

## Design of fine-bubble aeration systems for municipal WWTPs with high sea salt concentrations

S. Sander, J. Behnisch and M. Wagner

### ABSTRACT

The impact of sea salt on the aeration coefficient  $k_L a$  of fine-bubble aeration systems was investigated in clean water and in a model substance for simulating activated sludge. The investigations were carried out in laboratory and pilot scale. Sea salt was dosed in the form of artificial seawater, real seawater or simplified, by adding NaCl. The investigated media showed an increase of  $k_L a$  with rising sea salt concentrations up to 10 g/L. The ratio between  $k_L a$  and sea salt concentration is impaired by the air flow rate and the type of applied diffusers. Literature data shows, that the  $k_L a$  favourable effect of sea salt can also be found in the activated sludge of conventional municipal wastewater treatment plants. The results of this study allow the derivation of salt correction equations to be used in the design of fine-bubble aeration systems for taking into account the favourable impact of sea salt on  $k_L a$ . By that means, a more precise design of respective systems will be enabled.

**Key words** | aeration systems, design, municipal wastewater, oxygen transfer, saline wastewater

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### INTRODUCTION

On wastewater treatment plants (WWTPs), mainly fine-bubble aeration systems are used nowadays. The decisive design parameter with this type of aeration systems is the  $\alpha$ -value. It defines the ratio between the aeration coefficient  $k_L a$  in activated sludge and clean water. The  $\alpha$ -value is primarily dominated by the influence of surfactants (Wagner & Popel 1996) and the volume of activated sludge flocs (Henkel *et al.* 2009a). Generally, these effects negatively affect  $k_L a$  in activated sludge compared to clean water, so that the  $\alpha$ -value is normally set between 0.35 and 0.9 in the design of fine-bubble aeration systems (Wagner & Stenstrom 2015).

Most salts positively influence  $k_L a$  with fine-bubble aeration systems. This is valid for many single salts, but also mixed salt solutions (Henry *et al.* 2007) e.g. seawater (Cheuk 2006; Kawahara *et al.* 2009).  $k_L a$  is the product of the liquid-side mass transfer coefficient  $k_L$  and the liquid/gas interfacial area  $a$ . In most fine-bubble applications, with increasing sea salt concentration the  $k_L$ -value decreases slightly while the  $a$ -value increases significantly, resulting in a net increase of  $k_L a$  and  $\alpha$ . The effect of salts on the  $a$ -value is due to the inhibition of bubble coalescence, which is taking place in clean water (Deckwer *et al.* 1974;

Zlokarnik 1980; Cheuk 2006; Kawahara *et al.* 2009; Al Taweel *et al.* 2013; Baz-Rodriguez *et al.* 2014). Bubble coalescence describes the merging of two single bubbles after a collision into one larger bubble.

As the total concentration of dissolved salts  $S_{TDS}$  does not exceed 1 g/L in most municipal wastewaters (Tchobanoglous *et al.* 2014), the positive effect of most salts on the  $\alpha$ -value is not accounted for in the internationally applied design approaches for fine-bubble aeration systems (cf. DWA 2013; Tchobanoglous *et al.* 2014; Wagner & Stenstrom 2015). However, there are some case studies about municipal WWTPs that show significantly higher  $S_{TDS}$  concentrations: due to the application of seawater for toilet flushing, chlorine concentrations of 3.5–6.5 g/L have been measured in Hong Kong (Yu *et al.* 2002). Calculated as pure NaCl, these values correspond to  $S_{TDS}$  concentrations of 5.8–10.7 g/L. Wagner & Sander (2015) measured electrical conductivity values of more than 10,000  $\mu S/cm$  ( $\sim 6.7$  g  $S_{TDS}/L$ ) in activated sludge at a municipal WWTP in Southern China. Here, high  $S_{TDS}$  concentrations were caused by seawater intrusion into leaky sewers. The  $\alpha$ -values determined at that WWTP, exceeded a value of 1 and were therefore far higher than the above

described typical design  $\alpha$ -values with low  $S_{TDS}$  concentrations. It is assumed that these high  $\alpha$ -values are caused by the coalescence-inhibiting impact of sea salt in the plant influent. Design approaches for fine-bubble aeration systems do not include the effects of sea salts on the aeration coefficient, and are therefore oversizing the respective systems. Unnecessary investment and operational costs are the result.

The objective of the present study is to facilitate a more precise design of aeration systems for municipal WWTPs with high sea salt concentrations. Therefore, oxygen transfer tests were carried out to examine the effects of sea salts on  $k_{La}$ . From the collected data, salt correction equations were derived to be used in the design of aeration systems.

## MATERIALS AND METHODS

### Oxygen transfer test method

Oxygen transfer tests were performed using the desorption method via the application of pure oxygen in batch tests as described in [Wagner \*et al.\* \(1998\)](#). Dissolved oxygen in water was measured by means of up to four electrochemical oxygen sensors (Orbisphere, Hach Lange, Germany or Oxymax COS51D, Endress + Hauser, Germany). The software OCA (Fröse & Olderdissen, Germany) was used to calculate  $k_{LaT}$  values ( $k_{La}$  values at temperature  $T$ ) from the recorded signals according to the non-linear regression method. As described in [DWA \(2013\)](#), the  $k_{LaT}$  values were then converted to 20 °C according to the following equation:  $k_{La20} = k_{LaT} \cdot 1,024^{(20-T)} (1/h)$ . The air flow rate was measured with thermal flow sensors TA (Hoentzsch, Germany) of different sizes.

### Laboratory-scale experiments in clean water with artificial seawater

In order to describe the influence of sea salt on  $k_{La20}$  in clean water, desorption tests were carried out using a cylindrical test column made of acrylic glass with a diameter of 0.3 m and a water depth of 3 m. The column was equipped with a disc diffuser. Oxygen transfer tests were performed in clean water and at  $S_{TDS}$  concentrations of up to 27 g/L. Corresponding to the ionic composition in real seawater (e.g. [Xue \*et al.\* 2000](#)), artificial sea salt with a composition of 58.0% NaCl, 9.8%  $Na_2SO_4$ , 26.6%  $MgCl_2 \cdot 6 H_2O$ , 3.8%  $CaCl_2 \cdot 2 H_2O$ , and 1.8% KCl (percentages stated in mass-%) was used for the tests. For each  $S_{TDS}$  concentration,

oxygen transfer tests were conducted with at least three different target air flow rates (0.5, 1.0, and 1.5 m<sup>3</sup>/(diffuser · h)).

### Pilot-scale experiments in clean water with real seawater

In order to model the operating parameters of water depth and diffuser density (ratio between the surface of the diffusers and the floor area of the reactor) to full-scale conditions, additional oxygen transfer test were carried out using a pilot-scale test reactor. The test reactor showed a square base floor of 1.2 m × 1.2 m and a water depth of ~6 m. Three different diffuser densities were tested: 5%, 11%, and 28%. To do this, the required number of disc diffusers (1, 2, and 5), which are attached to a frame, were evenly distributed across the base floor of the reactor. Oxygen transfer tests were performed in clean water and at target  $S_{TDS}$  concentrations of 2, 4, 6, 8, and 10 g/L. To adjust  $S_{TDS}$ , real seawater from the South China Sea was mixed with local tap water. At each test setting,  $S_{TDS}$  was determined according to [APHA \(2005\)](#). Up to four different air flow rates were investigated for each test setting. The selected air flow rates were in the range between 1 and 12 m<sup>3</sup>/(diffuser · h), as recommended by the manufacturer. For technical reasons, at a diffuser density of 28% no higher total air flow rates than ~6.6 m<sup>3</sup>/(diffuser · h) could be achieved.

### Laboratory-scale experiments in a model substance to simulate activated sludge with NaCl

In this oxygen transfer test series, the effect of coalescence-inhibiting salts on  $k_{La}$  in the presence of further  $k_{La}$  active substances was investigated. This was done to transfer the results from the experiments in clean water with sea salt (see above) to the design approaches for fine-bubble aeration systems. As a further  $k_{La}$  active substance and model substance for activated sludge, iron hydroxide flocs as suggested by [Henkel \*et al.\* \(2009a\)](#) were used. In comparison to real activated sludge, it shows a decisive benefit: in real activated sludge,  $k_{La}$  is influenced by numerous effects (cf. introduction), which vary with time. Therefore, the net effect of activated sludge and its constituents on  $k_{La}$  at a specific time is hardly predictable. Iron hydroxide flocs as a biologically inactive substance do not show this disadvantages since the volume of their flocs remains stable with time.

Iron hydroxide flocs were prepared according to Henkel *et al.* (2009a): Trivalent iron chloride was dissolved in deionized water in a 500 L storage container. This strongly acidic generated solution was then dosed with sodium hydroxide until the pH-value was  $\sim 8$ . This way, iron hydroxide flocs precipitate, and dissolved NaCl is formed. In order to reduce  $S_{TDS}$ , iron hydroxide flocs were sedimented in the storage container, and the solids-free supernatant was replaced by deionized water. In order to increase  $S_{TDS}$ , pure NaCl was dosed. By using deionized water and pure NaCl, it was ensured that no other salts besides NaCl were dissolved. To determine the floc volume of iron hydroxide, a 1 L sample was taken and stored in a settling column (7 cm diameter). The volume of the iron hydroxide was measured when the sedimentation process of the sample was finished ( $\sim 48$  h). The measured value is termed the hydrostatic floc volume (HFV).

Oxygen transfer tests were conducted in a cylindrical test column made of acrylic glass with a diameter of 0.43 m and a water depth of 1 m. The column was equipped with a disc diffuser. Oxygen transfer tests were performed in clean water and at four different target  $S_{TDS}$  concentrations (0.5, 2.5, 5.0, and 7.5 g/L). For each  $S_{TDS}$  concentration, oxygen transfer measurements were carried out in the test column at three different target HFV concentrations (150, 300, and 450 mL/L). For each  $S_{TDS}$  resp. HFV test setting, oxygen transfer tests were conducted in the test column at three different air flow rates (0.5, 1.0, and 1.5 m<sup>3</sup>/(diffuser · h)) as double measurements. This way,  $k_L a$ -values in iron hydroxide flocs at different  $S_{TDS}$  concentrations were determined.

Following each test setting with iron hydroxide flocs and the respective  $S_{TDS}$  concentration, the sole effect of the dissolved salts on  $k_L a$  was determined. To do this, the iron hydroxide flocs were removed from the test column and returned to the storage container to settle. After sedimentation, the solids-free but saline supernatant was put back into the test column. Oxygen transfer tests were carried out in the supernatant at the same air flow rates as with the iron hydroxide flocs.  $S_{TDS}$  was determined in the iron hydroxide flocs and in the supernatant according to APHA (2005).

### Calculating the effect of sea salt on $k_L a$

The effect of sea salt on  $k_L a_{20}$  is expressed in this study as the ratio between  $k_L a_{20}$  in saline water and in clean water, determined in the same reactor with the same configuration.

The following applies:

$$f_S = \frac{k_L a_{20, \text{saline water}}}{k_L a_{20, \text{clean water}}}$$

According to DWA (2013), the  $\alpha$ -value is defined as the ratio between  $k_L a_{20}$  in activated sludge and in clean water. In this study, it is assumed that salts only contribute a negligible part of the  $\alpha$ -value. The following applies:

$$\alpha = \frac{k_L a_{20, \text{non-saline activated sludge}}}{k_L a_{20, \text{clean water}}}$$

In this study, oxygen transfer tests were also performed in a model substance for saline activated sludge. The salt effect on  $k_L a_{20}$  in this substance is described by the following equation:

$$\alpha f_S = \frac{k_L a_{20, \text{saline activated sludge}}}{k_L a_{20, \text{clean water}}}$$

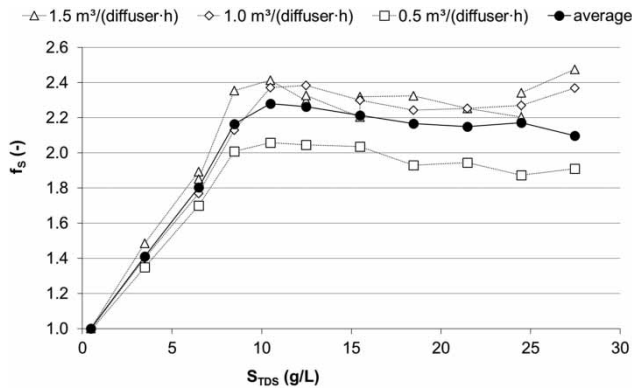
The parameters  $f_S$  and  $\alpha f_S$  are used in this study to describe the effect of sea salt on  $k_L a_{20}$  in different media. Until now, they have not been introduced to the design approaches of aeration systems.

As the air flow rate has a considerable influence on  $k_L a_{20}$ , the two  $k_L a_{20}$  values which are compared to each other in the equations have to relate to the same air flow rate. For technical reasons, it was not possible to conduct the oxygen transfer tests at exactly the same air flow rate. Therefore, the  $k_L a_{20}$ -values of the clean water tests were converted to the air flow rates applied in the oxygen transfer tests in saline/activated sludge media via a linear trend line using at least three determined  $k_L a_{20}$ /air flow rate pairs in clean water. This procedure was applied because  $k_L a_{20}$  was increasing linearly with the air flow rate in clean water. A comparable procedure to convert  $k_L a_{20}$ -values has been described by (Henkel *et al.* 2009b).

## RESULTS AND DISCUSSION

### Effect of sea salt in clean water

Figure 1 shows the  $f_S$ -values determined in the laboratory-scale experiments at different air flow rates as a function of  $S_{TDS}$ . For all investigated air flow rates, the following applies: due to the inhibition of bubble coalescence and



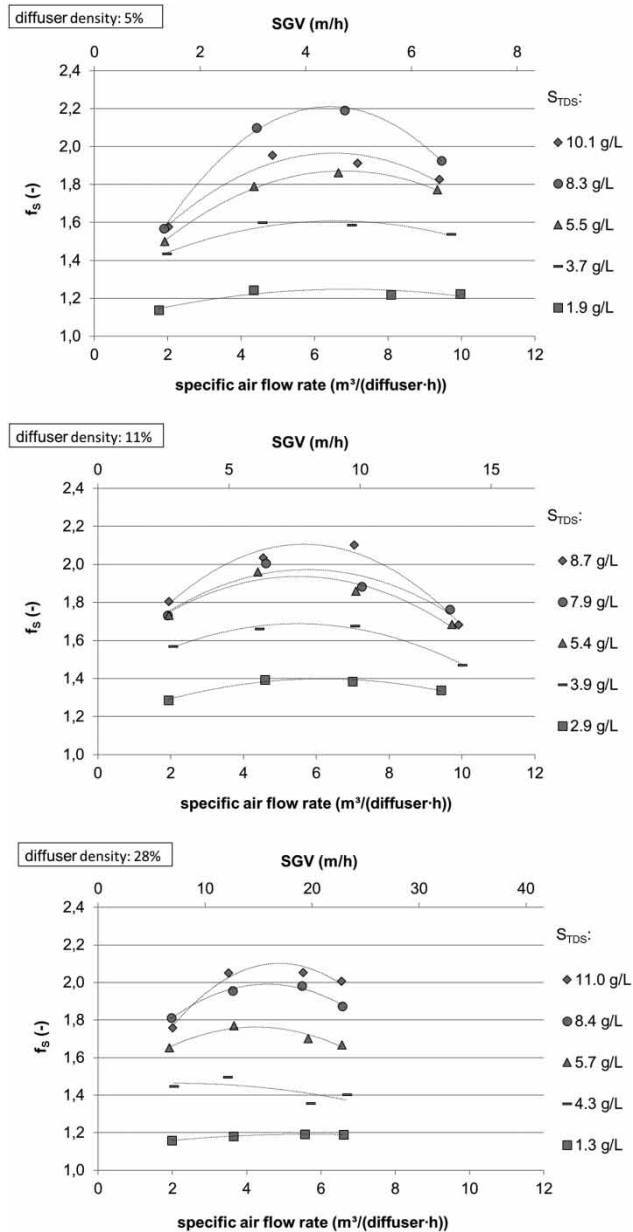
**Figure 1** |  $f_s$  as a function of  $S_{TDS}$  using artificial sea salt for different specific air flow rates.

rising  $a$ -values,  $f_s$ -values increase linearly up to  $S_{TDS}$  10 g/L. Here, the  $f_s$ -value reaches its maximum value ( $f_{s,max}$ ), and bubble coalescence is completely inhibited by the influence of the salts. At  $S_{TDS} > 10$  g/L,  $f_s$ -values slightly decrease again. This effect can be attributed to the negative effect of sea salt on  $k_L$ . However, this effect is quite low in comparison to the effect of  $S_{TDS}$  on the  $a$ -value in the tested  $S_{TDS}$  range of up to 27 g/L. The  $f_s$ -values determined at the low specific air flow rate of  $0.5 \text{ m}^3/(\text{diffuser} \cdot \text{h})$  were significantly lower than those determined at the two other air flow rates. The dependency between  $f_s$  and air flow rate is discussed below in context with the pilot-scale investigations.

In Figure 2,  $f_s$ -values determined in the pilot-scale experiments are presented as a function of the specific air flow rate and the superficial gas velocity (SGV) for the three tested diffuser densities of 5%, 11%, and 28%. The ratio between the specific air flow rate and SGV, as illustrated in the x-axes of Figure 2, depends on the diffuser density and therefore varies from diagram to diagram.

In general, with all three diffuser densities, the results from the laboratory-scale experiments were confirmed:  $f_s$  increases almost linearly with rising  $S_{TDS}$  concentrations up to  $\sim 10$  g/L. As an exception, the 5% diffuser density showed the highest  $f_s$ -values at  $S_{TDS}$  8.3 g/L. At the highest investigated  $S_{TDS}$  concentrations of 10.1 g/L, the  $f_s$ -value decreased again.

With all three diffuser densities, the highest  $f_s$ -values for one certain  $S_{TDS}$  concentration were determined in the medium range of specific air flow rates ( $\sim 5 \text{ m}^3/(\text{diffuser} \cdot \text{h})$ ). This becomes particularly evident with the higher  $S_{TDS}$  concentrations. The observation can be explained by factors, other than the bubble diameter, that influence  $f_s$ : (1) Low  $f_s$ -values at low air flow rates (here



**Figure 2** |  $f_s$  in the pilot-scale test plant as a function of air flow rate, utilizing real sea-water; diffuser densities: 5% (above), 11% (centre), 28% (below).

$\sim 2 \text{ m}^3/(\text{diffuser} \cdot \text{h})$ ) can be attributed to the isolated bubble rise behaviour at those conditions. Here, bubble coalescence does also occur less frequently in clean water (Baz-Rodriguez *et al.* 2014). (2) Low  $f_s$ -values at high air flow rates (here  $\sim 10 \text{ m}^3/(\text{diffuser} \cdot \text{h})$ ) can be attributed to the interactions of air flow rate, frequency of bubble formation, and primary bubble diameter (bubble diameter at the moment that the bubbles leave the diffuser). When the air flow rate increases, there is a specific stage where the frequency of bubble formation largely

remains constant and the primary bubble diameter increases (Kumar & Kuloor 1970). In coalescence inhibited systems, where air bubbles can remain at the primary bubble diameter during the entire ascent,  $f_S$  is thus reduced at air flow rates beyond this stage.

In this study,  $f_{S,max}$  was reached at  $S_{TDS} \sim 10$  g/L.  $f_{S,max}$  ranged between 1.8 and 2.5 (lab-scale experiments) and 1.6 and 2.2 (pilot-scale experiments). Comparable observations under comparable conditions in lab scale have been made by Cheuk (2006). A review of literature shows that this pattern does not necessarily have to apply at different conditions. Some examples are given:

The smaller the bubble diameter, the more salt is needed for complete inhibition of bubble coalescence (Tsang *et al.* 2004; Firouzi *et al.* 2015), i.e. when diffusers generating different bubble diameters than those in this study are used, complete bubble coalescence might also occur at sea salt concentrations higher or lower than 10 g/L.

The total value of  $f_{S,max}$  varies with the type of aeration system. Higher  $f_{S,max}$ -values are observed at systems generating smaller primary bubbles (Zlokarnik 1980). Therefore, it is obvious that in studies with micro-bubble generators, higher  $f_S$ -values than in the present study have been seen (Kawahara *et al.* 2009; Al Taweel *et al.* 2013) while in studies with nozzles, porous glass plates or perforated ring diffusers, slightly lower  $f_S$ -values than in the present study have been observed (Deckwer *et al.* 1974; Painmanakul & Hebrard 2008; Baz-Rodriguez *et al.* 2014). Ruen-ngam *et al.* (2008)

even found a negative effect of sea salt on  $k_La$ , resulting in  $f_S$ -values  $< 1$ . Those findings can be attributed to the use of an air-lift reactor in that particular study, where the coalescence behaviour thus completely differs from that in bubble columns.

However, the cited studies reflect conditions found at certain industrial applications but not at municipal WWTP. Therefore, for the further discussion only the data collected in the present study is used.

### Effect of NaCl in a model substance to simulate activated sludge

Table 1 summarizes the results of the oxygen transfer tests conducted with the iron hydroxide flocs ( $\alpha f_S$ -values) and the supernatants of the flocs ( $f_S$ -values). Also  $\alpha$ -values, which were calculated by two different mathematical approaches (multiplicative and additive), are given. The table shows that with an increase of HFV, lower  $\alpha f_S$ -values are obtained at a certain  $S_{TDS}$  concentration. This can be attributed to the reduction of interfacial renewal between air and water caused by the volume of the iron hydroxide flocs (Henkel *et al.* 2009a). However,  $\alpha f_S$  improves significantly with an increase of  $S_{TDS}$  at all investigated HFV settings. This shows that the floc volume of activated sludge, which was modelled here by iron hydroxide flocs, does not cancel out the  $k_La$  favourable effect of sea salt resp. NaCl.

**Table 1** |  $\alpha f_S$ -values measured in iron hydroxide flocs and  $f_S$ -values measured in the corresponding supernatants (mean values and standard deviation in brackets,  $n = 18$ )

Test results iron hydroxide flocs			Test results supernatants		Calculated $\alpha$ -values	
HFV mL/L	$S_{TDS}$ g/L	$\alpha f_S$ –	$S_{TDS}$ g/L	$f_S$ –	$\alpha = (\alpha f_S)/f_S$ –	$\alpha = (\alpha f_S) - f_S + 1$ –
130	0.52	0.81 (0.02)	0.53	0.96 (0.02)	0.84	0.85
120	2.48	0.95 (0.02)	2.52	1.11 (0.04)	0.86	0.84
120	4.90	1.32 (0.03)	4.95	1.44 (0.07)	0.92	0.88
120	7.57	1.49 (0.06)	7.62	1.58 (0.07)	0.94	0.91
310	0.48	0.68 (0.04)	0.54	1.04 (0.02)	0.65	0.64
290	2.50	0.74 (0.03)	2.49	1.16 (0.03)	0.64	0.58
300	4.89	1.07 (0.05)	4.92	1.51 (0.09)	0.71	0.56
330	7.63	1.32 (0.09)	7.67	1.85 (0.14)	0.71	0.47
450	0.52	0.60 (0.06)	0.49	0.94 (0.04)	0.64	0.66
480	2.57	0.63 (0.05)	2.58	1.23 (0.02)	0.51	0.40
480	4.91	0.79 (0.04)	4.99	1.47 (0.03)	0.54	0.32
450	7.56	1.12 (0.06)	7.50	1.78 (0.04)	0.63	0.34

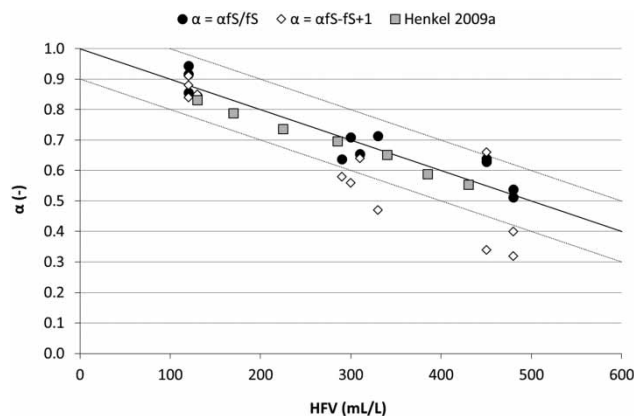
$\alpha$ -values calculated with mean values.



As can be seen in the table, the  $S_{TDS}$  concentrations in the iron hydroxide flocs and in the corresponding supernatants was almost the same. Also in the supernatants, the effect of NaCl on  $k_L a$  can be observed;  $S_{TDS}$  and  $f_S$  correlate positively with all observed supernatants. The standard deviations of the determined  $\alpha f_S$  and  $f_S$ -values tend to increase with the  $S_{TDS}$  concentration. This observation can be attributed to the growing influence of the air flow rate on  $k_L a$  with an increase of  $S_{TDS}$ .

The  $\alpha f_S$ -values determined in the iron hydroxide flocs describe the net effect of floc volume and NaCl on  $k_L a$ . The  $f_S$ -values determined in the supernatants represent the individual effect of NaCl. As besides floc volume and NaCl there were no further effects on  $k_L a$  in these experiments, the individual effect of the floc volume ( $\alpha$ -value) can be calculated via the determined data. In the table,  $\alpha$ -values were calculated by multiplicative ( $\alpha = (\alpha f_S)/f_S$ ) and additive linking ( $\alpha = (\alpha f_S) - f_S + 1$ ) of the determined mean  $\alpha f_S$  and  $f_S$ -values. It can be seen that with an increase of HFV and  $S_{TDS}$ , the  $\alpha$ -values calculated by the additive approach tend to be much lower than with the multiplicative approach. With the test setting HFV = 450 mL/L and  $S_{TDS} = 7.56$  g/L,  $\alpha$ -values calculated via the additive approach are only half of those calculated via the multiplicative approach.

Figure 3 illustrates the calculated  $\alpha$ -values of this study, literature data and the theoretical course of the  $\alpha$ -values, including a  $\pm 10\%$  confidence region as a function of HFV. The theoretical course is based on the theory that oxygen can only be introduced into a water body that contains free water. In the case of the free water content being 0 mL/L and HFV being 1,000 mL/L,  $k_L a$  is 0. The individual effect of floc volume HFV on  $\alpha$  can therefore be expressed via the equation:  $\alpha = -0.001 \cdot \text{HFV} + 1$  (Henkel et al.



**Figure 3** | Calculated  $\alpha$ -values for the iron hydroxide experiments, literature data and theoretical course of the  $\alpha$ -values with a  $\pm 10\%$  confidence region.

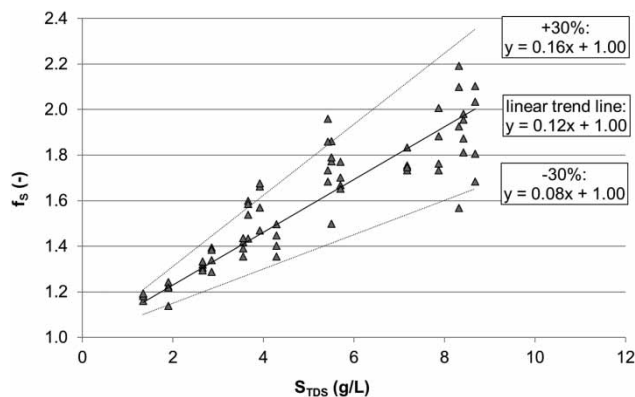
2009a). Across the complete investigated range and up to HFV 480 mL/L, the data calculated via the multiplicative approach follows the literature data more closely and stays within the  $\pm 10\%$  confidence region around the theoretical course.

In the design of aeration systems, the mathematical relation between  $\alpha$  and  $f_S$  is of particular interest. A discussion of the collected data shows that the net effect of sea salt, here simplified as NaCl, and the floc volume of activated sludge, here simplified as iron hydroxide flocs, can be expressed via multiplicative linking of the individual effects. In case the individual effects are calculated via additive linking, the influence of salt on the net effect is overestimated.

## Design recommendations

In order to include the positive effect of sea salt on  $k_L a$  in future design approaches of fine-bubble aeration systems, a correction equation for  $f_S$  was derived from the collected data. Only data collected in the clean water experiments in pilot-scale with real seawater was used for this purpose. This is because first, in these experiments fine-bubble diffusers made of elastomers, which are nowadays most frequently applied on municipal WWTPs, were used and second, because here the operating parameters diffuser density (5%–28%) and water depth ( $\sim 6$  m) were depicted in full-scale conditions. The relationship between  $f_S$  and  $S_{TDS}$  can be described as a linear function at  $S_{TDS} \leq 10$  g/L. Therefore, only data collected in this  $S_{TDS}$  range was selected for the derivation of the correction equation.

Figure 4 illustrates the selected  $f_S$ -values as a function of  $S_{TDS}$ . A linear trend line (solid line) is applied for the pairs of



**Figure 4** |  $f_S$ -values determined in pilot-scale investigations as a function of the  $S_{TDS}$  concentration; solid line indicates the linear trend line of the pairs of values; dotted lines indicate a confidence region of 30%.

values. For reasons of simplification, only two decimal digits are displayed for the trend line and the y-axis intercept of the trend line was set to 1. Due to the influence of the air flow rate, data scatter substantially around the trend line ( $R^2 = 0.778$ ). By multiplying the gradient of the trend line 0.12 with the factor 0.7 and 1.3 a confidence region of  $\pm 30\%$  was applied to the regression (dotted lines). Almost all pairs of values are within this confidence region.

In order not to oversize fine-bubble aeration systems for municipal WWTPs with high sea salt concentrations, in future it is recommended to include the  $f_{S,\alpha}$ -value into the equation used for the determination of the required standard oxygen transfer rate (SOTR) in design approaches. According to its definition, SOTR does not relate to a  $S_{TDS}$  concentration of 0 g/L, but to a manufacturer-specific value. This has consequences for the design: on the one hand, the effect of  $S_{TDS}$  under operating conditions has to be eliminated from the calculation of SOTR while on the other hand, SOTR has to be standardised to a manufacturer-specific value. Therefore, the design approach presented here distinguishes between  $S_{TDS}$  and  $f_S$  under operating conditions ( $S_{TDS,\alpha}$  and  $f_{S,\alpha}$ ) and under manufacturer-specific conditions ( $S_{TDS,st}$  and  $f_{S,st}$ ).

According to the design approach by DWA (2013) and by Wagner & Stenstrom (2015), aeration systems should be designed for different load cases. For the average load case 1, the average expected effect of  $S_{TDS}$  on  $f_S$  can be applied. At the maximum load case 2, where the upper capacity limit of the aeration system is defined, the effect of  $S_{TDS}$  should not be overestimated. Therefore, the lower limit of the confidence region should be applied for load case 2. The minimum load case 3 defines the lower capacity limit of the aeration system; therefore, the upper limit of the confidence region should be applied here. The load case-specific equations for considering the effect of the sea salt concentration under operational conditions  $S_{TDS,\alpha}$  on  $f_{S,\alpha}$  are:

$$f_{S,\alpha, \text{load case 1}} = 1 + 0.12 \cdot S_{TDS,\alpha} \quad (-)$$

$$f_{S,\alpha, \text{load case 2}} = 1 + 0.08 \cdot S_{TDS,\alpha} \quad (-)$$

$$f_{S,\alpha, \text{load case 3}} = 1 + 0.16 \cdot S_{TDS,\alpha} \quad (-)$$

The equations are valid for  $S_{TDS,\alpha} \leq 10$  g/L. In case  $S_{TDS,\alpha}$  exceeds 10 g/L,  $f_{S,\alpha, \text{load case 1}}$  has to be set at 2.2;  $f_{S,\alpha, \text{load case 2}}$  at 1.8 and  $f_{S,\alpha, \text{load case 3}}$  at 2.6. For reasons of simplification, the minor negative effect of  $S_{TDS}$  on  $k_L$  at  $S_{TDS} > 10$  g/L (cf. Figure 1) is not considered in the approaches.

Manufacturers generally determine the capacity of their systems in tap water with  $S_{TDS,st}$  concentrations  $< 1$  g/L. Therefore, the effect of  $S_{TDS,st}$  on SOTR is small and it not necessary to use load case specific equations for  $f_{S,st}$ . Simplified, the average expected effect of  $S_{TDS}$  on  $f_S$  can be applied for the calculation of  $f_{S,st}$ :

$$f_{S,st} = 1 + 0.12 \cdot S_{TDS,\alpha} \quad (-)$$

Transferring  $f_{S,\alpha}$  and  $f_{S,st}$  into the existing load case specific design approach, an extended equation for the calculation of SOTR is derived:

$$\text{SOTR} = \frac{f_d \cdot \beta_{St} \cdot C_{S,20} \cdot f_{S,st}}{\alpha \cdot f_{S,\alpha} \cdot \left( f_d \cdot \beta_{\alpha} \cdot C_{S,T} \cdot \frac{p_{\text{atm}}}{1.013} - C_X \right) \cdot \theta^{(T_w - 20)}} \cdot OV_h$$

(kg/h)

According to the findings with the model substance to simulate activated sludge,  $f_{S,\alpha}$  is multiplied with the  $\alpha$ -value in the equation. Abbreviations used in the equation are explained in Table 2. The extended SOTR equation may only be used in the design equation for fine-bubble aeration systems and only for WWTPs with (1) increased  $S_{TDS,\alpha}$  concentrations explicitly caused by seawater and (2) predominantly municipal wastewater. In addition, further surface-active substances should not occur in considerable concentrations in the activated sludge, as they might overlap the  $k_L a$  favourable effect of sea salt.

## CONCLUSIONS AND OUTLOOK

Sea salt, up to a  $S_{TDS}$  concentration of  $\sim 10$  g/L, has a positive effect on  $k_L a$  using fine-bubble diffusers. This applies to clean water, iron hydroxide flocs as a model substance to simulate activated sludge and – as literature shows – to real saline activated sludge from conventional municipal WWTPs. The relation between the sea salt concentration and  $k_L a$  depends also on the air flow rate and the type of applied diffuser. Overall, systems that enable the generation of smaller air bubbles are preferable in this context. The  $k_L a$  favourable effect of sea salt ( $f_S$ ) and the  $k_L a$  reducing effect of floc volume ( $\alpha$ ) show overall a multiplicative effect on  $k_L a$  ( $\alpha \cdot f_S$ ). To consider the effect of sea salt on  $k_L a$  in the design of fine-bubble aeration systems, the extended design approach derived in this paper can be used. The approach is valid for conventional fine-bubble diffusers and allows

**Table 2** | Nomenclature of the design equation SOTR with regard to fine-bubble aeration systems

Abbreviation	Unit	Description
$\alpha$	–	Interface factor
$\beta_{\alpha}$	–	Salt correction factor for oxygen saturation under operating conditions
$\beta_{St}$	–	Salt correction factor for oxygen saturation according to manufacturer specifications
$\theta$	–	Temperature correction factor
$C_{S,20}$	mg/l	Oxygen saturation concentration at 20 °C
$C_{S,T}$	mg/l	Oxygen saturation concentration at $T_W$
$C_X$	mg/L	Oxygen concentration in the aeration tank
$f_d$	–	Depth factor
$f_{S,\alpha}$	–	Salt correction factor $k_{La}$ under operating conditions
$f_{S,St}$	–	Salt correction factor $k_{La}$ according to manufacturer specifications
$h_{geo}$	m a.s.l.	Geodetic height of the WWTP
$OV_h$	kg/h	Oxygen consumption per hour
$p_{atm}$	hPa	Atmospheric pressure at plant location
SOTR	kg/h	Standard oxygen transfer in clean water
$T_W$	°C	Water temperature

for a load case-specific safety margin due to the effect of air flow rate on  $f_S$ .

In high-load or industrial WWTPs, there might be significantly higher concentrations of further surface active substances, extracellular proteins and other organics than in conventional municipal WWTPs. Those substances might override the positive effect of sea salt on  $k_{La}$ . The interactions of such substances and sea salt with regard to the resulting net effect on  $k_{La}$  deserves further investigation.

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