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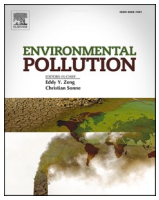

Review

Advances in identifying and managing emerging contaminants in aquatic ecosystems: Analytical approaches, toxicity assessment, transformation pathways, environmental fate, and remediation strategies☆

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ABSTRACT

Emerging contaminants (ECs) are increasingly recognized as threats to human health and ecosystems. This review evaluates advanced analytical methods, particularly mass spectrometry, for detecting ECs and understanding their toxicity, transformation pathways, and environmental distribution. Our findings underscore the reliability of current techniques and the potential of upcoming methods. The adverse effects of ECs on aquatic life necessitate both in vitro and in vivo toxicity assessments. Evaluating the distribution and degradation of ECs reveals that they undergo physical, chemical, and biological transformations. Remediation strategies such as advanced oxidation, adsorption, and membrane bioreactors effectively treat EC-contaminated waters, with combinations of these techniques showing the highest efficacy. To minimize the impact of ECs, a proactive approach involving monitoring, regulations, and public education is vital. Future research should prioritize the refining of detection methods and formulation of robust policies for EC management.

1. Introduction

Emerging contaminants (ECs) are a group of chemical compounds that have gained increased attention in recent years due to their potential to adversely affect human health and the environment (Cheng et al., 2021). These compounds are characterized by limited regulatory oversight, scarcity of data on their environmental occurrence, fate, and toxicological effects, and potential for causing adverse ecological impacts (Daughton, 2014). ECs encompass a wide range of substances originating from various anthropogenic activities, and can be categorized into several types including pharmaceuticals and personal care products (PPCPs), per- and polyfluoroalkyl substances (PFAS), microplastics, nanomaterials, and endocrine-disrupting chemicals (EDCs). PPCPs include prescription and over-the-counter drugs, veterinary pharmaceuticals, and a variety of personal care products, such as fragrances, sunscreens, and antimicrobial agents (Gavrilescu et al., 2015). Fig. 1 illustrates the major sources and compounds of ECs and showcases the diversity of ECs originating from multiple sources. These compounds enter aquatic environments through wastewater treatment plant effluents, leaching from landfills, and agricultural runoff from livestock manure.

PFAS are a group of synthetic chemicals, including perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS), which have been widely used in industrial and consumer products for their resistance to heat, water, and oil (Ramírez Carnero et al., 2021). PFAS can contaminate water sources through industrial discharges, firefighting foams, and the degradation of consumer products. Microplastics are small plastic particles (<5 mm) that originate from the breakdown of larger plastic debris, microbeads in personal care products, or synthetic fibres from textiles (Xu et al., 2020a). They can enter aquatic environments through

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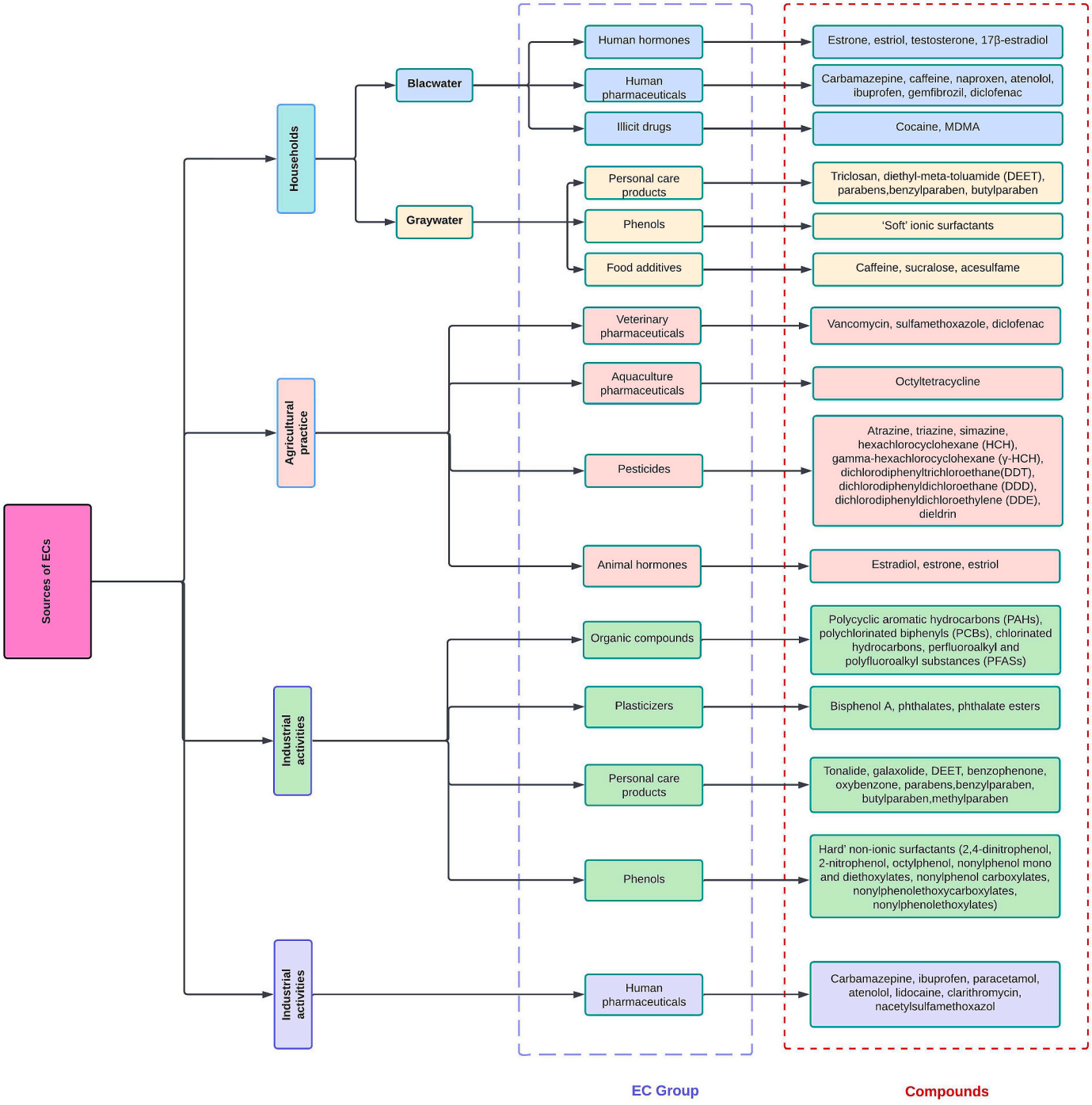

Fig. 1. Overview of the primary sources and categories of ECs in aquatic ecosystems.

wastewater treatment plant effluents, stormwater runoff, and atmospheric deposition. Nanomaterials are materials with at least one dimension between 1 and 100 nm that exhibit unique physicochemical properties due to their small size (Nanomaterials, 2020). They are used in various industrial and consumer products, such as electronics, cosmetics, and food packaging. The release of nanomaterials into aquatic environments can occur through wastewater treatment plant effluents, industrial discharges, and product degradation. EDCs are a diverse group of natural and synthetic chemicals that can interfere with the endocrine system of organisms, leading to developmental, reproductive, neurological, and immune system disorders (Maqbool et al., 2016). EDCs include some pesticides, industrial chemicals, and naturally occurring substances, such as hormones and phytoestrogens. They can enter aquatic environments through agricultural runoff, wastewater treatment plant effluents, and industrial discharges.

ECs can cause potential ecological and human health risks. Despite their low concentrations in the environment, ECs can exert chronic toxic effects on aquatic organisms, leading to endocrine disruption, reproductive impairment, and developmental abnormalities (Marlatt et al., 2022). In addition, some ECs can bioaccumulate and biomagnify in food webs, posing risks to higher trophic levels, including humans (Ward et al., 2010). Moreover, ECs can undergo complex transformation processes in the environment, generating potentially more toxic or persistent by-products (Sang et al., 2014). Therefore, understanding the

Table 1
Regulatory standards set by global authorities, accompanied by the predicted no effect concentration (PNEC) and derived drinking water equivalent limit (DWEL) values for chosen ECs (reprinted with permission of Elsevier from (Parida et al., 2021)).

Contaminants	Statutory guidelines (ng/L)				PNEC (ng/L)	Refs.	DWEL (μg/L) [h]				
	EU	US EPA	ADWG	WHO			Asia	Europe	America	Africa	Australia
17 β-estradiol		175[b]	175[d]	1	1.6	Zhou et al. (2019)	1.60	1.70	1.20	1.31	1.70
Atenolol	150,000[a]		——	——	100,000	Zhou et al. (2019)	87.60	93.60	69.70	71.00	92.70
Benzophenone-3	——	152,000[c]	——	——	6000	You et al. (2015)	64,902.90	69,342.20	51,641.20	52,648	68,723.40
Caffeine	87,000[b]	——	350[d]	——	320	Zhou et al. (2019)	18,4973.30	197,625.30	147,177.60	150,046.80	195,861.70
Carbamazepine	2,000[a]	——	100,000[d]	——	10	Zhou et al. (2019)	11.00	11.78	8.70	8.90	11.60
Ciprofloxacin	100	——	250,000[d]	——	100	Zhou et al. (2019)	51.90	55.40	41.30	42.10	54.90
Codeine	——	——	50,000[d]	——	976	Zhou et al. (2019)	64.90	69.34	51.64	52.65	68.72
Erythromycin	200	——	17,500[d]	——	103	Tran et al. (2018)	1298.00	1386.80	1032.80	1052.90	1374.40
Ibuprofen	11[a]	——	——	——	10	Rivera-Jaimes et al. (2018)	3569.60	3813.80	2840.20	2895.60	3779.70
Iohexol	——	——	——	——	$>1.0 \times 10^7$	Tran et al. (2018)	4,056,432.00	4,333,888.80	3,227,580.40	3,290,500	4,295,212.70
Iopromide	——	——	750,000[d]	——	3.7×10^8	Tran et al. (2018)	2,704,277.20	2,889,247.70	2,151,711.60	2,193,657.80	2,863,463.70
Norfloxacin	——	——	400,000[d]	——	1400	Zhou et al. (2019)	6165.70	6587.50	4905.90	5001.50	6528.70
Octocrylene	——	——	——	——	2300	Tran et al. (2018)	24,825.30	26,523.40	19,752.70	20,137.80	26,286.70
Paracetamol	——	——	175,000[d]	——	1400	You et al. (2015)	11,033.40	11,788.10	8779.0	8950.10	11,682.90
TCEP	4000	——	1,000[d]	——	——	Loos et al. (2017)	713.90	762.76	568.05	579.13	755.95
Trimethoprim	120,000[a]	——	70,000[d]	——	160	Tran et al. (2018)	136.20	145.60	108.40	110.50	144.30
Amphetamine	——	——	——	——	3800	Zhou et al. (2019)	9735.40	10,401.30	7746.10	7897.20	10,308.50
Bisphenol-A	240[a]	77,000[c]	200,000[d]	100	60	You et al. (2015)	1622.50	1733.50	1291.00	1316.20	1718.00
Diazinon	12[a], 100[g]	——	3,000[d]	——	0.26	Tsuda et al. (2011)	6.40	6.90	5.10	5.20	6.80
Diuron	70[a], 1,800[e]	——	200	——	65,000	(Mansano et al., 2016; Ramírez-Morales et al., 2021)	64.90	69.30	51.60	52.60	68.70

[f](Zini and Gutterres, 2021).
[a] (Korkaric et al., 2019).
[b] (Carvalho et al., 2016).
[c] (Careghini et al., 2015).
[d] (NHMRC, 2008).
[e] (Radley-Gardner et al., 2016).
[g] (Seibert et al., 2020).
[h] (Sharma et al., 2019).

Table 2
Various analytical approaches used for the detection of ECs.

Analytical approach	Technique	Principle	Targeted contaminants	Matrix	Instrument/ pre-treatment cost	Accuracy/ detection limit	Suitability for water types	Refs.
Chromatography	GC-MS	Separation of volatile compounds with mass spectrometry detection	PAHs, VOCs, PCBs	Air, Water, Soil	High	High/Ppb levels	Freshwater, wastewater	López et al. (2013)
	LC-MS/MS	Separation of non-volatile compounds with tandem mass spectrometry detection	PFAS, pharmaceuticals, hormones	Water, soil, biological samples	Very high	Very high/ Ppt levels	Drinking water, wastewater	Ammann et al. (2014)
Spectroscopy	UV–Vis	Absorption of ultraviolet–visible light	Metal ions, dyes	Water, soil	Moderate	Moderate/ Ppm levels	Surface water, groundwater	Zahmatkesh et al. (2022)
	FTIR	Absorption of infrared radiation	Organic compounds, polymers	Solids, liquids, gases	High	Moderate/ Ppm levels	Not typically suited for water	Quintelas et al. (2020)
	Raman	Inelastic scattering of monochromatic light	Inorganic compounds, crystal structures	Solids, liquids, gases	High	Moderate/ Ppm levels	Freshwater, saline water	Almaviva et al. (2022)
Electrochemical	Voltammetry	Measurement of the current as a function of applied voltage	Metal ions, organic compounds	Water, soil, biological samples	Moderate	High/Ppb levels	Freshwater, wastewater	Raymundo-Pereira et al. (2017)
	Amperometry	Measurement of current resulting from the redox reaction of an electroactive species	Organic compounds, metal ions	Water, Soil, Biological Samples	Low	High/Ppb levels	Freshwater, Drinking Water	Rodriguez-Mozaz et al. (2007)
	Potentiometry	Measurement of the potential of a solution between two electrodes	Ion concentrations, pH	Water, soil, biological samples	Low	Moderate/ Ppm levels	Any water type	Biyana Regasa and Nyokong (2022)
Immunoassays	ELISA	Enzyme-linked immunosorbent assay	Hormones, antibiotics, pesticides	Water, soil, food, biological samples	Moderate	High/Ppb levels	Drinking water, groundwater	Ding et al. (2021)
	Lateral Flow Assays	Chromatographic immunoassay with a visible endpoint	Pathogens, pesticides, toxins	Water, food, environmental samples	Low	Moderate/ Ppm to Ppb levels	Surface water, drinking water	Orlov et al. (2022)
Biosensors	Optical	Measurement of changes in optical properties (absorption, fluorescence, etc.)	Biomolecules, metal ions	Water, soil, air, biological samples	High	High/Ppb levels	Any water type	Zhou et al. (2021)
	Electrochemical	Measurement of changes in electrical properties (current, potential, etc.)	Biomolecules, metal ions	Water	Moderate	High/Ppb levels	Drinking water, wastewater	Kanoun et al. (2021)

sources, fate, and effects of ECs is essential for the development of appropriate monitoring, risk assessment, and management strategies. Despite the growing awareness of the hazards posed by ECs, gaps remain in our understanding of their detection, toxicity, transformation, fate, remediation, and mitigation. Advanced analytical techniques have been developed to identify and quantify ECs in various environmental matrices (Rasheed et al., 2019), but the continuous emergence of new contaminants and the complex nature of environmental samples pose significant analytical challenges (Lei et al., 2023). Furthermore, while various in vitro and in vivo models have been employed to assess the toxicity of ECs, extrapolating laboratory findings to ecological risk assessment in natural systems remains a major challenge (Archer et al., 2017; Rodrigues et al., 2015). Addressing these gaps in knowledge is essential for developing effective remediation strategies and mitigation measures. Current remediation technologies, such as advanced oxidation processes and adsorption, have shown promise for the removal of ECs from aqueous environments (Ahmed et al., 2021); however, the development of cost-effective, efficient, and sustainable treatment options remains an ongoing challenge (Sutherland and Ralph, 2019). Moreover, while some regulatory efforts have been made to control the release of ECs into aquatic environments, comprehensive monitoring and surveillance programs, as well as evidence-based policy recommendations, are necessary to minimize the impacts of these contaminants (Wielinga et al., 2014).

The aim of this paper is to provide a comprehensive overview of the current state of knowledge on emerging contaminants in aquatic ecosystems. It focuses on the detection methods, toxicity assessment, transformation pathways, environmental fate, remediation strategies, and mitigation measures for ECs. By integrating the latest research findings and highlighting novel approaches, we aim to inform future research and policy efforts aimed at protecting aquatic environments from the detrimental effects of emerging contaminants. The paper is organized as follows: Section 2 presents the regulatory frameworks for managing the environmental effects of emerging contaminants. Section 3 discusses the analytical approaches for the detection of emerging contaminants, including sample collection and preparation, advanced instrumental methods, and emerging techniques. Section 4 presents an overview of toxicity assessment methods including aquatic toxicity testing, in vitro and in vivo models, and ecotoxicological risk assessment. Section 4 delves into the transformation pathways and

environmental fate of ECs, covering physical, chemical, and biological processes. Section 5 examines the current remediation strategies for emerging contaminants, such as physical and chemical treatments, biological treatments, and combined and emerging remediation technologies. Section 6 presents mitigation measures and policy recommendations including source control, monitoring and surveillance programs, regulations, and public awareness and education. Finally, Section 7 concludes the paper by summarizing the key findings, discussing implications for environmental management and policy, and identifying future research directions.

2. Regulatory frameworks for managing the environmental effects of emerging contaminants

Various organizations have established guidelines to control the environmental impact of emerging contaminants. These standards define acceptable concentrations in water bodies, ensuring safety against prolonged exposure risks. There are no strict global guidelines for emerging contaminants (ECs) due to challenges like inadequate data and monitoring difficulties (Kovalakova et al., 2020). However, certain initiatives have been adopted in different countries.

- In 2007, the European Parliament Committee proposed the inclusion of specific ECs, like carbamazepine and bisphenol-A, in their priority substance list (Jensen, 2007).
- Various international bodies have proposed admissible EC concentrations in aquatic environments, as detailed in Table 1.
- The United States Environmental Protection Agency (USEPA) updates its priority substances list every five years based on human impact. They have identified some contaminants such as diuron and 17 β-estradiol, among others, as hazardous materials and highlighted them as priority pollutants in the fourth drinking water candidate list (USEPA, 2016).
- In 2015, USEPA highlighted the need for thorough research on the toxic effects of OP flame retardants (USEPA, 2015).
- The EU set limits for 33 ECs in aquatic environments in 2008. However, some contaminants in water bodies, such as ciprofloxacin and carbamazepine, often exceed the EU's guidelines (Samal et al., 2022).
- In 2011, WHO updated the Guidelines for Drinking Water Quality and included previously unlisted chemicals (Herschy et al., 2012).
- Australia's Drinking Water Guidelines (ADWG) under the National Health and Medical Research Council (NHMRC) has outlined guidelines for ECs in treated sewage, with many contaminants such as ciprofloxacin, carbamazepine, TCEP, and codeine found in Australian wastewater treatments below their maximum allowed concentrations (NHMRC, 2008).

3. Analytical approaches for the detection of emerging contaminants

Accurate detection and quantification of ECs in aquatic environments are critical to understanding their occurrence, fate, and potential ecological and human health risks. Over the past few decades, a wide array of analytical techniques has been developed to identify and measure ECs in various environmental matrices, such as water, sediment, and biota (Paszkiewicz et al., 2022). These techniques typically involve a combination of sample collection and preparation, separation, detection, and identification methods, which are continually evolving to meet the challenges posed by the complex nature of environmental samples and the continuous emergence of new contaminants (Shyamalagowri et al., 2021). Different analytical approaches in detecting ECs are presented in Table 2.

3.1. Sample collection and preparation

Sample collection and preparation are crucial steps in the analysis of ECs, as this significantly influences the quality and reliability of the obtained results. Depending on the type of environmental matrix and target analytes, different sampling strategies, such as grab sampling, composite sampling, and passive sampling, can be employed to obtain representative samples (Valenzuela et al., 2020). Following collection, samples often require pre-treatment steps to concentrate the target analytes and remove interfering substances. Common sample preparation techniques include solid-phase extraction (SPE), liquid-liquid extraction (LLE), and solid-phase microextraction (SPME) (da Silva Sousa et al., 2021). For microplastics in aquatic environments, established "gold" rules often guide sample collection and pre-treatment. Typically, grab sampling is preferred to capture a snapshot of microplastic contamination at a specific location and time. Pre-treatment may involve density separation methods to isolate microplastics from natural organic materials, followed by visual inspection and further analysis (Mattsson et al., 2022). For organic compounds like 1,4-dioxane, composite sampling might be the most representative method, especially in groundwater studies. Solid-phase extraction (SPE) is commonly used as a pre-treatment technique for these compounds (Andaluri and Suri, 2017). In the case of chlorinated volatile organic compounds (CVOCs), passive sampling is often used to monitor long-term exposure and pollutant levels in the environment. Liquid-liquid extraction (LLE) can be used as a pre-treatment method to concentrate CVOCs and remove potential contaminants (Park et al., 2005). For heavy metals such as lead and mercury, grab or composite sampling may be appropriate, depending on the environmental matrix. Solid phase microextraction (SPME) is often employed as a pre-treatment step to concentrate the metal ions and remove interfering ions or substances (Jagirani and Soylak, 2020). Careful sample collection is crucial when targeting pharmaceuticals and personal care products due to the low concentrations typically encountered in environmental samples. SPE or liquid chromatography are commonly employed as pre-treatment steps to concentrate the target PPCPs (Kosma et al., 2010; Yu et al., 2012). For pesticides, passive samplers that sequester hydrophobic organic compounds are often the method of choice, especially in aquatic ecosystems. LLE and SPE are common pre-treatment techniques to concentrate these compounds and remove co-extracted matrix components (Jordan et al., 2009). For per- and polyfluoroalkyl substances (PFAS), composite sampling methods are often used due to the persistence and bioaccumulative nature of these compounds. Anion-exchange solid-phase extraction (AX-SPE) is commonly used for the pre-treatment of PFAS (Nakayama et al., 2020).

3.2. Advanced instrumental method: mass spectrometry

Mass spectrometry coupled with liquid chromatography (LC-MS/MS) or gas chromatography (GC-MS/MS) have become the gold standard for the analysis of ECs due to their high sensitivity, selectivity, and wide applicability (Martín-Pozo et al., 2019). For example, (Ammann et al., 2014) demonstrated the applicability of LC-MS/MS in identifying and quantifying pharmaceuticals and EDCs in wastewater treatment plant effluents. Similarly, (Audy et al., 2018) employed GC-MS/MS to analyse perfluoroalkyl substances (PFAS) in indoor and outdoor air samples. LC-MS/MS is particularly suitable for the analysis of polar and non-volatile compounds, such as pharmaceuticals and EDCs, while GC-MS/MS is commonly used for less polar and volatile compounds, such as PFAS and some pesticides (Baroudi et al., 2020). The use of tandem mass spectrometry (MS/MS) allows for the identification and quantification of target analytes based on their specific mass-to-charge ratios and fragmentation patterns, providing a high degree of confidence in the obtained results (Bielawski et al., 2009). Another study by (Comtois-Marotte et al., 2017) highlighted the effectiveness of using high-resolution mass spectrometry (HRMS) for the detection and

identification of novel ECs in surface waters, further showcasing the versatility of mass spectrometry in EC analysis.

While mass spectrometry, particularly when coupled with liquid chromatography (LC-MS/MS) or gas chromatography (GC-MS/MS), has been lauded for its high sensitivity, selectivity, and broad applicability, it is not without limitations. One significant drawback is the high cost associated with the initial setup and ongoing operational expense, making it a less feasible option for smaller laboratories or facilities in developing countries (Stanczyk and Clarke, 2010). Furthermore, the operation of these advanced instruments demands specialized training and expertise, which can limit their routine use in less specialized settings. Matrix effects, wherein the chemical background interferes with the accurate quantification of target analytes, are also a concern. These effects can be particularly pronounced in LC-MS/MS and necessitate careful management to ensure accurate results (Núñez et al., 2005). Sample preparation, another important phase, can be both time-consuming and elaborate, potentially leading to the degradation of sensitive compounds. Additionally, the dynamic range of mass spectrometers can be limiting, affecting their ability to accurately quantify samples with highly variable analyte concentrations (Wang et al., 2017a). Regular calibration and maintenance are also required to keep these instruments functioning at their optimal level, which can be both time-intensive and costly. These limitations highlight the areas that emerging technologies in the field aim to address, i.e., they seek to improve on aspects like sensitivity, selectivity, and the method's applicability across a wide array of compounds and matrices.

3.3. Emerging instruments for EC detection and future perspectives

Recent advances in analytical chemistry have led to the development of novel techniques and methodologies for the detection of emerging contaminants. These emerging techniques aim to address the limitations of existing methods, such as sensitivity, selectivity, and applicability to a wide range of compounds and matrices. Below is a presentation of some of these emerging techniques.

3.3.1. Ambient mass spectrometry (AMS)

AMS techniques, such as direct analysis in real-time (DART) and desorption electrospray ionization (DESI), allow for the rapid and direct analysis of samples with minimal sample preparation (Cheng et al., 2019). These techniques have shown potential for the high-throughput screening of ECs in various environmental matrices, although their quantitative capabilities and applicability to complex samples still need further improvement (Aszyk et al., 2018). For instance, (Strittmatter et al., 2012) demonstrated the potential of DESI-MS for the analysis of pharmaceuticals in wastewater samples, while (Bridoux and Schramm, 2018) used DART-MS to detect and identify various contaminants in water and soil samples. Moreover, (Cheng et al., 2020) employed AMS techniques to analyse pesticides in fruits and vegetables, showcasing the broad applicability of these methods across different matrices. These studies underscore the potential of AMS in detecting and characterizing ECs, but also highlight the need for further research to enhance their quantitative performance and applicability to more complex environmental samples.

3.3.2. High-resolution mass spectrometry (HRMS)

HRMS instruments, such as Orbitrap and time-of-flight (TOF) mass spectrometers, offer significantly higher mass accuracy and resolving power compared to conventional MS/MS systems, enabling the identification of unknown contaminants through the elucidation of their chemical structures (Kaufmann and Teale, 2016). HRMS has been increasingly used for the non-targeted analysis of ECs in aquatic environments, facilitating the discovery of new contaminants and the investigation of their transformation products and metabolites (Zhang et al., 2023a). In recent studies, HRMS has been successfully employed to identify previously unreported contaminants in wastewater treatment plant effluents and trace the fate of polar organic contaminants in river systems (Muter and Bartkevics, 2020).

High-resolution mass spectrometry (HRMS) instruments, notably Orbitrap and time-of-flight (TOF) mass spectrometers, stand out with their elevated mass accuracy and resolving power in comparison to conventional MS/MS systems. This capability facilitates the identification of unknown contaminants and the detailed understanding of their chemical structures (Hernández et al., 2012). Over the years, the adoption of HRMS for the non-targeted analysis of ECs in aquatic settings has risen, paving the way to unveil new contaminants and decipher their transformation products and metabolites (Bajkacz et al., 2022). Recent research has extensively highlighted the capabilities of HRMS in the environmental domain. For instance, (Hernández et al., 2014) employed HRMS to discover a range of unreported pharmaceutical contaminants in water, which illuminated the expansive reach of such pollutants. Another study by (Philip et al., 2022) harnessed the potential of HRMS to trace the degradation pathways of complex pesticides in freshwater environments, providing valuable insights into their environmental fate.

A comparative analysis with other analytical methods showcases the advantages HRMS possesses. Its ability to conduct a non-targeted analysis is invaluable in detecting previously unidentified contaminants. However, while HRMS excels in identification, challenges persist. The technique often requires rigorous sample preparation and skilled analysts for accurate interpretation. Additionally, HRMS, although superior in resolving power, can be limited in its quantification capacity in some scenarios when juxtaposed with techniques such as liquid chromatography-mass spectrometry (LC-MS) (Nováková, 2013). Finally, while HRMS is a cornerstone in the realm of EC detection, offering profound insights into unknown pollutants and their transformations, it is imperative to consider its strengths and limitations in the broader analytical context. Future endeavours should aim to amalgamate its capabilities with other techniques, enhancing the precision and breadth of contaminant analysis.

3.3.3. Biosensors

Biosensors are analytical devices that combine a biological recognition element, such as an enzyme, antibody, or receptor, with a transducer that converts the biological response into a measurable signal (Badihi-Mossberg et al., 2007). Biosensors offer the advantages of rapid analysis, high specificity, and real-time monitoring, and have been developed for the detection of various ECs, such as pharmaceuticals, EDCs, and microplastics (Villalba-Rodríguez et al., 2022). However, the practical application of biosensors in environmental analysis is still limited by factors such as sensitivity, stability, and reproducibility, and requires further research and development (Liang et al., 2021). In recent years, biosensors have been employed in studies to detect trace levels of antibiotics in water samples (Zhou et al., 2021) and monitor the presence of endocrine disruptors in river waters (Rozi et al., 2022). These studies showcase the potential of biosensors in providing valuable data for the monitoring and management of ECs in environmental matrices.

Summary: To provide a more straightforward guide, it is crucial to link specific groups of ECs to the most suitable analytical techniques for their quantitative measurement. For the analysis of polar and non-volatile compounds like pharmaceuticals and endocrine disrupting compounds, liquid chromatography coupled with mass spectrometry is often the method of choice due to its high sensitivity and selectivity. On the other hand, gas chromatography coupled with mass spectrometry is more suitable for less polar and volatile compounds such as perfluoroalkyl substances and certain pesticides. Ambient mass spectrometry techniques have shown potential for high-throughput screening across various EC groups but require further validation for quantitative applications. High-resolution mass spectrometry is invaluable for the non-targeted analysis of ECs, facilitating the identification of new or unknown contaminants. Biosensors offer a real-time monitoring advantage and are especially useful for the detection of

pharmaceuticals, endocrine disrupting compounds, and microplastics. However, they are currently limited by sensitivity, stability, and reproducibility issues. By carefully matching the EC group with the appropriate analytical technique, researchers can obtain more reliable and accurate data, which is vital for the monitoring and management of these contaminants in various environmental matrices. Table 2 summarizes these recommended approaches for detecting various ECs.

4. Toxicity assessment of emerging contaminants

Toxicity assessment of ECs is crucial to understand their potential impacts on human health and the environment (Anand et al., 2021). Various methods have been employed to evaluate the toxic effects of ECs, ranging from traditional whole-organism bioassays to advance in vitro and in silico techniques. These methods provide valuable information on the dose-response relationships, mode of action, and potential risks associated with exposure to ECs (Escher et al., 2018). The toxicity of any compound can be attributed to a combination of factors, central among which are its chemical structure and dose, and the organism's ability to absorb, metabolize and detoxify the compound.

Detoxification refers to the body's mechanism to transform potentially harmful substances into less damaging forms, and then efficiently excrete them. Every compound possesses a unique molecular structure that directly influences its physical, chemical, and toxicological properties (Barratt, 2000). By analysing this structure, especially the reactivity of its functional groups and its potential decomposition products, one can gauge the toxicity it might induce. For instance, many organic emerging contaminants (ECs) break down into carbon monoxide (CO) and carbon dioxide (CO_2) since they inherently contain carbon atoms. However, ECs with nitrogen groups—examples include common substances like paracetamol, caffeine, and various antibiotics like ciprofloxacin and erythromycin—can decompose to release nitrogen oxides and toxic gases. Some compounds, like ciprofloxacin and norfloxacin, can also release hydrogen fluoride gas due to their fluoride content ().

There are numerous compounds with specific structural features that render them toxic. For example, ibuprofen, which originates from propionic acid, acts as a radical scavenger (Rodriguez-Mozaz et al., 2007). Ciprofloxacin, recognized for its toxicity, is classified as a xenobiotic because of its specific molecular substituents, including a cyclopropyl and carboxylic acid. Similarly, erythromycin is a cyclic ketone, and high concentrations of such ketone compounds can be harmful to the body. There are also certain sweeteners, like saccharin and acesulfame, deemed as toxic xenobiotics due to their specific groups. Compounds used in medical diagnostics, such as the X-ray contrast media iohexol, present toxicity primarily from their benzene group components.

Furthermore, certain compounds can lead to long-term health implications. Phenolic compounds, containing hydroxyl and carboxylic groups, have been linked to conditions like neurodegeneration and accelerated aging (Gay et al., 2018). This underscores the reason why substances like bisphenol A, a plasticizer containing phenolic groups, are considered toxic. Despite the knowledge we possess, comprehensive research that establishes the relationship between chemical structures and their toxicity in ECs remains lacking. A more in-depth exploration into how the molecular structure of ECs determines their toxicity can offer valuable insights into their potential environmental impacts.

4.1. Aquatic toxicity testing

4.1.1. Acute toxicity tests

Acute toxicity tests aim to evaluate the short-term effects of ECs on aquatic organisms, typically within a 48 to 96-h exposure period. These tests usually focus on mortality or immobilization as the primary endpoint, providing information on the concentration that causes a specific effect in 50% of the test organisms (e.g., LC50 or EC50) (Besser et al., 2005). Acute toxicity tests have been conducted on various aquatic species, including fish (e.g., zebrafish), invertebrates (e.g., *Daphnia magna*), and algae (e.g., *Pseudokirchneriella subcapitata*). For example, (Terasaki et al., 2009) investigated the acute toxicity of several PPCPs on *Daphnia magna* and found that some compounds, such as triclosan, exhibited high acute toxicity, whereas others, like ibuprofen, showed low acute toxicity. The toxicological mechanism of triclosan is related to its capability to interfere with fatty acid synthesis, which is vital for cellular function. Similarly, studies by (Mhadhbi and Beiras, 2012) and (Belanger et al., 2013) assessed the acute toxicity of various pesticides and industrial chemicals on aquatic organisms, highlighting the potential risks posed by these ECs. The toxicity often arises from mechanisms like enzyme inhibition, disruption of cellular membranes, and interference with neurotransmission in these aquatic species.

4.1.2. Chronic toxicity tests

Chronic toxicity tests assess the long-term effects of ECs on aquatic organisms, usually spanning a significant portion of the organism's life cycle. These tests provide information on sublethal endpoints, such as growth, reproduction, and behaviour, which can be more sensitive to EC exposure than acute toxicity endpoints (Rodrigues et al., 2020). Chronic toxicity tests have been performed on various aquatic organisms, including fish, invertebrates, and algae. For instance, (Vera-Chang et al., 2018) examined the chronic effects of the antidepressant fluoxetine on the reproduction and development of zebrafish and revealed reduced reproductive success and increased embryonic malformations. The mechanism behind this could be fluoxetine's interference with serotonin uptake, disrupting neurodevelopment and reproductive processes. (Morales et al., 2018) investigated the chronic toxicity of the endocrine disruptor bisphenol A on the freshwater snail *Physa acuta*, finding negative impacts on growth, reproduction, and survival. The toxicological mechanism underlying these effects is linked to bisphenol A's potential to mimic estrogens which are natural hormones. When organisms are exposed to bisphenol A, it can interfere with their endocrine system, disrupting normal hormonal processes, which in turn affects growth and reproductive functions (Vandenberg et al., 2010). Additionally, (Choi et al., 2020) evaluated the chronic effects of microplastics on the marine copepod *Tigriopus japonicus* and reported alterations in reproductive and feeding behaviour. The mechanism behind these effects can be attributed to microplastics acting as physical stressors and potential carriers of contaminants, which can disrupt hormonal balances and interfere with the copepod's neural functions responsible for feeding and reproduction.

4.2. In vitro and in vivo models

In vitro models, such as cell culture assays and enzyme inhibition tests, have emerged as valuable tools for assessing the toxic effects of ECs on specific molecular targets or cellular processes. These models can provide mechanistic insights into the mode of action of ECs and help prioritize which chemicals to test further in whole-organism bioassays (Schmidt et al., 2017). Similarly, in vivo models, such as zebrafish embryos or transgenic organisms, can be employed to investigate the effects of ECs on specific tissues, organs, or physiological processes (Ruchika et al., 2022). For example, (Fent et al., 2006) used an in vitro reporter gene assay to investigate the estrogenic activity of various ECs. Mechanistically, many of these ECs bind to estrogen receptors, mimicking or blocking natural hormones and disrupting normal endocrine functions.

(Kim et al., 2021) conducted a study where they employed a genetically modified zebrafish as their experimental model. With a specific focus on the nervous system, their objective was to explore the harmful effects caused by the fungicide azoxystrobin. In a different study, (Xu et al., 2020b) utilized a laboratory test based on yeast cells to investigate the potential of various emerging contaminants (ECs) to cause genetic damage. Their study particularly focused on contaminants found in pharmaceuticals and personal care products. The use of these models and assays allows researchers to better understand the potential harm

Table 3
Previous research on the toxicity assessment of ECs.

Assessment method	Principle	Contaminants	Test organisms	Endpoint	Ref.
Acute toxicity tests	Short-term exposure to a single or a mixture of contaminants	Pharmaceuticals, pesticides, nanoparticles	*Daphnia magna*, zebrafish, algae	Mortality, immobilization	Li et al. (2021a)
Chronic toxicity tests	Long-term exposure to a single or a mixture of contaminants	PFAS, endocrine disruptors, microplastics	Zebrafish, rats, mice	Reproduction, growth, development	Di Poi et al. (2018)
Genotoxicity tests	Assessment of DNA damage	Heavy metals, PAHs, genotoxic compounds	Bacteria, yeast, human cell lines	Mutations, chromosomal aberrations, DNA damage	Nugnes et al. (2022)
Cytotoxicity tests	Assessment of cell damage	Pharmaceuticals, nanoparticles, toxic compounds	Mammalian cell lines, fish cell lines	Cell viability, cell morphology	Araújo et al. (2023)
In toxicology area	Assessment of biological effects	Pesticides, hormones, endocrine disruptors	Algae, *Daphnia magna*, zebrafish	Bioluminescence, enzyme activity, gene expression	Di Paolo et al. (2016)
In vivo models	Assessment of toxicity in controlled laboratory conditions	Pharmaceuticals, nanomaterials, chemical mixtures	Cell lines, tissues, sub-cellular components	Cell viability, gene expression, protein function	Di Paolo et al. (2016)
	Assessment of toxicity in living organisms	Pharmaceuticals, pesticides, endocrine disruptors	Rodents, zebrafish, *Daphnia magna*	Mortality, behaviour, tissue damage	Herrero et al. (2012)
Omics techniques	Assessment of global changes in gene expression, proteins, and metabolites	Chemical mixtures, pharmaceuticals, nanomaterials	Microorganisms, plants, animals	Transcriptomics, proteomics, metabolomics	Wang et al. (2021a)
QSAR models	Quantitative structure-activity relationship models	Various chemical compounds	In silico	Predicted toxicity, bioactivity	Chen et al. (2023)
Ecotoxicity thresholds	Comparison of contaminant concentrations to established thresholds	Metals, pesticides, emerging contaminants	Various aquatic and terrestrial species	Threshold exceedance, predicted no-effect concentrations	Gredelj et al. (2018)
Zebrafish embryo toxicity test	Assessment of developmental toxicity	Pharmaceuticals, pesticides, nanomaterials	Zebrafish embryos	Malformations, hatching success	Sposito et al. (2018)
Terrestrial toxicity tests	Assessment of toxicity to soil-dwelling organisms	Pesticides, heavy metals, emerging contaminants	Earthworms, collembola, plants	Mortality, reproduction, growth	Herrero et al. (2012)
Microbial toxicity tests	Assessment of toxicity to microorganisms	Antibiotics, heavy metals, nanomaterials	Bacteria, fungi, protozoa	Inhibition of growth	Zhang et al. (2022)

these compounds can inflict on living organisms. In an in vivo study, (Couleau et al., 2012) explored the effects of titanium dioxide nanoparticles on the freshwater bivalve *Dreissena polymorpha*, reporting alterations in oxidative stress and the immune response. The mechanism behind these effects is believed to be the interaction of TiO_2 NPs with cellular components, leading to an increased production of reactive oxygen species (ROS). Elevated ROS can then damage cellular structures and disrupt normal cellular functions, triggering oxidative stress and compromising the immune system of the organism.

4.3. High-throughput screening in mechanistic toxicity analysis

The identification and understanding of ECs in aquatic systems have been greatly enhanced by the advent of high-throughput screening (HTS) technologies (Attene-Ramos et al., 2014). These sophisticated tools, capable of screening large numbers of substances rapidly and efficiently, have provided invaluable insights into the mechanistic toxicity of ECs (Mezencev and Subramaniam, 2019). Central to this technological renaissance are the "omics" technologies. The omics approaches, encompassing genomics, transcriptomics, proteomics, and metabolomics, are at the vanguard of this transformation. Leveraging the power of rapid and simultaneous assays of complex samples, these technologies facilitate in-depth analyses of organisms' responses to contaminants at the molecular and cellular levels (Dai and Shen, 2022).

Genomics offers insights into the entirety of an organism's DNA sequence, allowing researchers to identify genetic predispositions to certain toxic responses. For example, an aquatic species might possess a specific gene variant that makes it more susceptible to a particular EC. Transcriptomics, on the other hand, sheds light on the RNA transcripts produced from the DNA, providing real-time data on which genes are actively being expressed or suppressed when exposed to contaminants. This gives a snapshot of the functional elements of the genome under specific environmental conditions. Collectively, these cutting-edge HTS methods provide comprehensive insights into the intricate molecular pathways and mechanisms affected by contaminants, thus painting a holistic picture of their toxicological impacts. This holistic approach, underpinned by omics technologies, is paving the way for a more nuanced understanding of emerging contaminants' effects in aquatic systems (Sharma et al., 2022).

4.4. Ecotoxicological risk assessment

Ecotoxicological risk assessment aims to evaluate the potential adverse effects of ECs on ecosystems by integrating exposure and effect data. This process involves several steps, including hazard identification, dose-response assessment, exposure assessment, and risk characterization (Oliveira et al., 2020). Various approaches, such as species sensitivity distributions (SSDs), risk quotients (RQs), and population- and community-level models, can be employed to estimate the ecological risks posed by ECs. These assessments help guide decision-makers in the development of environmental regulations, monitoring programs, and remediation strategies (Dale et al., 2008). For example, (Guo et al., 2016) conducted an ecotoxicological risk assessment of pharmaceuticals in European surface waters, using the SSD approach, and found that several compounds, including diclofenac, posed a significant risk to aquatic ecosystems.

Diclofenac, a commonly used non-steroidal anti-inflammatory drug (NSAID), poses risks due to its persistence in water and its potential to bioaccumulate. Mechanistically, diclofenac can inhibit cyclooxygenase (COX) enzymes in aquatic organisms, leading to physiological disruptions and potential mortality. Similarly, (Li et al., 2023a) used the RQ approach to assess the ecological risks of perfluoroalkyl acids in Chinese surface waters, revealing that some substances, such as perfluorooctane sulfonate, posed a high risk to aquatic organisms. This high risk arises from PFOS's persistent nature, its ability to bioaccumulate, and its potential to disrupt endocrine systems in aquatic organisms by interfering with hormone production, distribution, and function. In another study, (Gu et al., 2023) used a combination of SSD and RQ approaches to evaluate the ecological risks of trace metals in estuarine ecosystems, identifying copper, zinc, and lead as priority contaminants. (Previšić et al., 2021) employed a population-level modelling approach to assess the potential impacts of ECs on aquatic invertebrates, demonstrating the

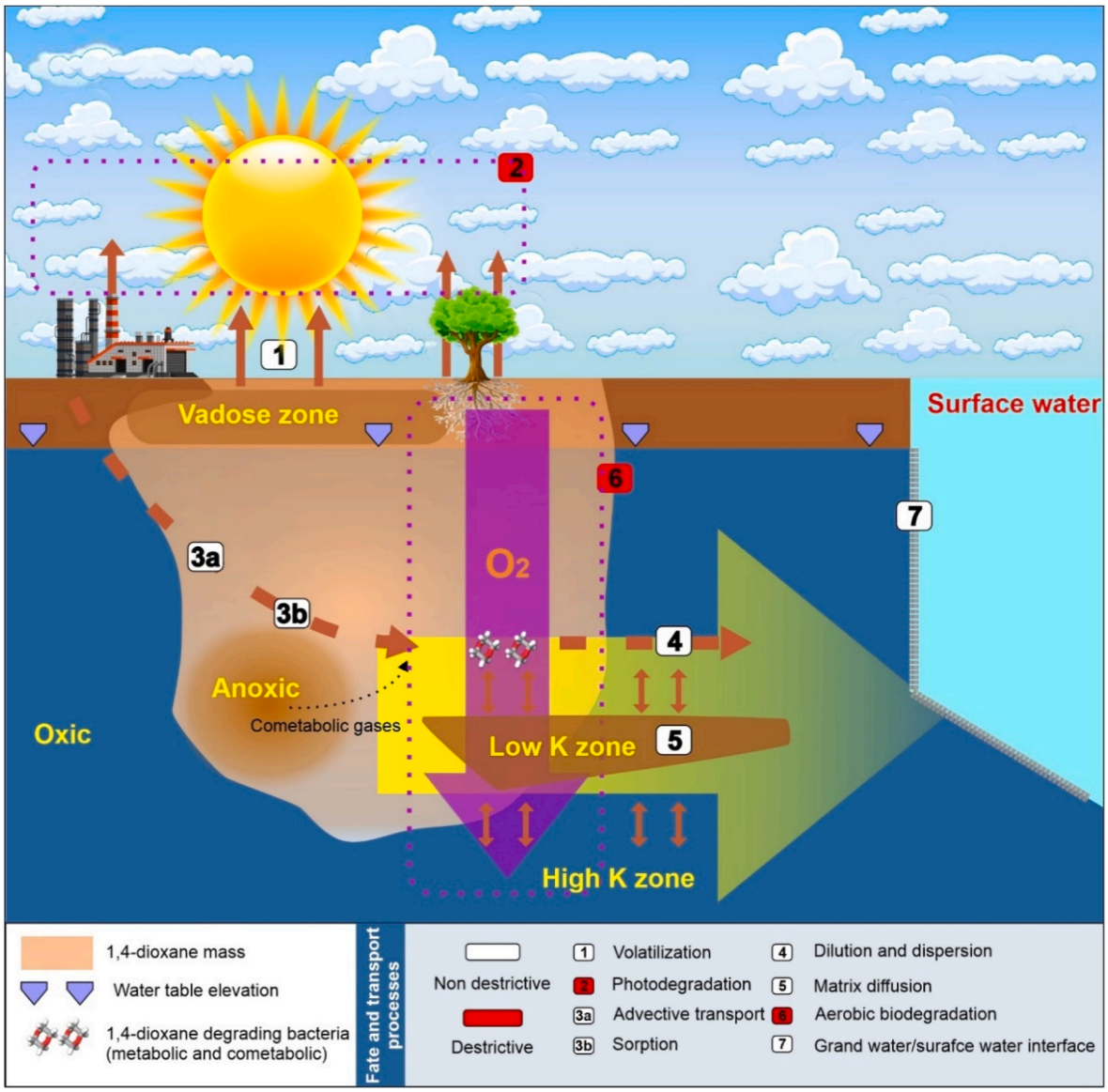

Fig. 2. The fate and transformation of one of the emerging contaminants (1,4-dioxane), redrawn from (ITRC, 2021). In the absence of water or soil moisture, this particular EC (1,4-dioxane) (1) volatilizes to the atmosphere where it is rapidly photodegraded. (2) In the presence of water advective flow drives EC into groundwater system or plants via uptake through plant root systems (3a) with little retardation from sorption into organic matter (3b). In the saturated zone, attenuation of EC occurs via dilution and dispersion (4), matrix diffusion (5), aerobic biodegradation mediated by microbes (6). Transport of undegraded EC to surface water may occur through groundwater-surface water interfaces (7).

importance of considering species-specific life history traits and exposure patterns in ecotoxicological risk assessments. A summary of previous research conducted on the toxicity assessment of ECs is presented in Table 3.

Summary: The toxicity assessment of emerging contaminants (ECs) is a critical endeavour that has far-reaching implications for both human health and environmental safety. This review has covered a multitude of methodologies including aquatic toxicity testing, in vitro and in vivo models, and ecotoxicological risk assessment, each with its own strengths and limitations. However, it is important to note that the collective findings from these studies reveal two major lessons. First, there is a pressing need for multi-disciplinary approaches that integrate various testing methods to create a more comprehensive understanding of EC toxicity. This is particularly relevant for informing effective management strategies and policy frameworks. Second, while advanced instrumentation such as LC-MS/MS and GC-MS/MS offer high sensitivity and selectivity in identifying ECs, they also come with limitations like cost and a high level of expertise, which calls for ongoing development in analytical technologies. These lessons make it evident that no single method is sufficient for toxicity assessment; rather, a combination of methods is essential. This nuanced understanding challenges us to develop more innovative and integrated strategies for assessing the risks associated with ECs, ultimately aiming to bolster efforts in environmental protection and public health. Recent advancements have ushered in high-throughput screening technologies that enable mechanistic toxicity analysis. Known as omics technologies, they allow researchers to explore sub-lethal toxic effects and cytotoxic exposures across all biological levels. Such deep insights offer a nuanced understanding of how emerging contaminants, even at trace levels, can disrupt biological systems.

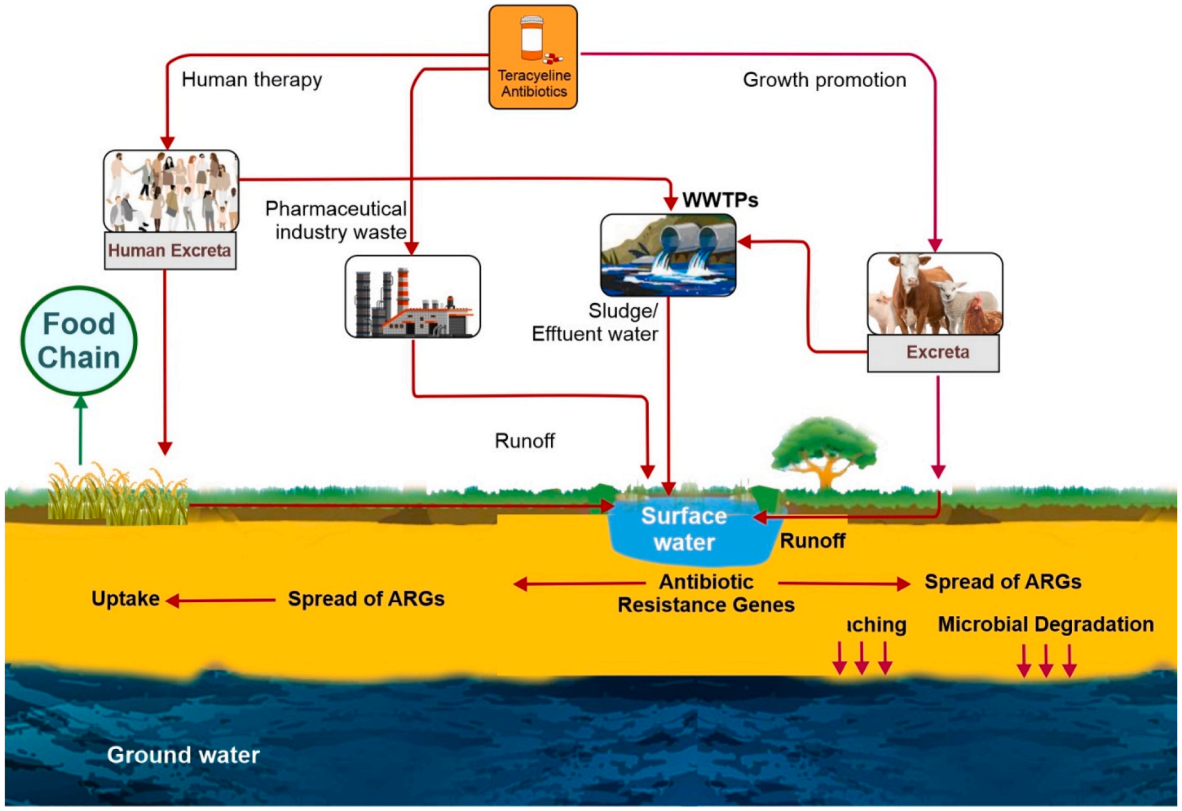

Fig. 3. Pathways of the emerging contaminant (a commonly used antibiotic, tetracycline) in both aquatic and terrestrial environments (Ahmad et al., 2021). The diagram depicts how tetracycline enters water bodies, its movement within these systems, and the potential impacts on aquatic life. It also shows how this contaminant affects soil, plants, and potentially animals.

5. Transport, transformation pathways and environmental fate of emerging contaminants

Understanding the transformation pathways and environmental fate of ECs in aquatic ecosystems is crucial for assessing their potential ecological and human health impacts, as well as for developing effective mitigation and remediation strategies. The behaviour and fate of ECs in the aqueous environment are governed by a complex interplay of physical, chemical, and biological processes, which can ultimately determine their persistence, bioavailability, and toxicity. Furthermore, these processes can be influenced by various environmental factors, such as temperature, pH, and the presence of other contaminants. The behaviour and movement of emerging contaminants after they are released into the environment are influenced by both the chemical and physical attributes of the compound and the characteristics of the surrounding environment. For instance, 1,4-dioxane, an emerging contaminant, is notable for its low capacity to bind to organic matter (indicated by its low organic carbon partitioning coefficient). This property makes it highly mobile in groundwater, raising concerns that the spread of 1,4-dioxane may exceed that of other typical co-contaminants like chlorinated volatile organic compounds (CVOCs) (Pica et al., 2021). This section provides a comprehensive overview of the major transformation pathways and environmental fate of ECs, including physical processes (adsorption, desorption, dilution, and dispersion), chemical processes (hydrolysis and photolysis), biological processes (biodegradation, bioaccumulation, and biomagnification), and the factors affecting their transformation and fate. Figs. 2 and 3 show the fate and transformation of the emerging contaminants.

5.1. Physical processes

5.1.1. Adsorption and desorption

Adsorption refers to the attachment of contaminants onto the surface of solid particles, such as suspended sediments and organic matter, whereas desorption is the release of contaminants from these surfaces back into the water column (Sastre et al., 2021). Adsorption and desorption processes can significantly affect the mobility, bioavailability, and persistence of ECs in aquatic systems. For instance, the adsorption of pharmaceuticals onto river sediments has been shown to reduce their bioavailability and potential toxicity to aquatic organisms (Radović et al., 2016). Similarly, studies conducted by (Yi et al., 2017) and (Martín et al., 2022) found that adsorption and desorption processes played a significant role in the environmental fate of various ECs, including antibiotics and perfluorinated compounds.

Adsorption is the process wherein contaminants adhere to the surface of solid particles, notably suspended sediments and organic matter. In contrast, desorption is the subsequent release of these contaminants from these solid surfaces back into the water column (Pignatello and Xing, 1996). The interplay between adsorption and desorption is pivotal in determining the mobility, bioavailability, and persistence of ECs within aquatic environments. The repercussions of adsorption on pharmaceutical contaminants in riverine systems have been particularly well-documented. These drugs, when adsorbed onto river sediments, tend to exhibit reduced bioavailability, thereby minimizing their toxicological impacts on aquatic biota (Patel et al., 2019). Complementing this, the investigative studies by (Christou et al., 2017) and (Zhao et al., 2012) illustrated the central role of adsorption and desorption processes in orchestrating the environmental fate of a spectrum of ECs, inclusive of antibiotics and perfluorinated compounds. Several other studies have further expanded our comprehension of these physical processes. (Zhang et al., 2014) explored how varying pH levels in water bodies

influenced the adsorption capacity of heavy metals onto sediments. Their findings emphasized that pH fluctuations can drastically modulate the rate and extent of metal adsorption, thereby reshaping their environmental distribution.

Moreover, the correlation between organic matter content in sediments and the adsorptive potential of endocrine-disrupting compounds (EDCs) was explored by (Styszko, 2016). They deduced that sediments with a higher organic content exhibited a pronounced affinity for EDCs, acting as reservoirs and subsequently impacting the contaminant's overall persistence in aquatic habitats. In another noteworthy study, (Lesan and Bhandari, 2003) underscored the temporal aspect of desorption, elucidating that over prolonged periods certain persistent organic pollutants undergo a transition from a readily desorbable state to a more tightly bound configuration. This aspect is crucial as it dictates the long-term bioavailability and potential risk of these compounds.

5.1.2. Dilution and dispersion

Dilution and dispersion processes are critical factors in determining the concentration and distribution of contaminants in aquatic environments (Chapman, 2007). Dilution occurs when a contaminant is released into a large water body, leading to a decrease in its concentration. Dispersion, on the other hand, refers to the spreading of contaminants in the water column due to natural processes, such as turbulent mixing and currents (Farré et al., 2008). These processes can either mitigate or exacerbate the ecological impacts of ECs, depending on the specific contaminant and environmental conditions. For example, (Tang et al., 2018) observed that dilution and dispersion played a major role in the distribution of polycyclic aromatic hydrocarbons (PAHs) in coastal waters, while (Mohapatra et al., 2021) reported similar findings for pharmaceutical compounds in river systems.

Dilution and dispersion processes are paramount in determining the concentration and distribution of contaminants in aquatic environments. Dilution manifests when a contaminant enters a vast water body, thereby reducing its concentration. Conversely, dispersion pertains to the diffusion of contaminants within the water column due to inherent natural phenomena, including turbulent mixing and water currents (Campos et al., 2022). Depending on the specific contaminant and environmental backdrop, these processes can either dampen or magnify the ecological ramifications of ECs.

A series of research endeavours have shed light on the significance of dilution and dispersion in contaminant distribution. (Sánchez-Brunete et al., 2007) pinpointed the pivotal role these processes have on the distribution of polycyclic aromatic hydrocarbons (PAHs) in marine realms. Likewise, (Rico et al., 2019) mirrored such findings for pharmaceutical residues in fluvial systems. In a landmark study, (Li et al., 2019) examined the dilution effects on heavy metal contaminants in estuarine zones. Their work highlighted that while dilution effectively diminished metal concentrations, dispersion led to a more widespread distribution, potentially affecting benthic organisms. Similarly, (He et al., 2021) investigated the dispersion patterns of microplastics in coastal regions. They underscored how these particles, despite being minuscule, underwent extensive dispersion due to tidal actions and wind-induced surface currents.

Further, a meta-analysis conducted by (Blanar et al., 2009) analysed numerous research studies on dilution and dispersion in freshwater lakes. They concluded that while dilution often led to a temporal decrease in pollutant concentrations, the long-term effects heavily depended on the extent of dispersion and the nature of the contaminant. In essence, while dilution and dispersion are natural processes acting on contaminants, their effects are multifaceted and vary depending on the contaminant type and the affected aquatic system. More comprehensive studies are warranted to understand their long-term impacts on aquatic ecology.

5.1.3. Hydrodynamic conditions

In static water bodies like ponds, lakes, and reservoirs, hydrodynamic models focus on three main processes to determine the transport of ECs: 1) advection; 2) diffusion; and 3) vertical mixing (Ji, 2017). For flowing waters like rivers, advection primarily drives the transport of ECs (Yearsley, 2009). These mechanisms have been studied to predict the behaviour of various contaminants or water quality elements in distinct environmental sections. Advection directs the long-range movement of ECs (Zoppou and Knight, 1997). Diffusion, on the other hand, causes ECs to move from areas of high to low concentration (Latour et al., 1994). We can further break down diffusion into turbulent and molecular types, both of which influence the lateral and longitudinal dispersion of ECs. The principal diffusion type, especially in the direction of water flow, is turbulent diffusion, which plays a pivotal role in EC transport.

Beyond the horizontal movement (advection and diffusion) of ECs, the vertical mixing attributes in aquatic settings are just as vital. Water pressure and gravitational changes primarily induce vertical transport in water. Turbulence also considerably affects vertical mixing, facilitating the vertical exchange of ECs in the water column due to the energy it provides (Ravens et al., 2000). Another significant aspect in surface waters is the secondary flow, a natural or induced occurrence near water boundaries, for example, near plants or rocks (Wang and Cheng, 2005). The existence and nature of these secondary flows can modify how ECs mix and move (Tong et al., 2019), yet their role has often been overlooked in studies exploring the behaviour of ECs in water environments.

Multiple factors, including topography, weather patterns, the roughness of the water bed, and aquatic plants, further modify the hydrodynamics of pollutant transport (Tong et al., 2021). Depending on the hydrodynamic context, ECs might exhibit diverse transport patterns. For instance, in a slow-moving water body, low pace and minimal mixing might result in elevated EC concentrations (Silva et al., 2011). In heavily urbanized areas, high EC levels in tributaries might be a result of nearby human activities, consistent hydrodynamic conditions marked by sluggish water movement, small water quantities, and inadequate mixing features. In shallow waters, wind plays a crucial role in the hydrodynamics of ECs (Jalil et al., 2018). Winds create currents when they blow across the water surface, transmitting both momentum and energy horizontally and vertically to the water body. This can affect the advection, diffusion, vertical mixing, and even the resuspension of sediments containing ECs (Li et al., 2017). Therefore, the impact of wind should not be underestimated when modelling EC transport dynamics.

5.2. Chemical processes

5.2.1. Hydrolysis

Hydrolysis is a chemical process by which a contaminant reacts with water, resulting in the cleavage of a chemical bond and the formation of new products (Al-Sabagh et al., 2016). Hydrolysis can significantly affect the stability, reactivity, and toxicity of contaminants in aquatic systems. For example, the hydrolysis of organophosphate pesticides in natural waters has been shown to produce less toxic and more biodegradable products (Pavez et al., 2020). Recent studies by (Jia et al., 2023) and (Xiong et al., 2021) demonstrated that hydrolysis was a key factor influencing the degradation of neonicotinoid insecticides and some pharmaceuticals in aquatic environments.

An example is seen in the hydrolysis of organophosphate pesticides in natural waters, which has been observed to yield products that are less toxic and more prone to biodegradation (Pehkonen and Zhang, 2002). Recently, the role of hydrolysis in the degradation of contaminants has gained significant attention in the literature. Notably, (Pietrzak et al., 2020) showcased hydrolysis as a predominant factor in degrading neonicotinoid insecticides in aquatic systems. Similarly, (Ahmed et al., 2020) identified that hydrolysis played a pivotal role in breaking down certain PFAS, offering new insights into the management of these persistent pollutants. (Kusema et al., 2010) delved into the hydrolysis rates of various industrial chemicals, suggesting a strong correlation between molecular structure and hydrolysis susceptibility.

Table 4
Advantages and disadvantages of different transformation pathways of ECs.

Transformation pathway	Description	Major ECs	Advantages	Disadvantages	Ref.
Adsorption	ECs attach to solid surfaces in the environment.	Pharmaceuticals, heavy metals	– Can remove contaminants from water. – Can be selective for certain contaminants. – May mitigate immediate risks.	– Some ECs can be released back into the environment. – May not be effective for all contaminants. – Can be a slow process.	(Sahoo et al., 2020; Sophia A and Lima, 2018)
Desorption	ECs detach from solid surfaces and return to the environment.	Pharmaceuticals, heavy Metals	– Can be part of remediation strategies. – Releases bound nutrients essential for the ecosystem.	– Can lead to re-pollution of the environment. – Can contribute to the spread of contamination.	(Ghosh et al., 2013; Vicente-Martínez et al., 2020)
Dilution	ECs are spread out in the environment, reducing concentration.	Pesticides, Industrial Chemicals	– Lowers immediate concentrations and potential toxicity. – Reduces risk of immediate exposure.	– Does not eliminate contaminants and may spread them further. – May prolong the environmental persistence of ECs.	(Varsha et al., 2022; Rathi et al., 2021)
Dispersion	ECs spread and mix due to environmental movements (e.g., water currents)	Oil spills, nutrients	– Can reduce local concentrations. – May disperse pollutants into non-critical areas.	– Contaminants may accumulate in certain areas. – Can contribute to the spread of contamination.	Zaulkiflee et al. (2022)
Hydrolysis	ECs react with water to break down into other substances.	Organophosphates, ester-based compounds	– This can lead to the degradation of harmful substances. – Often an abiotic and rapid process.	– Some products of hydrolysis may also be harmful. – Not effective for all ECs.	Rempel et al. (2021)
Photolysis	ECs break down when they absorb light, particularly UV.	Phenols, polycyclic aromatic hydrocarbons	– Can lead to the degradation of harmful substances. – Often an abiotic and rapid process.	– Some products of photolysis may be harmful. – Limited to ECs that absorb light.	(Rivas et al., 2012; Borges et al., 2015)
Biodegradation	Microorganisms break down ECs into other substances.	Petroleum hydrocarbons, biodegradable plastics	– Often results in less harmful substances. – Uses natural microbial processes. – Can be used in bioremediation.	– May be slow. – Not all ECs can be biodegraded. – Sometimes results in toxic metabolites.	(Matamoros et al., 2016; Maryjoseph and Ketheesan, 2020)
Bioaccumulation	ECs accumulate in organisms at concentrations higher than environment.	Heavy metals, persistent organic pollutants	– Helps understand toxicity in ecosystems. – Can be an indicator of environmental contamination.	– Can cause harm to organisms. – Can affect ecosystem balance. – May lead to health issues in animals and humans.	(Maryjoseph and Ketheesan, 2020; González-González et al., 2022)
Biomagnification	Concentration of ECs increases in food chains.	Heavy metals, persistent organic pollutants	– Useful to study ecological impacts. – Indicator of long-term environmental contamination.	– Can lead to high concentrations in apex predators. – Can affect entire ecosystems and human health.	(Yang et al., 2019; Clarke and Cummins, 2015)

This research underscores the potential for designing chemicals that are more environmentally friendly based on their hydrolytic profiles. Furthermore, (Wang et al., 2021b) reviewed the hydrolysis of common microplastics in marine environments, highlighting the transformation of these materials under different pH conditions. Their findings suggest that hydrolysis contributes to the mitigation of microplastic pollution in specific marine conditions.

5.2.2. *Photolysis*

Photolysis is the process by which contaminants are degraded through the absorption of sunlight, leading to the formation of new products (Weidauer et al., 2016). Photolysis can play a crucial role in the transformation and fate of many ECs, particularly those that are susceptible to direct or indirect photochemical reactions. For instance, the photolysis of some pharmaceuticals, such as sulfamethoxazole and carbamazepine, has been reported to contribute significantly to their degradation in surface waters (Fatta-Kassinos et al., 2011). Additionally, studies conducted by (Martínez-Zapata et al., 2013) and (Abdelraheem et al., 2015) found that photolysis was an important transformation pathway for various emerging contaminants, including endocrine-disrupting chemicals and organic UV filters.

Photolysis is a pivotal chemical process that induces the degradation of contaminants through sunlight absorption, subsequently leading to the formation of various byproducts (Koumaki et al., 2015). This photochemical route holds significance in dictating the environmental fate of a myriad of ECs, especially those which readily undergo direct or indirect photochemical reactions. The significance of photolysis in contaminant degradation is well-documented. Notably, several studies have demonstrated the considerable role of photolysis in the degradation of pharmaceuticals in aquatic environments. Compounds such as sulfamethoxazole and carbamazepine, once introduced to surface waters, have been observed to degrade primarily due to photolysis (Kim and Tanaka, 2009).

The research terrain is rich with explorations focusing on the photolytic behaviour of ECs. (Rafaie et al., 2023) and (Arman et al., 2021) both emphasized the importance of photolysis in managing

Table 5
Previous research findings on the transformation pathways and environmental fate of emerging contaminants.

Author, year	Contaminant(s)	Transformation pathway	Key findings
(Nguyen et al., 2019)	PPCPs	Biodegradation	Biodegradation by microorganisms is an important pathway for the removal of PPCPs from the environment, with varying efficiency depending on the compound and environmental conditions.
(Zhuo et al., 2012)	Perfluorinated compounds	Photolysis	Photolysis played a significant role in the degradation of perfluorinated compounds, with products showing decreased environmental persistence and toxicity compared to the parent compounds.
(Issac and Kandasubramanian, 2021)	Microplastics	Fragmentation	Microplastics in the environment undergo fragmentation processes, resulting in the formation of smaller particles that may be more easily ingested by aquatic organisms and have greater potential for bioaccumulation.
(Nikiforov, 2021)	PFASs	Hydrolysis	Hydrolysis was found to be an important degradation pathway for some PFASs, with varying rates depending on the specific compound and environmental conditions.
(Huang et al., 2019)	Nanoparticles	Adsorption/ desorption	Adsorption/desorption processes influence the transport and fate of nanoparticles in the environment, with factors such as particle size, surface charge, and the presence of natural organic matter affecting these processes.
(Gao et al., 2020)	Endocrine disruptors	Biodegradation	Certain endocrine-disrupting chemicals can be efficiently biodegraded by microorganisms under aerobic conditions, while others may persist or form transformation products with unknown properties.
(Lin et al., 2010)	Pharmaceuticals	Sorption to sediment	Pharmaceuticals can undergo sorption to sediment in aquatic environments, which may result in their removal from the water column but can also lead to their accumulation in bottom sediments.
(Porras et al., 2016)	Antibiotics	Photodegradation	Photodegradation is an important removal pathway for some antibiotics in aquatic environments, with the formation of transformation products that may have varying persistence and toxicity.
(Reungoat et al., 2010)	Micropollutants	Ozonation	Ozonation, a chemical oxidation process, is effective in removing a wide range of micropollutants from water, although some contaminants may exhibit resistance and require additional treatment.
(Barbieri et al., 2021)	Pesticides	Bioaccumulation	Pesticides can bioaccumulate in aquatic organisms, potentially causing adverse effects on their growth, reproduction, and survival, and leading to biomagnification of these contaminants through the food chain.

emerging contaminants, particularly spotlighting endocrine-disrupting chemicals and organic UV filters. In a similar vein, (Mei et al., 2021) investigated the degradation rates of several persistent organic pollutants (POPs) under varying sunlight conditions, concluding that photolysis played an essential role in their attenuation. Another avenue of research underscored the importance of understanding the byproducts formed post-photolysis. In their groundbreaking study, (Tang et al., 2012) provided insight into the transformation products of several industrial chemicals when subjected to direct sunlight, emphasizing the need to study these byproducts for potential secondary environmental or health risks. Furthermore, certain ECs exhibit varied degradation pathways based on the spectrum of light available. A notable study by (Yap et al., 2019) found that specific ECs degraded faster under UV light compared to visible light, emphasizing the influence of light quality on photolytic reactions.

5.3. Biological processes

5.3.1. Biodegradation

Biodegradation is the breakdown of contaminants by microorganisms, such as bacteria, fungi, and algae, through enzymatic processes (Bacha et al., 2023). Biodegradation can be an important factor in the removal and transformation of many ECs in aquatic systems, particularly those that are biologically labile. For example, the biodegradation of nonylphenol ethoxylates, a group of surfactants widely used in detergents, has been demonstrated to significantly reduce their persistence and ecological impacts in aquatic environments (He et al., 2020). Recent studies by (Park et al., 2017) and (Zhu and Chen, 2014) also revealed that biodegradation played a critical role in the removal of various pharmaceuticals and personal care products in wastewater treatment systems.

An instance can be observed in the biodegradation of nonylphenol ethoxylates, a group of surfactants ubiquitous in detergents. Their degradation considerably diminishes both their persistence and detrimental ecological consequences in aquatic habitats (Mann and Boddy, 2000). Recent studies by (Iatrou et al., 2017) and (Christofilopoulos et al., 2019) have also emphasized the cardinal role of biodegradation in eliminating an array of pharmaceuticals and personal care products from wastewater treatment systems. Moreover, the research conducted by (Cai et al., 2023) showcased how certain bacteria strains are uniquely efficient in breaking down POPs, which were once believed to be resistant to biodegradation. Their study underscored the importance of microbial diversity in enhancing biodegradation rates.

Triclosan, an antimicrobial and antifungal agent found in many personal care products, was previously recognized for its persistent nature in aquatic environments. However, a study by (Nandikes et al., 2022) discovered specific fungal species capable of degrading triclosan under anaerobic conditions, a significant revelation in understanding its environmental fate. (Zhou et al., 2022) investigated the role of biofilms in facilitating the biodegradation of ECs. Their study found that biofilms, comprising complex communities of microorganisms, could enhance the degradation of contaminants by offering a diverse array of enzymatic actions and pathways. A comparative study by (Ahmed et al., 2017) looked into the differences between aerobic and anaerobic biodegradation of various ECs. They found that while some contaminants were readily degraded under aerobic conditions, others were better suited for anaerobic environments, emphasizing the importance of understanding specific environmental contexts for effective biodegradation.

5.3.2. Bioaccumulation and biomagnification

Bioaccumulation refers to the uptake and retention of contaminants by organisms from their environment, whereas biomagnification is the process by which the concentration of contaminants increases in organisms at higher trophic levels in a food web (Szynkowska et al., 2018). Both bioaccumulation and biomagnification can have profound implications for the ecological and human health risks posed by ECs. For instance, the bioaccumulation and biomagnification of per- and polyfluoroalkyl substances (PFAS) in aquatic food webs have been linked to adverse health effects in both wildlife and humans, including reproductive, developmental, and immunological disorders (Liang et al., 2022). Additional studies by (Yang et al., 2019) and (Lin et al., 2018) demonstrated that the bioaccumulation and biomagnification of various ECs, such as organophosphate flame retardants and brominated flame retardants, could pose significant risks to wildlife populations and human health. The various advantages and disadvantages of different transformation pathways of ECs and a summary of previous research findings on the transformation pathways and environmental fate of ECs are presented in Table 4 and Table 5, respectively.

Table 6
Various factors affecting the transformation pathways and environmental fate of ECs.

Factor	Findings	Ref.
Environmental conditions	- Temperature affects biodegradation rates of pharmaceuticals	Sipma et al. (2010)
	- pH-dependent hydrolysis of estrogens and beta-blockers	Gabet et al. (2007)
	- Redox conditions influence antibiotic fate in aquatic environments	Jong et al. (2020)
	- Photodegradation of pharmaceuticals enhanced by organic matter	
Presence of other contaminants	- Accelerated photodegradation of triclosan in the presence of humic substances	Awfa et al. (2019) Chen et al. (2017)
	- Heavy metals inhibit bisphenol biodegradation in aquatic systems	Im and Löffler (2016)
Microbial community structure	- PAH biodegradation influenced by diversity and abundance of PAH-degrading bacteria	Haleyur et al. (2019)
	- Selective degradation of PCB components by specific bacterial strains	Egorova et al. (2021)
	- Degradation of endocrine-disrupting compounds affected by bacterial community composition	Zhang et al. (2020)

5.4. Factors affecting transformation and fate

Several factors can influence the transformation pathways and environmental fate of ECs in aquatic systems. Understanding the factors affecting the transformation and fate of emerging contaminants is crucial for predicting their environmental behaviour and developing effective strategies for their monitoring, risk assessment, and remediation. Table 6 shows how different factors affect the transformation and fate of ECs.

5.4.1. Environmental conditions

Factors such as temperature, pH, and redox conditions can significantly affect the rate and extent of chemical and biological transformations of ECs in aquatic environments (Varsha et al., 2022). For example, the hydrolysis of certain pharmaceuticals, such as estrogens and beta-blockers, has been shown to be highly pH-dependent, with faster degradation rates observed under acidic or alkaline conditions (Ikehata et al., 2006). Li et al. (Li et al., 2021b) demonstrated that temperature played a crucial role in the degradation of pharmaceuticals, with higher temperatures leading to increased biodegradation rates. Similarly, a study by Stadler et al. (Stadler et al., 2015) found that redox conditions significantly affected the fate of antibiotics in aquatic environments, with anaerobic conditions leading to the formation of more persistent transformation products.

5.4.2. Presence of other contaminants

The presence of other contaminants in aquatic systems can alter the transformation and fate of ECs through synergistic or antagonistic interactions (Lonappan et al., 2016). For instance, the photodegradation of some pharmaceuticals can be enhanced in the presence of naturally occurring organic matter, which can act as photosensitisers and promote indirect photochemical reactions (Sardana et al., 2022). In a study by Wang et al. (Wang et al., 2017b), the presence of humic substances was found to accelerate the photodegradation of triclosan in natural waters. Conversely, a study by Li et al. (Li et al., 2007) revealed that the presence of heavy metals could inhibit the biodegradation of bisphenol A in aquatic systems due to the toxic effects of metals on microbial communities.

5.4.3. Microbial community structure

The biodegradation of ECs in aquatic systems is highly dependent on the structure and activity of microbial communities, which can vary widely in response to environmental conditions, contaminant concentrations, and other biotic and abiotic factors (Gomes et al., 2020). For example, the biodegradation of polycyclic aromatic hydrocarbons (PAHs) in marine sediments has been found to be strongly influenced by

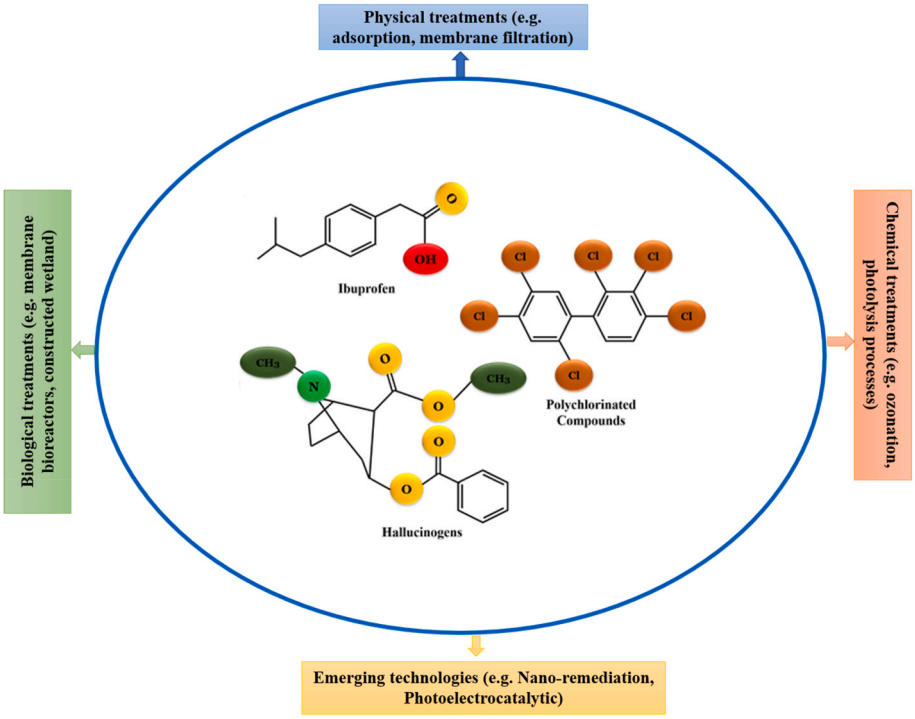

Fig. 4. Comprehensive overview of the key strategies for the removal of ECs, highlighting emerging technologies, chemical treatments, physical treatments, biological technologies.

Table 7
Advantages and disadvantages of different remediation strategies for ECs.

Remediation Technology	Description	Advantages	Disadvantages	Ref.
Physical: adsorption	Contaminants are attached to the surface of a solid material.	– Effective for a wide range of contaminants. – Simple to implement. – Can be used for both water and air treatment.	– Disposal of used adsorbents can be challenging. – Possible desorption of contaminants. – High cost and non-recyclable of certain adsorptive materials – May require periodic replacement of the adsorbent material.	(Shahid et al., 2021; Song et al., 2022)
Chemical: advanced oxidation processes	Utilizes strong oxidizing agents to degrade contaminants.	– Effective for degradation of recalcitrant contaminants. – Can be highly selective. – Capable of complete mineralization of contaminants.	– May produce toxic by-products. – Often requires high energy input. – Can be expensive to operate and maintain.	(Shahid et al., 2021; Morin-Crini et al., 2022)
Biological: bioremediation	Utilizes microorganisms to break down contaminants.	– Environmentally friendly. – Can be cost-effective. – Capable of degrading complex organic molecules.	– Can be slow. – Not effective for all contaminants. – Requires careful monitoring and control of environmental conditions. – Difficulty in obtaining highly effective microbial consortium – Low reproducibility	Dhangar and Kumar (2020)
Biological: constructed wetlands	Utilizes natural processes in wetlands to remove contaminants.	– Sustainable and low-cost. – Multipurpose (biodiversity, flood protection). – Can treat a wide range of pollutants.	– Requires large land area. – Maintenance and monitoring are required. – Efficiency can be affected by weather conditions.	(Chowdhury et al., 2022; Sánchez et al., 2022)
Combined: integrated AOPs and bioremediation	Combines chemical and biological processes.	– AOPs can break down contaminants into more biodegradable substances. – Potentially more effective than individual methods. – Can handle a wide range of contaminants.	– Complex to implement and monitor. – Possible negative interactions between processes. – May be expensive and require skilled operators.	Giwa et al. (2021)
Emerging: nano-remediation	Utilizes nanoparticles to degrade or remove contaminants.	– High surface area and reactivity. – Can target specific contaminants. – Capable of degrading contaminants at a molecular level.	– Potential environmental risks of nanoparticles. – High costs. – Uncertain long-term effects.	(Ganie et al., 2021; Pico et al., 2019)
Emerging: photoelectrocatalytic treatment	Combines photocatalysis and electrochemistry to degrade contaminants.	– Can be highly effective for certain contaminants. – Utilizes renewable solar energy. – Can achieve high degradation rates.	– Limited to contaminants that can be degraded photoelectrochemically. – Potential formation of harmful by-products. – Requires access to sunlight.	(Fang et al., 2020; Malinović et al., 2021)

the diversity and abundance of PAH-degrading bacteria (Botsou and Hatzianestis, 2012). Adebusoye et al. (Adebusoye et al., 2008) demonstrated that specific bacterial strains could selectively degrade different components of a complex mixture of polychlorinated biphenyls (PCBs). Furthermore, a study by Roccuzzo et al. (Roccuzzo et al., 2021) showed that the degradation of endocrine-disrupting compounds in wastewater treatment plants was significantly affected by the composition of the bacterial community, with certain bacterial species exhibiting a higher capacity to degrade these contaminants.

Summary: The analysis of existing literature on the environmental fate and transformation pathways of emerging contaminants (ECs) has revealed several key insights and limitations. Notably, it was found that individual approaches focusing solely on physical, chemical, or biological processes are insufficient to fully understand or mitigate the behaviour of ECs in aquatic systems. A robust argument is thus made for adopting an integrated, multi-dimensional research approach, which has been shown to offer greater efficacy in both understanding and tackling ECs. For instance, singular reliance on physical adsorption strategies has proven to be inadequate in controlling the mobility of ECs like 1,4-dioxane. Likewise, the efficacy of processes such as dilution and photolysis were found to be highly contingent on specific environmental conditions. This paper further argues that there is a critical need for lifecycle analyses, particularly for ECs with the potential for bioaccumulation and biomagnification, to include risk assessments that are both comprehensive and relevant to long-term ecosystem and human health.

6. Remediation strategies for emerging contaminants

Various remediation technologies have been developed and investigated to address the challenges posed by emerging contaminants in aquatic environments. These technologies aim to remove or degrade contaminants to minimize their potential impacts on human health and ecosystems. Recent studies have focused on the application of physical, chemical, and biological treatments, as well as combined and emerging remediation technologies. Fig. 4 illustrates various methods to remove ECs. In the realm of environmental management, particularly in the field of ECs treatment, a robust evaluation system is essential. The evaluation metrics included are: (1) Effectiveness, which gauges the extent to which a strategy successfully removes or degrades the target ECs; (2) Sustainability, examining both the environmental and economic feasibility of the strategy; (3) Applicability, assessing its versatility across various ECs; (4) Operational Complexity, considering the ease of both implementation and maintenance; (5) Cost Implication, focusing on the financial commitments required to both implement and maintain the strategy; and (6) Scalability, which evaluates the practicality of transitioning the strategy from the pilot stage to a full-scale application. In this paper the effectiveness of different remediation strategies has been presented. In addition, the various advantages and disadvantages of different remediation strategies for ECs and a summary of previous research findings on the different remediation strategies for ECs are presented in Tables 7 and 8, respectively.

Table 8
Summary of previous research findings on remediation strategies for ECs.

Remediation strategy	Contaminants studied	Key findings	Ref.
Advanced oxidation processes (AOPs)	PPCPs, endocrine disrupting compounds (EDCs)	AOPs effectively degrade a wide range of ECs; Fenton and photo-Fenton processes showed higher removal efficiency.	Camargo-Perea et al. (2020)
Adsorption and ion Exchange	Heavy metals, pharmaceuticals	Activated carbon and biochar were found to be highly effective adsorbents for the removal of various ECs.	Sophia A and Lima (2018)
Constructed wetlands	Pharmaceuticals, personal care products (PPCPs)	Constructed wetlands demonstrated good removal efficiency for PPCPs and EDCs, with over 80% reduction in some cases.	Yi et al. (2017)
Membrane bioreactors	Antibiotics, EDCs, PPCPs	Bioreactors effectively removed the majority of ECs, with up to 99% removal efficiency for some compounds.	Ji et al. (2020)
Membrane processes	PPCPs, EDCs, pesticides	Membrane processes, such as nanofiltration and reverse osmosis, showed high removal efficiency (80–100%) for various ECs.	Ramrakhiani et al. (2022)
Combined and emerging remediation technologies	PPCPs, EDCs, pesticides	Combined treatments, such as AOPs with biological processes, led to enhanced removal efficiency and reduced operational costs.	Paździor et al. (2019)
Electrochemical oxidation	Pharmaceuticals, EDCs, perfluorinated compounds	Electrochemical oxidation, using boron-doped diamond electrodes, demonstrated high removal efficiency for a variety of ECs, especially pharmaceuticals.	Wilk et al. (2022)
Phytoremediation	Pharmaceuticals, pesticides, EDCs	Some plant species were found to be effective at removing or degrading ECs, with certain species capable of reducing contaminant concentrations by up to 70%.	Shi et al. (2023)
Ultraviolet (UV) and UV/H_2O_2 processes	Pharmaceuticals, EDCs, pesticides	UV and UV/H2O2 processes showed significant removal of ECs, with higher removal efficiency observed when combined with H2O2.	Rott et al. (2018)
Ozonation	Pharmaceuticals, pesticides, EDCs	Ozonation proved to be an effective method for the degradation of various ECs, with removal efficiency ranging from 50% to over 90% depending on the contaminant.	Tanveer et al. (2022)
Solar photocatalysis	Pharmaceuticals, EDCs, pesticides	Solar photocatalysis, using TiO2 as a catalyst, showed promising results in degrading various ECs, with up to 99% removal efficiency for some compounds.	Sousa et al. (2012)
Electro-Fenton process	Pharmaceuticals, EDCs, pesticides	The Electro-Fenton process was effective in the degradation of various ECs, with high removal efficiency (80–100%) reported for some contaminants.	Li et al. (2023b)

6.1. Physical and chemical treatments

6.1.1. Advanced oxidation processes (AOPs)

AOPs are a group of chemical treatment methods that involve the generation of highly reactive hydroxyl radicals, which can effectively oxidize and degrade a wide range of organic contaminants (Andreozzi et al., 1999). AOPs include UV/H_2O_2, Fenton, and photocatalytic processes (Ghaly et al., 2001). These processes have been demonstrated to be effective in the removal of various ECs, such as pharmaceuticals, personal care products, and endocrine-disrupting chemicals. For example, Felis et al. (Felis et al., 2011) highlighted effective degradation of Bisphenol A using the UV/H_2O_2 process. This mechanism leverages the combination of ultraviolet light and hydrogen peroxide to produce highly reactive hydroxyl radicals. These radicals are capable of breaking down a myriad of organic contaminants, including bisphenol A (BPA). Such findings are consistent with earlier research conducted by Rosenfeldt and Linden (Rosenfeldt and Linden, 2004) where the UV/H_2O_2 process was established as an efficient tool for degrading endocrine-disrupting chemicals. Mackuľak et al. (Mackuľak et al., 2015) turned their attention to the removal of pharmaceuticals from wastewater using Fenton oxidation. This process relies on the reaction between hydrogen peroxide and iron salts, producing hydroxyl radicals like the UV/H_2O_2 process. These radicals are then involved in the oxidation of organic contaminants present in the wastewater. Historically, the Fenton process has demonstrated efficacy in addressing various pollutants, particularly in complex wastewater matrices. Furthermore, Trojanowicz et al. (Trojanowicz et al., 2018) shed light on its capability, particularly in degrading challenging pollutants like perfluorinated compounds. These are synthetic compounds known for their resistance to degradation due to the strength of their carbon-fluorine bonds. The photocatalytic process accelerates the breakdown of these bonds by generating electron-hole pairs when a semiconductor catalyst,

Table 9
Efficiency of different advanced oxidation processes in removing emerging contaminants.

Emerging contaminants	Influent concentration (μg/L)	Removal efficiency (%)	Refs.
Acetaminophen	1,209,304	98	de Luna et al. (2012)
Acetaminophen	1,209,304	97	de Luna et al. (2012)
Acetaminophen	5	90	Klamerth et al. (2012)
Aldrin	5000	90	Belgiorno et al. (2007)
Amoxicillin	104,000	100	Gaya and Abdullah (2008)
Ampicillin	105,000	100	Gaya and Abdullah (2008)
Antipyrine	5	89	Klamerth et al. (2012)
Atenolol	1	85	Prieto-Rodríguez et al. (2013)
Atrazine	5	60	Klamerth et al. (2012)
Bisphenol A	1	85	Prieto-Rodríguez et al. (2013)
Bisphenol A	26,938	80	(Andersen et al., 2003), (Torres et al., 2007)
Caffeine	5	90	Klamerth et al. (2012)
Caffeine	1	55	Prieto-Rodríguez et al. (2013)
Carbamazepine	5	90	Klamerth et al. (2012)
Cefalexin	200,000	100	Estrada et al. (2012)
Cloxacillin	10,300	100	Gaya and Abdullah (2008)
Diazinon	10,000	99	Belgiorno et al. (2007)
Diclofenac	5	90	Klamerth et al. (2012)
Diclofenac	1	85	Prieto-Rodríguez et al. (2013)
Diclofenac	50,000–100,000	90	(Tijani et al., 2013), (Hartmann et al., 2008)
Diuron	1	85	Prieto-Rodríguez et al. (2013)
Flumequine	5	90	Klamerth et al. (2012)
Gemfibrozil	1	85	Prieto-Rodríguez et al. (2013)
Hydrochlorothiazide	1	85	Prieto-Rodríguez et al. (2013)
Hydroxybiphenyl	5	85	Klamerth et al. (2012)
Ibuprofen	5	90	Klamerth et al. (2012)
Ibuprofen	1	85	Prieto-Rodríguez et al. (2013)
Isobroturum	5	90	Klamerth et al. (2012)
Ketoprofen	49.84	100	(Illés et al., 2012), (Domínguez et al., 2010)
Ketoralac	5	90	Klamerth et al. (2012)
Malathion	10,000	99	Belgiorno et al. (2007)
Naproxen	6.5	98	Jallouli et al. (2016)
Naproxen	1	85	Prieto-Rodríguez et al. (2013)
Ofloxacin	5	90	Klamerth et al. (2012)
Ofoxacin	1	85	Prieto-Rodríguez et al. (2013)
Paraxanthine	1	85	Prieto-Rodríguez et al. (2013)
Progesterone	5	90	Klamerth et al. (2012)
Sulfamethoxazole	52,700	100	Ganzenko et al. (2014)
Sulfamethoxazole	5	95	Klamerth et al. (2012)
Sulfamethoxazole	52,700	100	(Ganzenko et al., 2014), (Dirany et al., 2010)
Tetracycline	25,000	86	Ganzenko et al. (2014)
Triclosan	5	90	Klamerth et al. (2012)
Trimethoprim	1	85	Prieto-Rodríguez et al. (2013)

like titanium dioxide, is exposed to light. Earlier studies, such as those by Doll and Frimmel (Doll and Frimmel, 2005), have further corroborated the efficiency of photocatalysis in degrading persistent organic pollutants. The efficiency of different advanced oxidation processes in removing emerging contaminants is shown in Table 9.

6.1.2. Adsorption and ion exchange

Adsorption and ion exchange are widely used physical treatment methods for removing ECs from water. Adsorption involves the attachment of contaminants onto the surface of a solid adsorbent, such as activated carbon, biochar, or zeolites (Rathi and Kumar, 2021). Ion exchange, on the other hand, involves the exchange of ions between a solid resin and the contaminated solution (Vilensky et al., 2002). Both methods have been successfully applied in the removal of various ECs, including pharmaceuticals, perfluoroalkyl substances, and heavy metals. Snyder et al. (Snyder et al., 2007) are among numerous researchers who have recognized the remarkable effectiveness of activated carbon in eliminating pharmaceutical residues from wastewater. Their study is echoed by other extensive research that delineates the profound adsorption capacity of activated carbon in capturing diverse organic contaminants (Huang et al., 2021). Xie et al. (Xie et al., 2022) reported the resin's efficacy in removing perfluorooctanoic acid (PFOA) from solutions. The mechanism behind this is the resin's ability to swap benign ions from its structure with the target contaminant ions in the solution. Notably, the removal of PFOA, a persistent environmental pollutant, via ion exchange resins is a crucial finding, and such observations are supported by other studies that highlight the flexibility and effectiveness of ion-exchange methods in treating diverse pollutants (Amini et al., 2015). Wen et al. (Wen et al., 2016) highlighted the potential of zeolites in sequestering heavy metals from contaminated water. The process fundamentally involves the exchange of cations from the water with those present in the zeolite structure, leading to the entrapment of heavy metals. The consistent results presented by Wen et al. (Angaru et al., 2021) are also resonated in various studies emphasizing the merits of zeolites, with some even delving into modifications to enhance their efficiency. Table 10 shows the comparative efficiency of various absorbents in removing different emerging contaminants.

6.2. Biological treatments

6.2.1. Constructed wetlands

Constructed wetlands (CWs) are engineered systems designed to mimic the natural processes of wetlands, providing an effective and sustainable approach to treating contaminated water (Wu et al., 2018). CWs rely on the synergistic action of plants, microorganisms, and natural environmental conditions to remove contaminants through

Table 10
Comparative efficiency of various absorbents in removing different emerging contaminants.

Emerging contaminants	Adsorbent	Dosage	Removal efficiency	Refs.
1,8-dichlorooctane	AC	0.20 g/L	2699 mg/g	Patiño et al. (2015)
	CN	0.20 g/L	2740 mg/g	Patiño et al. (2015)
	Carbon nanofibers	0.20 g/L	2667 mg/g	Patiño et al. (2015)
	High surface area graphite's	0.20 g/L	2760 mg/g	Patiño et al. (2015)
Amoxicillin	AC	100 mg/L	228.29 mg/g	Benjedim et al. (2020)
Bisphenol-A	AC	8 cm Bed depth	74.70%	Katsigiannis et al. (2015)
Caffeine	Carbon xerogels	2.40 g/L	182.50 mg/g	Álvarez et al. (2015)
	AC	2.50 g/L	99.90%	Gil et al. (2018)
	CS	1 mg/L	71.30%	Rossner et al. (2009)
	Peach stone	0.12 g/50 mL	96%	Torrellas et al. (2015)
Carbamazepine	Wood	10 mg/L	63%	Mailler et al. (2016)
	Peach stone	0.12 g/50 mL	72%	Torrellas et al. (2015)
	Powered activated carbon	100 mg/L	220 mg/g	Delgado et al. (2019)
Ciprofloxacin	Wood	10 mg/L	77%	Mailler et al. (2016)
	Coal	10 mg/L	63%	Mailler et al. (2016)
	Peat	10 mg/L	62%	Mailler et al. (2016)
Clofibric acid	AC	2.50 g/L	63.33%	Gil et al. (2018)
	C_{18}-Mica− 4	2 g/L	81%	Martín et al. (2018)
Diclofenac	Carbon xerogels	2.40 g/L	80 mg/g	Álvarez et al. (2015)
	AC	2.50 g/L	99.90%	Gil et al. (2018)
	Mixed poly aluminium chloride and activated carbon	100 mg/L	85%	Jiang et al. (2015)
	Peach stone	0.12 g/50 mL	48%	Torrellas et al. (2015)
	C_{18}-Mica− 4	2 g/L	78%	Martín et al. (2018)
Dodecyl sulphate	C_{18}-Mica− 4	2 g/L	100%	Martín et al. (2018)
Erythromycin	Wood	10 mg/L	75%	Mailler et al. (2016)
	Coal	10 mg/L	66%	Mailler et al. (2016)
Estradiol	CS	1 mg/L	78.90%	Rossner et al. (2009)
Estrone	CS	1 mg/L	89.20%	Rossner et al. (2009)
Fluoxetine	CS	1 mg/L	98.40%	Rossner et al. (2009)
Fluoxetine	Carbonaceous resin	1 mg/L	75.80%	Rossner et al. (2009)
Gallic acid	AC	2.50 g/L	86.67%	Gil et al. (2018)
Gemfibrozil	C_{18}-Mica− 4	2 g/L	86%	Martín et al. (2018)
Hydrocodone	Coconut shell	1 mg/L	76.60%	Rossner et al. (2009)
Ibuprofen	AC	2.50 g/L	99.90%	Gil et al. (2018)
	Mixed poly aluminium chloride and activated carbon	100 mg/L	75%	Jiang et al. (2015)
	AC	8 cm Bed depth	57.40%	Katsigiannis et al. (2015)
	C_{18}-Mica− 4	2 g/L	78%	Martín et al. (2018)
Ketoprofen	AC	8 cm Bed depth	61.40%	Katsigiannis et al. (2015)
Methylparaben	Powdered activated carbon	0–20 mg/L	167 mg/g	Moreno-Marenco et al. (2022)
Methyl-phenoxy-ethanol	AC	0.20 g/L	239 mg/g	Patiño et al. (2015)
	CN	0.20 g/L	97 mg/g	Patiño et al. (2015)
	CN	0.20 g/L	63 mg/g	Patiño et al. (2015)
	High surface area graphite	0.20 g/L	99 mg/g	Patiño et al. (2015)
Nalidixic acid	AC	0.20 g/L	277 mg/g	Patiño et al. (2015)
	CN	0.20 g/L	186 mg/g	Patiño et al. (2015)
	CN	0.20 g/L	67 mg/g	Patiño et al. (2015)
	High surface area graphite	0.20 g/L	179 mg/g	Patiño et al. (2015)
Naproxen	AC	8 cm Bed depth	65.60%	Katsigiannis et al. (2015)
Nonylphenol	C_{18}-Mica− 4	2 g/L	99%	Martín et al. (2018)
Norfloxacin	Wood	10 mg/L	79%	Mailler et al. (2016)
	Coal	10 mg/L	69%	Mailler et al. (2016)
	Peat	10 mg/L	62%	Mailler et al. (2016)
Octadecyl sulphate	C_{18}-Mica− 4	2 g/L	100%	Martín et al. (2018)
Ofloxacin	Wood	10 mg/L	79%	Mailler et al. (2016)
	Coal	10 mg/L	71%	Mailler et al. (2016)
	Peat	10 mg/L	68%	Mailler et al. (2016)
Oxybenzone	CS	1 mg/L	99.80%	Rossner et al. (2009)
	Carbonaceous resin	1 mg/L	99.80%	Rossner et al. (2009)
Paracetamol	AC	100 mg/L	453.39 mg/g	Benjedim et al. (2020)
Pentoxifylline	CS	1 mg/L	69.90%	Rossner et al. (2009)
Perfluorobutanoic acid	C_{18}-Mica− 4	2 g/L	78%	Martín et al. (2018)
	C_{18}-Mica− 4	2 g/L	77%	Martín et al. (2018)
Perfluorooctane sulfonic acid	C_{18}-Mica− 4	2 g/L	100%	Martín et al. (2018)
Perfluorooctanoic acid	C_{18}-Mica− 4	2 g/L	91%	Martín et al. (2018)
Propranolol	Wood	10 mg/L	81%	Mailler et al. (2016)
	Coal	10 mg/L	76%	Mailler et al. (2016)
	Peat	10 mg/L	70%	Mailler et al. (2016)

(continued on next page)

Table 10 (*continued*)

Emerging contaminants	Adsorbent	Dosage	Removal efficiency	Refs.
Propylparaben	Powdered activated carbon	0–20 mg/L	151 mg/g	Moreno-Marenco et al. (2022)
	C_{18}-Mica− 4	2 g/L	78%	Martín et al. (2018)
Salicylic acid	AC	2.50 g/L	83.33%	Gil et al. (2018)
	Mixed poly aluminium chloride and activated carbon	100 mg/L	70%	Jiang et al. (2015)
	C_{18}-Mica− 4	2 g/L	90%	Martín et al. (2018)
Sildenafil citrate	Powered activated carbon	100 mg/L	395 mg/g	Delgado et al. (2019)
Sulfamethoxazole	CS	10 mg/L	60%	Mailler et al. (2016)
Triclosan	AC	8 cm Bed depth	86.70%	Katsigiannis et al. (2015)
	CS	1 mg/L	99.50%	Rossner et al. (2009)
	Carbonaceous resin	1 mg/L	99.80%	Rossner et al. (2009)
Trimethoprim	CS	1 mg/L	80.60%	Rossner et al. (2009)

AC: Activated carbon; CN: Carbon nanotube; CS: Coconut shell.

processes such as adsorption, precipitation, and biodegradation (Chen et al., 2019). Recent studies have demonstrated the potential of CWs for the removal of various ECs, including pharmaceuticals, personal care products, and endocrine-disrupting chemicals. For instance, Zhang et al. (Zhang et al., 2011) investigated the capability of CWs in treating water contaminated with pharmaceuticals, specifically noting the successful removal of ibuprofen, naproxen, and carbamazepine. The mechanism behind this effectiveness lies in the CWs' unique design which simulates natural wetlands. Here, the synergistic interplay between plants, microorganisms, and the physical properties of the wetland substrate results in processes like adsorption, biodegradation, and phytoremediation, which collaboratively contribute to contaminant removal. Ilyas et al. (Ilyas and van Hullebusch, 2020) documented the considerable potential of CWs in eliminating these compounds from water. This can be attributed to the biological processes within the wetland, where microbial communities can break down complex organic compounds present in personal care products. This finding was consistent with other research which suggested that CWs could be an economical and eco-friendly alternative for treating wastewater laced with these chemicals (Stefanakis, 2019). Furthermore, the menace posed by endocrine-disrupting chemicals (EDCs) in the water has been a growing concern due to their potential impact on aquatic life and human health. These chemicals can interfere with the endocrine system, leading to a host of adverse effects. Chen et al. (Chen et al., 2022) took a leap in this context, showcasing the viability of CWs in not just removing but also mitigating the effects of EDCs in contaminated water. The sorption of these chemicals onto plant roots and their subsequent degradation by associated microbial communities in the rhizosphere are potential mechanisms that assist in their removal (Haritash and Kaushik, 2009). Table 11 shows the comparative efficiency of various constructed wetlands in removing different emerging contaminants.

6.2.2. Membrane bioreactors

Bioreactors are engineered systems that utilize microorganisms to degrade contaminants through biological processes. They include aerobic and anaerobic treatment systems, such as activated sludge, membrane bioreactors, and biofilm reactors (Saidulu et al., 2021).

Table 11
Comparative efficiency of various constructed wetlands in removing different emerging contaminants.

Emerging contaminants	Constructed wetland type	Removal Efficiency (%)		Refs.
		Min	Max	
Acetaminophen	Horizontal subsurface flow	>90	–	Ranieri et al. (2011)
Atenolol	Horizontal subsurface flow	58	99	Chen et al. (2016)
	Surface flow	27	53	Breitholtz et al. (2012)
	Horizontal subsurface flow	48	–	Ramos et al. (2016)
Carbamazepine	Free water subsurface flow	35	71	Reyes-Contreras et al. (2012)
	Subsurface flow	16	–	Matamoros and Bayona (2008)
	Surface flow	32	37	Matamoros et al. (2008)
	Subsurface flow	27	28	Zhang et al. (2011)
Chlorpyrifos Mecoprop	Subsurface flow	>96		Agudelo et al. (2010)
	Surface flow	79	91	Matamoros et al. (2008)
	subsurface flow	22	–	Matamoros and Bayona (2008)
Diclofenac	Free water subsurface flow	17	26	(Reyes-Contreras et al., 2012), (Hijosa-Valsero et al., 2010)
	Free water subsurface flow	36	52	Hijosa-Valsero et al. (2010)
	Surface flow	20	50	Chen et al. (2016)
	Horizontal subsurface flow	24	93	Chen et al. (2016)
	Vertical subsurface flow	53	73	Chen et al. (2016)
	Surface flow	73	96	Matamoros et al. (2008)
	Surface flow	85		Llorens et al. (2009)
Furosemide	Horizontal subsurface flow	80	96	Chen et al. (2016)
	Horizontal subsurface flow	35	71	(Ramos et al., 2016) (Matamoros et al., 2009)
Galaxolide	Free water subsurface flow	67	82	Reyes-Contreras et al. (2012)
	Surface flow	88	90	Matamoros et al. (2008)
	Surface flow	87		Song et al. (2009)
Ibuprofen	Horizontal subsurface flow	74	99	Chen et al. (2016)
	Surface flow	45	95	Reyes-Contreras et al. (2012)
	Free water subsurface flow	27	74	Hijosa-Valsero et al. (2011)
	Free water subsurface flow	6	95	Hijosa-Valsero et al. (2010)
	Subsurface flow	71	–	Matamoros and Bayona (2008)
	Vertical subsurface flow	55	99	Chen et al. (2016)
	Surface flow	95	96	Llorens et al. (2009)
	Subsurface flow	71	80	Zhang et al. (2011)
	Surface flow	50	100	Ávila et al. (2013)
Ketoprofen	Free water subsurface flow	47	81	Reyes-Contreras et al. (2012)
	Free water subsurface flow	11	50	(Reyes-Contreras et al., 2012; Hijosa-Valsero et al., 2010)
	Surface flow	47	91	Chen et al. (2016)
	Horizontal subsurface flow	10	90	Chen et al. (2016)
Metoprolol	Horizontal subsurface flow	60	93	Chen et al. (2016)
	Surface flow	3	30	Breitholtz et al. (2012)
	Horizontal subsurface flow	11	–	Ramos et al. (2016)
Naproxen	Free water subsurface flow	58	81	Reyes-Contreras et al. (2012)

(*continued on next page*)

Table 11 (*continued*)

Emerging contaminants	Constructed wetland type	Removal Efficiency (%)		Refs.
		Min	Max	
	Free water surface flow	27	66	(Reyes-Contreras et al., 2012), (Hijosa-Valsero et al., 2010)
	Free water surface flow	27	83	NA
	Surface flow	75	76	Chen et al. (2016)
	Horizontal subsurface flow	76	97	Chen et al. (2016)
	Vertical subsurface flow	69	96	Chen et al. (2016)
	Subsurface flow	85		Matamoros and Bayona (2008)
	Surface flow	52	92	Ávila et al. (2013)
	Surface flow	83	91	Zhang et al. (2011)
Oxybenzone	Horizontal subsurface flow	>97	–	Matamoros et al. (2009)
Paracetamol	Horizontal subsurface flow	95	100	Chen et al. (2016)
	Horizontal subsurface flow	>90	–	Ranieri et al. (2011)
	Hybrid	>95–99	99	Ávila et al. (2013)
Tramadol	Surface flow	12	26	Breitholtz et al. (2012)
	Horizontal subsurface flow	54	85	Chen et al. (2016)
Triclosan	Horizontal subsurface flow	62	91	Chen et al. (2016)
	Reclamation pond	74	93	Matamoros and Salvadó (2012)

Bioreactors have been shown to be effective in the removal of various ECs, including pharmaceuticals, personal care products, and nanomaterials. Aboudalle et al. (Aboudalle et al., 2021) investigated the role of activated sludge systems in the removal of pharmaceutical compounds from wastewater. The activated sludge process involves the biological breakdown of organic contaminants using a mixed culture of microorganisms. These microorganisms consume the pharmaceutical compounds as a food source, thereby breaking them down into simpler, non-toxic compounds. Aboudalle et al.'s findings reinforced the potential of this method to effectively reduce pharmaceutical pollutants from wastewater streams. Goswami et al. (Goswami et al., 2018) conducted research on the efficacy of membrane bioreactors (MBRs) in treating wastewater laden with personal care products. MBRs combine the conventional biological treatment processes with membrane filtration. The microorganisms in the bioreactor break down the personal care products, while the membrane helps retain the microorganisms and remove any undegraded contaminants from the treated water. Goswami et al.'s study accentuated the robustness of MBRs in mitigating pollutants from personal care products in wastewater. Franca et al. (Franca et al., 2020) explored the use of biofilm reactors to address this issue. In a biofilm reactor, microorganisms attach to surfaces and form a dense, sticky film. These biofilms, given their enhanced surface area and metabolic pathways, can capture and degrade a wide array of contaminants, including engineered nanomaterials. Franca et al.'s research underscored the potential of biofilm reactors as a promising solution to degrade and remove engineered nanomaterials from wastewater.

6.2.3. Biosorption

Biosorption is a treatment process distinct from biodegradation. While both are biologically based, they operate on different principles. In the biosorption process, microorganisms are anchored or "immobilized" onto a material known as an adsorbent. This allows for two primary processes: sorption, where contaminants are collected on the surface of the adsorbent, and bio-oxidation, where chemical reactions involving the contaminants occur. Following this, the pollutants can accumulate and attach themselves to certain parts of the biomass cellular structure without the need for active uptake.

Nguyen et al. (Nguyen et al., 2014) conducted a study comparing biosorption and biodegradation's roles in removing emerging contaminants from wastewater. They used both live and sodium azide-inactivated white rot fungus (*T. versicolour*) cultures for their research. Another study by Banihashemi and Droste (Banihashemi and Droste, 2014) focused on the biosorption of specific ECs like EE2, bisphenol A, and benzophenone. Their findings indicated that the concentration of these contaminants in solution dropped quickly for certain pollutants (with the order being triclosan > EE2 > bisphenol A). Furthermore, the concentrations of these compounds in both liquid and solid phases continued to decline over the duration of their study, suggesting the potential biodegradation of these chemicals in both forms. Table 12 shows the efficiency in removing emerging contaminants using biosorption methods and membrane bioreactor systems.

6.3. Hybrid and emerging remediation technologies

Combined and emerging remediation technologies involve the integration of multiple treatment methods to enhance contaminant removal efficiency. These approaches may include the combination of physical, chemical, and biological treatments or the application of novel materials and processes, such as electrochemical oxidation, sonolysis, and photocatalysis (Shahid et al., 2021). Recent studies have demonstrated the potential of combined and emerging remediation technologies for the removal of various ECs, including pharmaceuticals, personal care products, and endocrine-disrupting chemicals. Michael et al. (Michael et al., 2019) conducted research on the amalgamation of two treatment methods: activated carbon adsorption and Fenton oxidation. Their findings revealed that this combined approach was particularly successful in eliminating pharmaceutical contaminants from water. Activated carbon adsorption functions by capturing contaminants on its porous surface, while Fenton oxidation employs a chemical reaction to degrade pollutants further. Dao et al. (Dao et al., 2020) focused on the utilization of electrochemical oxidation as a method for treating wastewater. Their findings confirmed its effectiveness, especially in removing personal care product residues. Electrochemical oxidation works by driving a current through water, resulting in the production of reactive species that can breakdown contaminants. Chiha et al. (Chiha et al., 2011) investigated the capabilities of sonolysis, a process that uses ultrasonic waves to induce chemical reactions. Their research underscored its potential in degrading endocrine-disrupting chemicals, which are compounds that can interfere with the hormonal systems of organisms. Alfred et al. (Alfred et al., 2020) presented findings on the potential of photocatalysis, a method that leverages light to spur chemical reactions, in treating water contaminants. Their research demonstrated the efficacy of this approach, specifically in removing various ECs from water. Photocatalysis, primarily driven by sunlight or artificial light sources, offers a sustainable avenue for water purification. Table 13 shows the comparison of the efficiency of various hybrid systems in removing a range of emerging contaminants.

6.4. Evaluation and optimization of remediation strategies

To ensure the effectiveness and sustainability of remediation strategies, it is essential to evaluate and optimize their performance based on factors such as contaminant removal efficiency, energy consumption, environmental impacts, and economic feasibility (Wang et al., 2021c). Life cycle assessment (LCA) is a widely used approach for evaluating the environmental impacts of remediation technologies throughout their life cycle, from raw material extraction to disposal (Visentin et al., 2019). Additionally, a cost-benefit analysis (CBA) can help compare the economic feasibility of different remediation strategies by considering both direct and indirect costs and benefits (Liu et al., 2016). Recent studies have focused on the optimization of remediation strategies through the application of various experimental designs and modelling

Table 12
Analysis of the efficiency in removing emerging contaminants using biosorption methods and membrane bioreactor systems.

Category	ECs	Biosorption				MBR		Refs.
		Live (fungus)		Inactivated (fungus)		Effluent (μg/L)	Removal (%)	
		Influent (μg/L)	Removal (%)	Influent (μg/L)	Removal (%)			
Antiplatelet agents	Codeine					0.338	88.0	Dolar et al. (2012)
	Paracetamol					1.675	58.10	Monsalvo et al. (2014)
Anxiety relievers	Clopidogrel					0.133	69.0	Dolar et al. (2012)
	Hydrocodone					0.111	93.0	Dolar et al. (2012)
Antagonists	Diazepam					0.019	67.0	Dolar et al. (2012)
Analgesics	Carbamazepine	50	01	50	07	0.099; 1.55; 0.9	51.00; 4.80; 100	(Monsalvo et al., 2014; de la Torre et al., 2015)
	Citalopram					0.295	91.80	Llorens-Blanch et al. (2015)
	Diclofenac	50	97	50	43	5; 1.1	15.30; 49	(Nguyen et al., 2014; de la Torre et al., 2015; Nguyen et al., 2013)
	Ibuprofen	50	100	50	27	5	96.0	(Nguyen et al., 2014; Nguyen et al., 2013)
	Lorazepan					0.0848	100	Llorens-Blanch et al. (2015)
	Metronidazole					1.14	96.0	Llorens-Blanch et al. (2015)
	Naprox	50	100	50	17	5; 1.85	45.0; 70.30	(Nguyen et al., 2014; Monsalvo et al., 2014)
	Primidone	50	12	50	27	5; 1.7	88.4; 1.80	(Nguyen et al., 2014; Monsalvo et al., 2014)
	Trazodone					0.0341	100	Llorens-Blanch et al. (2015)
Anti-depressants	Amitriptyline	50	05	50	09			Nguyen et al. (2014)
Anticonvulsants	Ketoprofen	50	22	50	11	1.625	14.90	(Nguyen et al., 2014; Monsalvo et al., 2014)
Antibiotics	Azithromycin					0.142	74.0	Dolar et al. (2012)
	Clarithromycin					2.722	87.0	Dolar et al. (2012)
	Erythromycin					0.08	79.0	Dolar et al. (2012)
	Ofloxacin					2.90	90.0	Dolar et al. (2012)
	Sulfamethaxazole					0.268; 1.625	69.0; 95.20	(Dolar et al., 2012; Monsalvo et al., 2014)
	Trimethoprim					1.55	36.80	Monsalvo et al. (2014)
Anti-inflammatory	Acetaminophen					5	87.10	Dolar et al. (2012)
Beta-blockers	Atendol					2.44; 1.775	97.1; 15.8	(Dolar et al., 2012; Monsalvo et al., 2014)
	Metoprolol					0.076	71.2	Dolar et al. (2012)
	Nadolol					0.048	67.0	Dolar et al. (2012)
	Propranolol					0.309; 0.4	70.9; 100	(Dolar et al., 2012; de la Torre et al., 2015)
	Sotalol					0.222	70.9	Dolar et al. (2012)
	Salbutamol					0.04	79.0	Dolar et al. (2012)
Diuretics	Hydrochlorothiazide					0.4075	99.10	Llorens-Blanch et al. (2015)
	Furosemide					0.356	78.70	Llorens-Blanch et al. (2015)
EDCs	Androstenedione					1.775	99	Monsalvo et al. (2014)
	Androsterone					1.775	98	Monsalvo et al. (2014)
	E1	50	72	50	31.50	5	96.50	(Monsalvo et al., 2014; Nguyen et al., 2013)
	E2	50	60.50	50	29.50	5	99.50	(Nguyen et al., 2014; Nguyen et al., 2013)
	EE2	50	62	1.50; 2.10	1.38; 2.76	5	92.90	(Nguyen et al., 2014; Banihashemi and Droste, 2014; Nguyen et al., 2013)
	E3	50	4.50	50	13	5	97.60	(Nguyen et al., 2014; Nguyen et al., 2013)
	17β-Estradiol-17-acetate	50	79	50	84	5	99.10	(Nguyen et al., 2014; Nguyen et al., 2013)
	Bisphenol A	50	65	1.50; 2.10	1.24; 2.59	5	94.20	(Nguyen et al., 2014; Banihashemi and Droste, 2014)
	4-*tert*-butylphenol	50	33	50	10.50	5	92.80	(Nguyen et al., 2014; Nguyen et al., 2013)
	Nonylphenol					1.58	98.70	Monsalvo et al. (2014)
	Octylphenol					1.725	70.20	Monsalvo et al. (2014)
	4-*tert*-octylphenol		90		82.50	5	96.50	(Nguyen et al., 2014; Nguyen et al., 2013)
	4-*n*-nonylphenol					5	97.20	Nguyen et al. (2013)
	Testosterone					1.85	99	Monsalvo et al. (2014)
Gastroesophageal	Lorazepam					0.082	48.0	Dolar et al. (2012)
	Ranitidine					0.932; 0.0319	89.0; 100	(Dolar et al., 2012; Llorens-Blanch et al., 2015)
Lipid regulators	Clofibric acid	50	06	50	18			Nguyen et al. (2014)
	Gemifibrozil	50	100	50	57.5	0.885	13.10	(Nguyen et al., 2014; Monsalvo et al., 2014)
Pain-relievers	Famotidine					0.132	84.0	Dolar et al. (2012)

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Table 12 (*continued*)

Category	ECs	Biosorption				MBR		Refs.
		Live (fungus)		Inactivated (fungus)		Effluent (μg/L)	Removal (%)	
		Influent (μg/L)	Removal (%)	Influent (μg/L)	Removal (%)			
PCPs	Benzophenone	50	40	1.50; 2.10	1.11.5; 2.83			(Nguyen et al., 2014; Banihashemi and Droste, 2014)
	Oxybenzene	50	54.50		59.50			Nguyen et al. (2014)
	Propylparaben					1.1	100	Trinh et al. (2016)
	Salicylic acid	50	68	50	0	5; 13.9	97.8; 100	(Nguyen et al., 2014; de la Torre et al., 2015; Nguyen et al., 2013)
Pesticides	Atrazine	50	18	50	9	1.65	6.80	(Nguyen et al., 2014; Monsalvo et al., 2014)
	Dicamba					310	69.0	Ghoshdastidar and Tong (2013)
	Fenoprop	50	01	50	0	5	19.0	(Nguyen et al., 2014; Nguyen et al., 2013)
	2,4-D					3580	99.0	Ghoshdastidar and Tong (2013)
	Mecoprop					1990	75.40	Ghoshdastidar and Tong (2013)
	Pentachlorophenol	50	63	50	96	5	60.80	(Nguyen et al., 2014; Nguyen et al., 2013)
	Triclosan	50	78.5	50	97	5; 0.594	99.10; 90.20	(Nguyen et al., 2014; Nguyen et al., 2013)
Stimulant	Caffeine					1.625	76.90	Monsalvo et al. (2014)

approaches, such as response surface methodology (RSM), artificial neural networks (ANNs), and genetic algorithms (GAs) (Mohammadi et al., 2019). These techniques can help identify the key factors affecting contaminant removal efficiency and provide insights into the interactions between process variables, ultimately guiding the development of more efficient and sustainable remediation strategies for emerging contaminants in aquatic environments.

Summary: While a broad range of remediation strategies for removing ECs from aquatic environments have been covered in the literature, this deeper evaluation reveals certain gaps. The commonality across physical, chemical, and biological treatments, as well as advanced technologies, is their variable effectiveness and sustainability which are contingent upon specific environmental factors and contaminant characteristics. A clear lesson from this is the inadequacy of a one-size-fits-all approach. For instance, biological treatments may be more suitable for biodegradable ECs but less effective for persistent compounds. Furthermore, isolated remediation techniques may be insufficient for complex contaminant mixtures, thereby demanding innovative combined or emerging technologies. The research landscape is ripe for the development of more targeted, adaptive, and context-specific remediation strategies. These findings make a compelling argument for an evidence-based overhaul of current environmental management practices and policies to better safeguard human health and aquatic ecosystems. This not only underscores the value of the extant research but also strongly advocates for targeted investigation to fill the existing gaps.

7. Mitigation measures and policy recommendations

7.1. Source control and pollution prevention

Source control and pollution prevention measures aim to reduce the release of ECs into the environment at their source. Examples of these measures include proper disposal of pharmaceuticals and personal care products (PPCPs) (Wang and Wang, 2016), the use of alternative, less harmful chemicals in industrial processes (Naidu et al., 2016), and the implementation of best management practices in agriculture to minimize pesticide and nutrient runoff (Mohapatra et al., 2016). Furthermore, studies by Kristensen et al. (Kristensen and Mosgaard, 2020), Goyal et al. (Goyal et al., 2021), and Joensuu et al. (Joensuu et al., 2020) emphasize the importance of promoting the circular economy concept, which focuses on waste reduction, resource efficiency, and product life extension, to help minimize the environmental impacts of ECs.

7.2. Monitoring and surveillance programs

Monitoring and surveillance programs are essential for tracking the occurrence, fate, and effects of ECs in aquatic environments. These programs can inform risk assessments, guide remediation efforts, and inform policy development (Tong et al., 2022). Recent studies by Tong et al. (Tong et al., 2022), Petrie et al. (Petrie et al., 2016), and Altenburger et al. (Altenburger et al., 2019) have highlighted the importance of developing standardized sampling and analysis methods, expanding monitoring efforts to cover a wider range of ECs, and utilizing emerging techniques, such as passive sampling and biomonitoring, to improve our understanding of ECs in the environment.

7.3. Regulations and guidelines for emerging contaminants

The development of regulations and guidelines specific to ECs is crucial for managing their potential risks to human health and the environment. Existing regulations, such as the European Union's Water Framework Directive (WFD) and the United States' Safe Drinking Water Act (SDWA), have started to incorporate the mitigation of some ECs, but more comprehensive and internationally harmonized approaches are needed (Gaston et al., 2019). This includes setting environmental quality standards, establishing maximum contaminant levels for drinking water, and promoting the adoption of green chemistry principles in product design and manufacturing. Research by Omer (Omer, 2008), Nikolopoulou et al. (Nikolopoulou and Ierapetritou, 2012), and Matus et al. (Matus et al., 2012) underscore the importance of such regulatory efforts.

7.4. Public awareness and education

Raising public awareness and promoting education about the sources, risks, and mitigation measures for ECs can lead to more sustainable consumption patterns and waste management practices (Barbir et al., 2021). Initiatives such as public information campaigns, school curricula, and community engagement programs can foster environmentally responsible behaviours, including proper disposal of pharmaceuticals, reduction of single-use plastics, and support for environmentally friendly products (Pettipas et al., 2016). Studies by Luis et al. (Luís et al., 2020), Ramirez-Andreotta et al. (Ramirez-Andreotta et al., 2014), and Ruhoy and Conti et al. (Conti et al., 2021) have demonstrated the effectiveness of such initiatives in changing public behaviour and attitudes towards ECs.

Table 13
Comparison of the efficiency of various hybrid systems in removing a range of emerging contaminants.

Emerging contaminants	Treatment process	Influent (μg/L)	Removal (%)	Refs.
Endocrine disruption chemicals				
E1	Membrane bioreactor + ultra-filtration	5.0	99.40	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	99.60	Nguyen et al. (2013)
	Flocculants + activated sludge + ultra-filtration	0.03	96.0	Melo-Guimarães et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	99.30	Nguyen et al. (2013)
E2	Membrane bioreactor + ultra-filtration	5.0	99.50	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	99.40	Nguyen et al. (2013)
	Flocculants + activated sludge + ultra-filtration	0.01	96.50	Melo-Guimarães et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	99.60	Nguyen et al. (2013)
EE2	Membrane bioreactor + ultra-filtration	5.0	95.50	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	94.0	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	93.61	Nguyen et al. (2013)
	Flocculants + Activated sludge + ultra-filtration	0.04	95.0	Melo-Guimarães et al. (2013)
E3	Membrane bioreactor + Nano-filtration	5.0	97.70	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	96.10	Nguyen et al. (2013)
	Membrane bioreactor + ultra-filtration	5.0	98.30	Nguyen et al. (2013)
17β-estradiol-17-acetate	Membrane bioreactor + ultra-filtration	5.0	99.0	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	99.30	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	100	Nguyen et al. (2013)
Bisphenol A	Flocculants + activated sludge + ultra-filtration	0.86	95.0	Melo-Guimarães et al. (2013)
	Membrane bioreactor + ultra-filtration	5.0	98.60	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	89.30	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	90.50	Nguyen et al. (2013)
4-*n*-nonylphenol	Membrane bioreactor + ultra-filtration	5.0	96.90	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	96.60	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	100	Nguyen et al. (2013)
	Flocculants + activated sludge + ultra-filtration	14.64	97.0	Melo-Guimarães et al. (2013)
4-*tert*-butylphenol	Membrane bioreactor + ultra-filtration	5.0	95.50	Nguyen et al. (2013)
	Membrane bioreactor + ultra-filtration	5.0	98.20	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	91.0	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	95.70	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	83.50	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	93.80	Nguyen et al. (2013)
	Flocculants + activated sludge + ultra-filtration	4.39	98.0	Melo-Guimarães et al. (2013)
Pesticides				
Atrazine	Ozonation + biological activated carbon	0.001	70.0	Reungoat et al. (2012)
2,4-D	Ozonation + biological activated carbon	0.10	92.90	Reungoat et al. (2012)
	Flocculants + activated sludge + ultra-filtration	0.42	78.0	Nguyen et al. (2013)
Diazinon	Ozonation + biological activated carbon	0.78	99.30	Reungoat et al. (2012)
Diuron	Ozonation + biological activated carbon	0.05	99.10	Reungoat et al. (2012)
Fenoprop	Membrane bioreactor + ultra-filtration	5.0	10.60	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	17.70	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	18.60	Nguyen et al. (2013)
Metolachlor	Ozonation + biological activated carbon	0.002	69.0	Reungoat et al. (2012)
Praziquantel	Ozonation + biological activated carbon	0.003	97.10	Reungoat et al. (2012)
Triclopyr	Ozonation + biological activated carbon	0.12	87.40	Reungoat et al. (2012)
Pentachlorophenol	Membrane bioreactor + ultra-filtration	5.0	78.0	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	50.80	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	59.10	Nguyen et al. (2013)
Triclosan	Membrane bioreactor + ultra-filtration	5.0	99.20	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	99.20	Nguyen et al. (2013)
	SFCW + HFCW	55	32–7	Reyes-Contreras et al. (2011)
	Membrane bioreactor + nano-filtration	5.0	98.70	Nguyen et al. (2013)
	Flocculants + activated sludge + ultra-filtration	0.99	98.0	Melo-Guimarães et al. (2013)
Beta blockers				
Atenolol	Membrane bioreactor + reverse osmosis	0.92–2.44	99.50	Dolar et al. (2012)
	Ozonation + biological activated carbon	0.60	99.80	Reungoat et al. (2012)
Metoprolol	Membrane bioreactor + reverse osmosis	0.08	99.50	Reyes-Contreras et al. (2011)
	Ozonation + biological activated carbon	0.92	99.97	Reungoat et al. (2012)
Nadolol	Membrane bioreactor + reverse osmosis	0.05	99.50	Dolar et al. (2012)
Propranolol	Membrane bioreactor + reverse osmosis	0.31	99.50	Dolar et al. (2012)
	Ozonation + biological activated carbon	0.06	99.70	Reungoat et al. (2012)
Sotalol	Membrane bioreactor + reverse osmosis	0.22	99.50	Dolar et al. (2012)
Salbutamol	Membrane bioreactor + reverse osmosis	0.04	99.50	Dolar et al. (2012)
DEHP	Flocculants + activated sludge + ultra-filtration	10.32	77.0	Melo-Guimarães et al. (2013)
Salicylic acid	Membrane bioreactor + ultra-filtration	5.0	92.60	Nguyen et al. (2013)
	Membrane bioreactor + nano-filtration	5.0	97.30	Nguyen et al. (2013)
	Membrane bioreactor + reverse osmosis	5.0	95.40	Nguyen et al. (2013)
	SFCW + HFCW	3.50	91.0	Reyes-Contreras et al. (2011)
	Flocculants + activated sludge + ultra-filtration	28.478	98.50	Melo-Guimarães et al. (2013)
	Ozonation + ultrasound	31	100	Ibáñez et al. (2013)
Pharmaceuticals				
Amoxicillin	Ultrafiltration + activated carbon + ultrasound	10,000	99.90	Secondes et al. (2014)
Azithromycin	Membrane bioreactor + reverse osmosis	0.14	99.50	Dolar et al. (2012)

(*continued on next page*)

Table 13 (*continued*)

Emerging contaminants	Treatment process	Influent (μg/L)	Removal (%)	Refs.
Clarithromycin	Membrane bioreactor + reverse osmosis	2.72	99.50	Dolar et al. (2012)
	Ozonation + ultrasound	0.35	94.30	Ibáñez et al. (2013)
Clindamycin	Ozonation + ultrasound	0.01	100	Ibáñez et al. (2013)
Erythromycin	Ozonation + biological activated carbon	0.17	98.40	(Reungoat et al., 2012; Garcia-Rodríguez et al., 2014)
	Membrane bioreactor + reverse osmosis	0.08	99.50	Dolar et al. (2012)
Lincomycin	Ozonation + biological activated carbon	0.003	80.60	(Reungoat et al., 2012; Garcia-Rodríguez et al., 2014)
	Ozonation + ultrasound	0.21	66.70	Ibáñez et al. (2013)
Ofloxacin	Membrane bioreactor + reverse osmosis	2.90	99.50	Dolar et al. (2012)
	Ozonation + ultrasound	0.35	42.90	Ibáñez et al. (2013)
Roxithromycin	Ozonation + biological activated carbon	0.08	97	(Reungoat et al., 2012; Garcia-Rodríguez et al., 2014)
Sulfamethaxazole	Membrane bioreactor + reverse osmosis	0.27	99.50	Dolar et al. (2012)
	Ozonation + biological activated carbon	0.02	77.40	(Reungoat et al., 2012; Garcia-Rodríguez et al., 2014)
Sulfamethazine	Ozonation + ultrasound	0.21	100	Ibáñez et al. (2013)
Sulfathiazole	Ozonation + biological activated carbon	0.001	58.30	(Garcia-Rodríguez et al., 2014; Suarez et al., 2010)
Trimethoprim	Ozonation + biological activated carbon	0.027	99.30	(Garcia-Rodríguez et al., 2014; Suarez et al., 2010)
Tylosin	Ozonation + biological activated carbon	0.004	85	(Garcia-Rodríguez et al., 2014; Suarez et al., 2010)
	Ozonation + biological activated carbon	0.096	87	(Garcia-Rodríguez et al., 2014; Suarez et al., 2010)
4-aminoantipyrine	Ozonation + ultrasound	0.94	100	Ibáñez et al. (2013)
Diazepam	Membrane bioreactor + reverse osmosis	0.013–0.02	99.50	Dolar et al. (2012)
Lorazepam	Membrane bioreactor + reverse osmosis	0.05–0.08	99.50	Dolar et al. (2012)
Famotidine	Membrane bioreactor + reverse osmosis	0.12–0.13	99.50	Dolar et al. (2012)
Ranitidine	Membrane bioreactor + reverse osmosis	0.20–0.93	99.50	Dolar et al. (2012)
	Ozonation + biological activated carbon	0.007	86.40	Reungoat et al. (2012)
Citalopram	Ozonation + biological activated carbon	0.19	99.60	Reungoat et al. (2012)
Dapsone	Ozonation + biological activated carbon	0.002	75	Reungoat et al. (2012)
Doxylamine	Ozonation + biological activated carbon	0.29	99.30	Reungoat et al. (2012)
Phenytoin	Ozonation + biological activated carbon	0.37	97.50	Reungoat et al. (2012)
Risperidone	Ozonation + biological activated carbon	0.001	88.80	Reungoat et al. (2012)
Sertraline	Ozonation + biological activated carbon	0.02	97.20	Reungoat et al. (2012)
Venlafaxine	Ozonation + biological activated carbon	0.47	99.60	Reungoat et al. (2012)
	Ozonation + ultrasound	0.92	2.20	Ibáñez et al. (2013)
Clopidogrel	Membrane bioreactor + reverse osmosis	0.11–0.13	99.50	Dolar et al. (2012)
Hydrochlorothiazide	Ozonation + biological activated carbon	0.81	99.90	Reungoat et al. (2012)
Enalapril	Ozonation + ultrasound	0.25	100	Ibáñez et al. (2013)
Perindopril	Ozonation + biological activated carbon	0.74	97.30	Reungoat et al. (2012)
Irbesartan	Ozonation + ultrasound	1.73	48.60	Ibáñez et al. (2013)
Norbenzoylecgonine	Ozonation + ultrasound	0.18	100	Ibáñez et al. (2013)
Valsartan	Ozonation + ultrasound	0.87	23	Ibáñez et al. (2013)

7.5. Research priorities and future challenges

Future research on ECs should prioritize the identification of novel contaminants and their transformation products, improvement of analytical techniques, and evaluation of the long-term ecological and human health effects (Zhang et al., 2023b). Additionally, research is needed to develop innovative, cost-effective, and sustainable remediation technologies, as well as assess the effectiveness of existing and emerging mitigation measures (Tak et al., 2020). Interdisciplinary collaboration among scientists, policymakers, industry, and other stakeholders is essential to address these complex challenges and develop evidence-based strategies for managing the risks associated with ECs in aquatic environments (Hargrove and Heyman, 2020). Studies by Kah (NanopesticidesNanofertilizers, 2015), Richardson et al. (Richardson and Ternes, 2011), and Megahed et al. (Megahed and Ghoneim, 2021) emphasize the importance of interdisciplinary research and collaboration to tackle the multifaceted challenges posed by ECs.

8. Conclusions and future research directions

This paper provided a comprehensive overview of emerging contaminants in aquatic ecosystems, covering their detection, toxicity assessment, transformation pathways, environmental fate, remediation strategies, and mitigation measures. It highlighted the importance of understanding the sources, fate, and effects of these contaminants for the development of appropriate monitoring, risk assessment, and management strategies. Advanced analytical techniques have been developed to identify and quantify ECs in various environmental matrices, while in vitro and in vivo models have been employed to assess their toxicity. Furthermore, it discussed the complex transformation processes and factors affecting the environmental fate of ECs, as well as the current and emerging remediation technologies for their removal from aqueous environments.

The findings of this paper have several implications for environmental management and policy. First, the development of sensitive and reliable analytical methods for the detection of ECs is essential for effective monitoring and surveillance programs. Such programs can provide valuable information on the spatial and temporal distribution of ECs, allowing for targeted management interventions and the evaluation of remediation efforts. Second, understanding the toxicity of ECs and their transformation products is crucial for establishing science-based regulations and guidelines that protect aquatic organisms and human health. Third, the development and optimization of remediation strategies should consider the specific characteristics of the contaminants and the receiving environments, as well as the potential trade-offs between treatment efficiency, cost, and sustainability. Lastly, the implementation of mitigation measures, such as source control, pollution prevention, and public awareness campaigns, can help to reduce the release of ECs into aquatic environments and minimize their impacts on ecosystems and human health.

Several future research directions can be identified to advance our understanding of emerging contaminants and inform management and policy decisions. These include.

i) Expanding the range of ECs studied: A key area of focus in this field is the expansion of research scope to include a broader range of ECs and their transformation products. This necessitates the development and application of advanced analytical techniques that can accurately detect and monitor contaminant mixtures in environmental samples. These methods need to accommodate for

the continuous emergence of new contaminants, considering the complex and diverse nature of environmental matrices. This could involve leveraging novel technologies such as high-resolution mass spectrometry, advanced chromatography techniques, or machine learning for data analysis.

ii) Improving the ecological relevance of toxicity assessments: There is a need to improve the ecological relevance of toxicity assessments to better predict the environmental impacts of ECs. Traditional laboratory toxicity testing often fails to replicate real-world scenarios, creating a gap between experimental findings and actual ecological risks. Future research should focus on developing toxicity testing methodologies that incorporate real-world complexities like environmental variability, multiple stressors, and species interactions. This could involve designing advanced models that reflect various exposure scenarios, inter-species sensitivity, and the influence of biotic and abiotic factors.

iii) Investigating the long-term impacts of emerging contaminants: While significant research has been conducted on the acute and immediate effects of ECs, there is a dearth of information on potential long-term impacts. Understanding the chronic impacts of these contaminants is crucial for comprehensive risk assessment and management. Future studies should explore the potential long-term effects of ECs, such as changes in biological community structure, disruption of ecosystem services, persistent bioaccumulation, and indirect impacts on human health through food chains.

iv) Developing innovative and sustainable remediation technologies: Addressing the issue of ECs effectively requires the development of innovative, efficient, and sustainable remediation technologies. These technologies should be capable of removing multiple contaminants from various environmental matrices, including water, soil, and air. The exploration of advanced remediation technologies such as bioremediation, nanotechnology-based methods, advanced oxidation processes, and hybrid systems could offer potential solutions.

v) Enhancing interdisciplinary collaboration and stakeholder engagement: The multifaceted nature of emerging contaminant issues calls for an interdisciplinary approach. Collaborative efforts among scientists from different disciplines, policymakers, industry stakeholders, and the public are essential to the formulation of effective management strategies and policies. Enhancing stakeholder engagement can ensure that the devised strategies are practical, socially acceptable, and adequately address the local and regional specificities of contaminant issues. Such a comprehensive approach is key to protecting both human health and the environment in the face of increasing EC challenges.

Author Contributions Statement

M. Mofijur: Writing-Original draft.; M.M. Hasan: Writing-Original draft.; Shams Forruque Ahmed: Writing-Review and Editing.; F. Djavanroodi: Writing-Review and Editing.; I.M.R. Fattah: Writing-Review and Editing; A.S. Silitonga: Writing-Review and Editing.; M.A. Kalam: Writing-Review and Editing.; John L. Zhou: Conceptualization.; T.M. Yunus Khan: Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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