



Hg²⁺ reduction and re-emission from simulated wet flue gas desulfurization liquors

Jingjing Wo, Meng Zhang, Xiaoya Cheng, Xiaohang Zhong, Jiang Xu, Xinhua Xu*

Department of Environmental Engineering, Zhejiang University, Hangzhou 310027, People's Republic of China

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ABSTRACT

In this study, considering that Hg²⁺ in wet flue gas desulfurization (FGD) systems can easily be reduced and then released into atmosphere, causing secondary pollution, the researches about Hg²⁺ reduction and Hg⁰ re-emission mechanism were carried out. The effects of several experimental parameters on the reduction were studied, including initial pH, temperature, and concentrations of Cl⁻ and S(IV). Our experimental results indicated that Cl⁻ had a restraining effect on the Hg²⁺ reduction and Hg⁰ re-emission, after 24 h reaction, only 20.5% of Hg²⁺ was reduced with 100 mM Cl⁻ in simulated desulfurization solution. Cl⁻ can slow Hg²⁺ reduction and Hg⁰ re-emissions dramatically through changing reaction mechanism, with formation of new intermediate: ClHgSO₃⁻, which can decompose to Hg⁰, but much more slowly than Hg(SO₃)₂²⁻ or HgSO₃. Simulating the conditions of the practical application (initial pH 5, T = 50 °C, S(IV) = 5 mM, Cl⁻ = 100 mM), we also found that Ca²⁺, NO₃⁻, F⁻, etc. all had obvious effects on reduction rates. Based on the material balance and characteristic of the reactants, the reduction emission mechanism of Hg²⁺ has been established, providing theoretical basis for industrial application of mercury control in wet FGD systems.

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1. Introduction

For the past several decades, mercury (Hg) has received increasing attention as a pollutant of serious concern [1]. The amount of mercury in earth's biosphere is increasing gradually due to both natural and anthropogenic emissions [2]. The major man-made sources of mercury emission into the atmosphere are the flue gases from coal combustors and municipal waste incinerators [3]. The large quantity and scale of boilers make coal burning the largest anthropogenic source of mercury emissions, emitting approximately 50 t of mercury into the atmosphere annually, in other words, about one-third of the total anthropogenic emission [1,2]. In China, coal now accounts for about 67% of the primary energy consumption and will continue to play an important role in energy uses for a long time to come [4]. Mercury contamination is a serious problem in China, particularly concentrated in southwestern China (particularly Guizhou Province) [5]. Pacyna and Pacyna estimated that China's mercury emissions from coal combustion contributed more than 25% to total global mercury emissions [6]. Wu et al. reported that Hg emissions from coal combustion in China increased from 202.4 t in 1995 to 256.7 t in 2003 at an average annual rate of 3.0%. Among all of the coal consumption sectors, the power sector is the leading one in Hg

emission growth, up by 5.9% annually and reaching 100.1 t in 2003 [7].

The emission of mercury into the atmosphere and its transport away from emission sources impose serious health concerns. Mercury buried deep in the ground or beneath lake and ocean floors does not appear to present a threat to either the environment or human health [2]. However, mercury released to the environment can be converted through biological process into an organic form, methyl mercury (MeHg), a neurotoxin that can bio-accumulate in the food chain [3]. After many years of coal burning and other industrial processes, the level of organic mercury in many water bodies rises to an extent that consumption of fish is unsafe [2]. Reports estimated that methyl mercury level in edible fish would double near a 1000 MW power plant [4]. Based on available evidences of the mercury effects on health effects, the US Environmental Protection Agency (EPA) issued the Clean Air Mercury Rule (CAMR) to regulate mercury release into the environment [4,8].

There are three forms of mercury which may be present in the coal combustion flue gas: elemental mercury (Hg⁰), oxidized mercury (Hg²⁺), and particle-bound mercury (Hg^p) [9]. Generally, oxidized mercury is water-soluble and can be efficiently removed in the wet flue gas desulfurization (FGD) processes of coal combustion, which exhibits up to 90% removal of Hg²⁺. Particle-bound mercury comprises only ca. 10% of the total, most can be captured by the typical air pollution control devices (APCD) such as electrostatic precipitators and baghouses. However, the elemental mercury is difficult to capture with typical APCD due to its volatility

* Corresponding author. Tel.: +86 571 87951239; fax: +86 571 87952771.
E-mail address: xuxinhua@zju.edu.cn (X. Xu).

and chemical stability [3,4,9]. Mercury in atmosphere is mostly in the form of elemental mercury vapor, it can circulate in the atmosphere for 1–2y, allowing its wide dispersion and long-distance transportation [10]. Of total coal combustion emission, the main mercury forms in China are Hg^{2+} and Hg^0 that about 61.2% of the Hg is released as Hg^{2+} , 22.5% as Hg^0 , and 16.3% as Hg^0 in 2000 [5,11].

Nowadays wet FGD systems have already obtained wide application in developed countries because of their high removal efficiency, reasonable cost, and wide applicability. If a high removal efficiency of mercury is also achieved, wet FGD systems can be used for cost-effective multi-pollutant control (operation cost is only 30–50% of activated carbon injection (ACI)), hence research of controlling the mercury emission also has received special attention. However, field data have shown that, in general, less than 70% mercury removal efficiency is achieved by wet FGD. There presences two problems as follows: firstly, Hg^0 can hardly be dissolved in desulfurization solutions, leading to poor removal efficiency of Hg^0 ; secondly, it is speculated that some of the absorbed mercury (e.g., ionic form of mercury (Hg^{2+}) such as mercuric chloride (HgCl_2)) can be converted back to elemental mercury (Hg^0) and re-emitted [12]. Therefore, it is necessary to control secondary pollution of mercury in desulfurization solutions or desulfurization byproducts.

In this study, a sequence of experiments was carried out in order to evaluate the influence of different operational parameters on Hg^{2+} reduction efficiency in a 1-L three-necked flask. In the end, the mechanism of the reactions between Hg^{2+} , SO_3^{2-} and Cl^- in the process was also hypothesized.

2. Experiments and methods

2.1. Chemicals

All of the chemicals, such as anhydrous sodium sulfite ($\geq 97.0\%$, AR), sodium chloride ($\geq 99.8\%$, GR), nitric acid (65–68%, GR), hydrochloric acid (36–38%, GR), sodium hydroxide ($\geq 96.0\%$, AR), stannous chloride ($\geq 98.0\%$, AR), stannum ($\geq 99.9\%$, AR), potassium dichromate (99.95–100.05%, GR), mercury chloride ($> 99.0\%$, AR), etc. were all used as received without further purification. The gases, such as Ar ($\geq 99.99\%$) and N_2 ($\geq 99.9\%$), stored in the steel cylinders were provided by Jingong Gas Co. Ltd.

2.2. Experimental procedure

The batch experiments for Hg^{2+} reduction in simulated desulfurization solutions proceeded in a 1-L three-necked flask inside a water bath maintained at the desired temperature, and HgCl_2 aqueous solution were added into the flask as the source of Hg^{2+} . Other chemicals, such as Na_2SO_3 , NaCl, CaCl_2 , MgCl_2 , NaNO_3 and NaF were selectively added to simulate the source of SO_3^{2-} , Cl^- , Ca^{2+} , Mg^{2+} , NO_3^- and F^- . The reaction solution was stirred under nitrogen atmosphere to remove production Hg^0 . Samples of 5 mL were periodically collected at selected time intervals, then it was diluted to 50 mL in the volumetric flask with 5 mL stationary liquid already in, ready for analysis.

2.3. Analytical method

Concentrations of Hg^{2+} in aqueous solution were analyzed by QM201C fluorescence arsenic/mercury analyzer (Qing'an Instrument Co., Suzhou, China). Samples and 7% (w/w) SnCl_2 solution were transferred by a peristaltic pump into mercury analyzer at a constant rate, superfluous SnCl_2 was used to reduce Hg^{2+} to Hg^0 . Mercury vapor produced was supplied with Ar as carrier gas and

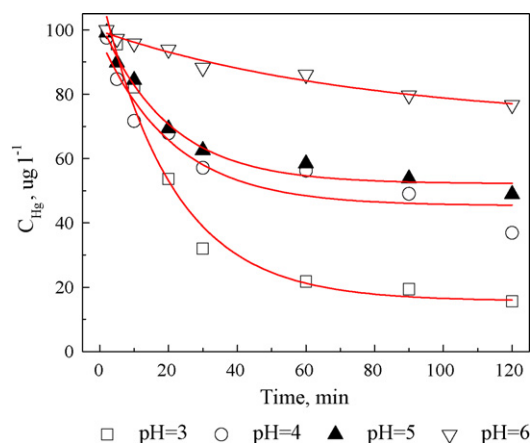


Fig. 1. Effect of initial pH on the Hg^{2+} reduction efficiency ($T = 50^\circ\text{C}$, $S(\text{IV}) = 5.0\text{ mM}$).

flushed out and was measured by cold vapor generation atomic absorption spectrometry.

The ultraviolet (UV) spectra characterization was analyzed by TU-1900 UV-VIS spectrophotometer (Beijing Purkinje General Instrument Co. Ltd.), and pH value was measured with a digital pH meter (Mettler-Toledo Instruments (Shanghai) Co. Ltd.).

3. Results and discussion

3.1. Effect of initial pH value

The initial pH value of solution is important in reduction of Hg^{2+} . Prior to starting reaction all reactant solutions were adjusted to different pH values by diluted H_2SO_4 and NaOH, and during the reaction pH values were not adjusted. As Fig. 1 shows, the Hg^{2+} concentration continued to decrease as the experiment proceeded, but the rate of decrease gradually slowed, especially after an hour. With the decrease of initial pH, mercury removal efficiency increased gradually. At higher initial pH value (pH 6), concentration of Hg^{2+} in simulated desulfurization solutions decreased more slowly, keeping relatively stable. When the initial pH values increased from 3.0, 4.0, 5.0 to 6.0, after 2 h, the Hg^{2+} reduction efficiencies were 84.4%, 63.1%, 51.1% and 23.3%, respectively. It is speculated that the presence of OH^- at higher pH, greatly reduced the possibility of reduction by transforming Hg^{2+} to $\text{Hg}(\text{OH})^+$ or $\text{Hg}(\text{OH})_2$. On the other hand, decreasing the liquor pH can decrease the sulfite-to-bisulfite ratio, leading to the reduction of Hg^{2+} . However, caution should be exercised for lower pH may decrease the SO_2 removal efficiency and encourage corrosion in wet flue gas desulfurization systems [13].

3.2. Effect of reaction temperature

The performances under different reaction temperatures (40–60°C) were analyzed to study the temperature effect on Hg^{2+} reduction. Results were shown in Fig. 2 that when initial pH value was 5.0 and $S(\text{IV})$ concentration reached 5.0 mM, Hg^{2+} reduction efficiency greatly improved with the increase of reaction temperature. When the temperature was raised from 40 to 60°C, reduction efficiency increased from 37.1% to 76.3% after 2 h reaction, which could be explained by the fact that reduction of the mercuric ion appears to proceed uniquely via the HgSO_3 complex whose decomposition rate is strongly temperature dependent [13]. Loon et al. has found that the rate constant k_0 roughly quadruples with each 10°C increase in temperature [14].

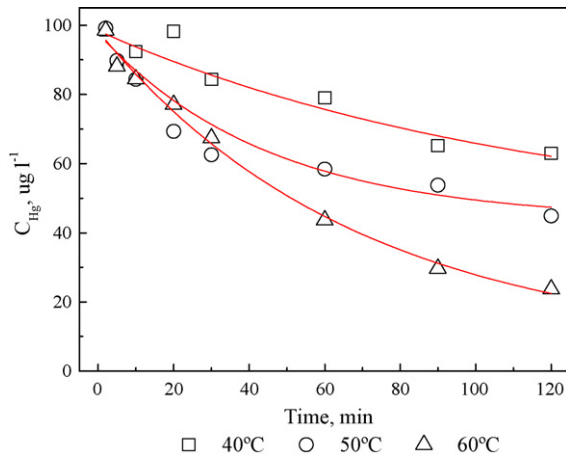
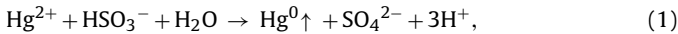


Fig. 2. Effect of temperature on the Hg²⁺ reduction efficiency (initial pH 5, S(IV)=5.0 mM).

3.3. Effect of S(IV) concentration

The mercury removal curves with different initial SO₃²⁻ concentrations are shown in Fig. 3. Evidently, increase in S(IV) concentration led to a slower reduction of mercury, when S(IV) concentration was at low level, Hg²⁺ concentration sharply decreased, for Hg²⁺ was reduced to Hg⁰ as the reaction continued. The removal percentage of Hg²⁺ reached 43.7% in 20 min with S(IV) concentration of 1 mM, and 91.8% Hg²⁺ was removed in 2 h. In contrast, only less than 35% of removal efficiency was obtained for 8 mM S(IV) after 2 h.

Equimolar amounts of Hg²⁺ and HSO₃⁻ in the solutions decompose to Hg⁰ and SO₄²⁻ within minutes at room temperature (E1) [14]. As is found, the reactions for Hg⁰ emission are as follows, the main pathway is through mercuric-sulfite complexes (E2, 3):



Only HgSO₃ decomposes to give reduction of Hg²⁺ with a half-life of approximately 1 min at 25 °C:

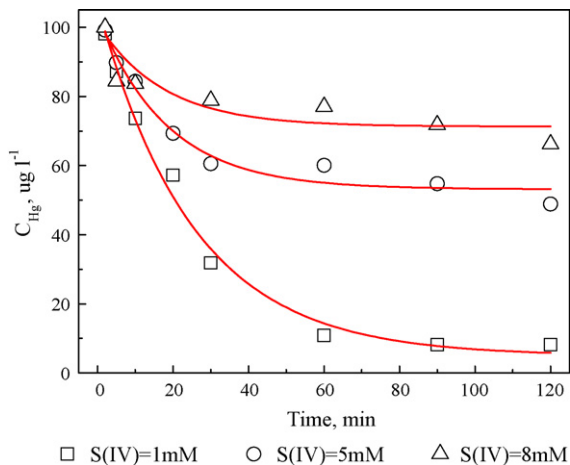
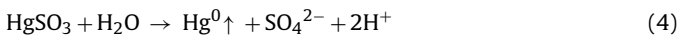


Fig. 3. Effect of S(IV) concentration on the Hg²⁺ reduction efficiency (initial pH 5, T=50 °C).

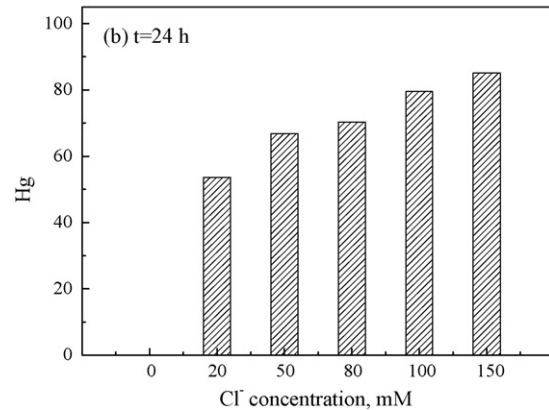
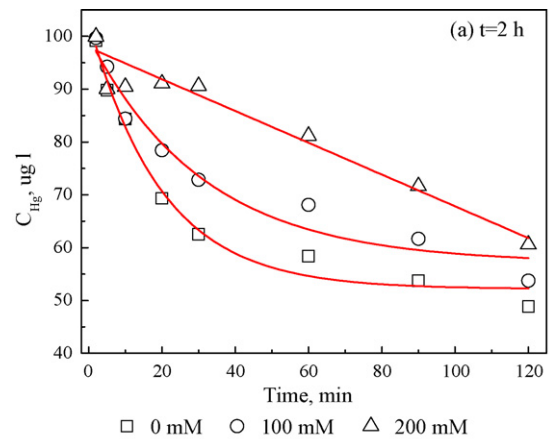


Fig. 4. Effect of Cl⁻ concentration on the Hg²⁺ reduction efficiency (initial pH 5, T=50 °C, S(IV)=5.0 mM).

However, HgSO₃ is not generally assumed to be the dominant mercuric species with excess sulfite, the major species is thought to be Hg(SO₃)₂²⁻ which is compared to be more stable than HgSO₃, and this can explain why rates generally decrease at higher S(IV) concentration.

3.4. Effect of Cl⁻ concentration

Fig. 4(a) and (b) presents the reduction of Hg²⁺ in simulated desulfurization solutions with chloride in it. Concentration of Cl⁻ had an evident effect on the reduction of Hg²⁺ that the simulated desulfurization solution without chloride showed fastest reduction rate, and higher Cl⁻ typically lowered re-emission. From Fig. 4(a), it can be seen that about 55.1% Hg²⁺ was reduced in solution without Cl⁻ while only 39.4% with 200 mM Cl⁻. Considering Hg²⁺ reduction rates were slow at practical conditions, runs were carried out for 24 h open without stirring. Oxygen in the atmosphere greatly depressed the reduction, the trend after 24 h successive reaction is presented in Fig. 4(b). With the increase of Cl⁻ concentration, the removal rate decreased correspondingly, and only 20.5% of Hg²⁺ was reduced with 100 mM Cl⁻ in simulated desulfurization solution even after 24 h while almost 100% reduction efficiency was obtained without Cl⁻ in it.

Chloride slows re-emissions dramatically through changing reaction mechanism, with formation of new intermediate: ClHgSO₃⁻, which can decompose to Hg⁰, but much more slowly than Hg(SO₃)₂²⁻ or HgSO₃. In addition, Cl₂HgSO₃²⁻ is formed reversibly at higher chloride concentration, which does not decompose to Hg⁰. The reactions are below:



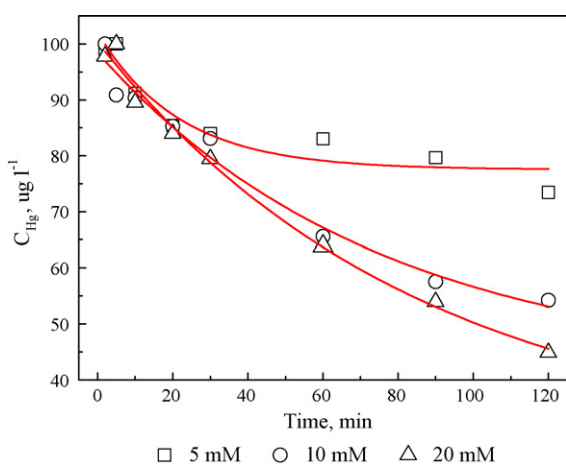


Fig. 5. Effect of Ca^{2+} concentration on the Hg^{2+} reduction efficiency ($T=50^\circ\text{C}$, initial pH 5, $\text{S(IV)}=5.0\text{ mM}$, $\text{Cl}^- = 100.0\text{ mM}$).



3.5. Effect of Ca^{2+} and Mg^{2+} concentration

Based on the references, we estimated the effect of other ions upon the baseline conditions of the practical application (initial pH 5, $T=50^\circ\text{C}$, $\text{S(IV)}=5\text{ mM}$, $\text{Cl}^- = 100\text{ mM}$). As is shown in Figs. 5 and 6, the addition of Ca^{2+} and Mg^{2+} accelerated the reaction and increased the mercury removal efficiency dramatically. When the initial SO_3^{2-} and Cl^- concentrations were 5.0 and 100.0 mM, the removal efficiency of mercury reached 26.5%, 45.8% and 55.1% with an initial Ca^{2+} concentration of 5, 10 and 20 mM, respectively. Similarly, the mercury removal efficiency reached 12.7%, 21.0% and 46.0% with an initial Mg^{2+} concentration of 10, 30 and 40 mM. However, Ca^{2+} provided better accelerated effect than Mg^{2+} . The results may be explained by considering that Ca^{2+} and Mg^{2+} would form ion pairs with sulfite, which can lower S(IV) concentration and affect mercury reduction rates. It also indicates that the concentration of the aqueous species S(IV) is of major importance. On the other hand, the effect of Ca^{2+} can be interfered by the low solubility product of CaSO_3 .

3.6. Effect of NO_3^- concentration

Our experimental results indicate that NO_3^- had an adverse effect on the Hg^0 re-emission, as increasing the NO_3^- concentration

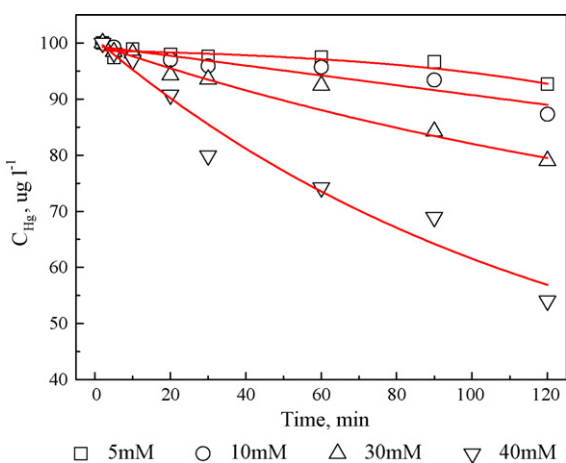


Fig. 6. Effect of Mg^{2+} concentration on the Hg^{2+} reduction efficiency ($T=50^\circ\text{C}$, initial pH 5, $\text{S(IV)}=5.0\text{ mM}$, $\text{Cl}^- = 100.0\text{ mM}$).

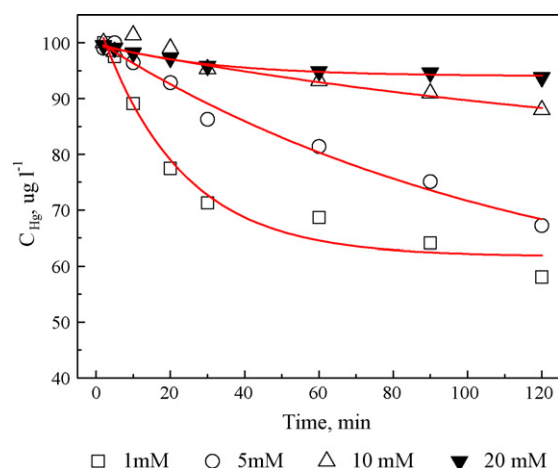


Fig. 7. Effect of NO_3^- concentration on the Hg^{2+} reduction efficiency ($T=50^\circ\text{C}$, initial pH 5, $\text{S(IV)}=5.0\text{ mM}$, $\text{Cl}^- = 100.0\text{ mM}$).

from 1 to 20 mg L^{-1} led to the decrease of the removal efficiency from 41.9% to 6.20% in 2 h (Fig. 7). The negative effect was serious especially at lower NO_3^- concentration. Based on the analysis above, we have guessed that the additive of NO_3^- can inhibit the formation of HgSO_3 complexes in Hg^{2+} emission.

3.7. Effect of F^- concentration

Fig. 8 illustrates the concentration depletion curves of Hg^{2+} in the presence of F^- . It is shown that the higher the F^- concentration was, the lower the mercury removal rate was. The removal percentage reached 57.2% in 2 h in solution with 0.5 mM F^- , but only less than 10.1% of removal efficiency was obtained with 4 mM F^- in it. Reasons have been presumed that stable chelate compound may be formed with Hg^{2+} and F^- in solution.

3.8. Reaction mechanism and model

UV spectrum was further investigated to make sure the change of the intermediate product in the Hg^{2+} reduction. In this study, the UV spectrum and rate curve was recorded by analyzing a solution containing 1 mg L^{-1} Hg^{2+} and 1.0 mM Na_2SO_3 in 1 cm square quartz cuvettes at pH 3 and room temperature, as is shown in Fig. 9. A peak was observed with λ_{max} 230 nm, the absorbance decreased from 0.073 to 0.023 in 19 min, and the rate curve at 230 nm varied curvi-

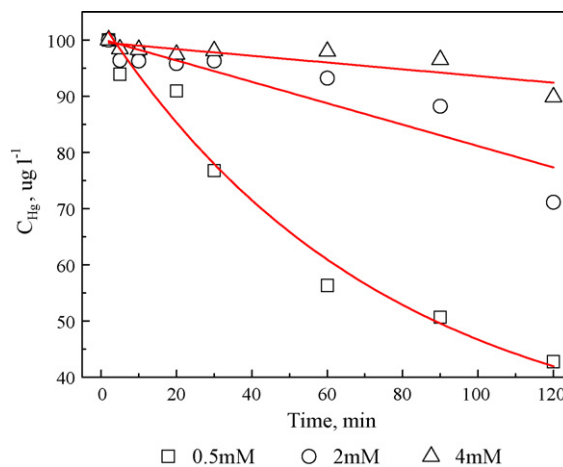


Fig. 8. Effect of F^- concentration on the Hg^{2+} reduction efficiency ($T=50^\circ\text{C}$, initial pH 5, $\text{S(IV)}=5.0\text{ mM}$, $\text{Cl}^- = 100.0\text{ mM}$).

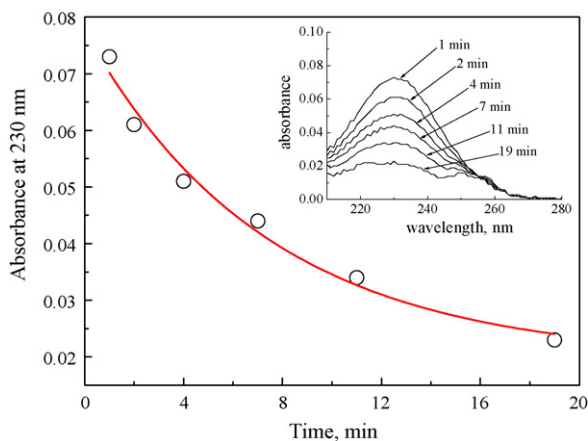


Fig. 9. UV spectra and rate curve of $1 \text{ mg L}^{-1} \text{ Hg}^{2+}$ at $1.0 \text{ mM Na}_2\text{SO}_3$ solution.

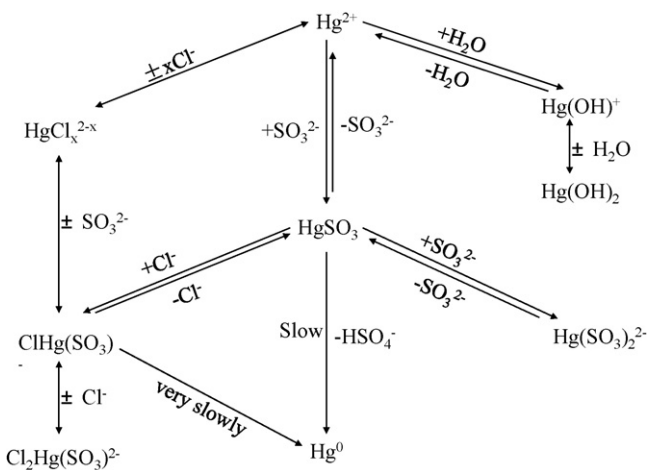
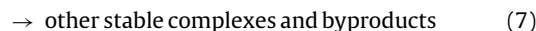
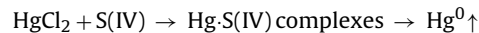


Fig. 10. Diagram of main reaction pathways of Hg^{2+} reduction and Hg^0 re-emission in desulfurization solutions.

linear with reaction time. The complex, two sulfite anions bound to the aqueous mercuric cation, forming in the reaction of the aqueous mercuric ion with excess sulfite, is believed to be $\text{Hg}(\text{SO}_3)_2^{2-}$ [13]. We can conclude that the intermediate product $\text{Hg}(\text{SO}_3)_2^{2-}$ was not stable indefinitely, they decreased as the reaction proceeded, for reduction of Hg^{2+} in the presence of excess hydrogen sulfite involves in dissociation of sulfite from $\text{Hg}(\text{SO}_3)_2^{2-}$, followed by decomposition of HgSO_3 . Rate constant can be obtained from non-linear least-squares fits to the first-order integrated rate equation [13].

Based on the above experiments, we know that the reaction mechanisms of Hg^{2+} absorbed in the wet FGD are complex, the main pathways are through mercuric-sulfite and chloro-mercuric-sulfite species, and it could be described as the reactions shown in Fig. 10. Formations of HgSO_3 , $\text{Hg}(\text{SO}_3)_2^{2-}$, ClHgSO_3^- and $\text{Cl}_2\text{HgSO}_3^{2-}$, with their different characters of generating Hg^0 (the sequence of the tendency in Hg^0 re-emission was $\text{HgSO}_3 > \text{Hg}(\text{SO}_3)_2^{2-} > \text{ClHgSO}_3^-$, $\text{Cl}_2\text{HgSO}_3^{2-}$ does not decompose to Hg^0), can explain how excess sulfite and chloride lead to a slower reduction of mercury. The observation agrees very well with a report previously published by others that Chang and Ghorishi suggested that only a portion of Hg^{2+} species absorbed in the scrubbing liquor functioned as the precursor of Hg^0 re-emissions, while the remaining portion was relatively stable and would not serve as the precursor of Hg^0 re-emissions, like represented by Eq. (7) [13,15]. In addition, byproducts of $\text{Hg}(\text{OH})^+$ or $\text{Hg}(\text{OH})_2$ may form in the presence of

OH^- , which greatly weaken the Hg^{2+} reduction. Preliminary interpretation of modeling results is that re-emissions rate is controlled by mercuric chloro-sulfite complexes at low pH but gradually by mercuric-sulfite complexes alone at higher pH.



4. Conclusion

An evaluation of the influence of different experimental parameters on mercury removal efficiency was carried out. The experimental results indicate that flue gas Hg^0 re-emission across a wet FGD scrubber can be reduced by increasing the initial pH value, concentration of S(IV), or lowering the temperature. Under simulated wet FGD conditions, Hg^{2+} reduction efficiency increased with the concentration of Ca^{2+} and Mg^{2+} increasing, and decreased with the increasing concentration of NO_3^- , F^- and Cl^- . The S(IV) induced Hg^{2+} reduction and Hg^0 re-emission mechanism is complex, the main pathways are through $\text{Hg}\cdot\text{S(IV)}$ complexes. Chloride slows re-emissions dramatically through changing reaction mechanism, with formation of new intermediate: ClHgSO_3^- , which can decompose to Hg^0 , but much more slowly than $\text{Hg}(\text{SO}_3)_2^{2-}$ or HgSO_3 . The present findings could be valuable for industrial application of characterizing and optimizing mercury control in wet FGD systems.

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References

- [1] A. Hoffart, W. Seames, E. Kozliak, S. Riedinger, J. Francini, C. Carlson, A two-step acid mercury removal process for pulverized coal, *Fuel* 85 (2006) 1166–1173.
- [2] H.Q. Yang, Z.H. Xu, M.H. Fan, A.E. Bland, R.R. Judkins, Adsorbents for capturing mercury in coal-fired boiler flue gas, *J. Hazard. Mater.* 146 (2007) 1–11.
- [3] T. Morimoto, S.J. Wu, M.A. Uddin, E. Sasaoka, Characteristics of the mercury vapor removal from coal combustion flue gas by activated carbon using H_2S , *Fuel* 84 (2005) 1968–1974.
- [4] X.H. Xu, Q.F. Ye, T.M. Tang, D.H. Wang, Hg^0 oxidative absorption by $\text{K}_2\text{S}_2\text{O}_8$ solution catalyzed by Ag^+ and Cu^{2+} , *J. Hazard. Mater.* 158 (2008) 410–416.
- [5] D.G. Streets, J.M. Hao, Y. Wu, J.K. Jiang, M. Chan, H.Z. Tian, X.B. Feng, Anthropogenic mercury emissions in China, *Atmos. Environ.* 39 (2005) 7789–7806.
- [6] E.G. Pacyna, J.M. Pacyna, Global emission of mercury from anthropogenic sources in 1995, *Water, Air, Soil Pollut.* 137 (2002) 149–165.
- [7] Y. Wu, S.X. Wang, D.G. Streets, J.M. Hao, J.K. Jiang, Trends in anthropogenic mercury emissions in China from 1995 to 2003, *Environ. Sci. Technol.* 40 (2006) 5312–5318.
- [8] Z.J. Mei, Z.M. Shen, Z.Y. Mei, Y.J. Zhang, F. Xiang, J.P. Chen, W.H. Wang, The effect of N-doping and halide-doping on the activity of CuCoO_4 for the oxidation of elemental mercury, *Appl. Catal., B* 78 (2008) 112–119.
- [9] S.J. Wu, M. Ozaki, M.A. Uddin, E. Sasaoka, Development of iron-based sorbents for Hg^0 removal from coal derived fuel gas: effect of hydrogen chloride, *Fuel* 87 (2008) 467–474.
- [10] G.C. Fang, Y.S. Wu, T.H. Chang, Comparison of atmospheric mercury (Hg) among Korea, Japan, China and Taiwan during 2000–2008, *J. Hazard. Mater.* 162 (2009) 607–615.
- [11] J.K. Jiang, J.M. Hao, Y. Wu, D.G. Streets, L. Duan, H.Z. Tian, Development of mercury emission inventory from coal combustion in China, *Environ. Sci.* 26 (2005) 34–39.
- [12] J.C.S. Chang, S.B. Ghorishi, Simulation and evaluation of elemental mercury concentration increase in flue gas across a wet scrubber, *Environ. Sci. Technol.* 37 (2003) 5763–5766.
- [13] L.L.V. Loon, E.A. Mader, S.L. Scott, Sulfite stabilization and reduction of the aqueous mercuric ion: kinetic determination of sequential formation constants, *J. Phys. Chem. A* 105 (2001) 3190–3195.
- [14] L.V. Loon, E. Mader, S.L. Scott, Reduction of the aqueous mercuric ion by sulfite: UV spectrum of HgSO_3 and its intramolecular redox reaction, *J. Phys. Chem. A* 104 (2000) 1621–1626.
- [15] J.C.S. Chang, Y.X. Zhao, Pilot plant testing of elemental mercury reemission from a wet scrubber, *Energy Fuels* 22 (2008) 338–342.