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Resilience of carbon and nitrogen removal due to aeration interruption in aerated treatment wetlands

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ABSTRACT

Treatment wetlands have long been used for domestic and industrial wastewater treatment. In recent decades, treatment wetland technology has evolved and now includes intensified designs such as aerated treatment wetlands. Aerated treatment wetlands are particularly dependent on aeration, which requires reliable air pumps and, in most cases, electricity. Whether aerated treatment wetlands are resilient to disturbances such as an aeration interruption is currently not well known.

In order to investigate this knowledge gap, we carried out a pilot-scale experiment on one aerated horizontal flow wetland and one aerated vertical flow wetland under warm (T_{water} > 17°C) and cold (T_{water} < 10°C) weather conditions. Both wetlands were monitored before, during and after an aeration interruption of 6 d by taking grab samples of the influent and effluent, as well as pore water. The resilience of organic carbon and nitrogen removal processes in the aerated treatment wetlands depended on system design (horizontal or vertical flow) and water temperature. Organic carbon and nitrogen removal for both systems severely deteriorated after 4 – 5 d of aeration interruption, resulting in water quality similar to that expected from a conventional horizontal sub-surface flow treatment wetland. Both experimental aerated treatment wetlands recovered their initial treatment performance within 3 – 4 d at T_{water} > 17°C (warm weather) and within 6 – 8 d (horizontal flow system) and

4 – 5 d (vertical flow system) at T_{water} < 10°C (cold weather). In the vertical flow system, DOC, DN and NH_4-N removal were less affected by low water temperatures; however, the decrease of DN removal in the vertical flow aerated wetland at T_{water} > 17°C was twice as high as in the horizontal flow aerated wetland. The quick recovery of treatment performance highlights the benefits of aerated treatment wetlands as resilient wastewater treatment technologies with high performance and low maintenance requirements.

Keywords

Aeration, constructed wetland, domestic wastewater, aeration stop, system dynamics

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Abbreviations

CSTR	continuous-stirred-tank-reactor
DO	dissolved oxygen
DOC	soluble organic carbon
DN	soluble nitrogen
d	days
EC	electric conductivity
HA	aerated horizontal sub-surface flow wetland
nHRT	nominal hydraulic retention time
ORP	redox potential
T_{water}	water temperature

t_R	recovery time
VA	aerated vertical sub-surface flow wetland

1. INTRODUCTION

The release of untreated and/or non-adequately treated wastewater still poses a threat to the protection of groundwater and natural waterways, especially in rural areas of less developed and semi-arid countries (WWAP, 2017). In such areas centralized sewer systems are often not feasible from an engineering or economical standpoint (Maurer et al., 2005). An interesting alternative to centralized wastewater treatment can be found in a decentralized approach using treatment wetlands (Zhang et al., 2014). In recent decades, treatment wetland technology has expanded to include more engineered or intensified designs such as aerated treatment wetlands (Ilyas and Masih, 2017). Aerated treatment wetlands provide high levels of treatment for organic carbon, nitrogen and pathogens in a cost-effective manner and have low operation and maintenance requirements in comparison to conventional wastewater treatment technologies. Opposed to completely passive treatment wetland designs, aerated treatment wetlands use pumps that move air within the system thereby reducing the system footprint but requiring a power source for operation. How does an aerated treatment wetland respond during and after a power disruption or air pump failure? The resilience of aerated treatment wetland technology is not well known.

In general, the resilience of a wastewater treatment system can be defined as the ability to maintain treatment performance during a disturbance or to recover initial treatment performance within a given time after a disturbance (Cuppens et al., 2012). However, theoretical concepts and metrics of resilience are still a widely disputed topic in the literature (Holling, 1973, 1996; Bruneau et al., 2003; Cuppens et al., 2012; Thorén, 2014). In a recent review, García et al. (2017) highlighted the importance of resilience in wastewater treatment, but reported that few publications directly address the topic of resilience.

The resilience of aerated treatment wetlands may be affected by a number of factors. Here, two factors (wetland design and temperature) are considered in detail. Wetland design can be classified in two main types, horizontal and vertical flow. These two designs are reported to exhibit different hydraulic characteristics: aerated vertical flow systems show hydraulic characteristics similar to one continuous-stirred-tank-reactor (CSTR) compared to three to four CSTRs in an aerated horizontal flow design (Boog, 2013; Boog et al., 2014). The wetland design controls the hydraulics and the wetland functioning, but may also affect resilience. Temperature has an influence on microbial processes in treatment wetlands in general (Kadlec and Wallace, 2009) and thus, may have a potential effect on the resilience of aerated treatment wetlands.

The resilience of treatment wetlands to shock loads or operational malfunctions has been addressed in very few studies (Zapater et al., 2011; Dotro et al., 2012; Butterworth et al., 2016). To the best of our knowledge, only one publication deals with resilience in aerated treatment wetlands (Murphy et al., 2016), which highlights an important research gap. Murphy et al. (2016) reported the resilience of nitrification due to a two–week–long aeration interruption in a full-scale aerated horizontal flow wetland. However, this study only investigated one aerated wetland design, and the question remains whether aerated vertical flow systems are resilient against aeration interruption and to which degree. Furthermore, the resilience of organic carbon removal in aerated treatment wetlands has not yet been investigated. The internal system behavior during transition from aerated to non-aerated phases has also not been reported in the literature; this information could reveal fundamental insights into the functioning of aerated treatment wetlands.

To address these open questions, this study investigates the resilience of organic carbon and nitrogen removal due to aeration interruption in aerated treatment wetlands. The goal of this paper is to assess the potential effect(s) of (1) system design (aerated horizontal or aerated vertical flow), and (2) temperature on the resilience of aerated treatment wetlands, and, (3) to investigate the spatio-temporal system behavior during the transition from aerated to non-aerated phases. To investigate the effect of system design and temperature, pilot-scale experiments including one horizontal and one vertical sub-surface flow aerated wetland were carried out under warm and cold weather conditions. To study the associated spatial dynamics, samples of the influent, effluent and pore water were taken over the course of the experiments.

2. MATERIALS AND METHODS

2.1 Experimental Methods

2.1.1. Site and system description

The pilot-scale experiments were carried out at the UFZ Ecotechnology Research Facility at Langenreichenbach, Germany. Two unplanted aerated sub-surface flow treatment wetlands were used as experimental units: a saturated horizontal flow system (HA) and a saturated vertical down-flow system (VA). Previous studies (Nivala et al., 2013b; Boog et al., 2013; Boog et al, 2014) at the site did not find significant differences in mass removal of bulk organic carbon and nitrogen between the two unplanted systems and planted replicates. This validated the use of unplanted systems to further investigate the resilience of organic carbon and nitrogen removal in aerated treatment wetlands. A detailed description of the experimental site and the two wetlands can be found in Nivala et al. (2013a). Basically, the horizontal flow system measured 4.7 m in length, 1.2 m in width with saturated depth of

1.0 m. The vertical flow system measured 2.7 m in length, 2.4 m in width with a saturated depth of 0.85 m. Both wetlands were filled with gravel (8 - 16 mm) as the main filter media. Coarse gravel (16 - 32 mm) was used as filter media in the influent and effluent zones for HA. Both systems were continuously aerated (24 h d^{-1}) through a network of drip irrigation tubing installed along the wetland bottom according to Wallace (2001). Air was provided by electric diaphragm pumps: one pump (*Mistral 4000, Aqua Medic*) for VA at an air flow rate of approximately 1.6 $m^3 h^{-1}$ and three pumps (*Mistral 2000, Aqua Medic*, two at the front and one in the back) for HA at flow rates of approximately 1.2 $m^3 h^{-1}$ (first half) and 1.0 $m^3 h^{-1}$ (second half). The two systems were loaded with domestic wastewater that was pretreated in a septic tank with a nominal hydraulic retention time (nHRT) of 3.5 d. The design loading rate of both wetlands was 576 L d^{-1}, resulting in an areal specific loading rate of 102 mm d^{-1} for the horizontal flow, and 95 mm d^{-1} in case of the vertical flow system. Wastewater was dosed every 30 min for HA and every hour for VA.

Both systems were established in September 2009 and started operation in June 2010. The aeration modes were changed between August 2012 and July 2014: the horizontal flow system was switched to a wind-powered air pump (Boog et al. 2016) and the vertical flow system to intermittent electric aeration (Boog et al. 2014). In August 2014, aeration in both systems was switched back to continuous electric aeration. It is noted here that the wind-driven aeration in the horizontal flow (HA) system turned out to be insufficient; the system became overloaded during that time which induced clogging of the aeration system (Boog et al., 2016). In autumn 2014, HA was drained and filled with clean water while compressed air at a pressure of 5 bar was injected into the aeration system in order to clean the clogged aeration orifices. Despite this, the system was at stable performance ~~during the~~ before the start of this study.

2.1.2 Experimental design

Two experimental series were carried out: one series during warm weather ($T_{water} > 17°C$, June-August 2015) and one series during cold weather conditions ($T_{water} < 10°C$, January-February 2016). Each series contained an experiment on the horizontal (HA) and the vertical flow (VA) aerated wetland. The warm weather experiments were conducted in series (first HA then VA) due to limitations in the number of available sensors and auto-samplers. The cold weather experiments were conducted side-by-side. Both systems were monitored during a four to six-week baseline phase to assess baseline performance, a six–day interruption phase without aeration and an eight–day recovery phase after restarting aeration. During the baseline phase, grab sampling of influent and effluent was done by hand on four individual days; during interruption and recovery phases influent samples were taken daily by hand and auto-samplers (*WaterSam WS 312*) were used to obtain effluent samples (one sample every 8 – 12 h). Pore water sampling along the main flow path during the experiments (four times during baseline, quasi-daily during interruption and recovery phases) was done manually using stainless steel piezometers and a peristaltic pump at a flow rate of 2 L min^{-1}. Pore water samples along the main flow path in HA were taken at 13, 25, 50 and 75 % of the length (depth of 0.5 m), and, in VA at 17, 50 and 84 % of the depth (at the middle of the system) (See supplementary information on **Figure S1**). During the interruption and recovery phases, additional pore water samples were taken for HA at 38 % of the main flow path length. Additional information about the experiments is given in the supplementary information (**Chapter S1**).

2.1.3 Water Quality Analysis

All grab samples were analyzed for redox potential ORP (*SenTix® ORP, WTW Weilheim*), electric conductivity (EC) and dissolved oxygen (DO) (*ConOx®, WTW Weilheim*), temperature (*T*) and pH (*SenTix® pH*) using a handheld meter (*Multi 350i®, WTW Weilheim*) and a pH meter;

five-day carbonaceous biochemical oxygen demand ($CBOD_5$*, DIN 38409 H52, WTW OxiTOP®*), dissolved organic carbon (DOC*, DIN EN 1484, Shimadzu TOC-VCSN, filtration by 0.45 µm ceramic filter*), dissolved nitrogen (DN*, DIN EN 12660, Shimadzu TNM-1, filtration by 0.45 µm ceramic filter*), ammonia nitrogen (NH_4-N*, DIN 38 406 E5, Thermo Fisher Scientific Gallery Plus*), nitrate nitrogen (NO_3-N, *DIN 38 405 D9, Thermo Fisher Scientific Gallery Plus*) and nitrite nitrogen (NO_2-N*, DIN 38 405 D10, Thermo Fisher Scientific Gallery Plus*). Inflow and outflow to the pilot-scale wetlands were measured with a magnet inductive flow meter (*Endress+Hauser, Promag 10*) and a tipping counter, respectively. During interruption and recovery phases, sensors for dissolved oxygen (*CellOx®, WTW Weilheim*), pH (*SenTix, WTW Weilheim*) and organic reduction potential (*SenTix ORP, WTW Weilheim*) were placed in the effluent stream of both wetlands for on-line monitoring.

2.2 Data Processing

2.2.1 Pre-processing

Outliers were identified by a graphical interpretation of the concentration time series. Detection limits of NH_4-N, NO_3-N and NO_2-N were 0.02, 0.07, 0.01 mg L^{-1}, respectively. To be conservative, analysis data reported as below detection limits were set to the value of the corresponding detection limit.

2.2.2 Mass removal

Areal and percentage mass removal rates were calculated according to **Equation 1** and **2**.

$$Areal\ Mass\ Removal = \frac{1}{A_{wetland}} \cdot (C_{in} \cdot Q_{in} - C_{out} \cdot Q_{out}) \quad (1)$$

$$Percentage\ Mass\ Removal = 100 \cdot \left(1 - \frac{C_{out} \cdot Q_{out}}{C_{in} \cdot Q_{in}}\right) \quad (2)$$

Areal mass removal is in g m^{-2} L^{-1}, percentage mass removal in %, the wetland surface area $A_{wetland}$ in m². C_{in} and C_{out} are the inflow and outflow concentrations in mg L^{-1}. Q_{in} and Q_{out} are the hydraulic inflow and outflow rates in L d^{-1}. For treatment wetlands it is recommended to calculate mass removal on flow and concentration averages of three to four nHRT periods (Kadlec and Wallace 2009). Mass removal of the baseline phases were calculated using averages of flow and concentration over the corresponding baseline phase. Mass removal during the interruption and recovery phase were calculated using daily averages of flow and concentration due to the short time frame.

2.2.3. Resilience Metrics

To quantitatively assess resilience, we defined three metrics: *1)* delay, defined as the time after aeration stop/restart at which a first change in treatment performance is detected, *2)* loss of treatment performance, defined as the change in effluent quality as compared to the baseline phase, and *3)* recovery time (t_R) , defined as the time from aeration restart until initial baseline performance is recovered.

3. RESULTS

To investigate the resilience of carbon and nitrogen removal due to aeration interruption in aerated treatment wetlands, two pilot-scale wetlands, one with horizontal the other with vertical flow, were monitored before (baseline phase), during (interruption phase) and after (recovery phase) a six-day-long aeration interruption under warm weather (T_{water} > 17°C) and cold weather (T_{water} < 10°C) conditions.

3.1 Water Temperature, Precipitation and Flow during the Interruption and Recovery phases

The average of effluent and pore water temperatures during the interruption and recovery phases for the horizontal (HA) and vertical flow (VA) systems at $T_{water} > 17°C$ were both 20°C respectively, and, at $T_{water} < 10°C$, were 6 and 5°C respectively. Pore water temperatures at the beginning of the cold weather trial were below 2°C. A precipitation event (11 mm d^{-1}) one day prior to aeration interruption in the warm water experiment in HA resulted in an outflow rate increase of 5% ($T_{water} > 17°C$) while the corresponding hydraulic outflow was stable at an average of 570 L d^{-1}. In contrast, during the warm weather experiment VA, precipitation events on Days 4 and 8 (25 and 50 mm d^{-1} respectively) resulted in an increased effluent flow rate of approximately 60 % on Day 4 and 50 % on Days 8 and 9. Despite this fact, effluent pollutant concentrations fluctuated only 10 – 15 % as a result of the rainfall. During the cold weather trial ($T_{water} < 10°C$), several precipitation events with a maximum of 4 mm d^{-1} occurred on Days 2, 4, 10 and 11, and one event with a precipitation of 13 mm d^{-1} on Day 13. The corresponding daily mean effluent flow rates for HA and VA were 586 and 579 L d^{-1}, respectively, except for Day 13, with 630 and 635 L d^{-1}. Additional data, including data for the baseline phases, are given in the supplementary information (**Table S2**, **Figures S2–S3**). The obtained results show that hydraulic flow was not in complete steady-state over the course of the experiments, however, this fact was considered as acceptable for the purpose of evaluating chemical water quality parameters.

3.2 Water Quality during Baseline Phases

The water quality of the influent to the wetlands during the different baseline phases showed typical characteristics of primary treated domestic wastewater, including low DO (< 1 mg L^{-1}), low ORP (< -200 mV), high concentration of DOC (> 80 mg L^{-1}), DN (> 60 mg L^{-1}) and NH_4-N

(> 50 mg L^{-1}) (**Table 1, Figure S4–S5**). Average pH values were close to 7.0, NO_2-N concentration were < 0.01 mg L^{-1} and DOC/DN were 1.0 - 1.5. In contrast, both wetlands consistently produced high quality effluents characterized by low concentrations of DOC (< 12 mg L^{-1}), DN (< 45 mg L^{-1}),NH_4-N (< 5 mg L^{-1}) and high concentrations of NO_3-N (< 50 mg L^{-1}), DO (> 5.0 mg L^{-1}), and ORP (> 20 mV) (**Table 1, Figure S4–S5**). Average pH values were 6.8 – 7.5, NO_2-N effluent concentration were < 0.01 mg L^{-1} and DOC/DN were 0.2 – 0.5. The corresponding mass removal rates for HA and VA respectively at T_{water} > 17°C were 89 and 93% (DOC), 49 and 72% (DN) as well as 100 and 96% (NH_4-N). The mass removal rates for HA and VA respectively at T_{water} < 10°C were 89 and 88% (DOC), 42 and 52% (DN) as well as 100 and 97% (NH_4-N) (**Table S1**). Furthermore, HA was characterized by stronger gradients of ORP, DO, DN, NH_4-N, NO_3-N and NO_2-N along the main flow direction compared to the VA. In HA, all NH_4-N was removed within 50 % of the length; however, the concentration of NO_3-N was only 60% of the initial NH_4-N concentration, indicating a simultaneous removal of NH_4-N and NO_3-N. In contrast, water quality gradients in the vertical flow system VA are closer to the inflow (within the first 10 % of the depth). For both wetlands, carbon and nitrogen profiles are well reflected by a sharp increase of ORP.

Table 1. Influent and effluent water quality during the baseline phases.

Sample	DO mg L^{-1}	ORP mV	DOC mg L^{-1}	DN mg L^{-1}	NH_4–N mg L^{-1}	NO_3–N mg L^{-1}	Flow L d^{-1}
Warm weather (T_{water} > 17°C)							
VA-In	0.7± 0.1	-254.2 ± 14.7	108.3 ± 34.9	83.5 ± 13.0	69.4 ± 7.7	0.1 ± 0.0	578.2 ± 0.2
VA-Out	5.4 ± 0.3	82.5 ± 62.2	7.2 ± 3.8	24.1 ± 11.0	2.9 ± 1.1	16.9 ± 9.4	573.6 ± 38.7
HA-In	0.6 ± 0.1	-234.9 ± 37.2	98.5 ± 27.7	74.0 ± 10.0	64.1 ± 7.2	0.1 ± 0.0	578.3 ± 0.1
HA-Out	8.6 ± 0.3	175.1 ± 83.6	10.4 ± 0.6	38.4 ± 4.3	0.0 ± 0.0	37.3 ± 2.4	594.0 ± 44.7
Cold weather (T_{water} < 10°C)							

In	0.9 ± 0.5	-231.7 ± 26.8	87.7 ± 25.9	66.8 ± 8.0	60.6 ± 13.0	0.1 ± 0.0	578.3 ± 0.1
HA-Out	12.8 ± 0.9	203.1 ± 18.5	9.4 ± 0.8	39.2 ± 7.5	0.0 ± 0.0	39.0 ± 9.1	580.1 ± 18.1
VA-Out	6.3 ± 1.9	201.8 ± 14.3	9.7 ± 1.0	32.2 ± 7.4	2.2 ± 1.5	29.2 ±6.9	584.8 ± 23.8

3.3 Water Quality during Interruption and Recovery Phases

The stop in aeration triggered a simultaneous decrease in DO and NO_3-N concentrations and an increase in DOC, DN and NH_4-N effluent and pore water concentrations in both the horizontal (HA) and vertical flow (VA) system during both trials. Effluent quality of both HA and VA deteriorated to the level of a conventional horizontal sub-surface flow wetland (conventional horizontal sub-surface flow wetlands systems at the same site were examined by Nivala (2013b) and Ayano (2014)) within 4 – 5 d after the aeration was switched off. This is reflected by a corresponding drop in ORP to approximately -400 mV ($T_{water} > 17$°C) and -300 – 130 mV ($T_{water} < 10$°C). After restarting aeration, both HA and VA recovered their initial effluent quality from the baseline phase within 3 – 4 d in warm weather ($T_{water} > 17$°C), and, within 6 – 8 d for HA and 4 – 5 d for VA in cold weather ($T_{water} < 10$°C) (**Figure 1).** It is to be noted that during the warm weather trial ($T_{water} > 17$°C) aeration was restarted 5 h later for VA than for HA. The corresponding times series of VA at $T_{water} > 17$◦°C was shifted back by 5 h to simplify the graphical comparison shown in **Figure 1**. Due to the fuzziness of the online sensor response and to simplify the visualization in **Figure 1** we extracted the data from the online sensor responses (**Figure S6)** at the same time that an effluent grab sample was taken.

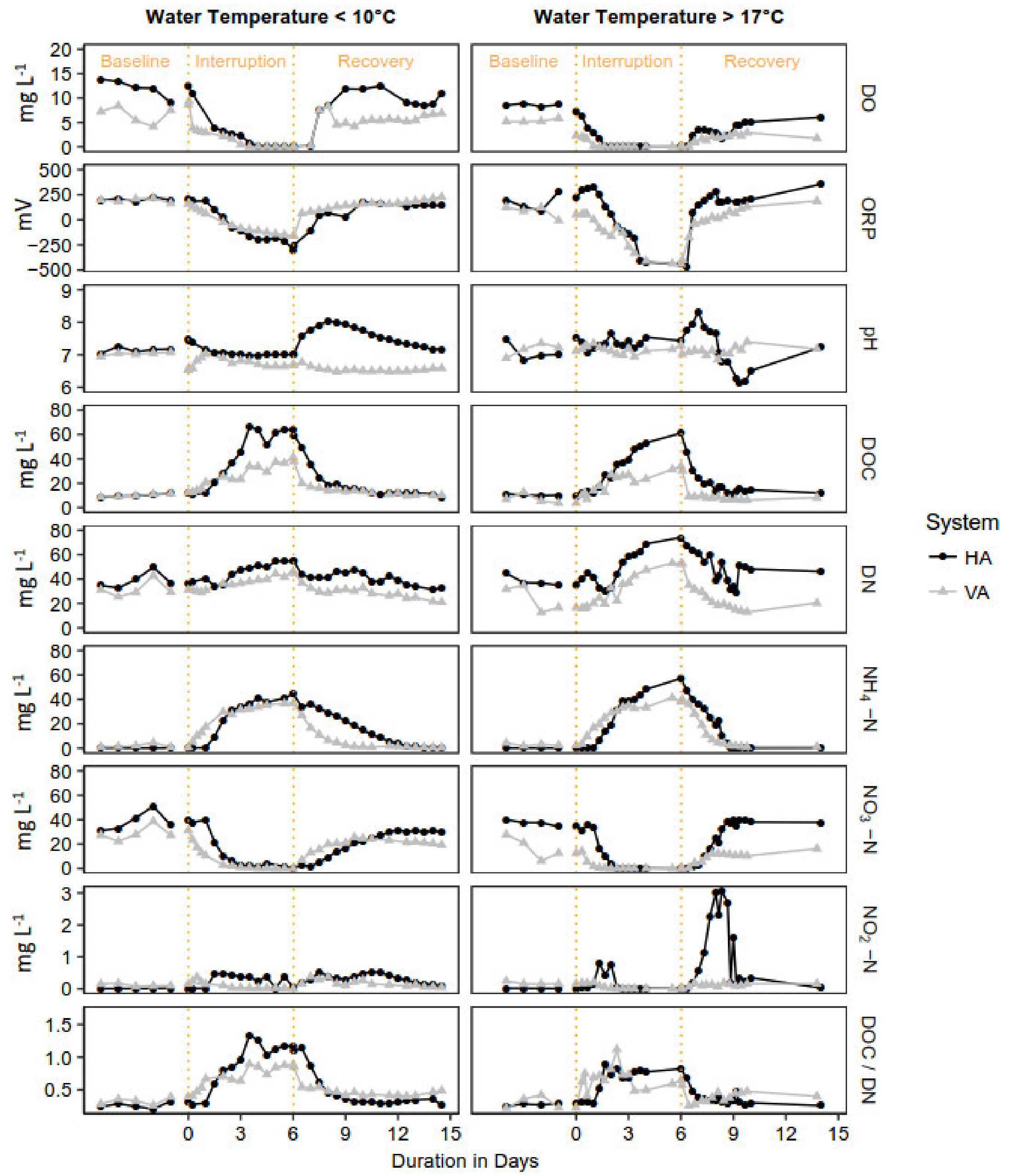

Figure 1. ***Effluent water quality of the horizontal (HA) and vertical flow (VA) wetlands during the cold (T_{water} < 10°C) and warm weather (T_{water} > 17°C) trials. Note, baseline performance is not to temporal scale.***

Influent quality during the interruption and recovery phases of all experiments were similar compared to their corresponding baseline phases (**Figures S4 – S5**). The depletion of effluent DO for both systems at $T_{water} < 10°C$ took twice as long as compared to the trials at $T_{water} > 17°C$. The longer depletion can be divided into two distinct parts: an initial drop-down with a steep concentration gradient and a second phase with a more gentle concentration gradient (**Figure 1**). The second phase might be caused by the measurement method itself and not by processes in the wetland systems. Because the measurement was conducted with DO online probes that were installed in a measurement cylinder located several meters of pipe after the actual location of the wetlands' outflow, re-aeration due to unsaturated flow in the pipes may have biased the measurement. This phenomenon would be more acute at low DO concentration.

In contrast to DO concentration, changes in ORP, DOC, NH_4-N and NO_3-N concentrations after switching off the air pump were delayed by approximately 1 d for HA and by less than 10 h for VA. The change in NO_2-N effluent concentration in HA at $T_{water} > 17°C$ was delayed by approximately 1 d. The changes in DOC/DN in HA were delayed about 1 d but did not show any delay in VA. There was no remarkable delay for pH, DO and DN. After switching the air pump back on, there was no delay for the change in water quality, except for NO_3-N and DOC/DN in HA at $T_{water} < 10°C$. The end of the interruption phase exhibited significant losses of treatment performance for both wetlands (**Table 2**). The recovery times of effluent concentrations for HA and VA differed with respect to the water quality parameters analyzed (**Table 3**). Associated DOC, DN and NH_4-N mass removal rates recovered within a similar timeframe as effluent concentrations (**Figure S7**).

Table 2. ***Loss of treatment performance after 6 d of non-aeration (concentration based reduction (CR), areal mass removal (MR))***

Design	DOC		DN		NH_4–N	
	CR (%)	MR (%)	CR (%)	MR (%)	CR (%)	MR (%)
Warm weather (T_{water} > 17°C)						
HA	50	44	31	19	76	70
VA	39	43	42	52	61	71
Cold weather (T_{water} < 10°C)						
HA	68	82	38	32	80	85
VA	43	43	29	25	71	62

Table 3. ***Recovery time (t_R) of effluent concentrations and ratio of recovery time to nominal hydraulic retention time (nHRT) after restarting aeration.***

Design	DOC		DN		DOC/DN		NH_4–N		NO_3–N	
	t_R (d)	t_R /nHRT	t_R (d)	t_R /nHRT	t_R (d)	t_R /nHRT	t_R (d)	t_R /nHRT	t_R (d)	t_R /nHRT
Warm weather (T_{water} > 17°C)										
HA	3.0	0.8	3.0	0.8	2.0	0.6	3.5	0.9	2.0	0.6
VA	1.0	0.3	3.0	0.9	1.0	0.3	3.5	1.0	2.0	0.6
Cold weather (T_{water} < 10°C)										
HA	4.5	1.2	3.5	0.9	3.5	0.9	7.5	2.0	6.0	1.6
VA	2.0	0.6	2.0	0.6	2.0	0.6	4.0	1.2	3.5	1.0

NO_3-N effluent concentrations after recovery (T_{water} < 10°C) were lower than during the baseline phase for HA and VA. This might be caused by a decreasing NH_4-N influent concentration from Day 11 onwards and/or the precipitation events from Days 9 – 12. Effluent

NO_2-N concentrations for HA at $T_{water} > 17°C$ peaked after switching aeration off (Day 1 – 2) and on again (Day 8 – 9). At $T_{water} < 10°C$ NO_2-N effluent concentrations in HA increased to approximately 0.5 mg L^{-1} but did not peak. Zones of NO_2-N accumulation were observed in HA; one occurred after the air pumps were switched off (between 10% and 60% of the fractional length) and another occurred after the air pumps were restarted (between 20% and 100% of the length) (**Figure 1**). VA exhibited only minor fluctuations in NO_2-N effluent concentrations during both trials (**Figure 1 - 2**).

Switching aeration off altered the baseline water quality patterns in both systems (**Figure 2**). The alteration intensity was higher in HA. Water quality turned, for both systems, from levels of ORP (> 100 mV), DO (> 3 mg L^{-1}) and NO_3-N (> 20 mg L^{-1}) to ORP levels below -100 mV and high concentration of DOC (> 30 mg L^{-1}), DN (> 35 mg L^{-1}) and NH_4-N (> 35 mg L^{-1}). Spatial gradients of DO, ORP, NO_2-N and NO_3-N concentration in both systems disappeared completely. The spatial gradients of DOC, DN and NH_4-N, in contrast, declined but where still observable. The span of DOC, DN and NH_4-N gradients in the horizontal flow (HA) system stretched from 30 – 40% of the length (baseline) to the whole flow path length within six day without aeration. The corresponding spatial gradients in the vertical flow system (VA) did not stretch, except for DN concentration at $T_{water} > 17°C$. After switching aeration back on, pore water quality patterns similar to the baseline phase were observed within six to eight days.

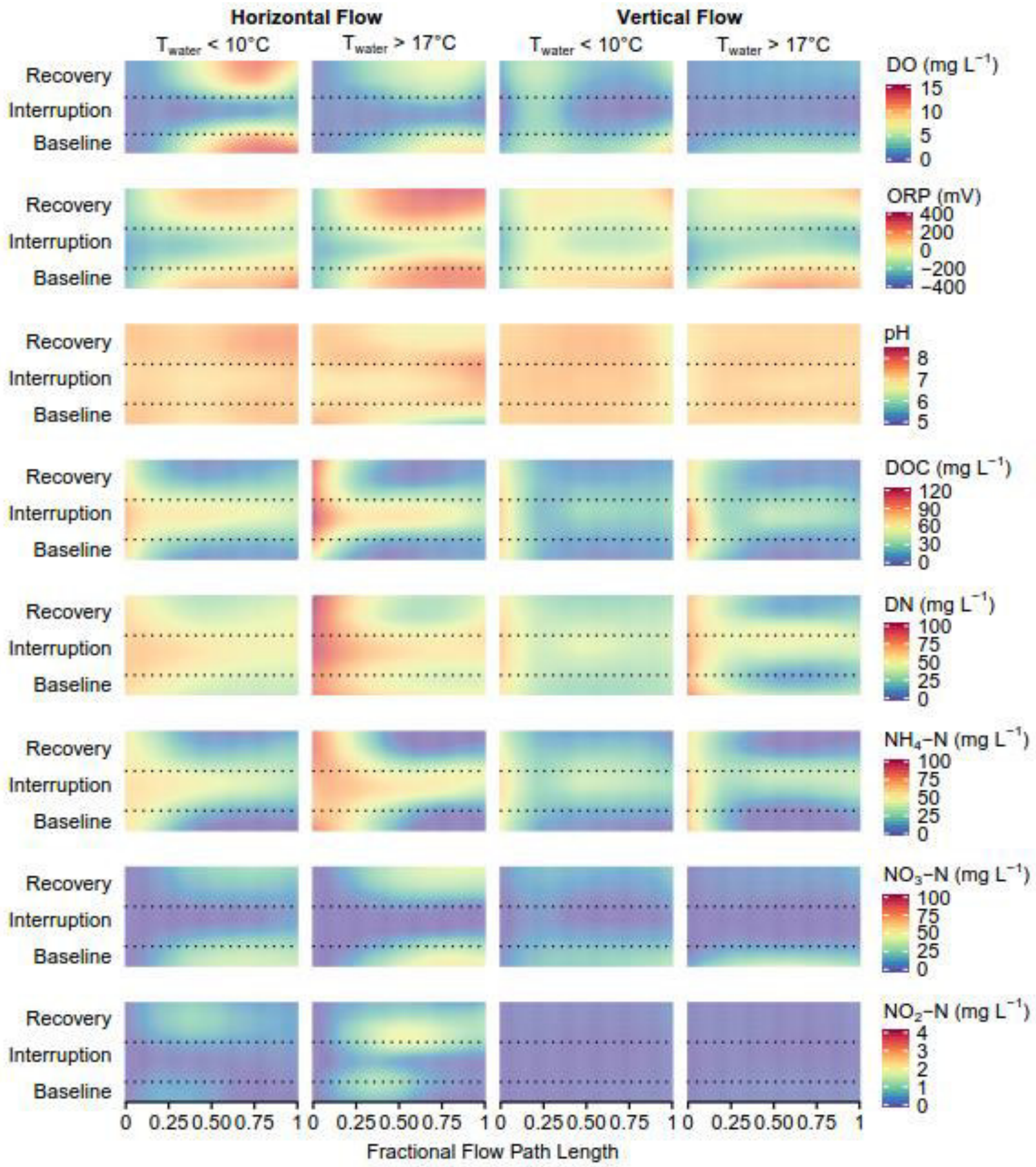

Figure 2. ***Water quality profiles of the horizontal (HA) and vertical flow (VA) aerated treatment wetlands across the fractional length of the main flow paths during baseline, interruption and recovery phases. Visualization of the baseline phases at T_{water} > 17°C are not to scale and should be only qualitatively interpreted (see Appendix A for further explanation).***

4. DISCUSSION

4.1 Factors affecting Resilience

Oxygen Transfer

After the baseline phase, aeration was switched off. Aerobic carbon removal and nitrification ceased, as well as the production of NO_3-N through nitrification. The reason was oxygen (DO) depletion as oxygen is a main driver of biochemical processes in biological wastewater treatment (Tchobanoglous et al., 2003) and treatment wetlands (Kadlec and Wallace, 2009). Oxygen is especially important regarding organic carbon removal and ammonia elimination via autotrophic nitrification (Gujer, 2010; Nivala et al., 2013b; Liu et al., 2016). During the interruption phase both systems shifted from aerobic to anoxic conditions (ORP > 100mV) then to rather anaerobic conditions (ORP < -100mV) which corresponded to a continuous deterioration in treatment performance (**Figure 1**). After restarting aeration, both systems fully recovered. Similar findings were also reported for a two-week long aeration interruption in a horizontal flow system (Murphy et al., 2016).

System Design

In steady-state (baseline phase), the main active zone for microbial degradation of organic carbon and nitrogen in the horizontal flow (HA) system was limited to the first half of the wetland, whereas, in the vertical flow (VA) system, the main zone was either limited to the inflow region or did not exist (**Figure 2**). It is to be noted that the pore water visualizations (**Figure 2**) were created by interpolating data from discrete sampling events including influent and effluent samples as well. Thus, pore water quality distribution at the in- and outflow region may not exactly represent reality. Despite these limitations, the visualizations are accurate enough to differentiate general system specific pore water quality patterns. Such patterns were lower water quality gradients and less variable water quality across the main flow path length in VA. This is probably caused by a lower degree of hydraulic mixing in aerated vertical flow wetlands (Boog et al., 2013) and a five times higher cross sectional loading area in VA compared to HA. In a former study in the same systems, (Button et al., 2015) reported a higher microbial activity in the first half of HA, whereas, activity was more equally distributed across the whole flow path length of VA. Strong pore water quality gradients in aerated horizontal flow wetlands were also reported in the literature (Li et al., 2014; Zhong et al., 2014). In general, biological reactors with higher hydraulic mixing exhibit less spatial concentrations gradients of certain components (Müller-Erlwein, 2007). As such, the results of baseline performance of this research comply with previous studies in the same systems and in similar aerated treatment wetlands at the same site (Boog, 2013; Boog et al., 2014).

The longer response delays after aeration stop for ORP, DOC, NH_4-N, and NO_3-N effluent concentrations in HA (≈ 1.0 d HA, < 10 h VA) were caused by the specific hydraulic characteristics of the design. In HA, the baseline pore water gradient for NO_3-N was at 30 – 50% of the length. The NO_3-N rich water at 50 – 100% of the length was discharged before the gradient could be detected in the effluent—which caused the delay and a smooth decline of NO_3-N effluent concentrations afterwards. In VA, the hydraulic mixing is higher (Boog et al., 2013), which may still be the case during non-aeration phases. Additionally, the outlet position in VA is closer to the inlet than in HA. This facilitates short-circuiting and would reduce any response delay. Hydraulic flow in sub–surface flow constructed wetlands is, in general, governed by advective–dispersive transport (Kadlec and Wallace, 2009). Thus, we believe that advective–dispersive transport is the dominant mechanism causing the delay and the smooth decline afterwards. Murphy et al. (2016) also observed a delay followed by a smooth decline of NO_3-N in a horizontal flow system and suggested diffusive transport of NO_3-N from the biofilm into the bulk water as the main mechanism. However, Murphy et al. (2016) did not monitor the change in pore water concentration. In conjunction with water temperature, system design also affected the recovery times (see paragraph **Water Temperature**).

Water Temperature

NH_4-N, DN and DOC removal in VA were less affected by low water temperature than in HA (**Figure 1 – 2** and **Figure S7**). This might be caused by a reduced microbiologically active zone in HA, as this zone was suggested to be limited to the front of the wetland during the baseline phase. Comparable results could not be found in the literature. DN removal in VA was affected to a greater extent than in HA at $T_{water} > 17°C$, which was due to the higher DN removal performance (72% in VA, 49% in HA) during the corresponding baseline phase.

Recovery time of NH_4-N and NO_3-N effluent concentrations in HA were longer during cold weather ($T_{water} < 10°C$) than during warm weather ($T_{water} > 17°C$). For VA this was similar but recovery times were shorter during cold weather. This phenomenon was reflected in the ORP values over the course of the experiments. The results suggest an effect of temperature on recovery time, and an interaction effect between system design and water temperature. Murphy et al. (2016) also suggested a temperature effect on the recovery time of a horizontal flow aerated wetland during cold weather. An effect of system design and water temperature is also noticeable for DOC recovery but the interaction is less pronounced. Effluent DO recovery was similar in both HA and VA in the warm weather trial but were shorter during the cold weather trial. This is due to the higher oxygen solubility at lower water temperatures.

Murphy et al. (2016) reported a recovery time of approximately 48 h for nitrification in a horizontal subsurface-flow wetland after two weeks of non-aeration at water temperatures down to 5°C. This shorter recovery time might be caused by a shorter nominal hydraulic retention time nHRT (≈ 2 d), by a lower NH_4-N influent concentration (average < 60 mg L^{-1}) and by a higher baseline NH_4-N effluent concentration (mean of 5.6 mg L^{-1}) for the experimental system of Murphy et al. (2016). Nitrogen removal recovered within 1.0 – 2.0 times the nHRT in the present study. The corresponding ratio of nitrogen removal recovery to nHRT reported by Murphy et al. (2016) would be approximatively 1.0. This indicates that recovery, besides microbial processes, may depend on the nHRT of the treatment wetland. Phan et al. (2015) observed a recovery time of 72 h for TN removal in a membrane bioreactor (nHRT = 1.5 d, mean NH_4-N inflow concentration of 50.0 mg L^{-1}) after 18 h of aeration interruption (T_{water} ≈ 18°C). This translates into a comparably lower recovery time relative to the nHRT.

DOC/DN-Ratio

An interaction of system design and temperature is definitely visible in the development of DOC/DN over the course of the experiments. The development was similar under warm weather conditions but different (higher DOC/DN for HA) under cold weather (**Figure 2**). The similarity (DOC/DN in HA and VA were 0.7 – 1.1) under warm weather conditions was biased by higher DN influent concentration for HA during that trial (that lowered the DOC/DN), otherwise the DOC/DN in HA might have been similar to the cold weather trial (DOC/DN in HA was 1.2 – 1.4). The higher ratio HA was the result of the higher decrease of DOC removal during the interruption phase, which was probably due to the lower microbial active area (that in turn might have lowered the treatment capacity under anaerobe to anoxic conditions) in HA. The lower microbial active area in HA could also explain the slower DOC/DN recovery in HA. The DOC/DN may also indicate the concurrence of heterotrophic and nitrifying bacteria with respect to DO consumption, especially during the recovery phases. In DO limited environments (such as the beginning of the recovery phase), heterotrophic bacteria are reported to outcompete autotrophic nitrifiers (Henze, 2008). This is reflected in the faster decrease of effluent DOC/DN compared to DN and NH_4-N effluent concentrations. A different DOC/DN at the time of aeration restart might have influenced the recovery times of DOC, DN and NH_4-N removal. Nevertheless, the DOC/DN development was probably driven by the break-down of aerobe removal processes and the DOC/DN of the influent instead of being a driver of recovery time (or resilience) itself.

PH Fluctuation

Effluent pH in HA for both trials markedly increased after aeration restart and peaked at a value of approximately 8.0 – 8.3 (**Figure 1**). A possible reason is that CO_2 had been accumulated in the pore water during the phase when aeration was switched off. This could have developed a new equilibrium between CO_2, HCO_3^- and CO_3^{2-}. By switching aeration on again CO_2 could have been quickly stripped, altering the developed equilibrium towards HCO_3^- and CO_3^{2-}. This is one possible cause for the rapid pH increase. The following pH decrease in HA during the warm water trial could have been the result of H_3O^+ production by reinitiated nitrification of NH_4-N that accumulated in the pore water during the interruption phase. Afterwards, pH returned to a level of approximately seven as the wetland and the initial equilibrium recovered. The absence of the rapid decrease in pH during the cold weather trial in HA could have been caused by a lower H_3O^+ production due to a lower NH_4-N pore water concentration and lower NH_4-N decrease (as evidenced by a longer recovery time). The increase in pH in HA might have increased ammonia removal by triggering ammonia volatilization. At 20 °C and a pH of 8.3, 20% of ammonia ions (10% at 10°C) would transform into the gaseous form NH_3 (Tchobanoglous et al., 2003). Additionally, the pH drop below 7.0 during warm weather experiment in HA might have reduced nitrification rates, as nitrification rates are reported to decrease outside of pH 7.0 – 8.0 (Henze, 2008). However, the two effects were difficult to observe in the effluent and pore water concentration series and it is not possible to identify them individually.

NO_2–N Accumulation

The accumulation of NO_2-N up to approximately 3.5 mg L^{-1} in HA during the aeration interruption and recovery phases of the warm weather trial might be caused by a more intense inhibition of nitrite compared to ammonia oxidation. Nitrite oxidizing bacteria are reported to be more susceptible to low DO concentrations and NH_3 (triggered by high pH values) (Okabe et al., 2011). Low DO concentrations and high pH values occurred during the recovery phase in HA. The lower NO_2-N accumulation in HA during the cold weather trial (T_{water} < 10°C) could be caused by reduced NH_4-N influent concentrations. The slight degree of NO_2-N accumulation in VA can be explained by the higher degree of mixing, which reduced the magnitude of the NO_2-N accumulation by dilution. Burgess et al. (2002) observed NO_2-N accumulation in an activated sludge system after switching aeration off. In contrast, Murphy et al. (2016) reported consistently low effluent concentrations of NO_2-N from an aerated horizontal flow wetland system subjected to a 14 d aeration interruption. NO_2-N accumulation may be more an indicator of the shift in nitrogen metabolism than having a direct effect on resilience.

4.2 Spatio-Temporal System Behavior

Pore water quality patterns differed between wetland designs (**Figure 2**) and were more variable in HA. Nevertheless, this variability declined within 5 – 6 d after aeration was switched off (**Figure 2**). In HA, the front part of the bed (up to 40% of the length) had already been anaerobic under baseline conditions and was, therefore, not severely affected by switching the aeration off while the back part (50 – 100% of the length) changed significantly. This is in contrast to VA, which changed almost entirely over the main flow path length. The most probable reason is the higher degree of mixing and the higher cross-sectional loading area in aerated vertical flow wetlands.

Pore water and effluent quality developments revealed the formation of two diametrically opposed groups of water quality parameters—independently of the water temperature. On one hand, high values for DO, ORP and NO_3-N dominated the effluents during the baseline and the end of the recovery phases, while on the other hand, high values for DOC, DN and NH_4-N dominated the interruption phases (**Figure 1**). The transition was smooth. High values of DO, ORP and NO_3-N are commonly observed in aerated treatment wetlands at steady–state (Butterworth et al., 2016; Fan et al., 2016; Wu et al., 2016; Ilyas and Masih, 2017). In contrast, high values of DOC, DN and NH_4-N are dominating passive horizontal flow sub-surface flow wetlands for secondary treatment (Kadlec and Wallace, 2009; (Vymazal, 2005). Thus, the change of aeration would allow controlling the environmental conditions in sub-surface flow wetlands, and thus, the water quality to various degrees. This is important when targeting the treatment of pollutants that require different environmental conditions. This possibility was already applied in intermittent aeration schemes of vertical sub-surface flow wetlands to increase TN removal (Fan et al., 2013; Uggetti et al., 2016) and could be further used in spatial aeration schemes for horizontal subsurface flow aerated wetlands.

4.3 Practical Implications

~~The most significant outcome of this study is the finding that both horizontal and vertical flow aerated wetland designs are able to fully recover from the simulated failure of an air pump. After a six-day period with no aeration, the systems required 8 d or less to fully recover, even with water temperatures as low as 2°C. However, under cold weather conditions, the vertical flow aerated wetland design (VA) recovered more rapidly than the horizontal flow aerated wetland design (HA).~~ This study also showed that aerated treatment wetlands can maintain near-complete nitrification and a total nitrogen removal of 60% after severe operational disruption at water temperatures as low as 2°C. Yet under aeration interruption, treatment performance deteriorates significantly while remaining within a similar range of conventional (passive) horizontal sub-surface flow wetland designs that were examined in previous studies at the same site (Nivala et al., 2013b; Ayano, 2014). It is to be highlighted that even in the midst of an aeration interruption the treatment performance of aerated wetlands still complies with the discharge limits for organic carbon removal according to German regulations DIN EN 12566-3 "Ablaufklasse C" for small-scale treatment systems (DIN EN 12566-3, 2016). It is also important to note that in case of an air pump failure or power interruption, maintenance is of course needed but immediate maintenance may not be necessary. Moreover, the technical integrity of the aerated wetland system will not be compromised if maintenance is carried out within a short time (on the order of one week). The duration of an aeration interruption might affect the resilience of aerated treatment wetlands, either by causing a more permanent shift in the microbial community, or by irreversible damage due to carbon or solids overload which may lead to clogging. Severe biomass accumulation in the horizontal flow wetland (HA) resulted from an aeration interruption of almost two years (Boog et al., 2016).

Future research should investigate the relationship between the length of aeration interruptions and the effect on treatment performance and clogging, and identify potential transition points from resilient to non-resilient behavior in aerated treatment wetlands. Future research should also address the influence of the hydraulic retention time on resilience and the possibility to increase the resilience of aerated treatment wetlands through exposure to stress situations. Cho et al. (2016) observed a reduction of nitrification recovery time in a bioreactor treating steel production wastewater after applying subsequent ammonia shock loads. This might also be the case for aerated treatment wetlands that would be subjected to subsequent aeration interruptions.

5. CONCLUSIONS

This study investigated the resilience of organic carbon and nitrogen removal due to aeration interruption in an aerated vertical flow and in an aerated horizontal flow treatment wetland under warm (T_{water} > 17°C) and cold weather (T_{water} < 10°C) conditions.

The most significant outcome of this study is the finding that both horizontal and vertical flow aerated treatment wetland designs are able to fully recover from the simulated failure of an air pump. After a six-day period with no aeration, the systems required 8 d or less to fully recover, even with water temperatures as low as 2°C. The response for ORP, DOC and nitrogen species clearly differed between the two aerated treatment wetland designs and between the two water temperature ranges. We identified the formation of two diametrically opposed water quality parameter groups (1^{st} DOC, DN, NH_4-N; 2^{nd} ORP, DO, NO_3-N) in the two aerated treatment wetland designs and found higher water quality variability and stronger water quality gradients in the horizontal flow design, which was attributed to the lower degree of hydraulic mixing compared to the vertical flow design.

This study strengthens the hypothesis that aerated treatment wetlands are resilient against disruptions in aeration. Aerated treatment wetlands will still achieve reasonable treatment performance during phases of non-aeration (equivalent to that of conventional horizontal sub-surface flow wetland), and are able to recover without any specific maintenance except the restart of the air pump. These findings identify aerated treatment wetlands as resilient wastewater treatment technologies with low maintenance requirements. ~~and encourage the use of aerated treatment wetlands for water quality improvement and to increase the protection of ground and surface water resources~~.

APPENDIX

To facilitate the understanding of the water quality profiles in **Figure 1**, **Figure A1** shows how to conceptualize the colormaps from more common line charts.

Colormaps in **Figure 1** and **Figure A** were created by interpolating influent, pore water, and effluent water quality data using a local polynomial regression (weighted least-squares, grid resolution 0.01). It is to be noted that the pore water samples during the baseline phases (**Figure 1**) at $T_{water} > 17°C$ were collected over the entire four-week long baseline phase and do not represent the pore water quality at the time directly prior to aeration stop (as is the case for the trial at $T_{water} < 10°C$).

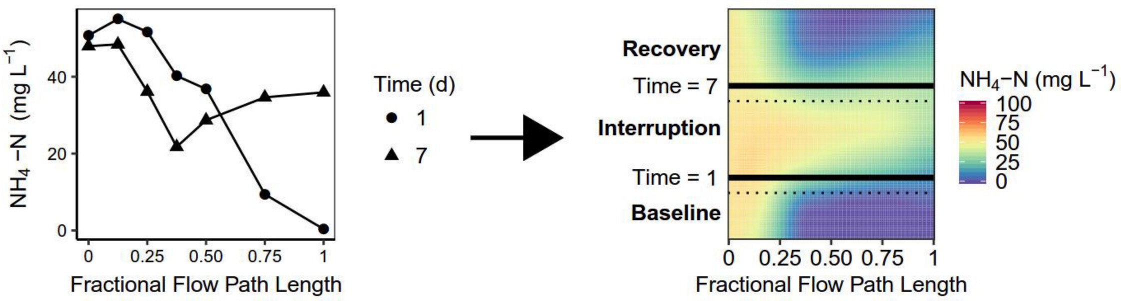

***Figure A1**. Water quality profiles along the fractional flow path length. A graph showing the NH_4-N concentration for a specific time (time = zero is at aeration stop) on the y-axis in a line chart (left plot) corresponces to the color gradient along a hypothetical horizontal line (here indicated in black) in the colormap (right plot).*

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