development and reproduction.

6/ KNOWLEDGE GAPS AND FURTHER RESEARCH REQUIRED

There are concerns that experiments undertaken to date have been largely ex-situ, and mainly at subcellular, cellular and organism level, with very few studies of the wider impacts of sunscreens and their UV filters. There is a lack of firm evidence of widespread negative impacts at reef community and/or ecosystem level. The evidence available may not properly reflect conditions on the reef, where pollutants may rapidly disperse and be diluted. Concentrations of UV filters used in experimental work have generally been higher than those likely to be encountered in the reef environment, although no study has assessed the levels of these chemicals in the tissues of long-lived species.

Further research needs include:

- Determining concentrations of different sunscreen components in the water column, sediment and biota in different locations, comparing sites that are heavily visited and/or affected by coastal run-off with those that are more remote.
- Investigating the effects of sunscreen pollutants at reef community and ecosystem level. Downs et al. (2015) linked some of the pathologies described above with reef degradation in the US Virgin Islands, and it is possible that whole-organism changes could affect nutrient and food web dynamics, biodiversity and community composition, habitat structure, and disease dynamics (Prichard and Granek 2016).
- Exploring links between organismal or subcellular/ cellular-level biomarkers since studying pharmaceutical effects on biodiversity/ community composition in complex systems is difficult (Prichard and Granek, 2016). Studies should focus on locations with high visitor numbers where sunscreens are suspected as a possible cause of coral bleaching or reef decline.
- Investigating UV filter toxicity in relation to ocean warming and acidification predictions, to assess the effects of global climate change on this issue. While sunscreen exposure might not cause evident stress in healthy corals, the response could be different if corals are already facing stressful conditions. Studies carried by Danovaro et al. (2008) showed that bleaching of corals and coral cells exposed to organic UV filters was faster at higher temperatures.
- Researching the extent and significance of bio-concentration and bio-accumulation of organic UV filters, and the consequences of long-term, chronic exposure to sunscreen pollutants.
- Studying the effects of mixtures of pharmaceutical and personal care products (PPCP) which would provide a more realistic picture of ecological risks, since organisms are rarely, if ever, exposed to single contaminants.
- Identifying those ingredients that are safe and those that pose a realistic threat to marine ecosystems. The organic UV filters which are widely recognised as being ‘of concern’, but there are other ingredients of sunscreens that could have impacts but about which little is known. Minetto et al. (2014) in at review of ecotoxicity of nanoparticle forms of zinc oxide and titanium dioxide and other mineral nanoparticles concluded that it was difficult to determine a clear framework about nano-ecosafety to saltwater organisms, because of fragmentary and incomplete information and sometimes profiles of limited ecological significance.

It is essential that reef scientists work with industry to ensure that all sunscreen ingredients are subjected to toxicological testing in a standardised manner. Some work is already underway to develop simple toxicity indicators for sunscreens, using coral nubbins exposed for periods of 5 weeks to low concentrations of UV filters (Fel et al., 2017). Given that different sites are exposed to different pollutants (as found in the US Virgin Islands), each site must be investigated independently (Downs et al., 2011). A more rigorous approach of this nature would enable managers to pinpoint where sunscreen pollution is having an impact and to focus attention on remedial measures. Biocides, fertilizers and pesticides leaching from the coast should be included in such studies.

7/ RECOMMENDATIONS

Over-exposure to UV solar radiation is indisputably a human health issue and the development of sunscreens with organic and inorganic UV filters has been a significant factor in reducing risks. However, considering the many stresses already faced by reefs and the current concerns about the toxicity of certain components of sunscreens to corals and other marine organisms, a proactive and precautionary approach is required, especially in areas with high levels of marine-based tourism. The key need is to reduce the amount of harmful sunscreen components that reach the reef environment. This will require the involvement of governments, reef managers, divers, snorkelers and the tourist and pharmaceutical industries.

Although this document focuses on the effects of sunscreen components on reef biota and the wider marine environment, it is important to note that there are also wider concerns about the effects of these chemicals on human health. Krause et al (2012) point out that human exposure to UV-filters in sunscreens is high because they are rapidly absorbed from the skin. They note that oxybenzone has been found in 96% of urine samples in the US and that several UV-filters have been found in 85% of Swiss breast milk samples.

Measures that can be taken include:

7.1. Encouraging the manufacture of reef-friendly sunscreens

IAs with all pharmaceutical products, sunscreen manufacturers are governed by detailed regulations that specify the limits of concentration of allowable UV filters before a sunscreen can be placed on the market.

The regulations vary in different countries. For example, in Europe, Regulation (EC) No 1223/2009 covers assessment of the potential impact of chemicals on human health and Regulation (EC) No 1907/2006 covers the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). The REACH regulation is aimed at ensuring a high level of protection of both human health and the environment and its provisions are underpinned by the precautionary principle (http://eur-lex.europa.eu/eli/reg/2006/1907/2014-04-100). The European Union (EU) lists 28 allowable UV filters, but in the USA, 24 ingredients are covered by the Food and Drug Administration with some of the filters allowed in the EU being excluded (Pirotta, 2015). In Japan and the USA, oxybenzone can be used as an active ingredient at levels of up to only 5-6% but in the EU levels of up to 10% are permitted (Kim et al.,, 2014).

These regulations have been established to ensure human safety and also take environmental concerns into account. It is important that research into impacts of sunscreens on reefs and reef biota continues to feed into the regulatory framework to which manufacturers have to abide. For example, a review by Ruszkiewicz et al., (2017) into neurotoxic effects of active ingredients in sunscreen products advocates revisiting the current safety and regulation of specific sunscreens and investing in alternative UV protection technologies.

7.2. Promoting the use of reef-friendly sunscreens and other methods of UV protection

A growing number of sunscreens are now on the market that are considered to be eco-friendly, safe and non-toxic to corals. Initiatives providing information on these include:

- The Environmental Working Group (EWG) in the USA: a non-profit organisation whose work includes advocacy in the area of toxic chemicals (https://www.ewg.org/sunscreen).
- MarineSafe (www.marinesafe.org): a campaign and certification scheme that aims to reduce the number of toxic chemicals and plastics finding their way into the ocean through replacement, management and user engagement. MarineSafe awards an assurance mark to products formulated without marine-toxic ingredients or interactions, and that are independently tested by certified laboratories to ensure that the finished product is not harmful.

Many local authorities, reef managers, tour organisers, dive companies and non-profit organisations have campaigns on environmental issues ranging from marine litter to protection of endangered species and reef etiquette, and these could be expanded to include sunscreens. A good example of this is the U.S. National Park Service, responsible for managing reefs in South Florida, Hawaii, U.S. Virgin Islands and American Samoa, who have produced an information sheet entitled Protect Yourself, Protect The Reef! The impacts of sunscreens on our coral reefs (noaa.gov/_docs/Site%20Bulletin_Sunscreen_final.pdf). This explains the issues and potential impacts of sunscreens, asking people to use ones that are less likely to damage corals.

Since the first warnings about the dangers of sunscreen toxicity (e.g. Danovaro et al., 2008) the environmental lobby has been highly active in calling for people to buy/use only non-toxic sunscreens. Dive magazines, dive centres and industry bodies such as PADI are also raising the issue and urging people to use eco-friendly products. According to Downs (2016) some resorts and dive shops are even proposing to offer 'coral safe' sunscreen for free to their guests. In this respect it is important that clear and accurate information is provided and that people read the label.

In Bonaire in the Caribbean, a seminar is planned in 2018, to raise awareness about the potential threats to reefs from sunscreens containing oxybenzone (Anon., 2017) and to encourage all retailers of sunscreen products on Bonaire to only sell sunscreens that are free of oxybenzone. Bonaire's economy depends largely on healthy coral reefs and therefore it is not unreasonable to expect local businesses and inhabitants to contribute pro-actively to solving this potential problem.

Although there are still unanswered questions regarding the eco-safety of the many sunscreen products on the market, those with higher levels of toxicity are known. One of the most important messages is to 'Read the Label', and check the ingredients to ensure that the product is 'reef safe' before buying or using it. People are also being urged to consider alternative protective measures, such as use of sun-protective clothing (McCoshum et al., 2016; https://www.ewg.org/sunscreen). By 'covering-up' using appropriate clothing and hats, finding shade and/or avoiding going out in the middle of the day, the need for sunscreens is significantly reduced and there will thus be less pollution. Recent recommendations for reducing sunscreen pollution in Jellyfish Lake (Palau) include similar practical advice and also that people should be educated in the most responsible use of eco-friendly sunscreen, which means application at least 20 minutes before entering the lake (Bell et al., 2017).

7.3. Regulating the sale and use of sunscreens containing toxins

This approach is being taken in Mexico and Hawaii

Mexico: Sunscreens that are known to contain toxic chemicals may not be used at the Xcaret and Xel-Há ecological reserves. At the park entrances, visitors must hand in their non-biodegradable sunscreens for the duration of their visit, and a sample of biodegradable sunscreen is provided; when they leave, their own sunscreen is returned[1]. At Xel-Há this initiative is enshrined in the sites Environmental Management System which covers a number of activities to reduce impacts on the environment. The "chemical-free sunscreen exchange program" at Xel-Há leads to about 100,000 chemical-free sunscreens being exchanged annually with visitors[2]. However, the best information may not be being used: sunscreens containing OC, any of the benzophenones, butyl methoxydibenzoylmethane, hexyldecanol, cetyl dimethicone, methylparaben, polyethylene, propylparaben, and butylcarbamate, are considered to be non-biodegradable and those containing titanium oxide and zinc oxide are considered biodegradable and safe[3] although mineral substances are by definition, non-biodegradable. In addition, the nanoparticle form of zinc oxide is now known to be toxic and not necessarily 'reef friendly'.

Hawaii: Efforts are underway to ban the sale of oxybenzone throughout the state. Legislation stalled in the final days of the 2017 session, but the effort continues, with state lawmakers already starting their push to pass the bill in 2018 (Yoshioka 2017). If this is successful it would ban sale of oxybenzone products but not entirely prevent their use because of international tourists who might carry sunscreens in with them from overseas. This highlights the importance of campaigns to raise awareness amongst visitors.

7.4. Exerting consumer pressure to encourage development and use of eco-friendly sunscreens

The major sunscreen manufacturers may be reluctant to make changes to tried and tested formulations and would probably not do so unless dictated by official regulations or if it becomes more profitable to produce and sell a different product. In this latter respect, consumer demand and consumer/retailer-led campaigns can be

1- http://blog.xcaret.com/en/biodegradable-sunscreen-everything-you-need-to-know/
2- http://www.xelha.com/social-responsability-xelha.php 3- http://blog.xcaret.com/en/biodegradable-sunscreen-everything-you-need-to-know

highly effective forces for change. This is well illustrated by sustainable seafood campaigns around the world that have successfully changed people's outlooks and buying habits and in doing so helped to promote the growth of sustainably-managed fisheries.

7.5. Introducing financial disincentives for manufacture and use of potentially damaging sunscreens

Because the use of UV filters is regulated, it would be possible to tax those considered to be environmentally damaging. This in turn might encourage manufacturers to reduce the amounts that they use and turn to eco-friendly options. Alternatively, if costs were passed on to the consumer, sunscreens containing damaging UV filters would become more expensive, so helping to persuade people to buy eco-friendly sunscreens that they might currently be avoiding due to their higher cost.

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