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防晒剂的海洋环境行为与生物毒性

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摘要: 防晒剂广泛添加于防晒霜中,通过人类的水中娱乐活动和污水处理厂排水等途径进入海洋环境,成为一种新型海洋污染物.由于防晒剂具有用途广泛、持续排放、难降解等特点,其对环境与生态的潜在风险已成为国内外环境领域的研究热点.本文分析了防晒剂在海洋中的迁移、转化和挥发等环境行为,总结了常见无机防晒剂(nTiO₂和 nZnO)以及有机防晒剂(二苯甲酮类、樟脑衍生物类和肉桂酸类)对海洋生物(藻类、贝类、鱼类、珊瑚、海胆等)的毒性效应,包括生长抑制、繁殖抑制、致死和致畸等,并从氧化损伤、神经毒性及内分泌干扰角度分析其内在的毒性作用机制,最后展望了本领域的研究前景与未来的研究方向,为有关防晒剂的科学研究与污染管控提供参考.

关键词:防晒霜;防晒剂;环境行为;海洋生物;生物毒性

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Fate and Toxicity of UV Filters in Marine Environments

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Abstract: UV filters have been widely used in sunscreen products, and they have partially ended up in the marine environment via human recreational activities and sewage treatment plant drainage, becoming one of the emerging marine pollutants. As UV filters have many characteristics, such as extensive use, continuous emissions, and stability, their potential risks to the environment and ecology have become a hot topic in the field of environmental research all over the world. This study analyzed the environmental behavior of UV filters in the ocean, such as migration, transformation, and volatilization. The toxic effects (i. e., growth inhibition, reproductive inhibition, death, and malformation) of the inorganic (mainly nano-TiO₂ and nano-ZnO) and organic UV filters (mainly benzophenones, camphor derivatives, and cinnamic acids) on marine organisms (i. e., algae, seashell, fish, coral, and sea urchin) were summarized. The research also analyzed the inherent toxicity mechanisms from the perspective of oxidative damage, neurotoxicity, and endocrine disability. The prospect and future directions in this field were also discussed. This review provides a reference for scientific research and pollution control related to UV filters.

Key words: sunscreen; UV filters; environmental behavior; marine organisms; biotoxicity

随着人们对紫外线辐射危害的认识不断提高以 及保健意识的增强,使用含有防晒剂的个人护理品 (如防晒霜)抵御紫外线危害已为大众所接受,这一 行为直接导致防晒霜的使用率和生产量迅速增 加[1~3]. 2012年, 防晒系列护肤品市场的销售额已 高达91 亿美元[2]. Gondikas 等指出[3], 仅老多瑙 河景观湖周边的娱乐活动就可带来每年8.1 t 的防 晒霜消费. 防晒剂的大规模生产和大量使用, 极大 地增加了其环境暴露的可能性, 而防晒剂具有持续 排放、难降解等特点, 在环境中易持久存在, 其对 环境与生态的潜在风险已成为国内外环境领域的研 究热点[4]. 值得注意的是, 在防晒霜的生产及使用 过程中, 防晒剂将不可避免地通过多种途径(如污 水处理厂排放、径流输入和水中娱乐活动等)进入 沿海水域[5,6],给海洋生态环境带来不可忽视的威 胁,成为一类新型的海洋污染物.实际上,近年来 的研究表明,海洋水体、沉积物和生物体内均已检测出防晒剂的存在^[7],其污染程度日益严重;防晒剂还可在海洋生物体内累积,并造成众多海洋生物如藻类、贝类、鱼类、珊瑚、海胆等中毒,其海洋生态效应已引起人们的广泛关注^[6].目前国内已有有关防晒剂水环境行为与毒性效应的专门报道^[4].但却缺乏有关防晒剂在海洋环境中迁移、转化和归趋及其海洋生态效应的进展报告.考虑到海水与淡水在成分、离子强度、pH等多项理化条件上具有显著差异,而这些水体性质又与防晒剂在水环境中的挥发、溶解、降解等行为及其水生生物毒性存在密

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作者简介: 朱小山(1977~), 男, 博士, 副研究员, 主要研究方向 为纳米材料的海洋生态毒理学, E-mail: zhu. xiaoshan@ sz. tsinghua. edu. cn 切关联^[8~11],防晒剂的海洋环境效应及其生态风险应有别于其进入淡水环境后的表现,值得进行单独报道.本文在总结防晒剂的物理化学性质基础上,分析了近年来常用防晒剂的海洋环境行为与生物毒性,并展望了本领域的研究前景和未来的研究方向,以期为有关防晒剂的科学研究与污染管控提供参考.

1 防晒剂的理化性质及其在防晒霜中的应用

使用以防晒霜为主的外用防晒产品是目前减小 紫外线伤害的主要方法[12]. 防晒霜可以分为物理 防晒霜和化学防晒霜. 物理防晒霜的主要防晒成分 为无机防晒剂,又名紫外线屏蔽剂.目前,最常用 的两种无机防晒剂是纳米 TiO,(nTiO,) 和纳米 ZnO (nZnO). nTiO₂ 和 nZnO 均属于宽禁带半导体, 其 紫外线屏蔽机理可用固体能带理论解释. 用于防晒 化妆品中的 nTiO₂ 一般为金红石晶型, 其禁带宽度 (Eg) 为 3.0 eV, nZnO 的 Eg 则为 3.2 eV, 它们的紫 外吸收波长分别为 413 nm 和 388 nm. 当受到高能 (hv≥Eg)紫外线照射时, 价带电子可吸收紫外线而 被激发到导带上,同时产生电子-空穴对,从而具有 吸收紫外线的功能[13]. 此外, 当紫外线照射于 nTiO₂ 和 nZnO 时,由于其粒径均小于紫外线的波 长, 粒子中的电子被迫振动(振动频率与入射光频 率相同),成为二次波源,向各个方向发射电磁波, 从而引起紫外光的散射^[14]. 据报道, nTiO₂在 2006 ~2011 年的商业化生产量为5 000 t·a⁻¹, 2011 ~ 2014年则达到了10000 t·a^{-1[15]}, 其中, 化妆品(包 括防晒霜)占据了 70%~80% 的用量[16]. nZnO 颗 粒的年产量达到 550t, 其中 70% 用于化妆品(包括 防晒霜)的生产[17]. 化学防晒霜的主要防晒成分为 有机防晒剂,又名紫外线吸收剂. 欧盟监管并授权 的有机紫外防晒剂有26种,几乎包含了所有广泛 使用并全球认可的防晒剂, 其理化性质见表 1[18]. 防晒霜中常用的有机防晒剂主要有7类,包括二苯 甲酮类、二苯甲酰基甲烷类、樟脑衍生物类、氰基 丙烯酸类、对氨基苯甲酸类、肉桂酸类和苯并三唑 类[19]. 其中, 2-羟基-4-甲氧基二苯甲酮(BP-3)、4-甲基苄亚甲基樟脑(4-MBC)、对甲氧基肉桂酸辛酯 (EHMC)等成分应用最为广泛^[20]. 有机防晒剂的防 晒机理是基于苯环上的羟基氢和相邻的羰基氧之间 通过分子内氢键形成螯合环, 当吸收紫外线后, 分 子发生热振动, 氢键断裂, 螯合环被打开, 形成离 子型化合物, 该化合物处于不稳定的高能态, 通过

自发释放能量,螯合闭环,恢复到稳定的低能态,这样周而复始的"开环-闭环"过程,将吸收的紫外线转化为热能等其他形式的能量释放,从而达到减弱紫外线伤害的目的.此外,有机防晒剂分子中的羰基官能团能被紫外线激发形成烯醇式结构,在羰基和烯醇的异构互变过程中可消耗部分能量[13].

2 防晒剂的海洋环境行为

海洋是防晒剂的重要归宿之一. 如图 1 所示,防晒剂进入海洋环境的途径主要包括直接的水中娱乐活动和间接的径流输运与污水处理厂出水等方式^[5]. 如今,沿海旅游和海上旅游是全球旅游产业中发展最快的部分,旅游人数有望在 2020 年达到15.6 亿^[24]. 海水浴场游泳、潜水、冲浪等海上娱乐活动是防晒霜进入海洋环境最直接的途径^[6,25]. 防晒霜也会在日常清洗中经下水道进入污水处理厂,但污水处理厂无法有效去除高浓度的有机防晒剂^[26],残留防晒剂将伴随污水处理厂出水最终进入海洋. 防晒剂一旦进入海洋, 便受到海流、波浪、高盐度等各种理化因素的影响,发生复杂的迁移、转化,其存在形式、分布状况及毒性效应不断改变. 此外,防晒剂易被海洋生物积累而进入食物链,可能造成更为广泛而复杂的生态效应^[27].

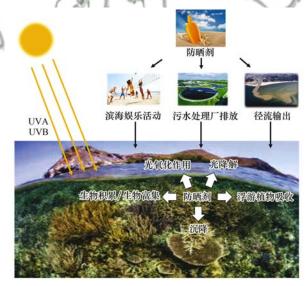


图 1 防晒剂进入海洋环境的途径

Fig. 1 Paths for UV filters to enter the ocean

2.1 防晒剂的迁移

防晒霜中的 nTiO₂ 和 nZnO 等无机防晒剂纳米颗粒,粒径小、疏水性强、难溶于水,且海水的高离子强度使其容易聚集并发生沉积. 但布朗运动、涡流运动以及径流剪切力等可以使部分进入海洋环

表 1 欧盟监管并授权的 26 种有机紫外防晒剂及其理化性质[18]

Table 1 Physicochemical properties of all the UV filters regulated and authorized in the EU

类别	名称	缩写	相对分子质量	$\lg K_{\mathrm{OW}}{}^{1)}$	lg BCF ^{2)[21]}	$\lg K_{\rm OC}^{3)[21]}$	溶解度 /g•L ⁻¹	$\lambda_{ m max}$ /nm
二苯甲酮类	Benzophenone-3	BP-3	228. 24	3. 79	1. 38	3. 10	0. 21	290
	Benzophenone-4	BP-4	308. 31	0.88	/	/	0.65	$240^{[22]}$; 288
对氨基苯甲酸	PABA	PABA	137. 14	0. 83	/	/	915	282
及其衍生物	PEG-25 PABA	P25	277. 41	/	/	/	/	$310^{[23]}$
	Ethylhexyl dimethyl PABA	EHD-PABA	277. 40	6. 15	3. 74	0.38	2. 1×10^{-3}	310
水杨酸盐类	Homosalate	HS	262. 35	6. 16	/	/	0.02	/
	Ethylhexyl salicylate	EHS	250. 34	5.77	/	/	0.028	$240^{\left[23\right]}$
肉桂酸类	Ethylhexyl methoxycinnamate	EHMC	290. 40	5. 80	5. 80	4. 10	0.015	306 ^[22]
	Isoamyl p-methoxycinnamate	IMC	248. 32	4.06	/	/	0.06	/
樟脑衍生物类	Camphor benzalkonium methosulfate	CBM	409. 55	0. 28	/	/	/	288 ^[23]
	Terephtalydene dicamphor sulfonic acid	TDSA	562. 69	1. 35	/	/	0.014	340 ^[23]
	Benzylidene camphor sulfonic acid	BCS	320. 40	2. 74	/	/	0.038	297 [23]
	Polyacrylamido methylbenzylidene camphor	PBC	/	/	/	/	/	1
	4-Methylbenzylidene camphor	4-MBC	254. 37	4. 95	3. 51	3. 89	5. 1×10^{-3}	300 [22]
	3-Benzylidene camphor	3-BC	240. 34	4.49	X	1/	9.9×10^{-3}	292 [23]
三嗪类	Ethylhexyl triazone	EHT	826. 10	15. 53	15	1/1	//	310 ^[22]
	Diethylhexyl butamido triazone	DBT	765. 98	11. 90	/ @ / \	\mathbb{N}	4.6×10^{-7}	// /)][
	Bis-Ethylhexyloxyphenol methoxyphenyl triazine	EMT	627. 81	13. 89	·Vi	$\int \mathcal{N}$	4. 9×10^{-8}	340 ^[22]
苯并三唑类	Drometrizole trisiloxane	DRT	225. 25	9. 79	D 00.	1	1. 3 × 10 ⁻⁵	344; 303
91	Methylene bis-benzotriazolyl Tetramethylbutylphenol	MBT	658. 87	14. 35	all!	3 1	3.0×10^{-8}	340 ^[22]
苯并咪唑衍	Phenyl benzimidazole sulfonic acid	///PBSA	274. 30	0. 01	0.50	2. 46	0. 26	300 [22]
生物	Disodium phenyl dibenzimidazole tetrasulfonate	PDT	674. 60	/ \/	1 1 1 1 1 1 1 1 1 1 1 1 1 1 1 1 1 1 1	1	/	250
二苯甲酰基	Butyl methoxydibenzoylmethane	BMDM	310. 39	2. 41	4. 51	3. 23	0. 037	358 ^[23]
甲烷类	Diethylamino hydroxyl benzoyl hexyl benzoate	DHH	397. 51	6. 93	/	/	9. 5 × 10 ⁻⁴	360 ^[22]
其他	Octocrylene	OCR	361.49	7. 35	/	/	2. 0 × 10 ⁻⁴	300 [22]
	Polysilicone-15	P15	/	/	/	/	/	313

¹⁾辛醇水分配系数($K_{\rm ow}$),平衡状态下有机化合物在N-辛醇和水两相平衡浓度之比;2)生物富集因子(BCF),生物组织(干重)中污染物的浓度和溶解在水中的浓度之比;3)有机碳分配系数($K_{\rm oc}$),污染物吸附在土壤中(每单位有机碳)的质量和溶液中平衡浓度的比值

境的 nTiO₂ 和 nZnO 保持悬浮状态,并在波浪和海流等的作用下进行远距离输运;此外,当纳米颗粒沉积或吸附到其它颗粒表面或者由生命体携带时,会在更广阔的空间范围内进行迁移^[28]. 当前研究发现,纳米颗粒可以通过多种途径与水生生物相互作用:纳米颗粒或其团聚物可以吸附或包裹在浮游植物或微生物表面,并最终进入生物体内;滤食性或吞食性动物,如浮游绕足类、底栖贝类和多毛类动物等可以直接滤食或吞食纳米颗粒;海洋生态系统中高营养级生物,通过直接饮用/食用含有 nTiO₂和 nZnO 的水或藻类等低营养级生物接触并富集纳

米颗粒,使纳米颗粒在食物网内发生传递甚至生物 放大,进而产生一系列难以预料的环境效应与生态 风险.

与无机防晒剂不同,水环境中的有机防晒剂主要通过底泥吸附、生物富集和挥发等方式进行迁移,但海水环境中其吸附及生物富集行为尚未见专门报道.当前研究显示,污水处理系统中去除的部分4-MBC 主要被吸附或沉降到污泥中,而非被降解去除^[29].弱极性的奥克立林(OC)容易经疏水相互作用被沉积物的有机碳部分吸附,从而在沉积物中富集.目前,已在伊瓜苏河及其支流的沉积物中

发现浓度最高可达 322.2 $\text{ng} \cdot \text{L}^{-1}$ 的 OC 存在[30]. 防 晒剂被底泥吸附的程度通常用固相-液相分配系数 K_a 表示, K_a 越大, 防晒剂越容易被污泥吸附. 有些 防晒剂难降解、K。低,它们则会通过摄食被生物富 集,并经食物链富集到生态系统不同营养级生物体 内, 甚至对人类自身造成潜在的危害. 此外, 研究 者发现暴露于 BP-3 120 h 后, 风车草对 BP-3 的 富集超过73.9% ±9.1%,暴露过程中根部能够连 续吸收 BP-3 并累积在植物组织中[31]. 以上研究对 于指导海水环境中防晒霜的行为研究具有重要意 义. 对于具有强挥发性的有机防晒剂, 如卤代脂肪 烃、硝基苯类化合物、醚等,挥发是其另一种重要 的迁移途径. 防晒剂的挥发速率依赖于其性质和水 体特征[8], Hudson 等认为[32], 海水对流层中挥发 性有机物分布均匀,扩散和重力沉降可忽略不计. 大气动力学过程和海面气体交换对海水表层挥发性 防晒剂的分布和浓度同样具有关键影响. 然而目 前,挥发对于海水中防晒剂浓度变化的贡献率及其 负面影响仍缺乏研究.

2.2 防晒剂的转化

2.2.1 无机防晒剂的转化

由于 nTiO₂ 的高度稳定性,目前尚未见到其进入海洋后的转化报道.需要注意的是已有大量研究发现,nZnO 可在水体中发生部分溶解,从颗粒态转变成离子态,其溶解率与 nZnO 的性质(如浓度、粒径、纯度)以及水体理化性质(例如 pH、天然有机化合物、电解质)有密切关系^[9]. Miller 等发现^[33],低浓度(0.1~1 mg·L⁻¹)nZnO 能快速溶解于海水中,大约70%颗粒物可溶解转化为Zn²⁺,而当浓度达到10 mg·L⁻¹时,仅有32%颗粒可溶解. Fairbairn等则指出^[34],nZnO 比普通粒径 ZnO 或 Fe 掺杂的nZnO 更易溶解于海水中. 水体理化性质亦会影响nZnO 的溶解度,Miao 等发现^[35],Zn²⁺的释放受水体 pH 值影响,其最大 Zn²⁺ 释放浓度与初始 pH 值导和关关系.

2.2.2 有机防晒剂的转化

有机防晒剂的光化学转化对其环境归趋具有决定性影响^[36]. 在海水表层,有机防晒剂可以通过直接光解、自敏化光解和间接光解等多种途径发生转化. 一种防晒剂可同时发生多种光化学反应,其反应动力学与防晒剂性质、反应类型及环境条件等因素相关^[4]. 理想情况下,防晒剂分子吸收光子跃迁至激发态后,会通过辐射(发射荧光或磷光)、无辐射跃迁(内转换或系间穿跃)等方式回到基态. 然

而,激发态分子的不稳定会引起光致裂解,导致防晒剂分子发生直接光解反应,常见有机防晒剂的反应遵循(准)一级动力学. 防晒剂在模拟日光照射下的光解半减期(t_{1/2})可表示为:

$$\begin{split} t_{1/2} &= \ln \left(2/k_{\rm sl} \right) \\ k_{\rm sl} &= 2.3 \ \varPhi \sum L_{\lambda} \varepsilon_{\lambda}^{p}/2 \end{split}$$

式中, k。为日光照射下光解反应的(准)一级速率常 数,与表观光解量子产率(Φ)、某纬度波长为 λ 的 太阳光光强 (L_{λ}) 、污染物在波长 λ 处的摩尔吸光系 数 $(\varepsilon_{\lambda}^{r})$ 均有关. 目前有机防晒剂的光解速率和 $t_{1/2}$ 数据很少, 远不能满足此类污染物生态风险评价的 需求,同时,防晒剂的结构、季节变化和纬度变化 等都对 $t_{1/2}$ 的测定带来不确定因素^[4]. 另有一些有 机防晒剂的激发态分子可敏化溶解氧或溶剂分子生 成 ROS, 在直接光解的同时发生自敏化光解. 张思 玉等[4]和 MacManus-Spencer 等[30]发现 PBSA、OMC 自敏化光解时的表观光解速率常数 k_{obs} 随 c_0 增大而 减小, 与四环素发生自敏化光解时的变化趋势不 同,因此,防晒剂自敏化光解的机理和动力学规律 较为复杂, 仍有待研究. 防晒剂的间接光解受水体 中溶解性物质的影响,溶解性物质主要通过光致产 生 ROS、激发三重态物种、水合电子等引发防晒剂 的间接光解. 通过对比有机防晒剂在太阳光照下的 直接光解半减期和 ROS 等活性物质引发的间接光 解半减期,发现在10,浓度较高的表层水中,10,引 发的 PABA 间接光解比直接光解更重要, 但其他活 性物质(如·OH)对 PABA 的联合作用仍不清楚[4]. 有机防晒剂的光化学转化产物不尽相同, IAMC、 EHMC 和 4-MBC 在光辐射下结构发生异构化或聚 合, OD-PABA 等则发生降解^[37]. 需要注意的是, 光解可能会加剧 BP-3 对海洋生物的负面影响. Downs 等发现^[38], 光照下 BP-3 对珊瑚 Stylophora pistillata 幼虫的致死性更为显著. Rodil 等[37]研究 BP-3 对栅藻属绿藻(Scenedesmus vacuolatus)的毒性 时也有类似发现. 但目前针对 BP-3 光降解产物的 研究极少, 仅有研究指出, BP-3 被白腐菌降解后产 生了少量 BP-1, 主要的中间代谢产物是糖缀合物 衍生物[39].

除光化学转化之外,海洋中数量庞大、种类繁多、功能各异的生物也会在有机防晒剂的海洋环境归趋中担当重要的角色^[40].有机防晒剂的海洋生物转化研究主要集中在其可生物降解性方面.海洋生物特性如细菌群落组成与结构,环境因子如 pH、

温度和光照等因素均与防晒剂的生物降解密切相关[10],此外不同种类的防晒剂表现出极大的生物降解差异性. 例如,Volpe 等的研究指出[41],EHDPAB 可被海洋沉积物中的微生物群落降解,在好氧和厌氧条件下其降解率均可达到90%;而4-MBC的降解则相对缓慢且降解不完全. 这种差异性可能是近海环境中防晒剂种类集中但浓度差异大的原因. 当前,海水生物对有机防晒剂的降解机制仍未清楚,污水环境中推测的降解机制包括共代谢作用和混合基质增长作用等,前者中微生物通过细胞代谢系统分解或部分转化有机防晒剂,后者中微生物利用防晒剂为碳源和能源,将其完全矿化[8]. 此外Nguyen 等发现[42],白腐菌可通过胞外酶和胞内酶系统(如细胞色素 P450)联合作用去除有机防晒剂,其中,带酚基、脂溶性的防晒剂能够更完全地被白

腐菌降解,这是由于酚类防晒剂可被白腐菌胞外酶 提取物漆酶降解,而脂溶性防晒剂易被生物吸收从 而发生生物降解.但上述转化与降解机制是否适用 于海水环境及其他防晒剂仍需验证.

3 防晒剂的海洋生物毒性

如前所述,防晒剂进入海洋环境后会发生复杂的迁移、转化过程,最终广泛分布于海水、沉积物甚至生物体内,其海洋环境效应与生态风险值得关注^[5,6].目前已发现常见的无机与有机防晒剂对海洋藻类、贝类、鱼类、珊瑚等均具有毒性效应(表2).但总体而言,在已有的研究中,所用的受试生物及防晒剂种类较少,且缺乏环境浓度下长期慢性暴露的毒性数据,相关的毒理学机制也有待进一步明确.

表 2 不同防晒剂对海洋生物的毒性影响

		Table 2 Toxio	ity of diff	ferent UV filters in	n marine organisms	Car &
项目	nTiO_2	nZnO)	BP-3	4-MBC	HS BP-1
藻类	生长抑制作用[43]	生长抑制作用、表现 出生态毒性和细胞毒 性 ^[44]			生长抑制作用, EC ₅₀ 为 171. 45 μg·L ^{-1[45]}	(0)
番 纤毛虫	光催化产生活性氧,导致氧化应激反应 ^[46] 光催化产生活性氧,导 致氧化应激反应 ^[46]	19/		2		影响早期 发育 ^[47]
珊瑚	导致珊瑚漂白[17]	导致珊瑚漂白 ^[17] 累 积在珊瑚胃皮层中并 导致细胞膜扰动 ^[48]	. 现传t	毒性、内分泌干 ^{改畸[38]}	(En) and	
海胆	与 BP-3 和 HS 等防晒剂 共同造成海胆幼虫骨骼 发育异常, 胆碱酯酶活 性下降 ^[49]	造成免疫细胞细胞核 受损,致畸 ^[50] ;景 响囊胚阶段的有丝分 裂活性并导致染色体 异常 ^[51]	河 町 列 川 田 州 山 田 州 山 田 州	iO ₂ 和 HS 等防 共同造成海胆幼 骼发育异常,胆 酶活性下降 ^[49]	生长抑制作用, EC_{50} 为 853. 74 $\mu g \cdot L^{-1[45]}$, 造成海胆幼虫发育异常 (尖端交叉、尖端分离等) $^{[52]}$	与 nTiO ₂ 和 BP-3 等 防晒剂共同造成海胆 幼虫骨骼发育异常, 胆 碱 酯 酶 活 性 下 降 ^[49]
贝类	短暂生物积累[53]	氧化应激、内质网应激、基因毒性 ^[54]	Ĺ		生长抑制作用, EC ₅₀ 为 587. 17 μg·L ^{-1[45]}	
鱼类		氧化应激、细胞损伤 ^[55]	Į			

3.1 无机防晒剂的海洋生物毒性

当前关于无机防晒剂的海洋生物毒性研究主要以 nTiO₂ 和 nZnO 这两类纳米材料为研究对象. 调查表明, 热带国家使用含 nTiO₂ 的防晒霜为16 000~25 000 t·a^{-1[17]}, 而涂抹在皮肤上的防晒霜至少有 25% 在游泳等人类活动过程中进入海洋^[43], 假设防晒霜中 nTiO₂ 含量为 4%^[46], 那么热带国家每年释放到近岸海域的 nTiO₂ 据估计可达 160~250 t. nTiO₂ 在海洋中的大量存在加强了对其生态风险进行评估的必要性. Sendra 等的实验表明^[56], 在直接的太阳光辐射下, 防晒霜和 nTiO₂ 的介入会对浮

游植物物种的数量及其演变造成影响,且其毒性水平与紫外线辐射显著相关。由于 nTiO₂ 的光化学性质,其在紫外线辐射下会产生高浓度 H₂O₂[1 g 商业防晒产品中的无机氧化物纳米粒子在海水中产生H₂O₂ 的速率高达 463 nmol·(L·h)^{-1[46]}],从而导致细胞膜或细胞壁损伤^[57]、脂质过氧化、生长抑制及微藻种群中健康细胞比例下降等毒性效应^[58]。此外,nTiO₂ 纳米颗粒在藻类细胞表面的吸附会造成遮光效应等物理伤害,进而抑制细胞生长^[59]。Wang等^[11]以及 Minetto等^[60]还发现 nTiO₂ 对海洋微藻的毒性,受 nTiO₂ 浓度、紫外照射、pH 值和离

子强度等因素的影响.考虑到 nTiO₂ 容易累积在海水表层和沉积物中^[5],其对海洋中浮游动物和底栖动物生长的影响值得重视.除单独的生态效应之外,nTiO₂ 与环境中其他污染物的相互作用同样值得关注.nTiO₂ 体积小、比表面积大、静电引力强,从而容易吸附其它环境污染物,并影响其迁移、归趋及其对水生生物的毒性.笔者的前期研究发现^[61],在2 mg·L⁻¹ nTiO₂ 存在下,三丁基锡(TBT)对九孔鲍(Haliotis diversicolor supertexta)的毒性提高了近20倍,这可能是由于TBT吸附于nTiO₂聚集体,而nTiO₂聚集体最终进入鲍鱼胚胎内.除此之外,nTiO₂与水环境中的菲(Phe)污染物形成nTiO₂-Phe 复合物,从而促进了海洋贝类毛蛤(Scapharca subcrenata)对Phe 的吸收^[62].然而截止目前还没有关于nTiO,长期毒性效应的综合数据^[43].

nZnO 可以吸收 UVA 和 UVB, 而 nTiO, 只能吸 收 UVB, 因此 nZnO 可以提供更好的紫外线防护功 效, 其在无机防晒剂中的使用量在未来甚至可能超 过 nTiO2[63]. 有研究表明, nZnO 对海洋藻类的毒性 远高于普通粒径的 ZnO^[55], 其对假微型海链藻 (Thalassiosira pseudonana)表现出显著的生长抑制。 效应(图2),且纳米材料的不同来源对其毒性效应 无明显影响[4]. 进一步的研究发现, nZnO 对藻类 的毒性与其浓度及暴露时间有关[41,64,65], 当 nZnO 浓度升高和暴露时间增加时, nZnO 对杜氏盐藻 (Dunaliella tertiolecta)、假微型海链藻(Thalassiosira pseudonana)和海洋小球藻(Chlorella vulgaris)的生 长抑制作用会增强,在300 mg·L⁻¹,72 h暴露条件 下时甚至可改变细胞的完整性并引起细胞膜的巨大 损伤. Tang 等[48]的研究同样表明, nZnO 可以通过 干扰脂质的正常代谢过程造成珊瑚细胞膜扰动,进 而造成潜在的慢性毒性. 而对海胆的研究表明, nZnO 可导致成年海胆的免疫细胞细胞核受损及幼 体畸形, 且其毒性效应与其尺寸具有密切关系, 100 nm 直径的 nZnO 颗粒造成的细胞核受损率 (33%)及致畸率(75.5%)显著高于14 nm 直径的 nZnO 颗粒造成的细胞核受损率(64%)及致畸率 (84.7%)^[50]. 此外, nZnO 在 30 μmol·L⁻¹的较高 浓度下可导致海胆囊胚出现有丝分裂活性下降及染 色体异常[51]. 相较于对藻类的毒性效应, nZnO 对 鱼类的毒性明显较小. Wong 等[55]利用超氧化物歧 化酶(SOD)和金属硫蛋白(MT)来评估4 mg·L⁻¹和 40 mg·L⁻¹的 nZnO 对青鳉鱼(Oryzias melastigma)的 亚致死毒性,暴露于 nZnO 的青鳉鱼(Oryzias

melastigma)相比于对照组并无明显的 SOD、MT上升表现. nZnO 的毒性与其溶解的 Zn²+相关,溶解度越高其毒性越大,考虑到 nZnO 在海水中的溶解度大于在淡水中的溶解度,其对海洋环境的潜在危害不容忽视^[66]. 但也有研究指出, nZnO 的生物毒性受纳米颗粒本身的毒性决定. 因此,针对 nZnO 的海洋生物毒性,如何区分溶解态和颗粒态的毒性贡献是未来研究的重点. 值得一提的是, nZnO 可抑制有害细菌的生长,这在一定程度上有利于自然水环境中和废水处理过程中的污染物净化^[64],其综合环境效应需要系统性评估.

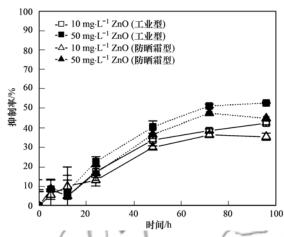


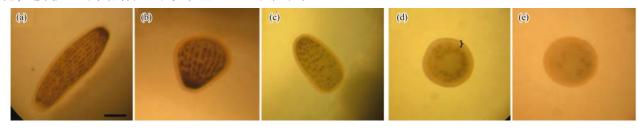
图 2 10 mg·L⁻¹和 50 mg·L⁻¹不同类型 ZnO(防晒霜型 和工业型) 对假微型海链藻的生长抑制作用^[44]

Fig. 2 Growth inhibition of *T. pseudonana* with ZnO (sunscreen and industrial) exposure

3.2 有机防晒剂的海洋生物毒性

目前, 二苯甲酮(BPs)、樟脑衍生物类和肉桂 酸类有机防晒剂在国内外防晒霜中的应用最为广 泛[12]. BPs 结构稳定, 是有机防晒剂中最重要的一 类^[20], 常见的种类有二苯甲酮(BP)、2-羟基二苯 甲酮(2-HB)、2-羟基-4-甲氧基二苯甲酮(BP-3)、 2-羟基-4-甲氧基二苯甲酮-5-磺酸(BP-4)等. 其中, BP-3 已在水、土壤、沉积物、污泥中被检测出来, 在淡水和海水中的最大检测浓度分别为 428 ng·L^{-1[65]}和3 300 ng·L^{-1[67]}. BP-3 在海水中的浓度 几乎是淡水中的 10 倍, 表明 BPs 对海洋生物的毒 性影响不容小觑. 但目前, BPs 对海洋生物的急性 毒性数据还很少, 仅有的研究主要关注贻贝、细菌 和珊瑚这3种生物. Paredes 等[45]的研究报道了 BP-3 对紫贻贝(Mytilus galloprovincialis)幼虫的半效 应浓度(EC₅₀)为 3.42 mg·L⁻¹. Liu 等^[68]则发现 BP、BP-1、BP-2、BP-3 对海洋发光细菌——明亮发 光杆菌(Photobacterium phosphoreum)的 15min-EC50 估计值分别为 34. 26、14. 21、8. 24、14. 27 mg·L⁻¹. Downs 等发现^[38], BP- 3 会使珊瑚 *Stylophora pistillata* 的幼虫从运动状态转变为畸形、无梗的状态,随着 BP-3 浓度的增加,幼虫会表现出更高比例的漂白现象,并在暴露 8h 后出现死亡(LC_{50} = 3. 1 mg·L⁻¹,图 3). 此外,BP-3 可导致珊瑚幼虫骨化,这或归因于其内分泌干扰效应. BPs 对非海洋

生物的毒性研究同样表明其毒性机制与其内分泌干扰特性相关. BPs 具有高脂溶性,可以通过口腔和真皮被生物体快速吸收^[69],进入生物体后表现出多重激素活性^[70],如激活双翅目昆虫溪流摇蚊(*Chironomus riparius*)中一组脱皮激素响应基因的表达^[71],影响鱼类的内分泌平衡和繁殖能力^[72],或影响生物体内性腺和甲状腺等激素的功能^[73].



(a) 对照组; (b) 22.8 $\mu g \cdot L^{-1}$ BP-3; (c) 228 $\mu g \cdot L^{-1}$ BP-3; (d) 2.28 $m g \cdot L^{-1}$ BP-3; (e) 22.8 $m g \cdot L^{-1}$ BP-3

图 3 珊瑚 Stylophora pistillata 幼虫在光照下于不同浓度 BP-3 暴露 8 h 后的显微图片 [38]

Fig. 3 Stylophora pistillata planulae exposed to various treatments of BP-3 for 8 h in light

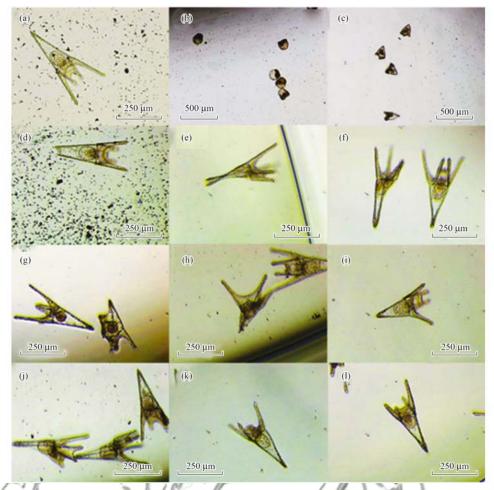
我国《化妆品卫生规范》(2007版)规定可以使 用 6 种樟脑衍生物类有机防晒剂, 分别是 3-亚苄基 樟脑(3-MC)、4-甲基苄亚基樟脑(4-MBC)、亚苄基 樟脑磺酸、樟脑苯扎铵甲基硫酸盐、聚丙烯酰胺甲 基亚苄基樟脑和对苯二亚甲基二樟脑磺酸. 樟脑衍 生物具有较高的光稳定性和化学稳定性, 不易生物 降解,环境累积效应高[19,37,74]. 其中 4-MBC 已在 河流、湖泊、海洋等自然水体中发现有 ng 级水平存 在, 其在近岸海水表层的日浓度变化已达到51.4~ 113.4 ng·L^{-1[6,75,76]}, 在生物体中的浓度高达1 800 ng·g-1, 在鱼体内的累积水平甚至与 DDT、PCBs 相 似[52], 故而在食物链中存在潜在的生物累积作用 和生物放大作用[77],相应的海洋生态风险不容忽 视. 但遗憾的是仅有少量有关 4-MBC 的海洋生物 毒性效应研究. Paredes 等[45] 发现 4-MBC 对海洋球 等鞭金藻(Isochrysis galbana)、紫贻贝(Mytilus galloprovincialis)、海胆(Paracentrotus lividus)以及刺 节糠虾(Siriella armata)的毒性远大于 BP-3、BP-4 等防晒剂, 其中对球等鞭金藻的毒性水平最高. Torres 等[52]以斑马鱼和海胆的胚胎作为生物模型, 发现 4-MBC 对两种胚胎的最大无影响浓度(NOEC) 分别为 50 μg·L⁻¹和 0.32 μg·L⁻¹, 均在环境相关浓 度下对胚胎发育产生影响. 其中海胆幼虫可以发生 尖端交叉、尖端分离等发育异常的情况(图4). 上 述生物毒性与樟脑衍生物类防晒剂的内分泌干扰效 应相关. 即 4-MBC 在低浓度水平下可干扰斑马鱼 和海胆胚胎发育相关激素调控的一系列生理过 程[52],此外,4-MBC 对非海洋生物的毒性研究显

示,其毒性机制还包括诱导溪流摇蚊(Chironomus riparius)胚胎和幼虫阶段蜕皮激素受体(EcR)和热休克蛋白(HSP70)的基因表达^[78],影响斑马鱼胚胎的肌肉和神经元发育^[79],这些研究结论或可指导 4-MBC 对海洋生物毒性机制的后续研究.

肉桂酸酯类有机防晒剂中以甲氧基肉桂酸乙基 己酯(EHMC)和 p-甲氧基肉桂酸异戊酯最为常见. 目前,大约70%的防晒化妆品都添加了EHMC[19]. EHMC 一般以反式异构体结构存在, 但光暴露下会 转变成顺式异构体,后者的环境风险是前者的1.7 倍,因此,光暴露将大大加剧 EHMC 的毒性[80].调 查表明,饮用水、沉积物、地表水、海水和水生生 物中均有 EHMC 存在, 其中沉积物是 EHMC 的主 要归宿. 例如, 东京湾沉积物中 EHMC 的浓度达到 79.0 μg·kg⁻¹, 远远高于其他防晒剂(BP-3 为 0.5 μg·kg⁻¹, ED-PABA 为 1.0 μg·kg⁻¹)^[81]. Bachelot 等[82]和 Sang 等[83]在法国和香港沿海水域的贻贝 中均检测出高浓度 EHMC 的存在. Tsui 等[84] 进一 步用研究中获得的测量环境浓度(MECs)除以预期 的无效浓度(PNECs),通过获得的风险商(HQs)评 估 EHMC 的海洋生态风险,发现其对腹足类海洋底 栖生物造成繁殖毒性的概率高达84%. 这些研究表 明 EHMC 的海洋生物毒性,尤其是对底栖生物的毒 性不容忽视. 但目前尚无直接的毒性实验数据, 相 关研究亟待开展.

3.3 防晒剂的致毒机制

如前所述, 当前研究发现不同类型防晒剂可对 多种海洋生物(细菌、藻类、珊瑚、贝类和鱼类等)



(a)长腕幼虫阶段的正常幼虫; (b) 原肠期; (c) 棱镜幼虫; (d) 尖端分离; (e)、(f) 尖端交叉;
 f)、(g) 臂融合; (g) 一般异常幼虫; (h)、(i) 臂定位异常; (j) 臂变形; (k) 臂缺失; (k)、(l) 臂不对称
 图 4 孵育 48 h 后,对照组及 4-MBC 暴露组中海胆 Paracentrotus lividus 幼虫的显微照片[52]

Fig. 4 Embryonic stages and developmental abnormalities of the sea urchin

Paracentrotus lividus observed after 48 h of incubation under controlled conditions of temperature and light

造成毒性(表2),这不仅证实了防晒剂的海洋生态 风险, 也表明亟需开展防晒剂的海洋生物毒性机制 研究. 目前, 在无机防晒剂的海洋生物毒性考察过 程中, 发现 nTiO, 对海洋生物的氧化损伤、物理伤 害及 nZnO 的 Zn2+离子释放在它们的毒性效应中具 有关键作用. nTiO, 等防晒剂的介入能够使得机体 自由基水平增高[46], 过量的自由基特别是 ROS 自 由基导致机体氧化与抗氧化失衡, 甚至可以攻击包 括 DNA 在内的几乎所有生物分子,产生肿瘤、畸 形、衰老、死亡等后果[85]. 另外, nTiO₂ 在太阳光 照尤其是紫外辐照下,可于水溶液中产生 H₂O₂,在 高浓度(> 300 μmol·L⁻¹)下可对生物有机体产生 不可逆损伤, 在低浓度下则能通过激活细胞内各种 信号转导机制造成对细胞的损害, 甚至还参与预适 应的形成, 其作用机制十分复杂[86]. 此外, 已有研 究发现 nTiO₂ 暴露导致神经细胞 D384 和 SH-SY5Y

的细胞膜呈现不完整状态,膜电势降低,细胞活性显著降低;或导致小神经胶质细胞内 IL-1β、IL-2、TNF-α、MCP-1、MIP-2等含量上调,诱发神经炎性^[87]. 但无机防晒剂对海洋生物造成的神经毒性尚缺乏直接的实验数据支持. 对于 nZnO,研究表明其毒性机制同样包括细胞膜扰动^[48],此外 nZnO 可以导致海胆成体细胞核受损及幼体畸形^[50]. 并在30 μmol·L⁻¹的较高浓度下导致海胆囊胚有丝分裂活性下降及染色体异常^[51].

有机防晒剂对海洋生物的毒性效应与其脂溶性和内分泌干扰特性相关^[69]. 防晒剂可通过受体介导反应、非受体介导反应、影响内分泌系统与神经系统和免疫系统的综合效应等机制对生物内分泌系统产生影响^[88]. 现有研究表明,有机防晒剂具有多种内分泌干扰效应. 在分子水平上诱导分泌 pS2 蛋白,影响多种激素受体如雌激素受体(ER)、雄激

素受体(AR) 以及孕激素受体(PR)的基因表达, 还 能诱导雄性动物产生卵黄蛋白原; 在细胞水平上影 响生殖细胞的形成,抑制细胞酶的活性;在个体水 平上影响生殖系统、青春期发育、子代成活率等方 面,不同有机防晒剂的内分泌干扰效应也有所不 同^[89]. 例如 3-亚苄基樟脑(3-BC)和二苯甲酮(BP-2) 可以引起雄性鱼类第二性征的雌性化, 进而降低 其生育率和繁殖率[90~92]. 有机防晒剂还能引起体 内外神经毒性, 其毒性机制包括细胞结构损伤、炎 症反应、神经细胞凋亡等. 例如 BP-3 可激活 caspase 家族和 Ca²⁺ 依赖的内质网应激, 引起小神 经胶质细胞 N9 和星形胶质细胞 U87 细胞的凋 亡^[73, 93]. 此外 BPs 能够在低浓度(1.0 μg·L⁻¹)引 起淡水原生动物嗜热四膜虫(Tetrahymena thermophile)体内的氧化应激反应^[94],使过氧化氢 酶(CAT)活性增强,谷胱甘肽(GSH)含量下降. 但 BPs 是否能对海洋生物引起相似的神经毒性及氧化 应激响应, 仍有待探索, 光化学转化过程也可显著 影响有机防晒剂的毒性,加剧或减弱其对海洋生物 的影响^[36]. 相较于无光照射条件, 光照下 BP-3 对 珊瑚 Stylophora pistillata 幼虫和栅藻属绿藻 (Scenedesmus vacuolatus)的毒性更强[37,38],而 Et-PABA 对海洋细菌 V. fischeri 的毒性下降, 光照 12h 后其毒性消失[95]

4 结论与展望

防晒霜的大量生产及使用会不可避免地造成其 环境泄漏, 尤其是在沿海水域中的泄漏, 因此防晒 剂的海洋生态效应已经引起广泛关注. 其中无机防 晒剂(nTiO,和 nZnO等)具有颗粒粒度小、比表面 积大、吸附能力强等纳米尺度特征, 有机防晒剂 (BP-3、4-MBC、EHMC等)具有脂溶性强、易生物 累积等特征, 两类防晒剂均对多种海洋生物(细菌、 鱼类、藻类、贝类、珊瑚等)具有显著毒性效应. 其 中无机防晒剂的毒性机制包括氧化损伤、物理伤 害、离子释放等,有机防晒霜则表现出氧化损伤、 内分泌干扰、神经毒性和生物累积等负面效应. 两 类防晒剂不尽相同的毒性机制表明其对海洋生物具 有不同威胁, 在研究防晒霜的生态危害时应对其分 别进行探索. 需要注意的是, 目前关于防晒霜海洋 生物毒性的研究仍比较匮乏, 受试生物种类、暴露 条件等均十分有限, 且缺乏真实海洋环境中防晒霜 生物毒性的报道, 这无疑不利于全面并准确评估防 晒霜的海洋生态毒性. 此外, 考虑到无机及有机防 晒剂的吸附能力,其进入海洋环境后易与其他环境 污染物发生相互作用,产生异于其单独毒性效应的 共同毒性效应,未来应对此方面展开更为深入地研 究,为防晒霜海洋生态毒理学的开展及其污染管控 奠定基础.

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