The application of thermally activated persulfate for degradation of Acid Blue 92 in aqueous solution

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Abstract

Thermally activated persulfate (TAP) was applied for the degradation of Acid Blue 92 (AB92) dye in its aqueous solution. The effects of pH (3–11), temperature (298–333 K), contact time (15–75 min), sodium persulfate (SPS) concentration (0.05–0.5 mM) and initial AB92 concentration (50–400 mg/L) on the degradation of AB92 using TAP were examined. The initial and residual AB92 concentrations were determined spectrophotometrically at the wavelength of 260 nm and the dye mineralization was examined via the total organic carbon analysis. In addition, the chemical oxygen demand was also measured. The activation energy (E_a) of AB92 degradation was calculated as 17.38 kJ mol⁻¹ based on the Arrhenius equation. Maximum degradation efficiency of 86.47% was reached after 75 min of treatment at a pH of 5, AB92 concentration of 200 mg/L, SPS concentration of 0.5 mM and temperature of 333 K. The degradation efficiency declined with the addition of different sodium chloride concentrations and organic radical scavengers. AB92 degradation was reduced from 86.5 to 74%, 65, and 59.1% using ethylenediaminetetraacetic acid, *tert*-butanol, and ethanol, respectively. A kinetic model was also developed to estimate the pseudo-first-order constants as a function of the main operational parameters (initial dye concentration and TAP concentration). Decolorization rate constants (k) of 0.0009, 0.001, 0.0012, 0.0014, and 0.0018 min⁻¹ were obtained at 303, 308, 313, 328, and 333 K, respectively, using the Langmuir–Hinshelwood kinetic model. The results obtained indicate that the TAP degradation process has great potential for the reduction of azo dyes in aqueous environments.

Keywords Acid Blue 92 · Thermally activated persulfate · Degradation efficiency · Total organic carbon · Chemical oxygen demand

Introduction

The textile industry is considered as a prominent dye production sector [1]. The utilization of various types of colors in addition to chemical substances in dyeing processes generates wastewater with unique characteristics such as pH, color, and composition [2]. The disposal of colored wastewater into the aquatic ecosystem significantly hinders the penetration of light into the deep waters [3, 4]. It may also disturb the process of photosynthesis; this can also lead to the obliteration of aquatic plants [4]. In addition, colored dye effluents are significantly hazardous to the environment even at lower concentrations [5]. Moreover, the majority of

dyes employed by textile industries are of organic origin; they are produced from phthalocyanine, diazo and anthraquinone salts which contain benzene rings that are highly carcinogenic and toxic in nature [6, 7]. An example of such dye is the C.I. Acid Blue 92 (AB92), which is utilized on a regular basis by textile industries. Many researchers have proven that dyes are not completely removed during biological treatment, and they enter into water resources via wastewater effluents originating from treatment plants [8]. These compounds are not eliminated effectively through traditional wastewater removal procedures since they are non-biodegradable.

Several techniques have been employed for the elimination of dyes from polluted waters including coagulation–flocculation [9, 10], chemical treatment [11], oxidation [12, 13], adsorption [14–22] and photocatalytic degradation [23–35]. Adsorption is the most widely used because of its simplicity, low cost and adsorption recovery properties in removing contaminants [36]. Adsorption is usually done

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with an adsorbent such as activated carbon to eliminate dyes, but this process only transfers pollutants from one phase to another [37].

Several advanced techniques in recent studies have been reported for wastewater treatment containing azo dyes [38–40]. Among various implemented methods for dye removal, the advanced oxidation process (AOP) is applied favorably to destroy resistant contaminants. This process fundamentally creates hydroxyl radicals ($\cdot\text{OH}$) and sulfate radicals ($\text{SO}_4^{\cdot-}$) [41]. The sulfate radical contains significant oxidation potential ($E_0=2.6\text{ V}$) in addition to being selective in reactions with organic compounds via electron transfer [42], whereas $\cdot\text{OH}$ is non-selective and reacts with various compounds [43–45]. Activated persulfate is applied extensively for environmental remediation because the produced radicals react easily with the organic compounds by complete or partial mineralization [46]. The radical is typically created from potassium peroxymonosulfate (KHSO_5 , PMS) or sodium persulfate ($\text{Na}_2\text{S}_2\text{O}_8$, SPS) using thermal, transition metal and UV activation methods [47–49]. In addition, they can be applied alone to eliminate pollutants of high concentrations, since they usually possess a strong adsorption capacity [43, 50]. Even though $\text{S}_2\text{O}_8^{2-}$ is a formidable oxidant, it reacts efficiently with pollutants at slow rates that are deemed impractical. This is due to the fact that $\text{S}_2\text{O}_8^{2-}$ may be activated into hydroxyl ($\cdot\text{OH}$) and sulfate ($\text{SO}_4^{\cdot-}$) radicals which are extremely powerful oxidants that react with pollutants within close diffusion-limited rates [51, 52]. The production of a highly reactive sulfate radical is depicted in Eqs. 1–6 [53, 54]:

$\text{SO}_4^{\cdot-}$ is comparably more stable to oxidize organic contaminants, thus providing the possibility for greater dispersion distance and improved mineralization in water [55, 56].

$\text{SO}_4^{\cdot-}$ has been proven to possess great potential for methylparaben degradation via UV-activated persulfate compared to other activators [57]. Cai et al. [58] proved that the bimetallic Fe–Co/GAC catalyst may be utilized to heterogeneously activate SPS oxidation for Acid Orange 7 degradation, which has also been proven in other studies. Also, very toxic persistent organic pollutants (POPs) can be decontaminated with persulfates [59].

The thermally activated persulfate (TAP) process is widely applied in ISCO procedures to treat hazardous and organic contaminants in water. Thermal activation may basically reduce the reaction time and cause higher drops in SPS usage in comparison to other methods [60]. When temperature increases, $\text{S}_2\text{O}_8^{2-}$ is disintegrated into sulfate ions (SO_4^{2-}) as seen in the reaction below [61]:

It is clear that SPS reaction with heat (Eq. 7) causes the generation of $\text{SO}_4^{\cdot-}$ [62].

Thus, in this research, the impact of heat activation on SPS for the elimination of AB92 from its aqueous solution was examined. The impact of different operating parameters such as pH, contact time, sodium persulfate concentration, and initial AB92 concentration on the degradation process was also examined. No prior studies on AB92 dye removal using the TAP process have been observed in the literature. The effect of different radical scavengers including ethylenediaminetetraacetic acid (EDTA), tert-butanol (TBA), and ethanol (EtOH) on the removal of AB92 was examined. The impact of sodium chloride concentrations was also considered.

Materials and methods

Chemicals

AB92 (Acid Blue 92) dye was purchased from Alvan Sabet Corporation, Hamadan, Iran, and used as the pollutant. All reagents [sodium persulfate (SPS) ($\text{Na}_2\text{S}_2\text{O}_8$, 99%, molecular weight: 238.09 g mol^{-1}), sodium hydroxide (NaOH , 98%), and sulfuric acid (H_2SO_4 , 99.99%)] were of analytical grade and purchased from Merck (Germany). The specifications of AB92 are shown in Table 1. All solutions were prepared

Table 1 Physical and chemical properties of AB92

Chemical name	Chemical structure	Molecular formula	Molecular weight (g mol^{-1})
Acid Blue 92		$\text{C}_{26}\text{H}_{16}\text{N}_3\text{Na}_3\text{O}_{10}\text{S}_3$	695.59

using de-ionized water. The pH of the solution was adjusted using HCl or NaOH 0.1 N solutions.

Experimental procedure and analysis

The effects of different parameters such as pH (3, 5, 7, 9, and 11), contact time (10, 20, 40, 60, and 80 min), dye concentration (50, 75, 100, 150, and 200 mg/L), SPS concentration (0.05, 0.07, 0.09, 0.3, and 0.5 mM), and temperature (313, 323, 333, 343 and 353 K) on AB92 degradation were studied.

Initially, the initial AB92 dye concentration, SPS concentration, pH, and temperature were set to be constant at 200 mg/L, 0.07 mM, 3, and 333 K, respectively. The effect of each parameter on the removal percentage of AB92 dye was evaluated. AB92 dye and SPS stock solutions were prepared using double-distilled water. The reactor was filled with 250 mL of the prepared AB92 solution. The initial and residual AB92 dye concentrations were determined using a UV–visible spectrophotometer (Shimadzu Model: CE-1021-UK). The COD concentration was determined using a spectrophotometer (COD VAQIO). The total organic carbon (TOC) was determined through the TOC analyzer (ANATOC Series II). Before adding SPS to the solution, the solution temperature was set using a shaker incubator (Froilabo EC180). To ensure the complete mixing of the mixture during the experiment, the shaker speed was set at 150 rpm.

The results were reported as the average of three replications of each experiment. The apparent activation energy, E_a (J mol⁻¹), required to decompose AB92 at the studied temperatures was calculated using the Arrhenius equation (Eq. 8) [46, 63, 64]:

$$k = A \exp\left(\frac{-E_a}{RT}\right), \quad (8)$$

where A is the frequency factor, R is the ideal gas constant (8.314 J mol⁻¹ K⁻¹), k is the rate constant and T is the absolute temperature (K).

The rate of AB92 decolorization was analyzed using the first-order and Langmuir–Hinshelwood kinetic models. The first-order equation is given as [65]:

$$\ln\left(\frac{C}{C_0}\right) = kt, \quad (9)$$

where C_0 ([AB92]₀) and C ([AB92]) are the concentrations at the initial time, $t=0$ and contact time t , respectively, and k is the first-order rate constant [27, 30, 31].

The Langmuir–Hinshelwood kinetic model is given as [64, 66]:

$$\ln\left(\frac{C_0}{C_t}\right) = kt, \quad (10)$$

where C_0 and C_t are the concentrations of the dye at the initial time ($t=0$) and contact time; the rate constant (k) is obtained from the slope of the straight-line plot of $\ln(C_0/C_t)$ versus t .

Results and discussion

Effect of pH

Solution pH is one of the most important parameters that influence SO₄-based oxidation of organic pollutants in wastewater or natural water. The effect of solution pH (3–11) on the degradation of AB92 was studied at initial dye concentration ([AB92]₀) of 100 mg/L, initial persulfate concentration ([SPS]₀) of 0.5 mM and temperature of 333 K. Highest removal of AB92 was observed at pH of 5 (Fig. 1). COD percentage removals of 55, 65.5, 54.2, 53, and 59.7% were obtained at pH of 3, 5, 7, 9, and 11, respectively, at a reaction time of 15 min. Maximum COD removal was observed at a pH of 5. According to the reactions below (Eqs. 11–13), in an acidic environment, SPS was decomposed to bisulfate (HSO₄⁻) and tetra sulfate oxide (SO₄). Also, SO₄ decomposed to O₂ and H₂SO₅ in a weak and strong acidic environment, respectively.

Fig. 1 Effect of pH on AB92 and COD degradation by TAP. ([AB92]₀:100 mg/L, temperature: 333 K, [SPS]₀: 0.5 Mm)

Therefore, it was predicted that dyes at pH of 3 and 4 were less decomposed than at other conditions because of catalytic–acidic decomposition of persulfate. The more neutral and alkaline the pH of the environment, the more was the reaction of the sulfate radical with $\text{OH}^-/\text{H}_2\text{O}$ resulting in the production of hydroxyl radicals (Eqs. 14 and 15).

Although the hydroxyl radicals showed high oxidation potential ($E_0 = 2.8$ V), they were attracted by non-target species or converted to hydrogen peroxide ($E_0 = 1.8$ V) because of their non-selective reaction [43, 45]. Sulfate radical showed selective reactions too. The difference in reaction mechanisms could differentiate the dye decomposition rate in alkaline and acidic conditions because electron transfer is the main mechanism of sulfate radicals' reaction [67]. Ji et al. [68] showed that the maximum rate of trichloroethylene (atrazine) removal using persulfate occurred at a pH of 5.

Effect of initial dye concentration

The effect of AB92 concentration (50, 100, 200, 300, and 400 mg/L) on the removal efficiency of AB92 was investigated at the reaction time of 15 min and optimum pH of 5. Figure 2 shows that the efficiency of AB92 decomposition was increased with increasing dye concentration. Therefore, AB92 radical reaction with hydroxyl radicals at low AB92 concentration would be increased, which will result in an increase of AB92 decomposition by the free radical [69]. Previous studies also observed that an increase in the concentration of organic compounds accelerated the rate

Fig. 2 The effect of initial AB92 concentration on AB92 and COD degradation using TAP (Temperature: 333 K, [SPS]₀: 0.5 mM, pH 5)

of $\text{S}_2\text{O}_8^{2-}$ loss, presumably due to radical chain reactions that propagated $\text{S}_2\text{O}_8^{2-}$ decomposition [70, 71].

The effect of SPS concentration

Sodium persulfate concentration is regarded as a vital and influential factor for thermally activated SPS oxidation due to its direct impact on the sulfate radical equilibrium. Thus, the impact of SPS concentration on dye removal was investigated. As depicted in Fig. 3, AB92 was eliminated to a high extent by increasing the SPS concentration from 0.05 to 0.5 mM. The sulfate radical showed increased reduction/oxidation properties compared to the persulfate anion and was able to disintegrate a high amount of organic compounds. Radicals' performance was commensurate to the persulfate concentration at a constant temperature. Based on Eqs. 16 and 17, higher concentrations of SPS caused extra sulfate radicals' diffusion, thus acting as a scavenger that was effective in decomposing the pollutants [72]. These sulfate radicals may react with each other to form persulfate ion over again [73].

Furthermore, as depicted in Fig. 3, the average percentage of COD elimination was obtained as 12.21, 33.05, 33.26, 39.5, and 58.2% at initial SPS concentration of 0.05, 0.07, 0.09, 0.3, and 0.5 mM, respectively. The highest COD removal rate was achieved at 0.5 mM. The increased AB92 and COD removal efficiency may be attributed to higher

Fig. 3 The effect of initial SPS concentration on AB92 and COD degradation using TAP ([AB92]₀: 200 mg/L, temperature: 333 K, pH 3)

$\text{SO}_4^{\cdot-}$ production with increasing SPS concentrations. The created sulfate radicals disintegrated the AB92 molecules via hydrogen abstraction effecting the double bonds and electron transfer. In addition, based on Eq. 17, the SPS may react directly with AB92 to generate sulfate radicals [74, 75], which entails the gradual decomposition of AB92. Based on Eqs. 18–28, the generated free radicals can cause a series of chain reactions in radical production and attacks on AB92 molecules resulting in AB92 degradation [76, 77].

Yang et al. [48] proved that when the initial persulfate concentration exceeded a particular point, the azo dye, Acid Orange 7 decomposition rate is marginally slowed down.

Effect of temperature

Temperature plays an important role in the decomposition of pollutants using activated persulfate. The effect of temperature on the degradation of AB92 was studied at temperatures of 303, 308, 313, 328, and 333 K at constant $[\text{AB92}]_0$ of 200 mg/L, $[\text{SPS}]_0$ of 0.5 mM and pH 5. As seen in Fig. 4, AB92 and COD removal efficiency was higher at the temperature of 333 K. The decomposition of persulfate to sulfate radicals could be increased by increasing the temperature (Fig. 4). The efficiency of dye removal increased linearly with increasing temperature. The bond fission of O–O in SPS caused by increased temperature resulted in an increased rate of AB92 decomposition [78, 79]. The high temperature provided enough energy to separate the O–O bond in the SPS leading to the production of the active species such as $\cdot\text{OH}$ and $\text{SO}_4^{\cdot-}$ [80], which led to the pollutants' removal. Although the high temperature was useful in the decomposition of non-organic compounds using active SPS,

Fig. 4 The effect of temperature on AB92 and COD degradation by TAP. ($[\text{AB92}]_0$: 200 mg/L, $[\text{SPS}]_0$: 0.5 mM, pH 5)

Fig. 5 Arrhenius plot for AB92 degradation

the attraction and reduction reactions of SPS would proceed simultaneously. Therefore, the optimal temperature has to be determined [67]. A study by Potakis et al. [81] on the bisphenol oxidation using active persulfate showed that increasing temperature from 313 to 343 K accelerated the removal of bisphenyl at the same rate as that seen in the current study.

The reaction time is another parameter of importance in SPS oxidation with heat [82]. After some time, the production of sulfate radicals increased parallel to the rate of pollutants' removal.

The activation energy ($E_a = -\text{Slope} \times R$) was calculated as $17.38 \text{ kJ mol}^{-1}$ by plotting $\ln k$ against $1/T$ (Fig. 5). In addition, the change in enthalpy (ΔH^0) and change in entropy (ΔS^0) were calculated as $2.091 \text{ kJ mol}^{-1}$ and $-1.011 \text{ kJ mol}^{-1} \text{ K}^{-1}$, respectively, using the Eyring equation [83, 84]:

$$k = \frac{k_B T}{h} e^{-\frac{\Delta H^\ddagger}{RT}} e^{-\frac{\Delta S^\ddagger}{R}}, \quad (29)$$

where k_B is the Boltzmann's constant ($1.381 \times 10^{-23} \text{ J/K}$), h is the Planck's constant ($6.626 \times 10^{-34} \text{ Js}$), R is the universal gas constant ($8.314 \text{ J mol}^{-1} \text{ K}^{-1}$), and T is the absolute temperature (K). $\Delta H^\ddagger$ is the enthalpy of activation (the same as ΔH^0) and $\Delta S^\ddagger$ is the entropy of activation (the same as ΔS^0).

Equation 30 shows the relationship between temperature (T) and free energy (ΔG^0)

$$\Delta G^0 = -2.091 + T \times 1.011. \quad (30)$$

Ghauch and Tuqan [63] proved that bisoprolol decomposition via heated persulfate/ H_2O requires an activation energy of $119.8 \text{ kJ mol}^{-1}$.

Degradation kinetics

The plot of $\ln(C_0/C_t)$ versus contact time, t , at various temperatures is shown in Fig. 6. The rate constant, k , estimated from the slopes of the plots were 0.0009, 0.001, 0.0012, 0.0014, and 0.0018 min^{-1} at temperatures of 303, 308, 313, 328, and 333 K, respectively. As seen in the study, an increase in the temperature of persulfate activation could lead to an increase in the degradation rate constant. The correlation coefficients,

Fig. 6 The degradation kinetics of AB92 using heat-activated persulfate

R^2 , show that the data follow the Langmuir–Hinshelwood equation at all temperatures except 308 K.

TOC removal

To ascertain the TAP process potential for AB92 mineralization, TOC removal was assessed at the initial AB92 concentration of 200 mg/L , SPS concentration of 0.5 mM , pH of 5, and temperature of 333 K . As presented in Fig. 7, AB92 mineralization levels were obtained as 55.38, 65.5, 67, 77.44, and 89.2% at the reaction times of 15, 30, 45, 60, and 75 min, respectively. Even though significant AB92 removal efficiency was acquired through the TAP process, AB92 mineralization may not be complete and may decompose into various products. The results obtained from this study proved that AB92 and its by-products were aptly mineralized via the TAP process. Hence, at a reaction time of 75 min, 89.2% of the TOC was decomposed.

Effect of chloride ion

The impact of chloride on the degradation of AB92 using the TAP process was studied in the range of $50\text{--}500 \text{ mg/L}$ at different reaction times (Fig. 8). From Fig. 8, it can be seen that the degradation of AB92 was better without the addition of chloride ion ($[\text{NaCl}]_0 = 0 \text{ mg/L}$). The removal efficiency was decreased with the addition of sodium chloride and decreased further as the concentration and reaction time were increased. Chloride (Cl^-) is pervasive in environmental systems and often hinders the degradation of contaminants in some cases while expediting degradation in others [70, 85]. Chloride is vital since it is reactive with SO_4^{2-} , with $\text{KSO}_4^{2-} = 2.6 \times 10^8 \text{ M}^{-1} \text{ s}^{-1}$ [86] and is a typical co-contaminant for chlorinated pollutants. In relation to chloride, SO_4^{2-} and Cl^- reaction creates $\text{Cl}^\cdot$ that

Fig. 7 TOC removal during the TAP process. ($[\text{AB92}]_0 = 200 \text{ mg/L}$, temperature = 333 K , pH 5, $[\text{SPS}]_0 = 0.5 \text{ mM}$)

Fig. 8 Effect of NaCl concentration on AB92 degradation. ([AB92]₀ = 200 mg/L, pH 5, [SPS]₀ = 0.5 mM, pH 5)

subsequently intervenes in a series of reactions (Eqs. 31–41) which create either ClO_3^- or $\cdot\text{OH}$ [86–89]:

The $\text{SO}_4^{\cdot-}$ and Cl^- reaction at neutral pH creates $\cdot\text{OH}$. Thus, Cl^- is not regarded as a competing solute, since the result of this reaction is another profound oxidant that can cause a reaction with the contaminants. The Cl^- is regarded as a radical scavenger, that is, a computing solute at $\text{pH} < 5$ since the reaction result was not a secondary radical applicable to oxidizing the contaminants. Furthermore, at this pH, the reaction creates ClO_3^- which is possibly a toxic compound.

Fig. 9 Effect of organic and inorganic scavengers on AB92 degradation. ([AB92]₀ = 200 mg/L, [SPS]₀ = 0.5 mM, pH 5, temperature = 353 K and [scavenger] = 10 mg/L)

Effect of inorganic and organic radical scavengers

The effect of various organic radical scavengers (EDTA, tert-butanol, and ethanol) on AB92 removal with time was examined at a scavenger concentration of 10 mg/L (Fig. 9). As shown in Fig. 9, the degradation efficiency was reduced from 86.5% to 74, 65, and 59.1%, respectively, using ethylenediaminetetraacetic acid (EDTA), tert-butanol (TBA), and ethanol (EtOH). The impact of the organic scavengers (TBA, and EtOH) on AB92 degradation efficiency was lower compared to the inorganic scavenger (EDTA), which implies that EtOH may efficiently hinder the effects of the oxidizing species. Based on Fig. 9, this research showed that the assessed organic species may significantly suppress the studied procedure due to their scavenging impact on reactive radicals' reaction [85, 90]. Based on Eqs. 41 and 44, EtOH reaction rate with $\text{OH}^\cdot$ [$k = (1.2\text{--}2.8) \times 10^9 \text{ M}^{-1} \text{ s}^{-1}$] is 50 times greater compared to that of $\text{SO}_4^{\cdot-}$ [$k = (1.6\text{--}7.7) \times 10^7 \text{ M}^{-1} \text{ s}^{-1}$]. Furthermore, TBA is regarded as an influential scavenger for $\text{OH}^\cdot$ [$k = (3.8\text{--}7.6) \times 10^8 \text{ M}^{-1} \text{ s}^{-1}$]. Based on Eqs. 42 and 45, it can quickly react with $\text{OH}^\cdot$ [60, 68]

In addition, EDTA is an organic molecule that can react with $\cdot\text{OH}$ and $\text{SO}_4^{\cdot-}$. Therefore, the addition of an excess amount of

Table 2 The summary of the AOP degradation processes applied in the treatment of the AB92 dye

AOP	Optimum conditions	Method of analysis	Removal efficiency	References
Sonocatalytic	Initial dye concentration = 10 mg/L Catalyst dosage (Sm-doped ZnO) = 1 g/L Dopant percentage = 6% Ultrasonic power = 150 W Reaction time = 150 min Frequency = 36 kHz	UV–Vis spectrophotometer (WPA Light wave S2000, England)	DE = 90.10%	[91]
Ozonation	Initial dye concentration = 40 mg/L pH = 10 Time = 20 min	UV–vis spectrophotometer (PG, T80)	DE = 96.30%	[92]
O ₃ /H ₂ O ₂	Initial dye concentration = 40 mg/L H ₂ O ₂ concentration = 6 mg/L pH = 10 Time = 30 min	UV–Vis spectrophotometer (PG, T80)	DE = 97.8%	[92]
O ₃ /activated carbon (AC)	Initial dye concentration = 40 mg/L Carbon concentration = 1 g/L (for DE) Carbon concentration = 2 g/L (for COD) pH = 4 Time = 5 min	UV–Vis spectrophotometer (PG, T80) and Open reflux method (ORM)	DE = 98.2%; COD = 100%	[92]
Laccase-catalyzed decolorization	pH = 8 Laccase activity of 2.5 U/mL Dye concentration = 75 mg/mL	UVD 2950, Labomed, Culver City, USA	DE = 94.1%	[93]
Thermal activation process	Initial dye concentration = 200 mg/L Persulfate concentration = 0.5 mM pH = 5 Time = 75 min Temperature = 333 K	UV–visible spectrophotometer (Shimadzu Model: CE-1021-UK) and COD VAQIO	DE = 86.47%; COD = 74.92%	This study

EDTA should be avoided to prevent the EDTA from competing with the contaminant for the reactive radical species.

Comparison with other AOPs for the degradation of AB92

As presented in Table 2, numerous researchers have assessed the removal of AB92 from aqueous environments using several modern oxidation procedures. Compared to the prior studies, the TAP process is a beneficial method to eliminate AB92. Other advantages of this method include favorable removal efficiency, cost-effectiveness, non-toxicity, and aqueous solubility.

Conclusion

The applicability of thermally activated sodium persulfate (SPS) for the degradation of Acid Blue 92 (AB92) from its aqueous solution was investigated. The effects of pH (3–11), temperature

(298–333 K), contact time (15–75 min), SPS concentration (0.05–0.5 mM), and initial AB92 concentrations (50–400 mg/L) on the degradation of AB92 were examined. The AB92 disintegration kinetics was studied. The degradation of AB92 by SPS was fitted into the Langmuir–Hinshelwood model equation. The effect of various organic radical scavengers (EDTA, tert-butanol, and ethanol) and chloride ion concentration on AB92 removal was examined. Optimum conditions of pH 5, SPS concentration of 0.5 mM, reaction time of 75 min, and temperature of 333 K and initial AB92 concentration of 200 mg/L were obtained for the AB92 removal using the TAP process, which gave a removal efficiency of 86.74%. AB19 degradation efficiency declined with the addition of radical scavengers and chloride ion. The TAP degradation process was employed efficiently for the treatment of azo dyes in aqueous environments.

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Author contribution SA and CAI Conceived and designed the experiments; Analyzed and interpreted the data; Wrote the paper. Contributed reagents, materials, analysis tools or data. SR Performed the experiments; Analyzed and interpreted the data.

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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