

Article

Influence of Temperature on the Removal Efficiency of Organic Matter and Ammonia from Micro-Polluted Source Water

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Abstract: Temperature is an important factor influencing the treatment effect of biological aerated filters (BAFs). In this study, BAFs incorporating biological manganese oxides (BMOs) were used to treat micro-polluted source water containing organic masses and ammonia, and the influence of temperature on the removal efficiency of the pollutants was investigated. The results showed that after the formation of biogenic manganese oxides (BMOs) in the filter layer, the removal efficiency of COD_{Mn} significantly improved. When the water temperature was approximately 24 °C, 16 °C, and 6–8 °C, the removal rates of COD_{Mn}, ammonia, and manganese were 60.64, 42.55, and 20.48; 98.40, 95.58, and 85.04; and 98.70, 97.63, and 96.38%, respectively. The influence of water temperature on the removal efficiency of the pollutants was hierarchically structured as follows: COD_{Mn} > ammonia > manganese. Analysis of the removal efficiencies of the pollutants along the filter layer showed that COD_{Mn} had been eliminated in every filtration layer, and ammonia and manganese were mainly removed in the 0~0.4 m and 0~0.8 m regions of the filter layer, respectively. With a decreasing water temperature, the concentrations of COD_{Mn}, ammonia, and manganese along the filter layer increased. The biological COD_{Mn}, manganese, and ammonia removal processes were all first-order kinetic reactions. With a decreasing water temperature, the kinetic constant *k* gradually decreased, and the reaction half-life (*t*_{1/2}) gradually increased.

Keywords: organic mass and ammonia; micro-polluted source water; water temperature; biological aerated filter



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1. Introduction

With the extensive use of fertilizers and pesticides in the processes of industrial and agricultural production, as well as the emission of household sewage and the “three wastes” of industry that lack precise treatment, the number of organic pollutants—including pesticides, products from the chemical industry, pharmaceuticals, personal care products, and endocrine disruptors in surface water bodies is increasing [1–3]. Although the concentrations of many contaminants are low, they have little chance of being biodegraded and are very harmful [4,5]. The number of toxicological indicators of organic pollution in the Standards for Drinking Water Quality (GB 5749-2006) has increased to 53. Therefore, the efficient removal of trace organic compounds from slightly polluted source water has become a hot research topic.

BMOs are the product of the oxidation of Mn(II) by manganese-oxidizing bacteria. A BMO’s particle size is on the nanometer scale, and it has a high specific surface area and remarkable oxidation ability [6–8]. BMOs are reduced to Mn(II) after the catalysis and oxidation of an organic mass, and Mn(II) can be oxidized to higher-valence manganese oxide by manganese-oxidizing bacteria, thereby achieving long-lasting catalyzed oxidation of organic matter. Microorganisms are capable of further metabolizing and degrading the catalytic oxidation products of organic matter [9]. Cheng et al. [10] discovered that

the rate of the oxidative degradation of diclofenac by BMOs was much faster than that of the oxidation of diclofenac by MnO_2 produced through chemical processes. It was also found that BMOs could efficiently remove 14 drugs, including ibuprofen and others [11]. Ammonia-N is another important pollution indicator in surface water, and it is often associated with organic matter. Accordingly, in this research, BAFs with BMOs were used to treat micro-polluted source water containing ammonia-N and organic matter. The catalytic oxidation ability of BMOs was fully utilized to realize the synergistic removal of organic mass via BMOs and microorganisms to improve the efficiency of the removal of organic masses and ammonia-N from micro-polluted source water. At the same time, the influence of temperature on the efficiency of removing organic masses and ammonia-N was investigated.

2. Materials and Methods

2.1. Test Device

A flow chart of the biological aerated filter device is shown in Figure 1. The height of the biological aerated filter was 2.5 m, and its inner diameter was 0.1 m. The filter material was manganese sand with a particle size of 4–6 mm and a thickness of 1.5 m. The upper part of the filter was equipped with a mixer for mixing tap water and lake water + manganese sulfate; the bottom part was equipped with an aeration head to supply dissolved oxygen. Water-sampling ports were sequentially arranged down the filter layer of the biological aerated filter with an interval of 0.1 m. The mixture of lake water, manganese sulfate (Shanghai Macklin Biochemical Technology Co. Ltd., Shanghai, China), and ammonium chloride (Shanghai Macklin Biochemical Technology Co. Ltd., Shanghai, China) in the water tank was pumped into the mixer of the filter column via a peristaltic pump, and the volume of the water tank was 500 L.

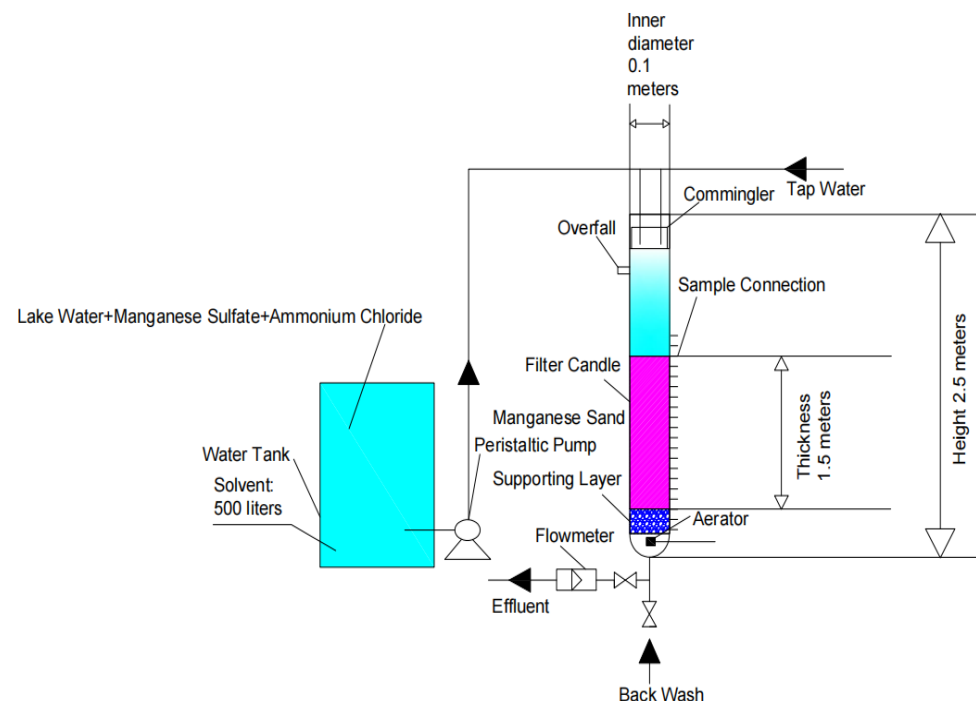


Figure 1. Schematic diagram of the biological aerated filter system.

2.2. Inlet Water Quality

In the experiment, the influent consisted of simulated micro-polluted source water. This water was prepared by combining lake water, manganese sulfate, ammonium chloride, and tap water in precise ratios. The lake water used in the mixture was collected from an artificial lake situated on the campus at which this experiment was conducted. The test

was conducted at temperatures ranging from 3 to 25 °C, and the quality of the influent water can be found in Table 1.

Table 1. Ranges and average values of parameters of the experimental influent water.

Project		COD _{Mn} (mg L ⁻¹)	NH ⁴⁺ -N (mg L ⁻¹)	Mn (mg L ⁻¹)	pH
Intake water quality	range	7.71~10.43	1.30~1.63	0.94~1.12	6.87~7.43
	average value	9.15	1.47	1.02	7.17

2.3. Test Methods

In this experiment, a mature and stable biological aerated filter was used to treat simulated, slightly polluted source water. A mixture of lake water and manganese sulfate was pumped into the mixer of the biological aerated filter using a submersible pump, where it was mixed with tap water, which served as the influent in this experiment. An aeration head was arranged at the bottom of the biological filter column, and the amount of dissolved oxygen in the filter column was equal to 2–3 mg L⁻¹. This experiment started at the end of August and continued until the start of February of the following year, spanning a total of 166 days. During the test, the water temperature, which was not artificially changed, dropped from approximately 24 °C to 4 °C. The influence of water temperature on the removal efficiency with respect to organic mass, ammonia-N, and manganese and the variation trends of organic matter, ammonia-N, and manganese during the experiment were investigated. The aeration biofilter columns had a filtration rate of 3 m/h. They were backwashed with an intensity of 10 L/(s·m²) for 3 min, and the backwash cycle occurred every 14 days. Daily measurements were conducted regarding the influent and effluent concentrations of COD_{Mn}, ammonia nitrogen, manganese, and temperature. Additionally, the concentrations of COD_{Mn}, ammonia nitrogen, and manganese along the filter were measured twice a week.

2.4. Detection Methods

COD_{Mn}—acid potassium permanganate method, ammonia-N—salicylic acid spectrophotometry, manganese—ammonium persulfate spectrophotometry, DO—portable DO analyser, and pH—pH meter.

3. Results and Discussion

3.1. The Influence of Water Temperature on the Removal Efficiency with Respect to COD_{Mn}, Ammonia-N, and Manganese

3.1.1. Effect of Water Temperature on COD_{Mn} Removal Efficiency

In this experiment, a mature biological aerated filter was used, and many microorganisms, such as heterotrophic bacteria that oxidize organic matter, nitrifying bacteria that oxidize ammonia-N, and manganese-oxidizing bacteria that oxidize manganese, were attached to its filter material. In the initial stage of the experiment (Days 1~21), the water temperature in the biological aerated filter was approximately 24 °C, the influent COD_{Mn} concentration was 8.05~10.43 mg L⁻¹, the effluent COD_{Mn} decreased to less than 4 mg L⁻¹, the average concentration was 3.56 mg L⁻¹, and the average removal rate reached 60.64% (Figure 2). The biological aerated filters' removal efficiency with respect to COD_{Mn} in the slightly polluted source water was generally poor. Nengzi et al. [12] used a ceramsite biofilter to treat river water, and the removal rate of COD_{Mn} was 20~35%, with an average removal rate of 25%. Wang et al. [13] used a biological method to treat polluted water sources in the Yangtze River Delta and found that the removal rate of COD_{Mn} was 18%. Research on the treatment of slightly polluted Dongjiang water with light filter material biofilters showed that the average removal rate of COD_{Mn} was 28% [14]. This result was obtained due to the fact that the organic compounds in the micro-polluted source water were mainly humic acid, fulvic acid, lignin, and other refractory organic compounds [15]. In this experiment, the removal rate of COD_{Mn} was very high. The reason for this result

was that manganese was removed at the same time as organic mass and ammonia-N. The bivalent manganese in the influent was oxidized by manganese-oxidizing bacteria attached to the filter material with high-valence BMOs. These BMOs have high oxidation ability; they have the ability to catalyze and oxidize recalcitrant organic compounds in the influent, transforming them into more easily degradable organic matter [16–18], which can then be utilized by heterotrophic microorganisms [17], thus improving the removal efficiency with respect to COD_{Mn} .

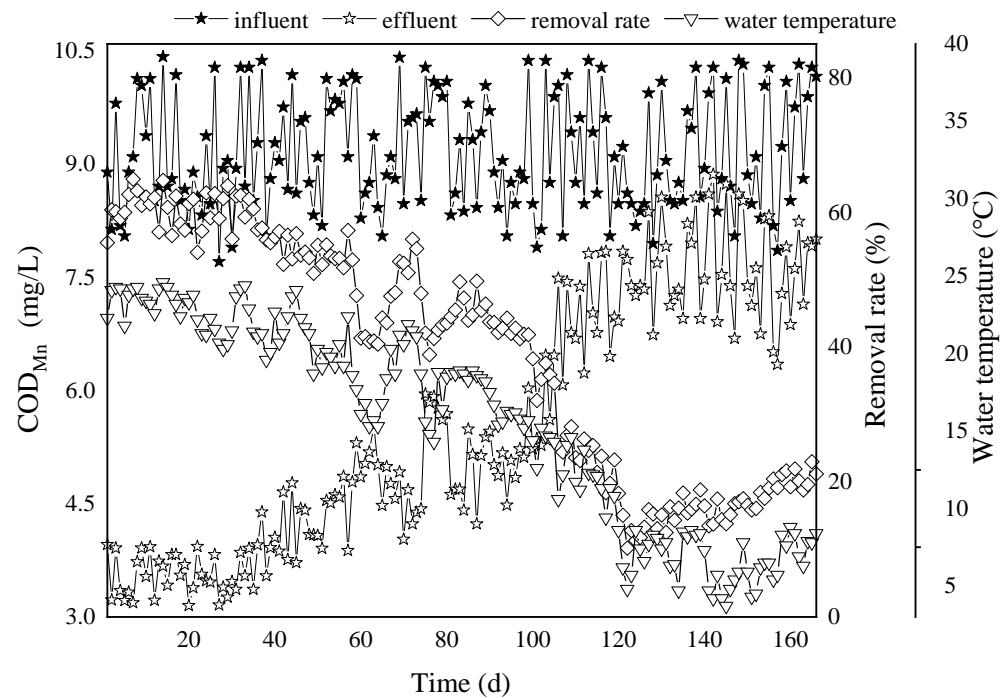


Figure 2. COD_{Mn} removal efficiency of the biological aerated filter at different water temperatures.

As the water temperature gradually decreased, the amount of effluent COD_{Mn} slightly increased. When the water temperature dropped rapidly from 22.4 °C (on the 57th day) to 15.3 °C (on the 62nd day), the effluent COD_{Mn} increased from 3.88 to 5.19 mg L⁻¹, and the removal rate decreased from 57.32% to 40.82%. When the water temperature was above 20 °C, the water temperature had little impact on the removal efficiency with respect to COD_{Mn} . However, when the water temperature suddenly dropped to approximately 15.3 °C, the removal efficiency with regard to COD_{Mn} decreased markedly. Then, with the adaptation of heterotrophic bacteria to the water temperature and the recovery of the water temperature, the removal rate regarding COD_{Mn} considerably increased to approximately 50%. When the water temperature dropped to approximately 16 °C (from the 91st day to the 99th day), the removal rate with respect to COD_{Mn} fluctuated between 41.26% and 44.33%, and the average rate of removal was 42.55%. When the water temperature suddenly dropped to 10.6 °C and 4.8 °C (on the 106th day and 122nd day), the effluent COD_{Mn} levels increased to 7.49 mg L⁻¹ and 7.74 mg L⁻¹, respectively, and the removal rate in terms of COD_{Mn} decreased to 25.43% and 10.21%, respectively. With the adaptation of heterotrophic bacteria to water temperature, the COD_{Mn} removal efficiency tended to stabilize. When the water temperature was approximately 6–8 °C (day 155–day 166), the concentration of COD_{Mn} in the effluent was 6.35–8.32 mg/L, which exceeded the national drinking water standard. The COD_{Mn} removal rate was 17.61–23.02%, and the average rate of removal was 20.48%. Water temperature is an important factor affecting the removal efficiency regarding COD_{Mn} because a decrease in water temperature inhibits the activity of heterotrophic bacteria. When the water temperature suddenly dropped to approximately 15 °C, 10 °C, and 5 °C, the removal rate in relation to COD_{Mn} decreased markedly. BMOs produced in a biological filter column can effectively improve removal efficiency in relation to COD_{Mn} .

3.1.2. Influence of Water Temperature on Ammonia-N Removal Efficiency

When the water temperature in the biological filter column was approximately 24 °C (day 1~day 21), the influent ammonia-N concentration was 1.35~1.61 mg L⁻¹. The mature filter layer contained many nitrifying bacteria, which could completely oxidize the influent ammonia-N. The effluent ammonia-N concentration was below 0.04 mg L⁻¹, and the ammonia-N removal rate was 97.72~99.17%. As illustrated in Figure 3, the average rate of removal was 98.40%. The biologically aerated filter presented a good removal effect with respect to ammonia-N, and this efficiency was ascribed to the synergistic oxidation of nitrifying bacteria and BMOs. When the water temperature dropped to 20~24 °C, the nitrification rate of ammonia-N was not affected, and the effluent ammonia-N concentration did not increase. Then, as the water temperature dropped below 20 °C, the ammonia-N in the effluent increased markedly. When the water temperature was approximately 16 °C (day 91 to day 99), the effluent ammonia-N significantly increased to 0.050~0.083 mg L⁻¹, and the average removal rate of ammonia-N was 95.58%.

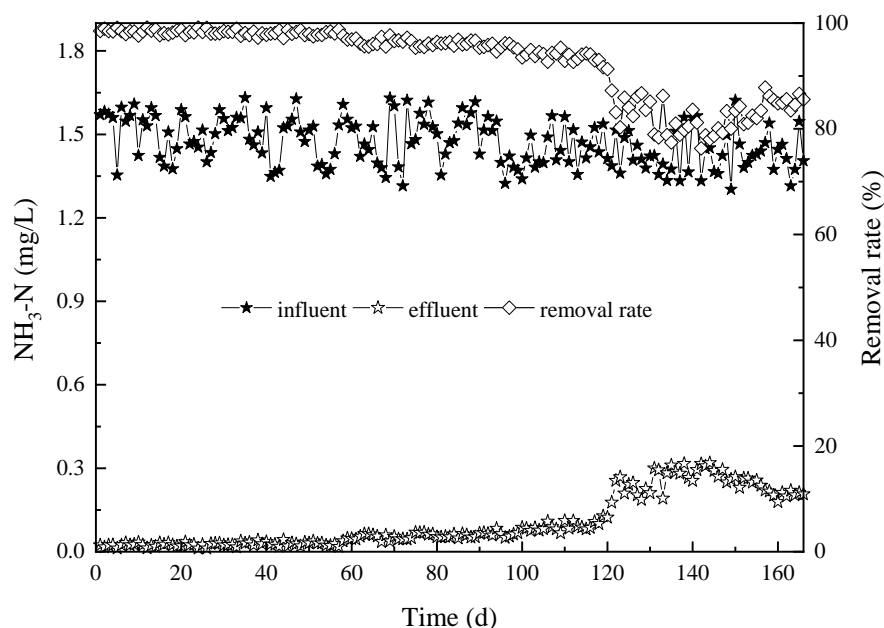


Figure 3. Ammonia removal efficiency of the biological aerated filter at different water temperatures.

When the water temperature suddenly dropped to 10.6 °C and 4.8 °C on the 106th and 122nd days, the ammonia-N levels in the effluent increased to 0.11 mg L⁻¹ and 0.26 mg L⁻¹, respectively, and the removal rate of ammonia-N decreased to 92.71% and 83.14%, respectively. The activity of nitrifying bacteria was significantly inhibited. However, as the nitrifying bacteria acclimated to the changes in water temperature, the removal efficiency with respect to ammonia nitrogen showed a slight improvement and eventually stabilized. When the water temperature was approximately 6~8 °C (day 155~day 166), the effluent ammonia-N concentration was 0.18~0.26 mg L⁻¹, the average concentration was 0.21 mg L⁻¹, and the average removal rate was 85.04%. When the water temperature was 20~24 °C, the nitrification rate of ammonia-N was not affected. When the water temperature dropped to approximately 15 °C, 10 °C, and 5 °C, the nitrification rate of ammonia-N was markedly inhibited, and the removal rate of ammonia-N was markedly reduced. As the water temperature dropped, although the removal rate of ammonia-N markedly decreased, the ammonia-N concentration in the effluent was always less than 0.5 mg L⁻¹, which is less than the national drinking water standard. Wang et al. [19–21] used a biological filter column to remove ammonia-N, iron, and manganese in groundwater when the water temperature was approximately 8 °C. The experimental results showed that the ammonia-N concentration in the effluent was less than 0.2 mg L⁻¹. Nitrifying bacteria exhibit high activity at low temperatures and can oxidize ammonia-N.

3.1.3. Influence of Water Temperature on Manganese Removal Efficiency

The influent concentration of manganese was $0.94\text{--}1.12\text{ mg L}^{-1}$. As illustrated in Figure 4, When the water temperature was approximately $24\text{ }^{\circ}\text{C}$ (day 1~day 21), the biologically aerated filter presented a good removal effect with respect to manganese. This was because a mature filter column was used in this experiment, and many manganese-oxidizing bacteria were attached to its filter layer. Under the catalysis of extracellular enzymes produced by manganese-oxidizing bacteria, bivalent manganese in an influent is oxidized into high-valence BMOs, thus allowing for the removal of bivalent manganese [22,23]. BMOs exhibit strong oxidizing ability and can catalyze the oxidation of organic masses and ammonia-N, thereby improving removal efficiency with respect to COD_{Mn} and ammonia-N. The concentration of Mn in the effluent dropped below 0.02 mg L^{-1} , and the removal rate of manganese was $98.09\text{--}99.27\%$, with an average removal rate of 98.70% .

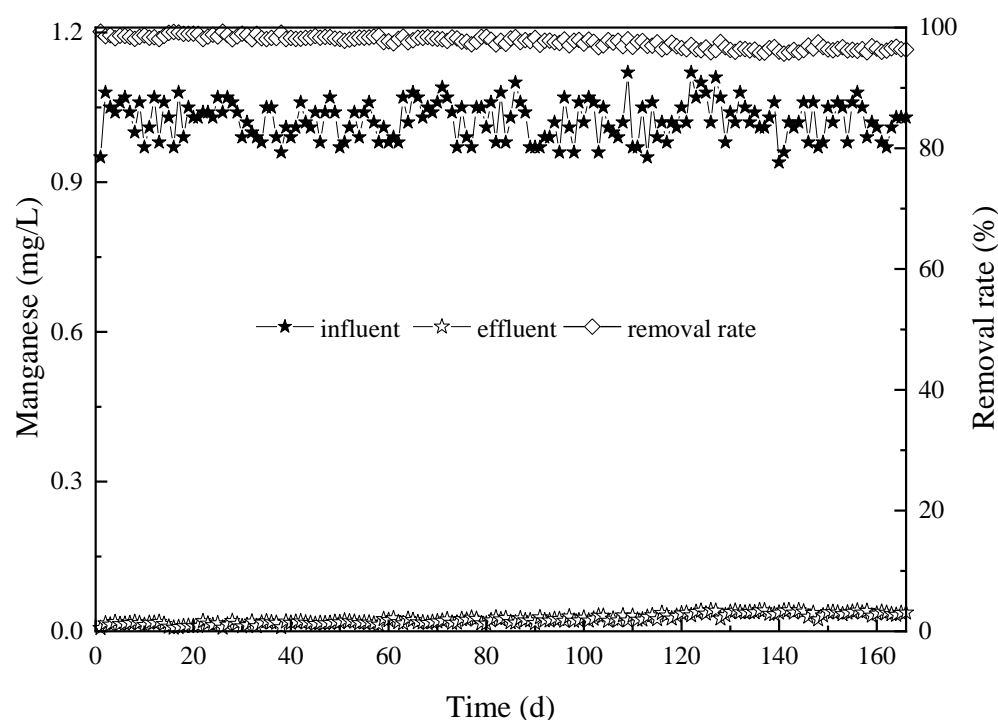


Figure 4. Manganese removal efficiency of the biological aerated filter at different water temperatures.

When the water temperature gradually dropped to approximately $15\text{ }^{\circ}\text{C}$ (from the 91st day to the 99th day), the manganese in the effluent slightly rose to $0.021\text{--}0.030\text{ mg L}^{-1}$, and the removal rate dropped to $97.06\text{--}98.04\%$, with an average removal rate of 97.63% . When the water temperature suddenly dropped to $10.6\text{ }^{\circ}\text{C}$ and $4.8\text{ }^{\circ}\text{C}$ on the 106th and 122nd days, the manganese in the effluent did not decrease markedly. This was because a manganese sand filter material was used in this experiment, which has a strong adsorption capacity for manganese [24,25] and can adsorb unoxidized divalent manganese. When the water temperature was approximately $6\text{--}8\text{ }^{\circ}\text{C}$ (day 155~day 166), the manganese content in the effluent was $0.031\text{--}0.042\text{ mg L}^{-1}$, which is less than the 0.1 mg L^{-1} national drinking water standard. The average concentration was 0.037 mg L^{-1} , and the average removal rate was 96.38% . When the test water temperature was approximately $8\text{ }^{\circ}\text{C}$, the biological filter column reduced the iron, Mn, and ammonia-N concentrations in actual groundwater to less than 0.1 mg L^{-1} , 0.05 mg L^{-1} , and 0.1 mg L^{-1} , respectively [25]. When the water temperature was $18\text{--}22\text{ }^{\circ}\text{C}$, this biofilter also had a good removal effect with respect to iron, Mn, and ammonia-N in simulated groundwater [26]. Manganese-oxidizing bacteria can not only oxidize divalent manganese at a constant water temperature but also oxidize divalent manganese at a fluctuating water

temperature (4–25 °C). This shows that manganese-oxidizing bacteria are highly adaptable to water temperature and that biological manganese (BIO-Mn) removal can be applied in treatment plants treating groundwater containing Mn in different areas and at different water temperatures [27]. In terms of potency, the effect of water temperature on the removal efficiency with respect to COD_{Mn} , ammonia-N, and Mn corresponded to the following order: $\text{COD}_{\text{Mn}} > \text{ammonia-N} > \text{Mn}$.

3.2. Change and Kinetic Analysis of COD_{Mn} , Ammonia-N, and Mn during the Experiment

3.2.1. Change in COD_{Mn} Levels along the Treatment Path and Kinetic Analysis

On the 10th day, 93rd day, and 158th day, the water temperatures were 23.4 °C, 15.6 °C, and 8.3 °C, respectively. The COD_{Mn} concentrations in the influent were 9.38, 9.05, and 9.24 mg L⁻¹, respectively (Figure 5). At 0.4 m along the filter layer, the concentration of COD_{Mn} decreased to 7.18, 7.49, and 8.66 mg L⁻¹, and at 0.8 m, it decreased to 5.55, 6.58, and 8.09 mg L⁻¹; the effluent COD_{Mn} was 3.54, 5.17, and 7.29 mg L⁻¹, respectively. COD_{Mn} was removed in all sections of the filter layer, and the removal amount was the largest in the 0–0.4 m section of the filter layer. This was because the biodegradability of organic mass in the lake water was poor, and a small amount of easily degradable organic mass was preferentially removed in the upper filter material. In the lower areas of the filter layer, the biodegradability of organic mass became increasingly worse, and the amount of COD_{Mn} removed gradually decreased. Water temperature had an obvious influence on the removal efficiency with respect to COD_{Mn} along the treatment path. With a decreasing water temperature, the concentration of COD_{Mn} along the treatment path gradually increased.

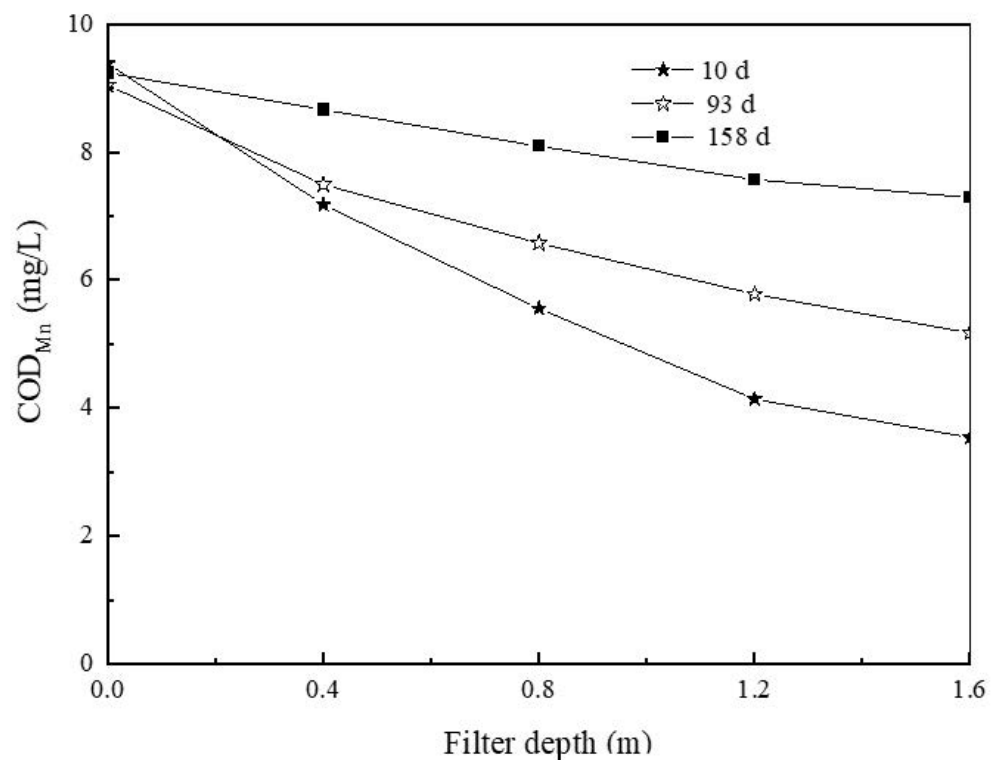


Figure 5. Variation in COD_{Mn} concentration along the filter layer at different water temperatures.

The results of a kinetic study of the pollutants in water can determine the time required to effectively remove a particular pollutant, and this reaction time can provide parameters for the design of pollutant treatment equipment. [28]. To better guide the design of treatment plants for the treatment of water containing COD_{Mn} and ammonia-N, the kinetic characteristics of the removal of COD_{Mn} , ammonia-N, and Mn by biological aerated filters at different water temperatures were studied by taking simulated lake water

containing COD_{Mn} and ammonia-N as a research object. When the water temperatures were 23.4, 15.6, and 8.3 °C, the biological removal of COD_{Mn} was consistent with first-order reaction kinetics (Table 2). The values of the kinetic constant k were 0.0328, 0.0167, and 0.00790 min⁻¹, and the values of the reaction half-life $t_{1/2}$ were 21.13 min, 41.51 min, and 84.74 min, respectively. The reaction half-life regarding the biological removal of COD_{Mn} was greatly prolonged with a decreasing water temperature, which indicated that the activity of heterotrophic microorganisms was very sensitive to water temperature, and the removal effect of COD_{Mn} was easily affected by water temperature.

Table 2. Linear regression analysis of biological COD_{Mn} removal in biological aerated filters at different water temperatures.

Water Temperature (°C)	Kinetic Constant k (min ⁻¹)	Reaction Half-Life $t_{1/2}$ (min)	Formula	R ²
23.4	0.0328	21.13	$y = -0.0328 \times -0.00710$	0.997
15.6	0.0167	41.51	$y = -0.0167 \times -0.0522$	0.999
8.3	0.00790	87.74	$y = -0.00790 \times -0.00370$	0.997

3.2.2. Change along the Treatment Path and Kinetic Analysis of Ammonia-N

The change in the ammonia-N concentration along the filter layer of the biological aerated filter at different water temperatures is shown in Figure 6. On the 10th, 93rd, and 158th days, the influent ammonia-N concentrations were 1.42, 1.51, and 1.54 mg L⁻¹, respectively. On the 10th day, the water temperature was 23.4 °C, and the ammonia-N concentration rapidly decreased to 0.37 and 0.19 mg L⁻¹ at 0.4 and 0.8 m in the filter layer, respectively. Ammonia-N was mainly removed in the 0~0.4 m section of the filter layer. When the water temperature dropped to 15.6 °C and 8.3 °C, the ammonia-N concentration at 0.4 and 0.8 m in the filter layer rapidly increased to 0.50, 0.67, 0.25, and 0.46 mg L⁻¹, respectively, and the amount of ammonia-N removed at a depth of 0~0.4 m in the filter layer decreased markedly. Water temperature is an important factor affecting the nitrification effect of ammonia-N, and the concentration of ammonia-N along the treatment path increased markedly with a decreasing water temperature.

A first-order kinetic model was used to fit the biological removal of ammonia-N at different water temperatures (Table 3). The results showed that the nitrification of ammonia-N in the biological aerated filter was in accordance with first-order reaction kinetics with kinetic constants K of 0.110, 0.0995, and 0.0513 min⁻¹ and reaction half-lives $t_{1/2}$ of 6.32 min, 6.97 min, and 13.51 min, respectively. With a decreasing water temperature, the nitrification rate of ammonia-N decreased markedly, its kinetic constant K decreased markedly, and its $t_{1/2}$ was markedly prolonged.

3.2.3. Changes in Concentrations and Kinetic Characteristics of Manganese along the Treatment Path

On the 10th, 93rd, and 158th days, the concentrations of manganese in the influent were 0.97, 0.99, and 0.99 mg L⁻¹, respectively (Figure 7). The manganese concentration decreased to 0.49, 0.54, and 0.57 mg L⁻¹ at 0.4 m and to 0.12, 0.16, and 0.24 mg L⁻¹ at 0.8 m in the filter layer, respectively. Mn was mainly removed at 0~0.8 m of the filter layer, and its concentration along the treatment path increased with a decreasing temperature. The manganese concentration at 1.2 m in the filter layer decreased below 0.1 mg L⁻¹ and reached the drinking water standard. The effect of a decreasing water temperature on manganese was considerably less than that on COD_{Mn} and ammonia-N, which indicated that manganese-oxidizing bacteria have stronger adaptability to water temperature.

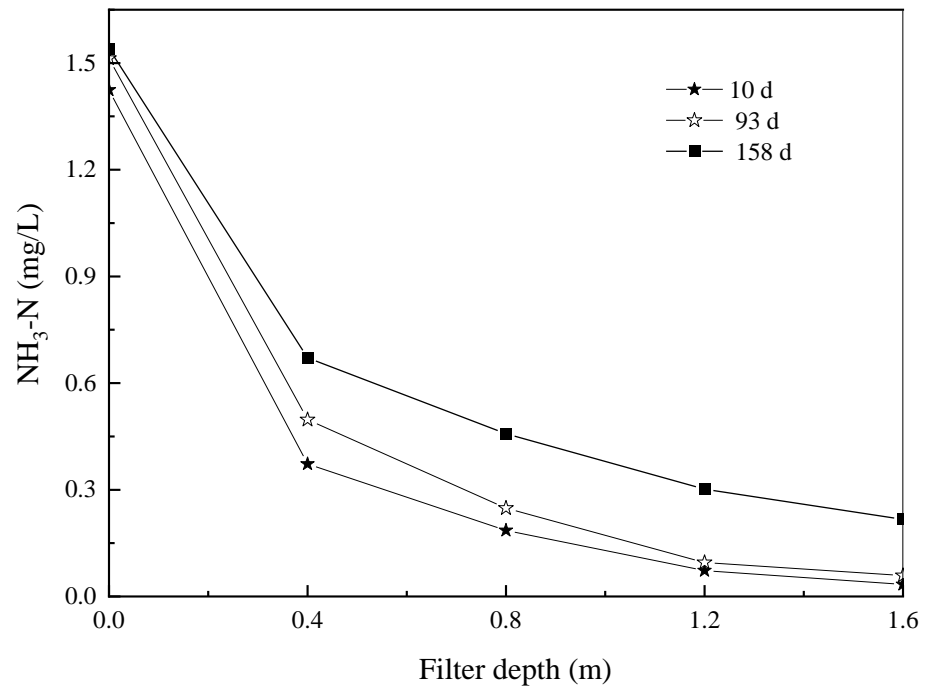


Figure 6. Variation in ammonia along the filter layer at different water temperatures.

Table 3. Linear regression analysis of biological ammonia removal in biological aerated filters at different water temperatures.

Water Temperature (°C)	Kinetic Constant k (min^{-1})	Reaction Half-Life $t_{1/2}$ (min)	Formula	R^2
23.4	0.110	6.32	$y = -0.110 \times -0.386$	0.993
15.6	0.0995	6.97	$y = -0.0995 \times -0.291$	0.995
8.3	0.0513	13.51	$y = -0.0513 \times -0.408$	0.999

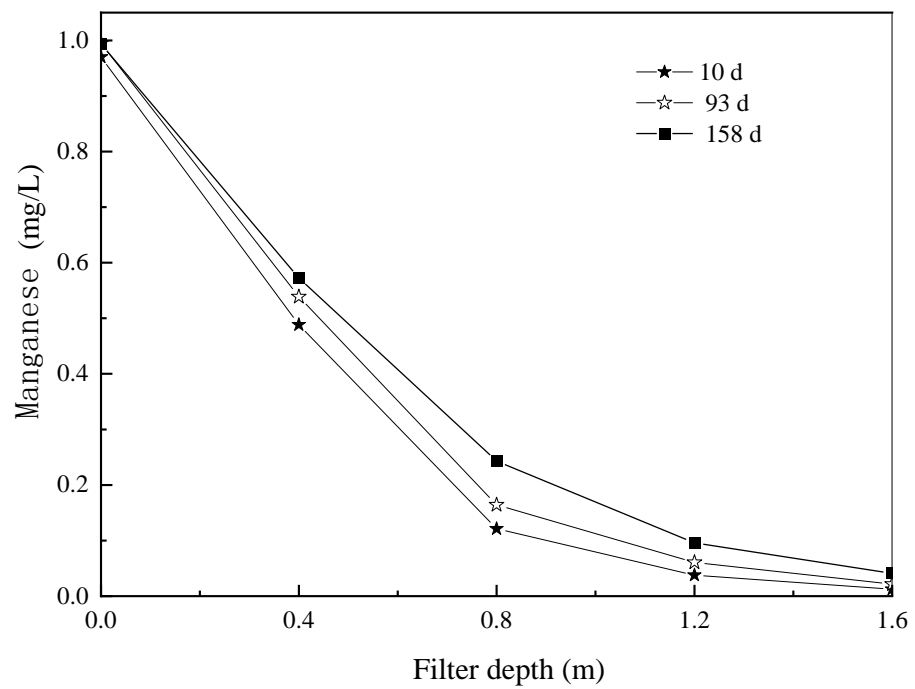


Figure 7. Variation in Mn along the filter layer at different water temperatures.

Liu et al. [23] and Zhang et al. [28] used biofilter columns to treat groundwater containing iron, manganese, and ammonia-N and studied the kinetic characteristics of BIO-Mn removal. The results showed that the nature of BIO-Mn removal was in accordance with first-order reaction kinetics (Table 4). In this research, a first-order kinetic model was used to fit the BIO-Mn removal results at different temperatures. The results showed that BIO-Mn removal in the biological aerated filter was also consistent with first-order reaction kinetics. The kinetic constants (k) of BIO-Mn removal were 0.164 min^{-1} , 0.144 min^{-1} , and 0.119 min^{-1} , and the reaction half-lives ($t_{1/2}$) were 4.22 min, 4.83 min, and 5.82 min, respectively. With a decreasing water temperature, the kinetic constant K gradually decreased, and the reaction half-life $t_{1/2}$ gradually extended. Compared with the literature [23,28,29], the kinetic constant K in this study was smaller, and the half-reaction time $t_{1/2}$ was longer. For example, the kinetic constant K reported by Cheng et al. [29] was 0.687, and the half-reaction time $t_{1/2}$ was only 1.010 min, which was due to the smaller filter material particle size (0.8~1 mm) and the constant water temperature. When the water temperature decreased, the activities of microorganisms (such as heterotrophic bacteria, nitrifying bacteria, and manganese-oxidizing bacteria) were inhibited. Manganese-oxidizing bacteria have stronger adaptability to water temperature changes, so the influence of water temperature changes on BIO-Mn removal was less than that of organic mass oxidation and ammonia-N nitrification.

Table 4. Linear regression analysis of biological manganese removal.

Water Temperature (°C)	Kinetic Constant k (min^{-1})	Reaction Half-Life $t_{1/2}$ (min)	Formula	R^2
23.4	0.164	4.22	$y = -0.164 \times +0.612$	0.998
15.6	0.144	4.83	$y = -0.144 \times +0.540$	0.997
8.3	0.119	5.82	$y = -0.119 \times +0.451$	0.996

4. Conclusions

After BMOs were generated in the biological aerated filter, the removal effect in relation to COD_{Mn} markedly improved. When the water temperature dropped from approximately 24 °C to approximately 6~8 °C, the concentrations of COD_{Mn} , ammonia-N, and manganese in the effluent of the biological aerated filter increased greatly, markedly, and slightly, respectively. With a sudden decrease in water temperature, the activities of heterotrophic bacteria and nitrifying bacteria that oxidize organic mass were markedly inhibited. When these microorganisms adapted to the water temperature, the removal efficiency with respect to COD_{Mn} and ammonia-N markedly increased.

The analysis of the results along the treatment path showed that COD_{Mn} was markedly removed in each section of the filter layer, and the amount removed gradually decreased when moving along the filter layer. Ammonia-N was mainly removed in the 0~0.4 m section of the filter layer, while manganese was mainly removed in the 0~0.8 m section of the filter layer. With a decreasing water temperature, the concentrations of COD_{Mn} , ammonia-N, and manganese increased.

Kinetic analysis showed that the biological removal of COD_{Mn} , BIO-Mn removal, and the nitrification of ammonia-N all adhered to first-order reaction kinetics. When the water temperatures were 23.4, 15.6, and 8.3 °C, the kinetic constants K of COD_{Mn} , ammonia-N, and manganese were 0.0328, 0.0167, 0.00790, 0.110, 0.0995, and 0.0513 min^{-1} , respectively. The reaction half-lives $t_{1/2}$ were 21.13, 41.51, 87.74, 6.32, 6.97, 13.51, 4.22, 4.83, and 5.82 min, respectively. With a decreasing water temperature, the kinetic constant K gradually decreased, and the reaction half-life $t_{1/2}$ gradually extended.

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