Rapid and selective electrochemical transformation of ammonia to N₂ by substoichiometric TiO₂-based electrochemical system†

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In this study, we have developed a continuous-flow electrochemical system towards the rapid and selective conversion of ammonia to N₂, based on a tubular substoichiometric titanium dioxide (Ti₄O₇) anode and a Pd–Cu co-modified Ni foam (Pd–Cu/NF) cathode, both of which are indispensable. Under the action of a suitable anode potential, the Ti₄O₇ anode enables the conversion of Cl⁻ to chloride radicals (Cl'), which could selectively react with ammonia to produce N₂. The anodic byproducts, e.g. NO₃⁻, were further reduced to N₂ at the Pd–Cu/NF cathode. EPR and scavenger experiments confirmed the dominant role of Cl' in ammonia conversion. Complete transformation of 30 mg L⁻¹ ammonia could be obtained over 40 min of continuous operation under optimal conditions. The proposed electrochemical system also exhibits enhanced oxidation kinetics compared to conventional batch systems. This study provides new insights into the rational design of a high-performance electrochemical system to address the challenging issue of ammonia pollution.

1. Introduction

The excessive discharge of ammonia wastewater accelerates the eutrophication of water bodies and causes the serious water pollution we are currently facing. To address this issue, various radical-based catalytic processes have been extensively studied. The hydroxyl radical (OH') and sulfate radical (SO₄') are two types of highly potent radicals that have been widely applied. However, another important but often neglected type of radical, chloride radicals (Cl'), have received only limited attention. These Cl' are selective oxidants (ΔG° = 2.4 V vs. NHE) that react with electron-rich groups like ammonia through single electron oxidation, H-abstraction, and addition to unsaturated C–C bonds. We have previously developed a flow-through electrochemical system enabling the sustainable production of Cl' and further converting ammonia to N₂.

Unfortunately, the poor stability of the SnO₂-modified carbon nanotube anode under elevated anode potentials significantly limits its wide application. In separate study, Zhang et al. employed a coupled Ti/IrO₂–RuO₂ anode and a Ti cathode and also achieved conversion of ammonia into N₂ within a conventional batch reactor. They believed that the produced active chlorine was responsible for the oxidation of ammonia. But the treatment kinetics were far from satisfactory. It is, therefore, highly desirable to develop efficient, environmentally friendly and affordable electrochemical systems for ammonia decontamination.

In recent years, Magnéli phase titanium sub-oxides have been considered as promising electrode materials for electrochemical applications, due to their increased electrical conductivity, chemical stability, strong corrosion resistance and limited production cost. These substoichiometric titanium oxides are commercially referred to as Ebonex®, with the generic formula of TiₓO₇₋₂ₓ₁ where x is an integer between 4 and 10. Magnéli phase TiₓO₇ possesses the highest electrical conductivity (e.g., 1500 S cm⁻¹) as well as comparable or even higher oxidative performance compared with other state-of-the-art electrode materials. These intriguing characteristics indicate that TiₓO₇ may serve as an alternative electrode material for various environmental applications.

Herein, an electrochemical system composed of a TiₓO₇-tubular anode and a Pd–Cu co-modified nickel foam (Pd–Cu/NF) cathode was rationally designed. Both are indispensable. The TiₓO₇ electrode exhibits high stability under high potentials (e.g., 3.0 V vs. Ag/AgCl) and may oxidize Cl⁻ into chlorine...
The surface morphology and crystallinity of the Ti4O7 electrode was examined by a field emission scanning electron microscope (FESEM, Model No.: Hitachi S-4800, Japan) and an X-ray diffractometer (XRD, Model No.: Rigaku D/max-2550/PC), respectively. X-ray photoelectron spectroscopy was obtained from a Thermo Fisher Scientific Escalab 250Xi X-ray Photoelectron Spectrometer (XPS) under high vacuum (1 × 10⁻⁹ torr). The total organic carbon of real water samples was measured with a multi N/C 2100 TOC Analyzer (Analytik Jena AG, Germany). An MS5000 spectrometer (Freiberg Instruments Inc., Germany) was employed to obtain the electron paramagnetic resonance (EPR) spectra using 5,5-dimethyl-1-pyrroline-N-oxide (DMPO) as a radical trap. Ammonia concentration was measured by the Nessler reagent method using a TU-1810 UV-vis spectrophotometer (Yixin Instruments Equipment Co., Ltd., Shenzhen) at a wavelength of 420 nm. The amount of N₂ produced was calculated from the total nitrogen (30 mg L⁻¹) ammonia minus the measured concentration of ammonia and nitrate. NO₃⁻ and NO₂⁻ concentrations were determined with a DionexTM ICS-5000 ion chromatography system (IC, Thermo Fisher, USA). NB concentration was quantified by waters 3000 high performance liquid chromatography system (UHPLC, USA) coupled with a C18 column at 262 nm.

2.3 Fabrication of the Pd–Cu/NF cathode

The fabrication of a Pd–Cu/NF electrode followed a previously reported electrodeposition method. Briefly, the NF support was cleaned by sequential sonication in H₂SO₄, acetone, and DI water to remove the surface oxidized layer before use. The electrolyte solution consisted of 2 mM PdCl₂, 4 mM CuSO₄·5H₂O and 0.1 M HCl. The electrodeposition experiment was conducted in a typical three-electrode system using an NF working electrode, a Pt counter electrode, and a saturated Ag/AgCl electrode reference. The deposition potential was controlled at −1.0 V vs. Ag/AgCl for 30 min using a CHI 760E electrochemical workstation.

2.4 Electrochemical experiments

The ammonia oxidation experiment was performed in flow-by mode (Fig. S1) using a TiO₂ tube and a Pd–Cu/NF or a carbon rod cathode. The cathode was arranged at the center of the TiO₂ tube and sealed with silicone rubber to avoid any leakage (Fig. 1). In a typical experiment, 30 mL of 30 mg L⁻¹ (NH₄)₂SO₄ solution with certain NaCl electrolyte was passed through the reactor and returned. The flow rate (2–6 mL min⁻¹) was controlled by an ISM833C-230V peristaltic pump (Cole-Parmer, USA). Besides the flow circulation, no other mixing was provided in the reactor. The cell potential (1.5–3.2 V vs. Ag/AgCl) was provided by a CHI 760E electrochemical workstation. Cyclic voltammetry (CV) experiments were performed using a three-electrode system with a TiO₂ working electrode, a saturated Ag/AgCl electrode reference electrode and a carbon rod counter electrode. The scan rate was set at 20 mV s⁻¹ using 0.12 M NaCl or 0.12 M Na₂SO₄ electrolyte.

3. Results and discussion

The surface morphology of the TiO₂ tubular electrode was firstly characterized by FESEM. The results showed that the internal surface of the TiO₂ tubular electrode presented a typical interconnected porous structure (Fig. S2a).
average pore size was determined to be 2.9 ± 0.62 μm (Fig. S2b†). The macropores provide sufficient electroactive sites and are beneficial for mass transport of target molecules.17 XRD patterns of the tubular electrode agree well with the Magnéli phase Ti₄O₇ standard pattern18 (Fig. S3†). This confirmed that the main component of the anode was Ti₄O₇. The XPS technique clearly demonstrated the presence of characteristic peaks of O 1s, Ti 2p and C 1s, with a super characteristic peaks centred at binding energies of 456.2 eV (2p₃/2) and 461.9 eV (2p₁/2) can be attributed to Ti⁴⁺ and Two Ti 2p characteristic peaks centred at binding energies of 459 eV (2p₃/2) and 464.6 eV (2p₁/2) can be assigned to Ti³⁺ (Fig. S4b†). The cathodic Pd–Cu/NF exhibited a cross-linked grid structure with certain visible particle agglomerations on the NF surface (Fig. S5†). NF showed three characteristic XRD peaks centred at 44.5°, 51.8°, and 76.4°. A typical Pd peak was observed at 40.1° (Fig. S6f). However, no characteristic diffraction of Cu was observed. This may be due to the low crystallinity of the copper phase and the peaks centred at 42° and 48° of the Cu₃Pd phase being masked by the strong peak of Ni at 44.5°.22 The presence of Pd and Cu elements on NF can also be verified by the XPS technique. The binding energies of the spin–orbit coupling of the metal (Pd⁰) were at 340.5 and 335.2 eV, respectively. Another pair of Pd⁰ signals was observed at 342.2 and 337.0 eV (Fig. S7a†). The Cu 2p spectrum centred at 931.8, 934.4, and 942.7 eV, is attributed to the presence of Cu⁰, Cu⁺, and Cu⁵⁺, respectively (Fig. S7b†).

The anodic potential is an essential factor affecting the ammonia conversion performance. As can be seen from Fig. 2a, only negligible ammonia conversion was observed at an anode potential of 1.5 or 2.0 V vs. Ag/AgCl (<3%). Once the anode potential increased to 2.5 V vs. Ag/AgCl, a decreased concentration of ammonia was observed, indicating that an ammonia oxidation reaction had occurred. This may be explained by the occurrence of mono-electron chloride oxidation (E° = 2.4 V vs. NHE) under these conditions (eqn (1)).23 The as-produced Cl⁻ may selectively react with ammonia to produce NO₃⁻ or N₂. The ammonia removal performance increased with the anode potential, and complete ammonia transformation within 40 min was observed at an applied anode potential of 3.0 V vs. Ag/AgCl. This indicates that the ammonia conversion efficiency could be improved by increasing the applied voltage. Further increasing the anode potential to 3.2 V vs. Ag/AgCl did not further accelerate the ammonia oxidation kinetics, which may be due to the occurrence of other side reactions, like water splitting. This may produce gas bubbles, e.g. O₂, at the anode surface, blocking the surface-active sites. To alleviate such negative consequences, we adopted an upflow configuration of the reactor. An increased flow rate (e.g., 4 mL min⁻¹) also facilitates the rapid sweeping out of bubbles.24,25 Therefore, 3.0 V vs. Ag/AgCl was selected as optimal for subsequent investigations.

The [Cl⁻] was another key parameter impacting the ammonia conversion.26 As can be seen from Fig. 2b, the ammonia conversion rate increased with increasing [Cl⁻] from 0.08 M to 0.12 M. This may be attributed to the production of more chloride radicals with increased [Cl⁻] (eqn (2)). However, further increasing the [Cl⁻] to 0.14 M caused a slight decline in ammonia oxidation efficiency. Excessive [Cl⁻] may cause unnecessary side reactions (eqn (2) and (3)) leading to the consumption of Cl⁻ as well as a decreased ammonia conversion efficiency.

Fig. 2 The effect of different operational factors on ammonia removal: impact of (a) anode potential, (b) [Cl⁻], (c) flow rate, and (d) solution pH.
The flow rate is another important factor. As presented in Fig. 2c, the ammonia conversion efficiency increased with flow rate from 2 to 4 mL min\(^{-1}\). This can be explained by the fact that these ammonia ions have more chances to come into contact with these active sites at an increased flow rate in the recirculated filtration mode. Whereas at a flow rate of 6 mL min\(^{-1}\), the ammonia oxidation efficiency showed a decreasing trend. This may be ascribed to the decreased residence time within the reactor (or insufficient contact with the reaction sites) and only an incomplete ammonia oxidation was achieved. We thus chose an optimal flow rate of 4 mL min\(^{-1}\) for subsequent study, considering the "trade-off" effect between flow rate and residence time of ammonia.

The impact of solution pH on ammonia conversion was further studied under optimized experimental conditions. Interestingly, as shown in Fig. 2d, the best ammonia conversion performance was obtained at neutral pH. For example, over 40 min of continuous filtration reaction, complete ammonia conversion was obtained at pH 7. Whereas only 87%, 90% and 96% were achieved at pH values of 3, 5, and 9, respectively. This indicates that the designed electrochemical system has great prospects for the treatment of environmental water matrices. The pH-dependent equilibrium between NH\(_2\) and NH\(_4^+\) is responsible for the increase in ammonia conversion at neutral pH. According to a previous report, ammonia in aqueous solution can exist in either un-ionized form (NH\(_3\)) or ionized form (NH\(_4^+\)). In these two forms of ammonia, the un-ionized one is more susceptible to oxidation and more abundant under neutral pH conditions. Similar results were previously reported by Zhu et al. who found that NH\(_2\)-N was completely removed in a neutral solution but hardly removed in an acidic solution. Conversely, Ji et al. reported a highly selective method for transforming ammonia to N\(_2\), based on a novel solar-driven photoelectrochemical coupled chlorine radical reaction system, in which the ammonia removal efficiency was enhanced when the solution acidity increased from pH 6 to 2.

The impact of different operation systems on the conversion of ammonia was further investigated. As presented in Fig. S8, compared with a conventional batch reactor, the present continuous-flow system demonstrated obviously enhanced kinetics (-99.9% vs. 62.3%). This could be due to the convection-enhanced mass transport within the proposed electrochemical system, while diffusion alone dominates the mass transport within the batch system. Moreover, the as-produced gas bubbles may easily block the surface-active sites within the batch reactor, further damaging the overall performance.

The Cl\(^-\) can be produced by direct oxidation by Ti\(_4\)O\(_7\) or through the instantaneous adduct of Cl\(^-\) with OH\(^-\). While other side reactions may also consume Cl\(^-\), such as Cl\(^-\) reacting with Cl\(^-\) to form Cl\(_2\)\(^-\) and further reacting to produce HOCl/OCl\(^-\) (eqn (2)-(4)). It is known that OH\(^-\), Cl\(^-\) and HClO are strong oxidants that can react with NH\(_4^+\). In particular, the Cl\(^-\) could react selectively with NH\(_4^+\) (\(k_{\text{Cl}^-} = 5 \times 10^{15} \text{M}^{-1} \text{s}^{-1}\)) with a much higher reaction rate compared with that of OH\(^-\) (\(k_{\text{OH}^-} = 3 \times 10^8 \text{M}^{-1} \text{s}^{-1}\)). These Cl\(^-\) can selectively convert NH\(_4^+\) into N\(_2\), NO\(_3^-\), or NO\(_2^-\) (eqn (5)-(7)). However, the contribution of ammonia conversion from other reactive chlorine species cannot be eliminated, which will be discussed later.

\[
\begin{align*}
\text{Cl}^- + h^+(\text{hole}) & \rightarrow \text{Cl}^- \quad (1) \\
\text{Cl}^- + \text{Cl}^- & \rightarrow \text{Cl}_2^- \quad (2) \\
\text{Cl}_2^- + \text{H}_2\text{O} & \rightarrow \text{HOCl} + \text{Cl}^- + H^+ + e^- \quad (3)
\end{align*}
\]

\[
\text{HClO} \leftrightarrow \text{ClO}^- + \text{H}^+ \quad (4)
\]

\[
\begin{align*}
2\text{NH}_4^+ + 6\text{Cl}^- & \rightarrow \text{N}_2 + 6\text{Cl}^- + 8\text{H}^+ \quad (5) \\
\text{NH}_4^+ + 8\text{Cl}^- + 3\text{H}_2\text{O} & \rightarrow \text{NO}_3^- + 8\text{Cl}^- + 10\text{H}^+ \quad (6) \\
\text{NH}_4^+ + 6\text{Cl}^- + 2\text{H}_2\text{O} & \rightarrow \text{NO}_2^- + 6\text{Cl}^- + 8\text{H}^+ \quad (7)
\end{align*}
\]

The changes in ammonia conversion products over time were investigated under an optimized anode potential of 3.0 V vs. Ag/AgCl, [Cl\(^-\)] of 0.12 M, flow rate of 4 mL min\(^{-1}\) and pH of 7. As shown in Fig. 3a, NO\(_3^-\) and N\(_2\) content increased gradually with time to 12 mg L\(^{-1}\) and 18 mg L\(^{-1}\), respectively, over 40 min when using a carbon rod cathode. NO\(_3^-\) was not detected during this process, possibly due to the strong oxidative ability of Ti\(_4\)O\(_7\). However, the ideal product of the present system is N\(_2\), rather than NO\(_3^-\). Since NO\(_3^-\) is negatively charged, it has a strong electrostatic repulsion from the cathode surface. Moreover, these NO\(_3^-\) ions can also be converted back to NH\(_4^+\) (eqn (8)).

\[
\text{NO}_3^- + 7\text{H}_2\text{O} + 8\text{e}^- \rightarrow \text{NH}_4^+ + 10\text{OH}^- \quad (8)
\]

To address this issue, a Pd–Cu/NF cathode was further employed to replace the carbon rod cathode, NF, with excellent electrical conductivity and a three-dimensional network, is considered to be a promising cathode material. Moreover, Pd and Cu have been proven to be effective promoters for NO\(_3^-\) reduction reaction to N\(_2\). In a recent study, a Pd–Cu/NF cathode demonstrated an evidently enhanced N\(_2\) yield compared with a Pd/NF or a Cu/NF cathode in an SnO\(_2\)-CNT based electrochemical system. Importantly, the as-fabricated Pd–Cu/NF surface is positively charged. Hence, the negatively-charged NO\(_3^-\) can be easily absorbed onto the cathode surface via electrostatic attraction. Therefore, another set of experiments was performed using the Pd–Cu/NF cathode coupled with the Ti\(_4\)O\(_7\) anode. The results showed that ammonia was gradually converted to NO\(_3^-\) and N\(_2\) over time. Compared with the carbon rod cathode, the NO\(_3^-\) removal rate increased by 50% over 40 min of reaction when using the Pd–Cu/NF cathode (Fig. 3b). In addition, Pd and Cu coatings are also essential. The results indicate that similar ammonia removal rates can be obtained with Pd/NF or Cu/NF cathodes. However, the presence of Pd or Cu alone only partially contributed to the conversion of NO\(_3^-\) to N\(_2\) (Fig. S9†). This confirmed the capacitive characteristics of the Pd–Cu/NF electrode, which can rapidly adsorb NO\(_3^-\) onto the cathode via electrostatic attraction. NO\(_3^-\) can be reduced to NO\(_2^-\) after

\[
\begin{align*}
\text{Cl}^- + h^+(\text{hole}) & \rightarrow \text{Cl}^- \quad (1) \\
\text{Cl}^- + \text{Cl}^- & \rightarrow \text{Cl}_2^- \quad (2) \\
\text{Cl}_2^- + \text{H}_2\text{O} & \rightarrow \text{HOCl} + \text{Cl}^- + H^+ + e^- \quad (3)
\end{align*}
\]
adsorption on the Cu site, and then further reduced to N$_2$ on the Pd site (eqn (9)–(12)).

\[
\begin{align*}
\text{NO}_3^- + \text{Cu} &\leftrightarrow \text{Cu} - \text{NO}_3^{\text{ads}} \\
\text{Cu} - \text{NO}_3^{\text{ads}}^- + \text{H}_2\text{O} + 2e^- &\rightarrow \text{NO}_2^- + \text{Cu} + 2\text{OH}^- \\
\text{NO}_2^- + \text{Pd} &\leftrightarrow \text{Pd} - \text{NO}_2^{\text{ads}}^- \\
2\text{Pd} - \text{NO}_2^{\text{ads}}^- + 4\text{H}_2\text{O} + 6e^- &\rightarrow \text{N}_2 + 2\text{Pd} + 8\text{OH}^- 
\end{align*}
\]

(9)  
(10)  
(11)  
(12)

To distinguish the dominant active species on ammonia conversion, we spiked a few scavengers into the ammonia solution before flowing it into the reactor. We chose nitrobenzene (NB) as an OH$^-$ quenching agent with a $k$ of 3.9 $\times$ 10$^9$ M$^{-1}$ s$^{-1}$ and tert-butanol ($t$BuOH) as a joint quenching agent of Cl$^-$ and OH$^-$ with $k$ values of 1.9 $\times$ 10$^9$ and 6 $\times$ 10$^8$ M$^{-1}$ s$^{-1}$, respectively. As shown in Fig. 4a, complete ammonia conversion was observed within 40 min of reaction in the absence of any scavengers. However, the ammonia conversion decreased by 74% in the presence of $t$BuOH within the same period. This suggests that the ammonia removal process indeed involved radicals and that HOCl plays only a minor role. To quantitatively determine the amount of [OH$^-$], we used an indirect method by adding 1 mM NB into the reaction medium due to their high affinity. According to eqn (13), [OH$^-$] is determined to be 1.06 $\times$ 10$^{-11}$ M with a $K_{\text{NB}}$ of 0.04133 min$^{-1}$ (Fig. S10†). While the ammonia conversion decreased by only 17% when NB was added, proving that Cl$^-$ may play a dominant role in the process of ammonia oxidation. It is of note that other radicals such as NO$_3^-$ may also exist in the solution and contribute to ammonia conversion. However, the reaction rate constant of NO$_3^-$ with $t$BuOH ($k = 5.7 \times 10^4$ M$^{-1}$ s$^{-1}$)$^{38}$ was 5 orders of magnitude lower than that of Cl$^-$. It is thus reasonable to believe that the contribution from NO$_3^-$ is negligible.

\[
K_{\text{NB}} = k_{\text{OH},\text{NB}}[\text{OH}^-] 
\]

(13)

The production of Cl$^-$ and OH$^-$ during electrochemical filtration was further determined by the EPR technique. As presented in Fig. 4b, the typical 4-peak spectrum of OH$^-$ with an intensity ratio of 1 : 2 : 2 : 1 was detected with DMPO as a spin trapping agent.$^{39}$ However, an alternative 11-peak spectrum was detected at an increased anode potential of 2.5 or 3.0 V vs. Ag/AgCl. This 11-peak spectrum can be divided into a 4-peak (DMPO-OH$^-$) signal and another 7-peak signal. According to previous studies, these 7-peak signals may be ascribed to DMPO-Cl, which are enhanced with an applied potential up to 3.0 V vs. Ag/AgCl.$^{40}$ (Fig. S11†). Similar results were previously
reported by Li et al.\textsuperscript{14} This confirmed the formation of OH\textsuperscript{−} and Cl\textsuperscript{−} within the present electrochemical flow-by system. In addition, we performed another control experiment using Na\textsubscript{2}SO\textsubscript{4} to replace NaCl. As expected, only negligible ammonia conversion (<3%) was observed, regardless of the anode potentials (Fig. S12†), further indicating the essential role of Cl\textsuperscript{−}. The CV spectrum in 0.12 M of NaCl and Na\textsubscript{2}SO\textsubscript{4} solution also indicates that an oxidation reaction occurred at an anode potential of 2.92 V vs. Ag/AgCl using an NaCl electrolyte, as evidenced by the obvious oxidative peak. This oxidative peak could be due to the ammonia oxidation reaction in the presence of Cl\textsuperscript{−}, while no visible redox peak was observed when using the Na\textsubscript{2}SO\textsubscript{4} electrolyte (Fig. S13†). This indirectly verifies the essential role of Cl\textsuperscript{−} for ammonia transformation in this reaction system.

To examine the potential applications of the proposed system, ammonia conversion with different background solutions was investigated. To do this, 30 mg L\textsuperscript{−1} of (NH\textsubscript{4})\textsubscript{2}SO\textsubscript{4} were added to local tap and lake water samples before flowing into the electrochemical system. A similar conversion trend for ammonia was observed in all cases (Fig. S14†). Complete ammonia conversion could still be obtained in tap water with a background total organic carbon (TOC) of 1.8 mg L\textsuperscript{−1}. In addition, the removal of ammonia was about 97% when challenged with lake water with a background TOC of 93.2 mg L\textsuperscript{−1}. The effect of selected common anions (e.g., carbonate, phosphate and silicate) on ammonia conversion was further investigated under optimal operating conditions (Fig. S15†). The results showed that such inhibition was negligible in the presence of carbonate at varying concentrations. Silicate and phosphate posed a slight inhibitory effect on ammonia conversion. When the concentration increased from 5 to 20 mM, the ammonia conversion efficiency decreased from 100% to 86.5% and 82.3%, respectively. This indicates that the complex composition of natural water and the presence of some competitive ions produces only negligible inhibition of ammonia conversion. This result can be explained by the fact that the Cl\textsuperscript{−} react selectively with other electron-rich groups (e.g., NH\textsubscript{4}\textsuperscript{+}). The stability of Ti\textsubscript{4}O\textsubscript{7} was further evaluated by performing four consecutive cycles of ammonia removal experiments in DI-H\textsubscript{2}O conditions. The results showed an average 98.7 ± 2.2% ammonia removal efficiency, indicating the excellent anodic stability of the Ti\textsubscript{4}O\textsubscript{7} electrode (Fig. S16†). These positive outcomes indicate that the proposed electrochemical system is a potential technology for removing ammonia from various water environments. It should be noted that the toxicity of Cl\textsuperscript{−} is beyond the scope of this work. We would like to leave this key question for further exploration.

4. Conclusions

In summary, a continuous-flow electrochemical system was developed towards effective and exhaustive ammonia removal. The proposed electrochemical system employs a highly stable tubular Magnéli phase Ti\textsubscript{4}O\textsubscript{7} anode coupled with a Pd–Cu co-modified NF cathode. The as-produced Cl\textsuperscript{−} can react selectively with NH\textsubscript{4}\textsuperscript{+} to produce N\textsubscript{2} as it flows through the electrochemical system. This flow-by electrochemical system also demonstrated enhanced oxidative kinetics compared with a conventional batch system.

Conflicts of interest

There are no conflicts to declare.

Acknowledgements

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References

1 S. Y. Weon, S. I. Lee and B. Koopman, Environ. Technol., 2004, 25, 1211–1219.
2 L. Chmielarz and M. Jablonska, RSC Adv., 2015, 5, 43408–43431.
3 Z. J. Li, X. P. Zhang, H. F. Dong, X. C. Zhang, H. S. Gao, S. J. Zhang, J. W. Li and C. M. Wang, RSC Adv., 2015, 5, 81362–81370.
4 W. J. Tang, Y. Zhang, J. Bai, J. H. Li, J. C. Wang, L. S. Li, T. S. Zhou, S. Chen, M. Rahim and B. X. Zhou, Sep. Purif. Technol., 2020, 234, 9.
5 J. Lee, H. Park and W. Choi, Environ. Sci. Technol., 2002, 36, 5462–5468.
6 X. R. Zhang, W. G. Li, E. R. Blatchley, X. J. Wang and P. F. Ren, Water Res., 2015, 68, 804–811.
7 Y. Y. Xiang, J. Y. Fang and C. Shang, Water Res., 2016, 90, 301–308.
8 Y. Yang, J. J. Pignatello, J. Ma and W. A. Mitch, Environ. Sci. Technol., 2014, 48, 2344–2351.
9 F. Li, X. Peng, Y. B. Liu, J. C. Mei, L. W. Sun, C. S. Shen, C. Y. Ma, M. H. Huang, Z. W. Wang and W. G. Sand, Chemosphere, 2019, 229, 383–391.
10 C. Y. Zhang, D. He, J. X. Ma and T. D. Waite, Water Res., 2018, 145, 220–230.
11 F. C. Walsh and R. G. A. Wills, Electrochim. Acta, 2010, 55, 6342–6351.
12 S. Nayak and B. P. Chaplin, Electrochim. Acta, 2018, 263, 299–310.
13 C. Trellu, C. Coetsier, J. C. Rouch, R. Esamilre, M. Cretin and C. Causerand, Water Res., 2017, 131, 310–319.
14 F. Li, X. Peng, Y. B. Liu, J. Mei, L. Sun, C. Shen, C. Ma, M. Huang, Z. Wang and W. Sand, Chemosphere, 2019, 229, 383–391.
15 K. Fang, H. Gong, W. Y. He, F. Peng, C. H. He and K. J. Wang, Chem. Eng. J., 2018, 348, 301–309.
16 Y. Zhang, J. Li, J. Bai, Z. Shen, L. Li, L. Xia, S. Chen and B. Zhou, Environ. Sci. Technol., 2018, 52, 1413–1420.
17 L. J. Janssen and L. Koene, Chem. Eng. J., 2002, 85, 137–146.
18 L. Guo, Y. Jing and B. P. Chaplin, Environ. Sci. Technol., 2016, 50, 1428–1436.
19 X. Y. Tao, J. G. Wang, Z. G. Ying, Q. Cai, G. Zheng, Y. Gan, H. Huang, Y. Xia, C. Liang, W. Zhang and Y. Cui, *Nano Lett.*, 2014, 14, 5288–5294.
20 X. X. Li, A. L. Zhu, W. Qu, H. Wang, R. Hui, L. Zhang and J. Zhang, *Electrochim. Acta*, 2010, 55, 5891–5898.
21 Z. Q. Zhang, Y. P. Xu, W. X. Shi, W. Wang, R. J. Zhang, X. Bao, B. Zhang, L. Li and F. Y. Cui, *Chem. Eng. J.*, 2016, 290, 201–208.
22 Y. Yoshinaga, T. Akita, I. Mikami and T. Okuhara, *J. Catal.*, 2002, 207, 37–45.
23 C. D. Vecitis, M. H. Schnoor, M. S. Rahaman, J. D. Schifffman and M. Elimelech, *Environ. Sci. Technol.*, 2011, 45, 3672–3679.
24 P. Gayen, C. Chen, J. T. Abiade and B. P. Chaplin, *Environ. Sci. Technol.*, 2018, 52, 12675–12684.
25 T. X. H. Le, H. Haich, A. D. Shah and B. P. Chaplin, *Environ. Sci. Technol. Lett.*, 2019, 6, 504–510.
26 J. Ding, Q. L. Zhao, J. Q. Jiang, L. Wei, K. Wang, Y. Zhang, W. Hou and H. Yu, *Environ. Sci. Pollut. Res.*, 2017, 24, 5152–5158.
27 K. W. Kim, Y. J. Kim, I. T. Kim, G. I. Park and E. H. Lee, *Electrochim. Acta*, 2005, 50, 4356–4364.
28 Y. B. Liu, P. Wu, F. Q. Liu, F. Li, X. An, J. Liu, Z. Wang, C. Shen and W. Sand, *Environ. Sci. Technol.*, 2019, 53, 1527–1535.
29 S. H. Lin and C. L. Wu, *Water Res.*, 1996, 30, 715–721.
30 X. Zhu, J. Ni and P. Lai, *Water Res.*, 2009, 43, 4347–4355.
31 Y. Z. Ji, J. Bai, J. H. Li, T. Luo, L. Qiao, Q. Zeng and B. Zhou, *Water Res.*, 2017, 125, 512–519.
32 B. P. Chaplin, *Acc. Chem. Res.*, 2019, 52, 596–604.
33 Y. D. Gao, I. M. Alecu, P. C. Hsieh, B. P. Morgan, P. Marshall and L. N. Krasnoperov, *J. Phys. Chem. A*, 2006, 110, 6844–6850.
34 L. Jing, B. Chen, D. Y. Wen, J. S. Zheng and B. Y. Zhang, *Environ. Sci. Pollut. Res.*, 2018, 25, 2691–2701.
35 X. R. Zhang, W. G. Li, E. R. Blatchley III, X. Wang and P. Ren, *Water Res.*, 2015, 68, 804–811.
36 Y. Wang, J. H. Qu and H. J. Liu, *Chin. Chem. Lett.*, 2006, 17, 61–64.
37 M. J. Watts and K. G. Linden, *Water Res.*, 2007, 41, 2871–2878.
38 S. P. Mezyk, T. D. Cullen, K. A. Rickman and B. J. Mincher, *Int. J. Chem. Kinet.*, 2017, 49, 635–642.
39 X. R. Wang, Y. Li, L. P. Tang, W. Gan, W. Zhou, Y. F. Zhao and D. S. Bai, *Chin. Chem. Lett.*, 2017, 28, 394–399.
40 T. Li, Y. Jiang, X. Q. An, H. Liu, C. Hu and J. Qu, *Water Res.*, 2016, 102, 421–427.