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Improve low−temperature selective catalytic reduction of NOx with NH_3 by ozone injection

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Abstract

Low-temperature selective catalytic reduction (SCR) is an important technology for the abatement of the nitrogen oxides (NOx) from stationary sources. The research of fast SCR process has attracted many attentions. In this study, the ozone generated by dielectric barrier discharge oxidized NO to achieve the fast SCR process. We found that the combination of O_3 and SCR can effectively reduce NOx emissions at low temperature (≤ 250 °C). At 250 °C, the removal efficiency of NOx in O_3−NH_3−SCR process (molar ratio of O_3/NO = 0.5) reached 88.3%, while the standard SCR was only 56.8%. Furthermore, the effects of H_2O and SO_2 on NOx removal using ozone assisted SCR system were also explored.

Keywords: Dielectric-barrier discharge, O_3, CeO_2 catalyst, SCR, De−NOx.

1. Introduction

According to the US Environmental Protection Agency (EPA), the sources of nitrogen oxides [NOx (x = 1, 2)] mainly include the combustion of electric and thermal fuel, the discharge of transportation processes and industrial production processes [1]. Usually, more than 90% of the NOx in the flue gas is NO, which can pose serious environmental problems such as acid rain, photochemical smog, ozone depletion, the greenhouse effect, etc. [2]. Selective catalytic reduction (SCR) technology is widely used in coal-fired flue gas for the removal of nitrogen oxides. The typical standard SCR process ($4NO + 4NH_3 + O_2 \rightarrow 6H_2O + 4N_2$) uses NH_3 as a reducing agent to convert NOx to nitrogen and water in combination with the catalyst [3−5]. However, the problems of narrow operating temperature window of catalysts and low catalytic activity below 250 °C have been widely reported [6, 7]. One way to improve efficiency is to make the ratio of NO to NO_2 about 1:1, which is called fast SCR process ($4NH_3 + 2NO_2 + 2NO \rightarrow 4N_2 + 6H_2O$) [8−10]. The fast SCR process can be achieved by converting 50% NO to NO_2 before contacting the catalyst [11, 12]. Therefore, the oxidation of NO to NO_2 is a key factor and rate determining step for entire fast SCR reaction. In recent years, fast SCR has received much attention due to its potential in low temperature SCR process [13]. The research and development of fast SCR technology aim to improve the SCR denitration efficiency at low temperature and reduce the amount of catalysts [14]. At present, pre-oxidation techniques are effective method to realize the fast SCR process, which include non-thermal plasma oxidation [15], electron beam flue gas treatment [16], photocatalytic oxidation absorption [17] and direct oxidation [18−21]. The NOx oxidation method by ozone injection has several advantages compared with other techniques, for examples, the ozone injection method keeps the plasma discharge clean compared to the direct oxidation in an oxygen-containing flue gas using an oxidation reactor, which also has a stronger oxidation selectivity for NO [22, 23]. Despite of these positive points, the low-temperature oxidation technology is still relatively expensive and requires further optimization for the overall process. The main methods for generating ozone include ultraviolet irradiation, electrolytic method, radiochemical method, and dielectric barrier discharge. Dielectric barrier discharge method dissociates oxygen

molecules by high-energy electrons generated in a gas by a high-voltage electric field and polymerizes into ozone molecules by collision. This method has been widely used in industry due to its advantages such as relatively low energy consumption and large ozone output [24].

Vanadium-based catalysts have achieved great success only in the field of high-temperature SCR, and it is more important to develop new catalysts with low temperature and high activity. Some transition metals such as Mn, Ce, Cu, Fe are believed to exhibit higher activity during the selective catalytic reduction of NO with NH_3. Luo *et al*. reported that the 5 wt% γ-MnO_2/TiO_2 supported catalysts can achieve almost 100% NOx conversion at 200−300 °C in SCR experiment [25]. Jesús *et al*. used CuO/NiO monolithic catalysts to achieve a NOx conversion of about 90 % at 230 °C [26]. CeO_2 is a non-toxic and environment friendly transition oxide, due to its superior strong oxygen storage and release property via the redox cycle between Ce^{4+} and Ce^{3+}, which has been widely used for NOx removal [27]. Shan *et al*. reported that Ce−Ti based catalyst showed over 90% NOx conversion at a temperature of 450 °C under a GHSV of 20000 h^{-1} [28]. Gao *et al*. found that Ce–Cu–Ti has the largest specific surface area and pore volume, and presents high activity of NO removal at the temperature range from 200 °C to 400 °C [29]. Li *et al*. prepared a novel Ce−Mo−Ox catalyst, which showed excellent activity and maintained NOx removal efficiency above 90% in the temperature range of 200−350 °C [30].

Although many studies have reported the method of NO pretreatment by ozone oxidation, few studies have reported the effect of ozone injection location on NO oxidation and NOx removal. Furthermore, the effects of H_2O and SO_2 on NOx removal by O_3 assisted SCR flue gas are also rarely reported. In this paper, an experimental study was carried out to improve the efficiency of NOx removal at low temperature by ozone assisted SCR process. The effects of simulated flue temperature, O_3/NO molar ratio, and residence time on NO oxidation efficiency were studied. Moreover, the effects of O_3/NO molar ratio, O_3 injection location, H_2O and SO_2 on the ozone assisted SCR process were also investigated.

2. Experimental system

2.1 Catalyst Preparation

CeO_2 catalyst was prepared by using cerium nitrate ($Ce(NO_3)_3·6H_2O$) as precursor and oxalic acid ($H_2C_2O_2·2H_2O$) as precipitant. A certain amount of cerium nitrate was dissolved in the solution, then the excess oxalic acid was added to the solution under strong agitation and mixed evenly until the solution was completely precipitated. The sample was filtered and dried at 100 °C for 10 h, and successively calcined at 500 °C for 5h. The obtained catalyst was denoted as CeO_2. The CeO_2 powder catalyst was pressed and crushed to 20−40 mesh before the activity tests.

2.2 Experimental setup

The schematic diagram of the experimental device is presented in Fig. 1. It consists of a simulated flue gas mixing unit, a temperature control device, an ozone generator, a SCR reactor and gas analysis devices. The simulated flue gas was from gas cylinders (N_2, 99.99%; O_2, 99.99%; NO, 5%, balanced with N_2; SO_2, 99.99%; NH_3, 1.5%, balanced with Ar). The typical simulated flue gas composition was as follows: [NO] = [NH_3] = 500 ppm, [O_2] = 3 vol.%, [SO_2] = 200 ppm (when used), and N_2 was the carrier gas. The total flow rate of the gas was maintained at 1.0 L·min^{-1}. In order to determine the influences of operating temperature on each process, the mixing units and SCR reactor were placed in the tubular furnace. The simulated flue gas was mixed and heated to the target temperature in the mixing preheater, then the simulated gas was fed into the ozone oxidation system and the SCR reactor. The SCR reactor was made of a quartz tube (40 cm length ×16 mm i.d.) filled with CeO_2 catalyst. The Gas hourly specific velocity (GHSV) was calculated based on the residence time of the gas in the catalyst. The ozone generator structure used in the experiment was a coaxial cylindrical dielectric barrier discharge reactor. A stainless steel rod with a diameter of 4 mm and a length of 370 mm was used as a high-voltage electrode, a ceramic tube with a length of 250 mm and an outer diameter of 10 mm was used as a dielectric. The ceramic tube was provided with a plexiglass tube with an outer diameter of 30 mm and a diameter of 150 mm, and circulating cooling water between them served as a ground electrode. Power supply used in this experiment was high frequency AC power supply (CTP-2000K). Water vapor was introduced by inletting simulated gas through temperature-controlled bubble system containing deionized

water and the water content was varied from 0 vol.% to 6 vol.%. A flue gas analyzer (Testo 350) was used to monitor the concentration of NO and NO_2 in the simulated flue gas online. Ozone was monitored online by an ozone analyzer (2B Technologies), SO_2 and the composition of gas phase were determined by Fourier transform infrared spectroscopy (FTIR, Nicolet 6700), and flue gas humidity was measured using a thermo-hygrometer (HP21, Rotronic).

The NO oxidation efficiency and the NOx removal efficiency were calculated according to the following equation:

$$NO_{conversion}(\%) = \frac{[NO]_{in} - [NO]_{out}}{[NO]_{in}} \times 100\% \quad (1)$$

$$NOx_{removal}(\%) = \frac{[NO_x]_{in} - [NO_x]_{out}}{[NO_x]_{in}} \times 100\% \quad (2)$$

Where $[NO]_{in}$, $[NOx]_{in}$ and $[NO]_{out}$, $[NOx]_{out}$ represent the concentrations of NO and NOx in the inlet and outlet gases respectively.

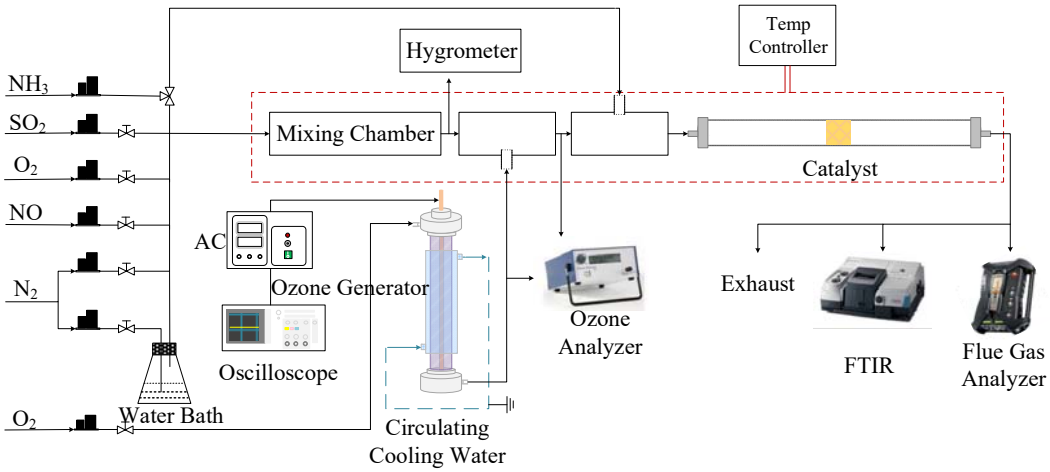

Fig.1. Schematic of the experimental setup for O_3 assisted SCR system.

3. Results and discussion

3.1 The influence of NO oxidation

Fig. 2 shows the ozone production and O_3/NO molar ratio as a function of discharge voltage. As shown in Fig. 2, the ozone concentration increased with the discharge voltage increased, while the discharge voltage increased from 5.6 kV to 6.4 kV with the O_3/NO molar ratio increased from 0.2 to 0.8. These results indicated that the enhancement of O_3 concentration can effectively oxidize NO. In this experiment, the initial simulated flue gas only contained NO, so the oxidation of NO was the key to achieve fast SCR and efficient NOx removal. Fig. 3 shows the effect of molar ratio of O_3/NO on the oxidation efficiency of NO. As the residence time of the simulated flue gas in the heating furnace increased from 2 s to 6 s, the oxidation efficiency of NO did not change significantly. There was a certain deviation of NO oxidation efficiency within 6 s, which might be due to fluctuations in simulated smoke concentration or ozone generation. The oxidation products of NO at different O_3/NO molar ratios were analyzed by FTIR. As shown in Fig. 4, with the O_3/NO molar ratio (≤ 0.8) increased, the peak intensity at 1900 cm^{-1} declined gradually, while the peak intensity at 1600 cm^{-1} and 2919 cm^{-1} increased. Correspondingly, the characteristic peak of NO gradually disappeared, while the intensity of NO_2 characteristic gradually improved. Only NO_2 was detected in the oxidation products, and no characteristic peaks of NO_3 and N_2O_5 were observed in FTIR. These results suggested that the increase of the molar ratio of O_3/NO (≤ 0.8) can effectively oxidize NO to NO_2. In addition, a series of kinetic model parameters during NO oxidation in this experiment were obtained from data in the National Institute of Standards and Technology (NIST) database, as shown in formulas (3)−(5).

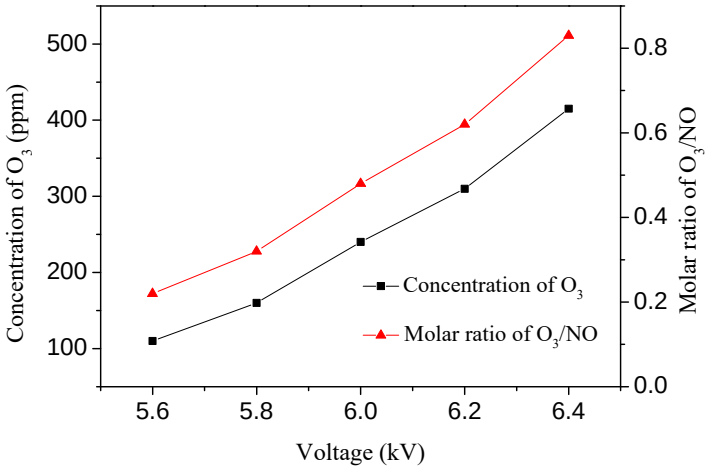

Fig. 2. O_3 concentration at different voltages. Conditions: $[NO]_{in}$ = 500 ppm, O_2 flow rate = 10 mL·min^{-1}, and flue gas flow rate = 1.0 L·min^{-1}.

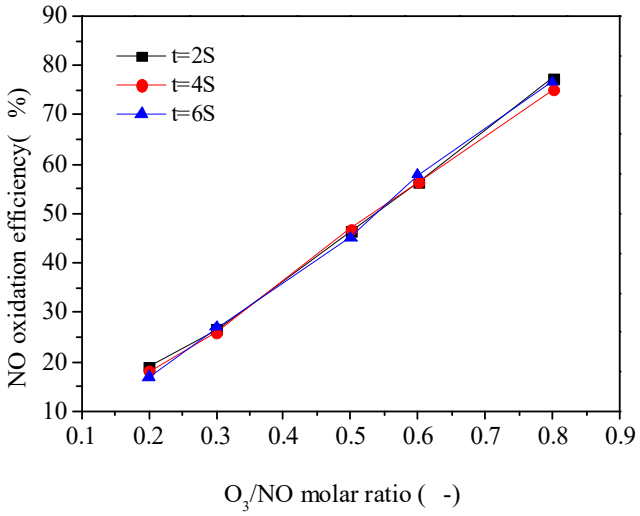

Fig. 3. Effect of the molar ratio of O_3/NO on NO oxidation efficiency. Conditions: $[NO]_{in}$ = 500 ppm, flue gas temperature = 150 °C, and flue gas flow rate = 1.0 L·min^{-1}.

$$O_3 + NO = NO_2 + O_2 \qquad k=1.80\times10^{-14}cm^3\cdot molecule^{-1}\cdot s^{-1} \qquad (3)$$

$$O_3 + NO_2 = NO_3 + O_2 \qquad k=2.50\times10^{-17}cm^3\cdot molecule^{-1}\cdot s^{-1} \qquad (4)$$

$$NO_2 + NO_3 = N_2O_5 \qquad k=3.60\times10^{-30}cm^3\cdot molecule^{-1}\cdot s^{-1} \qquad (5)$$

Considering the temperature range of standard SCR flue gas is usually within 400 °C, the effect of reaction temperature on NO oxidation efficiency by O_3 was investigated, as shown in Fig. 5, When the simulated flue gas temperature was 100 °C and the molar ratio of O_3/NO was 0.5, the oxidation efficiency of NO was 45.36%. However, the oxidation efficiency of NO decreased significantly with the increase of reaction temperature. In particular, the oxidation efficiency of NO was only 3.55% at 400 °C. With the temperature of the flue gas raised, the self-decomposition rate of ozone accelerated, which led to the decrease of NO oxidation efficiency. Therefore, an appropriate temperature condition should be selected to reduce the loss of thermal decomposition of O_3 during NO oxidation process.

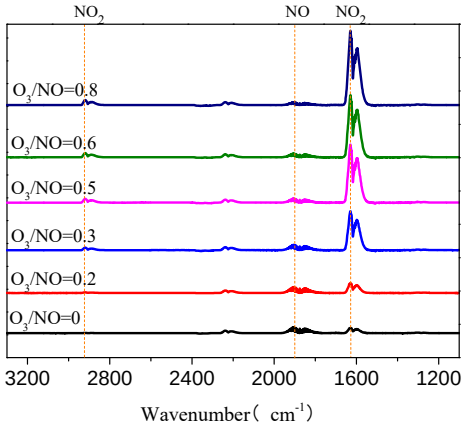

Fig. 4. Effect of the molar ratio of O_3/NO on NO oxidation products.
Conditions: $[NO]_{in}$ = 500 ppm, flue gas temperature = 150°C, residence time = 2 s, and flue gas flow rate = 1.0 L·min^{-1}.

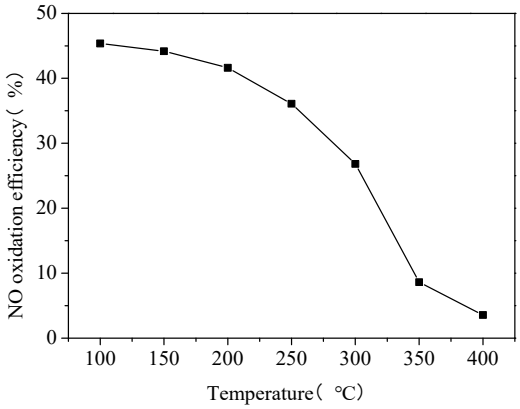

Fig. 5. Effect of gas temperature on NO oxidation efficiency by O_3.
Conditions: $[NO]_{in}$ = 500 ppm, O_3 concentration= 250 ppm, residence time = 2 s, and flue gas flow rate = 1.0 L·min^{-1}.

3.2 Effect of O_3/NO molar ratio on the NOx removal

The effect of the molar ratio of O_3/NO on NOx removal efficiency in ozone assisted SCR system is investigated in Fig. 6. As the temperature raised from 100 °C to 350 °C, the NOx removal efficiency increased significantly from 8.8% to 71.8%. However, the NOx removal efficiency started to decline rapidly in the SCR process, when the temperature reached 400 °C. This result can be attributed to the activation energy required for the removal of NOx by the standard SCR process was higher. Since the activation temperature of the catalyst did not reach at a low temperature, the efficiency of NOx removal was low. As the temperature of the flue gas gradually increased, the catalytic activity was excited, which enhanced the NOx removal efficiency under the presence of NH_3 in formula 6. When the temperature exceeded 400 °C, the NOx removal efficiency significantly decreased, which can be attributed to the deactivation of catalyst due to the high temperatures. In addition, the NOx removal efficiency could markedly enhance with the injection of O_3, and that increased from 67.4% to 88.0% with the O_3/NO molar ratio increased from 0.2 to 0.5 at a temperature of 250 °C. However, when the O_3/NO molar ratio further increased by 0.6, the NOx removal efficiency slightly decreased. This result indicated that the O_3/NO molar ratio of 0.5 might be the optimal value to realize the fast SCR process (Formula 7).

$$4NH_3 + 4NO + O_2 \xrightarrow{catalyst} 4N_2 + H_2O \quad (6)$$

$$2NH_3 + NO + NO_2 \rightarrow 2N_2 + 3H_2O \quad (7)$$

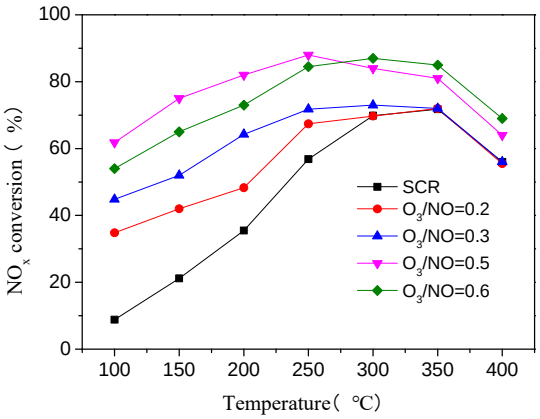

Fig. 6. Effect of O_3/NO molar ratio on removal of NOx by O_3−NH_3−SCR process.
Conditions: $[NO]_{in}$ = $[NH_3]$ = 500 ppm, $[O_2]$ concentration = 3%, residence time = 2 s, and flue gas flow rate = 1.0 L·min^{-1}, GHSV = 25000 h^{-1}.

3.3 Effect of O_3 assisted NH_3−SCR on NOx removal

Fig. 7 shows the effect of two injection locations of O_3 upstream and downstream (denoted as O_3−NH_3−SCR and NH_3−O_3−SCR, respectively) NH_3 on NOx removal of flue gas in O_3 assisted SCR process. The result showed that the NH_3−O_3−SCR method and the O_3−NH_3−SCR method had a NOx removal efficiency of 82.0% and 88.3% at 250 °C, respectively, while the best removal efficiency of standard SCR at 350 °C was only 71.8%. It can be seen that, compared with the standard SCR process, the injection of O_3 can significantly enhance the NOx removal efficiency at the same temperature and the temperature of largest removal efficiency could be effectively decreased in O_3 assisted SCR process. Moreover, the NOx removal efficiency in O_3−NH_3−SCR system was larger than that in NH_3−O_3−SCR system. When the temperature exceeded 250 °C, both the NOx removal efficiency in O_3−NH_3−SCR and NH_3−O_3−SCR system began to decrease, which can be described to the O_3 rapid decomposition due to the high reaction temperature.

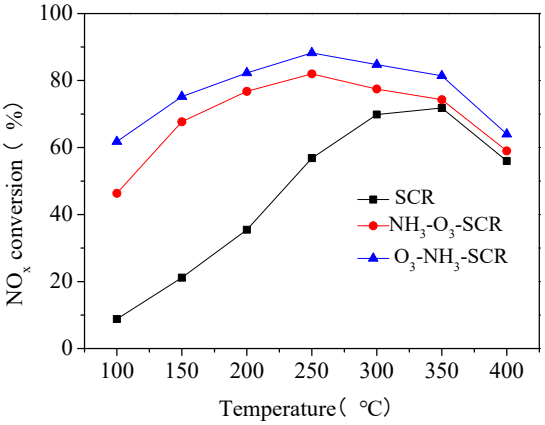

Fig. 7. Effect of O_3 injection position assisted SCR on NOx removal efficiency.
Conditions: $[NO]_{in}$ = $[NH_3]$ = 500 ppm, O_3 concentration = 250 ppm, flue gas flow rate = 1.0 L·min^{-1}, and GHSV = 25000 h^{-1}.

In order to analyze the possible reasons for the difference in NO removal efficiency between the NH_3−O_3−SCR process and the O_3−NH_3−SCR process, the effect of NO oxidation efficiency by ozone and NH_3 injection locations in simulated flue gas was analyzed. As shown in Fig. 8, the NO oxidation effect of O_3−NH_3 system was better than NH_3−O_3 system in the range of the O_3/NO molar ratio from 0.2 to 0.8. This result might be described to the oxidation reaction of ozone with ammonia consumed part of ozone, resulting in a decrease in the oxidation efficiency of NO in NH_3−O_3 system (Formula 8) [31].

$$2NH_3 + 4O_3 \rightarrow 4O_2 + NH_4NO_3 + H_2O \quad (8)$$

In order to verify the presence of ammonium nitrate, the NO_3^- content of the exhaust gas was measured. The method of collecting ammonium nitrate was to filter the gas online at the outlet during the reaction by using a micron-sized fiber gas particulate filter. Then, the filter was placed in excess water for sonication to sufficiently dissolve the ammonium nitrate adhering to the filter. The NO_3^- in the dissolved ammonium nitrate solution was finally determined by ion chromatography to determine the yield of ammonium nitrate. The yield of ammonium nitrate was 2.2 mg h^{-1} at simulated flue gas temperatures of 150 °C. This study suggested that the NO oxidation efficiency in the O_3–NH_3 system was better than that in the NH_3–O_3 system. When ozone was injected into the simulated flue gas after ammonia, ozone could react with ammonia to produce ammonium nitrate. In this process, ammonia could compete with NO for ozone, which affected the production of NO_2 and reduced the removal efficiency of NOx in the fast SCR process.

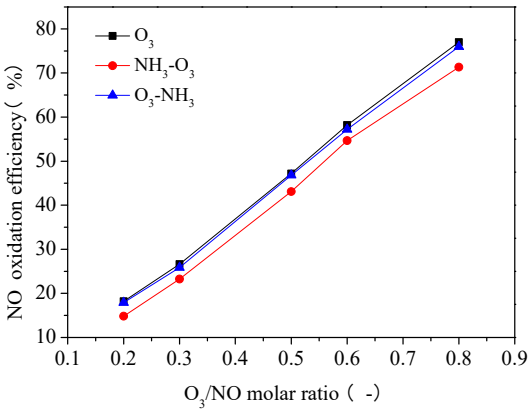

Fig. 8. Effect of O_3 injection position on NO oxidation efficiency.
Conditions: $[NO]_{in}$ = $[NH_3]$ = 500 ppm, O_2 content = 3 vol.%, flue gas temperature = 150 °C, residence time = 2 s, flue gas flow rate = 1.0 L·min^{-1}.

3.4 Effect of H_2O on the NOx removal

Fig. 9 shows the effect of H_2O on NO oxidation efficiency in simulated flue gas. The temperature of flue gas was kept at 150 °C. The study showed that the addition of water vapor in the simulated flue gas reduced the oxidation efficiency of NO by O_3. When the content of water vapor in carrier gas was 0 vol.%, 2 vol.%, and 6 vol.%, the NO oxidation efficiency was 81.4%, 77.6%, and 70.0% at the O_3/NO molar ratio of 0.8, respectively. This result could be attributed to the loss of ozone in the experiment, because the ozone could react with H_2O and H_2O_2 (Formula (9) and (10)) [32].

$$O_3 + H_2O \rightarrow H_2O_2 + O_2 \qquad k_{O_3+H_2O} < 1.1\times10^{-22} cm^{-3}\cdot molecule^{-1}\cdot s^{-1} \qquad (9)$$

$$O_3 + H_2O_2 \rightarrow H_2O + 2O_2 \qquad k_{O_3+H_2O_2} < 4.0\times10^{-20} cm^{-3}\cdot molecule^{-1}\cdot s^{-1} \qquad (10)$$

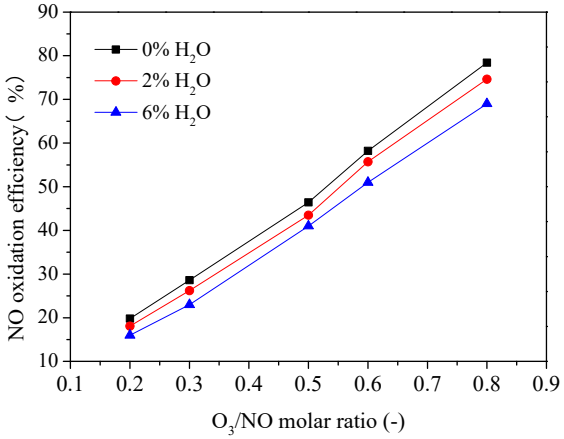

Fig. 9. Effect of H_2O on NO oxidation efficiency of O_3 oxidation.
Conditions: $[NO]_{in}$ = 500 ppm, O_2 content = 3 vol.%, flue gas temperature = 150 °C, flue gas flow rate = 1.0 L·min^{-1}.

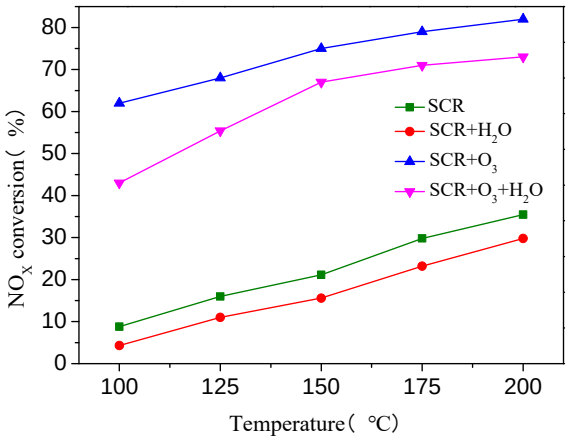

Fig. 10. Effect of H_2O on NOx removal efficiency in O_3−NH_3−SCR system.
Conditions: $[NO]_{in}$ = $[NH_3]$ = 500 ppm, O_3 concentration = 250 ppm, O_2 content = 3 vol.%, H_2O content = 6 vol.%, flue gas temperature = 150 °C, flue gas flow rate = 1.0 L·min^{-1}., and GHSV = 25000 h^{-1}.

Fig. 10 shows the effect of the presence of H_2O in the flue gas on the degradation efficiency of the O_3−NH_3−SCR process and the SCR process. At the temperature of 150 °C, the NOx removal efficiency with 0 vol.% and 6 vol.% H_2O in the SCR process was 21.15%, 15.6%, respectively. The temperature was in the range of 100 °C to 200 °C, and NOx removal efficiency in the flue gas with 6 vol.% H_2O was less than that with 0 vol.% H_2O. This result can be probably explained by the competitive adsorption of H_2O with NO, NH_3 and other reactive molecules on the catalyst surface. Another reason was that the hydroxylation of H_2O occurred on the catalyst surface, resulting in a decrease in the redox capacity of the catalyst. And hydroxylation reaction was generally non-renewable for low temperature NH_3−SCR process [33, 34]. However, compared with SCR, O_3−NH_3−SCR process had a significantly improved NOx removal efficiency at the temperature of 150 °C, and the NOx removal efficiency with the presence of H_2O of 0 vol.% and 6 vol.% were 75.0% and 67.0%, respectively. Experiment showed that the presence of water vapor reduced the NOx removal efficiency in O_3−NH_3−SCR system. The reason for this phenomenon was that water vapor affected the process of NO conversion to NO_2 and had a toxic effect on the catalyst surface. Therefore, the addition of ozone only enhanced the NOx degradation effect, but could not improve the water resistance of the catalyst.

3.5 Effect of SO_2 on the NOx removal

Fig. 11 shows the effect of the presence of SO_2 in the simulated flue gas on the oxidation efficiency of NO. The concentration of O_3 in the simulated flue gas was 250 ppm, and the concentration of SO_2 was 200 ppm. The composition of outlet exhaust gas was determined by FTIR. When SO_2 was contained in the simulated flue gas, no SO_3 was detected in all cases, which might be attributed to the oxidation rate of SO_2 was very low and the concentration of SO_3 produced was under the detection limit [35]. According to the formula 11, the reaction of O_3 and SO_2 was difficult to carry out. Therefore, the presence of SO_2 under this condition hardly affected the oxidation efficiency of NO by ozone oxidation.

$$O_3 + SO_2 = SO_3 + O_2 \qquad k=1.95\times10^{-38}cm^3\cdot molecule^{-1}\cdot s^{-1} \qquad (11)$$

The effect of SO_2 in simulated flue gas on the NOx removal efficiency in the O_3−NH_3−SCR and SCR process is shown in Fig. 12. In SCR and O_3−NH_3−SCR systems, the catalyst still maintained a stable NOx removal efficiency after 60 min on stream. However, with the addition of SO_2, the NOx removal efficiency of the SCR system decreased from 21.4% to 9.9% after 120 minutes. Similarly, the NOx removal efficiency of the O_3−NH_3−SCR system reduced from 75.8% to 55.9%. And the O_3−NH_3−SCR system exhibited superior SO_2 resistance than SCR system. The reason of decrease in NOx removal efficiency might be described to the formation of sulfate which could cover the surface active center of catalyst and result in the deactivate of catalyst [36]. Therefore, when SO_2 was contained in the flue gas, O_3−NH_3−SCR process can significantly

improve the NOx degradation efficiency compared with the SCR process, but the inhibition effect of the SO_2 on the catalyst still existed.

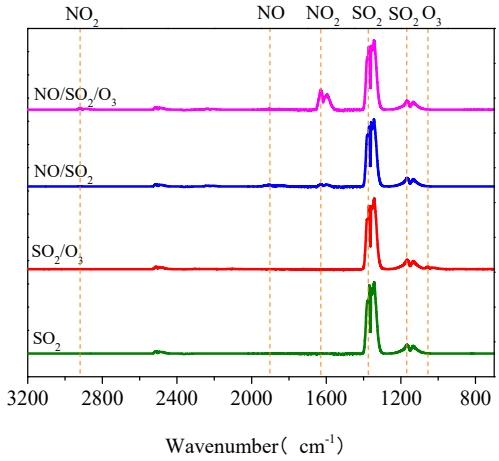

Fig. 11. Effect of SO_2 on NO oxidation efficiency.
Conditions: $[NO]_{in}$ = 500 ppm, O_3 concentration = 250 ppm, O_2 content = 3 vol.%, SO_2 concentration = 250 ppm, flue gas temperature = 150 °C, flue gas flow rate = 1.0 L·min^{-1}.

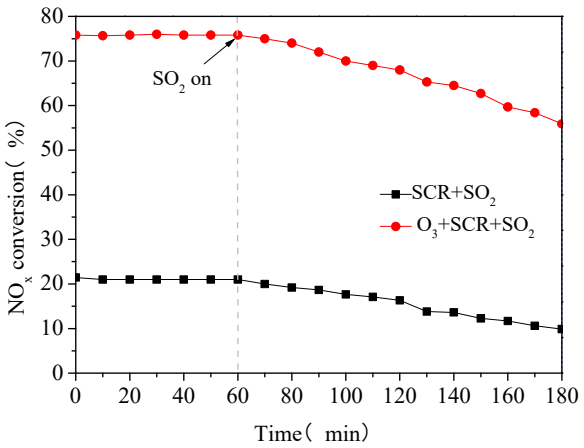

Fig. 12. Effect of SO_2 on NOx removal efficiency in O_3−NH_3−SCR system.
Conditions: $[NO]_{in}$ = $[NH_3]$ = 500 ppm, O_3 concentration = 250 ppm, O_2 content = 3 vol.%, SO_2 concentration = 250 ppm, flue gas temperature = 150 °C, flue gas flow rate = 1.0 L·min^{-1}., and GHSV = 25000 h^{-1}.

4. Conclusion

In this study, the key influencing factors on the oxidation of NO by ozone were studied, including O_3/NO molar ratio, residence time and reaction temperature. Reaction temperature of the flue gas was the main factor affecting the decomposition of O_3. The decomposition rate of O_3 was slow at low temperature and increased significantly when temperature exceeded 150 °C. In addition, when the O_3/NO molar ratio was below 0.8, the oxidation product of NO was only NO_2.

The effect of O_3 injection position and other components, such as O_3/NO molar ratio, H_2O and SO_2, on the NOx removal efficiency of O_3 assisted SCR system were also studied. The results indicated that the O_3 assisted SCR process obviously improved the removal efficiency of NOx in the standard SCR process at the temperature below 250 °C. In addition, the O_3−NH_3−SCR system had better NOx removal effect than the NH_3−O_3−SCR system. SO_2 had almost no effect on NO oxidation efficiency, while H_2O had a certain inhibitory effect on NO oxidation efficiency. The NOx removal of O_3−NH_3−SCR system was also inhibited by H_2O and SO_2 at low temperature, but the NOx removal of O_3−NH_3−SCR process was still significantly better than the SCR process.

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