New approaches to enhance pollutant removal in artificially aerated wastewater treatment systems

Andrew I. Freeman [a], Ben W.J. Surridge [a]*, Mike Matthews [b], Mark Stewart [c], Philip M. Haygarth [a].

[a] Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK.

[b] Peak Associates Environmental Consultants ltd, Lancaster Office, Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK.

[c] Manchester Airport Group Plc, Manchester Airport, Water Services Department, Building 30, M90 1AA, UK.

* Corresponding Author. Tel: +44 (0) 1524 594516. Email: b.surridge@lancaster.ac.uk (B.W.J. Surridge).

Abstract

Freshwater ecosystems sustain human society through the provision of a range of services. However, the status of these ecosystems is threatened by a multitude of pressures, including point sources of wastewater. Future treatment of wastewater will increasingly require new forms of decentralised infrastructure. The research reported here sought to enhance pollutant removal within a novel wastewater treatment technology, based on un-planted, artificially aerated, horizontal subsurface flow constructed wetlands. The potential for these systems to treat de-icer contaminated runoff from airports, a source of wastewater that is likely to grow in importance alongside the expansion of air travel and under future climate scenarios, was evaluated. A new configuration for the delivery of air to aerated treatment systems was developed and tested, based on a phased-aeration approach. This new aeration approach significantly improved pollutant removal efficiency compared to alternative aeration configurations, achieving > 90 % removal of influent load for COD, BOD_5 and TOC. Optimised operating conditions under phased aeration were also determined. Based on a hydraulic retention time of 1.5 d and a pollutant mass loading rate of

0.10 kg d^{-1} m^{-2} BOD_5, > 95 % BOD_5 removal, alongside final effluent BOD_5 concentrations < 21 mg L^{-1}, could be achieved from an influent characterised by a BOD_5 concentration > 800 mg L^{-1}. Key controls on oxygen transfer efficiency within the aerated treatment system were also determined, revealing that standard oxygen transfer efficiency was inversely related to aeration rate between 1 L and 3 L min^{-1} and positively related to bed media depth between 1,500 mm and 3,000 mm. The research reported here highlights the potential for optimisation and subsequent widespread application of the aerated wetland technology, in order to protect and restore freshwater ecosystems and the services that they provide to human society.

Statement of Contributions: AF conceived the research, led the design and implementation of the experiments, analysed samples, undertook statistical analysis and co-led the writing of the manuscript. BS contributed to the conception of the research and design of the experiments and co-led the writing of the manuscript with AF. PH, MS and MM supported the design and implementation of the research and edited earlier versions of the manuscript.

Key Words

Aerated constructed wetlands

De-icer contaminated runoff

Phased artificial aeration

Oxygen transfer efficiency

Organic pollutants

Freshwater ecosystems

1. Introduction

Freshwater ecosystems provide services that are critical for human society (Dodds et al., 2013, UNESCO, 2015, Durance et al., 2016). However, these ecosystems also face diverse pressures resulting from population growth, urbanisation, industrial development and a changing global climate (Vorosmarty et al., 2000, Ormerod et al., 2010, Vorosmarty et al., 2010, Matthews, 2016). In consequence, contemporary rates of degradation within freshwater ecosystems significantly exceed historical rates, but also contemporary rates of degradation within other ecosystems (Barnosky et al., 2011, Valiente-Banuet et al., 2015). The changes in ecosystem structure and function that are associated with degradation threaten the integrity of freshwater ecosystems, but also constrain the potential for human society to benefit from the services that could potentially be provided by freshwaters (Gleick, 1998, World Health Organization, 2015).

Therefore, there is a growing imperative to protect and restore the status of freshwater ecosystems globally. Chemical water quality is a fundmanetal control on ecosystem status and remains subject to significant anthropogenic pressure (Smith and Schindler, 2009, Schindler, 2012, Malaj et al., 2014, Jekel et al., 2015, Van Meter et al., 2016). Point sources have been recognised as a major contributor of pollutants to freshwaters for several decades in many countries (EEA, 2007, DEFRA, 2012, EEA, 2015), with centralised or decentralised wastewater treatment systems being widely used to improve the quality of wastewater entering freshwaters from point sources. However, the energy demands, greenhouse gas emissions and whole life costs associated with many traditional wastewater treatment technologies are subject to increasing scrutiny (Henriques and Catarino, 2017, Rajasulochana and Preethy, 2016). Alternative treatment technologies, characterised by relatively low energy consumption, by simple operating principles and by minimal whole life costs are increasingly required. Such technologies provide a potentially more sustainable means of protecting or enhancing ecosystem services compared to conventional wastewater treatment technologies. Further, such technologies would support enhanced treatment of point sources in countries where significant investment in centralised wastewater treatment infrastructure cannot be made, alongside the treatment of smaller, micro-point sources of wastewater for which traditional technologies may be inappropriate or disproportionately costly. In this context, the research reported here developed a novel treatment technology for wastewater, based on un-planted, artificially aerated, horizontal subsurface flow (HSSF) constructed wetlands. It is recognised that the treatment systems considered in this research do not include the vegetation planting schemes that are common in many constructed wetland designs. This reflects the specific application of the technology to the treatment of wastewater at airports, for which planted systems are potentially inappropriate (see below). However, the HSSF constructed wetland terminology is used within this paper, reflecting the fact that in hydraulic, microbial and bed media geochemical terms, the treatment systems described here share many features that are common with planted HSSF constructed wetlands.

Treatment technologies that rely on natural, passive pollutant degradation processes, including constructed wetlands, offer environmental and economic advantages over many traditional wastewater treatment systems (Castro et al., 2005, Vymazal et al., 2006, Kadlec and Wallace, 2009, Vymazal and Kröpfelová, 2009, Freeman et al., 2015, Wu et al., 2015). However, the availability of dissolved oxygen (DO) is frequently a fundmanetal limit on pollutant removal within traditional constructed wetland designs (Wallace et al., 2007, Kadlec and Wallace, 2009, Nivala et al., 2013). In saturated HSSF wetlands, 0.12 g m^{-2} d^{-1} – 12.11 g m^{-2} d^{-1} O_2 may be transported into a system through the combination of direct diffusion from the atmosphere and diffusion from the sub-surface root network of wetland vegetation (Armstrong et al., 1990, Brix and Schierup, 1990, Brix, 1997, Bezbaruah and Zhang, 2005, Nivala et al., 2013). Vertical flow constructed wetlands achive higher diffusion rates of 28.4 g m^{-2} d^{-1} – 156 g m^{-2} d^{-1} O_2 for saturated systems and up to 482 g m^{-2} d^{-1} O_2 in fill and drain systems, primarily due to the draw down of air into the bed during sequential filling and draining of wastewater through the wetland substrate (Cooper, 2005, Fan et al., 2013a, Nivala et al., 2013). However, the rate of DO supply via these mechanisms within traditional constructed wetland designs is often negligible when compared to the rate of DO consumption associated with many raw wastewaters (Nivala et al., 2013). Whilst anaerobic respiration of some pollutants occurs, the resulting pollutant removal rates are often lower than under aerobic conditions, meaning that treatment efficiency is significantly reduced (Huang et al., 2005, Ouellet-Plamondon et al., 2006, Fan et al., 2013b, Nivala et al., 2013, Murphy et al., 2015, Uggetti et al., 2016). The need to improve rates of DO supply in order to enhance pollutant removal in traditional constructed wetlands has led to the development and commercialisation of aerated wetlands for a range of applications across the globe (Wallace, 2001). Aerated wetlands involve the active supply of DO into a self-contained, media-filled treatment bed, to maintain aerobic conditions within the wetland by meeting the DO demand exerted by wastewater. With sufficient DO supplied to the system through aeration, the role of wetland vegetation root transfer for this purpose is significantly reduced and systems can remain un-planted to serve applications in which attacting wildlife is undesirable. However, there is currently no recognised design standard for aerated wetlands (Nivala et al., 2013), alongside limited empirical data to support understanding of how factors such as the configuration of aeration devices, wetland bed depth or aeration rate impact the availability of DO and, ultimately, pollutant removal. In alternative aerobic wastewater treatment systems, such as activated sludge plants, the energy consumed by aeration devices typically accounts for 45 % to 80 % of the total operating cost (Stenstrom and Rosso, 2006, Zhou et al., 2013). Optimisation of aeration systems within wastewater treatment plants can typically achieve energy efficiency improvements of 20 % to 40 % (Henriques and Catarino, 2017). Therefore, the design and optimisation of aeration systems to ensure maximum O_2 transfer from the gaseous to the liquid phase is central to achieving low cost, sustainable treatment solutions through the use of aerated wetlands.

The research reported here developed and tested novel, artificially aerated HSSF constructed wetland designs to enhance pollutant removal efficiency from wastewater. The specific context for the research was the need for new technological approaches to treat surface water runoff from airports following contamination with chemical de-icers. At international hub airports, > 1 M L of chemical de-icers are applied annually to aircraft, aprons, taxiways and runways to facilitate safe winter operations, potentially contaminating large volumes of storm water runoff (CAA, 2000, ACRP, 2008, Erdogan, 2008, Association of European Airlines, 2012, ISO, 2012, Freeman et al., 2015). De-icer application volumes at UK airports are forecast to increase in-line with an increase in aircraft movements of 1 % to 3 % annually up to 2050 (DFT, 2003, DFT, 2013). Simultaneously, expansion of airports to meet this demand alongside a changing global climate will likely generate increasing volumes of surface water runoff, placing unprecedented pressure on existing wastewater infrastructure and pollution prevention strategies, potentially threatening the status of freshwater ecosystems.

The primary environmental concern associated with airport de-icing activities is DO depletion in surface waters that receive storm water runoff, due to the high DO demand exerted by propylene glycol ($C_3H_8O_2$) or potassium acetate ($C_2H_3KO_2$) contained within chemical de-icers (ACRP, 2008, ACRP, 2009). Both $C_3H_8O_2$ and $C_2H_3KO_2$ are associated with extremely high five-day biochemical oxygen demand (BOD_5) and, when diluted within snow melt or rainfall, can potentially achieve BOD_5 concentrations > 20,000 mg L^{-1} within storm water runoff (US EPA, 2000, Corsi et al., 2012). Coupled with high temporal variability in pollutant and hydraulic loads, de-icer contaminated storm water runoff from airports represents a significant threat to freshwater ecosystems if discharged without appropriate treatment (Corsi et al., 2001, ACRP, 2008, ACRP, 2012).

De-centralised treatment of de-icer contaminated storm water runoff at airports using aerated constructed wetlands represents a novel and potentially sustainable approach to the treatment of this source of wastewater. Artificially aerated constructed wetlands appear to offer several advantages in the context of treating de-icer contaminated runoff compared to conventional wastewater treatment technologies, including: simple design; low maintenance; low operational and whole life costs; no regular sludge disposal requirements and resilience to fluctuating hydraulic and pollutant loads (Kadlec and Wallace, 2009, Freeman et al., 2015). The limitations of artificially aerated wetlands for airport applications include the large footprint required to manage large storm water runoff volumes and the attraction of wildlife to wetland habitats which can pose a possible bird strike hazard for aircraft (Blackwell et al., 2008). Despite such limitations, many airports have large areas of land in close proximity to runways that could represent suitable sites for un-planted HSSF aerated wetlands (Wallace and Liner, 2011a, Wallace and Liner, 2011b). In order to inform the application of artificially aerated wetlands at airports, the objectives of the research reported here were: (i) to evaluate the impact of media depth and aeration rates on standard oxygen transfer efficiency (SOTE); (ii) to determine the impact of different aeration diffuser configurations on pollutant removal efficiency; and (iii) to determine optimal pollutant loading rates for effective treatment of de-icer contaminated runoff using aerated wetlands.

2. Methodology and Materials

2.1. Standard Oxygen Transfer Efficiency Tests

Tests were conducted to determine standard oxygen transfer efficiency within four experimental columns filled with 10 mm to 20 mm washed angular limestone gravel, to depths of 1,500 mm, 2,000 mm, 2,500 mm and 3,000 mm (Figure 1). Each individual column was constructed from 220 mm internal diameter medium density polyethylene gas pipe and included sample ports, sample valves, ceramic disc diffusers and airlines. The columns were sealed at the base with electrofusion couplings and bolted stainless steel end caps, with drain valves and 200 mm fine bubble ceramic disc diffusers positioned near to the base. Air was delivered into the columns using an Airmaster model 8/36, 1.5 hp, 24 L oil-free compressor. A 0 – 15 L min^{-1} flow meter was positioned on the compressor outlet and used to regulate the aeration rate delivered into each column. Paired 10 mm internal diameter sample ports were installed on opposite sides of the columns at elevations from the column base equating to 25 %, 50 %, 75 % and 100 % of the total media depth. Sample ports protruded into the column by 20 mm, to avoid sampling water from the internal column wall.

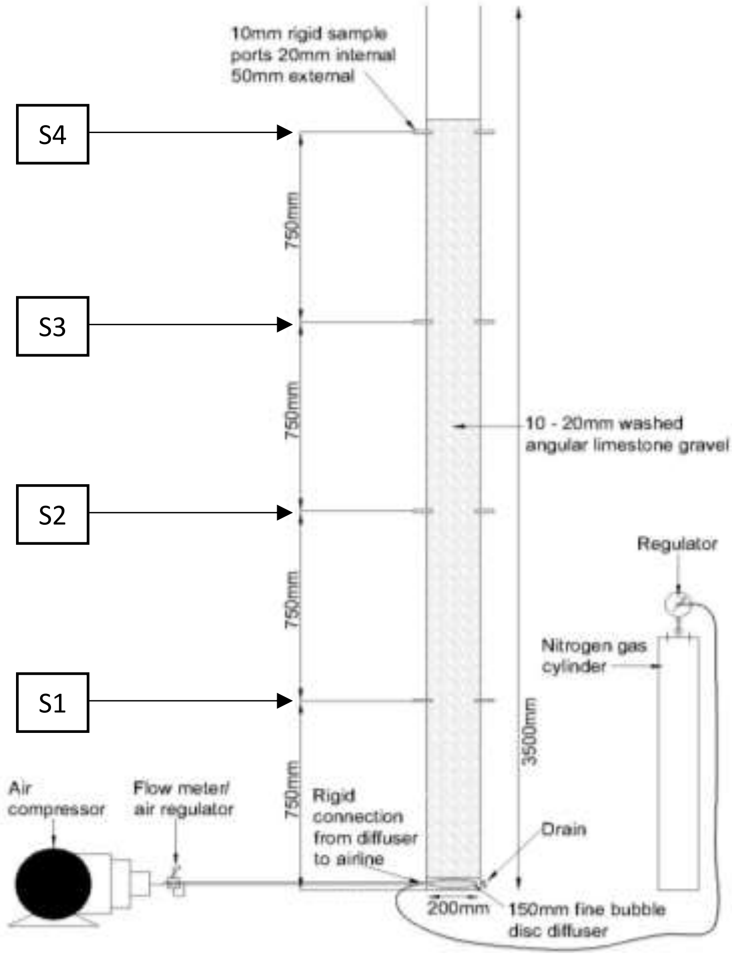

Figure 1. Cross-section of experimental column design for standard oxygen transfer efficiency tests, given the example of a 3,000 mm deep column (not to scale). S1 – S4 = sample locations comprising 10 mm diameter rigid sample ports protruding externally by 50 mm and internally by 20 mm.

Standard oxygen transfer efficiency tests were conducted following procedures described within the ASCE standard (ASCE, 2007). Briefly, DO concentration profiles were generated for each test, which first involved purging nitrogen gas through the column to deoxygenate the potable water within the media pore spaces in the column to < 0.5 mg L^{-1} DO (Ghaly and Kok, 1988). The time taken for DO concentrations to reach the steady-state saturation point during re-aeration via an aeration device at a pre-calibrated flow rate was subsequently measured (Figure 2). The DO concentration data from the point of re-aeration to steady state saturation were analysed using the ASCE-approved DOPar3-0-3 programme non-linear regression model and standardised to conditions of 20 °C water temperature and 1,000 mbar barometric pressure (Stenstrom et al., 2006, ASCE, 2007).

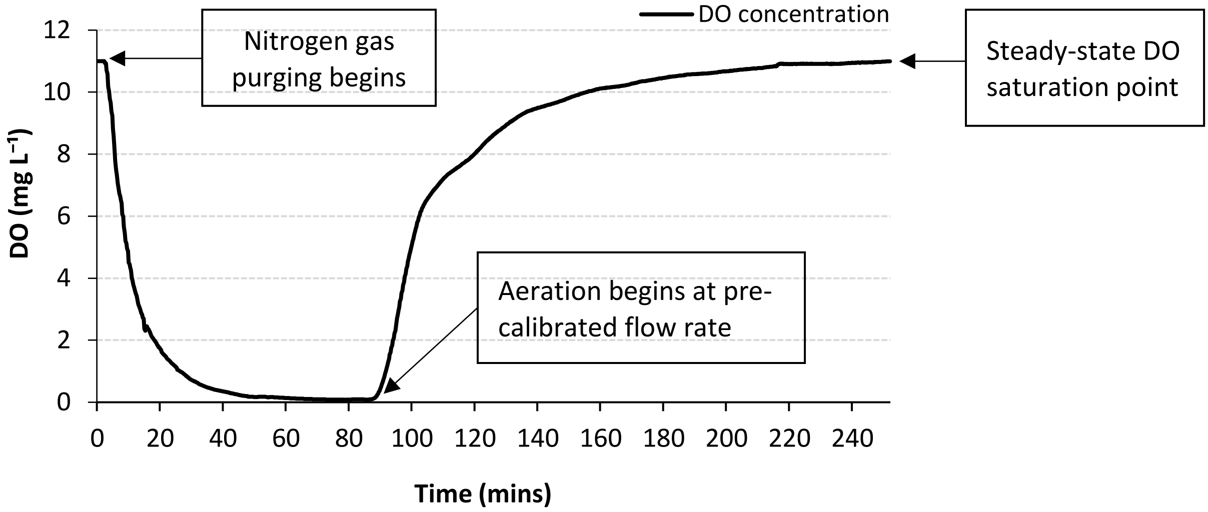

Figure 2. Example dissolved oxygen (DO) profile demonstrating the changing DO concentration during standard oxygen transfer efficiency tests. Example shows the DO profile at sample location S1 (see Figure 1) within the 3,000 mm deep column.

Dissolved oxygen concentrations were measured at ten-second intervals simultaneously at the four sample locations (S1 – S4) using optical multi-parameter probes installed within flow cells (Figure 3). Water was pumped through each flow cell at a rate of approximately 16 ml min^{-1} using a peristaltic pump to create a sealed, self-contained sample loop through which the test water was continuously circulated. Pump tubing was purged prior to each test to remove any trapped air resulting from filling of the column or calibration of the probes. Each probe was calibrated prior to each test following a two-stage calibration procedure for DO, involving the atmospheric saturation point followed by a zero-point calibration within a 1000 mg L^{-1} Na_2SO_3 and 1 mg L^{-1} $CoSO_4$ solution. Three aeration rates (1 L min^{-1}, 2 L min^{-1} and 3 L min^{-1}) were tested with each of the four media depths, to assess the impact of aeration rate and total media depth on SOTE (Table 1). Tests were conducted between 15/11/2014 – 05/01/2015, with each combination of media depth and aeration rate repeated in triplicate. Temperature, DO, redox potential and total dissolved solids concentration were recorded within the potable water prior to the start of each test, confirming that no significant changes in the quality of the potable water occurred between individual tests.

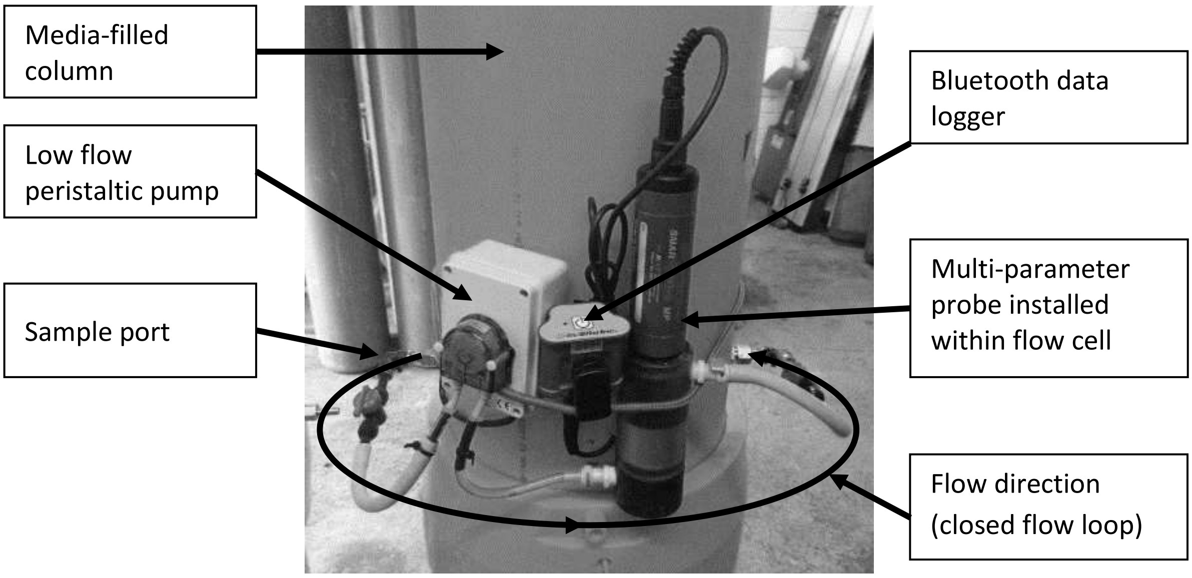

Figure 3. Photograph of an experimental column and the low flow recirculation pump, Smartroll ™ RDO ® multi-parameter probe and flow cell setup to ensure a closed, self-contained flow loop. This setup was replicated at each of the four sample locations during testing to obtain representative DO concentrations at different depths within the column.

Table 1. Experimental design for standard oxygen transfer efficiency tests in media-filled columns with total depths of 1,500 mm to 3,000 mm, operating at aeration rates of 1 L min^{-1} to 3 L min^{-1}

Test No.	Total media depth (mm)	Aeration rate (L min^{-1})
1	1,500	1
2	1,500	2
3	1,500	3
4	2,000	1
5	2,000	2
6	2,000	3
7	2,500	1
8	2,500	2
9	2,500	3
10	3,000	1
11	3,000	2
12	3,000	3

2.2. Pilot-Scale Aerated Wetland Tests

Aeration configuration and optimisation tests were conducted within an un-planted, pilot-scale system that closely replicated an artificially aerated HSSF constructed wetland (Figure 4), located at Manchester Airport, UK (53.356235 °N -2.282445 °W). The pilot-scale system comprised a 1,000 L mixing tank and three cylindrical tanks (1,600 mm deep x 1,400 mm diameter) each of 2,500 L capacity, replicating three aerated wetland treatment cells. A Marlow Watson 520R peristaltic pump was used to dose wastewater from the mixing tank into the first treatment cell. The three treatment cells were positioned in series and connected with 50 mm internal diameter flexi-hose. The elevation of each successive cell decreased by 250 mm, allowing gravity to drive water flow through the treatment system. Each cell comprised a non-insulated narrow inlet distribution zone containing 40 mm to 100 mm diameter crushed brick and a main treatment zone containing the same 10 mm to 20 mm diameter angular limestone gravel media used within the SOTE tests described in Section 2.1. Media within the main treatment zone of each cell was separated from the inlet zone by a slotted mesh screen. As an alternative to wetland vegetation, the media was capped with a porous membrane and a 200 mm deep layer of bark chippings to provide insulation. The total media depth within each cell was 1,400 mm, corresponding to a total media volume of 6.45 m^3 across the three treatment cells. Three 30 mm internal diameter piezometers with 50 mm long screens at the base were installed within each treatment cell to depths of -250mm, -750mm and -1,250mm below the gravel media surface, enabling measurement of physicochemical conditions within each cell. A 210 w Charles Austen ET200 linear diaphragm blower was used to deliver up to 200 L min^{-1} (45 L min^{-1} m^{-3} of media) of air into the system at 0.15 bar of pressure. Braided PVC airlines of 10 mm internal diameter connected the blower to uniformly distributed tubular fine bubble membrane diffusers, which were positioned at the base of each cell. A manifold system was fitted to the aeration line to control the delivery and spatial distribution of aeration volumes into each treatment cell. Prior to undertaking each individual aeration configuration and optimisation test, the system was conditioned for twice the hydraulic retention time (HRT) using the test mass loading rate (MLR) to promote steady-state conditions and microbial acclimatisation. Each aeration configuration and optimisation test was repeated in triplicate.

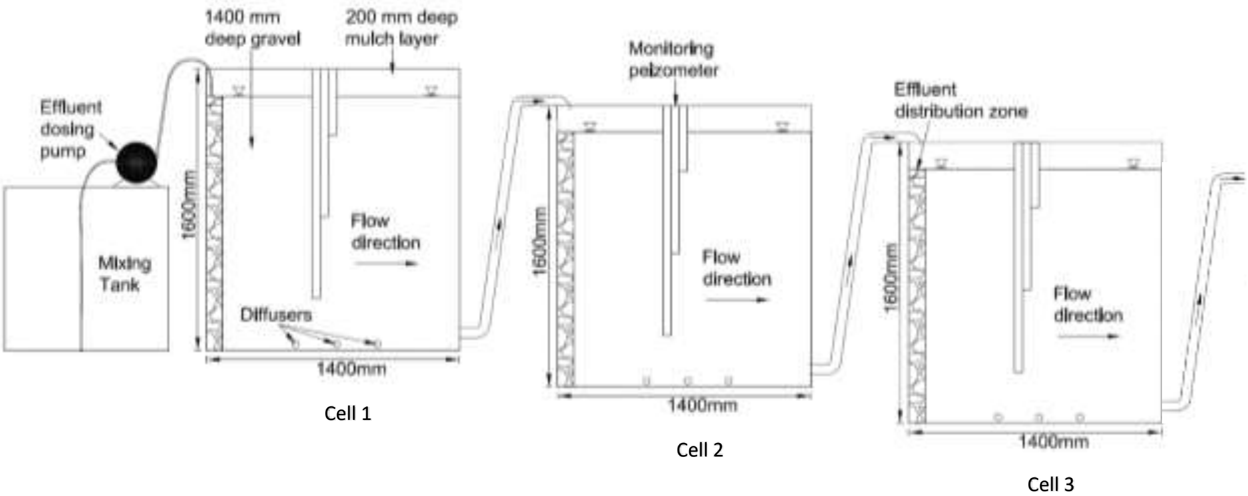

Figure 4. Cross-section of the un-planted pilot-scale system, replicating an artificially aerated horizontal subsurface flow constructed wetland, used for aeration configuration and optimisation tests.

2.2.1. Procedures for Aeration Configuration and Optimisation Tests

Synthetic influent was created to replicate typical chemical oxygen demand (COD), BOD_5 and total organic carbon (TOC) concentrations within winter storm water runoff from airports. The synthetic influent comprised base-flow runoff from the airfield catchment at Manchester Airport containing minimal masses of de-icers and consequently relatively low mean background pollutant concentrations (Table 2).

Table 2. Characteristics and composition of baseflow runoff for Manchester Airport's airfield catchment and aircraft (ADF) and pavement (PDF) de-icers which were combined to create the synthetic influent used within the aerated wetland tests

Parameter	Unit	Base-flow runoff [a]	ADF [b]	PDF [c]	Experiment synthetic influent		
					L	M	H
BOD	mg/l	52.4	354,000	270,000	831	1,355	1,853
COD	mg/l	109	834,000	330,000	1,206	2,405	3,404
TOC	mg/l	40.6	-	-	424	1,104	1,534
TSS	mg/l	21.9	-	-	61	123	162
pH		7.6	7	10.6	7.1	6.8	7

[a] Data recorded from samples collected from Manchester Airport's airfield catchment 12/11/2014 - 27/03/15 during baseflow conditions, defined as catchment discharge volume < 1.00 L sec^{-1} (n = 123) [b] Values for neat Kilfrost ABC-S Plus aircraft de-icing fluid recorded from material data safety sheet. Active ingredient is propylene glycol [c] Values for neat Safegrip pavement de-icing fluid recorded from material data safety sheet. Active ingredient is potassium acetate plus corrosion inhibitors. L, M, H = low, medium and high respectively, defined as 0.2%, 0.3% and 0.4% volume of de-icer: volume of baseflow runoff water respectively (-) Data unknown or not applicable.

Other contaminants potentially present within the baseflow runoff include surfactants, solvents, triazoles, polycitric aromatic hydrocarbons (PAHs), aldehyde, benzine, volatile organic compounds (VOCs) and sulphates, which are deposited within airport catchments during standard airport operations such as aircraft and ground vehicle washing and maintenance, refuling and combustion of aviation fuels (Sulej et al., 2011, Sulej et al., 2012). Whilst these contaminants were not directly measured in the research reported here, it is assumed that only low concentrations would have been present within the baseflow runoff because mobilisation and transport from within airport catchments mainly occurs during storm water runoff events (Sulej et al., 2012). Baseflow runoff was spiked with aircraft and pavement de-icing fluids sourced from Manchester Airport and used widely within the aviation industry in order to create the synthetic influent used within the aerated wetland tests. Three individual influent concentrations were created, defined as low (L), medium (M) and high (H) strength, containing 0.2 %, 0.3 % and 0.4 % volume of de-icer:volume of runoff respectively (Table 2). A nutrient solution of urea and ammonium phosphate was added to the synthetic influent at concentrations consistent with nutrient requirements for optimal microbial growth (Grady et al., 1999, Wallace and Liner, 2010, Wallace and Liner, 2011a). The approximate ratio of BOD_5:N:P was kept constant by adjusting the volume of the supplementary nutrient solution in relation to the BOD_5 MLR for a test, ensuring that microbial processes were not constrained by N or P availability. The volume of influent dosed into the system for each individual test was adjusted in order to maintain the desired HRT within the three treatment cells.

2.2.2. Aeration Configuration Tests within Pilot-Scale Aerated Wetland

Four aeration configurations were tested to establish their impact on pollutant removal efficiency: phased aeration (PA), uniform aeration (UA), inlet-only aeration (IA) and no aeration (NA). The individual configurations were achieved by adjusting the manifold system to alter the spatial distribution and volume of air delivered into each of the three cells (Table 3). Each of these tests was dosed with the L strength influent with a mean BOD_5 concentration of 810 ± 60 mg L^{-1} and an areal MLR of 0.09 ± 0.01 kg d^{-1} m^{-2} BOD_5, alongside a HRT of 1.5 d across all three cells taken together.

Table 3. Summary of hydraulic retention time (HRT), five-day biochemical oxygen demand (BOD_5) influent concentration, areal mass loading rates and air volume distribution throughout the pilot system during aeration configuration tests

Aeration Configuration	HRT [(a)] (days)	BOD_5 (mg L^{-1})	BOD_5 mass (kg d^{-1} m^{-2})	Air vol. (L min^{-1}) distribution across the system		
				Cell 1	Cell 2	Cell 3
Phased (PA)	1.5	834	0.1	100	66.6	33.3
Uniform (UA)	1.5	727	0.08	66.6	66.6	66.6
Inlet-only (IA)	1.5	812	0.09	200	0	0
None (NA)	1.5	868	0.1	0	0	0
Mean		810 ± 60	0.09 ± 0.01			

[(a)] Hydraulic retention time across all three cells taken together calculated following Equation 3. ± 1 standard deviation of the mean.

2.2.3. Optimisation Tests within Pilot-Scale Aerated Wetland

Further to the aeration configuration tests, nine separate tests were conducted between 17/02/2015 – 14/06/2015 to determine the effect of wetland operating conditions on pollutant removal (Table 4). The novel PA configuration, as opposed to UA, IA or NA described in Table 3, was maintained throughout these optimisation tests. Three different HRTs (2.2 d, 1.5 d and 1.1 d) across all three cells taken together were tested to assess the impact of HRT on pollutant removal efficiency. Each of the three HRTs was tested with L, M and H strength influent as described within Section 2.2.1., with respective BOD_5 concentrations of 831 ± 35 mg L^{-1}, 1,355 ± 81 mg L^{-1} and 1,853 ± 99 mg L^{-1}. During optimisation tests, operating conditions were equivalent to mean areal MLRs of 0.07 to 0.28 kg d^{-1} m^{-2} BOD_5, within the typical range of areal MLRs (0.05 to 0.28 kg d^{-1} m^{-2} BOD_5) identified from the literature for uniformly aerated wetlands (Envirodynamics Consulting, 2012, Moshiri, 1993).

Table 4. Summary of operating conditions including hydraulic loading rate (HLR), hydraulic retention time (HRT), five-day biochemical oxygen demand (BOD_5) influent concentration and influent areal mass loading rates used during aerated wetland optimisation tests one to nine [(a)]

Test No.	HLR (m^3 d^{-1})	HRT [(b)] (days)	BOD_5 [(c)] (mg L^{-1})	BOD_5 (kg d^{-1} m^{-2})
1	1	2.24	864 (L)	0.07
2	1.5	1.49	834 (L)	0.10
3	2	1.12	795 (L)	0.12
Mean (tests 1 - 3)			**831 ± 35**	**0.10 ± 0.03**
4	1	2.24	1,286 (M)	0.10
5	1.5	1.49	1,444 (M)	0.17
6	2	1.12	1,335 (M)	0.21
Mean (tests 4 - 6)			**1,355 ± 81**	**0.16 ± 0.05**
7	1	2.24	1,812 (H)	0.14
8	1.5	1.49	1,967 (H)	0.23
9	2	1.12	1,782 (H)	0.28
Mean (tests 7 - 9)			**1,853 ± 99**	**0.22 0.69**

[(a)] Tests conducted with a phased aeration (PA) configuration and aeration rate of 44.64 m^3 d^{-1} m^{-3} of media across all three cells taken togeather,
[(b)] hydraulic retention time across all three cells taken together calculated following Equation 3,
[(c)] five-day biochemical oxygen demand (BOD_5) influent concentrations interpreted as L = low, M = medium and H = high strength. ± 1 standard deviation of the mean.

2.2.4. Data and Sample Collection for Pilot-Scale Aerated Wetland Tests

Mean DO concentrations were determined by measuring DO in samples collected from the piezometers installed within each cell on three occassions: at the start; mid-point; and the end of each individual test. Dissolved oxygen was measured using a Hannah 9828 multi-parameter probe within a sealed flow cell, through which approximately 16 ml min^{-1} of sample was pumped from the piezometers using a peristaltic pump. Results were recorded when the probe readings had stabilised, following purging of stagnant water from each piezometer. Further, a total of four spot samples were collected for each individual aeration configuration and optimisation test, involving one sample of the influent and one sample from each of the three treatment cell outlets. Samples from the cell outlets were taken based on calculations of the HRT within each cell and assuming steady state conditions and laminar throughflow within each treatment cell. Samples were collected either manually into new, one litre plastic bottles, or via Aquacell P2 portable water samplers which were connected to the treatment cell outlets.

2.2.5. Chemical Analysis of Water Samples from Pilot-Scale Aerated Wetland Tests

Water samples from the aeration configuration and the optimisation tests were transported to an on-site laboratory and analysed within 48 hours of collection for BOD_5, COD and TOC using standard laboratory procedures. Several duplicate samples were also analysed by an independent UKAS-accredited laboratory for BOD_5 following the ISO/IEC 17025 standard to verify the BOD_5 data that were determined on-site. Digestion of samples on-site for COD and TOC was performed using a LT200 instrument followed by colorimetric determination using a DR2800 photospectrometer. Hach methods were used to standards ISO 6060-1989 for COD and EN 1484 purging method for TOC. Hach Addista LCA standards of 50 ± 4 mg L^{-1} COD and 16.5 ± 3 mg L^{-1} TOC were used to verify COD and TOC results for each batch of samples processed. The analytical limit of detection was 15 mg L^{-1} and 3 mg L^{-1} for COD and TOC respectively. All samples outside of the method range of 3 mg L^{-1} – 150 mg L^{-1} COD and 1.5 mg L^{-1} – 30 mg L^{-1} TOC were discarded and repeated following dilution with deionised water. Analysis for BOD_5 was performed at 20°C using a BODTrak ™ II instrument. Samples were inoculated with a seed solution (PolySeed®) prior to incubation and analysed in accordance to the Hach standard manometric sample dilution, five day test procedure method 8043 (Hach, 2013). Blanks comprising de-ionised water, one nutrient buffer pillow and 35 ml of seed solution were frequently tested and discarded if the BOD_5 concentration was > 0.2 mg L^{-1}. In addition to external laboratory verification, results for BOD_5 were verified on-site using glucose and glutamic acid (GGA) standards of 300 mg L^{-1}, inoculated with 35 ml of PolySeed® solution and incubated at 20 ˚C for five days following appropriate dilution. All GGA standard results were within the maximum standard deviation of the method (± 30.5 mg L^{-1}).

2.2.6. Data Interpretation

Pollutant removal efficiency for the tests described in Sections 2.2.2 and 2.2.3 was calculated as a cumulative percent removal, R, between the influent and the final effluent leaving the third treatment cell following Equation 1, assuming that the system was in equilibrium at the time of sample collection:

$$R = \frac{C_\mathrm{i} - C_\mathrm{o}}{C_\mathrm{i}} * 100 \qquad (1)$$

where:

R = pollutant removal efficiency (%)

C_i = mean influent concentration across triplicate tests (mg L^{-1})

C_o = mean final effluent concentration across triplicate tests (mg L^{-1})

Areal mass pollutant loading rates (kg d^{-1} m^{-2}) were calculated in accordance with Equation 2 (Kadlec and Wallace, 2009):

$$MLR = \frac{Q * C_\mathrm{i}}{A} \qquad (2)$$

where:

MLR = mass pollutant loading rate (kg d^{-1} m^{-2})

Q = volumetric flow rate (m^3 d^{-1})

C_i = influent pollutant concentration, i.e. BOD_5 (mg L^{-1})

A = wetland area (m^2)

Hydraulic retention time (HRT) was calculated in accordance with Equation 3 (Çakir et al., 2015, Metcalf and Eddy Inc, 1991):

$$HRT = \frac{\pi\, r^2 * \phi * d}{Q} \qquad (3)$$

where:

HRT = hydraulic retention time (days)

$\boldsymbol{\pi}$ = pi

r = cell radius (m)

ϕ = media porosity (40 %)

d = media depth (m)

Q = influent flow rate ($m^3\ d^{-1}$)

2.3. Statistical Analysis

Two-way analysis of variance (ANOVA) and Tukey's-b tests were used to test the effects of media depth and aeration rate on SOTE within the experimental column tests. One-way ANOVA and Tukey's-b tests were performed on sample data from the aeration configuration tests to assess the effects of aeration configuration within the system on COD, BOD_5 and TOC removal efficiency. Separate two-way ANOVA and Tukey's-b tests were performed on data from the aerated wetland optimisation tests to assess the effects of HRT and influent strength on COD, BOD_5 and TOC removal efficiency. Data normality and homogeneity of variances were determined by Shapiro-Wilk and Levenne tests respectively, revealing normal distributions and variances to be homogeneous within all datasets. All statistical analyses were conducted using IBM SPSS 20 and significant effects were accepted at $p < 0.05$.

3. Results

3.1. Effects of Aeration Rate and Media Depth on Standard Oxygen Transfer Efficiency in Media-filled Columns

Aeration rate was inversely related to SOTE ($F(2,24) = 28.13$, $MSE = 14.10$, $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed that aeration rates of 1 L min^{-1} resulted in significantly higher SOTEs compared to aeration rates of 3 L min^{-1}. No significant difference was found between SOTEs under aeration rates of 1 L min^{-1} compared to 2 L min^{-1} or aeration rates of 2 L min^{-1} compared to 3 L min^{-1}. The effect of media depth on SOTE was not significant, nor was there a significant interaction effect between aeration rate and media depth on SOTE. Whilst there was no significant effect of media depth on SOTE, a consistent trend was observed with increasing media depth resulting in an increase in mean SOTE under each of the three aeration rates (Figure 5).

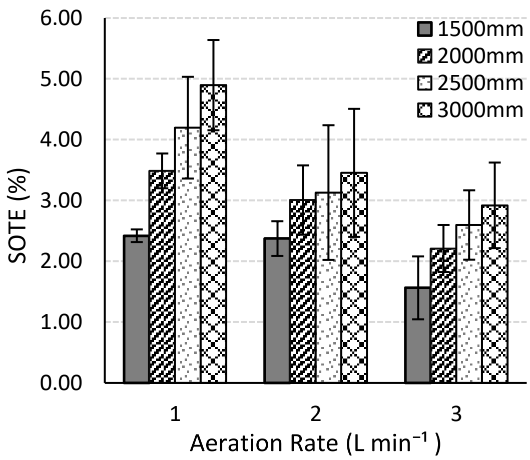

Figure 5. Standard oxygen transfer efficiency (SOTE) in media-filled columns of 1,500 mm to 3,000 mm depth at aeration rates of 1 L min^{-1} to 3 L min^{-1}. Columns represent the mean SOTE for each depth and each aeration rate tested (n = 3 for each combination of aeration rate and media depth). Error bars represent ± 1 standard deviation of the mean.

3.2. Impact of Aeration Configuration and Sample Position on Dissolved Oxygen Concentration within the Pilot-Scale Aerated Wetland

The effect of aeration configuration on mean DO concentrations within pore water in the pilot-scale aerated wetland cells was significant ($F(3,24) = 84.19$, $MSE = 2{,}158$, $p \leq 0.0001$). Post-hoc Tukey's-b tests indicated that the PA configuration resulted in significantly higher mean DO concentrations within the treatment cells in comparison to UA, IA or NA configurations, whilst there was no significant difference in mean DO concentrations between UA, IA and NA configurations. The effect of sample position (cell number) on DO concentration was also significant ($F(2,24) = 57.19$, $MSE = 1{,}466$, $p \leq 0.0001$). Post-hoc Tukey's-b tests indicated that no significant difference in DO concentrations was observed between cells one and two, however DO concentrations were significantly higher in cell three compared with cells one and two. The interaction between aeration configuration and sample position also had a significant effect on DO concentration ($F(6,24) = 45.61$, $MSE = 1{,}169$, $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed that significant differences in DO concentrations across the three cells were observed in PA and UA tests, but not within either IA and NA tests in which DO concentrations remained at 0 mg L^{-1} across all three cells (Table 5).

Table 5. Summary of mean dissolved oxygen (DO) concentrations (mg L^{-1}) determined from piezometer samples within treatment cells one to three during aeration configuration tests. n = 9 for each aeration configuration and cell position combination

Aeration configuration	**Position within system**		
	Cell 1	**Cell 2**	**Cell 3**
Phased (PA)	0	2.1 ± 1.2	8.3 ± 0.2
Uniform (UA)	0	0.7 ± 0.4	0.8 ± 0.4
Inlet-only (IA)	0	0	0
None (NA)	0	0	0

± 1 standard deviation of the mean

3.3. Impact of Aeration Configuration and Sample Position on Pollutant Removal within the Pilot-Scale Aerated Wetland

Aeration configuration had a significant effect on COD ($F(3,24) = 327.57$, $MSE = 3,403$, $p \leq 0.0001$), BOD_5 ($F(3,24) = 361.21$, $MSE = 3,665$, $p \leq 0.0001$) and TOC ($F(3,24) = 98.81$, $MSE = 2,412$, $p \leq 0.0001$) removal efficiency within the pilot-scale wetland (Table 6, Figure 6). Post-hoc Tukey's-b tests confirmed that significant increases in removal efficiency for each of these pollutants occurred in the order NA < IA < UA < PA. Across the NA to PA aeration configurations, the mean removal efficiency increased from 37 % to 92 % for COD, from 45 % to 98 % for BOD_5 and from 46 % to 92 % for TOC. In parallel with increased pollutant removal efficiency, final effluent concentrations of COD, BOD_5 and TOC decreased significantly across the NA-IA-UA-PA aeration configurations, with the lowest final effluent concentration for each parameter achieved under the PA configuration (Table 6). Sample position also significantly influenced pollutant removal efficiency for COD ($F(2,24) = 364.47$, $MSE = 3,787$, $p \leq 0.0001$), BOD_5 ($F(2,24) = 512.26$, $MSE = 5,197$, $p \leq 0.0001$) and TOC ($F(2,24) = 197.77$, $MSE = 4,828$, $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed significantly higher COD, BOD_5 and TOC removal efficiencies within cells one and two compared to cell three. Further, a significant interaction effect between aeration configuration and sample position through the system was also observed in terms of pollutant removal efficiency for COD ($F(6,24) = 62.18$, $MSE = 645.98$, $p \leq 0.0001$), BOD_5 ($F(6,24) = 82.00$, $MSE = 831.95$, $p \leq 0.0001$) and TOC ($F(6,24) = 35.55$, $MSE = 819.09$, $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed that pollutant removal efficiency was significantly higher within treatment cells one and two compared to cell three for COD, BOD_5 and TOC under all aeration configurations, except for the IA configuration in which no significant difference between treatment cells two and three was observed.

Table 6. Summary of influent concentration, final effluent concentration and mean pollutant removal efficiency (%) for chemical oxygen demand (COD), five-day biochemical oxygen demand (BOD_5) and total organic carbon (TOC) within a pilot-scale aerated wetland operating under four different aeration configurations (n = 3 for each aeration configuration)

Aeration configuration	COD			BOD_5			TOC		
	Influent (mg L^{-1})	Final effluent (mg L^{-1})	Removal efficiency (%) [(a)]	Influent (mg L^{-1})	Final effluent (mg L^{-1})	Removal efficiency (%) [(a)]	Influent (mg L^{-1})	Final effluent (mg L^{-1})	Removal efficiency (%) [(a)]
Phased (PA)	1,217 ± 28	98 ± 13	92 ± 1	834 ± 63	21 ± 5	98 ± 1	430 ± 11	35 ± 6	92 ± 2
Uniform (UA)	1,130 ± 48	246 ± 25	78 ± 1	727 ± 19	36 ± 6	95 ± 1	575 ± 106	68 ± 13	88 ± 4
Inlet-only (IA)	1,193 ± 6	676 ± 21	43 ± 2	812 ± 29	421 ± 61	48 ± 7	544 ± 57	266 ± 10	51 ± 6
None (NA)	1,161 ± 23	730 ± 44	37 ± 4	868 ± 21	477 ± 55	45 ± 6	796 ± 18	428 ± 35	46 ± 6

[(a)] Cumulative pollutant removal efficiency (%) determined from influent to final effluent concentration, see Equation 1

± 1 standard deviation of the mean

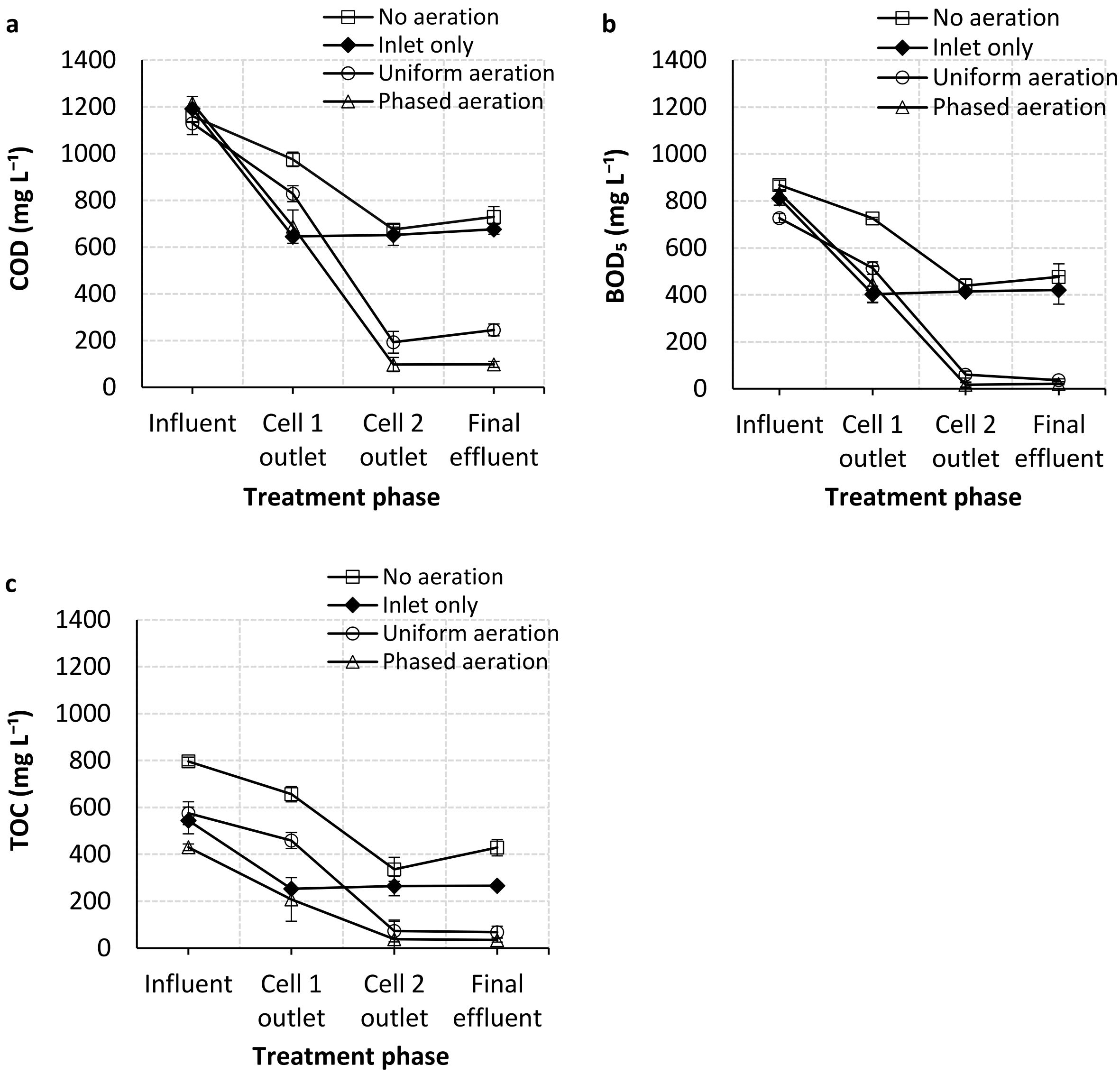

Figure 6. Mean reduction of (a) chemical oxygen demand (COD), (b) five day biochemical oxygen demand (BOD_5) and (c) total organic carbon (TOC), concentrations (mg L^{-1}) throughout the pilot-

scale aerated wetland from influent to final effluent (cell 3 outlet, Figure 4), when tested under different aeration configurations.

3.4. Impact of Hydraulic Loading Rate and Influent Strength on Pollutant Removal Efficiency

A summary of the results from the nine different optimisation tests (three hydraulic retention times * three influent strengths, see Table 4) is reported in Figure 7. A significant effect of HRT on pollutant removal efficiency was observed for COD ($F(2,18) = 105.40$, $MSE = 1{,}467$, $p \leq 0.0001$), BOD_5 ($F(2,18) = 98.40$, $MSE = 1{,}892$, $p \leq 0.0001$) and TOC ($F(2,18) = 28.00$, $MSE = 989.39$ $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed significantly higher pollutant removal efficiencies within tests operating with a HRT of 2.24 d and 1.49 d for all three parameters, compared to tests with a HRT of 1.14 d. No significant effect of influent concentration on pollutant removal efficiency was observed for COD, BOD_5 and TOC, whilst there was also no significant interaction effect between HRT and influent concentration on the removal efficiency of these parameters.

A significant effect of HRT on the final effluent concentrations from the outlet of treatment cell three was observed for COD ($F(2,18) = 69.11$, $MSE = 565{,}484$, $p \leq 0.0001$), BOD_5 ($F(2,18) = 53.50$, $MSE = 302{,}871$, $p \leq 0.0001$) and TOC ($F(2,18) = 18.86$, $MSE = 79{,}994$, $p \leq 0.0001$). Post-hoc Tukey's-b tests revealed that final effluent concentrations were significantly lower for COD, BOD_5 and TOC during tests with HRTs of 2.24 d and 1.49 d, compared to tests with an HRT of 1.14 d. In contrast to pollutant removal efficiency, influent concentration had a significant effect on final effluent concentration for COD ($F(2,18) = 21.59$, $MSE = 176{,}618$, $p \leq 0.0001$), BOD_5 ($F(2,18) = 13.23$, $MSE = 74{,}891$, $p \leq 0.0001$) and TOC ($F(2,18) = 7.59$, $MSE = 32{,}183$, $p = 0.004$). Post-hoc tests revealed that final effluent concentrations were significantly lower during tests conducted with L influent strength compared to tests with M and H strength influents for both COD and TOC. Final effluent BOD_5 concentrations were significantly lower in tests conducted with L and M strength influents in comparison to H strength influents. Despite the significant effect of HRT and influent concentration on final effluent concentrations, no significant interaction effect between these factors was observed in terms of the final effluent concentrations of COD, BOD_5 and TOC.

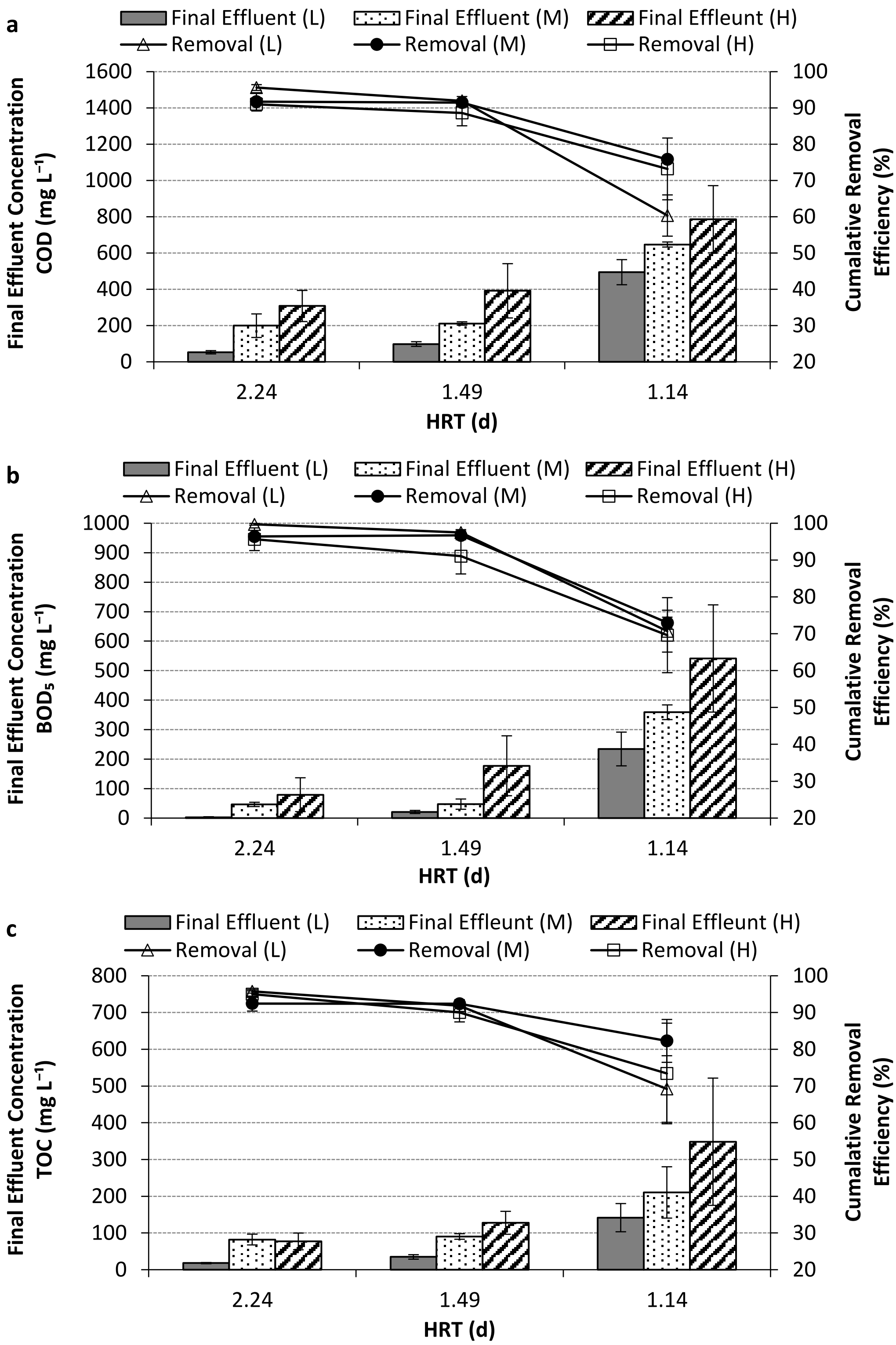

Figure 7. Results of pilot-scale aerated wetland optimisation tests for (a) chemical oxygen demand (COD), (b) five day biochemical oxygen demand (BOD_5) and (c) total organic carbon (TOC), cumulative removal efficiency (%) determined following Equation 1 and final effluent concentration (mg L^{-1}), when tested with mean BOD_5 influent concentrations of 831 mg L^{-1}, 1,355 mg L^{-1} and 1,853

mg L^{-1} representing low (L), medium (M) and high (H) influent strength and three different hydraulic retention times (1.14 d, 1.49 d and 2.24 d).

4. Discussion

4.1. The Impact of Artificial Aeration on Pollutant Removal within Constructed Wetlands

The research reported here demonstrates the essential role of artificial aeration within HSSF constructed wetlands treating high-strength organic wastewater, in order to optimise pollutant removal efficiency and to achieve pollutant concentrations in final effluent that meet the typical requirements of legislation. Without artificial aeration, anaerobic conditions developed throughout the pilot-scale aerated wetland at Manchester Airport, indicating limited availability of DO to support aerobic respiration and thereby resulting in sub-optimal removal efficiencies for COD, BOD_5 and TOC (< 51 % pollutant removal compared to influent conditions). These findings are consistent with the results of pilot-scale constructed wetlands dosed with de-icer contaminated runoff from Edmonton Airport in Canada and Buffalo Airport in the USA, where mean BOD_5 removal efficiencies of only 55 % and 68 % respectively were achieved in the absence of artificial aeration (Higgins et al., 2007).

However, our research also reveals that the precise configuration of artificial aeration can significantly influence pollutant removal within HSSF constructed wetlands. Specifically, and for the first time in an aerated wetland, we provide evidence for the advantages of a phased aeration configuration compared to alternative aeration configurations. In contrast to the IA configuration, both PA and UA configurations resulted in significantly higher pollutant removal efficiencies for COD, BOD_5 and TOC. Whilst high removal efficiencies were achieved for these pollutants under the UA configuration, Figure 6 emphasises that the removal of organic compounds occurs predominantly within the first two-thirds of the treatment system. This is consistent with findings from previous research in aerated wetlands, where up to two-thirds of organic matter was removed within the first quarter of a system (Akratos and Tsihrintzis, 2007, Zhang et al., 2010). In fixed film systems, such as aerated wetlands, biomass growth and microbial respiration of BOD_5 decrease exponentially towards the system outlet, associated with progressive filtration of particulate organic matter and declining concentrations of biodegradable organic carbon (Kadlec and Wallace, 2009). These characteristics potentially result in sub-optimal operating conditions under the UA configuration, involving under-aeration at the inlet zone, resulting in the development of anaerobic conditions, alongside over-aeration towards the outlet of the system resulting in unnecessary aeration, energy consumption and operating costs. In contrast, PA better aligns the delivery and demand for aeration across a treatment system, for example delivering 50 % of the total aeration to the first third of the pilot-scale system used in the research reported here. The PA approach has similarities with tapered aeration designs in biological reactors, such as the activated sludge treatment process, in which 55 % to 70 % of the total air input is typically applied to the first half of a system to address the high O_2 demand near to the inlet (Wolter and Hahn, 1995).

The PA configuration evaluated in our research enhanced pollutant removal efficiency by 15 %, 3 % and 5 % for COD, BOD_5 and TOC, compared to the results obtained under the more conventional UA configuration, although pollutant removal efficiency was only statistically higher for COD under the PA configuration. Despite the enhanced performance of PA compared to UA, IA or NA configurations, DO concentrations within the first treatment cell remained at 0 mg L^{-1}. This indicates high rates of aerobic respiration and insufficient input of air to meet the O_2 demand exerted by the influent within this cell, even with PA. Further, mean DO concentrations within the media pore space of cell three were high (8.3 mg L^{-1}) during PA tests, suggesting excessive inputs of air compared to the O_2 demand exerted by the waste water within this cell. Therefore, opportunities remain to further optimise the delivery of aeration as part of a PA configuration, in order to better match the supply of and demand for DO through a treatment system. For example, aeration devices could be automated to operate only when DO concentrations are within a pre-defined range, switching off when DO concentrations are outside of this range to prevent excessive input of air to a treatment system.

4.2. Impact of Aeration Rate and Media Depth on Standard Oxygen Transfer Efficiency

Whilst our research demonstrates the important role that artificial aeration plays in optimising constructed wetlands for the treatment of high-strength wastewaters, typical aeration devices consume approximately 0.2 kWh of energy per m^3 of water treated (Wallace et al., 2006, Murphy et al., 2012) and can contribute up to 80 % of the total operating costs of a system. The optimisation of design factors that control the efficiency of O_2 transfer from the gaseous to the liquid phase is therefore integral to achieving low cost, sustainable treatment solutions. Our research demonstrates that aeration rate was an important control on SOTE within the media-filled column experiments, in which SOTE decreased significantly from 2.4 % to 1.6 % in 1,500 mm deep columns and from 4.9 % to 2.9 % in 3,000 mm deep columns when aeration rate increased from 1 L min^{-1} to 3 L min^{-1}. The response of SOTE to increasing aeration rate is consistent with previous results within 1,000 mm deep, 2.25 m^3 media-filled tanks in which SOTE decreased from 14.0 % to 5.5 % when airflow rates increased from 10 L min^{-1} to 20 L min^{-1} (Butterworth et al., 2013). These authors also showed that SOTE decreased when media diameter increased (Butterworth et al., 2013), explaining the higher SOTE with the 2 mm diameter media used by these authors compared to the results reported in the current paper which used 10 mm to 15 mm diameter media. The response of SOTE to increasing aeration rate is also consistent with previous studies conducted within open-water diffused aeration systems, which have been examined more extensively than media-filled systems. For example, SOTE decreased from 23.6 % to 18.3 % when aeration rate increased from 0.4 L min^{-1} to 2.3 L min^{-1} within a full-scale, 2,700 mm deep oxidation ditch operating under an extended aeration configuration (Gillot and Héduit, 2000). Further, SOTE decreased from 8.9 % to 7.1 % in a 1,500 mm deep system and from 6.0 % to 4.5 % in a 2,900 mm system when aeration rate increased from 10 L min^{-1} to 40 L min^{-1} within a 3,000 mm deep pilot-scale hypolimnetic aeration system (Ashley et al., 2008).

In open water systems, the aeration rate influences the fluid dynamics of air bubbles, with larger bubbles being produced when the aeration rate through an orifice or diffuser is increased (Ashley et al., 2008). As bubble diameter increases, the buoyancy force increases bubble terminal velocity, which in turn minimises bubble retention time within a water column. Further, the bubble surface area to volume ratio decreases as bubble diameter increases, resulting in a reduction in the relative surface area across which the mass transfer of O_2 from the gaseous to the liquid phase can take place (Gillot and Héduit, 2000, Ashley et al., 2008, Henze et al., 2008). Bubbles also form more slowly when air comes into contact with water at a diffuser orifice under low airflow rates compared to high airflow rates (Davidson and Schüler, 1997), presumably resulting in greater O_2 transfer during the formation of each individual bubble at the diffuser orifice location (Ashley et al., 1991, Gillot and Héduit, 2000). Further, more uniform distribution of air bubbles released from a diffuser orifice can be achieved under low airflow rates, resulting in a more uniform distribution and greater separation between individual bubbles rising through the water column. This serves to reduce bubble coalescence in open-water systems (Ashley et al., 1991, Gillot and Héduit, 2000, Butterworth et al., 2013). However, this is likely to be less important in media-filled systems where bubble hold-up within the media pore space increases bubble coalescence (Fujie et al., 1992, Collingnon, 2006, Butterworth et al., 2013).

Generally, higher SOTEs have been reported within open water systems compared to media-filled systems. However, direct comparisons between the two types of system are complicated by the very different physical characteristics of open water and media-filled systems. A previous study that directly compared SOTE between open water systems and media-filled systems within 1,000 mm deep tanks, established that SOTE was enhanced within the media-filled system compared to the open water system under aeration rates ranging from 10 L min^{-1} to 60 L min^{-1} (Butterworth et al., 2013). In media-filled systems, the media pore space can enhance bubble hold-up compared to open water systems, thereby increasing bubble retention time and potential O_2 mass transfer from the gaseous to the liquid phase. The effect of increased bubble retention time in the research reported by Butterworth et al appeared sufficient to negate the adverse effects of bubble coalescence on SOTE within media-filled systems compared to open water systems. Regardless of SOTE, media-filled systems offer several advantages compared to open water systems for the treatment of high-strength wastewater, such as enhanced sedimentation and filtration of particulate load (Faulwetter et al., 2009). The presence of media also serves to provide a more robust and stable attached microbial population, due to the high surface area of the media surfaces compared to open water systems where microbial colonies are typically in suspension and can be more difficult to maintain (Metcalf and Eddy Inc, 1991). Further, media-filled systems such as aerated wetlands typically provide a higher and more consistent pollutant removal efficiency, alongside lower final effluent pollutant concentrations, compared to conventional open water systems such as lagoons (ACRP, 2013, Freeman et al., 2015).

Another means of enhancing SOTE is to increase bubble retention time by increasing the depth of a treatment system, thereby prolonging the time taken for a gas bubble to rise through a water or media-filled column. For example, our research demonstrated that SOTE more than doubled from 2.42 % to 4.90 % when media depth increased from 1,500 mm to 3,000 mm at airflow rates of 1 L min^{-1} within the media-filled columns, although we note that no statistically significant effect of media depth on SOTE was found. Whilst only limited research has assessed the effect of media depth on SOTE, the increase in SOTE with media depth reported in this paper is generally consistent with the results of previous work in open water systems. For instance, SOTE increased from 4.0 % to 4.6 %, when diffuser depth increased from 0.24 m to 0.32 m below the media surface within a 240 mm internal diameter laboratory scale column operating under aeration rates of 1.6 L min^{-1} (Zhen et al., 2003). Similar findings were observed during laboratory tests conducted within 300 mm internal diameter columns characterised by low water depths and air flow rates of 1 L min^{-1}, in which SOTE increased from 3.9 % to 4.2 % when depth increased from 0.45 m to 0.60 m (Atta et al., 2011).

The substantial increase in SOTE with increased media depth that is reported here suggests that more efficient, cost-effective and sustainable treatment of wastewater could be achieved through increasing the media depth within artificially aerated HSSF wetlands. However, there remain important practical challenges to constructing artificially aerated wetlands at depths > 1,500 mm. Firstly, health and safety issues surrounding the structural stability of excavations (HSE, 2016) would need to be addressed, resulting in stabilisation potentially being required during construction. Secondly, an economic assessment would be required to determine the cost-effectiveness of increasing media depth, given the additional costs for material excavation and disposal where the excavated material cannot be reused on-site. Finally, issues with groundwater levels creating hydraulic pressure beneath a treatment system, potentially damaging any impermeable liner, would need to be considered. However, even relatively small increases in media depth were shown in our research to generate substantial increases in SOTE, for example a 44 % increase in SOTE, from 2.4 % to 3.5 % when media depth increased from 1,500 mm to 2,000 mm at airflow rates of 1 L min^{-1}.

4.3. Optimisation of Hydraulic Retention Time and Pollutant Mass Loading Rate within Artificially Aerated Wetlands

Significantly higher pollutant removal efficiencies for COD, BOD_5 and TOC were achieved under HRTs of 2.2 d and 1.5 d in comparison to an HRT of 1.1 d within the research reported here. No significant difference in pollutant removal efficiency between 2.2 d and 1.5 d HRT suggests that the optimal HRT within the treatment system evaluated here was 1.5 d. This is within the 1.2 d to 6.1 d range of HRTs reported previously for aerated wetlands treating effluents characterised by high influent ammonia or BOD_5 concentrations (Wallace et al., 2006, Wallace and Liner, 2011a, Murphy et al., 2016, Uggetti et al., 2016). Although pollutant removal efficiency was not significantly affected by the range of influent concentrations tested, the concentrations of pollutants in the final effluent were significantly lower when the experimental system was dosed with low and medium strength influents, in contrast to high strength influents. These observations suggest that influent concentration and therefore pollutant MLR is a key factor determining the optimal operation of aerated wetland systems. Optimisation of biological wastewater treatment systems is typically achieved when steady-state MLRs are maintained, thereby facilitating the establishment of a microbial biomass that is fully acclimated to MLR of the influent. The results reported here indicate that optimal areal MLRs are 0.10 kg d^{-1} m^{-2} BOD_5, if the objective is to achieve a low final effluent concentration compliant with typical UK BOD_5 environmental permit limits. However, the treatment system evaluated here also performed at > 91% BOD_5 removal under areal MLRs of up to 0.23 kg d^{-1} m^{-2} BOD_5, although mean final effluent BOD_5 concentrations were 177 mg L^{-1} which exceeds typical environmental permit limits and would therefore require tertiary treatment or discharge as a trade effluent. The long-term operation of aerated wetlands exceeding 0.20 kg d^{-1} m^{-2} BOD_5 is not recommended, due to the potential for microbial clogging of the media pore space, resulting in operational issues including hydraulic malfunctioning, surface flooding or reductions in pollutant removal efficiency (Nivala et al., 2012, Pedescoll et al., 2013).

5. Conclusion

Global population growth, urbanisation, industrialisation and climate change represent significant threats to the ability of freshwater ecosystems to provide critical services to human society. New forms of decentralised treatment, such as artificially aerated wetlands, represent a potentially sustainable approach for mitigating the impacts of wastewater derived from sources such as airports thereby protecting and enhancing freshwater ecosystems and the services that they provide to human society.

The research reported here examined how new approaches, based on artificially aerated HSSF constructed wetlands, can enhance the treatment of high-strength wastewaters from sources such as airports. Using a novel phased-aeration approach, we demonstrate how pollutant removal efficiency per unit of aeration supplied can be significantly enhanced, potentially reducing the treatment costs associated with artificially aerated HSSF constructed wetlands. Optimal operating conditions for a pilot-scale system replicating an aerated HSSF constructed wetland were defined, resulting in > 90% removal of COD, BOD_5 and TOC with a hydraulic residence time of 1.5 d and a mass loading rate of 0.10 kg d^{-1} m^{-2} BOD_5. Further, reduced aeration rate and increased bed media depth were shown to enhance the transfer of oxygen from gaseous to liquid phases, thereby promoting aerobic pollutant degradation processes within aerated wetlands. This research highlights the potential of decentralised, aerated wetland technology to successfully treat high-strength wastewaters, providing additional support for future development and application of the aerated wetland technology to protect and restore freshwater ecosystems.

Acknowledgements

This research was funded by the European Reginal Development Fund (ERDF), through the Centre for Global Eco-innovation (CGE), grant reference X02646PR. This research is published with the permission of Peak Associates Environmental Consultants Ltd and The Manchester Airports Group Plc. We would like to specifically acknowledge the environment, health and safety department at Manchester Airport for their financial support towards materials and equipment and provision of resources including site and laboratory access. We also thank three anyonymous reviewers for their helpful comments which significantly improved this paper.

The authors declare no actual or potential conflict of interest related to the research reported here.

References

ACRP (2008) *Formulations for Aircraft and Airfield De-icing and Anti-icing: Aquatic Toxicity and Biochemical Oxygen Demand*, http://onlinepubs.trb.org/onlinepubs/acrp/acrp_webdoc_003.pdfInterim report 02-01). Available at: http://onlinepubs.trb.org/onlinepubs/acrp/acrp_webdoc_003.pdf.

ACRP (2009) *Deicing Planning Guidelines and Practicies for Stormwater Management Systems*, Washington, US: Trasportation Research BoardReport 14).

ACRP (2012) *Guidebook for Selecting Methods to Monitor Airport and Aircraft Deicing materials*, Washington, D. C.Report 72). Available at: http://www.trb.org/ACRP/Blurbs/167504.aspx.

ACRP 2013. De-icer treatment technologies. http://onlinepubs.trb.org/onlinepubs/acrp/acrp_rpt_099FactSheets.pdf: Transportation Research Board.

Akratos, C. S. and Tsihrintzis, V. A. (2007) 'Effect of temperature, HRT, vegetation and porous media on removal efficiency of pilot-scale horizontal subsurface flow constructed wetlands', *Ecological Engineering,* 29(2), pp. 173-191.

Armstrong, W., Armstrong, J. and Beckett, P. M. (1990) 'Measurement and Modelling of Oxygen Release from Roots of Phragmites Australis', *Constructed Wetlands in Water Pollution Control*: Pergamon, pp. 41-51.

Author (2007)*: Measurement of Oxygen Transfer in Clean Water*. Virginia, US: American Society of Civil Engineers.

Ashley, K. I., Hall, K. J. and Mavinic, D. S. (1991) 'Factors influencing oxygen transfer in fine pore diffused aeration', *Water Research,* 25(12), pp. 1479-1486.

Ashley, K. I., Mavinic, D. S. and Hall, K. J. (2008) 'Effect of orifice diameter, depth of air injection, and air flow rate on oxygen transfer in a pilot-scale, full lift, hypolimnetic aerator', *Canadian Journal of Civil Engineering,* 36(1), pp. 137-147.

Association of European Airlines (2012) *Recommendations for De-icing / Anti-icing Aeroplanes on the Ground*. Available at: http://skybrary.aero/bookshelf/books/2169.pdf.

Atta, N. N., Elbaz, A. A. and Sakr, A. H. (2011) 'Effect of Pressure, Water Depth and Water Flow Rate on Oxygen Saturation Level in Activated Sludge Process', *American Journal of Engineering and Applied Sciences,* 4(4).

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., Marshall, C., McGuire, J. L., Lindsey, E. L., Maguire, K. C., Mersey, B. and Ferrer, E. A. (2011) 'Has the Earth/'s sixth mass extinction already arrived?', *Nature,* 471(7336), pp. 51-57.

Bezbaruah, A. N. and Zhang, T. C. (2005) 'Quantification of oxygen release by bulrush (Scirpus validus) roots in a constructed treatment wetland', *Biotechnology and Bioengineering,* 89(3), pp. 308-318.

Blackwell, B. F., Schafer, L. M., Helon, D. A. and Linnell, M. A. (2008) 'Bird use of stormwater-management ponds: Decreasing avian attractants on airports', *Landscape and Urban Planning,* 86(2), pp. 162-170.

Brix, H. (1997) 'Do macrophytes play a role in constructed treatment wetlands?', *Water Science and Technology,* 35(5), pp. 11-17.

Brix, H. and Schierup, H.-H. (1990) 'Soil oxygenation in constructed reed beds: the role of macrophyte and-atmosphere interface oxygen transport', in Cooper, P.F. & Findlater, B.C. (eds.) *Constructed Wetlands in Water Pollution Control*: Pergamon, pp. 53-66.

Butterworth, E., Dotro, G., Jones, M., Richards, A., Onunkwo, P., Narroway, Y. and Jefferson, B. (2013) 'Effect of artificial aeration on tertiary nitrification in a full-scale subsurface horizontal flow constructed wetland', *Ecological Engineering,* 54(0), pp. 236-244.

CAA (2000) *Aircraft Icing Handbook*: Civil Aviation Authority. Available at: http://www.caa.govt.nz/safety_info/GAPs/Aircraft_Icing_Handbook.pdf (Accessed: 17/12/2012.

Castro, S., Davis, L. C. and Erickson, L. E. (2005) 'Natural, cost-effective, and sustainable alternatives for treatment of aircraft deicing fluid waste', *Environmental Progress,* 24(1), pp. 26-33.

Collingnon, D. (2006) *Insight into oxygen transfer in IFAS processes.* MSc, Cranfield University, Cranfield, UK.

Cooper, P. F. (2005) 'The performance of vertical flow constructed wetland systems with special reference to the significance of oxygen transfer and hydraulic loading rates ', *Water and Science Technology,* 51(9), pp. 81 - 90.

Corsi, S., Mericas, D. and Bowman, G. (2012) 'Oxygen Demand of Aircraft and Airfield Pavement Deicers and Alternative Freezing Point Depressants', *Water, Air, & Soil Pollution,* 223(5), pp. 2447-2461.

Corsi, S. R., Booth, N. L. and Hall, D. W. (2001) 'Aircraft and runway deicers at General Mitchell International Airport, Milwaukee, Wisconsin, USA. 1. Biochemical oxygen demand and dissolved oxygen in receiving streams', *Environmental Toxicology and Chemistry,* 20(7), pp. 1474-1482.

Davidson, J. F. and Schüler, B. O. G. (1997) 'Bubble formation at an orifice in a viscous liquid', *Chemical Engineering Research and Design,* 75, pp. S105-S115.

DEFRA (2012) *Tackling Water Pollution from the Urban Environment.* https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/82602/consult-udwp-doc-20121120.pdf: Department for Environment, Food and Rural Affairs.

DFT, Transport, D.f. (2003) *The future of air transport.* London, UK.

DFT, Transport, D.f. (2013) *UK aviation forecasts.* London, UK: Department for Transport.

Dodds, W. K., Perkin, J. S. and Gerken, J. E. (2013) 'Human Impact on Freshwater Ecosystem Services: A Global Perspective', *Environmental Science & Technology,* 47(16), pp. 9061-9068.

Durance, I., Bruford, M. W., Chalmers, R., Chappell, N. A., Christie, M., Cosby, B. J., Noble, D., Ormerod, S. J., Prosser, H., Weightman, A. and Woodward, G. (2016) 'Chapter Three - The Challenges of Linking Ecosystem Services to Biodiversity: Lessons from a Large-Scale Freshwater Study', in Guy Woodward and David, A.B. (ed.) *Advances in Ecological Research*: Academic Press, pp. 87-134.

EEA 2007. Diffuse Sources. http://www.eea.europa.eu/themes/water/water-pollution/diffuse-sources: European Environment Agency.

EEA 2015. Freshwater Quality. http://www.eea.europa.eu/data-and-maps/indicators/freshwater-quality/freshwater-quality-assessment-published-may-2: European Environment Agency.

Envirodynamics Consulting (2012) *Aerated gravel beds for de-icing waste treatment*, http://vsgc.odu.edu/ACRPDesignCompetition/competitionwinners/2012/2012Environmental_thirdplace1.pdf.

Erdogan, B. 2008. Production and development of de/anti icing fluids for aircraft. *In:* Uludag, Y. (ed.). METU.

Fan, J., Liang, S., Zhang, B. and Zhang, J. (2013a) 'Enhanced organics and nitrogen removal in batch-operated vertical flow constructed wetlands by combination of intermittent aeration and step feeding strategy', *Environmental Science and Pollution Research,* 20(4), pp. 2448-2455.

Fan, J., Zhang, B., Zhang, J., Ngo, H. H., Guo, W., Liu, F., Guo, Y. and Wu, H. (2013b) 'Intermittent aeration strategy to enhance organics and nitrogen removal in subsurface flow constructed wetlands', *Bioresource Technology,* 141, pp. 117-122.

Faulwetter, J. L., Gagnon, V., Sundberg, C., Chazarenc, F., Burr, M. D., Brisson, J., Camper, A. K. and Stein, O. R. (2009) 'Microbial processes influencing performance of treatment wetlands: A review', *Ecological Engineering,* 35(6), pp. 987-1004.

Freeman, A. I., Surridge, B. W. J., Matthews, M., Stewart, M. and Haygarth, P. M. (2015) 'Understanding and managing de-icer contamination of airport surface waters: A synthesis and future perspectives', *Environmental Technology & Innovation,* 3(0), pp. 46-62.

Fujie, K., Hu, H.-Y., Ikeda, Y. and Urano, K. (1992) 'Gas-liquid oxygen transfer characteristics in an aerobic submerged biofilter for the wastewater treatment', *Chemical Engineering Science,* 47(13), pp. 3745-3752.

Ghaly, A. E. and Kok, R. (1988) 'The effect of sodium sulfite and cobalt chloride on the oxygen transfer coefficient', *Applied Biochemistry and Biotechnology,* 19(3), pp. 259-270.

Gillot, S. and Héduit, A. (2000) 'Effect of air flow rate on oxygen transfer in an oxidation ditch equipped with fine bubble diffusers and slow speed mixers', *Water Research,* 34(5), pp. 1756-1762.

Gleick, P. H. (1998) 'Water In Crisis: Paths to Sustainable Water Use', *Ecological Applications,* 8(3), pp. 571-579.

Grady, C. P. L., Daigger, G. T. and Lim, H. C. (1999) *Biological Wastewater Treatment.* New York: Marcell Decker Inc.

Hach 2013. BODTrak II™ - Hach user manual. http://www.manualsdir.com/manuals/332793/hach-lange-bodtrak-ii-respirometric-biochemical-oxygen-demand-bod.html: Hach Lange.

Henriques, J. and Catarino, J. (2017) 'Sustainable value – An energy efficiency indicator in wastewater treatment plants', *Journal of Cleaner Production,* 142(Part 1), pp. 323-330.

Henze, M., Loosdrecht, M. C. M., Ekama, G. A. and Brdjanovic, D. (2008) *Biological Wastewater Treatment, Principals, Modelling and Design.* Cambridge, UK: IWA Publishing.

Higgins, J., Dechaine, L., Minkel, K., Liner, M. and Wallace, S. 2007. Subsurface flow wetlands to treat Glycol-contaminated stormwater from aircraft de-icing operations, BATEA. US: Water Environment Federation.

HSE 2016. Construction: Structural Stability during Excavations. http://www.hse.gov.uk/construction/safetytopics/excavations.htm: Health and Safety Executive.

Huang, Y.-L., Li, Q.-B., Deng, X., Lu, Y.-H., Liao, X.-K., Hong, M.-Y. and Wang, Y. (2005) 'Aerobic and anaerobic biodegradation of polyethylene glycols using sludge microbes', *Process Biochemistry,* 40(1), pp. 207-211.

ISO 2012. **ISO 11076:2012(en)** Aircraft — De-icing/anti-icing methods on the ground. https://www.iso.org/obp/ui/#iso:std:iso:11076:ed-5:v1:en: International Organisation of Standardisation.

Jekel, M., Dott, W., Bergmann, A., Dünnbier, U., Gnirß, R., Haist-Gulde, B., Hamscher, G., Letzel, M., Licha, T., Lyko, S., Miehe, U., Sacher, F., Scheurer, M., Schmidt, C. K., Reemtsma, T. and Ruhl, A. S. (2015) 'Selection of organic process and source indicator substances for the anthropogenically influenced water cycle', *Chemosphere,* 125, pp. 155-167.

Kadlec, R. H. and Wallace, S. D. (2009) *Treatment Wetlands.* 2nd Edition edn. US: Boca Raton, FL : CRC Press.

Malaj, E., von der Ohe, P. C., Grote, M., Kühne, R., Mondy, C. P., Usseglio-Polatera, P., Brack, W. and Schäfer, R. B. (2014) 'Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale', *Proceedings of the National Academy of Sciences,* 111(26), pp. 9549-9554.

Matthews, N. (2016) 'People and Fresh Water Ecosystems: Pressures, Responses and Resilience', *Aquatic Procedia,* 6, pp. 99-105.

Metcalf and Eddy Inc 1991. Wastewater Engineering: Treatment Disposal and Reuse. 3 ed. New York, NY: McGraw-Hill Education.

Moshiri, G. A. (1993) *Constructed Wetlands for Water Quality Improvements.* US: CRC Press.

Murphy, C., Nivala, J. and Wallace, S. 'Aerated Constructed Wetlands Principle, Performance and Application'. ***6th European Waste Water Management Conference & Exhibition***, Manchester: EWWM Aqua Enviro.

Murphy, C., Rajabzadeh, A. R., Weber, K. P., Nivala, J., Wallace, S. D. and Cooper, D. J. (2016) 'Nitrification cessation and recovery in an aerated saturated vertical subsurface flow treatment wetland: Field studies and microscale biofilm modeling', *Bioresource Technology,* 209, pp. 125-132.

Murphy, C., Wallace, S., Knight, R., Cooper, D. and Sellers, T. (2015) 'Treatment performance of an aerated constructed wetland treating glycol from de-icing operations at a UK airport', *Ecological Engineering,* 80, pp. 117-124.

Nivala, J., Knowles, P., Dotro, G., García, J. and Wallace, S. (2012) 'Clogging in subsurface-flow treatment wetlands: Measurement, modeling and management', *Water Research,* 46(6), pp. 1625-1640.

Nivala, J., Wallace, S., Headley, T., Kassa, K., Brix, H., van Afferden, M. and Müller, R. (2013) 'Oxygen transfer and consumption in subsurface flow treatment wetlands', *Ecological Engineering,* 61, pp. 544-554.

Ormerod, S. J., Dobson, M., Hildrew, A. G. and Townsend, C. R. (2010) 'Multiple stressors in freshwater ecosystems', *Freshwater Biology,* 55, pp. 1-4.

Ouellet-Plamondon, C., Chazarenc, F., Comeau, Y. and Brisson, J. (2006) 'Artificial aeration to increase pollutant removal efficiency of constructed wetlands in cold climate', *Ecological Engineering,* 27(3), pp. 258-264.

Pedescoll, A., Sidrach-Cardona, R., Sánchez, J. C., Carretero, J., Garfi, M. and Bécares, E. (2013) 'Design configurations affecting flow pattern and solids accumulation in horizontal free water and subsurface flow constructed wetlands', *Water Research,* 47(3), pp. 1448-1458.

Rajasulochana, P. and Preethy, V. (2016) 'Comparison on efficiency of various techniques in treatment of waste and sewage water – A comprehensive review', *Resource-Efficient Technologies,* 2(4), pp. 175-184.

Schindler, D. W. (2012) 'The dilemma of controlling cultural eutrophication of lakes', *Proceedings of the Royal Society B: Biological Sciences*.

Smith, V. H. and Schindler, D. W. (2009) 'Eutrophication science: where do we go from here?', *Trends in Ecology & Evolution,* 24(4), pp. 201-207.

Stenstrom, K. M. and Rosso, D. 'Oxygen Transfer in a Biological Aerated Filter'. *WEFTEC*, Washington D. C: Water Environment Foundation, 175 - 184.

Stenstrom, M. K., Leu, S. Y. and Jiang, P. 'Theory to Practice: Oxygen Transfer and the New ASCE Standard', *Water Environment Federation Technical Exhibition and Conference*, Dallas, US.

Sulej, A. M., Polkowska, Z. and Namiesnik, J. (2011) 'Analysis of Airport Runoff Waters', *Critical Reviews in Analytical Chemistry,* 41(3), pp. 190-213.

Sulej, A. M., Polkowska, Z. and Namiesnik, J. (2012) 'Pollutants in Airport Runoff Waters', *Critical Reviews in Environmental Science and Technology,* 42(16), pp. 1691-1734.

Uggetti, E., Hughes-Riley, T., Morris, R. H., Newton, M. I., Trabi, C. L., Hawes, P., Puigagut, J. and García, J. (2016) 'Intermittent aeration to improve wastewater treatment efficiency in pilot-scale constructed wetland', *Science of The Total Environment,* 559, pp. 212-217.

UNESCO (2015) *Water Ecosystem Services. A global Perspective. International Hydrology Series* Cambridge, UK.: United Nations Educational, Scientific and cultural organization (UNESCO) and Cambridge University Press.

US EPA (2000) *Preliminary Data Summary Airport De-icing Operations (Revised)*, Washington DCEPA-821-R-00-016). Available at: http://water.epa.gov/scitech/wastetech/guide/airport/upload/2005_10_07_guide_airport_airport.pdf.

Valiente-Banuet, A., Aizen, M. A., Alcántara, J. M., Arroyo, J., Cocucci, A., Galetti, M., García, M. B., García, D., Gómez, J. M., Jordano, P., Medel, R., Navarro, L., Obeso, J. R., Oviedo, R., Ramírez, N., Rey, P. J., Traveset, A., Verdú, M. and Zamora, R. (2015) 'Beyond species loss: the extinction of ecological interactions in a changing world', *Functional Ecology,* 29(3), pp. 299-307.

Van Meter, K., Thompson, S. E. and Basu, N. B. (2016) 'Chapter 11 - Human Impacts on Stream Hydrology and Water Quality A2 - Jones, Jeremy B', in Stanley, E.H. (ed.) *Stream Ecosystems in a Changing Environment*. Boston: Academic Press, pp. 441-490.

Vorosmarty, C. J., Green, P., Salisbury, J. and Lammers, R. B. (2000) 'Global Water Resources: Vulnerability from Climate Change and Population Growth', *Science,* 289(5477), pp. 284.

Vorosmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R. and Davies, P. M. (2010) 'Global threats to human water security and river biodiversity', *Nature,* 467(7315), pp. 555-561.

Vymazal, J., Greenway, M., Tonderski, K., Brix, H. and Mander, Ü. (2006) 'Constructed Wetlands for Wastewater Treatment', in Verhoeven, J.A., Beltman, B., Bobbink, R. & Whigham, D. (eds.) *Wetlands and Natural Resource Management Ecological Studies*: Springer Berlin Heidelberg, pp. 69-96.

Vymazal, J. and Kröpfelová, L. (2009) 'Removal of organics in constructed wetlands with horizontal sub-surface flow: A review of the field experience', *Science of The Total Environment,* 407(13), pp. 3911-3922.

Wallace, S. (2001) *System for removing pollutants from water*. [Online].

Wallace, S., Higgins, J., Crolla, A., Kinsley, C., Bachand, A. and Vercijls, S. 'High rate ammonia removal in aerated engineered wetlands', *10th International Conference on Wetland Systems for Water Pollution Control*, Lisbon, Portugal.

Wallace, S. and Liner, M. 'Nutrient Limitations in Industrial Treatment Wetlands', *12th International Conference on Wetland Systems for Water Pollution Control*, Venice, Itally: Naturally Wallace Consulting.

Wallace, S. and Liner, M. 2011a. Design and performance of the wetland treatment system at the Buffalo Niagra International Airport http://naturallywallace.com/projects/docs/23_Design%20and%20Performance%20BNIA%2052011.pdf: Naturally Walace Consulting.

Wallace, S. and Liner, M. 2011b. Under ground treatment of airport de-icing fluid. http://www.naturallywallace.com/pub/CategoryView.aspx?id=4.

Wallace, S. D., Liner, M. O., Redmon, D. and Hilderbrand, M. 'Oxygen transfer efficiency in aerated subsurface flow wetlands'. *2nd International Symposium on Wetland Pollutant Dynamics and Control*, University of Tartua, Estonia.

Wolter, C. and Hahn, H. H. (1995) 'Modelling of activated sludge systems Von Derin Orhon und Nazik Artan Technomic Publishing AG, Basel, 1994, ISBN 1-56676-101-8, 601 S., hardcover, sfr 181,—', *Acta hydrochimica et hydrobiologica,* 23(4), pp. 187-187.

World Health Organization 2015. Drinking Water. http://www.who.int/mediacentre/factsheets/fs391/en/.

Wu, S., Wallace, S., Brix, H., Kuschk, P., Kirui, W. K., Masi, F. and Dong, R. (2015) 'Treatment of industrial effluents in constructed wetlands: Challenges, operational strategies and overall performance', *Environmental Pollution,* 201, pp. 107-120.

Zhang, L.-y., Zhang, L., Liu, Y.-d., Shen, Y.-w., Liu, H. and Xiong, Y. (2010) 'Effect of limited artificial aeration on constructed wetland treatment of domestic wastewater', *Desalination,* 250(3), pp. 915-920.

Zhen, H., Petiraksakul, A. and Meesapya, W. (2003) 'Oxygen Transfer Measurement in Clean Water', *KMITNB,* 13(1), pp. 14 - 19.

Zhou, X., Wu, Y., Shi, H. and Song, Y. (2013) 'Evaluation of oxygen transfer parameters of fine-bubble aeration system in plug flow aeration tank of wastewater treatment plant', *Journal of Environmental Sciences,* 25(2), pp. 295-301.

Çakir, R., Gidirislioglu, A. and Çebi, U. (2015) 'A study on the effects of different hydraulic loading rates (HLR) on pollutant removal efficiency of subsurface horizontal-flow constructed wetlands used for treatment of domestic wastewaters', *Journal of Environmental Management,* 164, pp. 121-128.