5 Advanced Oxidation Processes and Their Applications

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CONTENTS

5.1 INTRODUCTION

As a result of rising industrialization, wastewater production has increased dramatically, necessitating effective treatment to safeguard ground and surface water pollution (Khan et al., 2015). On the other hand, the heightened demand for water reclamation and reuse requires innovative treatment techniques. Besides, the stringent discharge standards further enhance the requirement for advanced treatment

DOI: 10.1201/9781003202431-5

options. The ineffective treatment creates many problems for the environment and pollutes the water bodies, which is problematic for aquatic life and public health (Mahtab et al., 2021). In recent years, trace organic compounds (TOrCs) have been reported in the aquatic environment, including pharmaceuticals, consumer items, and industrial chemicals (Khan et al., 2021). Aside from urban and agricultural run-offs, wastewater treatment plant effluents are thought to be the largest source of TOrC emissions (Gros et al., 2010; Luo et al., 2014). Because conventional physical and biological wastewater treatment can only partially remove TOrCs, they remain in wastewater treatment plant effluents released into surface waters (Luo et al., 2014). As a result, environmentalists and researchers are concerned about their suitable treatment. Advanced oxidation processes (AOPs) are considered highly effective and viable choices for the degradation and removal of a wide range of contaminants and TOrCs in these situations (Comninellis et al., 2008; Klavarioti et al., 2009; Yang et al., 2014; Stefan, 2018; Hussain et al., 2020; Hussain et al., 2021).

AOPs form powerful oxidants in situ to oxidize organic compounds (Huang et al., 1993; Hussain et al., 2020). These include processes that use OHradicals (•OH), which account for most AOPs, and those that use other oxidizing species, such as sulfate or chlorine radicals. Figure 5.1 depicts various oxidizing agents. Hydroxyl radical-based treatments have multiple advantages, as widely documented in earlier research. The hydroxyl radicals (•OH) are the principal reactive species in AOPs, and Figure 5.2 depicts several favorable characteristics of the hydroxyl radicals (•OH).

A comprehensive analysis of AOPs regarding running costs, sustainability, and general viability make selecting the most appropriate treatment techniques among AOPs challenging. Several AOPs are well-established and in full-scale operation in drinking water treatment and water reuse facilities, notably those involving ozonation and UV irradiation. Various researchers are constantly presenting innovative investigations of a range of evolving AOPs for water treatment (for example, electrochemical AOPs, plasma, electron beam, ultrasound, or microwave-based AOPs)

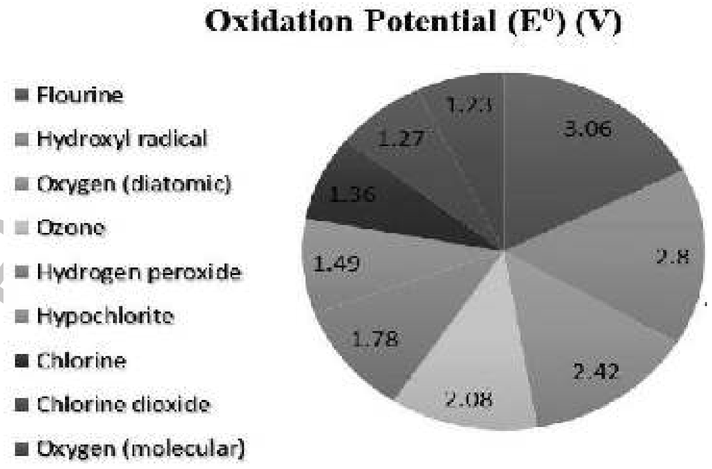

FIGURE 5.1 Various oxidizing agents' oxidation potential.

FIGURE 5.2 Characteristics of hydroxyl radicals.

TABLE 5.1
List of Various Advanced Oxidation Processes Reported in the Literature

Type of AOPs	Classification
Fenton-based	Fe^{2+}/H_2O_2, Fenton like
Ozone-based	O_3/H_2O_2, O_3/UV
Photochemical	UV/H_2O_2, UV/O_3, Photo-Fenton, Photocatalysis
Sonochemical	US/H_2O_2, US/O_3, Sono-Fenton
Electrochemical	Electro-Fenton
Sono-Photo Chemical	Sono-Photo-Fenton
Photo-Electro Chemical	Photo-Electro-Fenton
Sono-Electro Chemical	Sono-Electro-Fenton

(Stefan, 2018). When it comes to choosing the optimal treatment options, the characteristics of wastewater samples are crucial.

For samples with high biodegradable contents, such as high biological oxygen demand (BOD) and low toxicity, traditional biological treatment approaches (aerobic or anaerobic) are preferable. However, samples with low biodegradability will very often require chemical treatment. AOPs are used to treat complex wastewater containing refractory chemicals in general. AOPs' applicability is further enhanced by their high treatment efficacies and fast treatment times. These AOPs take less time to complete than conventional treatment techniques. Table 5.1 lists the many forms of AOPs that have been reported in the literature. The generation of reactive oxidative species in situ and the interaction of oxidants with target pollutants are two stages in all AOPs. The mechanisms of radical production are influenced by the proposed system and water quality and are dependent on process-specific characteristics. Other factors influence the effectiveness of contaminant removal in addition to radical scavenging.

5.2 TYPES OF DIFFERENT ADVANCED OXIDATION PROCESSES

5.2.1 Fenton-based AOPs

5.2.1.1 Classical Fenton Process (CFP)

The Fenton process has been considered the oldest method among AOPs given by the British chemist H.J.H. Fenton in 1894. The process was well-utilized for several wastewater treatments containing recalcitrant compounds. The process only utilizes two chemicals, namely ferrous ion (as the catalyst) and hydrogen peroxide (as an oxidizing agent). The combined chemicals (Fe^{2+} and H_2O_2) of the CFP are called Fenton's reagent. The advantages of the CFP are well-reported in the literature, like the ease in application, fewer chemicals requirement, quick degradation of a variety of pollutants, readily available and non-toxic chemicals requirement, etc.

On the other hand, the reported drawbacks of the CFP are restricting its widespread applications, especially for full-scale. The reported disadvantages are a large amount of iron sludge production after the treatment, a narrow working pH range requirement, and the high dosage of chemicals required for high treatment efficacies. The dosage of the reagents varies depending on the sample type and required treatment efficacies. Reaction pH and reagent dosage are the main influencing factors in the CFP. The pH of around 2.5–4 is effective, as per the reported studies. The reactions involved in the CFP in the absence of the organic compounds have been summarized below:

$$\mathrm{Fe(II)} + \mathrm{H_2O_2} \rightarrow \mathrm{Fe(III)} + {}^{\bullet}\mathrm{OH} + \mathrm{OH^-} \tag{5.1}$$

$$\mathrm{Fe(III)} + \mathrm{H_2O_2} \rightarrow \mathrm{Fe(II)} + \mathrm{HOO^{\bullet}} + \mathrm{H^+} \tag{5.2}$$

$$\mathrm{H_2O_2} + {}^{\bullet}\mathrm{OH} \rightarrow \mathrm{HOO^{\bullet}} + \mathrm{H_2O} \tag{5.3}$$

$$\mathrm{Fe(III)} + \mathrm{HOO^{\bullet}} \rightarrow \mathrm{Fe(II)} + \mathrm{O_2} + \mathrm{H^+} \tag{5.4}$$

$$\mathrm{Fe(II)} + {}^{\bullet}\mathrm{OH} \rightarrow \mathrm{Fe(III)} + \mathrm{OH^-} \tag{5.5}$$

$$\mathrm{Fe(II)} + \mathrm{HOO^{\bullet}} \rightarrow \mathrm{Fe(III)} + \mathrm{HO_2} \tag{5.6}$$

$$\mathrm{HOO^{\bullet}} + \mathrm{HOO^{\bullet}} \rightarrow \mathrm{H_2O_2} + \mathrm{O_2} \tag{5.7}$$

$${}^{\bullet}\mathrm{OH} + \mathrm{HOO^{\bullet}} \rightarrow \mathrm{H_2O} + \mathrm{O_2} \tag{5.8}$$

$$ {}^{\bullet}OH + {}^{\bullet}OH \rightarrow H_2O_2 \tag{5.9}$$

The hydroxyl radicals degrade the organic compounds by the following three mechanisms, i.e. hydrogen abstraction, hydroxyl addition, and electron transfer, as shown in reactions (11–13) (Huang et al., 1993; Bello et al., 2019).

$$ {}^{\bullet}OH + \text{Organic compounds} \rightarrow \text{Products} \tag{5.10}$$

$$ {}^{\bullet}OH + RH \rightarrow R^{\bullet} + H_2O \tag{5.11}$$

$$ {}^{\bullet}OH + R \rightarrow R(OH) \tag{5.12}$$

$$ {}^{\bullet}OH + Fe^{2+} \rightarrow Fe^{3+} + OH^{-} \tag{5.13}$$

5.2.1.2 Fenton-like Process

Several wastewaters have been treated using the Fenton reaction and the Fenton-like reaction initiated by Fe^{3+} and H_2O_2. Differentiating these processes is pointless from a mechanistic perspective because Fe^{2+} and Fe^{3+} are present in the chain of Fenton reactions depicted in the initial reactions. Once Fenton oxidation begins, all initially added Fe^{2+} is quickly oxidized to Fe^{3+}, resulting in a system that acts independently of iron oxidation states (Pignatello et al., 2006). However, a significant distinction in actuality is that at the start of Fenton oxidation, the rapid development of hydroxyl radicals may occur. In contrast, Fenton-like oxidation has a moderate generation rate of hydroxyl radicals. Because the rate constant in reaction (1) is substantially greater than in reaction (2), the latter reaction becomes a rate-limiting step, slowing the release of hydroxyl radicals. Fenton and Fenton-like reactions have similar organics removal efficiency, according to Rivas et al. (2003). According to Kim et al. (2001), the Fenton reaction removed more COD and had a higher BOD5/COD ratio than the Fenton-like reaction. Furthermore, the optimal pH of 3.0 for Fenton oxidation was lower than the optimal pH of 4.5 for the Fenton-like reaction.

5.2.1.3 Photo-Fenton Process

This process involves UV light radiations to attain the higher production of ${}^{\bullet}OH$ and to regenerate Fe^{2+} ions (Kim et al., 1997). The UV or visible light radiation of wavelength below 450 nm is preferably used in the process (Zepp et al., 1992; Mahtab and Farooqi, 2020). In this process, the photoreduction of Fe^{3+} by UV irradiation causes the photochemical regeneration of Fe^{2+}, which reacts with H_2O_2 and produces ${}^{\bullet}OH$ and Fe^{3+} ions, as shown in Eq. (5.14). The regeneration of Fe^{3+} continues the cycle and leads to higher ${}^{\bullet}OH$ production, which enhances the Fenton's oxidation performance (Faust and Hoigné, 1990). The process also accompanies the direct photolysis of H_2O_2 to generate the ${}^{\bullet}OH$, as shown in Eq. (5.15). However, the presence of iron complexes in a solution absorbs a large part of radiation and affects the photolysis of H_2O_2 (Safarzadeh-Amiri et al., 1997). The role of pH is vital in the photo-Fenton process (PFP), which determines the formation of different iron complexes. At a pH value of 3, Fe^{3+} ions effectively converted into the most photoreactive ferric ion water complex, i.e. $[Fe(OH)]^{2+}$ species. The metal charge transfer excitation of $[Fe(OH)]^{2+}$ by UV radiation regenerates Fe^{2+} and produces $^{\bullet}OH$, as shown in Eq. (5.16) (Faust and Hoigné, 1990; Avetta et al., 2015). Acidic conditions ($pH = 3$) also favor the conversion of carbonates and bicarbonates into carbonic acid, which comparatively exhibits low susceptibility toward $^{\bullet}OH$ radicals (Legrini et al., 1993).

$$Fe^{3+} + hv + H_2O \rightarrow Fe^{2+} + {}^{\bullet}OH + H^{+} \tag{5.14}$$

$$H_2O_2 + hv \rightarrow 2\,{}^{\bullet}OH \tag{5.15}$$

$$\left[Fe(OH)\right]^{2+} + hv \rightarrow Fe^{2+} + {}^{\bullet}OH \tag{5.16}$$

The addition of ligands may further enhance the regeneration of Fe^{2+}. These complexes under UV irradiation follow the ligand to metal charge transfer step and regenerate Fe^{2+} ions, as shown in Eq. (5.17). In general, the combination of photochemistry and the Fenton process is a very compelling technology.

$$Fe^{3+} - L + hv \rightarrow Fe^{2+} + L^{\bullet +} \tag{5.17}$$

5.2.1.4 Electro-Fenton Process

This process involves using electrons to complement the CFP. The Electro-Fenton process (EFP) works on the principle of cathodic reduction of Fe^{3+} and O_2 to generate Fenton's reagents, i.e., Fe^{2+} and H_2O_2 (He and Zhou, 2017). The EFP can be classified into four types based on Fenton's reagent formation. Type 1 involves using oxygen sparging cathode and sacrificial anode for the generation of H_2O_2 and Fe^{2+}, respectively, with no external addition of reagents (Ting et al., 2008). In Type 2, Fe^{2+} is generated from the sacrificial anode while H_2O_2 is externally added. In Type 3, oxygen sparging cathodes are used for the electro-generation of H_2O_2, and Fe^{2+} is externally added (Bello et al., 2019). Type 4 involves the electrolytic regeneration of Fe^{2+} by the cathodic reduction of Fe^{3+} ions (Zhang et al., 2006). However, type 3 is the most popular EFP that is used for the continuous electro-generation of H_2O_2. In a typical process, a constant oxygen gas supply at the cathode in an acidic medium causes its two-electron reduction. It leads to the formation of $H_2O_{2,}$ as shown in Eq. (5.18) (Pliego et al., 2015). Initially, a small quantity of ferrous salts is added to the cell to react with H_2O_2 and generate Fe^{3+}, which continues the cathodic electro-regeneration of Fe^{2+}, as shown in Eq. (5.19) (Brillas et al., 2009). The sacrificial anode oxidation of iron is also significant in terms of the production of $Fe^{2+,}$ as shown in Eq. (5.20) (Varank et al., 2020).

$$O_2 + 2H^{+} + 2e^{-} \rightarrow H_2O_2 \tag{5.18}$$

$$Fe^{3+} + e^{-} \rightarrow H_2O_2 \tag{5.19}$$

$$Fe \rightarrow Fe^{2+} + 2e^{-} \tag{5.20}$$

The process is very advantageous over the CFP. The electro-generation of H_2O_2 could lead to an 80% cost reduction and save the associated transport and handling cost. The effective utilization of Fe^{3+} and continuous regeneration of Fe^{2+} minimizes the problem of sludge production and enhances the production of $^{\bullet}OH$ (Huang and Chu, 2012; Pliego et al., 2015). However, several factors like pH, current density, dissolved oxygen level, catalyst concentration, electrolytes, electrode nature, and temperature affect the efficiency of the process. EFP showed the same trend of results for solution pH, temperature, and initial concentration of pollutants as exhibited by the CFP. Applied current is an essential factor determining the electron generation and regeneration of H_2O_2 and Fe^{2+}, respectively. The higher applied current leads to higher efficiency but up to a specific limit. A value higher than certain pre-determined levels causes parasitic reactions and adversely affects the performance of the process. Lin and Chang (2000) have reported results in 69% of COD removal and 15.82% of NH_3-N reduction for treating landfill leachate by the EFP. The process further increased the biodegradability of leachate from a value of 0.1 to 0.29. In general, the EFP with the high in-situ generation of H_2O_2, electro-regeneration of Fe^{2+}, and low sludge production is very advantageous but requires high energy. The operational costs involved in the EFP include labor, material, cost of energy consumption, fixed, and disposal costs, but the major part of these costs in the EFP comes from the consumption of electric energy (Tirado et al., 2018). The high treatment cost due to the high electricity consumption is considered the main drawback of the EFP. The higher duration of treatment for the adequate mineralization of the resistant intermediates formed in the process leads to higher associated treatment costs (Monteil et al., 2019). The higher currents lead to the adequate mineralization of contaminants and add up to the higher electric energy consumption. Hence, it is essentially required to correctly set the applied current density that marks the balance between the energy-related costs and the efficiency of the process (He and Zhou, 2017). The electricity consumption in the electro-Fenton treatment process is analyzed by Eq. (5.21) (Tirado et al., 2018).

$$\text{Energy consumption} = \left\{U \cdot I \cdot T \cdot 1{,}000 / \left(\text{COD}_\text{o} - \text{COD}\right)V\right\} \tag{5.21}$$

where U = consumed electric energy (kWh/kg COD), I = current intensity (Amp), T = time (h), V = volume of water (L), COD_o = initial COD (mg/L), and COD = final COD (mg/L).

5.2.1.5 Heterogeneous Fenton Catalysis

On the other hand, heterogeneous Fenton catalysis, one of the sophisticated oxidation technologies, is of great interest for pollutant removal due to its intrinsic procedure and extensive application (Xia et al., 2011). Heterogeneous Fenton-like reactions on solid catalysts may efficiently catalyze the oxidation of organic pollutants over a wide pH range, which is beneficial for in-situ treatment of polluted groundwater and soil and can be reused for consecutive cycles. A surface-controlled reaction, a heterogeneous Fenton-like response, is governed by the catalyst surface area, H_2O_2 concentration, reaction temperature, pH, and ionic strength of the solution (Matta et al., 2007). When only Fe^{3+} is present initially, Fe^{2+} slowly regenerates and commences oxidation processes.

5.2.2 Ozon-based AOPs

Ozone-based AOPs are also widely used to treat a variety of wastewaters. The oxidative power of ozone is high ($E^{\circ} = 2.08\,\text{V}$) (Figure 5.1). The molecular structure of organic substances is altered by ozone, which oxidizes them into more biodegradable compounds that may be eliminated by biological treatment. Ozone-based AOPs significantly reduce COD and BOD levels in leachate and other wastewaters. Rivas et al. (2003) used an ozone dosage of 1.3–1.5 g O_3/g of COD for 1 hour to produce a 30% COD reduction, but Hagman et al. (2008) used 4 g/L O_3 to obtain a 22% COD reduction at pH 8–9. Wang et al. (2004) also found a drop in leachate alkalinity from 4,030 to 2,900 mg/L by 12.5 g O_3/L. The ozonation process is pH-dependent and can take place either through molecular ozone reactions (direct electrophilic attack on refractory contaminants) or by the formation of •OH radicals (indirect attack due to ozone breakdown) (Kurniawan et al., 2006). The following are the reactions that occur during the ozonation process:

$$O_3 + OH^- \rightarrow {}^{\bullet}HO_2 + {}^{\bullet}O_2^- \tag{5.22}$$

$$HO_2 \leftrightarrow {}^{\bullet}O_2^- + H^+ \tag{5.23}$$

$$O_3 + {}^{\bullet}O_2^- \rightarrow {}^{\bullet}O_3^- + O_2 \tag{5.24}$$

$${}^{\bullet}O_3^- + H^+ \leftrightarrow {}^{\bullet}HO_3 \tag{5.25}$$

$${}^{\bullet}HO_3 \rightarrow {}^{\bullet}OH + O_2 \tag{5.26}$$

$${}^{\bullet}OH + O_3 \rightarrow {}^{\bullet}HO_4 \tag{5.27}$$

$${}^{\bullet}HO_4 \rightarrow {}^{\bullet}HO_2 + O_2 \tag{5.28}$$

$${}^{\bullet}HO_4 + {}^{\bullet}HO_4 \rightarrow H_2O_2 + 2O_3 \tag{5.29}$$

$${}^{\bullet}HO_4 + {}^{\bullet}HO_3 \rightarrow H_2O_2 + O_3 + O_2 \tag{5.30}$$

When the pH of a reaction increases over 9.0, ozone-resistant compounds known as hydroxyl radical scavengers form, which prevent oxidation; as illustrated in reactions (31) and (32), carbonate ions generated from bicarbonate ions act as scavengers, slowing down the rate of oxidation (Kurniawan et al., 2006).

$$CO_3^{-2} + {}^{\bullet}OH \rightarrow OH^- + {}^{\bullet}CO_3^- \tag{5.31}$$

$$HCO_3^- + {}^{\bullet}OH \rightarrow OH^- + HCO_3^{\bullet} \tag{5.32}$$

5.2.2.1 Peroxone Process (O_3/H_2O_2)

The use of hydrogen peroxide in conjunction with ozonation resulted in a more significant reduction in COD. The introduction of the radical system can be used to oxidize refractory compounds because it allows for selective molecular ozone reactions before the process is changed to free radical attack (non-selective). Ozone and hydrogen peroxide combine in a complicated series of reactions to produce •OH radicals, as demonstrated in reactions (33–39) (Langlais et al., 1991). Two hydroxyl radicals are produced as a result of these reactions, which comprise one H_2O_2 and two O_3 molecules (Schulte et al., 1995).

$$H_2O_2 + H_2O \leftrightarrow {}^{\bullet}HO_2^- + H_3O^+ \tag{5.33}$$

$$O_3 + {}^{\bullet}HO_2^- \rightarrow {}^{\bullet}OH + {}^{\bullet}O_2^- + O_2 \tag{5.34}$$

$${}^{\bullet}O_2^- + H^+ \leftrightarrow {}^{\bullet}HO_2 \tag{5.35}$$

$${}^{\bullet}O_2^- + O_3 \rightarrow {}^{\bullet}O_3^- + O_2 \tag{5.36}$$

$${}^{\bullet}O_3^- + H^+ \leftrightarrow {}^{\bullet}HO_3 \tag{5.37}$$

$${}^{\bullet}HO_3 \rightarrow {}^{\bullet}OH + O_2 \tag{5.38}$$

$$H_2O_2 + 2O_3 \rightarrow 2{}^{\bullet}OH + 3O_2 \tag{5.39}$$

At an initial pH of 7, 60 minutes of 5.6 gm O_3/hour ozone injection followed by 400 mg/L H_2O_2 resulted in a 72% COD reduction and an increase in the BOD_5/COD ratio from 0.01 to 0.24 (Cortez et al., 2011). Ozone has long been utilized in water treatment as an oxidant and disinfectant. Ozone is an electron-rich oxidant that mostly affects double bonds, amines, and activated aromatic rings (e.g., phenol). Because its reactions in actual aqueous solutions frequently contain the formation of •OH, ozonation is commonly classified as an AOP or AOP-like process. The reaction's initiation, on the other hand, is rather sluggish, with a second-order rate constant of 70 $M^{-1}\,s^{-1}$.

Although peroxide is produced due to ozone interactions with the aqueous matrix, its contribution to the overall •OH formation during wastewater ozonation is insignificant (Nöthe et al., 2009). The O_3/H_2O_2 technique is well-established in water treatment and reuse applications. However, studies have shown that its benefits for wastewater applications are restricted due to severe competitive reactions (Hübner et al., 2015). It may, however, be a viable treatment option for reducing bromate generation during ozonation.

5.2.2.2 Ozonation at Elevated pH

Ozonation at high pH is a useful AOP (Buffle et al., 2006). The pH of treated water influences the effectiveness of direct ozonation (Calderara et al., 2002). If calcium carbonate precipitation is not a problem, ozonation at pH > 8 may be feasible. Because leachate is a complex matrix with high organic content, ozone treatment cannot satisfy the discharge standards alone. A high ozone dose is required to reduce COD, which makes this approach energy-intensive. Because some ozone is lost in the off-gas entering the ozone reactor, all ozone-based AOPs have a lower ozone mass transfer from gas to liquid. Although efforts are being made to increase ozone mass transfer efficiency, ozonation remains a viable treatment option as a pre- or post-treatment of leachate (Miklos et al., 2018).

5.3 APPLICATIONS OF ADVANCED OXIDATION PROCESSES

The diverse uses of different AOPs for several wastewater treatments, including landfill leachate, pharmaceutical wastewater, municipal wastewater, textile wastewater, and other industrial effluents, have been described in the literature. Table 5.2 summarizes previous studies on AOPs.

TABLE 5.2
The Various Applications of AOPs Reported in the Literature

Sample/Wastewater Type	Type of AOPs	References
Municipal landfill leachate	Classical Fenton process	Pieczykolan et al. (2013)
Sanitary landfill leachate	Coagulation-Fenton process	Moradi and Ghanbari (2014)
Urban landfill leachate	Fenton-like process	Martins et al. (2012)
Landfill leachate	Photo-electro-Fenton process	Altin (2008)
Municipal landfill leachate	Classical Fenton process	Gau and Chang (1996)
Municipal landfill leachate	Classical Fenton process	Kim and Huh (1997)
Biologically pre-treated leachate	Classical Fenton process	Welander and Henrysson (1998)
Pre-treated leachate	Electro-Fenton process	Lin and Chang (2000)
Biologically pre-treated leachate	Photo-Fenton process	Lau et al. (2002)
Mature leachate	Photo-Fenton process	De Morais and Zamora (2005)
Landfill leachate	Electro-Fenton process	Zhang et al. (2006)
Mature landfill leachate	Classical Fenton process	Deng (2007)
Mature leachate	Integrated Fenton-ultrafiltration process	Primo et al. (2008)
Fenton-ultrafiltration process	Classical Fenton process	Cortez et al. (2010)
Municipal landfill leachate	Classical Fenton process	Yilmaz et al. (2010)
Stabilized landfill leachate	Classical Fenton process	Mohajeri et al. (2011)
Mature landfill leachate	Photo-Fenton process	Jain et al. (2018)
Raw leachate	Sono-photo-Fenton process	Zha et al. (2016)
Pre-coagulated leachate membrane concentrates	Classical Fenton process	Xu et al. (2009)

(*Continued*)

TABLE 5.2 (*Continued*)
The Various Applications of AOPs Reported in the Literature

Sample/Wastewater Type	Type of AOPs	References
Stabilized landfill leachate	Heterogeneous electro-Fenton process	Sruthi et al. (2018)
Stabilized landfill leachate	Heterogeneous Fenton process	Niveditha and Gandhimathi (2020)
Municipal wastewater	Ozonation process	Tofani and Richard (1995)
Synthetic wastewater	Ozonation process	Beltran et al. (2000)
Landfill leachate	Classical Fenton process	Mahtab et al. (2021)
Pharmaceutical wastewater	Ozonation and peroxonation process	Alaton et al. (2004)
Industrial wastewater (textile)	Ozonation + ferrous sulfate (coagulation) process	Selcuk (2005)
Industrial wastewater (pharmaceuticals)	Ozonation process	Hernando et al. (2007)
Landfill leachate	Ozone/hydrogen peroxide process	Tizaoui et al. (2007)
Industrial wastewater (steel)	Ozonation process	Chang et al. (2008)
Effluent (dyes and textile industries)	Ozonation process	Pachhade et al. (2009)
Wastewater	Ozonation process	Turhan and Ozturkcan (2013)
Landfill leachate	Ozonation process	Amr et al. (2014)
Winery wastewater	Photo-electro-Fenton process	Díez et al. (2016)
Winery wastewater	Adsorption and photo-Fenton process	Guimarães et al. (2019)
Winery wastewater	Electro-Fenton-photolytic reactor	Díez et al. (2017)
Olive oil mill wastewater origin	Photo-Fenton	García and Hodaifa (2017)
Olive oil mill wastewater origin	Coagulation/flocculation followed by solar photo-Fenton oxidation	Ioannou-Ttofa et al. (2017)
Olive oil mill wastewater origin	Coagulation/flocculation followed by Fenton oxidation and biological treatment (only industry)	Amaral-Silva et al. (2017)
Olive oil mill wastewater Origin	Combined electrocoagulation (ECR)-photocatalytic (PCR) degradation system	Ates et al. (2017)
Olive oil mill wastewater origin	Combined ozone/Fenton process	Kirmaci et al. (2018)
Meat processing plants wastewater origin	Classical Fenton process	Masoumi et al. (2015)
Dairy wastewater	Electro-Fenton process	Davarnejad and Nikseresht (2016)
Dairy wastewater	Electro-Fenton process	Akkaya et al. (2019)
Food industry	Ultrasonic irradiation	Yılmaz and Fındık (2017)
Food industry	Ozone-based processes	Guzmán et al. (2016)
Food dye	Heterogeneous E-Fenton process	Barros et al. (2016)
Food dye	Photocatalytic process	Júnior et al. (2019)

5.4 CONCLUSION

This chapter overviews various AOPs' basic details and applications. The general reaction mechanisms and specific information regarding AOPs are also highlighted. It can be concluded that the versatile applications of AOPs are still in demand and need to be further explored to reduce the associated drawbacks. The studies on the disadvantages of specific AOPs and their sustainable solutions could be the likely domain for further research. The extensive full-scale applications of AOPs are limited and need to be explored further. It was observed that integrated treatment technologies are much more efficient and environmentally friendly. Hence, the combined treatment technologies should be implemented for complex wastewater treatments. The single treatment options are difficult to achieve stringent discharge standards, which further favors the requirement of combined treatment technologies. The generated sludge after the treatment should also be efficiently handled or managed to avoid the secondary pollution of soil, ground, and surface water. Several AOPs are efficient only as a pre- or post-treatment option; hence, a suitable selection of AOPs is essential for overall processes' performance.

ACKNOWLEDGMENT

The authors would like to express their gratitude to the Department of Civil Engineering (Aligarh Muslim University) for providing technical assistance with this chapter.

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