A review on heterogeneous oxidation of acetaminophen based on micro and nanoparticles catalyzed by different activators

Abstract: Emerging contaminants are the contaminants that newly identified their adverse effects on the environment. Pharmaceutical compounds have gained researchers' attention among developing organic pollutants as the demand for pharmaceutical compounds has increased, implying their continuing release into the environment. Acetaminophen (ACT) is a popular drug that is wildly used without prescription for the relief of headaches and rheumatic pains. In some places, the detected values of ACT are more than the natural values, which may seriously threaten the environment. Many methods have been applied to remove ACT from water. The advanced oxidation process (AOP) based on micro and nanoparticles has shown promising results to remove ACT from an aqueous medium. This review provides a summary and an organization of the scattered available information related to studies that investigated the removal of ACT from water by an AOP based on micro and nanoparticles. Many topics investigated in this review include the influence of temperature, pH, catalyst concentration, pollutant concentration, the effects of scavengers and oxidants, the stability of the catalyst, and doping ratio. The main results obtained for the removal of ACT by using micro and nanoparticles have been discussed in this review.

Keywords: advanced oxidation process, acetaminophen, paracetamol, heterogeneous, microparticles, nanoparticles

Abbreviations

ACT acetaminophen
AO ammonium oxalate
AOP advanced oxidation process
BQ benzoquinone
CF NPs cobalt ferrite magnetic nanoparticles
CNT carbon nanotube
CoAl-LDH CoAl-layered double hydroxide
CS-Fe chitosan-Fe
CWAO catalytic wet peroxide oxidation
DO dissolved oxygen
ECDs endocrine disrupting compounds
EDTA-2Na ethylenediaminetetracetic acid disodium
EOCs emerging organic contaminants
EtOH ethanol
EPR electron paramagnetic resonance
Fe/N-CNT Fe and N codoped carbon nanotube
GAC granular activated carbon
GR green rust
IPA isopropanol
l-his l-histidine
MCM mobil composition of matter
MGF Fe2O3
MGN nanostructured magnetite Fe2O3 powders
MnCN Mn-doped graphite phase carbon nitride
MNP@β-CDP magnetite nanoparticles modified
β-cyclodextrin crosslinked polymeric
MNPs@C magnetic mesoporous carbon

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Mpg-C$_3$N$_4$  graphene and mesoporous graphitic carbon nitride  
NS-CMK-3  N/S copped ordered mesoporous carbon  
OVPTCN  oxygen vacancies and phosphorus coppeded black titania-coated carbon nanotube composites  
PAA  application of peracetic acid  
PDS  sodium persulfate  
PILC  pillared clays  
PMS  peroxymonomosulfate  
POD  horseradish peroxidase  
PS  persulfate  
PPCPs  pharmaceutical and personal care products  
PSW  powdered stone waste  
rGO  reduced graphene oxide  
SBA-15  Santa Barbara amorphous 15  
SOD  superoxide dismutase  
TBA  tert butyl alcohol  
TCuO50-GO  tomato skin inspired copper oxide-graphene oxide  
TEMPOL  4-hydroxy-2,2,6,6-tetramethylpiperidine-1-oxyl  
TEOA  triethanolamine  
TFSZ  titania loaded fibrous silica  
TiO$_2$ NT  titanium dioxide nanotube  
US  ultrasound  
UV  ultraviolet light  
UVA  ultraviolet with wavelength between 315 and 400 nm  
UVC  ultraviolet with wavelength between 100 and 280 nm  
WWTPs  wastewater treatment plants  
ZIF-8  zeolitic imidazolate frameworks-8  
ZSM-5  zeolite socony mobil-5  
ZVC  zero-valent copper  
ZVI  zero-valent iron

1 Introduction

Emerging organic contaminants (EOCs) have recently gained attention due to their resistance to oxidation and negative impact on the environment [1]. EOCs groups including pharmaceuticals and personal care products, pesticide, disinfection byproducts, wood preservation, endocrine disrupting compounds, bacteria, cyanotoxins, and industrial chemicals [2]. The continuous introduction of these bioactivity compounds into the environment in many ways. Even in low concentrations, they attracted the regulatory organization and governments [3]. Among EOCs compounds, pharmaceutical compounds have attracted real attention because of their negative impacts on public health and the environment [4,5]. Many pharmaceutical products are widely used as anti-inflammatories, analgesic, lipid regulators, antibiotics, antiepileptics, antiseptics, and disinfections. Nowadays, a large number of prescription and nonprescription cure have been used around the world [6–8]. Recently, the consumption of pharmaceutical compounds has increased, which means the continuous release of them into the environment. In the European Union, the use of pharmaceutical compounds could reach thousands of tons per year [9]. With passing time, they may reach a specific concentration causing chronic toxicity effects for humans and the organisms. Moreover, pharmaceutical compounds may enter into the human body through inhalation, ingestion, or transdermal delivery leading to accumulation in tissues, reproductive damage, inhibition of cell proliferation, and behavioral changes [10–12]. It should be noted that there is no standard set for the discharge limits of the pharmaceuticals in an aqueous medium, and the researches that examined pharmaceutical concentrations in water are limited [13]. The sources of pharmaceutical compounds are varied: they could be released from wastewater treatment plants (WWTPs), hospitals, medical care centers, landfills, domestic sewage systems (the drugs could discharge with the urine and manure of the human body through the sewer system), and industries (unused, expired, and residual); as a result, pharmaceutical compounds have been found in the surface water and groundwater [14,15].

Acetaminophen (ACT) or paracetamol (C$_8$H$_9$NO$_2$, $M_w = 151.163$, $N$-(4-hydroxyphenyl)ethanamide) is a popular drug that is widely used without prescription for the relief of headache, migraine, neuralgia, backache, and rheumatic pains [16,17]. The natural value of ACT in the surface water and municipal WWTPs has been detected to be less than 100 ng/L. However, the detected values of ACT in the industrial effluent from a few to tens of milligrams per liter, which may seriously threaten the aquatic organisms and the environment [18–20]. In addition, ACT has some toxic byproducts, such as 1,4-benzoquinone and N-acetyl-P-benzoquinone, which can harm the kidney and liver in humans [21]. Freshwater scarcity, and the rising drinking water demand, is considered one of the environmental problems in the twenty-first century according to its effect on environment, economic, and society [22]. The increase in drinking water demand may attribute to many reasons, such as industrialization, the increase in population, environmental change, and environmental demand. To fulfill the increasing demand for drinking water and to skip any further
accumulation of pollutants in the environment, it has become important to treat contaminated water, and also it helps to remove the pollutants mixing with clean water sources [23]. Many approaches have been applied to remove ACT from water [24–27]. Thus, approaches are classified into three major processes: physical, chemical, and biological process. Among these approaches, chemical oxidation proved its ability to degrade recalcitrant compounds, such as ACT, that resist the bioremediation [18]. The researchers’ concern for the ACT degradation is evident from the number of research publications on ACT removal per year, as shown in Figure 1. Last 50 year survey on oxidation of ACT showed 1,620 documents in scopus.com, out of which 56% (900) publications were reported in the last 10 years. Data analysis on the literature survey showed that 92.2% research articles, 4.3% review articles, and 1.6% conference articles were published during this period. This current study, aims to review the recent studies that deal with advanced oxidation process (AOP) based on micro and nanoparticles to remove ACT from an aqueous medium. Also, this study provides a summary and an organization of the scattered available information related to this subject. Many topics investigated in this review include the influence of temperature, pH, catalyst concentration, pollutant concentration, the effects of additives (scavengers and oxidants), the reusability and durability of the catalyst, and doping ratio. The main results obtained for the removal of ACT by using micro and nanoparticles have been discussed in this review.

2 AOP

The mechanism of AOP relies on the activation of some molecules to create reactive species called radicals. The examples of AOP techniques are varied such as Fenton, metal/PS, electrochemical, ultrasound/oxidant, ultrasonic irradiation, ozonation in the presence of particles, and nanoparticles/ultraviolet light (UV). Application of these processes into the water medium generates radicals with a high oxidation–reduction potential that can oxidize different organic compounds [28–31]. There are two mechanisms for the AOP: heterogeneous and homogeneous. In the microcomposite-based and nanocomposite-based catalyst systems, oxidation mainly occurs on the catalyst surface; therefore, the heterogeneous reaction pathway is dominant. Many heterogeneous mechanisms were proposed, for example, singlet oxygen, surface-activated complex, surface-confined sulfate radicals, and surface electron transfer (catalysts as an electron conductor) [32]. Chemical Eqs. (1)–(3) represent iron oxide-based catalyst for heterogeneous nonradical-based reaction and Eqs. (4)–(12) represent iron oxide-based catalyst and persulfate (PS) as an oxidant for homogeneous radical-based reaction [5].

![Figure 1](https://example.com/f1.png)  
**Figure 1:** (a) The number of documents of ACT degradation by year and (b) the percentage of each type of documents. Source: Scopus database.
\[\text{Fe}(II) + S_2O_8^{2-} \rightarrow \text{Fe}(III) + SO_4^{2-}\]  
(1)

\[\text{Fe}(III) + S_2O_8^{2-} \rightarrow \text{Fe}(II) + S_2O_6^{2-}\]  
(2)

\[\text{Fe}(II) + SO_4^{2-} \rightarrow \text{Fe}(III) + SO_2^{2-}\]  
(3)

\[SO_4^{2-} + H_2O \rightarrow SO_2^{2-} + H^+ + OH^-\]  
(4)

\[SO_2^{2-} + S_2O_8^{2-} \rightarrow SO_2^{2-} + S_2O_6^{2-}\]  
(5)

\[OH^- + S_2O_8^{2-} \rightarrow HSO_4^- + SO_4^{2-} + \frac{1}{2}O_2\]  
(6)

\[OH^- + S_2O_8^{2-} \rightarrow OH^- + S_2O_6^{2-}\]  
(7)

\[SO_2^{2-} + OH^- \rightarrow SO_2^{2-} + OH^-\]  
(8)

\[SO_2^{2-} + SO_2^{2-} \rightarrow S_2O_5^{2-}\]  
(9)

\[SO_2^{2-} + OH^- \rightarrow HSO_4^-\]  
(10)

\[SO_2^{2-} + \text{Pollutant} \rightarrow \text{Byproducts}\]  
(11)

\[OH^- + \text{Pollutant} \rightarrow \text{Byproducts}\]  
(12)

A wide variety of AOP systems, including homogeneous and heterogeneous mechanisms, have been applied to degrade ACT from an aqueous medium. AOP systems that are based on homogeneous mechanisms have many drawbacks detected, such as the difficulty of catalyst recovery and metal ions leaching in the reaction media, which caused secondary water pollution [33]. As revealed, heterogeneous catalytic systems such as AOP based on micro and nanomaterials have shown promising results to remove ACT from water [34]. Many semiconductors, such as WO_3, ZnO, SnO_2, TiO_2, CeO_2, Bi_2O_3, and C_3N_4, and metals, such as Fe_2O_3, FeO_2, Al_2O_3, zero-valent aluminum, and Cu_2O, have been used as heterogeneous catalysts. Also, the doping between semiconductors and metals has attracted the researcher’s attention according to their advantages, such as reduction in the bandgap of the semiconductors, increase in the bandwidth of absorbance light of the semiconductor surface, and reusability of the composites many times after regeneration.

3 The degradation mechanism of ACT by nanoparticles

The oxidation of ACT based on semiconductors and metal nanomaterials has been gaining the attention because of its high degradation performance, low toxicity, low cost, and ability to function under different conditions. A wide range of activators have been used to catalyze the nanoparticles, such as ultrasound, irradiation (UV and visible light), plasma, and oxidants. The oxidation mechanisms for ACT degradation were varied. Most of the studies observed that the main degradation mechanism was based on the formation of superoxide radicals.

3.1 Mechanism by nanoparticles catalyzed by UV and visible light

Semiconductor nanoparticles capable to generate a hole (h^+) and electrons (e^-) after illuminated with UV or visible light make it the most promising oxidation process as electrons act as reduction agents, whereas holes act as oxidation sites [2]. The mechanism of semiconductors that catalyzed by irradiation, based on photoexcited of the electrons that exist on the catalyst surface, leading to the movement of electrons (e^-) from valance band to conduction band leaving positives holes (h^+). Both electrons and holes can start the redox reactions and oxidize ACT. The mechanism of nanoparticles catalyzed by UV and visible light is depicted in Figure 2.

The equations (13)–(18) represent the possible chemical reactions of TiO_2 catalyzed by UV or visible light system [35].

\[\text{TiO}_2 + h^+ \rightarrow e_{cb}^- + h_{vb}^+\]  
(13)

\[H^+ + e^- \rightarrow \text{recombination}\]  
(14)

\[h_{vb}^+ + H_2O \rightarrow OH^- + H^+\]  
(15)

\[h_{vb}^+ + OH^- \rightarrow OH^-\]  
(16)

\[ACT + h^+ \rightarrow \text{direct oxidation}\]  
(17)

\[OH^- + ACT \rightarrow \text{direct oxidation}\]  
(18)

Some disadvantages have been observed related to photodegradation in the presence of semiconductor TiO_2. For example, the high rate of recombination of electrons–holes leading to minimize the degradation of ACT, a wide bandgap of TiO_2, low ionic and electrical conductivity, slow charge transfer rate limits the quantum efficiency of TiO_2 in the photocatalytic reactions, limited adsorption capacity and porosity, and lower efficiency under solar irradiation restricts the application of this system. To decrease the bandgap and enhance the absorption of irradiation of surface catalyst, the researchers have doped semiconductors with transition metals (Co, Fe, Ni, Ag, Au, Cu, Mg, Pt, Zn, Sn, S, and Al) and nonmetals (N and Cl) [17,36–43]. Kohantorabi et al. [38] revealed that ACT was completely degraded after 15 min of reaction, and the mineralization was 63% within 1 h when 1.0% w/w Ag/ZnO@NiFe_2O_4/PMS/UVA was applied. In addition, Yang et al. [44] examined TiO_2 nanoparticles activated by UVC to degrade ACT from a liquid.
medium. In this system, the initial concentration of ACT was 2.0 mM, and the catalyst concentration was 0.4 g/L. The degradation of ACT was 96% after 80 min of reaction at a pH range of 5.1–3.2. Moreover, Montenegro-Ayo et al. [45] applied TiO₂ nanoparticles/UV/0.02 M Na₂SO₄ system to oxidize ACT. Around 72% of ACT was degraded after 300 min. Sun et al. [46] applied 0.5 mM N/S codoped ordered mesoporous carbon to catalyze peroxymonosulfate (PMS) for ACT degradation. Around 50 mg/L of ACT was totally oxidized within 30 min, and the mineralization was 27%. The kinetic reaction of ACT in this system was $2.4 \times 10^{-1}/\text{min}$. Wang et al. [47] examined BiOCl/UV/Persulfate system to oxidize ACT from an aqueous medium. The results showed that 50 µM of ACT was completely degraded within 150 min, and the mineralization was 83% after 180 min at pH = 5.4. The kinetic reaction was $7.13 \times 10^{-4}/\text{min}$. The studies that applied semiconductor as a catalyst and activated by UVA or UVC to degrade ACT from an aqueous medium have summarized in Table 1.

As the catalyzing of nanoparticles by using UV is costly, to avoid that, the researchers intensified on an alternative photocatalytic method such as visible light. As mentioned, TiO₂ powder has a limited absorption capacity of solar light: just around 5% could be absorbed by TiO₂ powder. To enhance the optical absorption of the catalyst, the researchers examined different approaches to improve the TiO₂ performance. Gómez-Avilès et al. [2] used C-modified TiO₂ nanoparticles/solar irradiation system for ACT degradation. In this system, ACT was completely removed within 60 min, and the mineralization was 20.4% after 120 min. Da Silva et al. [40] studied the degradation of ACT by using 2 g/L of 25% MgO doped with TiO₂ catalyzed by the solar light system. About 48.3% of ACT was degraded within 1 h at pH = 7. Furthermore, Aziz et al. [43] applied 5 wt% of TiO₂ doping with SiO₂-ZSM-5/visible light system to remove ACT from an aqueous medium. The results showed around 96% of ACT was oxidized after 180 min, and the mineralization was 77.8%. In addition, 0.1 wt% Cu-doped TiO₂/visible light system was applied by Lin and Yang [48]. The results indicated that 50 mg/L ACT was completely decomposed after 3 h of reaction at pH = 6, and the degradation rate was 0.0243/min. Also, Feng et al. [49] examined oxygen vacancies and phosphorus codoped black titania-coated carbon nanotube composites (OVPTCN) activated by a visible light system to degrade and mineralize ACT from a liquid medium. The results were 96 and 20.4% of degradation and mineralization, respectively. Table 2 includes the studies that investigated the semiconductor particles/visible light systems to remove ACT from an aqueous medium.

### 3.2 Mechanism by nanoparticles catalyzed by oxidants

Many studies investigated the oxidation of ACT by using synthesized particles in the presence of oxidants. For example, Ikhlaq et al. [50] examined the oxidation of ACT by zeolite/O₃ system. They proposed that both ACT and O₃ were adsorbed on the surface of zeolite then react with each other, which supports that the nonradical mechanism was dominant. Figure 3 illustrated the adsorption of oxidant and ACT on the catalyst surface, then the oxidant attacked ACT, which resulted in the degradation of the pollutant via the heterogeneous or nonradical mechanism.

In addition, heterogeneous and homogeneous could happen together. Mashayekh-Salehi et al. [3] applied MgO/O₃ system to oxidize ACT in an aqueous solution. They proposed the following chemical equations (19)–(25) that might happen while the degradation reaction is running.

- Direct oxidation with O₃ molecules on MgO surface:
  \[
  \text{MgO} - \text{O}_3 + \text{ACT} \rightarrow \text{H}_2\text{O}_2 + \text{CO}_2 + \text{Intermediates} \quad (19)
  \]
# Table 1: Summary of studies that investigated semiconductor/UV systems for removal of ACT

| Systems                                      | BET surface area (m²/g) | Particle size (nm) | Pore volume/pore diameter/pore size | pH | Mineralization                  | Removal | Time (min) | [ACT] | Catalyst | Ref. |
|----------------------------------------------|-------------------------|--------------------|-------------------------------------|----|----------------------------------|---------|------------|-------|----------|------|
| TiO₂                                         | *                       | *                  | *                                   | 7.9| 59% within 5 h, 72% within 4 h | 96%     | 300        | 50 ppm | 2 g/L    | [8]  |
| TiO₂ with flow O₂ (100 mL/min)               |                         | *                  | *                                   | 4.3| *                                | 100%    | 240        |       |          |      |
| 25% Mg–SiTi                                    | 382                     | *                  | 0.09 cm³/g, 16.8 nm                 | 5.1–3 | *                                | 60%     | 60         | 20 mg/L| 0.7 g/L  | [40] |
| TiO₂ with O₂ concentration 34 mg/L            | 50                      | 21                 | *                                   |    | 60% after 300 min and 85% within 450 min | 95%     | 80         | 2.0 mM | 0.4 g/L  | [44] |
| BI OCI/UV/PS                                  | 20.11                   | *                  | 0.1005 cm³/g, 19.98 nm              | 5.4| 83% after 180 min                | 100%    | 150        | 50 µM  | 0 g/L    | [47] |
| TiO₂                                         | *                       | *                  | *                                   | 7  | *                                | 72.7%   | 60         | 0.017 mM| 0.3 g/L  | [54] |
| Combination of TiO₂ with graphene oxide TiO₂@rGO | 114.3                   | 20                 | 8.3 nm                              | 5.4| 92% of ACT was mineralized within 50 min | 100%    | 50         | 50 mg/L| 2 g/L    | [59] |
| Combination of TiO₂ with graphene oxide TiO₂@rGO | 139                     |                   | 5.4 nm                              |    |                                   | 87%     | 50         |        |          |      |
| Combination of TiO₂ with graphene oxide TiO₂@rGO | 166.5                   |                   | 2.6 nm                              |    |                                   | 70%     | 50         |        |          |      |
| 15% TiO₂/Fe₂O₃ core–shell nanostructure       | 59.32                   | Nanoparticles      | 0.165 cm³/g, 11.12 nm               | *  | *                                | 64.5%   | 90         | 50 mg/L| 0.1 g/L  | [60] |
| 33% TiO₂/Fe₂O₃ core–shell nanostructure       | 70.78                   |                   | 0.169 cm³/g, 9.56 nm                | *  |                                   | 75%     | 90         |        |          |      |
| 50% TiO₂/Fe₂O₃ core–shell nanostructure       | 125                     |                   | 0.245 cm³/g, 7.813 nm               | *  | With purge 100 mL/min of O₂ TOC 66% | 87.8%   | 90         |        |          |      |
| Anatase TiO₂ hollow sphere fabricated through a template solvothermal route | 164                     | 4.0–5.0 nm         | 0.32 cm³/g                          | *  | *                                | 94%     | 60         | 50 mg/L| 0.1 g/L  | [61] |
| ZnO/PSW-contained sonoreactor                | 12.773                  | *                  | 2.934 cm³/g                         | Natural pH | 40% 240 min          | 98.1%   | 60         | 10 mg/L| 0.7 g/L  | [73] |
| CNT10                                        | 348–421                 | 5–6 nm             | 0.284 cm³/g                         | 7  | 61.2% after 60 min             | 81.6 ± 0.6% | 60         | 10 mg/L| *        | [75] |
| TiO₂–Rutile                                   | *                       | *                  | *                                   | *  | *                                | 70%     | 50 h       | 1 mM   | 1 g/L    | [76] |
| TiO₂–Anatase                                  |                         |                    |                                     |    | More than 50%                  |         |            |        |          |      |
| TiO₂                                         | 51–55                   | 30–50              | *                                   | Natural pH | 11% within 72 min   | 100%    | 72         | 10 mg/L| 200 mg/L | [77] |
| 80% Anatase, 20% rutile TiO₂/UV in wastewater |                         |                    |                                     |    | 21% within 72 min            | 100%    | 24         |        |          |      |

(Continued)
Table 1: Continued

| Systems | BET surface area (m²/g) | Particle size (nm) | Pore volume/pore diameter/pore size | pH | Mineralization | Removal | Time (min) | [ACT] | [Catalyst] | Ref. |
|---------|------------------------|-------------------|-----------------------------------|----|----------------|---------|------------|-------|-----------|------|
| TiO₂ thin films calcinated at 650°C | * | Nanoparticles | 800 nm | * | 100% | 150 | 1 mg/L | * | | [78] |
| TiO₂ modified with electrolysis/ H₂O₂/UV | 115.4 | 9.3 nm | 0.2 µm | 5.5 | 26.20% | 76% | 90 | 0.1 g/L | 0.5 g/L | [79] |
| Cu/iron-pillared clay/UV | 110 | * | 1.76 cm³/g, 3.82 nm | 2.7–3 | 80% after 180 min | 100% | 60 | 100 mg/L | 0.5 g/L | [80] |
| Sonophotocatalytic/MnOₓ/TiO₂ | 132 | 181 µm | 0.14–0.21 cm³/g | * | * | 26% | 180 | 25 ppm | 0.1 g/L | [81] |
| PAA/UVC-LED/Fe(ii) | * | * | * | * | 95% | 30 | 20 mg/L | | | |
| Photo-Fenton process | * | 14 nm | * | 3.5 | * | 85% | * | 20 mg/L | 0.2 g/L | [83] |
| Photo-Fenton | * | * | * | 2.8 | 77% after 75 min | 100% | 15 | 0.62 mM | 0.18 mM | [84] |
| MNPs@C/UV/PMS | 572.6 | * | 0.379 cm³/g | 6 | 63.5% after 40 min | 97.4% | 40 | 20 mg/L | 0.16 g/L | [85] |
| Photo Fenton-like oxidation process | Between 1 and 6 | * | * | Natural pH | * | 86.9% | 60 | 10 mg/L | 0.1 g/L | [86] |
| Zeolite-TiO₂ | 49 | * | * | 5 | * | 44.3% | 180 | 20 mg/L | 1 g/L | [87] |
| Zeolite-ZnO | 98 | * | * | 5 | * | 58.7% | 180 | 20 mg/L | 1 g/L | [87] |
| 3% (w/w) of WO₃/TiO₂/SiO₂ composite under UV-VIS irradiation | 167.30 | * | 0.67 cm³/g | 9 | * | Higher 95% | 240 | 5 mg/L | 1.5 g/L | [88] |
| Combining ZVI reduction with photo-Fenton | * | * | * | 3.5 | * | 75% | 60 | 5 mg/L | * | [89] |
| La-doped BiFeO₃/rGO | 31.79 | 5–6 nm | 7.31 cm³/g | * | * | 98.6% | 30 | 50 ppm | 0.2 g | [90] |

*Means data not available; BET: Brunauer, Emmett and Teller.
| Catalysts | BET surface area (m²/g) | Particle size (nm) | Pore volume/pore diameter/pore size | Removal (%) | Time (min) | pH | [ACT] | [Catalyst] | Mineralization | Ref. |
|------------|--------------------------|-------------------|-----------------------------------|-------------|-----------|-----|-------|---------|----------------|------|
| C-modified TiO₂ calcinated at 400°C | 147 | 11.8 | 0.87 cm³/g | 100 | 60 | * | 5 mg/L | 250 mg/L | * | [2] |
| C-modified TiO₂ calcinated at 500°C | 111 | 17 | 0.9 cm³/g | | | | | | | |
| 25% Mg-SiTi | 382 | * | 0.09 cm³/g, 16.8 nm | 48.3 | 60 | 4.3 | 20 mg/L | 0.7 g/L | * | [40] |
| 4% Ag-g-C₃N₄/O₂ | 6.3 | * | * | * | * | * | 7, 8, and 9 | 10 mg/L | 0.25 g/L | 83% within 120 min | [41] |
| TiO₂-fibrous silica (0.1 M) | 531 | * | 1.215 cm³/g | 88 | 180 | 5 | 10 mg/L | 0.5 g/L | * | [43] |
| TiO₂-fibrous silica (0.2 M) | 525 | 1.102 cm³/g | 96 | 180 | | | | | | |
| TiO₂-fibrous silica (0.3 M) | 633 | 1.309 cm³/g | 67 | 180 | | | | | | |
| 0.1 wt% Cu-doped TiO₂ | 120 | 8.2 | 5.6 nm | 100 | 180 | 6 | 50 mg/L | 4 g/L | * | [48] |
| OVPTCN | 104.71 cm²/g | * | * | 96 | 120 | * | 5 ppm | 0.5 g/L | 20.4% after 120 min | [49] |
| BaTiO₃/TiO₂ (1:1) | | | | | | | | | | |
| BaTiO₃/TiO₂ (1:3) | | | | | | | | | | |
| BaTiO₃/TiO₂ (3:1) | | | | | | | | | | |
| Ag/AgCl at ZIF-8 | 367.4 | * | * | 99 | 90 | 5 | 1 mg/L | 0.5 g/L | * | [57] |
| TiO₂ modified with electrolysis/visible light/H₂O₂ | 115.4 | 9.3 nm | 0.2 µm | 60 | 90 | 5.5 | 0.1 g/L | 0.5 g/L | 20.50% within 90 min | [79] |
| K₂S₂O₈-doped TiO₂ | * | * | * | 100 | 540 | 6.9 | 0.1 mM | 1 g/L | * | [91] |
| Photo-Fenton using FeO₄²⁻ | * | * | * | 100 | 180 | 2.5 | 0.05 mM | 50 mg/L | 58% after 300 min | [92] |
| Photo-Fenton using FeSO₄ | * | * | * | 100 | 120 | | | | 79% after 300 min | [93] |
| β-Bi₂O₃ | * | 70 | * | 93.6 | 180 | * | 10 mg/L | 50 mg | 89.5% after 240 min | [94] |
| Photo-Fenton solar process/synthesized wastewater | * | * | * | 90 | 36 | Neutral pH | 1 ppm | 3 ppm | 26.5% | [95] |
| Novel siligraphene/g-C₃N₄ composites | 173.44 | * | 0.29 cm³/g | 80 | 20 | Less than 6 | 5 mg/L | 0.015 g | * | [95] |
| UIO-66-NH₂ | 904 | * | 0.48 cm³/g | 90 | 6 h | * | 5 mg/L | 250 mg/L | * | [96] |
| Fe₃O₄–TiO₂ nanocomposites by | 84 | * | 0.115 cm³/g | 95.85 | 180 | 11 | 30 mg/L | 1.25 g/L | * | [97] |

*Means data not available.
MgO – ACT + O₃ → H₂O₂ + CO₂ + Intermediates (20)

- Radical type catalytic oxidation on MgO surface:
  
  MgO – S + O₃ → MgO₅⁺ + O²⁻ (21)
  MgO – S⁺ + 2(H₂O) + O₃ → MgO – SΟΗ + 3(OH)⁻ + 2O₂ (22)

  MgO – S(OH)₃⁺ + ACT → H₂O₂ + CO₂ + Intermediates (23)

- Direct oxidation with O₃ molecules in the bulk solution:
  
  O₃ + ACT → H₂O₂ + CO₂ + Intermediates (24)

- Radical type catalytic oxidation in the bulk solution:
  
  OH⁻ + ACT → H₂O₂ + CO₂ + Intermediates (25)

The symbol S in the Eqs. (21)–(23) represents Lewis acid sites on the surface of MgO composites, which were available for reacting with ozone. Hydrogen peroxide (H₂O₂) has been used as an oxidant to promote the degradation of ACT. For example, the doping of bimetallic iron–copper in the presence of SO₂⁻ and H₂O₂ has been applied.

The Eqs. (26)–(31) represent the main chemical equations of generating hydroxyl radicals onto iron and copper oxide surfaces [51]:

≡Cu⁺⁰ + 1/2 H₂O₂ + Hc⁺ → ≡Cu⁺ + H₂O (26)
≡Cu⁺ + H₂O₂ → ≡Cu³⁺ + OH⁻ + OH⁻ (27)
≡Cu²⁺ + H₂O₂ → ≡Cu⁺ + HO₂⁻ + H⁺ (28)
≡Fe²⁺ + H₂O₂ → ≡Fe²⁺ + OH⁻ + OH⁻ (29)
≡Fe³⁺ + H₂O₂ → ≡Fe³⁺ + HO₂⁻ + H⁺ (30)
≡Fe³⁺ + ≡Cu⁺ → ≡Fe³⁺ + ≡Cu²⁺ (31)

The possible reactions in the presence of PS are presented in Eqs. (32)–(36) [52]:

≡Cu⁺ + S₂O₅²⁻ → ≡Cu³⁺ + SO₄⁻ (32)
≡Cu²⁺ + S₂O₅²⁻ → ≡Cu³⁺ + SO₄⁻ (33)
≡Fe²⁺ + S₂O₅²⁻ → ≡Fe³⁺ + SO₄⁻ + SO₄⁻ (34)
≡Fe³⁺ + HSO₅⁻ → ≡Fe³⁺ + SO₄⁻ + H⁺ (35)
≡Fe³⁺ – (OH⁻) + HSO₅⁻ → ≡Fe³⁺ + SO₄⁻ + OH⁻ (36)

The possible reactions Eqs. (37)–(39) in the presence of PMS are as follows [53]:

≡Fe²⁺ + HSO₅⁻ → ≡Fe²⁺ – (OH⁻)OSO₅⁻ + OH⁻ (37)
≡Fe²⁺ – (OH⁻) + HSO₅⁻ → ≡Fe³⁺ + SO₄⁻ + OH⁻ (38)
≡Fe³⁺ – (OH⁻)OSO₅⁻ → ≡Fe³⁺ – (OH⁻) + SO₄⁻ (39)

Furthermore, in the systems that rely on PS and PMS as a catalyst, the reaction may go further to produce hydrogen peroxide (OH⁻) as in the following Eqs. (40)–(44) [5].

SO₄⁻ + H₂O → SO₄⁻ + H⁺ + OH⁻ (40)
S₂O₅²⁻ + OH⁻ → SO₄⁻ + S₂O₅²⁻ (41)
OH⁻ + S₂O₅²⁻ → HSO₄⁻ + SO₄⁻ + 1/2 O₂ (42)
OH⁻ + S₂O₅²⁻ → OH⁻ + S₂O₅²⁻ (43)
OH⁻ + SO₄⁻ → SO₄⁻ + OH⁻ (44)

Table 3 lists the studies that applied metals and semiconductor particles catalyzed by oxidants to remove ACT from the aqueous medium.

### 4 The influence of different parameters on ACT degradation

Many different parameters that influence the degradation of ACT such as temperature, pH, catalyst concentration, pollutant concentration, effects of additives (scavengers...
Table 3: Summary of the studies that investigated the removal of ACT by synthesized particles

| Catalyst                                      | BET surface area (m²/g) | Particle size (nm) | Pore volume/and pore diameter/pore size | Oxidant concentration                  | Removal (%) | Time (min) | Mineralization | [ACT] | [Catalyst] | pH | Ref. |
|----------------------------------------------|-------------------------|-------------------|-----------------------------------------|----------------------------------------|-------------|------------|----------------|-------|------------|----|------|
| Pt-supported nanocomposites of Al₂O₃         | 25.01                   | Nano rod          | 6.98 cm³/g                               | [O₃] = 3 mg/min                         | 100         | 9          | 24% after 60 min | 35 µM | 5 mg/L     | 7  | [17] |
| 1.0% w/w Ag/ZnO0.04@NiFe₂O₄/UVA             | 111.06                  | 14.8              | 0.69 cm³/g, 12.49 nm                     | [PMS] = 1.5 mM                          | 92          | 25         | 65% after 60 min | 15 mg/L | 40 mg/L   | 7  | [39] |
| N/S codoped ordered mesoporous carbon       | 545.6                   | *                 | 2–50 nm                                  | [PMS] = 0.5 mM                          | 100         | 30         | 27%            | 50 mg/L | 100 mg/L   | 3.5–9 | [46] |
| Bismuth oxychloride BiOCl                   | 20.11                   | *                 | 0.1005 cm³/g, 19.98 nm                   | [Na₂S₂O₈] = 1 mM                        | 100         | 150        | More than 85% after 180 min | 50 µM | 0.3 g/L | 5.4 | [47] |
| Zeolite                                      |                         | 91.35             | 4 Å                                      | [O₃] = 0.9 mg/min [H₂O₂] = 15 mM       | 90.68       | 60         | *             | 50 mg/L | 5 g        | 7.12 | [50] |
| Fe₃O₄@SiO₂@Cu yolk–shell nanostructures      |                         | 458.86            | 0.41 cm³/g                               | [PS] = 0.6 g/L [PMS] = 0.2 mM          | 90          | 40         | *             | 100 mg/L | 0.3 g/L    | 6.5  | [51] |
| Fe₂O₃ at Cu₂O                               |                         | *                 | 2–2.5 µm                                 | [PS] = 0.8 g/L                          | 94          | 30         | *             | 50 mg/L | 2 g/L      | Around 7 | [52] |
| Fe₂O₃ magnetic nanoparticles                | 85.2                    | 20                | 0.302 cm³/g, 13.8 nm                     | [PMS] = 0.2 mM                          | 75          | 120        | *             | 10 g/mg/L | 0.8 g/L    | 4.3  | [53] |
| Modified MgO nanoparticles                  | 257.3                   | 23.6              | 0.22 µm                                  | [O₃] = 1.8 mg/min [PMS] = 0.2 mM       | 94          | 30         | *             | 100 mg/L | 2 g/L      | 5.4  | [56] |
| Mn Fe₃O₄                                    |                         | *                 | *                                       | [PMS] = 0.2 mM                          | 100         | 60         | *             | 50 mg/L | 0.2 g/L    | 4.3  | [58] |
| Co Fe₃O₄                                    |                         | *                 | 0.302 cm³/g, 13.8 nm                     | [PMS] = 0.2 mM                          | 90          | 90         | *             | 50 mg/L | *          | 6.5  | [63] |
| Fe₂O₃ at Cu₂O                               |                         | *                 | 15 µm                                    | [H₂O₂] = 5 mM [PDS] = 5 mM             | 100         | 180        | *             | 50 mg/L | *          | 4    | [64] |
| Pyrite                                       |                         | *                 | *                                       | [PDS] = 5 mM                            | 96          | 180        | *             | 50 mg/L | *          | 6.5  | [67] |
| Pyrite                                       |                         | *                 | *                                       | *                                       | 82          | 360        | *             | 50 mg/L | [Fe(III)]  | 6    | [68] |
| Green rust SO₄/Cu(II)                       |                         | *                 | *                                       | *                                       | 28          | 28         | *             | 50 mg/L | [Cu(II)]   | 6.5  | [69] |
| Green rust CO₃/Cu(II)                       |                         | *                 | *                                       | *                                       | 74          | 74         | *             | 50 mg/L | [Cu(II)]   | 6.5  | [70] |
| Green rust Cl/Cu(II)                        |                         | *                 | *                                       | *                                       | 74          | 74         | *             | 50 mg/L | [Cu(II)]   | 6.5  | [70] |
| Zero-valent aluminum                        |                         | *                 | Micro pores                              | [PS] = 2 mM [H₂O₂] = 153 mM/L          | 98.5        | 35         | 41% after 6 h  | 30 mg/L | 2 g/L      | 7    | [65] |
| Nanostructured magnetite (Fe₂O₃) powders MGN1| 39.6                    | 29                | *                                       | [PS] = 2 mM [H₂O₂] = 153 mM/L          | 100         | 300        | *             | 100 mg/L | 6 g/L      | 2.6  | [70] |
| MGN2                                         | 5.84                    | 208               | 34% after 6 h                            | 39% after 6 h                           | 7           | 7          | *             | 6 g/L   | *          | 2.6  | [70] |
| MGM                                          | 48                      | 35                | *                                       | [H₂O₂] = 0.08 mM                        | 98.4        | 30         | *             | 10 mg/L | 0.5 g/L    | 2.0–8.2 | [74] |
| Fe/N–CNT                                     |                         | *                 | *                                       | [H₂O₂] = 0.08 mM                        | 98.4        | 30         | *             | 10 mg/L | 0.5 g/L    | 2.7  | [80] |
| Cu/Fe–PILC                                   | 110                     | *                 | 1.76 cm³/g, 3.82 nm                      | [H₂O₂] = 17 mM                          | 80          | 20–30      | *             | 20 mg/L | 0.3 g/L    | Natural pH | [98] |
| Iron oxide–silica                           | 734                     | *                 | 0.95 cm³/g                               | [H₂O₂] = 17 mM                          | 80          | 20–30      | *             | 20 mg/L | 0.3 g/L    | Natural pH | [98] |

(Continued)
| Catalyst | BET surface area (m²/g) | Particle size (nm) | Pore volume and pore diameter/pore size | Oxidant concentration | Removal (%) | Time (min) | Mineralization | ACT | Catalyst | pH | Ref. |
|----------|-------------------------|-------------------|----------------------------------------|-----------------------|-------------|------------|---------------|-----|----------|----|------|
| US/Fenton/TiO₂ NT | * | * | * | US1000 kHz | 85.3 | 30 | * | 3 µM | Fe²⁺·H₂O₂ ratio of 20:4 | 3 | [99] |
| Iron ore tailings | 74 | * | 1.2 nm | [H₂O₂] = 6.2 mol/L | 100 | 30 | * | 2 mg/L | 250 mg | 7.6 | [100] |
| Zero-valent aluminum under air-equilibrated acidic conditions | * | 75–150 µm | * | * | 99 | 960 | * | 20 mg/L | 2 g/L | 1.5 | [101] |
| Bicarbonate | * | * | * | [PS] = 5 mM | More than 50 | 8 h | * | 10 µM | 25 mM | 8.3 | [102] |
| Fenton oxidation system | * | * | * | [H₂O₂] = 15 mM | 91.2 | 60 | * | 5 mM | 0.055 mM | 3 | [103] |
| Fenton process system | * | * | * | [H₂O₂] = 13.8 mM | 92 | 30 | 34% after 60 min | 200 ppm | 5 mM | 3 | [104] |
| Aerated Fenton reactor system | * | * | * | [H₂O₂] = 25 mM | 99 | 40 | 14% after 40 min | 5 mM | 0.1 mM | 3 | [105] |
| GAC | 939.38 | * | 0.54 cm³/g | [PS] = 0.21 mM | 100 | 90 | * | 10 mg/L | 1 g/L | 3–7 | [106] |
| CNT | 201.04 | 1.91 cm³/g | * | 100 | 60 | * | 0.1 g/L | 0.1 g/L | 7 | [107] |
| Co-FeOCl | 10.39 | 0.073 cm³/g | * | [H₂O₂] = 0.5 mM | 87.5 | 60 | * | 10 µM | 0.2 g/L | 7 | [108] |
| TCuO50-GO/CWAO | 7.74 | * | * | 52.1% after 60 min | 96.2 | 60 | * | 100 mg/L | 0.5 g/L | 5.5 | [109] |
| Fe-SBA-15(20)Fe₂(SO₄)₃ | 801 | * | * | [H₂O₂] = 25 mg/L | 90.8 | 120 | 64% after 240 min | 20 mg/L | 1,000 mg/L | 4.5 | [110] |
| Fe-SBA-15(20)FeCl₃ | 705 | * | * | 10.5 | | | | | | |
| MNPs/β-CD | 112 | * | 0.138 cm³/g | [KMnO₄] = 21.7 µM | 95.6 | 10 | * | 3.31 µM | MNPs/β-CD | Initial 7 | [111] |
| 0.5-MnCN | * | * | * | [PMS] = 0.8 g/L | 100 | 15 | * | 20 mg/L | 200 mg/L | 6.5 | [112] |
| TiO₂-rGO 5%/plasma | * | * | * | 92 | 18 | * | 20 mg/L | 0.25 g/L | 9.5 | [113] |
| ZVC | * | * | O₃ | 100 | 4 h | * | 50 mg/L | 5 g/L | 3 | [114] |
| CoAl-LDH | 52.9 | * | 0.122 cm³/g | [PMS] = 0.5 mM | 97.5 | 10 | * | 10 mg/L | 0.15 g/L | 6 | [115] |
| CoAl-CLDH-300 | 251.4 | 0.263 cm³/g | 96.9 | 6.0% after 4 h | | | | | | |
| SnO₂ | 81 | 5.5 | O₃ | 1.6 mg/min | 98 | 20 | * | 84% after 30 min | 50 mg/L | 1.3 g/L | 7 | [116] |
| Zn/MCM | 650 | * | 0.643 cm³/g | Stochiometric amount of [H₂O₂] | 100 | 240 | 200 ppm | 5 mg/L | 1 g/L | 3 | [117] |
| Fe/MCM | 640 | 0.738 cm³/g | * | More than 50 | | | | | | |
| Cu/MCM | 950 | 0.727 cm³/g | * | More than 50 | | | | | | |
| Cr/MCM | 780 | 0.694 cm³/g | * | More than 50 | | | | | | |

*Means data not available.
and oxidants), flow $O_2$, and doping ratio have been investigated.

4.1 The influence of ACT concentration

Major studies that have applied AOP systems in the presence of synthesized particles for ACT oxidation pointed out that when the ACT concentration increases, the degradation efficiency decreases. For systems based on oxidants such as hydrogen peroxide, PS, and PMS, high ACT concentrations may adsorb and cover a wide number of the active sites on the catalyst’s surface, consequently, suppressing the production of super oxidant radicals. Also, for the systems that depend on UV or visible lights as a catalyst, a high ACT concentration may accumulate on the catalyst surface and prevent the penetration of the irradiation, which may reduce the photocatalytic efficiency. For example, Yang et al. [44] applied TiO$_2$ catalyzed by UVA and UVC. They carried out different ACT initial concentrations from 2 to 10 mg/L. The removal decreased from 95% to less than 20%. Montenegro-Ayo et al. [45] studied the changes in the degradation rate when the initial concentration of ACT increased from 5 to 50 mg/L. The results showed a decrease in the oxidation rate from $2.05 \times 10^{-4}$ to $2.86 \times 10^{-5}$, respectively. Also, Tan et al. [54] examined TiO$_2$, activated by UV system to eliminate ACT from liquid medium. The oxidation of ACT was declined from 72.7 to 40.2% when the ACT initial concentration increased from 0.017 to 0.067 mM, respectively. Furthermore, Kurniawan et al. [55] applied photocatalytic of BaTiO$_3$/TiO$_2$ composites to remove ACT; 5 and 25 mg/L of initial ACT concentration were implemented, and the results were a decline in the degradation efficiency from 81 to 19%. The same results were observed when Yaghmaeian et al. [56] carried out modified MgO nanoparticles catalyzed by ozone. When the initial concentration of ACT was 10, 50, 100, and 200 mg/L, the removal was 99.5, 99.4, 77, and 45%, respectively. However, Fan et al. [57] observed that when the initial ACT concentrations were 0.5, 1.0, and 1.5 mg/L at Ag/AgCl@ZIF-8/visible light system, the removal efficiency was stable at 99% after 60 min. They reported that when the concentration was between 0.5 and 1.5, the reaction rate reached the fastest at 1.5 mg/L. However, when the initial concentration exceeds 2.0 mg/L, the reaction rates decrease because the permeability of the photon would reduce when the substrate concentration was too high.

4.2 The influence of semiconductor and metal dosages

Most studies agreed that when the catalyst concentration increases to a certain level, it may benefit and increase the degradation reaction. However, if an excessive amount of catalyst adds, that maybe affected adversely on the degradation performance, or at least the degradation performance stays similar. For systems based on photocatalytic, that might interpret because, at high catalyst concentrations, the agglomeration and the shielding effect of the suspended catalyst is due to increased turbidity and impedes the light penetration, which reduces the accessible light to the catalyst surface resulted decreasing in the photocatalytic. For AOP-nanocomposite systems that based on irradiation activation such as hydrogen peroxide, PMS, and PS, high catalyst concentration leading to an increase in the number of activated radicals, which leads to the self-consumption of generated radicals. Below are some studies that investigated different catalyst concentrations. Hassani et al. [39] carried out different concentrations of CoFe$_2$O$_4$/mpg-C$_3$N$_4$ nanoparticles to activate PMS when the catalyst concentration increased from 10 to 40 mg/L, the removal enhanced from 53 to 85% within 25 min. Although the concentration increased from 40 to 60 mg/L, the oxidation was still the same. Zhang et al. [52] implemented iron–copper bimetallic/PS system to degrade ACT and observed improvement in the degradation percentage at catalyst concentrations between 0.1 and 0.3 g/L, the removal increased from 38.6 to 90%. When 0.4 g/L of catalyst was added, there was no significant change in the degradation performance. Thus, it might be because the excessive amount of SO$_4^-$ increases the self-consumption as Eq. (45):

$$\text{SO}_4^- + \text{SO}_4^- \rightarrow \text{S}_2\text{O}_6^{2-}$$

Furthermore, MnFe$_2$O$_4$/PMS and CoFe$_2$O$_4$/PMS systems were applied by Tan et al. [58], and the results were 96.7, 100, 100, and 100% for MnFe$_2$O$_4$/PMS system and 61, 99.5, 100, and 100% for CoFe$_2$O$_4$/PMS at 0.1, 0.2, 0.3, and 0.4 g/L, respectively. Chashme Khavar et al. [59] mentioned the effect of catalyst dosage when the LED/titanium dioxide doped with graphene oxide (TiO$_2$@rGO) system was applied to remove ACT. They implemented TiO$_2$@rGO concentrations from 0.4 to 4 g/L. It observed that when the catalyst concentrations were 0.4, 1, and 2 g/L, the removal was 53, 57, and 81%, respectively. Meanwhile, when the concentration of TiO$_2$@rGO increased from 2 to 4 g/L, there was no improvement in the degradation
of ACT. Moreover, Abdel-Wahab et al. [60] examined the TiO2/Fe2O3/UV system that when the catalyst concentration increased from 0.1 to 2 g/L, the reaction rate was strongly affected, while concentrations between 0.1 and 1.2 g/L, the removal rate increased because the number of active sites and activated radicals increased. Hence, at 2 g/L of TiO2/Fe2O3 was applied, the degradation rate was declined. Also, different K2S2O8-doped TiO2 dosages have been applied. From 0.25 to 0.5 g/L concentration, the oxidation of ACT increased from 90 to 100% after 9 h. When the dosage increases from 0.5 to 1.5 g/L, the removal was kept around 100% and reached the fastest reaction rate at 1 g/L. While at 2 g/L was applied, the degradation dropped from 100 to 97% [61].

4.3 The influence of pH

pH is a significant factor in the AOP based on semiconductors and metal systems. The effect of pH in (AOP/composites) system for the degradation of ACT is widely investigated. It has been observed that ACT has two chemical forms depending on the pH: (i) nonionic form when the pK_a is under 9.4 and (ii) ionic form when pK_a is more than pH 9.4. In addition, pH_zpc and the type of composite were the main variables to define the oxidation performance of ACT [62]. For example, Ziylan-Yavaş and Ince [17] studied the degradation of ACT by using Pt-supported nanocomposites of the Al2O3/O3 system. The results showed that at the base and neutral pH conditions, ACT was eliminated in 7 and 10 min, respectively. The carbon mineralization increased when increasing the pH because the consumption of ozone increased with increasing pH. That might be attributed to the increase in O3 that converted to OH at high pH atmosphere. The impact of pH between 4 and 10 on the oxidation of ACT by using 1.0% w/w Ag/ZnO0.4@NiFe2O4/PMS/UV system was investigated. The best result was obtained at pH between 6 and 7. Because at this pH, the HSO5− ions can be attracted to the positive surface of the catalyst, which improve the oxidation efficiency by the production of OH−/SO5−. At acidic conditions, the activation of PMS was decreased due to the H-bond formation between H^+ and O−O group of HSO5−, which decreased the ACT removal, and both SO4− and OH react with H^+ resulted in reducing in the degradation of ACT. In alkaline conditions, there were some possible reasons responsible for decreasing ACT degradation: (1) converting SO4− radicals to OH− species with relatively lower redox potential by OH− ions, (2) formation of metal hydroxide complexes of CFNPs and subsequently decrease of PMS decomposition reactions, and (3) self-decomposition of PMS to water and sulfate ions. Ling et al. [41] examined the influence of pH on the mineralization of ACT when (solar/4% Ag-g-C3N4/O3) system was implemented. They applied the following pH: 3, 5, 7, 9, and 11. From pH 3 to 7, the degradation of ACT was enhanced because the increase of pH value increases the conversion of O3 to OH−, which degrades more ACT effectively when the pH increased from 7 to 11, the removal of ACT kept constant because at these pH values, the Ag-g-C3N4 and ACT have the same charge, and the repulsion force was dominant. Sun et al. [46] applied pH from 3.5 to 9 on N/S codoped/PMS system. The results showed that when the pH increased, the degradation was improving because the high concentration of OH− caused the decomposition of PMS to produce SO4−. Ikhlaq et al. [50] studied the oxidation of ACT by using a zeolite/O3 system at pH 3, 7, and 10. In this study, the optimum pH for zeolite/O3 was 7.12 because, at this pH value, the ACT and hydroxyl groups on the zeolite surface were protonated. However, at pH 3 and 10, the oxidation of ACT decreased because ACT and zeolite at this pH have the same charge. Zhang et al. [52] studied the pH in the iron−copper bimetallic system activated by PS to remove ACT. The best pH values were between 5 and 7. The strong acid and alkaline conditions were not favorable for this system. Moreover, the effect of pH in Fe3O4 magnetic nanoparticles investigated by Tan et al. [53]. They reported that there were two effects of pH on the experiment: (1) Different PMS fractions would be affected by pH. In acidic conditions, HSO5− predominated, while SO4− predominated in alkaline conditions. (2) The electrostatic point of Fe3O4 had a pH effect on the catalyst surface charge of 7.3. At acidic pH, less PMS could catalyze on the catalyst surface because of the inhabitation effect of pH the H-bond formation between H^+ and O−O group of HSO5 and positively charged catalyst surface. Kurniawan et al. [55] examined a wide range of pH values from 3 to 11 for ACT degradation using BaTiO3/TiO2/UV. From pH 3 to 7, the degradation enhanced from 7 to 95%, and the optimum value for ACT oxidation was 7; meanwhile, in alkaline conditions, the degradation decreased from 95 to 54% because both the catalyst and the ACT molecules had negative charges in alkaline conditions. As a result, the catalyst’s surface was repelled to the negatively charged ACT molecules, leading to a low ACT removal. Yaghmaeian et al. [56] investigated the influence of different pH to oxidize
ACT by using modified MgO nanoparticles catalyzed by ozone. They observed that the consumption of ozone was related to the increasing of pH: when the pH increased from 2 to 8, the consumption of ozone increased from 17 to 41.5%, and from 75 to 90% when the pH increased from 9 to 10. Also, more O\textsubscript{3} consumption means more OH production. OH\textsuperscript{−} has a high oxidation potential, which is more than O\textsubscript{3}. The best pH value in this system was the natural pH solution close to 5.4; at this pH, the ACT molecule was mostly in its molecular form and could better interact with OH. In acidic conditions, there was no O\textsubscript{3} converted to OH enough. In the alkaline conditions, the isoelectric point of modified MgO and ACT were 10.4 and 9.4, respectively, which means the catalyst m-MgO significantly promoted the decomposition of O\textsubscript{3}. In addition, Fan et al. [57] mentioned the influence of pH on the Ag/AgCl@ZIF8 system for oxidation of ACT. In this system, the optimum pH value was 5. The pH values between 7 and 9.4 were not desired because of weaker electrostatic integration between ACT and Ag/AgCl@ZIF8. For pH, more than 9.4 any pH values less than 7 were favorable for degradation ACT in this system. However, they noted that Ag/AgCl@ZIF8 dissolved in strong acid, which decreased the efficiency of this system for ACT removal. The influence of pH for decomposition of ACT in TiO\textsubscript{2}@rGO nanoparticle system was studied by Cheshme Khavar et al. [59]. They revealed that when the pH increased from 4 to 9, the degradation of ACT promoted from 68 to 93%, respectively. Thus, it can be explained that at the natural pH solution, the surface catalyst has a negative charge, the anion species could not adsorb at the catalyst surface, allowing higher functional group interaction of OH\textsuperscript{−}, which resulted in a high amount of OH\textsuperscript{−} converting to OH\textsuperscript{−} and finally enhanced the degradation of ACT. Zhang et al. [63] observed that when the pH increased from pH 3 to 6.5, the degradation efficiency increased from 80 to 91%, respectively. In the acidic atmosphere, the oxidation of Fe\textsuperscript{2+} to Fe\textsuperscript{3+} was slower, according to the increasing of H\textsuperscript{+}, which inhibited the catalyst, resulting in a decrease in the degradation of ACT. When the initial pH value was 11, it has been observed that the pH decreased to be 3. They explained that because PS produced a large amount of H\textsuperscript{+}, the pH was drastically reduced. At pH 11, the OH\textsuperscript{−} molecules combined with Fe\textsuperscript{3+} caused a rapid reduction of Fe\textsuperscript{2+}, resulting in an insufficient coupling effect of Fe\textsuperscript{2+} and CuO, thus limiting the oxidation of ACT. Peng et al. [64] carried out different pH conditions. They revealed that pyrite/PDS could apply for a wide pH range, whereas pyrite/H\textsubscript{2}O\textsubscript{x} in a narrow range. When pH 6 was implemented, the degradation of ACT in the pyrite/PDS system was 50%, whereas 0% when pyrite/H\textsubscript{2}O\textsubscript{x} system was applied. Also, at pH 8, the results were 10 and 0%, respectively. It is believed that pH playing a central role in the determination of radical species of PDS, from pH 2 to 7 SO\textsubscript{4}\textsuperscript{2−}, from 8 to 10 OH\textsuperscript{−}/SO\textsubscript{4}\textsuperscript{2−}, and from 10 to 12 OH\textsuperscript{−} was dominant. Also, Dong et al. [65] applied pH 3, 9, and 12. The results were 98.5, 79.5, and 13.5%, respectively. They observed, when the initial pH was 12, the pH value was decreasing to 3. This, because SO\textsubscript{4}\textsuperscript{2−} would react with OH\textsuperscript{−} and H\textsubscript{2}O resulting in the removal of H\textsuperscript{+} and consumption of OH\textsuperscript{−}. In the base conditions, OH\textsuperscript{−} combined with Fe\textsuperscript{2+} to form oxyhydroxides and leads to precipitation. The absence of OH\textsuperscript{−} leading to insufficient activation of PS, which was due to a decrease in the degradation of ACT. Zhang et al. [66] also mentioned the influence of pH by using a S-doped graphene/Pt/TiO\textsubscript{2} system. When pH value increased from 4, 8, and 10, the removal was 99.9, 95.3, and 95.1%, respectively. In acidic conditions, hydroxyl radical behaved like a weak acid and reacted with hydroxyl ions under neutral and alkaline conditions, which was due to decreasing degradation reaction. The effect of pH on green rust coupled with Cu(II) has been reported by Zhao et al. [67]. They investigated three systems GR\textsubscript{SO4}/Cu(II), GR\textsubscript{CO3}/Cu(II), and GR\textsubscript{Cl}/Cu(II), and it was observed that when pH 6 was applied on GR\textsubscript{SO4}/Cu(II), and GR\textsubscript{Cl}/Cu(II), the pH declined to 4, and 5.4, respectively. However, when pH 6 was applied to GR\textsubscript{CO3}/Cu(II), the pH increased to 6.4 and then decreased to 4.2. This pattern may be due to the buffering effect of CO\textsubscript{3} in this H\textsuperscript{+} system. Hydrolysis due to the rise in the pH, but as more accumulated, the buffering ability was exceeded, and then the pH decreased. When the pH decreased in the efficiency of GR\textsubscript{SO4}/Cu(II) and GR\textsubscript{CO3}/Cu(II) decreased from 100 to 82% and 84 to 28%, respectively.

### 4.4 The influence of scavengers

There are two mechanisms for the AOP: heterogeneous and homogeneous. In the mineral-based catalyst systems, oxidation mainly occurs on the catalyst surface; therefore, the heterogeneous reaction pathway is dominant. The mechanism of AOP is complex because some radical species are generating in parallel or series. To identify the radicals that are responsible for the degradation of ACT, the researchers added some substance called scavengers or quenching agents acting to trap the activated radicals: after adding these scavengers, the oxidation significantly declines, which means the radical trapped is responsible for the oxidation process. Table 4 represents the radicals generated in the AOP-synthesized particle-based system for degradation of ACT. Many scavengers such as isopropanol (IPA), tert butyl alcohol (TBA), methanol, salicylic acid, benzoquinone (BQ), KI (potassium iodide), triethanolamine (TEOA), ammonium oxalate, ethylenediaminetetraacetic acid di-sodium (EDTA-2Na), ethanol (EtOH), sodium oxalate, N\textsubscript{2},
| Systems | Scavenger | Main radical | pH | Remarks | Ref. |
|---------|-----------|--------------|----|---------|------|
| C-modified TiO₂/visible light | 0.06 M IPA for OH 0.06 M BQ for O₂⁻ | OH⁻ and O₂⁻ | Neutral pH | When IPA and BQ were added, the degradation for ACT significantly decreased, which means that OH⁻ and O₂⁻ were responsible of ACT degradation. Meanwhile, there was no effect for KI | [2] |
| Ag/ZnO@NiFe₂O₄/PMS/UVA | Oxalate for h⁺ TBA for OH ETOH for OH⁻ and SO₄⁻⁻ | O₂⁻, SO₄⁻OH⁻, and h⁺ | 7 | All of the scavengers that were added to this system was significantly decreased the degradation of ACT | [38] |
| CoFe₂O₄/mpg-C₃N₄/PMS | TBA for OH ETOH for OH⁻ and SO₄⁻⁻ | SO₄⁻⁻ | 7 | There was a lower impact in the presence of TBA on the degradation of ACT | [39] |
| Solar light/Ag-g-C₃N₄/O₃ | 5 mM/L TBA for OH 5 mM/L TEOA for holes | Both of them were responsible for ACT degradation | 7–9 | When TBA and TEOH were added to the reaction, the oxidation decreased from 98 to 54.8 and 43.7%, respectively | [41] |
| N/S codoped ordered mesoporous carbon/PMS | ETOH for SO₄⁻⁻ p-BQ for OH L-His for O₂⁻ | Singlet oxygen and catalyst surface-bound reactive PMS complexes | 3.5–9 | The effect of ETOH and p-BQ on the degradation of ACT were neglected while L-His significantly inhibited the reaction | [46] |
| BiOCl/UVA | TBA for OH BQ for O₂⁻ | OH⁻ and O₂⁻ | 5.4 | The obvious inhabitation in N₂ purging conditions verifies that HO₂⁻/O₂⁻ originated from the reaction between DO and electron | [47] |
| OVPTCN/visible light | EDTA-2Na for holes Tert-butanol for OH p-BQ for O₂⁻ | OH⁻ was the main active radical, then holes | * | EDTA-2Na and tert-butanol were significantly affected on the ACT degradation, while p-BQ was a negligible effect | [49] |
| Iron–copper/PS | 100 mM TBA for OH⁻ and SO₄⁻⁻ 100 mM methanol for SO₄⁻⁻ | SO₄⁻⁻ and OH⁻ | 6.5 | SO₄⁻⁻ was the main radical responsible for ACT oxidation, while OH⁻ just 20% from the total oxidation | [52] |
| Fe₃O₄ magnetic nanoparticles/PMS | ETOH TBA | Both of SO₄⁻⁻ and OH⁻ | 7 | In the initial phase, only OH⁻ were generated, and then after 5 min or later, SO₄⁻⁻ started to generate | [53] |
| Ag/AGCl@ZIF8/visible light | 1 mM AO for h⁺ 1 mM IPA for OH | O₂⁻ | 7 | BQ inhibited the reaction significantly, which means O₂⁻ was the main active substance responsible for the degradation of ACT | [57] |
| TiO₂@rGO/UV-LED | Methanol | OH⁻ | 5.4 | The oxidation percentage declined from 81 to 34, 26, and 2% after the addition of methanol, tert-butanol, and salicylic acid, respectively | [59] |
| Systems | Scavenger | Main radical | pH | Remarks | Ref. |
|---------|-----------|--------------|----|---------|------|
| Fe²⁺/CuO/PS | TBA for OH Methanol for SO₄⁻ | SO₄⁻ and OH | 6.5 | After adding TBA and methanol, the removal declined to 79 and 89%, respectively | [63] |
| Pyrite/PDS and pyrite/H₂O₂ | TBA for OH Ethanol for SO₄⁻ | OH⁻ was the dominant in pyrite/H₂O₂ and SO₄⁻ was in pyrite/PDS system | 4 | H₂O₂ could not be generated at high pH, whereas PDS could be applied in wide pH ranges | [64] |
| CS-Fe/PS | Methanol OH and SO₄⁻ TBA for OH Nitrobenzene for SO₄⁻ | OH⁻ and SO₄⁻ | 7 | TBA and NB exhibited slight inhabitation on ACT removal | [65] |
| Ferrous ion/copper oxide/O₂ | BQ | OH⁻ | 3 | After BQ was added, the degradation of ACT and the generation of H₂O₂ were completely suppressed | [71] |
| Fe/N–CNT/PS | MeOH and TBA Nonradical pathway via electron transfer | * | 5.5 | Oxidation of ACT by holes was the main oxidation mechanism, then O₂⁻ and OH radicals were generated | [74] |
| ZnO/PSW/US/UV CNT10 | tert-butanol IPA, BQ, and EDTA-2Na | h⁺, O₂⁻, and OH | 7 | The oxidation performance was decreased from 97.4 to 58 and 71.5% after addition of MeOH and TBA, respectively | [75] |
| TiO₂ crystalline/UV | IPA | OH⁻ | * | The removal decreased to 44, 52, and 67% in the presence of IPA, BQ, and AO as OH⁻, O₂⁻, and h⁺ scavenger, respectively | [76] |
| PAA/UVC-LED/Fe(II) | TBA, BQ, and MeOH | OH⁻ | 5 | When IPA was added, the degradation of ACT completely suppressed | [82] |
| MNPs@C/UV/PMS | MeOH and TBA | OH⁻ and SO₄⁻ | 6 | The oxidation performance was decreased from 97.4 to 58 and 71.5% after addition of MeOH and TBA, respectively | [85] |
| B-Bi₂O₃/visible light | IPA for OH Sodium oxalate for h⁺ TEMPOL for O₂⁻ N₂ for DO | h⁺ and O₂⁻ | * | The scavenger tests that none of the following radicals (SO₄⁻ and OH⁻ nor superoxide O₂⁻) was responsible for ACT degradation | [93] |
| Visible light/novel siligraphene/g-C₃N₄ composites | IPA, AO, and BQ | OH⁻, O₂⁻, and h⁺ | Less than 6 | When sodium oxalate was added the degradation was affected significantly. Also, the degradation was a decline after TEMPOL and N₂ were added | [95] |
| Photo Fenton-like oxidation process | p-benzoquinone (p-BQ), 2-propanol, AgNO₃ and triethylamine | O₂⁻ | * | The scavenger tests that none of the following radicals (SO₄⁻ and OH⁻ nor superoxide O₂⁻) was responsible for ACT degradation | [96] |
| US/Fenton/TiO₂ NT process | t-BuOH | OH⁻ | 3 | * | [99] |
| Bicarbonate-activated PS | Methanol, SOD, and FFA | Singlet oxygen (¹O₂) | 8.3 | | [102] |
| Systems               | Scavenger                      | Main radical | pH | Remarks                                                                 | Ref. |
|-----------------------|--------------------------------|--------------|----|-------------------------------------------------------------------------|------|
| GAC/PS                | MeOH and TBA                   | SO$_2^\cdot$OH | 3–7 | Free radicals were not generated in the aqueous phase, whereas they were produced on the surface catalyst | [106] |
| CNT/PS                |                                | Electron transfer process |     | Scavengers were not affected on the ACT degradation                      |      |
| Co-FeOCl/H$_2$O$_2$   | TBA                            | Mainly OH$^\cdot$ and O$_2^\cdot$ was detected | 7   | Methanol acting to trap superoxide radicals O$_2^\cdot$                   | [107] |
| 0.5-MnCN/PMS          | EtOH, TBA, p-BQ, and FFA       | O$_2^\cdot$  | 6.5 | The addition of TBA and EtOH, which were 1,000 times the concentration of ACT, had negligible effects on ACT removal, whereas FFA and p-BQ were significantly inhibited the removal of ACT | [111] |
| CoAl-LDH/PMS          | Electron paramagnetic resonance| $^{1}$O$_2$  | 6   | *                                                                         | [114] |
| SnO$_2$/O$_3$         | Tert-butanol, Salicylic acid, Carbonate, Chloride, Sulfate, Phosphate | OH$^\cdot$  | 7   | *                                                                         | [115] |

*Means data not available.
4-hydroxy-2,2,6,6 tertamethylpiperidinyloxy (TEMPOL), and L-histidine (L-his) have been applied.

4.5 The influence of doping ratio

The impact of the doping ratio in the synthesized particles, such as TiO2@rGO, Ag-g-C3N4, ZSM-5/TiO2, BaTiO3/TiO2, La-doped ZnO, TiO2/Fe2O3, Fe3O4@SiO2, Mg/SiO2, and iron-copper bimetallic doped with silica (Cu/Fe3O4@SiO2), for the ACT removal has been investigated. Most of these studies agreed that when the doping ratio increases, the degradation of ACT increases. But if the doping ratio increased above the threshold, it might negatively impact the degradation performance. For example, Ling et al. [41] applied the Ag-g-C3N4 system to degrade ACT. The excessive amount of Ag might be accumulated on the g-C3N4 nanoparticle surface and cover the active sites, which increase the recombination of photogenerated charges. Lin and Yang [48] examined Cu-doped TiO2 to eliminate ACT, many doping ratios were applied, 0.1, 1, and 10 wt%. The best result was 100% of ACT degradation within 3 h by using 0.1% of Cu. They revealed that when the Cu ratio increased, it may generate isolated CuO aggregates expelled from the Cu–TiO2 framework to the pore channels, which may act as a center of charge recombination and decline the mass transport between the pore channels. Also, Kurniawan et al. [55] prepared BaTiO3/TiO2 composites catalyzed by solar irradiation. Three different ratios of BaTiO3/TiO2 were applied, composite-A (1:3), composite-B (1:1), and composite-C (3:1). After 4 h of reaction, the results were 76, 39, and 26%, respectively. Cheshme Khavar et al. [59] applied TiO2@rGO nanocomposites catalyzed by UVA to oxidize ACT. Different doping ratios starting from 0, 1, 3, 5, 7.5, and 10 wt% of rGO were applied, and the degradation of ACT was 53, 83, 100, 87, 76, and 70%, respectively. These clearly showed that 3 wt% was the best doping ratio in the TiO2@rGO system. If the doping ratio exceeds 3 wt%, the degradation efficiencies begin to decrease because a large amount of ACT adsorbs on the catalyst surfaces, which due to some occupied active sites of TiO2 resulted in decreasing the UV light that reaches to TiO2 surface and reduces the photocatalytic activity. In addition, when the UV transmission decreased by TiO2, it might increase the recombination rate. Thi and Lee [68] observed that the presence of La doped on ZnO nanoparticles enhanced the photocatalytic activity and reduced the bandgap energy when applied 0.5 and 1.0 wt% La-doped ZnO. Meanwhile, too much adding like 1.5 wt% of La may adversely affect the system performance and increase the bandgap energy because, in each semiconductor, there is an energy level named Fermi, which is the highest energy level occupied by electrons in a particular site. In ZnO, the Fermi level is between the conduction band and valance band. When more than 1.0 wt% of La doped onto ZnO, the bandgap was increased because Burstein–Moss effect. These states could push the Fermi level to a higher energy position, and then the Fermi level would lie in the conduction band the process depicted in Figure 4.

Aziz et al. [69] studied TiO2 doped onto fibrous silica ZSM-5 system catalyzed by solar light to oxidize ACT. Different doping ratios starting from 1, 3, and 5 wt%, of TiO2 were carried out, and the results were 65, 90, and 71%, respectively. The results indicated that 3 wt% was the best doping ratio in the ZSM-5/TiO2 system. They explained that TiO2 might agglomerate on the surface of fibrous silica and cover the active sites, which caused low penetration of the visible irradiation. This effect was detected when electron carrier concentration exceeded

Figure 4: (a) Band gap narrowing of 0.5–1.0 wt% La-doped ZnO, and (b) band gap broadening for 1.5 wt% of La-doped ZnO.
the conduction band edge density of the state. However, some studies did not observe any decreases with an increase in the doping ratio. For example, Da Silva et al. [40] applied Mg/SiO₂/UV system. Many Mg concentrations experimented with to oxidize ACT, 1, 2, 10, and 25 wt%. The best result was 60% of ACT was removed within 60 min obtained when 25 wt% Mg concentration was applied. Moreover, Abdel-Wahab et al. [60] applied TiO₂/Fe₂O₃ with different TiO₂ ratios, 15, 33, and 50 wt%. They observed that when the concentration of TiO₂ increased from 15 to 50%, the degradation increased from 52.5 to 98%, respectively. That was because ACT could quickly and efficiently adsorb on the catalyst and thus increase the catalyst activity. Do et al. [51] observed that when the molar ratio of Cu increased from 2.1 to 2.94, the reaction rate begins to slow down because of the dispersion morphology of Cu nanocomposites of the surface of iron doped with silica (Fe₃O₄@SiO₂).

### 4.6 The influence of oxidants dosage

The influence of addition and the concentrations of PMS, PS, oxygen (O₃), ozone (O₃), and hydrogen peroxide (H₂O₂) have been widely investigated. All the studies agreed that the addition of oxidants enhances the degradation and the reaction rates of ACT. However, when an excessive amount of oxidant is added, it may impact adversely on the degradation performance. This attributed to many reasons: (1) the excessive amount of oxidants generate more radicals, these radicals may consume each other as shown in the Eqs. (46–50):

\[
\begin{align*}
\text{OH}^- + \text{OH}^- & \rightarrow \text{H₂O} & (46) \\
\text{SO}_4^- + \text{SO}_4^- & \rightarrow \text{S}_2\text{O}_8^{2-} & (47) \\
\text{SO}_4^- + \text{OH}^- & \rightarrow \text{HSO}_4^- + \frac{1}{2}\text{O}_2 & (48) \\
\text{HSO}_3^- + \text{OH}^- & \rightarrow \text{SO}_4^- + \text{H}_2\text{O} & (49) \\
\text{HSO}_4^- + \text{SO}_4^- & \rightarrow \text{SO}_4^- + \text{SO}_4^- + \text{H}^+ & (50)
\end{align*}
\]

(2) The limitation of active sites on the catalyst surface according to the presence of a high concentration of oxidant, and (3) if the excessive concentration of H₂O₂ was added, the generated hydroxyl radicals may react with H₂O₂ to produce H₂O₃, which contributed less oxidation potential than OH⁻. Several studies investigated the addition of oxidant in AOP based on composites systems. Ziylan-Yavaş and Ince [17] observed that when the ozone flow increased from 3, 6, and 9 mg/min on Pt/Al₂O₃/O₃ system, the oxidation was enhanced, and 9 mg/min contributed the best ozone flow in this system. That may refer to the excessive amount of H₂O₂ produced from the oxidation of ozone, which in turn increases the OH⁻ that attacks ACT resulting in increased ACT degradation. Hassani et al. [39] also examined different PMS dosages in CoFe₂O₄/mg-C₇N₄ catalyzed by PMS to degrade ACT. At 0.5 mM of PMS, the degradation efficiency was 60.9% after 25 min reaction. At 1.5 mM of PMS, the efficiency increased to 92%. They mentioned that higher PMS concentration was not favorable in this system because of the reasons mentioned above. Moreover, Sun et al. [46] examined many concentrations of PMS on N/S codoped ordered mesoporous carbon system. When 0.25, 0.5, and 1.0 mM of PMS were applied, the k values increased from 2.0 ± 0.04 × 10⁻² to 2.4 ± 0.06 × 10⁻¹ and 3.7 ± 0.2 × 10⁻¹, respectively. Also, Wang et al. [47] pointed out the addition of Na₂S₂O₈ and H₂O₂ on the BiOCl/UVA system. When Na₂S₂O₈ was added, the degradation rate and the mineralization were accelerated and enhanced. They attributed the improvement to three reasons: (1) the direct reaction between the photon and PS molecules, which results in generating sulfate radicals, (2) also, PS may react with conduction band electrons yielding the formation of sulfate radicals, and (3) sulfate radicals may generate from the reaction of O₂⁻ with PS. However, when H₂O₂ was added, the total organic carbon (TOC) removal was slowed down because the higher O–O bond energy of H₂O₂ compared with the band in the free PS ions under the natural pH might interpret why PS was easier to activate than H₂O₂. Zhang et al. [52] studied the effect of different PS dosages, from 0.2 to 0.6 g/L, on the Fe₂O₃@Cu₂O system to degrade ACT. They noted that a further increase in PS concentration might be less effective because sulfate radicals consumed each other. Tan et al. [53] studied the influence of adding different PMS concentrations on the degradation rate in Fe₃O₄ magnetic nanoparticles/PMS system. They observed that when the initial concentration of PMS increased from 0.0 to 0.2 mM, the reaction rate was promoted from 0.23 × 10⁻² to 1.22 × 10⁻²/min. However, when the initial concentration of PMS increased from 0.2 to 0.5 mM, the reaction rate slightly decreased from 1.45 × 10⁻² to be 1.13 × 10⁻²/min. The increase of degradation rate in the initial concentration 0.2 mM was attributed to the availability of PMS. At this concentration,
PMS acting as a limiting factor controlling the yield of radicals. Furthermore, Tan et al. [58] applied different PMS dosages on MnFe₂O₄ and CoFe₂O₄ to eliminate ACT. The dosages were 0.05, 0.1, 0.15, and 0.2 g/L, and the removal of ACT when MnFe₂O₄ and CoFe₂O₄ were used were 89, 100, 100, 100% and 55.6, 85.7, 94, 100%, respectively. They noted that when the initial concentration of PMS increased to 0.4 mM, the degradation rate started to decline. In this study, there was no adverse effect observed because the initial concentration of PMS did not reach the threshold level. Dong et al. [65] mentioned that when higher PS concentration applied, PS got in the micropores in the catalyst and would react with SO₄²⁻ and led to PS consumption, thereby causing the undesired inhibiting effect. In addition, Velichkova et al. [70] demonstrated that at 153 mM of H₂O₂ in MGN1, MGN2, and MGM systems, the reaction efficiency started to decrease, according to the reaction of OH⁻ with H₂O₂ as shown in equation (51):

\[
\text{OH}^- + \text{H}_2\text{O}_2 \rightarrow \text{H}_2\text{O}_2^- + \text{HO}_2
\]  

As mentioned above, HO_2^- contributed less oxidation potential than OH⁻, which adversely affect the degradation performance.

### 4.7 The influence of oxygen

The impact of the oxygen for ACT degradation by using AOP based on composites has been investigated. Moctezuma et al. [8] revealed that oxygen has a strong effect on photocatalytic degradation. Bubbling O₂ acted to trap the free electrons to inhabit the recombination of (e⁻/h⁺), which affect positively on the degradation performance. Also, Yang et al. [44] reported that O₂ increased the degradation of ACT more than six times. O₂ could inhibit electron–hole recombination as O₂ consumes conduction band electrons allowing valance band holes too, directly and indirectly as shown in Eqs. (52)–(55).

\[
\text{O}_2^- + \text{H}^+ \rightarrow \text{HO}_2^-
\]  

\[
\text{HO}_2^- + \text{H}^+ + e_{cb} \rightarrow \text{H}_2\text{O}_2
\]  

\[
\text{H}_2\text{O}_2 + e_{cb} \rightarrow \text{OH}^- + \text{OH}^-
\]  

\[
\text{H}_2\text{O}_2 + \text{hv} \rightarrow \text{O}_2^-
\]  

Zhang et al. [52] demonstrated that low O₂ concentration resulted in a lower ACT oxidation rate in the iron–copper/PS system. Also, the dissolved oxygen (DO) in the solution related adversely with N₂ purging, which decreased SO₄²⁻ resulted in the decrease of ACT oxidation. It should be noted that the formation of O₂⁻ through the reduction of O₂ by photoinduced electrons in the conduction band and subsequent inhabitation of (e⁻/h⁺) recombination. Furthermore, O₂⁻ may react and generate more oxidizing species such as OH⁻, H₂O₂, and HO₂. Moreover, a high concentration of oxygen and a small amount of UV 185 nm induce the formation of O₃ followed by the generation of H₂O₂ to produce more OH-. Abdel-Wahab et al. [60] examined two oxygenating systems: (1) static O₂ atmosphere and (2) purging 100 mL/min of O₂. The results showed that after 90 min of reaction, the removal of ACT was 75 and 99%, respectively. The enhancement in the degradation performance may be attributed to the formation of O₂⁻ by photoinduced electrons in the conduction band and subsequent inhabitation of (e⁻/h⁺) recombination. Furthermore, O₂⁻ may react and generate more oxidizing species such as OH⁻, H₂O₂, and HO₂. Zhang et al. [63] investigated the effect of DO in the degradation of ACT by using Fe²⁺/CuO/PS system. The result showed that the degradation decreased from 92 to 70% when the system purged with N₂, which means that oxygen played a major role in this system. As DO decreased, O₂⁻, which functioned between Cu²⁺ and Cu⁺, decreased according to Eqs. (56)–(59). Thus, resulting in a decrease in OH⁻, which was due to a decline in the degradation of ACT.

\[
\text{Cu}^+ + \text{O}_2 \rightarrow \text{Cu}^{2+} + \text{O}_2^-
\]  

\[
\text{Cu}^+ + 2\text{H}^+ + \text{O}_2 \rightarrow \text{Cu}^{2+} + \text{H}_2\text{O}_2
\]  

\[
\text{Cu}^+ + \text{H}_2\text{O}_2 \rightarrow \text{Cu}^{2+} + \text{OH}^- + \text{OH}^-
\]  

\[
\text{Cu}^{2+} + \text{O}_2^- \rightarrow \text{Cu}^{2+} + \text{O}_2
\]  

Zhang et al. [71] pointed out, that DO plays an important role in the oxidation of ACT and for radical’s generation when Fe²⁺/CuO was applied. When Fe²⁺/CuO was added, the concentration of DO decreased from 9.48 to 4.85 mg/L in the first 10 min of reaction and then increased to 7.64 mg/L after 6 h reaction. In addition, they observed that the removal of DO completely inhibits the degradation of ACT.

### 4.8 The influence of temperature

According to the literature, the temperature is directly proportional to the removal of pollutant because the pollutant migrates from the bulk solution to the gas–liquid interface region where temperature and OH⁻ are high. Moreover, it has been proposed that the optimum temperature was based on the characteristic of organic matter and the kinetics of the reaction between OH⁻ and pollutant. Velichkova et al. [70] studied the effect of temperature
Table 5: Summary of the studies that applied degradation cycles to examine the durability of the catalysts

| Catalysts                                      | Running times | Remarks                                                                 | Ref. |
|------------------------------------------------|---------------|-------------------------------------------------------------------------|------|
| Carbon-doped Ti calcinated at 500°C air        | 4             | No considerable loss of activity after four cycles                      | [2]  |
| MgO                                            | 4             | The efficiency dropped from 99.3 to 90%                                 | [3]  |
| 1.0% w/w Ag/ZnO                               | 4             | The degradation was 100, 100, 100, and 95%, respectively. Also, there was a small amount of leaching in Ag, Fe, Ni, and Zn | [38] |
| 0.04@NiFe2O4 mpg C3N4 nanocomposite            | 3             | During the three degradation cycles, the degradation performance was constant, but in the fourth cycle, the performance dropped according to the leaching of iron and cobalt | [39] |
| 4% Ag-g-C3N4/O3                               | 4             | The mineralization efficiency slightly dropped from 83.1 to 79.9%        | [41] |
| 0.2-TFSZ                                       | 5             | After the fifth cycle, the removal declined from 96 to 84%. The decline may attribute to the loss of catalyst mass during the recycling | [43] |
| NS-CMK-3                                       | 5             | This catalyst showed superior stability and the degradation percentage was kept constant at 98% | [46] |
| 0.1 wt% Cu-doped TiO2                         | 10            | The efficiency was still constant even after ten times of repetitions   | [48] |
| OVPTCN                                         | 4             | There was a negligible decrease in ACT removal after four running times. Also, a trace amount of was TiO2 leached | [49] |
| Fe3O4@SiO2-Cu 1.04                            | 6             | This catalyst showed superior stability and the degradation percentage was kept constant at 100% within 60 min after six degradation cycles | [51] |
| Fe3O4 magnetic nanoparticles                   | 4             | The reaction rates decreased after each recycle. The degradation was 74.7, 51.9, 39.7, and 29.5%, respectively | [53] |
| Ag/AgCl at ZIF-8                               | 3             | After three times of reaction, the degradation rate decreases just 5% without any change in the catalyst morphology | [57] |
| MnFe2O4                                        | 3             | After 60 minutes, the degradation of MnFe2O4 Composites was 100% for all cycles. CoFe2O4 removal rates were 100, 85.4, and 67.1%, respectively. | [58] |
| And CoFe2O4                                     |               |                                                                          |      |
| TiO2@rGO                                       | 5             | Five successive recycles with little decrease in degradation. The degradation after fifth recycle was 88% | [59] |
| 33% TiO2/Fe2O3                                 | 5             | After the fourth cycle, there was no change in the catalyst performance. But in the fifth cycle, the degradation declined to 57.5% | [60] |
| Hollow mesoporous TiO2                         | 10            | After ten times, there was no significant deactivation                  | [61] |
| Pyrite                                         | 4             | In both systems, after each run, the catalyst lost around 5% of its weight. The removal efficiency after four cycles was 90% | [64] |
| CS-Fe                                          | 5             | Still above 90% after the fifth cycle                                   | [65] |
| MGN1, MGN2, and MGM                            | 2             | All catalyst demonstrated a good stability                              | [70] |
| Ferrous ion and copper oxide                   | 3             | After three times of recycles, the efficiency dropped according to the leaching of copper and/or accumulation of iron precipitates | [71] |
| ZnO/PSW                                        | 4             | For the first three running times, the oxidation percentage just declined 10%, whereas at the end of the fourth cycle, the efficiency decreased 20% | [73] |
| Fe/N-CNT                                       | 10            | ACT degradation percentage kept at 99.8% after the tenth cycle           | [74] |
| TiO2 thin films calcinated at 650°C            | 1             | After 6 hrs of reaction, there was no significant change in ACT oxidation | [78] |
| Cu/Fe-PILC                                     | 2             | Only 3% of losing in mineralization efficiency                          | [80] |
| MNPs@C                                         | 5             | After five running times, the degradation decreased to 91%, which reflects good catalyst stability | [85] |
| ZnO-Z and TiO2-Z                               | 4             | A slight decline in efficiency has been observed%                       | [87] |

(Continued)
on the degradation of ACT by three types of nanoparticles of iron oxide/H₂O₂ system. They revealed that the increase in the temperature from 30 to 60°C had a beneficial effect for all examined conditions. Also, a higher temperature increases the rate of \( \text{OH}^\cdot \) formation. In contrast, high temperature increases the decomposition of H₂O₂ into O₂ and water, which reduces the removal efficiency of ACT. In addition, Im et al. \cite{72} studied the effect of temperature on the removal of ACT by ultrasound. Two ultrasound waves were applied at 28 and 1,000 kHz, and the best temperature was 25 and 35°C, respectively. They mentioned that beyond the optimum temperature leads to an increase in bubble vapor and then bubble collapse due to the reaction of net energy and free radicals. Tan et al. \cite{53} applied different temperatures from 30 to 70°C in Fe₃O₄ magnetic nanoparticles/H₂O₂ and PMS system. The results showed that temperature affects positively on the elimination of ACT and the kinetic increased from \( 1.5 \times 10^{-2} \) to \( 10 \times 10^{-2}/\text{min} \), for 30 and 70°C, respectively. In addition, Sun et al. \cite{46} reported that when N/S codoped ordered mesoporous carbon was applied for ACT degradation at 25°C, the removal was 100% after 30 min, whereas at 45°C, ACT was completely degraded after 20 min, and the kinetic at 25 and 45°C were \( 2.4 \times 10^{-1} \) and \( 3.5 \times 10^{-1}/\text{min} \), respectively. The activation energy, \( E_a \), of the oxidation system was calculated as 13.8 kJ/mol, which suggested the reaction temperature does not have a significant effect on the oxidation reaction.

### 5 Stability and reusability of the catalysts

One of the main advantages of synthesized particles, which use in AOP systems, is their durability and reusability without any considerable change in the degradation performance. According to the literature, the reduction in the degradation efficiency attributes to the following reasons: (1) metals such as Ag, Zn, Fe, Cu, and Ni could be

| Catalysts | Running times | Remarks | Ref. |
|-----------|---------------|---------|------|
| 3% (w/w) of WO₃/TiO₂/SiO₂ composite | 5 | There was a slight change in the degradation performance after fourth running time | [95] |
| Novel siligraphene/g-C₃N₄ composites | 30 h | There was a slight decrease after five cycles which confirms the good durability of the catalyst composites | [96] |
| UiO-66-NH₂ | 5 | The decomposition of ACT was more than 80% after the fifth cycle | [106] |
| GAC and CNT | 3 | ACT oxidation performance significantly declined from the second run 87.5 to 41.3%, whereas there was no significant change after the third cycle | [107] |
| Co-FeOCl | 3 | After three running times, the degradation efficiency is still more than 95% | [108] |
| TCO50-GO | 5 | After the third cycle the efficiency dropped from 87 to 80% | [109] |
| Fe-SBA-15(2)/FeCl₃ | 3 | After the fifth cycle, the degradation percentage slightly decreased from 94.6 to 84.4% | [110] |
| Fe-SBA-15(20)/FeCl₃ and Fe-SBA-15(20)/Fe₂(SO₄)₃ MNPs/β-CD/KMnO₄ | 5 | After the sixth cycle, the degradation percentage decreased 23% | [112] |
| ZVC | 6 | Between the first and fourth cycles, the degradation efficiency just decreased less than 3% | [114] |
| CoAl-LDH | 4 | After the fourth cycle, the ACT oxidation was not change significantly | [115] |
| SnO₂ | 4 | After the seventh day of reaction, the catalyst was capable to maintain a high degradation performance | [116] |
| Cr/MCM-41 and Fe/MCM-41 | 7 days | The degradation was 98, 81.5, and 76.2%. No observable change in the morphology | [117] |
| Mn₂O₃ | 3 | There was no observable decline in the degradation efficiency after the sixth running time | [118] |

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Table 6: Summary of the main byproducts that detected after ACT degradation

| Systems                                      | Byproducts                                                                 | Ref. |
|----------------------------------------------|-----------------------------------------------------------------------------|------|
| Modified MgO nanoparticles catalyzed by O<sub>3</sub> | Carboxylic acid derivatives were the main byproducts such as malonic, succinic acid, malic acid, formic acid, hydroxy-acetic acid, acetamide, and nitrite | [3]  |
| UV-TiO<sub>2</sub>                            | p-Aminophenol and p-nitrophenol                                              | [8]  |
| Ultrasound/ Pt-supported nanocomposites of Al<sub>2</sub>O<sub>3</sub> | Small carboxylic compounds                                                  | [17] |
| 1.0% w/w Ag/ZnO 0.04@NiFe<sub>2</sub>O<sub>4</sub>/PMS/UVA | Acetic acid, 1,2 dihydroxylndenzene, glycolic acid, α-hydrogen acid, 1,4-benzoquinone, and 3-hydroxypropanic acid | [38] |
| Solar/4% Ag-g-C<sub>3</sub>N<sub>4</sub>/O<sub>3</sub> | 1,4-Hydroquinone, dihydroxylated or trihydroxylated                          | [41] |
| UV-TiO<sub>2</sub>                            | Formic acid, oxamic acid, acetamide, hydroxy-acetic acid, malonic acid, butenedioic acid, succinic acid, malic acid, hydroquinone, acetamide, N-(2,4-dihydroxyphenyl), acetamide, N-(3,4-dihydroxyphenyl) | [44] |
| TiO<sub>2</sub> nanotube/UV | Formic acid, oxamic acid, and oxalic acid                                      | [45] |
| OVPTCN/visible light                        | Hydroquinone, 1,4-benzoquinone, 4-methoxyphenol, 2-hexenoic, and malic acid | [49] |
| Photocatalytic/BaTiO<sub>3</sub>/TiO<sub>2</sub> | Hydroquinone and 1,4-benzoquinone                                            | [55] |
| Ag/AgCl at ZIF-8/visible light              | Salicylaldehyde, acetamide and phenol, lactic acid, succinic acid, malic acid, and maleic acid | [57] |
| Photocatalyst/35% TiO<sub>2</sub>/Fe<sub>2</sub>O<sub>3</sub> | 4-Acetamidoresorcinol, 4-acetamidocatechol, 1,2,4-benzentriol, hydroquinone, acetamide, carboxylic acid derivatives have been observed like tartaric acid, malic acid, maleic acid, succinic acid, malonic acid, oxalic acid, and oxamic acid | [60] |
| Photocatalytic of hollow mesoporous TiO<sub>2</sub> | Oxalic acid and oxamic acid                                                  | [61] |
| Fe<sub>3</sub>O<sub>4</sub>/Cu<sub>2</sub>O/PS | N-(Aminocarboxyl)-acetamide, hydroquinone acetate, and 2-pentanol acetate | [63] |
| PDS/pyrite                                  | Hydroquinone, acetamide, acetic acid, and nitric acid                        | [64] |
| H<sub>2</sub>O<sub>2</sub>/pyrite            | Hydroquinone, acetamide, acetic acid, and nitrate                            | [65] |
| CS-Fe/PS                                    | 1,4-Dihydroxybenzene, p-BQ, and p-hydroxyphenol                             | [68] |
| Photocatalyst of 1% La-doped ZnO             | Hydroquinone, oxamic acid, acetic acid, butyric acid, 2-amino-5-methyl benzoic acid, and benzoic acid | [68] |
| Ferrous ion and copper oxide                | Hydroquinone, ammonium, carboxylic acid derivatives; oxalic acid, formic acid, and acetic acid | [71] |
| Fe/N-CNT/PS                                 | Oxaloacetic acid and 4-nitrophenol                                           | [74] |
| Solar photocatalytic of TiO<sub>2</sub>      | Acetic acid, oxalic acid, maleic acid, propionic acid, pyruvic acid, and formic acid | [77] |
| Photo Fenton process catalyzed by a Cu/Fe-PILC | 1,2-Benzoquinone, 1,4-benzoquinone, oxamic acid, acetamide, and hydroquinone | [80] |
| PAA/UVC-LED/Fe(n)                           | 4-Nitrophenol and hydroquinone                                              | [82] |
| Photocatalyst/β-Bi<sub>2</sub>O<sub>3</sub>   | Hydroquinone, formic acid, succinic acid, and hydroxy-acetic acid            | [93] |
| Photo-Fenton                                | Acetate, oxalate, formate, and propionate                                    | [92] |
| Photo-Fenton solar process/synthesized wastewater | Hydroquinone and monohydroxilated derivative                                | [94] |
| Solar light/Fe<sub>3</sub>O<sub>2</sub>–TiO<sub>2</sub> nanocomposites | Acetic acid, oxamic acid, oxalic acid, butyric acid, acetamide, and hydroquinone | [97] |
| Zero-valent aluminum under air-equilibrated acidic conditions | Hydroquinone and anionic derivatives; nitrate, acetate | [101] |
| Fenton oxidation                            | Hydroquinone, benzaldehydes, benzoic acids, include alcohols, ketones, aldehydes, and carboxylic acids | [103] |
| Fenton process                               | p-BQ, hydroquinone (1,4-dihydroxybenzene), catechol (1,2-dihydroxybenzene), resorcinol (1,3-dihydroxybenzene), and 2,4-dinitrophenol | [104] |
| Aerated Fenton reactor                       | Oxalic acids                                                                | [105] |
| 0.5-MnCN/PMS                                 | Hydroquinone, acetyl, and aminophenol                                        | [111] |
| TiO<sub>2</sub>-GO 5%/plasma                 | Hydroquinone, 4-methoxyphenol, malic acid, 4-heptanol, and 2-hexenoic acid   | [112] |
| ZnO/PSW-contained sono-reactor was irradiated by UVC light | Hydroquinone, oxalic acid, formic acic, acetic acid, and ammonium | [113] |
| SmO<sub>2</sub>/O<sub>3</sub>                | Carboxylic acids                                                            | [115] |
| Fe/MCM-41 and Cr/MCM-41/Catalytic wet peroxide oxidation | Oxalic acid, acetic acid, and formic acid                                   | [116] |
| Direct electron transfer by reactive Mn<sub>2</sub>O<sub>3</sub> | Acetic acid and α-nitrosofenol                                               | [117] |
leaching during the reaction, or/and (2) loss of the catalyst weight during the regeneration, or/and (3) agglomeration during the reaction. Most applied experiments using the following procedure to recover the catalyst: washing the catalyst three times with deionized water and dry them at a temperature between 80 and 100°C for 24 h. Sun et al. [46] used NS-CMK-3 catalyst to remove ACT. After five times of running, the catalyst showed high durability to oxidize ACT from an aqueous medium. Soltani et al. [73] applied ZnO/PSW nanopowder for four times to remove ACT. The authors revealed the first three times of running, the catalyst showed good stability, whereas at the end of the fourth cycle, the efficiency decreased to 20%. Moreover, Pham et al. [74] examined the durability of Fe/N-CNT particles to degrade ACT. Palas recovered and reused the catalyst for ten times. The catalyst showed superior stability even after ten times of degradation cycle, the oxidation efficiency was kept as 99.8%. In the Table 5, some studies have examined the catalyst for many cycles to oxidize ACT.

6 Byproducts formation

As mentioned in the introduction, some ACT byproducts such as 1,4-benzoquinone and N-acetyl-P-benzoquinone have negative effects on human health. Most of the byproducts mainly consist of hydroquinone and carboxylic acid derivatives. Ling et al. [41] applied Ag–G–C3N4/O3 catalyzed by Vis-UV light to oxidize ACT from a liquid medium. After the degradation process, the main byproducts were hydroquinone, di-hydroxyphenyl, and tri-hydroxyphenyl. Moreover, Montenegro-Ayo et al. [45] observed small byproducts such as formic acid, oxamic acid, and oxalic acid after ACT oxidation by using TiO2/UV system. In addition, Zhang et al. [71] detected oxalic acid, hydroquinone, formic acid, acetic acid, and ammonium as byproducts after ACT degradation by using ferrous ion and copper oxide/O2 system. Table 6 lists the studies, which monitored the byproducts of ACT after treatment processes.

7 Conclusions and future prospective

This review article has attempted to cover a wide range of state-of-the-art studies related to the oxidation of ACT by using semiconductor and metal catalysts. Hydroxyl radical was the most dominant superoxide responsible for ACT degradation because hydroxyl radical could be generating in all AOP systems, such as ultrasound systems, photocatalytic systems, and AOP-based oxidants systems, that were investigated for ACT degradation. Also, the stability and reusability of the catalysts have been studied. Most of the semiconductor catalysts have shown good stability, but Fe and N codoped carbon nanotube/PS system have shown superior stability, and the degradation efficiency is still 99.8% after the tenth cycle. pH has played a central role in ACT degradation by control of zero of point charge of the catalyst, pH of ACT, and the formation of radicals. In addition, the influence of catalyst, ACT, and oxidant concentrations has been reported. The increase of catalyst concentration was beneficial, but if the catalyst concentration exceeds the threshold point, it adversely impacted the degradation efficiency. Because of high catalyst concentrations, the agglomeration and the shielding effect of the suspended catalyst are due to increased turbidity and low light penetration, which reduces the accessible light to the catalyst surface resulting in decreased photocatalytic activity. High catalyst concentration leads to an increase in the number of activated radicals, which results in the self-consumption of generated radicals. If excessive amount of ACT was added that is due to, for systems based on oxidants such as hydrogen peroxide, PS, and PMS, high ACT concentrations may adsorb and cover a wide number of the active sites on the catalyst’s surface consequently, suppresses the production of super oxidant radicals. Moreover, for the systems that depend on UV or visible lights as a catalyst, a high ACT concentration may accumulate on the catalyst surface and prevent the penetration of the irradiation, which may reduce the photocatalytic efficiency. For oxidants, the excessive amount of oxidants (1) generate more radicals, which may consume each other, (2) the limitation of active sites on the surface catalyst according to the presence of a high concentration of oxidant, and (3) if the excessive concentration of H2O2 was added, the generated hydroxyl radicals might react with H2O2 to produce HO2, which contributed less oxidation potential than OH. It has been noted that the increase in doping ratio was not beneficial because the agglomerate on the surface covers the active sites, which caused low penetration of the irradiation. The studies agreed that DO improved ACT degradation, which attributed to the reaction between O2 and generated radical yielding to the formation of superoxide radicals. A high degree of temperature was not good for ACT degradation because high temperature increases the decomposition of H2O2 into O2 and water, which reduces
the removal efficiency of ACT. Finally, most of the byproducts mainly consist of hydroquinone and carboxylic acid derivatives. AOP systems based on micro and nanoparticles are considered a promising method for ACT degradation. There is a deficiency in the literature about the prediction of the oxidation mechanism of ACT in the presence of nanomaterials. This topic needs further investigation. In addition, the threshold of concentrations and the ratio of oxidants, pollutants, and catalyst need further investigation to expect the optimum ratio between them.

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