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Microbial Biodegradation of Sunscreen Agents: Mechanisms, Enzymatic Pathways, and Environmental Implications

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ABSTRACT: Environmental contamination from sunscreen ingredients such as oxybenzone and octinoxate has become an increasing concern due to their persistence and toxicity, even at trace concentrations. Continuous sunscreen usage leads to the constant release of these pollutants into the environment, where they can bioaccumulate and resist degradation. The novelty of this review lies in its focused synthesis of recent studies on the microbial and enzymatic degradation mechanisms of sunscreen contaminants, particularly oxybenzone and octinoxate, which exhibit high persistence and bioaccumulative potential. Microbial degradation offers a promising biological approach for the breakdown of these organic pollutants, as microorganisms have demonstrated strong biodegradative capabilities toward various environmental contaminants. This process relies on microbial enzymes that transform or mineralize pollutants into less toxic and simpler compounds. Key enzymes involved include laccase, cytochrome P450, and monooxygenase, which catalyze oxidation, reduction, and hydroxylation reactions. The article further examines these organic pollutants in terms of their persistence, environmental occurrence, degradation mechanisms, and pathways, while also addressing their ecological and health impacts. Moreover, different microbial-based treatment technologies are evaluated, highlighting their respective strengths and limitations. Finally, the review emphasizes the need for continued research into organic pollutant behavior and bioremediation technologies to deepen understanding and mitigate the adverse effects of these contaminants on the environment.

KEYWORDS: Environmental pollution; microbial degradation; enzymatic biodegradation; oxybenzone; octinoxate

1. Introduction

The increasing use of personal care products, particularly sunscreen formulations, has become a significant source of aquatic environmental contamination, primarily when residues are discharged as effluents from wastewater treatment facilities [1]. Oxybenzone (benzophenone-3) and octinoxate (ethylhexyl methoxycinnamate) are among the primary organic compounds commonly used in sunscreens, serving as ultraviolet (UV) radiation filters [2]. In the United States, these ingredients are permitted within regulated limits in commercial sunscreen formulations [3, 4]. Data from the Centers for Disease Control and Prevention (CDC) further reveal that exposure to oxybenzone is widespread among the U.S. population [5]. These compounds are presently categorized as Category III ingredients under the U.S. Food and Drug Administration (FDA) GRASE (Generally Recognized as Safe and Effective) classification. This designation indicates that they are not yet fully approved but are provisionally allowed in current sunscreen formulations while additional safety assessments are conducted [6]. In contrast, only inorganic UV filters, specifically zinc oxide and titanium dioxide in both micro- and nano-particulate forms, have received full approval for worldwide use in sunscreen products [7]. Several regions, including the Republic of Palau and the State of Hawaii, have implemented bans on sunscreen products containing oxybenzone and octinoxate due to their adverse ecological impacts. In Palau, strict legislation prohibits the importation, distribution, sale, and use of such sunscreens [8], while in the European Union, oxybenzone use has been heavily restricted and subjected to ongoing regulatory review [7].

The persistence and high-water solubility of these compounds underscore the urgency of assessing their ecological effects and developing effective treatment technologies for their removal [1, 2]. Table 1 presents the general physicochemical properties of both oxybenzone and octinoxate. Beyond their use in sunscreens, oxybenzone also functions as a UV stabiliser in various consumer products, including plastics, food packaging, furniture, and electronic coatings [9]. It is estimated that approximately 81% of personal care products marketed in the United States and China contain oxybenzone [10]. Survey data further highlight the widespread usage of sunscreens globally; for instance, 79.4% of 3,000 respondents in Germany reported frequent sunscreen use [11], while 75% of 2,321 participants in Saudi Arabia indicated regular application [12]. Many consumers remain unaware that these products contain potentially hazardous compounds with toxicological implications for both environmental and human health [2].

Both oxybenzone and octinoxate are classified as endocrine-disrupting contaminants. They exhibit a range of hormonal activities, including anti-androgenic, pro-estrogenic, and anti-estrogenic effects [13]. Experimental studies have shown that exposure to these chemicals can subtly alter concentrations of key sex hormones such as testosterone, estradiol, and inhibin B [14]. Octinoxate has been found to disrupt thyroid hormone regulation in Rainbow trout after six weeks of exposure, a concerning effect given the essential role of thyroid hormones in metabolism and immune function [2, 15]. Additional reported impacts include impaired antioxidant defenses, alterations in white blood cell counts, disruptions to metabolic processes, and histopathological damage in liver tissues [2].

Considering their harmful ecological and physiological effects, effective treatment strategies are required to degrade or detoxify these compounds. However, conventional wastewater treatment systems are generally ineffective at removing such persistent organic pollutants, allowing their accumulation in aquatic ecosystems and biota [16]. To address this

challenge, microbial degradation has emerged as a promising bioremediation approach due to the ability of certain microorganisms to mineralize organic contaminants at high degradation rates [1, 17]. Innovative microbe-based technologies such as revolving algal biofilms and fluidized bed bioreactors, have been explored for this purpose [17]. Despite their potential, microbial systems face limitations, including sensitivity to environmental conditions and slower degradation rates compared to physicochemical techniques like advanced oxidation processes [18].

The novelty of our review is the focused synthesis of recent findings on the microbial and enzymatic degradation mechanisms of sunscreen contaminants, particularly oxybenzone and octinoxate, which are highly persistent and bioaccumulative even at low concentrations. Unlike previous reviews, which primarily discussed the environmental occurrence or toxicological effects of UV filters, this article presents an integrated perspective combining environmental persistence, dispersion pathways (in soil, air, and aquatic environments), and biodegradation mechanisms involving key microbial enzymes such as laccase, cytochrome P450, and monooxygenase. Furthermore, the review evaluates microbial-based technologies for contaminant remediation, comparing their advantages, limitations, and potential future applications. This comprehensive approach provides new insights into how microbial degradation can be effectively used to remediate organic contaminants from sunscreens. Furthermore, the review discusses the broader environmental and health impacts of these contaminants on aquatic ecosystems and humans, thereby providing insights into future directions for mitigating their risks.

2. Environmental Occurrence of Oxybenzone and Octinoxate

The extensive use of sunscreen products has led to the continuous release of oxybenzone and octinoxate into multiple environmental compartments. These compounds have been identified in wastewater treatment plant (WWTP) influents and effluents, surface waters, sediments, and even drinking water, posing serious environmental and health risks due to their persistence and toxicity. Studies have consistently reported the presence of oxybenzone in WWTPs across various regions, reflecting its widespread occurrence. Seasonal variations have also been observed, with higher detection rates during summer months when sunscreen use is more prevalent. Although WWTPs are capable of partially removing these contaminants, significant amounts still escape into receiving water bodies. Alarmingly, oxybenzone has also been detected in drinking water and indoor dust, suggesting additional contamination pathways beyond aquatic systems. On a global scale, thousands of tons of sunscreen-derived compounds are released into coral reef environments each year, contributing to the accumulation of these pollutants in marine ecosystems [16]. In addition, oxybenzone can infiltrate groundwater through leaky sewage systems and polluted urban runoff, as demonstrated by detection at 7.23 ng/l in groundwater samples from Spain [17]. Recreational waters such as swimming pools and ponds in South Bohemia have also shown high concentrations, reaching up to 620 ng/l and 550 ng/l, respectively, during peak summer [18]. Similarly, coastal waters near tourist-heavy areas like Majorca recorded 577.5 ng/l of oxybenzone, far higher than concentrations in rivers or lakes [19]. These findings confirm the persistent and ubiquitous nature of sunscreen-derived pollutants.

3. Sources and Transport Pathways

The entry of oxybenzone and octinoxate into the environment occurs through multiple interconnected pathways, including direct release, indirect discharge, and secondary redistribution via environmental media. Direct inputs occur during swimming, bathing, and recreational activities, as sunscreen residues are washed off the skin. It is estimated that up to 96% of oxybenzone is rinsed off during bathing, with residues flowing into wastewater systems or directly reaching coastal waters, as observed at Hanauma Bay [6]. Sunscreen can be washed off with or without soap or shampoo, and other personal care products like shampoos and lotions may also contain UV filters. Indirect inputs mainly come from WWTP effluents, municipal sewage discharges, cesspits, and septic systems, which serve as non-point sources of contamination [80]. Due to their lipophilic properties, oxybenzone and octinoxate tend to adsorb to sediments, soils, and sludge. Agricultural reuse of treated wastewater or sludge can further introduce these compounds into terrestrial systems, where they may leach to groundwater. New contamination pathways have also been identified. For instance, plastic debris in aquatic environments can adsorb and subsequently release UV filters, acting as a secondary source of pollution. Additionally, aerosolized sunscreen particles may disperse through volatilization from wastewater treatment plants, wave activity along coastlines, and spray from recreational boats, introducing an atmospheric route for their transport [19]. Human activities further contribute to environmental accumulation, as oxybenzone has been consistently detected in urine, breast milk, and placental tissues, indicating the potential for maternal–fetal and maternal–infant transfer. Indoor dust in South Korea and the United States has also shown the presence of oxybenzone, suggesting that household dust represents another overlooked reservoir of contamination [12].

4. Environmental Fate and Accumulation

Once released into the environment, UV filters such as Oxybenzone and Octinoxate undergo a series of complex physicochemical and biological fate processes that dictate their persistence and ecological risk. Their relatively hydrophobic nature (high log $K_o w$), coupled with chemical stability, promotes their partitioning from the aqueous phase into organic-rich sediments and suspended particulate matter [20]. This sediment association not only shields the compounds from direct photolysis but also allows their long-term retention within benthic environments.
In aquatic systems, these compounds are subjected to photodegradation under sunlight exposure, especially in surface waters. However, the degradation rates are often slow, and the transformation products can retain toxicity or estrogenic activity, potentially compounding ecological impacts [21]. Biotic degradation is also limited, as conventional wastewater treatment plants show low mineralization efficiency, resulting in continuous discharge into natural waters [12].

Sedimentation processes facilitate the vertical transport of sunscreen-derived pollutants from surface waters to benthic habitats. This creates reservoirs of contaminants in sediments, particularly in enclosed or low-flushing marine systems such as coral reef lagoons and coastal embayments [22]. Over time, the pollutants can be resuspended by hydrodynamic disturbances, resulting in recurring exposure to pelagic organisms.

Furthermore, these UV filters exhibit bioaccumulative tendencies, as evidenced by their detection in a range of aquatic organisms including plankton, bivalves, crustaceans, and fish [23]. Lipophilic accumulation in tissues raises concerns about chronic exposure and potential biomagnification along trophic levels. Organisms inhabiting contaminated sediments, such as

benthic invertebrates, may act as entry points for the pollutants into the food web, transferring them to higher trophic organisms including fish and marine mammals [24]. The combined processes of adsorption, photodegradation, sedimentation, resuspension, and bioaccumulation contribute to the long-term persistence and cycling of these sunscreen contaminants in aquatic ecosystems. A conceptual overview of the transport pathways and environmental fate of sunscreen-derived UV filters is presented in Figure 1, summarizing their movement across water, sediments, biota, and atmospheric interfaces.

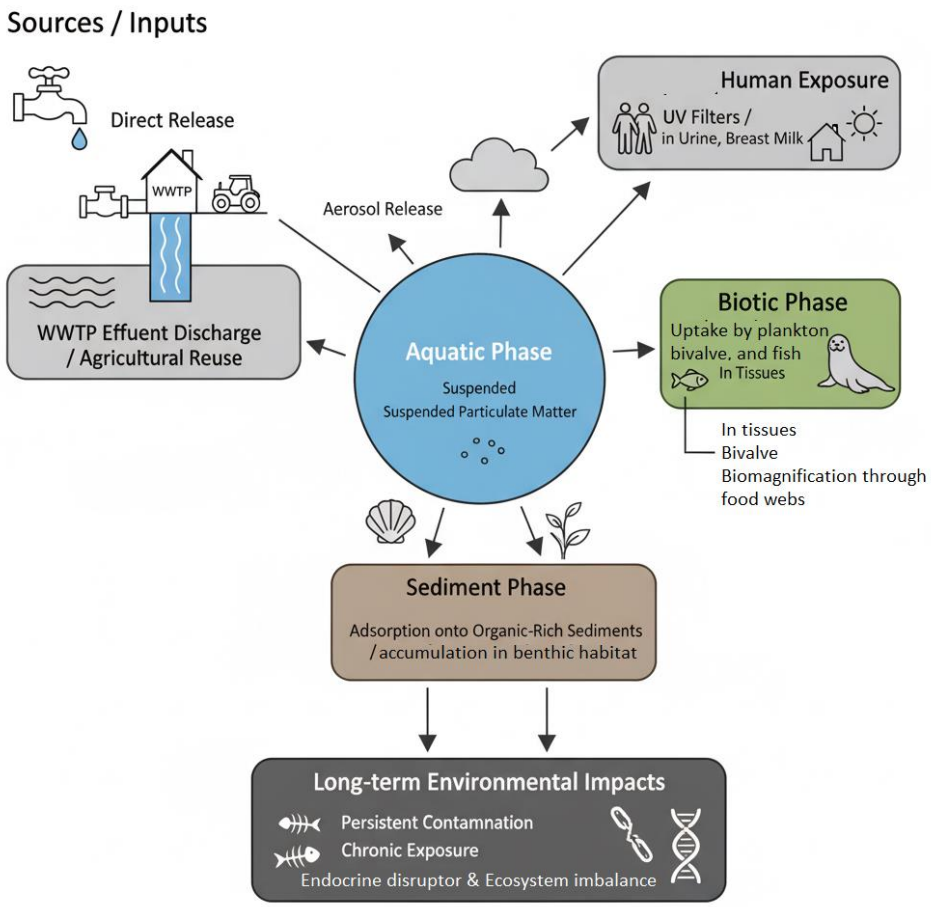

Figure 1. Conceptual pathways and environmental fate of sunscreen-derived UV filters.

5. Impacts of Oxybenzone and Octinoxate on Marine Organisms

The sunscreen compound oxybenzone (benzophenone-3) exhibits multiple adverse effects on marine species, ranging from microalgae to coral reefs and fish. Its toxicity arises from its ability to generate reactive oxygen species (ROS), disrupt photosynthesis, interfere with DNA, and induce endocrine disruption in marine organisms.

5.1. Effects on marine flora and microalgae.

Oxybenzone is acutely toxic to photosynthetic organisms, including microalgae and seagrasses. In *Posidonia oceanica*, benzophenone-3 exposure elevated ROS levels, leading to irreversible damage in chloroplast ribosomes and thylakoid membranes. Excess ROS caused biomolecular damage, altered membrane fluidity, disrupted ion transport, induced protein cross-linking, and

inhibited protein synthesis, ultimately resulting in cell death [24]. In *Cucumis sativus L.*, oxybenzone exposure inhibited photosynthetic electron transport (PET) activity, which increased with higher concentrations verproduction of ROS secondary to UV filter exposure further suppressed photosynthesis [25]. Additionally, oxybenzone has genotoxic effects, causing oxidative DNA damage, formation of DNA abasic sites, strand breaks, DNA-protein crosslinks, and cyclobutene pyrimidine dimers, often via CYP450-mediated bioactivation [26].

5.2. Coral reef impacts.

Corals are highly sensitive to oxybenzone. Exposure decreases algal pigment content and induces oxidative stress, causing coral bleaching by prompting expulsion of symbiotic zooxanthellae [11, 21, 27]. Chronic exposure, compounded by elevated ocean temperatures, accelerates coral mortality. Experimental studies on *Cladocopium goreaui* and *E. voratum* indicate that higher concentrations (≥500 µg/l) severely reduce cell density and chlorophyll-a content, leading to complete bleaching in *C. goreaui* [28]. Oxybenzone can also trigger viral lytic cycles in symbiotic algae, exacerbating coral damage [21].

5.3. Effects on marine fauna.

Fish are affected behaviorally and physiologically. In zebrafish (*Danio rerio*), exposure to 10–1000 µg/L oxybenzone for 15 days reduced locomotor activity, increased anxiety-like behavior, decreased aggressiveness, and delayed hatching [29]. These effects are attributed to the anti-androgenic and estrogenic activity of oxybenzone, demonstrating endocrine disruption in aquatic vertebrates [29]. Summary of oxybenzone impacts on marine species is shown in Table 1.

Table 1. Summary of oxybenzone impacts on marine species.

Organism/Species	Exposure Concentration	Observed Effect	Mechanism	Reference
Posidonia oceanica	5–500 µg/L	ROS increase, chloroplast damage, cell death	Disruption of photosynthesis, ROS-mediated biomolecular damage	[24]
Cucumis sativus L.	50–5000 µg/l	PET inhibition, photosynthesis suppression	ROS overproduction, chloroplast electron transport disruption	[25]
DNA in various species	N/A	DNA strand breaks, crosslinking, mutagenesis	CYP450-mediated bioactivation, genotoxicity	[26]
Cladocopium goreaui	500–5000 µg/l	Chlorophyll-a depletion, cell density decline, bleaching	Oxidative stress, pigment biosynthesis inhibition	[28]
E. voratum	500–5000 µg/l	Reduced cell density, chlorophyll-a decrease	ROS-mediated damage	[28]
Coral species	33–50 ppm	Coral bleaching, symbiont expulsion	Oxidative stress, viral lytic activation	[21, 25, 27]
Zebrafish (*Danio rerio*)	10–1000 µg/l	Reduced locomotion, delayed hatching, altered aggression	Endocrine disruption (anti-androgenic/estrogenic effects)	[29]

6. Mechanisms of Microbial Degradation of Sunscreen Compounds

Microbial degradation of persistent organic pollutants such as oxybenzone and octinoxate involves a series of enzymatic reactions that break down complex aromatic compounds into simpler, non-toxic metabolites. This process is critical for environmental remediation because

these compounds are highly resistant to conventional wastewater treatment. Microorganisms employ several mechanisms, including hydroxylation, oxidation, and the β-ketoadipate pathway, to facilitate the complete mineralization of the pollutants.

6.1. Hydroxylation.

Hydroxylation is the primary step in microbial degradation, where hydroxyl groups (-OH) are introduced into the aromatic structure of the pollutants, usually by replacing hydrogen atoms or functional groups [27]. This reaction is commonly catalyzed by the cytochrome P450 (CYP450) family of enzymes, which are widespread in bacteria and fungi [30]. By adding hydroxyl groups, the hydrophobic benzophenone compounds are transformed into more hydrophilic metabolites, making them more soluble in water and amenable to further degradation [31]. The key hydroxylated metabolites identified include 2,4-dihydroxybenzophenone (DHB), 2,2′-dihydroxy-4-methoxybenzophenone (DHMB), and 2,3,4-trihydroxybenzophenone (THB). DHB is formed through O-dealkylation of the methoxy side chain on ring A of 2-hydroxy-4-methoxybenzophenone (HMB), whereas DHMB results from aromatic hydroxylation of ring B at the ortho position. THB is produced by meta-hydroxylation of ring A of DHB [32]. These hydroxylated metabolites are less lipophilic and more chemically reactive, facilitating subsequent enzymatic oxidation and ring cleavage. Hydroxylation not only increases the solubility of the compounds but also reduces their bioaccumulation potential, a critical factor in limiting ecological toxicity. This step is often the rate-limiting stage in microbial degradation, as the efficiency of CYP450 enzymes can be influenced by environmental conditions such as pH, temperature, and the presence of co-substrates.

6.2. Oxidation.

After hydroxylation, the aromatic metabolites undergo oxidation, a crucial step where molecular oxygen is incorporated into the compound, typically leading to ring cleavage [33]. This reaction is primarily catalyzed by dioxygenases, enzymes belonging to the oxygenase class, which can simultaneously introduce two oxygen atoms into the aromatic ring [34]. Ring-cleavage dioxygenases such as catechol-1,2-dioxygenase (C12DO) and catechol-2,3-dioxygenase (C23DO) are responsible for breaking the hydroxylated intermediates at the ortho or meta positions, respectively [35]. The oxidation reaction transforms the planar aromatic structures into linear, open-chain compounds such as cis,cis-muconate, which are highly reactive and can be further metabolized by microbial enzymes. This step is essential because the aromatic ring is chemically stable and resistant to natural breakdown; oxidative cleavage is what enables microbes to funnel the products into central metabolic pathways. The efficiency of oxidation is influenced by enzyme specificity, substrate availability, and environmental factors. Oxidation also generates reactive intermediates that can undergo non-enzymatic transformations, further expanding the diversity of degradation products and facilitating complete mineralization into CO_2, water, and inorganic ions.

6.3. β-Ketoadipate pathway (β-KAP).

The β-KAP is a well-characterized microbial route for degrading aromatic compounds into central metabolites such as acetyl-CoA and succinyl-CoA [36]. This pathway is employed by

both bacteria and fungi, including species like *Cochliobolus heterostrophus*, *Phanerochaete chrysosporium*, and *Fusarium solani*. Initially, diverse aromatic pollutants, such as lignin-derived compounds (coniferyl alcohol) and chlorinated aromatics (4-chlorobenzoate), are converted into catechol or protocatechuate, which serve as key intermediates. The ring cleavage is catalyzed by mononuclear non-heme iron enzymes like C12DO or protocatechuate 3,4-dioxygenase, producing cis,cis-muconate or β-carboxymuconate. These intermediates are further processed via enol-lactone hydrolase and hydrolase enzymes to form β-ketoadipate. Finally, β-ketoadipate is converted into β-ketoadipyl-CoA by β-ketoadipate succinyl-CoA transferase, and subsequently into succinyl-CoA and acetyl-CoA via β-ketoadipyl-CoA thiolase, entering the tricarboxylic acid (TCA) cycle and fatty acid biosynthesis [36]. The β-KAP allows microbes to degrade a wide range of aromatic pollutants, converting recalcitrant compounds into energy-rich metabolites that sustain microbial growth while detoxifying the environment. This pathway demonstrates the versatility and efficiency of microbial metabolism in bioremediation applications, especially for persistent compounds like oxybenzone [37]. Mechanisms of microbial degradation is summarize in Figure 2.

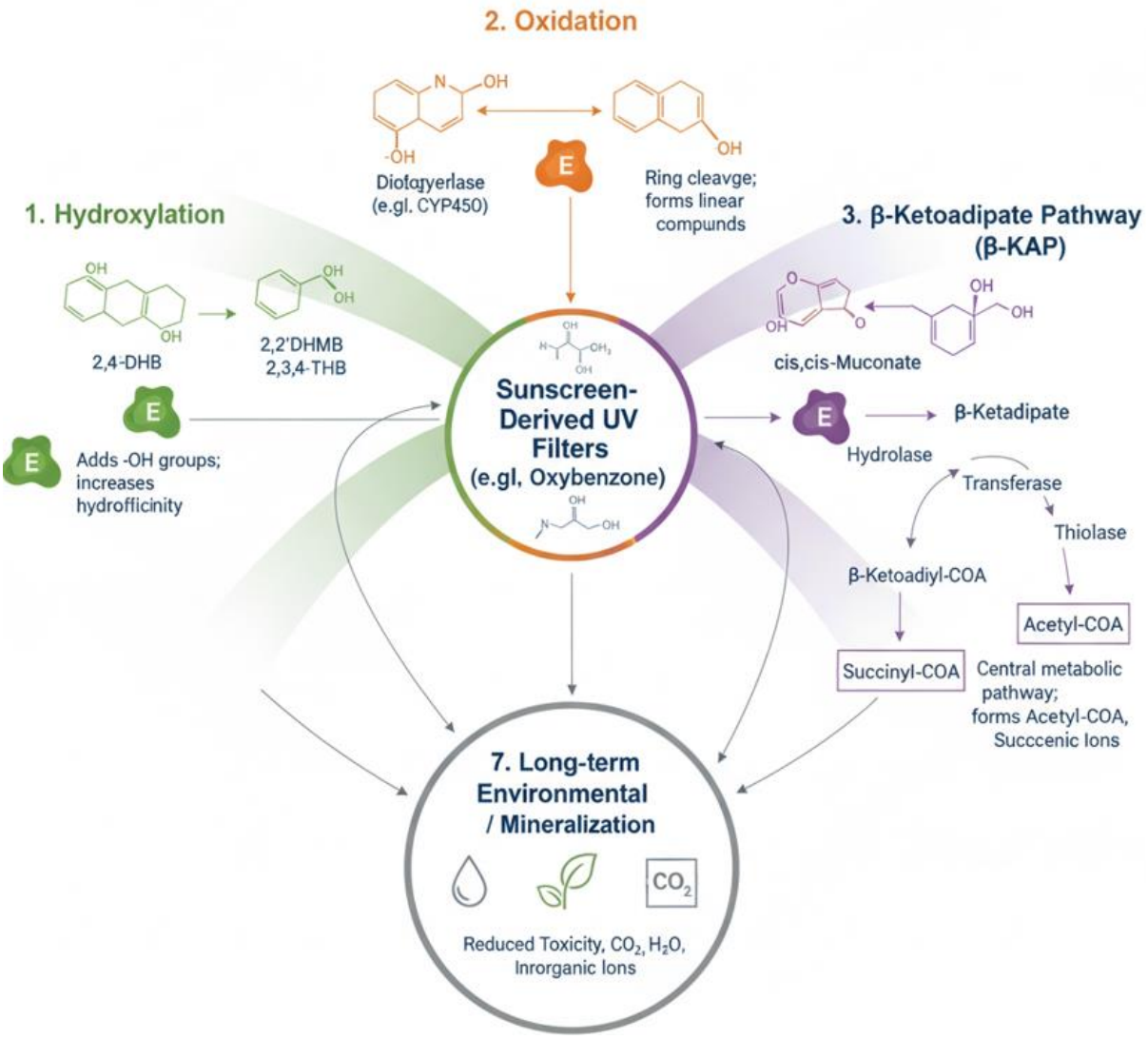

Figure 2. Mechanisms of microbial degradation.

7. Treatment Technologies for UV Filter Degradation

The removal of persistent UV filters, such as oxybenzone and octinoxate, from aquatic environments has become a critical concern due to their toxicological effects on marine ecosystems. Several biological treatment approaches have been developed, ranging from microbial bioaugmentation to advanced bioreactors. These strategies aim to exploit the enzymatic and metabolic capabilities of microorganisms or algae to degrade or transform these contaminants into non-toxic metabolites. Treatment technologies for biodegradation of sunscreen agents is summarized in Table 2.

Table 2. Treatment technologies for biodegradation of sunscreen agents.

Technology	Mechanism	Microorganisms / Biocatalysts	Key Reactions	Efficiency / Highlights	Reference
Non-Sterile Culture (Bioaugmentation)	Biological degradation in non-sterile conditions	*Klebsiella huaxiensis W2*	CYP450-mediated hydroxylation, dealkylation, C–C cleavage	99.3% degradation in 120 h; non-toxic byproducts	[33,37]
Revolving Algae Bioreactor (RAB)	Algal biofilm absorption and enzymatic degradation	Green algae, diatoms, cyanobacteria	Phase I oxidation/hydrolysis; Phase II conjugation; Phase III sequestration	94–98% degradation; low bioaccumulation	[38–42]
Fluidized Bed Bioreactor (FBBR)	Attached-growth system with fluidized media; mediator-assisted enzymatic oxidation	*Trametes versicolor*	Glycosylation; CYP450 oxidation; laccase-mediated oxidation	95–100% removal; scalable; robust biofilm formation	[43–46]

7.1. Bioaugmentation.

Bioaugmentation involves the introduction of a pollutant-degrading microorganism into a contaminated environment or medium to accelerate the degradation process. In a recent study [33], the bacterial strain *Klebsiella huaxiensis* W2 was isolated from activated sludge in a wastewater treatment plant in Turkey and demonstrated high potential for degrading oxybenzone under non-sterile conditions. The strain was selected through rigorous laboratory isolation and screening to identify its superior degradation capabilities. Wastewater medium containing 1 g/l of oxybenzone was inoculated with varying amounts of the preculture without sterilization to simulate real non-sterile environments, which also contained endogenous microorganisms. The inoculum size significantly influenced the degradation efficiency: 1 ml, 1.5 mL, and 2 ml of inoculum resulted in 83.14%, 98.02%, and 99.3% degradation, respectively. Maximum degradation reached 99.33% within 120 hours, with most of the oxybenzone removed during the first 72 hours. Interestingly, hydrolytic enzymes such as laccase and manganese peroxidase were not involved in this process; instead, *Klebsiella huaxiensis* utilized CYP450 monooxygenases to catalyze reactions including hydroxylation, epoxidation, dealkylation, and carbon–carbon bond cleavage. Two byproducts were identified and confirmed to be non-toxic to fibroblast cell lines, indicating both efficiency and safety. The study also demonstrated that increasing inoculum size could enhance the dominance of the target strain over endogenous microorganisms, optimizing biodegradation in practical applications.

7.2.Revolving Algae Bioreactor (RAB).

The RAB is a biological treatment system primarily used for nutrient removal but has shown potential for degrading UV filters (Figure 3). The system operates using a revolving belt as a growth substrate for algae, which rotates through a wastewater reservoir. This movement allows algae to absorb nutrients and contaminants from wastewater while being exposed to sunlight and air for photosynthesis and respiration [38, 39]. In a study on oxybenzone biodegradation [40], a mixture of pharmaceuticals and personal care products (PPCPs) was added to Bold's Basal Medium (BMM) to simulate real wastewater conditions. The RAB system contained a mixed culture of green algae, diatoms, and cyanobacteria. After biofilm stabilization, the system was operated over four weekly cycles. The results showed that, despite low bioaccumulation of oxybenzone (2–6% in biomass), the degradation efficiency was remarkably high: 98% for high pollutant loading and 94% for low loading. The biodegradation process is primarily enzymatic, involving three phases: Phase I (oxidation, reduction, hydrolysis to increase hydrophilicity), Phase II (conjugation with glucoside, malonyl-glucoside, or sulfate), and Phase III (sequestration into vacuoles or cell walls) [41,42]. The RAB system offers sustainable, low-energy treatment potential for persistent UV filters, leveraging algae metabolism for pollutant removal.

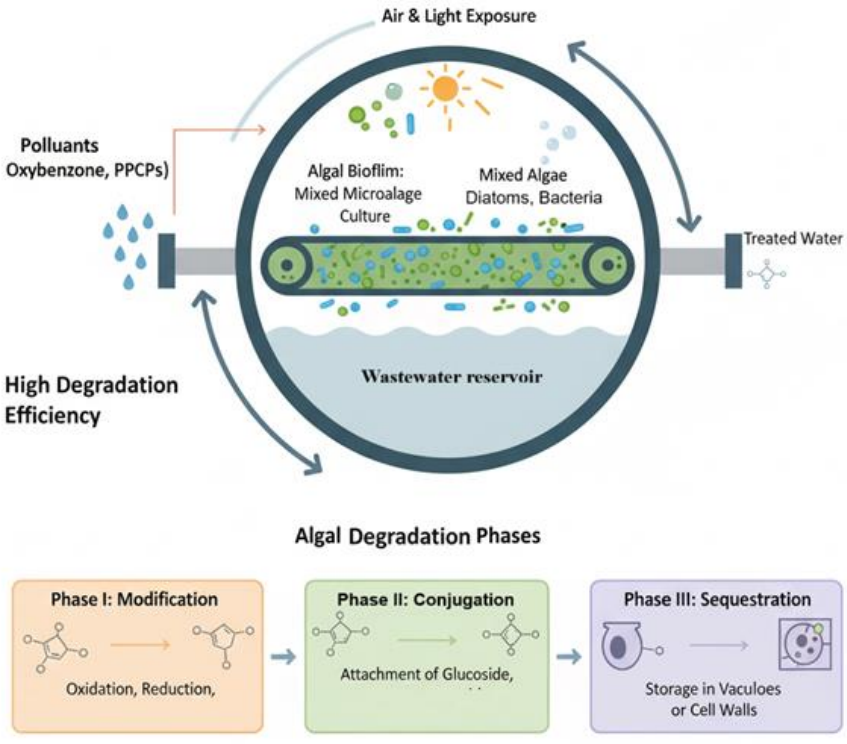

Figure 3. Mechanism of RAB.

7.3. Fluidized Bed Bioreactor (FBBR).

The FBBR is an advanced system for wastewater treatment, particularly effective for degrading complex organic pollutants such as UV filters. The FBBR contains solid particles or biomass carriers that are suspended by upward-flowing fluid, providing excellent mixing and enhanced contact between microorganisms and contaminants [43, 44]. This attached-growth system enables the formation of thick biofilms with increased biomass concentration and surface area,

optimizing degradation efficiency. Trametes versicolor, a white-rot fungus, is commonly employed in FBBRs due to its non-specific enzymatic capabilities, including extracellular ligninolytic enzymes such as laccases, lignin peroxidases, and manganese peroxidases [45]. Laccase-mediated oxidation, particularly in the presence of mediators, can effectively degrade oxybenzone, achieving 95–100% removal [46]. Biodegradation often begins with glycosylation, attaching sugar residues to the hydroxyl groups of oxybenzone (via UDP-glucosyltransferase or UDP-xylosyltransferase), followed by oxidation through CYP450 monooxygenases. Metabolites such as benzophenone-1 (BP1), 4,4-dihydroxybenzophenone (4DHB), and 4-hydroxybenzophenone (4HB) are formed during this process. The FBBR combines high microbial attachment, enhanced mass transfer, and mediator-assisted enzymatic activity, making it a highly efficient and scalable system for UV filter remediation (Figure 4).

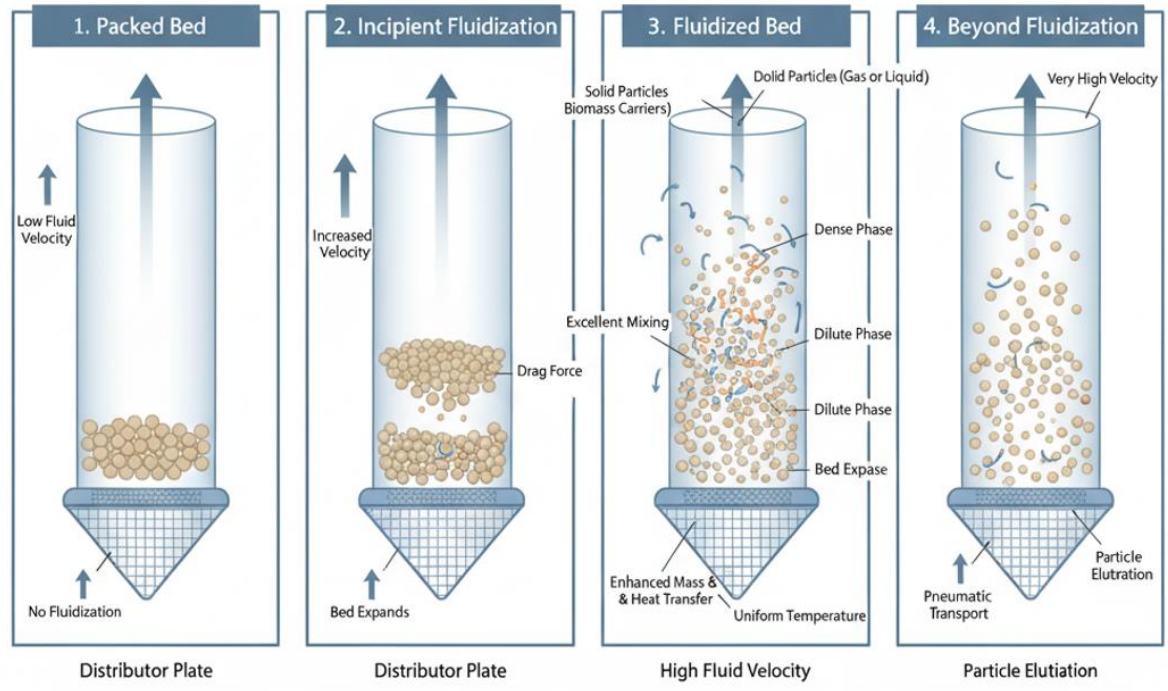

Figure 4. Mechanism of FBBR.

8. Advantages and Limitations of Microbial-Based UV Filter Treatment Technologies

Biological treatment technologies, including microbial and algae-based systems, are increasingly investigated as alternatives for removing persistent UV filters from wastewater and natural water bodies. While these approaches offer significant environmental and operational benefits, they also have inherent limitations that must be considered for real-world application.

8.1.Advantages.

8.1.1. High biodegradability and pollutant removal efficiency.

Conventional wastewater treatment processes, including primary sedimentation and secondary activated sludge treatment, are often insufficient for eliminating UV filters such as benzophenone-3 (BP3). Studies indicate that primary treatment alone can discharge UV filters at rates 7–13 times higher than combined primary and secondary treatment, even at low

concentrations [47, 48]. Advanced treatment techniques such as coagulation, flocculation, microfiltration, and ozonation achieve only 28–31% removal of BP3 [49], while activated sludge combined with trickling filters removes less than 15% of these compounds [50]. These processes largely transfer organic contaminants to another phase rather than fully degrade them [51].

In contrast, microbial degradation demonstrates markedly higher efficiency due to the enzymatic capabilities of microorganisms. Certain bacteria, such as *Rhodococcus sp. S2-17* and *Sphingomonas wittichii*, have been shown to completely degrade BP3 within 7–10 days under laboratory conditions [52, 53]. Enzymes like CYP450 monooxygenases catalyze the cleavage of aromatic rings in UV filter molecules, converting them into smaller, less harmful metabolites. Microbial degradation, therefore, not only achieves higher removal rates compared to conventional wastewater treatment but also transforms the contaminants into more environmentally benign substances, making it a sustainable and effective approach.

8.1.2. Environmentally sustainable technology.

Microbial and algae-based treatment systems offer eco-friendly advantages beyond high degradation efficiency. Microorganisms secrete enzymes that operate under mild conditions, are highly specific, and can be genetically modified to target a range of pollutants [54, 55]. The production and application of these enzymes in industrial-scale bioreactors are cost-effective, allowing continuous cultivation, easy handling, and the possibility of enzyme reuse [55]. For example, the RAB can simultaneously remove UV filters and heavy metals, including zinc, chromium, nickel, and manganese, with near-complete elimination of chromium and nickel, and significant removal of manganese and zinc [38]. The RAB system is energy-efficient, requiring less power than advanced oxidation processes (AOP), which are energy-intensive and can produce secondary contaminants if not carefully managed [56, 57]. The vertical belt configuration of the RAB enhances sunlight exposure, gaseous exchange, and nutrient absorption, leading to higher biomass productivity. Biofilm harvesting through scraping reduces both operational and energy costs, lowering production expenses by approximately 30% while maintaining yield [40, 58]. Additionally, the resulting algal biomass, which is over 80% water, can be reused directly as feedstock without costly dewatering, providing added economic and environmental benefits [59, 60].

8.2.Limitations

8.2.1. Sensitivity to environmental conditions.

Despite their advantages, microbial enzymes and organisms are highly sensitive to environmental factors, which can limit their effectiveness in practical applications. Variations in temperature, pH, and oxygen availability can significantly affect enzymatic activity and microbial growth [61]. High temperatures may denature enzymes, while suboptimal pH levels can alter the surface charge of microbial cells, influencing biofilm formation [62]. For RAB systems, light distribution is critical, as microalgae require sunlight for photosynthesis. Uneven light can cause photoinhibition at the surface and photolimitation at deeper layers, reducing biofilm growth [62]. Similarly, carbon dioxide availability directly affects microalgal biomass and metabolite production. Studies have shown that *Chlorella vulgaris* achieves maximum biomass at 13% CO_2, whereas lower CO_2 concentrations result in reduced growth [64].

Optimizing light and carbon simultaneously is therefore essential for maintaining efficient biodegradation and maximizing productivity [63, 65].

8.2.2. Time-Intensive degradation processes.

Although microbial systems can achieve high removal rates, the degradation process is relatively slow compared to chemical treatments such as AOP. Advanced oxidation processes generate reactive oxygen species that rapidly degrade organic contaminants, often achieving fast and high removal rates]. In comparison, algae-based RAB systems or microbial cultures may require several weeks to fully degrade contaminants like octinoxate, with some studies reporting a 30-day treatment period [57]. While AOP offers speed and efficiency, it comes with drawbacks such as high energy consumption and potential formation of secondary pollutants, which may pose additional environmental risks.

9. Conclusion

Microbial and algae-based treatment technologies represent promising alternatives to conventional wastewater treatments for removing persistent UV filters, such as BP3 and octinoxate. These biological approaches demonstrate high biodegradability due to the secretion of potent enzymes like CYP450 monooxygenases, laccases, and peroxidases, which can catalyze the breakdown of aromatic compounds into smaller, less toxic metabolites. Algae-based systems, such as the RAB, further contribute by simultaneously removing heavy metals, enhancing biomass productivity, and offering a cost-efficient, low-energy treatment option. The biodegradation processes are environmentally sustainable, operating under mild conditions with minimal secondary pollution, and allowing for the reuse of microbial or algal biomass as a valuable resource. However, these technologies also present practical limitations. Microbial and enzymatic activity is highly sensitive to environmental factors, including temperature, pH, oxygen availability, light intensity, and carbon dioxide concentration, which can affect degradation efficiency and biofilm formation. Furthermore, microbial-based treatments generally require longer timeframes compared to chemical methods like AOP, which can rapidly degrade organic pollutants but may produce secondary contaminants and demand high energy inputs. Microbial and algae-based systems provide an eco-friendly, effective, and scalable approach for the removal of UV filters from wastewater, but optimization of environmental conditions and operational parameters is essential to maximize efficiency. These technologies hold significant potential for sustainable water treatment, particularly in contexts where conventional methods are inadequate for eliminating persistent organic contaminants. Integrating microbial degradation with engineered bioreactors could achieve higher removal efficiencies while reducing environmental impacts and operational costs.

Competing Interests

The authors declare no competing financial or non-financial interests related to this work.

Author Contribution

All authors conceptualized the study, literature review, data analysis, and approved the final version.

Data Availability

All data generated or analyzed during this study are included in this manuscript. Additional datasets are available from the corresponding author upon reasonable request.

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