CRANFIELD UNIVERSITY

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THE USE OF ARTIFICIAL AERATION IN HORIZONTAL SUB-SURFACE FLOW CONSTRUCTED WETLANDS FOR TERTIARY NITRIFICATION

School of Applied Sciences

PhD Thesis

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Academic year 2013-2014

Supervisors:
Dr. Gabriela Dotro and Prof. Bruce Jefferson

March 2014

This thesis is submitted in partial fulfilment of the requirements for the Degree of Doctor of Philosophy

Abstract

Increased treatment capability is required on small sewage treatment works to meet ammonium consents that are tightening to effluent concentrations of below 5 mgNH_4^+-N/L and in some cases as low as 0.5 mgNH_4^+-N/L. Optimisation of existing assets is preferential over the addition or expansion of the works to minimise associated costs and energy usage. Many small works in the UK currently employ horizontal sub-surface flow constructed wetlands (HSSF CWs) that have restricted capability to nitrify due to limited oxygen transfer and as such artificial aeration has been proposed as a potential upgrade technology. To assess the performance of the technology, full-scale sites were monitored in terms of ammonium and solids removal and hydraulic characterisation over 3 years. Supporting pilot studies were carried out to assess the effect of aeration on the planted vegetation and to determine optimum transfer efficiencies. Results indicate aeration of HSSF CWS is a successful technology with regards to enhanced nitrification on tertiary small works sites, evidenced by all tested scenarios achieving ammonium effluent concentrations below 3 mg/L on sites receiving 0.1-13.0 gNH_4^+-N/m^2/d, and provides comparative treatment to higher energy technologies such as rotating biological contactors and activated sludge processes with the benefit of being able to be retrofitted into existing HSSF CWS sites. The presence of media was found to enhance oxygen transfer efficiency by a factor of 3.2 in a packed system compared to a control with optimal operation observed at orifice sizes below 1 mm diameter and air flow rates of 10-20 L/min. The presence of artificial aeration was detrimental to the growth of *Phragmites autralis* and *Typha latifolia* in terms of stunted growth and yellowing of leaves. Potential problems were found in the response to shock ammonium loadings at lowly loaded sites, such as an increase in influent ammonium concentration from 1.2±4.0 mgNH_4^+-N/L to 16.0±8.4 mgNH_4^+-N/L with a peak concentration of 33.2 mgNH_4^+-N/L. Further optimisation to rectify the issue is proposed in terms of increasing the abundance and activity of the inherent nitrifying population.

Acknowledgements

Above all I want to thank my supervisor Professor Bruce Jefferson for always manning up and for going above and beyond in his efforts to ensure this thesis became a reality. Absolutely forever indebted.

Of the many others to whom I am indebted include Dr. Gabriela Dotro, my second supervisor and no doubt the hardest worker I shall have the pleasure of meeting. Your patience and on-going enthusiasm were always inspiring. Nigel Janes; I know the Pilot Hall of Dreams isn't always your favourite place, but without you we would all be in a big mess. Thank-you for your speed and efficiency and patience in helping me out. Over in building 244 Andrew Richards, Jan Bingham and Maria Biskupsa for help with plant analysis and for having such a well run lab.

Industrial sponsors Yvonne Narroway in the early days, Andrew Richards and Mark Jones. Thanks to Mark in particular for always putting our well being first. Friends and family in particular Dad and Christine, my brother and sister -despite being on the other side of the world, Sarah Garforth and Melanie Lath for constant and unwavering support and encouragement. And Andy Pandy Mcleod, for solidarity and providing a laugh.

And to my partner in crime Peter Cruddas; although the end of the PhD inevitably won't mean the end of constant mood swings; thank-you for your constant unwavering love and understanding always.

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Nomenclature

α = Affect of process water characteristics on mass transfer coefficient

α^M = Affect of media on standard aeration efficiency

β = Effect of process water characteristics on oxygen saturation

Ω = Effect of pressure on oxygen saturation

θ = Effect of temperature on mass transfer coefficient

τ = Effect of temperature on oxygen saturation

τ_t = Tracer detention time

τn = Nominal residence time

ρ = oxygen density (gO_2/m^3)

2ndry = secondary

3ry = tertiary

AA = Artificial aeration

AE = aeration efficiency

AHFCW = Aerated horizontal flow constructed wetland

APHA = American public health association

ASCE = American Society of Civil Engineers

ABG = Above ground biomass

ASP = Activated sludge plant

ANOVA =Analysis of variance

AOB = ammonia oxidising bacteria

BAF = Biological aerated filter

BGB = Below ground biomass

C = Carbon

C = Oxygen concentration in tank (g/m^3)

C_0= oxygen concentration at beginning of test (ppm)

C^* = oxygen saturation concentration (ppm)

C^*_{20} = oxygen saturation concentration at standard temperature (20°C)

Ca = Calcium

$CBOD_5$ = 5 day carbonaceous biochemical oxygen demand

CO_2 = Carbon dioxide

COD = Chemical oxygen demand (mg/L)

Cu = Copper

CSO = Combined sewer overflow

CST = Capillary suction testing

CSTR = Continuous stirred-tank reactor

CW = Constructed wetland

D = Dispersion coefficient

DEFRA = Department for environment, food and agriculture

DM = Dry matter (g)

DO = Dissolved oxygen (mg/L)

e =Blower efficiency (%)

EA = Environment Agency

Fe = Iron

FISH = fluorescence in situ hybridisation

HSSF = Horizontal sub-surface flow

IFAS = Integrated fixed-film activated sludge

K = Potassium

k = Hydraulic conductivity (m/d)

KLa = Volumetric mass transfer coefficient (h^{-1})

$KLa_{(20)}$ = Standard volumetric mass transfer coefficient (h^{-1})

LDPE = Low density polyethylene

MDI = Morrill dispersion index

Mg = Magnesium

Mn = Manganese

N = Nitrogen

Na = Sodium

NOB = nitrite oxidising bacteria

Ni = Nickel

NH_4^+-N = Ammonical nitrogen (mg/L)

N_2O = Nitrous oxide

NO_3-N = Nitrate-nitrogen

NO_2-N = Nitrite-nitrogen

NR = Nitrification rate ($g/m^2/d$)

OTE = oxygen transfer efficiency (%)

OTR = Oxygen transfer rate ($kgO_2/m^2/d$)

OE = oxygenation efficiency (%)

OUR = Oxygen uptake rate ($kg/m^3/d$)

P = Phosphorous

P. australis = *Phragmites australis*

P_1 = Absolute inlet pressure (bar)

P_2 = Absolute outlet pressure (bar)

P_{act} =Actual pressure (bar)

P_{atm} = Atmospheric pressure (bar)

P_w = Power requirement of blower (kW)

p.e. = Population equivalent

PF = Plug flow

Q = Water flow rate (m^3/d)

Q_a = Air flow rate at standard conditions (m^3/d)

Q_{atm} = Air flow at atmospheric conditions (m^3/d)

qPCR = quantitative polymerase chain reaction

R = Gas constant for air (8.314 kJ/k mol K)

RBC = Rotating biological contactor

RTD = Residence time distribution

SAE = Standard aeration efficiency (kgO_2/kWh)

SAF = Submerged aerated filter

SOTR = Standard oxygen transfer rate

SOTE = Standard oxygen transfer efficiency (%/m)

SRF = Specific resistance to filtration

SVI = Sludge volume index

T = Temperature

T. latifolia = *Typha latifolia*

TF = Trickling filter

TKN = otal Kjeldhal nitrogen

TN = Total nitrogen

TSS = total suspended solids (mg/L)

UKWIR = United Kingdom Water Industry Research

V = volume of tank (m^3)

V_a = Active volume of the tank (m^3)

VSSF = Vertical flow sub-surface flow

w = Weight of flow of air (kg/d)

w_0= Mass flow of oxygen in the air stream (kg/d)

Zn = Zinc

Chapter 1

Introduction

Introduction

1.1 Background

The traditional approach to meeting tightening discharge consents in wastewater treatment has focussed on asset replacement and/or inclusion of additional end of pipe assets. Whilst generally successful at meeting the treatment requirements it can incur substantial increases in cost and energy and hence is not sustainable as a long term strategy. This is perhaps most apparent in the case of small rural sewage works where there is a desire for low energy and low maintenance treatment through the use of passive technologies (Brookes, 2013). Continuance of the philosophy is being challenged by the need to meeting tightening ammonium standards as set under the Water Framework Directive (2000) where traditionally, existing sites would be retrofitted with intensive technologies such as activated sludge processes or upgraded with tertiary nitrification processes such as submerged aerated filters or tertiary rotating biological contactors.

The majority of nitrogen entering wastewater treatment plants exist in either organic nitrogen or ammoniacal-nitrogen species with the latter representing the predominate component at between 66%-90% of the total nitrogen in raw wastewater (Sattayatewa et al, 2010; Gajewsko, 2011). The Organic N is further readily transformed to inorganic ammonia through the biological process of ammonification. In wastewater, typically at a pH of 6-9, ammonia predominately exits in its ionised state (ammonium – NH_4^+). Nitrifiers then convert NH_4^+ to nitrite (NO_2) and then to nitrate (NO_3) utilising available dissolved oxygen that is accordingly depleted by the nitrifiers and other organisms present unless replenished at a sufficient rate. In the absence of oxygen, denitrifiers then utilise the available nitrate molecules for respiration resulting in the production of nitrogen gas which is readily released into the atmosphere.

Ammonium-nitrogen (NH_4^+-N) is principally removed by the process of nitrification. Nitrification is a two-step, microbially mediated process requiring a constant supply of oxygen and describes the autotrophic biological oxidation of ammonium (NH_4^+) via nitrite (NO_2^-) (Equation 1) to nitrate (NO_3^-)(Equation 2):

$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O$ (mainly carried out by *Nitrosomonas*) (1)

$NO_2^- + 0.5O_2 \rightarrow NO_3^-$ (mainly carried out by *Nitrobacter*) (2)

The process requires 4.57 kg of oxygen to remove 1 kg NH_4^+-N in addition to the demand exerted by the faster growing heterotrophic bacteria for removal of organic matter. As such, the process requires the addition of oxygen to the system resulting in the use of energy intensive technologies such as activated sludge plants (ASPs) typically operating on large-scale works. In comparison, small works (serving a < 2000 population equivalent (p.e.)) contain predominately low energy technologies, favouring trickling filters (TFs) and rotating biological contactors (RBCs) as the main stage of treatment. These works account for a significant number of assets in the UK, where it is estimated that approximately 79 % of all works fit this description (DEFRA, 2012). Compared to the large-scale works, these sites offer a greater capacity to re-develop and adapt existing technologies. A typical small works process flow sheet consists of a primary settlement tank for capture of solids, followed by a secondary biological system for removal of biodegradable organic matter, suspended solids and nutrients and tertiary treatment for final polishing (Figure 1.1).

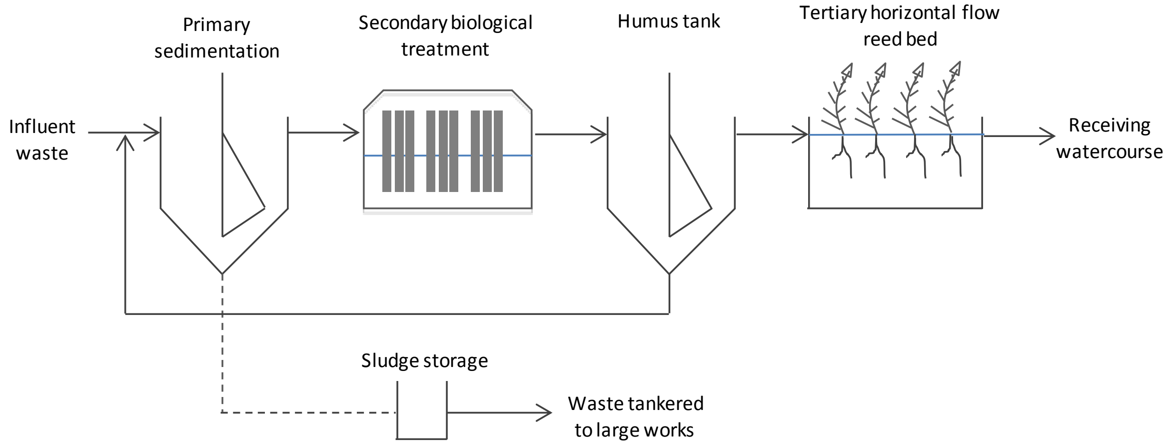

Figure 1.1 Typical process flow sheet for a small sewage works

The technologies generally achieve adequate organics and solids removal and can achieve NH_4^+-N removal down to 5 mgNH_4^+-N/L. However, with ammonium consents typically dropping below this and some cases down to as low as 0.5 mgNH_4^+-N/L (Pearce, 2003) the capacity of the works requires further expansion. One of the most common options in the small works context that has the potential to fill this space are constructed wetlands (CWs) which are now a well-established low energy technology utilised on small wastewater treatment plants.

Horizontal sub-surface flow (HSSF) constructed wetlands are passive wastewater treatment systems commonly used for tertiary suspended solids, organic matter, and nitrate removal. Partially treated wastewater is fed to the bed via a 'v' notched influent distribution trough onto cobbles of 50-200 mm to enable even distribution of the influent. The remainder of the bed is filled with 6-12 mm river gravel through which the wastewater flows horizontally through the permanently saturated media to improve quality via physical, chemical and biological processes (Figure 1.2). The wastewater is kept below the gravel surface (controlled by an adjustable standpipe) to ensure contact with the biofilm attached to the gravel media and to mitigate against overland flow and consequent by-passing of the major treatment pathways. The effluent is then captured in a collection chamber and discharged to the receiving water course.

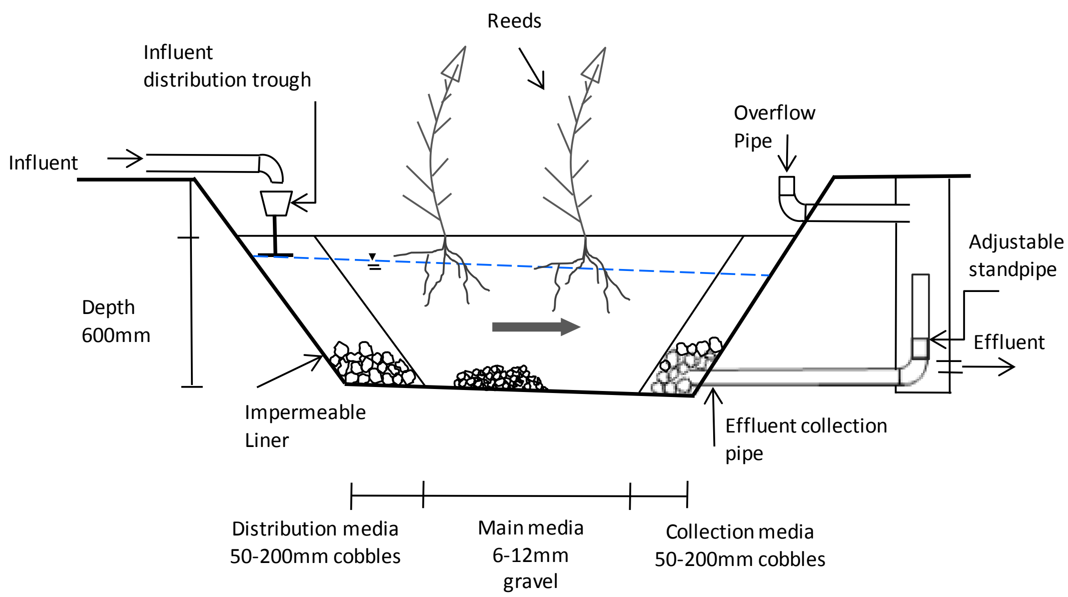

Figure 1.2 Schematic of a typical HSSF CW (Adapted from Knowles et al, 2011)

Oxygen transfer in conventional HSSF CWs occurs through convection and diffusion from the air to the surface water. Transfer is generally poor with estimated oxygen transfer rates of 0.3-3.2 $gO_2/m^2/d$ (Kadlec and Wallace, 2009;

Tyroller et al, 2011) resulting in a residual dissolved oxygen (DO) below 0.5 mg/L leading to low oxygen availability in HSSF CWs being cited as the primary cause of their poor nitrification capabilities (Cottingham et al, 1999, Vymazal and Kröpfelová, 2008). It follows that existing HSSF CWs sites are not equipped to meet future tightening NH_4^+-N consents facing small sewage treatment works. To illustrate, 10 years of historical data from a small works site serving 400 p.e. (Severn Trent Water data) shows adequate organics removal as measured by 5 day carbonaceous biochemical oxygen demand ($CBOD_5$) and total suspended solids (TSS) to a commonly used 20/30 mg/L standard 100 % of the time. Ammonium removal was shown to be less stable and exceeding a future target of 4 mgNH_4^+-N/L over 50 % of the time (Figure 1.3).

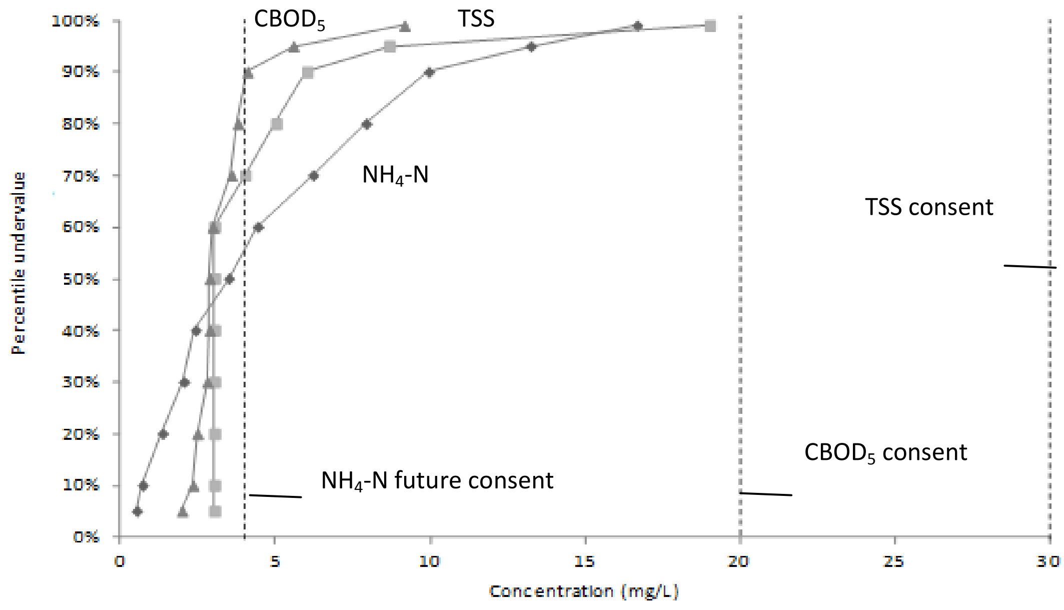

Figure 1.3 $CBOD_5$, TSS and NH_4-N effluent concentrations and typical future consents for HSSF for a small sewage works (p.e. 400). Severn Trent Water data, 2000-2010)

In response to the issue, recent innovations, coupled to the desire for more sustainable treatment approaches, has seen consideration of the CW technology for nitrification increase in terms of both scientific investigation (Ouellet-Plamondon et al, 2006, Loupasaki and Diamadopoulos, 2013) and implementation (Pearce, 2013). Increased oxygen transfer has been investigated in terms of vertical flow systems (and associated variants) that allows oxygen transfer via intermittent loading achieving oxygen transfer rates of 28-100 $gO_2/m^2/d$ or artificial aeration of HSSF wetlands, via a blower and coarse bubble air diffusers (Cottingham et al, 1999; Ouellet-Plamondon et al, 2006; Wallace, 2001). The latter technology has been adopted in the US with successful results (Kadlec and Wallace, 2009) where the systems are designed for low-moderate hydraulic and solids loading rates; typical median values in the US are 0.02 $m^3/m^2/d$ and 2 $g/m^2/d$ (Knowles et al, 2011). However, in many countries, loading rates are considerably higher with, for instance, the equivalent UK figures being 0.12 $m^3/m^2/d$ and 7 $g/m^2/d$ for hydraulic and solids loading respectively (Knowles et al, 2011). Highlighting that direct technology transfer may not be applicable and further investigation of the technology is required under UK conditions. This extends to understanding the relative benefits and weaknesses of the technology over and above alternative options such as tertiary RBCs, TFs and submerged aerated filters (SAFs) (Boller et al, 1994).

1.2 Project aims and objectives

1.2.1 Aim

Investigate the processes affecting the performance of artificial aeration in horizontal sub-surface flow constructed wetlands and establish the feasibility and appropriateness of its application for tertiary nitrification on small sewage treatment works.

1.2.2 Objectives

1. Review the available literature in order to understand nitrification potential in sub-surface flow constructed wetlands and to discuss their future outlook and challenges with a focus towards tertiary nitrification;
2. Assess the appropriateness of aerated CWs for delivery of tertiary nitrification across a range of typical situations through case study assessment to understand the overall efficacy and potential challenges with the adoption of the technology;
3. Investigate the effect of media presence on the efficacy and design of the aeration system;
4. Determine the effect of artificial aeration on plant growth and development;
5. Combine the findings of the three years investigation to advise on the impact on the design basis of an aerated HSSF CW, including suggestions of process optimisation and the role of aerated CWs within current UK.

1.3 Thesis structure

The thesis is presented as a series of chapters formatted as papers for publication (Figure 1.4, Table 1). All papers were written by the first author, Eleanor Butterworth and have been edited by Prof. Bruce Jefferson and Dr. Gaby Dotro. All experimental work was undertaken by Eleanor Butterworth with the following exceptions: Chapter 3: Hydraulic conductivity, tracer tests and solids characterisation by Philip Onunkwo as part of his MSc thesis. Chapter 4: Capture of video footage of bubble pathways in a visualisation column plus assistance with oxygen transfer experiments by Melanie Beneteau as part of her placement requirements. Chapter 6: Hydraulic conductivity and tracer tests by Gabriela Mansi as part of her placement requirements.

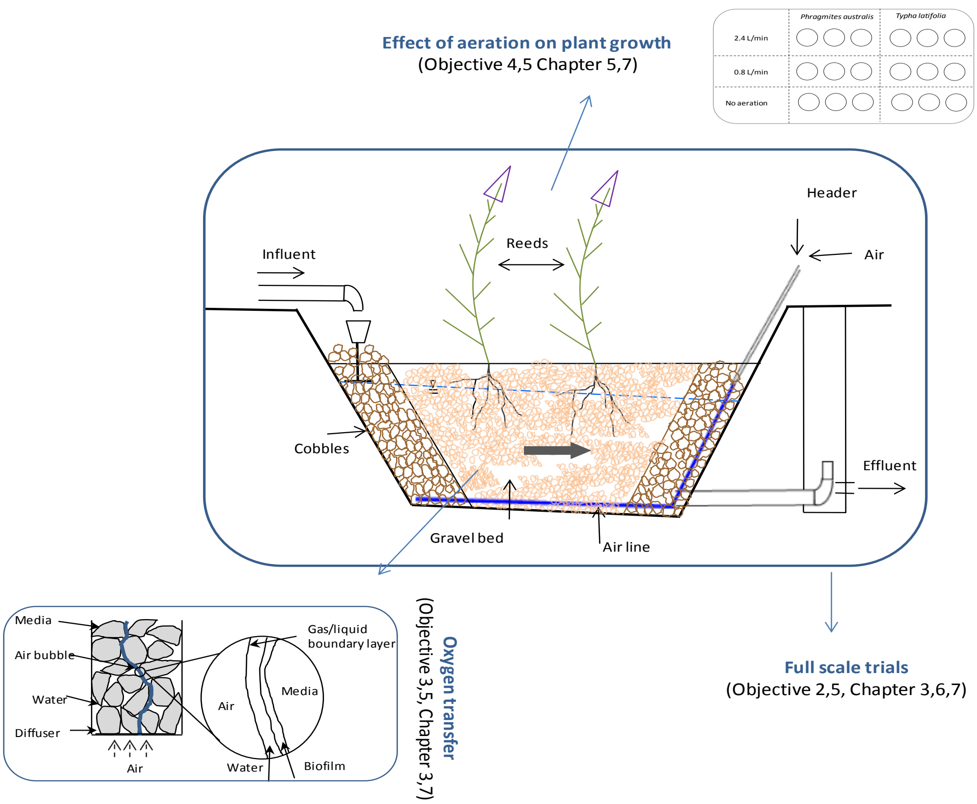

Figure 1.4 Thesis overview

A literature review was carried out in Chapter 2 in order to understand nitrification potential in sub-surface flow constructed wetlands. This was achieved by an overview of oxygen transfer theory and the factors affecting it,. Findings then informed a discussion of their future outlook and challenges with a focus towards tertiary nitrification (Chapter 2, Butterworth E., Dotro G., Richards A., Jones M., B Jefferson. (2014) Considering the potential for tertiary nitrification in sub-surface flow constructed wetlands: a review. *Ecological Engineering*. In preparation).

Chapter 3 assessed the initial performance of a full scale aerated HSSF CW by measurement of contaminant removal, hydraulic behaviour and accumulated solids characterisation in comparison to a non-aerated control. A cost comparison of implementing the technology against traditional options was then carried out to understand the opportunity for use of the process compared to the existing alternative process flow-sheet options (Chapter 3, Butterworth E., Dotro G., Jones M., Richard A., Onunkwo P., Narroway Y., Jefferson B. (2013) Effect of artificial aeration on tertiary nitrification in a full-scale sub-surface horizontal flow constructed wetland**.** *Ecological Engineering* 54:236-244).

The focus of Chapter 4 was on the effect of media presence on oxygen transfer efficiency (OTE). The aim of this study was to determine the difference in oxygen transfer rates in a densely media packed bed vs. a non-media system using a non-porous diffuser. Various configurations of diffuser orifice diameter size and air flow rate were tested to determine their affect on associated aeration efficiency.

Supporting bubble visualisation experiments were designed to provide further insight and explanation of the mechanisms occurring within the system. Results then provided the basis for recommendation around energy reduction in artificially aerated wetlands (Chapter 4, Butterworth E., Beneteau M., Dotro G., Jefferson B. (2014) Effect of media on oxygen transfer in packed beds. *Water Research*. Submitted)

Chapter 5 then responded to observations made during the full scale trials (Chapter 3) at the side-by-side test site where poor establishment of the commonly used plant *Phragmites australis* was observed in the presence of aeration; an observation shared elsewhere in the wetland community. Direct quantification of the effect of artificial aeration on plant growth in constructed wetlands in terms of above and below ground biomass and nutrient uptake of two macrophyte species *Phragmites australis* and *Typha latifolia* was carried out to provide quantitative, mechanistic evidence to support any differences between the systems (Butterworth E., Brix H., Dotro G., Jefferson B. (2014) Impact of aeration on macrophyte establishment in sub-surface constructed wetlands used for tertiary treatment of sewage. *Ecological Engineering*. In preparation).

Chapter 6 assessed the translation of artificial aeration to full scale systems of various configurations including intermittent aeration, tertiary combined storm flow and a secondary bed; determined the efficacy at high and variable ammonia loading rates not yet reported in the literature, and further, reported on the longer term trends of initial performance. Performance was assessed in terms of ammonia and solids removal, hydraulic conductivity and mixing patterns and in terms of process robustness to determine whether the use of aerated HSSF CWs is viable as a reliable tertiary process for ammonium removal on small sites (Butterworth E., Mansi G., Richards A., Jines M., Dotro G., Jefferson B (2014) Ammonia removal at four full-scale artificially aerated horizontal flow constructed wetlands. *Ecological Engineering* In preparation).

Table 1.1 Thesis structure with journal submission plan for each chapter

Chapter/ Paper	Objective addressed	Title	Target Journal	Status
2	1,5	Considering the potential for tertiary nitrification in sub-surface flow constructed wetlands: a review	Ecological Engineering	In preparation
3	2,5	Effect of artificial aeration on tertiary nitrification in a full-scale sub-surface horizontal flow constructed wetland	Ecological Engineering	Published
4	3,5	Effect of media on oxygen transfer in packed beds	Water Research	Submitted
5	4,5	Impact of aeration on macrophyte establishment in sub-surface constructed wetlands used for tertiary treatment of sewage	Ecological Engineering	In preparation
6	2,5	Ammonia removal at four full-scale artificially aerated horizontal flow constructed wetlands	Ecological Engineering	In preparation

The overall impact of the research with respect to artificial aeration for tertiary nitrification on small works was then discussed with a focus on the implications of the work, its applicability and potential for further work (Chapter 7). The aim of the final chapter was to bring all the previous chapters together to establish the feasibility and appropriateness of its application for tertiary nitrification on small sewage treatment works within the current process flowsheet.

1.4 References

Boller M., Gujer W., Tschui M. (1994) Parameters affecting nitrifying biofilm reactors. *Water Science and Technology* 29(10–11):1–11

Brookes A. (2013) Optimising LETS: Radical versus progressive innovation. Proceedings of the *2nd Conference on low energy wastewater treatment systems*, 24th September. Solihull, UK

Cottingham P., Davies T., Hart B. (1999) Aeration to promote nitrification in constructed wetlands. *Environmental Technology* 20(1):69–75

DEFRA. (2012) Wastewater treatment in the United Kingdom: Implementation of the European Union Urban Waste Water Treatment Directive – *91/271/EEC*. DEFRA: London, UK

Gajewska, M. (2011) Fluctuation of nitrogen fraction during wastewater treatment in a multistage treatment wetland. *Journal of Environment Protection Engineering* 37:119-128

Kadlec R., Wallace S. (2009) *Treatment Wetlands* (2nd edn) Taylor & Francis Group: Florida, US

Knowles P., Dotro G., Nivala J., Garcia J. (2011) Clogging in subsurface-flow treatment wetlands: Occurrence and contributing factors. *Ecological Engineering* 37(2):99-112

Loupasaki E., Diamadopoulos E. (2013) Attached growth systems for wastewater treatment in small and rural communities: A review. *Journal of Chemical Technology and Biotechnology* 88(2):190-204

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

Pearce P. (2013) Sustainable processes to meet low ammonia consents – the small works challenge. Proceedings of the *2nd Conference on low energy wastewater treatment systems*, 24th September. Solihull, UK

Sattayatewa, C., Pagilla, K., Sharo, R and Pitt, P. (2010) Fate of organic nitrogen in four biological nutrient removal wastewater treatment plants. *Water Environment Research* 82:2306-2315

Tyroller L., Rousseau D., Santa S., García, J. (2010) Application of the gas tracer method for measuring oxygen transfer rates in subsurface flow constructed wetlands. *Water Research* 44(14):4217-4225

Vymazal J., Kröpfelová L. (2008) *Wastewater treatment in Constructed Wetlands with Horizontal Sub-Surface Flow*. Springer

Wallace S. (2001) *System for removing pollutants from water*. US Patent: US 6,200,469 B1

Chapter 2

Considering the potential for tertiary nitrification in sub-surface flow constructed wetlands: a review

Considering the potential for tertiary nitrification in sub-surface flow constructed wetlands: a review

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2.1 Abstract

The challenge of how to maintain or improve wastewater treatment performance without causing an excessive increase in energy or costs is increasingly focussed towards ammonia. On small sewage treatment works in the UK solutions have historically been energy intensive: to divert waste to a larger plant, add a polishing step to the end of the process flow sheet, or upgrade and replace upstream processes. A large proportion of these sites employ constructed wetlands for tertiary treatment and as such this review explores oxygen transfer theory; nitrification performance of existing constructed wetland systems and affecting factors to reveal reduced energy and cost options are available. Consequently, future perspectives include the use of artificial aeration and greater consideration of vertical sub-surface flow systems as they achieve the nitrification capacity in a smaller footprint than horizontal flow systems and where suitable hydraulics permit, can be operated under very low energy demand.

Keywords: Constructed wetlands, nitrification, tertiary treatment

2.2 Introduction

One of the most important challenges facing the management of wastewater discharges concerns the maintenance or improvement of treatment performance, without causing an excessive increase in energy usage or costs. Such a challenge sits at the heart of the water-energy nexus, requiring innovation to offset the negative impacts associated with population growth, energy production and climatic change on the quality and availability of water resources for human consumption and ecosystems alike (Vörösmarty et al, 2000).

Meeting the challenge is of increasing urgency as regulatory organisations around the world are reducing allowable discharge concentrations in order to preserve the receiving bodies and enhance the natural capital associated to them. For example, within the EU, more stringent environmental quality standards are being set as part of the Water Framework Directive (adopted in 2000), requiring discharges to preserve the ecological status of receiving bodies to a level which would occur with minimal anthropogenic impact. Previous activities have largely resolved such concerns in terms of nitrogen at large sewage works, but pressures remain in terms of phosphorus and hazardous chemicals (Gardner et al, 2013). At small–medium scale works, the challenge extends to nitrogen, and in many small works (sub 2000 population equivalence) ammonia.

The traditional solution to the issue has largely resulted in asset replacement or inclusion of additional end of pipe assets, increasing both cost and energy usage. To illustrate, without innovation, the treatment required to meet aforementioned goals has been estimated by the UK Environment Agency to increase CO_2 emissions in the UK by over 110,000 tonnes per year; more than doubling operational and carbon emissions of individual works that require upgrading (Georges et al, 2009). Consequently, re-development and adaption of existing treatment processes is becoming increasingly desired and implemented (Brookes, 2013). This is perhaps most apparent at small works, as they represent a large percentage of the total works and contain predominately low energy technologies. For instance, in the UK, it is estimated that approximately 79 % of all works fit this description (DEFRA, 2012) and are based around trickling filters (TFs) or rotating biological contactors (RBCs) as the main biological treatment stage. Typical effluent quality exceeds the 20/30 (5 day carbonaceous biochemical oxygen demand ($CBOD_5$)/total suspended solids (TSS)) standard they were originally intended to meet and many provide reasonable levels of ammonium removal down to discharge levels of 5 $mgNH_4^+$-N/L. To date, upgrade of the treatment capacity has focussed on enhanced solids removal in processes such as depth filters, micro-screens and horizontal sub-surface flow (HSSF) wetlands in response to the need to improve suspended solids and $CBOD_5$ discharge levels. Looking forward, the challenge is increasingly focussed towards ammonia, with required discharge levels reduced to below 5 $mgNH_4^+$-N/L and in some cases down as low as 0.5 $mgNH_4^+$-N/L (Pearce, 2013). Ammonia removal is predominately derived through aerobic biological degradation and consequently, traditional tertiary treatment processes are ill equipped to achieve this by providing limited pathways for nitrification. Accordingly, upgrade refers to inclusion of tertiary aerobic biological processes such as submerged aerated filter (SAFs) and TFs or replacement / enhancement of the secondary treatment process, with the potential to switch to activated sludge plants (ASPs) or membrane bioreactors where meeting discharge consents is perceived to be particularly challenging. This potentially deviates from the philosophy of small works by failing to meet the aspiration to deliver appropriate treatment whilst maintaining a low impact in terms of energy, chemical usage, maintenance and costs. The divergence between aspiration and treatment need creates an opportunity to consider innovations in existing options that can be adapted to deliver the required pathways for ammonia removal.

One of the most common options in the small works context that has the potential to fill this space are constructed wetlands (CWs) which are now a well-established low energy technology utilised on small wastewater treatment plants. Constructed wetlands are traditionally passive wastewater treatment systems that consist of a lined excavation filled with porous media, planted with emergent macrophytes. Evolution of the concept has produced a variety of CW configurations capable of varying degrees of treatment that can be tailored to specific needs in terms of organics, solids or nutrient removal (Figure 2.1).

In HSSF CWs (Figure 2.1a), wastewater flows horizontally through the permanently saturated media to improve water quality via physical, chemical and biological pathways and is dominated by strong reducing conditions due to limited oxygen availability and hence such configurations have an inherent limit on their nitrification capacity. In contrast, vertical sub-surface flow (VSSF) CWs (Figure 2.1b) intermittently dose the bed from the surface. During draining, the biofilm surrounding the substrate media is exposed to the atmosphere enabling diffusion of the oxygen through the biofilm, creating the conditions required for aerobic biological wastewater treatment to occur (Cooper et al, 1997). The resultant increase in ammonium removal rate has led the technology to become increasingly popular as a secondary treatment process where limited operator input is desired and sufficient land is available (Kadlec and Wallace, 2009; Moir, 2013).

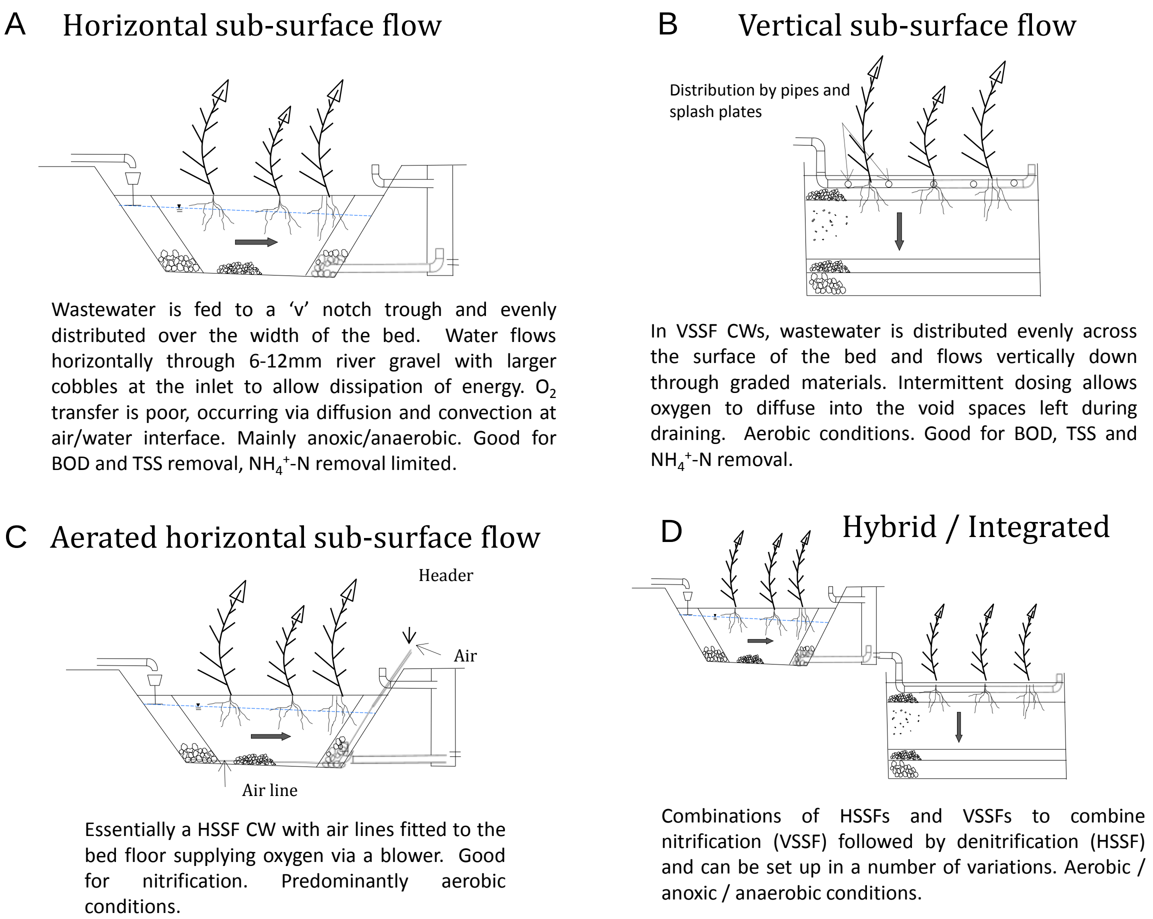

Figure 2.1 Common constructed wetland configurations: a. Horizontal sub-surface flow; b. Vertical sub-surface flow; c. Aerated horizontal sub-surface flow; d. Hybrid/integrated systems

Tidal flow and reciprocating wetlands are classifications of flood and drain systems based on the vertical flow design, whereby the length and frequency of the flood and drain cycles are varied to achieve the desired redox conditions to allow treatment via aerobic and anoxic processes. Where continuous aerobic conditions are required, artificial aeration has become popular (Figure 2.1c); supplying air via the addition of blowers and diffusers placed on the wetland bed (Ouellet-Plamondon et al*,* 2006; Wallace, 2006, Zhang et al, 2010). In addition to these classifications, numerous integrated or hybrid systems have been developed that combine variations of wetland design. Traditionally, vertical flow beds are used for nitrification followed by a horizontal bed for denitrification and solids removal (Figure 2.1d).

Ammonification occurs more rapidly than nitrification, dependant on pH, temperature and C/N ratio. Ammonium (NH_4^+–N) in CW systems can also be reduced by adsorption, plant uptake and volatilization. Very few studies complete N mass balances in tertiary CWs utilised for municipal wastewater treatment due to concerns over effective representation of the different components and the overall complexity (Chen et al, 2014) however it is generally believed that the contribution of these processes to the NH_4^+–N removal is very limited compared with nitrification (Lee et al, 2009).

Recent innovations, coupled to the desire for more sustainable treatment approaches, have seen consideration of the CW technology for nitrification increase in terms of both scientific investigation (Butterworth et al, 2013; Loupasaki and Diamadopoulos, 2013) and implementation (Pearce, 2013). Accordingly, the current paper aims to review the available literature in order to understand nitrification potential in sub-surface flow constructed wetlands and to discuss their future outlook and challenges with a focus towards tertiary nitrification. This is achieved by an overview of oxygen transfer theory and the factors affecting it, followed by a review of ammonium removal performance from the literature and a discussion of the influences affecting ammonium removal rates. The paper then concludes with a discussion of the outlook and challenges with regard to tertiary nitrification on small sewage treatment works.

2.3 Oxygen transfer

The ability to deliver sufficient oxygen to drive nitrification is based on the combination of the demand exerted by the nitrifying biofilms and the diffusional rate of transfer across the stagnant boundary layers surrounding the biofilms. The former constitutes the oxygen uptake rate (OUR) by the microorganism for growth, maintenance and production and is hence linked to the loading rate of the system (Garcia-Ochoa and Gomez, 2009), whilst the latter is known to be rate limiting once the bulk dissolved oxygen (DO) falls below 2 mgO_2/L (Nowak, 2000). The rate and efficiency of oxygen transfer is described in different ways including; the mass of oxygen transferred per unit time (oxygenation capacity, kgO_2/h); the percentage of the oxygen transfer compared to that available (oxygen transfer efficiency – OTE), commonly measured per metre of submergence to normalise against different studies; mass of oxygen transfer per unit of energy consumed (oxygenation efficiency - OE) and the aeration efficiency (AE) both measured in kgO_2/kWh. Details of the respective equations are provided in Chapter 4, Section 4.3.1.

The rate of oxygen transfer is proportional to the area of contact between the liquid and gas phases (ASCE, 1988). Consequently, aerobic processes are designed to maximise this feature in one of two ways: falling films or rising bubbles (Figure 2.2). Falling film systems occur in non-flooded tanks such that the majority of the void space is filled with air. Water is then passed over the biofilm enabling both oxygen and substrate to diffuse into the biofilms that are held in place on packing materials (Figure 2.2a). Typical examples of this technology include TFs, RBCs and VSSF CWs; listed in increasing order of packing density. In all cases the rate of transfer is operationally controlled through the wetting rate, with each packing system having a minimum liquid rate for effective use (Coulson et al, 2002). Reported oxygen transfer efficiencies for such systems are in the region of 5 %/m (Metcalf and Eddy, 2003; Medoza-Espinosa and Stephenson, 1999).

In contrast, rising bubble systems (Figure 2.2b) operate in flooded tanks where small bubbles of air are added at the bottom of the tank and allowed to rise to the surface under the action of gravity. Typical systems include flocculent processes such as activated sludge or sequenced batch reactors as well as biofilm processes such as SAFs, biologically aerated filters (BAFs) and artificially aerated CWs. Transfer rates are controlled by the contact time between the air bubble and the bulk liquid and the specific surface area of the gas/liquid interface. Consequently, smaller bubbles enhance transfer through an increase in both the specific surface area and the contact time such that fine bubble systems (2-5 mm bubble size, Mueller et al, 2002) are preferred over coarse bubble (6-10 mm bubble size, Mueller et al, 2002) in flocculent systems. Operationally this is influenced though the depth of submergence, air flow rate, type of diffuser (material and hole size) and the diffuser density (ASCE, 1988) with, for example, typical oxygen transfer efficiencies in the range of 8-12 %/m for activated sludge systems (Metcalf and Eddy, 2003).

The importance of initial bubble size is less clear in fixed-film rising bubble systems (Figure 2.2c) as media presence can cause the coalescence of fine bubbles, decreasing bubble surface area, resulting in a lower OTE and the break-up of coarse bubbles, increasing OTE (Fujie et al, 1992; Harris et al, 1996). For instance, no real benefit in OTE was observed with respect to the presence of media in pilot trials of an integrated fixed-film activated sludge (IFAS) process where rates remained around 4-7 %/m (Collignon, 2006). In high density packing systems, such as aerated wetlands, an additional impact is seen as the apparent rise rate of the bubbles can be reduced due to bubble hold up in the spaces between the media grains (Butterworth et al, 2014). Direct oxygen transfer measurements in open and packed tanks have shown a 53 % increase in OTE for the latter, although the impacts were strongly linked to gas flow rate and orifice size (Butterworth et al, 2014).

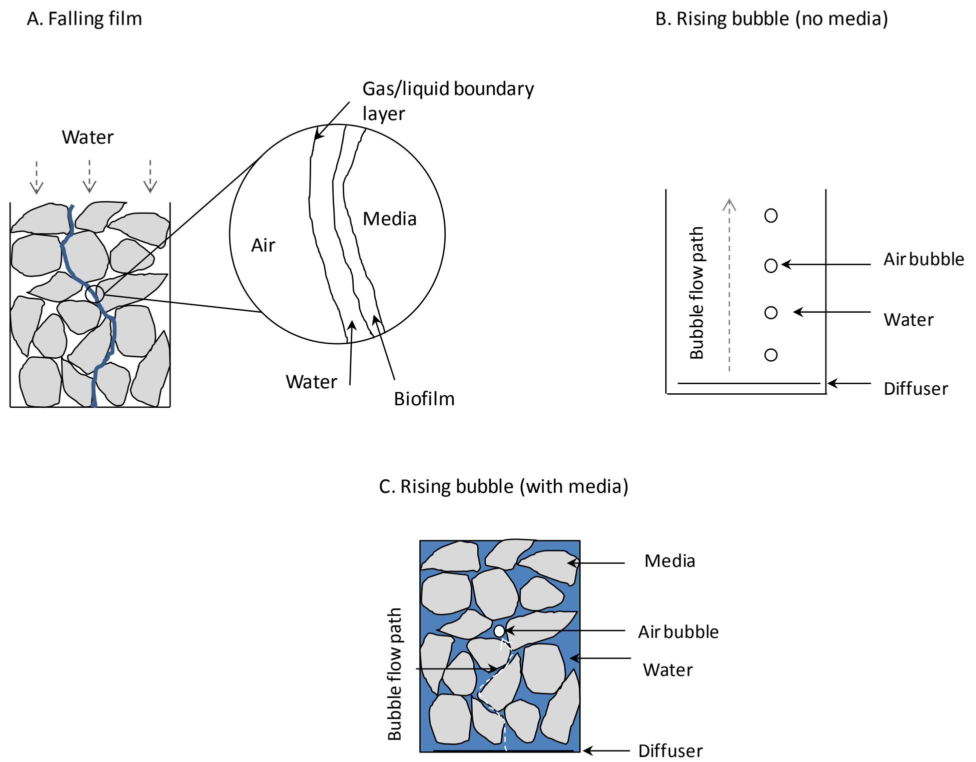

Figure 2.2 Common methods of air delivery to wastewater: a. Falling films; b. Rising bubbles (non-media system) and c. Rising bubbles (media system)

2.3.1 Oxygen transfer in constructed wetlands

In CWs the retrospective oxygen consumption rates are commonly based on a mass balance using water quality data, which has led to overestimation of oxygen transfer rates (Nivala et al, 2012). To illustrate, direct measurement of the OTR by gas tracer methods has shown OTRs of around 0.3-3.2 $gO_2/m^2/d$ in HSSF CWs – lower than previous estimates of 5.4 to 100 $gO_2/m^2/d$ based on mass balances and theoretical calculations (Tyroller et al, 2010). Respirometry techniques have also been adapted from use in activated sludge and applied to measure biological kinetics in VSSF CW systems (Andreottola et al, 2007). Samples are aerated to reach endogenous conditions and the OUR is determined from the response of the dynamic DO profile to a substrate spike. The method has calculated maximum OURs of 2.5-4.4 g $O_2/m^3/h$ and an ammonium removal rate (AR) of 1.8 gNH_4^+-$N/m^3/h$ (Andreottola et al, 2007; Ortigara et al, 2010).

2.3.2 Horizontal sub-surface flow

Oxygen supply in conventional HSSF CWs (Figure 2.1a) is poor and variable, occurring primarily via convection and diffusion from the air to the surface water, with estimated transfer rates of 0.3-3.2 $gO_2/m^2.d$ (Troyller et al, 2010) compared to required consumption rates of 2.4-11.6 $gO_2/m^2.d$ (Nivala et al, 2012). Oxygen transfer from plants in excess of plant respiration requirements are uncertain, but considered insignificant (Brix, 1990; Bezbaruah and Zhang, 2005). Such low oxygen transfer rates lead to a residual DO of around 0.1-0.9 mgO_2/L (Kaseva, 2004; Butterworth et al, 2013); insufficient for nitrification (Nowak, 2000) and consequently, complete nitrification is only considered achievable in lowly loaded systems of up to 2 gNH_4^+-$N/m^2.d$ (Butterworth et al, 2013).

2.3.3 Vertical sub-surface flow

Vertical sub-surface flow CWs (Figure 2.1b) are falling film systems (Figure 2.2a) reported to consume 5.7-156 $gO_2/m^2.d$ (Nivala et al, 2012) and maintain a residual DO of 4.3-6.5 mgO_2/L (García-Pérez et al, 2009; Sousa et al, 2011). Part of the oxygen demand is met via nitrate utilisation during the flooded phases that is subsequently released into the flow during the drain phase (Tanner et al, 1999; Austin, 2006). Effective delivery of the aerobic environments occurs when the sand is not saturated; thus good drainage and distribution is critical (Torrens et al, 2009). The oxygenation processes are affected by the applied hydraulic loading rate in terms of the batch feeding volumes; at any hydraulic loading, larger batch volumes favour oxygen diffusion but reduce retention time and hence treatment (Torrens et al, 2009). Variation in loading approaches has led to a range of estimated average OTRs between 50-90 $gO_2/m^2.d$ (Cooper et al, 1997). Tidal flow (Zhao et al, 2004) and reciprocating operating strategies (Behrends et al, 2001) are based on vertical flow systems, with several flood and drain cycles occurring daily, designed to enhance oxygen transfer and therefore increase nitrification (Tanner et al, 1999). To illustrate, a laboratory study run with a 3 hr fill : 3 hr drain cycle demonstrated that the oxygen demand in the tidal flow system was fully met with OTRs reaching 450 $gO_2/m^2.d$ (Wu et al, 2011). The systems have been reported to deliver the required oxygen quickly with saturation of the biofilm occurring in less than 1 minute (Behrends, 1999).

2.3.4 Artificially aerated systems

Artificially aerated (AA) systems (Figure 2.1c) are fixed-film, rising bubble systems (Figure 2.2c) reported to achieve residual DOs of 3.3-7.0 mgO_2/L (Bezbaruah and

Zhang, 2005, Muñoz et al, 2006) and meet oxygen demand rates of 50-1027 $gO_2/m^2.d$ (Nivala et al, 2012). Studies have been increasing in this area since 1999 (Cottingham et al, 1999; Chazarenc et al, 2009; Fan et al, 2013) and generally relate to aerated HSSF systems, although more recently include flooded vertical flow systems (Tang et al, 2008; Stefanski and Tsihrintzis, 2009). The air delivery configuration varies between systems and includes the use of a 20 cm diameter air diffuser placed at the inlet of the mescosm with air supplied at 6.7 $L/min.m^3$ bed (Ouellet-Plamondon et al, 2006); a 90 mm slotted PVC pipe across the width of the bed at two locations along a 30 m bed, delivering 4.2 $L/min.m^3$ of bed (Cottingham et al, 1999) and 125 mm diameter perforated pipe placed along the width of the bed at four locations across a length of 15.5 m, delivering 32.3 $L/min.m^3$ bed 12 hours a day (Nivala et al, 2007). The latter system used an orifice size of 3 mm indicating coarse bubble aeration akin to those used in SAFs whereas the former systems use fine bubble aeration as used in ASPs.

A paucity of reported data on the relative efficacy of the different designs indicates limited optimisation to date. However, pilot investigations into the impact of hole size and air flow rate have revealed low air flow rates (10-20 L/min) and small hole sizes (0.5-0.8 mm) produced higher SOTEs/m than high flow rates (40-100 L/min) and larger hole sizes (2-3 mm) (Butterworth et al, 2014). The time required to reach the maximum DO took around 5 minutes which compared to 20 minutes reported in a pilot aerated flooded vertical flow systems used for secondary treatment (Fan et al, 2013). The system was intermittently aerated at a range of 1±0.5 $L/min.m^3$ and stopped once the DO reached 3.5 mgO_2/L. DO decline took 60 minutes to drop below 1.0 mgO_2/L enabling total nitrification and good denitrification to occur.

2.4 Nitrification performance

Comparison of data from the collated studies of different wetland configurations revealed median ammonium removal rates of 2.8 gNH_4^+-N/m^2.d for VSSFs, 2.0 gNH_4^+-N/m^2.d for AA systems and 0.5 gNH_4^+-N/m^2.d for the HSSFs, with integrated systems operating at higher removal rates of 9.3 gNH_4^+-N/m^2.d (Table A, supplementary information 2.1). Variation of reported ARs was highest in the integrated (0.4-12.8 gNH_4^+-N/m^2.d) and vertical flow systems (0.1-79.3 gNH_4^+-N/m^2.d) with a two-sided Grubbs test (0.05 significance level) indicating no statistical outliers in the dataset. Variation in both sets is due to the VSSF component and reflects differences in set up, loading rate and the fact that ammonia load is not the rate limiting component in system design and operation. Supporting this, the range of reported ARs for HSSFs was 0.03-7.2 gNH_4^+-N/m^2.d verifying the systems are able to nitrify when operated under appropriate loadings to enable sufficient oxygen transfer and hence were more related to hydraulic and solids loading rate than ammonia loading rate.

Comparison of the tested systems revealed full scale studies utilising bed depths of 0.6-0.7 m and areas of 4-10,000 m^2 whilst pilot/lab scale systems ranged from depths of 0.2-0.6 m and areas of 0.1-5.9 m^2. Analysis of the data indicated general lower ammonium removal rates in lab/pilot systems compared to full scale (Table 1). For instance, underestimations appeared in the cases of VSSFs (31 %) and HSSFs (27 %) although a much closer translation between scales of operation appears to exist in the case of aerated HSSFs (17 %). However, in the case of integrated systems ARs were found to be 6.2 times higher at pilot than at full scale. This is likely to be due to the multiple stages common in integrated systems and the higher ammonium loading rates used in the pilot/lab scale integrated systems compared to those in the full scale (Table

A, supplementary information 1). As such, only data relating to full scale beds and outdoor systems were taken into account for the remainder of the data analyses.

Table 2.1 Comparison of ammonium removal rates (AR) in various wetland systems at full and pilot scale

System	Full scale		Pilot/lab scale			
	Number	**AR** (g/m²/d)	**Number**	**AR** (g/m²/d)	**Total number**	**ΔAR** (g/m²/d)
HSSF	16	0.71	15	0.52	31	-0.19
VSSF	13	2.19	6	1.52	21	-0.67
Integrated	3	1.65	12	11.9	15	+10.3
Aerated HSSF	7	2.30	17	1.91	13	-0.39

For full scale systems, median ammonium removal rates of 2.3 gNH_4^+-N/m^2.d for the aerated systems; 2.2 gNH_4^+-N/m^2.d for VSSFs and 0.7 gNH_4^+-N/m^2.d for the HSSFs were calculated; with integrated systems decreasing from 9.3 to 1.7 gNH_4^+-N/m^2.d (Figure 2.3).

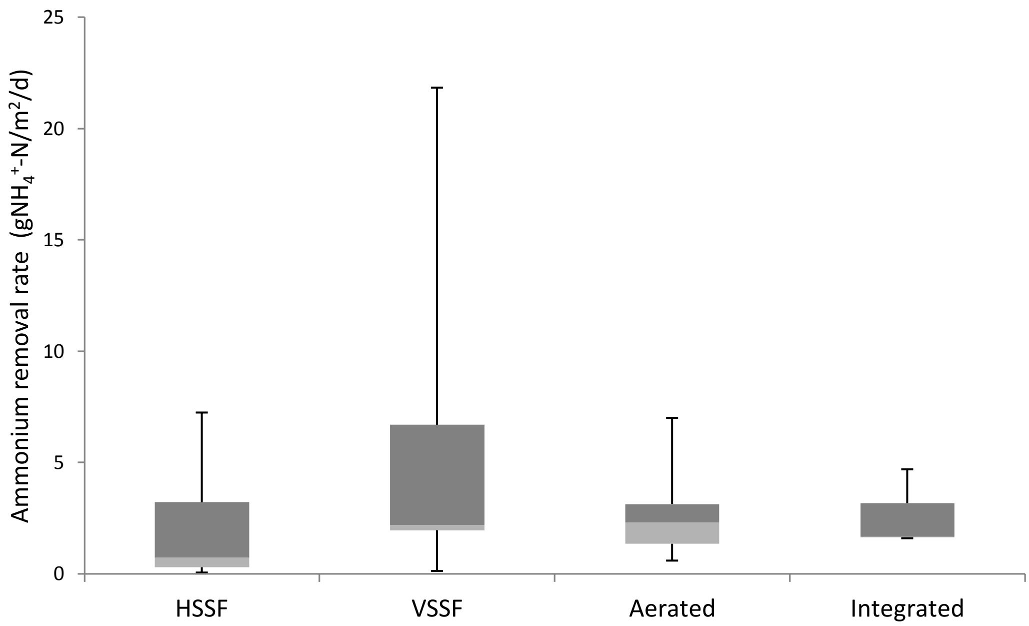

Figure 2.3 Comparison of nitrification rates calculated for various full scale wetland systems. The 'box' represents the 50th percentile range, the line the median and the 'whiskers' the upper 5th and 95th percentiles

2.5 Influences on ammonium removal rates

2.5.1 Loading rates

A strong correlation between NH_4^+-N loading and removal rate was found in all systems (Figure 2.4) in agreement with findings from Tanner et al, (2002) who showed rates of NH_4^+-N removal increased with mass loading for an HSSF CW. Strong correlations were also observed for each of the individual configurations; for instance increasing ammonium loading rates between 0.8-8.0 gNH_4^+-N/m^2.d corresponded to ARs of 0.7-7.2 gNH_4^+-N/m^2.d (R^2=0.99) for HSSF systems older than two years (Figure 2.4). Inclusion of the data from the younger systems (less than 2 years) weakened the relationship, producing an R^2 value of 0.46, as the younger systems produced a maximum AR of 1.8 gNH_4^+-N/m^2.d at a loading rate of 7.7 gNH_4^+-N/m^2.d. The capacity to nitrify also appears related to hydraulic residence time as a study by Trang et al, (2010) revealed a reduction in removal between 0 % and 91 % for a fixed inlet concentration when changing the hydraulic loading rate from 31-146 mm/d corresponding to N loading rates of 1.5 and 6.9 gNH_4^+-N/m^2.d.

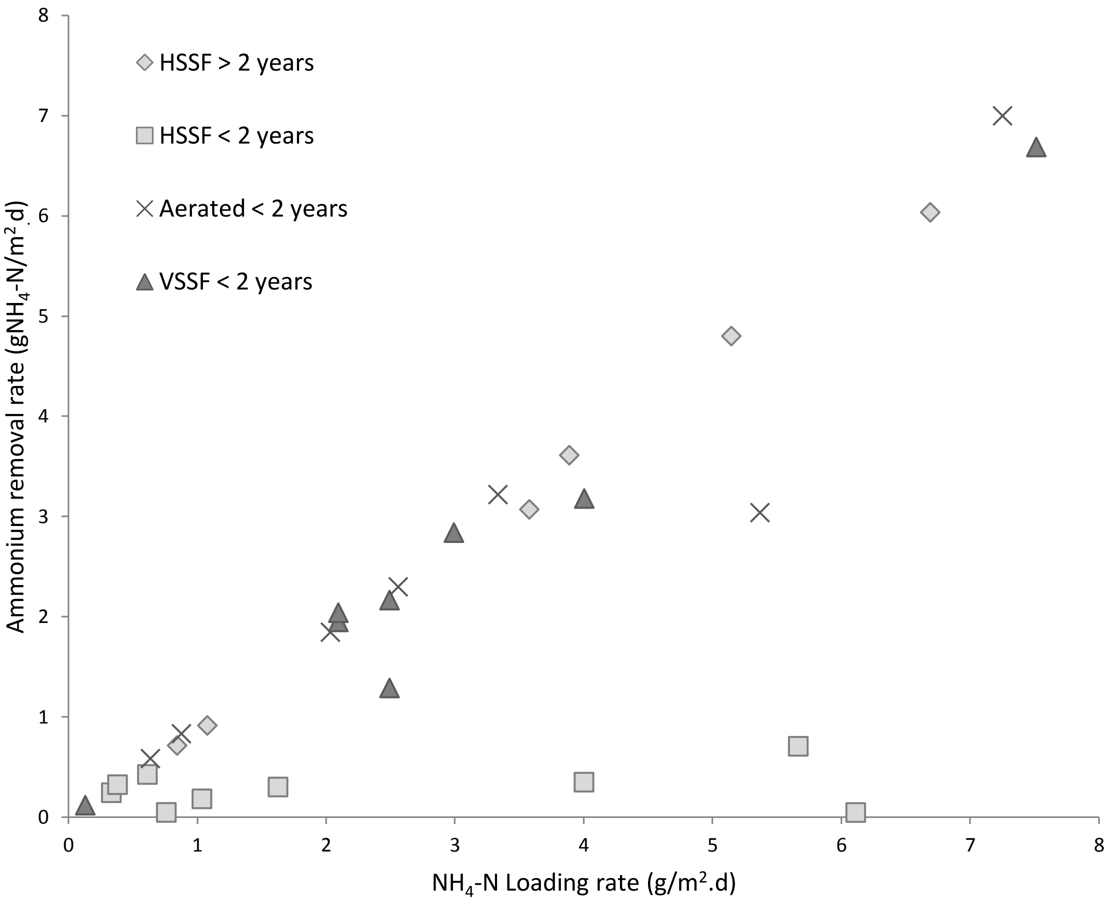

Figure 2.4 Relationship between ammonium removal rate and NH_4-N loading rate from long and short term performance data from full scale HSSF CWs, and full scale aerated and VSSF

*All ARs and loading rates relate to data from the full scale studies taken reported in Table 2. HSSF systems older than 2 years old are considered established. All other data points are from full scale systems less than 2 years old plotted up to loading rates of a maximum of 9 gNH4/m2.d

The majority of studies report on CWs were used for secondary treatment, whilst full scale tertiary systems appear under-represented in the literature (Table A, supplementary information 2.1). Typical removal efficiencies in full scale secondary HSSF CWs varied from 6.5-93.4 % corresponding to effluent ammonia concentrations predominately above 5 mg/L with a range between 3-61 mgNH_4^+-N/L. In comparison, effluent ammonia in the aerobic secondary CWs was lower and ranged between 1.5-

12.0 mgNH_4^+-N/L for VSSFs and 1.0-9.5 mgNH_4^+-N/L for AA HSSFs corresponding to removal efficiencies of 51.8-97.5 % and 95.3-96.7 % respectively.

Lower ARs were reported for full scale tertiary HSSF CWs (0.05-0.22 gNH_4^+-N/m^2/d) with corresponding hydraulic loading rates of 0.01-0.28 m^3/m^2/d (Table A, supplementary information 2.1). In one study, tracer tests identified that the preferential flow paths and dead zones that occurred in the systems were not the cause of the poor efficiency (O'Luanaigh et al, 2010). Further investigation showed the NH_4^+-N:COD ratio was high (1:9), suggesting poor performance could be due to competition from the faster growing heterotrophs utilising available oxygen to degrade organic material, leaving insufficient for nitrifying bacteria to degrade ammonia. Influent NH_4^+-N and organic-N, in contrast, were changing form on a cyclic basis through processes such as mineralisation, immobilisation and plant uptake**.** Equivalent loading rates in tertiary VSSF and AA HSSF CWs ranged from between 0.03-0.53 m^3/m^2/d treating ammonia concentrations up to 50.5 mgNH_4^+-N/L resulting in effluent ammonia concentrations of 0.2-29.2 mgNH_4^+-N/L (Table A, supplementary information 2.1).

2.5.2 Operation

Operational practice has also been shown to influence capacity at lab/pilot scale. For instance, an increase in ammonia removal from 70 % to over 91 % was observed in a continuous compared to an intermittently run (24 hr fill:24 hr drain) VSSF wetland (Zhang et al, 2005). Alteration of the dosing frequency has been shown to influence treatment through its impact on hydraulic retention time in the bed (Stevik et al, 1999). When the dose is applied in fewer, larger volumes the retention time is reduced due to the greater hydraulic driving pressure applied which correspondingly inhibits pollutant contact with the biofilm by reduction in the exchange between the mobile and less mobile water fractions in the bed (Torrens et al, 2009). However, oxygenation is reduced as dosing frequency is increased as it is controlled by the time between batches such that increased frequency can reduce nitrification, requiring a balance to be reached (Molle et al, 2006).

A study of the effect of the flood/drain ratio on performance of VSSF systems resulted in 94 %, 91 % and 63 % ammonium removal in 1:2, 2:1 and 3:0 systems, (days to flood: days to drain), however, total nitrogen removal was highest in the 3:0 system and lowest in the 1:2, whilst COD and total phosphorous removal did not differ significantly between the different ratios (Jia et al, 2010). The study also documented decreased nitrification rates during a period of high BOD_5 (330 $g/m^2/d$) loading rates due to increased competition for oxygen and the formation of thicker heterotrophic biofilms that buried the slow growing nitrifiers and contributed to clogging of the system.

Artificial aeration has been shown consistently to enhance nitrification congruent with the negation of the oxygen limitation in systems operated at high loading rates. To illustrate, a full scale system treating landfill leachate (Nivala et al, 2007) recorded 95 % ammonium removal efficiency (average loading 81 gNH_4^+-N/d) compared to a yearly average of 32 % removal in the system during periods of no aeration (average loading 29 gNH_4^+-N/d). The same has been reported for municipal sewage treatment where a full scale system operating at a loading rate of 5.5 gNH_4^+-N/m^2.d enabled 68 % ammonium removal during aeration compared to 15 % without (Cottingham et al, 1999). Further, Ouellet-Plamondon et al, (2006) recorded summer mass removals of

99 % and 94 % in an aerated system compared to a non-aerated control compared to lower removals of 94 % and 65 % respectively over winter, with systems loaded at 0.7 (summer) and 0.2 (winter) gNH_4^+-N/m^2.d. Equivalent findings have been reported in tertiary nitrification systems where a direct comparison of full scale aerated and non-aerated beds on the same site revealed a difference in effluent ammonia of 0.1±0.05 mgNH_4^+-N/L in the aerated bed compared to 8.6±6.4 mgNH_4^+-N/L in the non-aerated bed (Butterworth et al, 2013).

2.6 Outlook and challenges

In the current context of nitrification, delivery of sufficient air enables CW technology to provide effective treatment of ammonia at either a secondary or tertiary treatment stage in a wastewater flow sheet. Implementation for secondary treatment applications is becoming more commonplace at small rural municipal works in parts of Europe (Kadlec and Wallace, 2009) and private treatment systems more generally (Moir, 2013). In both cases, vertical or artificially aerated horizontal beds are used to ensure sufficient oxygen transfer to drive nitrification. Both systems are shown to be able to reduce ammonium to below 5 mgNH_4^+-N/L at a 95th percentile when operated as a secondary process (Table A, supplementary information 1) unless high hydraulic loading rates (0.53 m^3/m^2.d) or difficult wastewaters (e.g. leachate) are considered. Accordingly, when discussing future outlook it is more pertinent to discuss the potential for use of the technology for tertiary applications where the current preference for HSSF CW results in limited ammonium removal unless they are very lightly loaded (2 gNH_4^+-N/m^2.d).

A relative paucity of data exists for aerobic CWs used for tertiary nitrification but in both VSSF and AA CWs, where studies exist, the data indicates effective treatment congruent with lack of oxygen being the inhibitory factor. Data from both real and synthetic trials suggest effluent ammonia concentrations of less than 1 mgNH_4^+-N/L are possible whilst maintaining treatment levels in terms of solids and $CBOD_5$ compared to non-aerated systems by just aerating standard designs of HSSF CWs (Butterworth et al, 2013). Consequently, the question is about the relative comparison with alternative options to understand the opportunity space that can be occupied.

On sites that already contain HSSF CWs a significant advantage can be attributed to artificial aeration as the upgrade can be conducted as part of the routine maintenance cycle significantly reducing cost and negating the need for new assets. This was confirmed during a recent feasibility assessment of upgrading options on a small sewage treatment plant with an existing HSSF CW (Butterworth et al, 2013). Upgrading with artificial aeration was a more viable option in terms of cost, land and footprint than the traditional options of RBCs, SAFs or TFs.

Whilst the efficacy of treatment is becoming more established, challenges exist in relation to robustness to dynamic events, energy demand and the impacts of aeration on solids accumulation and hydraulic conductivity. Operational experience suggests that the longer HRTs used in artificially aerated wetlands provide enhanced resilience against cold temperatures compared to high rate equivalents such as SAFs and TFs (Pearce, 2013). However, wetlands suffer from the same challenge as all tertiary nitrification systems related to the low substrate concentration encountered during much of the year. For instance, previous studies indicate that feed ammonia concentrations rarely exceed 5 mgNH_4^+-N/L and are often substantially lower (Butterworth et al, 2013; Table A, supplementary information 1). This prohibits establishment of large communities of ammonia oxidising bacteria (AOB) within the beds such that during periods of increased load, available substrate may exceed the cell specific ammonia-oxidation rate of the community [4-10 fmol/cell/hr, (Belluci et al, 2011)]. No direct studies on bacterial abundance or community profile have been reported for aerobic wetlands used for tertiary nitrification systems but investigations on established HSSF CWs revealed 1-3 % of the total community being related to AOB (Krasnits et al, 2009) which increases to around 16 % when assessing aerated secondary VSSF beds (Fan et al, 2013). These represent the limits between which tertiary systems will likely sit and research is required to understand how to increase the active AOB population size in order to enhance nitrification resilience during increased loads. Once a sufficient community exists the technology may be able to emulate the IFAS process whereby nitrification rate can be turned up and down by controlling DO and hence provide a means of dynamic control against variable nitrification demand (Pearce, 2013).

Artificially aerated tertiary systems currently installed use small fixed speed blowers such that excess air needs to be vented making direct measurement of the required energy complex (site observations, author). However, a recent report based around a 750 population equivalent site revealed the use of a 1.5 kW blower (Pearce, 2013). Whilst this generates a very small energy cost (less than £1000 /yr based on UK prices) it represents a high relative energy demand per person at around 2.43 Wh/person comparable to typical levels for activated sludge of 2.45 Wh/person. Further, all energy use on the site was compared which revealed that the aeration system accounted for between 40-50 % of the total energy demand, the second being heating in the operators building. In contrast, energy demand for aeration based on oxygen transfer experiments indicates that only 0.7 kW is required to maintain an adequate DO and so the potential for optimisation exists (Butterworth et al, 2014). Whilst the small size of the blowers restricts concern on an individual site basis, once scaled up across all small works within a region the impact becomes significant. Reduction in the actual power demand also enhances the opportunities for using localised renewable energy sources providing a route for future off grid operation of such sites which would enhance uptake still further.

The majority of systems have been recently installed such that longer term impacts remain unclear on issues associated to solids accumulation, mixing and hydraulic conductivity. Previous mesocosm studies have indicated that the impact of artificial aeration reduces solids accumulation (Chazarenc et al, 2009) and enhances hydraulic conductivity during the initial years of operation (Butterworth et al, 2013). The reduction coincides with changes in characteristics of the solids in terms of volatile solids, specific filtration resistance and sludge volume index suggesting that the solids have transformed. This offers the possibility of extended bed life in aerated systems compared to un-aerated ones but validation is required through long term observations as recent results indicate that the benefits dissipate as the bed ages (Mansi et al, 2013). The equivalent information has not been reported for VSSF systems and so further research is required to understand how tertiary aerobic pathways influence long term operation of CWs in relation to clogging and solids accumulation.

2.7 Conclusions

The potential to successfully treat ammonia in CWs is apparent once sufficient oxygen is supplied to enable aerobic conditions to predominate in the bed. This can be achieved through either reduced loading in HSSF CWs or increased oxygen transfer through the use of aeration in the form of passive (VSSF) or artificial (aerated HSSF) aeration. Either way, a strong evidence base exists to demonstrate the capability of CWs to meet nitrification needs during secondary treatment and accordingly the technology is increasingly used in small rural sites for this function.

The future growth outlook is then towards tertiary nitrification where increasing numbers of small works are requiring upgrade, but application of CWs for the purpose is currently limited. Existing sites provide evidence that CWs can be an effective choice, thus, the challenge for growth increasingly relates comparison to the alternative technologies as well as overcoming the uncertainties associated with solids and hydraulic conductivity and the wider consideration of energy use. Consequently, future perspectives include greater consideration of VSSF systems as they achieve the nitrification capacity in a smaller footprint than HSSF systems and where suitable hydraulics permit, can be operated under a very low energy demand. However, if total nitrogen removal is required, aerated HSSF CWs appear more suitable as delivery of multiple redox environments within a single system can be simply achieved by aerating only a fraction of the bed, enabling total nitrogen removal.

2.8 Acknowledgements

The authors would like to express their gratitude for the financial and resource support of the work from Severn Trent Water Limited and to the EPSRC for their allocation of funding and support of the project.

2.9 References

Agudelo R., Penuela G., Aguirre N., Morató J., Jaramillo M. (2010) Simultaneous removal of chlorpyrifos and dissolved organic carbon using horizontal sub-surface flow pilot wetlands. *Ecological Engineering* 36(10):1401–1408

Alburquerque A., Oliveira J., Semitela S., Amaral L. (2009) Influence of bed media characteristics on ammonia and nitrate removal in shallow horizontal subsurface flow constructed wetlands. *Bioresource Technology* 100(24):6269–6277

American Society of Civil Engineers (ASCE). (1988) Manuals and reports on engineering practise No.68 Aeration. *WEF Manual of practise FD13*. USA

Andreottola G., Oliveira E., Foladori P., Peterlini R., Ziglio G. (2007) Respirometric techniques for assessment of biological kinetics in constructed wetland. *Water Science and Technology* 56(3):255-261

Austin D. (2006) Influence of cation exchange capacity (CEC) in a tidal flow, flood and drain wastewater treatment wetland. *Ecological Engineering* 28(1):35-43

Babatunde A., Zhao Y., Zhao X-H. (2010) Alum sludge-based constructed wetland system for enhanced removal of P and OM from wastewater: Concept, design and performance analysis. *Bioresource Technology* 101(16):6576-6579

Behrends L. (1999) Reciprocating sub-surface flow constructed wetlands for improving wastewater treatment. US Patent: US 5863433 A.

Behrends L., Houke L., Bailey E., Jansen P., Brown D. (2001) Reciprocating constructed wetlands for treating industrial, municipal and agricultural wastewater. *Water Science and Technology* 44(11-12):399-405

Bellucci M., Ofiteru I., Graham D., Head I., Curtis T. (2011) Low dissolved oxygen nitrifying systems exploit ammonia-oxidizing bacteria with unusually high yield. *Applied and Environmental Microbiology* 77(21):7787-7796

Bezbaruah A., Zhang T. (2005) Quantification of oxygen release by Bulrush (*Scripus validus*) roots in a constructed treatment wetland. *Biotechnology and Bioprocess Engineering* 89 (3):308-318

Brix H. (1990) Gas exchange through the soil-atmosphere interphase and through dead culms of *Phragmites australis* in a constructed reed bed receiving domestic sewage. *Water Research* 24(2):259-266

Bruch I., Fritsche J., Bänninger D., Alewell U., Sendelov M. (2011) Improving the treatment efficiency of constructed wetlands with zeolite-containing filter sands. *Bioresource Technology* 102:937-941

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

Butterworth E., Beneteau M., Dotro G., Jefferson B. (2014) Effect of media on oxygen transfer in packed beds. *Water Research* Submitted

Caselles-Osorio A., García J. (2006) Performance of experimental horizontal subsurface flow constructed wetlands fed with dissolved or particulate organic matter. *Water Research* 40(19):3603-3611

Caselles-Osorio A., García J. (2007) Effect of physico-chemical pretreatment on the removal efficiency of horizontal subsurface-flow constructed wetlands. *Environmental Pollution* 146(1):55-63

Chazarenc F., Gagnon V., Comeau Y., Brisson J. (2009) Effect of plant and artificial aeration on solids accumulation and biological activities in constructed wetlands. *Ecological Engineering* 35(6):1005-1010

Chen Y., Wen Y., Zhou Q., Vymazal J. (2014) Effects of plant biomass on nitrogen transformation in subsurface batch constructed wetlands: A stable isotope and mass balance assessment. *Water Research* 63: 158-167

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

Cooper P., Smith M., Maynard H. (1997) The design and performance of a nitrifying vertical-flow reed bed treatment system. *Water Science and Technology* 35(5):215-221

Cottingham P., Davies T., Hart B. (1999) Aeration to promote nitrification in constructed wetlands. *Environmental Technology* 20(1):69–75

Coulson J., Richardson J., Backhurst J., Harker J. (2002) *Chemical Engineering (5th edn)*. Butterworth-Heinemann: Oxford, UK

Curia A., Koppe J., Costa J., Feris L., Gerber W. (2011) Application of pilot-scale-constructed wetland as tertiary treatment system of wastewater for phosphorus and nitrogen removal. *Water, Air and Soil Pollution* 218(1-4):131-143

DEFRA. (2012) Wastewater treatment in the United Kingdom: Implementation of the European Union Urban Waste Water Treatment Directive – *91/271/EEC*. DEFRA: London, UK

Dong H., Qiang Z, Li T., Jin H., Chen W. (2012) Effect of artificial aeration on the performance of vertical-flow constructed wetland treating heavily polluted river water. *Journal of Environmental Sciences* 24(4):596-601

Fan J., Wang W., Zhang B., Guo Y., Ngo H-H., Guo W., Zhang J., Wu H. (2013) Nitrogen removal in intermittently aerated vertical flow constructed wetlands: Impact of influent COD/N ratios. *Bioresource Technology* 143:461–466

Foladori P., Ruaben J., Ortigara R. (2013) Recirculation or artificial aeration in vertical flow constructed wetlands: A comparative study for treating high load wastewater. *Bioresource Technology* 149:398-405

Fujie K., Hu H-Y., Ikeda Y., Urano K. (1992) Gas-liquid oxygen transfer characteristics in an aerobic submerged biofilter for wastewater treatment. *Chemical Engineering Science* 47(13-14):3745-3752

Garcia-Ochoa F., Gomez E. (2009) Bioreactor scale-up and oxygen transfer rate in microbial processes: An overview. *Biotechnology Advances* (27):153–176

García-Pérez A., Harrison M., Grant B. (2009) Re-circulating vertical flow constructed wetlands: green alternative to treating both human and animal sewage. *Environmental Health* 72(4):17-20

Gardner M., Jones V., Comber S., Scrimshaw M., Coelle-Garcia T., Cartmell E., Lester J., Ellor B. (2013) Performance of UK wastewater treatment works with respect to trace contaminants. *Science of the Total Environment* 456-457:359-369

Georges K., Thornton A., Sadler R. (2009) Evidence: Transforming wastewater treatment to reduce carbon emissions. *Environment Agency Report: SC070010/R2*

Green M., Friedler E., Ruskol Y., Safrai I. (1997) Investigation of alternative method for nitrification in constructed wetlands. *Water Science and Technology* 35(5):63-70

Harris S., Stephenson T., Pearce P. (1996) Aeration investigation of biological aerated filters using off-gas analysis. *Water Science and Technology* 34(3-4):307-314

Ibàñez C., Caiola N. (2012) Water, energy and limits to growth. Proceedings of the *International Conference on Water Security, Risk and Society,* April 16-18th. University of Oxford, Oxford, UK

Jia W., Zhang J., Wu J., Xie H., Zhang B. (2010) Effect of intermittent operation on contaminant removal and plant growth in vertical flow constructed wetlands: A microcosm experiment. *Desalination* 262(1-3):202–208

Johnson M, Mara D (2005) Aerated rock filters for enhanced nitrogen and faecal coliform removal from facultative waste stabilization pond effluents *Water Science & Technology* 55(11):135–142

Johnson M, Camargo Valero M and Mara D (2011) Maturation ponds, rock filters and reedbeds in the UK:statistical analysis of winter performance *Water Science & Technology* 55(11):135–142

Johansen N., Brix H. (1996) Design criteria for a two-stage constructed wetland. Proceedings of the *5th International Conference on Wetland Systems for Water Pollution Control*, 15-19th September. Vienna, Austria

Kadlec R., Wallace S. (2009) *Treatment Wetlands* (2nd edn) Taylor & Francis Group: Florida, US

Kaseva M. (2004) Performance of a sub-surface flow constructed wetland in polishing pre-treated wastewater—a tropical case study. *Water Research* 38(3):681–687

Kayser K., Kunst S. (2005) Processes in vertical-flow reed beds: nitrification, oxygen transfer and soil clogging. *Water Science and Technology* 51(9):177-84

Krasnits E., Friedler E., Sabbah I., Beliavski M., Tarre S., Green M. (2009) Spatial distribution of microbial populations in a well established constructed wetland treating municipal wastewater. *Ecological Engineering* 35(7):1085-1089

Langergraber G., Tietz A., Haber R. (2007) Comparison of measured and simulated distribution of microbial biomass in sub-surface vertical flow constructed wetlands. *Water Science and Technology* 56(3):233-240

Lee C., Fletcher T., Sun G (2009) Nitrogen removal in constructed wetlands. *Engineering in Life Sciences* 9(1): 11-22

Llorens E., Matamoros V., Domingo V., Bayona J., García J. (2009) Water quality improvement in a full-scale tertiary constructed wetland: Effects on conventional and specific organic contaminants. *Science of the Total Environment* 407(8):2517-2524

Loupasaki E., Diamadopoulos E. (2013) Attached growth systems for wastewater treatment in small and rural communities: A review. *Journal of Chemical Technology and Biotechnology* 88(2):190-204

Maltais-Landry G., Maranger R., Brisson J. (2009a) Effect of artificial aeration and macrophyte species on nitrogen cycling and gas flux in CWs. *Ecological Engineering* 35(2):221-229

Maltais-Landry G., Maranger R,. Chazarenc F. (2009b) Nitrogen transformations and retention in planted and artificially aerated constructed wetlands. *Water Research* 43(2):535-545

Maltais-Landry G., Maranger R., Brisson J., Chazarenc F. (2009c) Greenhouse gas production and efficiency of planted and artificially aerated constructed wetlands. *Environmental Pollution* 157(3):748–754

Mansi G., Butterworth E., Ranieri E., Jefferson B., Dotro G. (2013) Effect of artificial aeration on hydraulic behavior of horizontal subsurface flow wetlands. Proceedings of the *2nd Conference on low energy wastewater treatment systems,* 24th September. Solihull, UK

Matamoros V., García J., Bayona J. (2008) Organic micropollutant removal in a full-scale surface flow constructed wetland fed with secondary effluent. *Water Research* 42(3):653-660

Matthys A., Parkin G., Wallace S. (2000) A comparison of constructed wetlands used to treat domestic wastes: conventional, drawdown and aerated systems. Proceedings of the *7th International Conference on Wetland Systems for Water Pollution Control*; *International Water Association,* 11th-16th November. Colchester, UK

Medoza-Espinosa L., Stephenson T. (1999) A review of biological aerated filters (BAFs) for wastewater treatment. *Environmental Engineering Science* 16(3):201-216

Metcalf & Eddy., Tchobanoglous G., Burton F., Stensel, H. (2003) *Wastewater engineering: treatment and reuse (4th edn.)*, Metcalf Eddy Inc., McGraw-Hill: New York, US

Mitterer-Reichmann G. (2002) Data evaluation of constructed wetlands for treatment of domestic wastewater. Proceedings of the *8th international conference on wetland treatment for water pollution control,* 16th-19th September. Arusha, Tanzania

Molle P., Liénard A., Grasmick, A., Iwema, A. (2006) Effect of reeds and feeding operations on hydraulic behaviour of vertical flow constructed wetlands under hydraulic overloads. *Water Research* 40(3):606-612

Moir S. (2013) Reedbeds vs. Package Treatment Plants – how do they shape up? Proceedings of the *Constructed Wetlands Association annual conference Wetlands for Water Management* 26th–27th June. Cardiff University, Cardiff, UK

Mueller J., Boyle W., Pöpel H. (2002) *Aeration: principles and practice*. CRC Press: London

Muñoz P., Drizo A., Cully Hession W. (2006) Flow patterns of dairy wastewater constructed wetlands in a cold climate. *Water Research* 40(17):3209-3218

Nivala J., Hoos M., Cross C., Wallace C., Parkin S. (2007) Treatment of landfill leachate using an aerated, horizontal subsurface-flow constructed wetland. *Science of the Total Environment* 380(1-3):19–27

Nivala J., Wallace S., Headley Y., Kassa K., Brix H., van Afferden M., Müller R. (2012) Oxygen transfer and consumption in subsurface flow treatment wetlands. *Ecological Engineering* 61(B):544-554

Nowak O. (2000) Upgrading of wastewater treatment plants equipped with rotating biological contactors to nitrification and P removal. *Water Science and Technology* 41(1):145-153

O' Hogain S. (2003) The design, operation and performance of a municipal hybrid reed bed treatment system. *Water Science and Technology* 48(5):119-126

O'Luanaigh N., Goodhue R., Gill L. (2010) Nutrient removal from on-site domestic wastewater in horizontal subsurface flow reed beds in Ireland. *Ecological Engineering* 36(10):1266–1276

Ong S., Uchiyama K., Inadama D., Ishida Y., Yamagiwa K. (2010) Performance evaluation of laboratory scale up-flow constructed wetlands with different designs and emergent plants. *Bioresource Technology* 101(19):7239–7244

Ortigara A., Foladori P., Andreottola G. (2010) Kinetics of heterotrophic biomass and storage mechanism in wetland cores measured by respirometry. *Water Science and Technology* 64(2):409-415

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

Palmer H. (2008) *High rates of ammonia removal in constructed treatment wetland mesocosms using oxygenation*. MSc Thesis. Washington State University, US

Pearce P. (2013) Sustainable processes to meet low ammonia consents – the small works challenge. Proceedings of the *2nd Conference on low energy wastewater treatment systems*, 24th September. Solihull, UK

Sklarz M., Gross A., Yakirevich A., Soares M. (2009) A recirculating vertical flow constructed wetland for the treatment of domestic wastewater. *Desalination* 246(1-3): 617–624

Sousa W., Panitz C., Thomaz S. (2011) Performance of pilot-scale vertical flow constructed wetlands with and without the emergent macrophyte *Spartina alterniflora* treating mariculture effluent. *Brazilian Archives of Biology and Technology* 54(2):405-413

Stefanski A., Tsihrintzis V. (2009) Effect of outlet water level raising and effluent recirculation on removal efficiency of pilot-scale, horizontal sub-surface flow constructed wetlands. *Desalination* 248(1-3):961-976

Stevik T., Ausland G., Hanssen J., Jenssen P. (1999) The influence of physical and chemical factors on the transport of *E. coli* through biological filters for wastewater purification. *Water Research* 33 (18):3701–3706

Sun G., Gray K., Biddlestone A. (1999) Treatment of agricultural wastewater in a pilot-scale tidal flow reed bed system. *Environmental Technology* 20(3):233–237

Sun G., Zhao Y., Allen S. (2005) Enhanced removal of organic matter and ammonical-nitrogen in a column experiment of tidal flow constructed wetland system. *Journal of Biotechnology* 115(2):189-197

Tang X., Huang S., Scholz, M. (2008) Nutrient removal in wetlands during intermittent artificial aeration. *Environmental Engineering Science* 25(9):1279-1290

Tanner C., D'Eugenio J., McBride G., Sukias J., Thompson K. (1999) Effect of water level fluctuation on nitrogen removal from constructed wetland mesocosm. *Ecological Engineering* 12(1-2):67-92

Tanner C. (2001) Plants as ecosystem engineers in subsurface-flow treatment wetlands. *Water Science and Technology* 44(11-12):9-17

Tanner C., Kadlec R., Gibbs M., Sukias, J., Nguyen M. (2002) Nitrogen processing gradients in subsurface-flow treatment wetlands—influence of wastewater characteristics. *Ecological Engineering* 18(4):499–520

Torrens A., Molle P., Boutin C., Salgot M. (2009) Impact of design and operation variables on the performance of vertical-flow constructed wetlands and intermittent sand filters treating pond effluent. *Water Research* 43(7):1851-1858

Trang N., Konnerup D., Schierup H., Chiem N., Tuan L., Brix, H. (2010) Kinetics of pollutant removal from domestic wastewater in a tropical horizontal subsurface flow constructed wetland system: Effects of hydraulic loading rate. *Ecological Engineering* 36(4):527–535

Tyroller L., Rousseau D., Santa S., García, J. (2010) Application of the gas tracer method for measuring oxygen transfer rates in subsurface flow constructed wetlands. *Water Research* 44:4217-4225

Vörösmarty C., Green P., Salisbury J., Lammers, R. (2000) Global water resources: Vulnerability from climate change and population growth. *Science* 289(5477):284-288

Vymazal J. (2005) Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. *Ecological Engineering* 25(5):478–490

Vymazal J. (2011) Long-term performance of constructed wetlands with horizontal sub-surface flow: Ten case studies from the Czech Republic. *Ecological Engineering* 37 (1):54-63

Wallace S., Higgins J., Crolla A., Kinsley C., Bachand A., Verkuijl S. (2006) High rate ammonia removal in aerated engineered wetlands. Proceedings of the *10th International Conference on Wetland Systems for Water Pollution Control,* 23rd-29th September. Lisbon, Portugal

Weedon C. (2003) Compact vertical flow constructed wetland systems – first two years' performance. *Water Science and Technology* 48(5):15-23

Wu S., Zhang D., Austin D., Dong R., Pang C. (2011) Evaluation of a lab-scale tidal flow constructed wetland performance: Oxygen transfer capacity, organic matter and ammonium removal. *Ecological Engineering* 37(11):1789-1795

Yalcuk A., Ugurlu A. (2009) Comparison of horizontal and vertical constructed wetland systems for landfill leachate treatment. *Bioresource Technology* 100(9):2521–2526

Yousefi Z., Mohseni-Bandpei A. (2010) Nitrogen and phosphorus removal from wastewater by subsurface wetlands planted with *Iris pseudacorus*. *Ecological Engineering* 36(6):777–782

Zhang J., Huang X., Liu C., Shi H., Hu Y. (2005) Nitrogen removal enhanced by intermittent operation in a sub-surface wastewater infiltration system. *Ecological Engineering* 25(4):419-428

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

Zhao Y., Sun G., Allen S. (2004) Purification capacity of a highly loaded laboratory scale tidal flow reed bed system with effluent recirculation. *Science of the Total Environment* 330(1-3):1-8

Zhao Y., Babatunde A., Hu Y., Kumar J., Zhao X. (2011) Pilot field-scale demonstration of a novel alum sludge-based constructed wetland system for enhanced wastewater treatment. *Process Biochemistry* 46(1):278–283

Supplementary information 2.1

Table A Ammonium removal rates and site details

Reference	Area	V_e	Q	HRT	HLR	NH_4^+ loading rate	Inlet	Outlet	AR	Treatment
	m^2	m^3	m^3/d	d	$m^3/m^2/d$	$g/m^2/d$	mg/L	mg/L	$g/m^2/d$	
HSSF										
Llorens, 2009	10000	2800	100	30.0	0.01	0.33	33.0	6.8	0.24	Tertiary
Matamoros, 2008	10000	2800	100	30.0	0.01	0.38	37.8	3.0	0.32	Secondary
Matamoros, 2008	10000	2800	100	30.0	0.01	0.61	61.0	15.3	0.43	Secondary
Cottingham et al, 1999	150	36.00	14.4	3.00	0.10	5.66	59.0	50.2	0.71	Primary
Vymazal, 2011	2500	600.00	203	2.96	0.08	6.69	82.4	8.0	6.04	Secondary
Vymazal, 2011	704	168.96	80.2	2.11	0.11	8.02	70.4	6.8	7.24	Secondary
Vymazal, 2011	18.0	4.32	0.7	6.17	0.04	1.08	27.7	4.1	0.92	Secondary
Vymazal, 2011	2100	504.00	36.3	13.9	0.02	3.89	225	15.8	3.61	Secondary
Vymazal, 2011	750	180.00	22.8	7.89	0.03	0.84	27.7	4.10	0.72	Secondary
Vymazal, 2011	2160	518.40	231	2.24	0.11	3.58	33.4	4.71	3.07	Secondary
Vymazal, 2011	3040	729.60	233	3.13	0.08	5.14	67.1	4.46	4.80	Secondary
STW Site 1, (unpublished)	100	24.00	28.0	0.86	0.28	3.33	11.9	11.1	0.22	Tertiary
STW Site 2, (unpublished)	200	48.00	100	0.48	0.50	7.70	15.40	11.90	1.75	Secondary
Vymazal, 2005	18.0	4.32	0.30	14.4	0.02	1.03	62.0	51.0	0.18	Secondary
O'Luanaigh, 2010	15.1	2.09	0.33	6.40	0.02	1.62	74.9	61.0	0.30	Secondary
O'Luanaigh, 2010	4.00	0.69	0.14	5.00	0.03	0.76	22.1	20.7	0.05	Tertiary

Caselles-Osario and Garcia,2007	0.55	0.08	0.01	6.00	0.02	0.45	25.0	3.40	0.50	Secondary
Caselles-Osario and Garcia,2007	0.55	0.08	0.02	3.00	0.04	1.26	34.0	2.60	1.47	Secondary
Caselles-Osario and Garcia,2007	0.55	0.08	0.03	2.00	0.06	2.04	37.0	13.0	1.68	Secondary
Caselles-Osario and Garcia,2006	0.54	0.06	0.20	3.00	0.37	4.30	11.6	8.30	0.13	Secondary
Caselles-Osario and Garcia,2006	0.54	0.06	0.10	6.00	0.19	4.28	23.1	10.0	0.26	Secondary
Matthys et al,2000	0.07	0.01	0.00	3.50	0.03	5.44	156	126	1.04	Secondary
Agudelo, 2010	0.60	0.05	0.01	7.25	0.01	1.38	125	115	0.10	Secondary
Kaseva, 2004	5.88	1.23	0.63	1.98	0.11	2.19	20.6	15.1	0.58	Secondary
Kaseva, 2004	5.88	1.23	0.64	1.93	0.11	2.24	20.6	15.9	0.52	Secondary
Maltais – Landry et al, 2009c	1.00	0.10	0.06	1.73	0.06	0.12	2.00	0.01	0.12	Tertiary
Yalcuk, 2009	0.31	0.13	0.01	12.5	0.03	3.90	122	75.0	1.50	Secondary
Albuquerque et al, 2009	1.60	0.14	0.02	5.70	0.02	0.53	35.5	5.00	0.48	Tertiary
Albuquerque et al, 2009	1.60	0.13	0.02	5.10	0.02	0.80	53.6	31.1	0.35	Tertiary
Yousefi and Mohseni-Bandpei, 2010	0.13	0.02	0.08	4.00	0.58	14.8	25.3	14.4	0.40	Tertiary
Yousefi and Mohseni-Bandpei, 2010	0.13	0.02	0.09	4.00	0.69	17.5	25.3	19.0	0.27	Tertiary
VSSF										
Bruch et al, 2011	50.0	15.8	5.00	3.2	0.10	2.99	29.9	1.50	2.84	Secondary
Bruch et al, 2011	30.0	5.40	3.00	1.8	0.10	2.49	24.9	12.0	1.29	Secondary
Bruch et al, 2011	30.0	5.40	3.00	1.8	0.10	2.49	24.9	3.23	2.17	Secondary
Cooper et al, 1997	16.0	3.36	8.40	0.4	0.53	26.5	50.5	29.2	11.2	Tertiary
Cooper et al, 1997	16.0	3.36	8.40	0.4	0.53	15.3	29.2	14.0	7.98	Tertiary
Cooper et al, 1997	16.0	3.36	8.40	0.4	0.53	3.34	6.4	2.2	2.19	Tertiary

Cooper et al, 1997	16.0	3.36	8.40	0.4	0.53	1.16	2.2	1.4	0.43	Tertiary
Mitterer-reichmann, 2002	275.0	82.50	7.43	11	0.03	2.09	77.5	5.30	1.95	Secondary
Mitterer-reichmann, 2002	275.0	165.0	7.43	22	0.03	2.09	77.5	1.90	2.04	Secondary
Weedon, 2003	16.7	5.01	1.34	3.8	0.08	7.51	93.9	10.3	6.69	Secondary
Kayser and Kunst, 2005	2250	540.0	0.30	0.5	0.29	0.01	50.0	4.5	21.8	Secondary
Foladori et al, 2013	2.25	1.8	0.16	11.6	0.07	4.00	58.0	11.9	3.18	Secondary
Sklarz et al, 2009	0.99	0.36	0.30	1.2	0.30	11.2	37.0	3.00	10.3	Primary
Langergraber et al, 2007	20.0	26.0	4.00	1.0	0.20	13.5	67.6	6.60	79.3	Secondary
Langergraber et al, 2007	20.0	26.0	4.00	1.0	0.20	13.5	67.6	17.5	65.1	Secondary
Langergraber et al, 2007	20.0	26.0	4.00	1.0	0.20	13.5	67.6	36.9	39.9	Secondary
Yalcuk, 2009	0.50	0.11	0.01	11	0.02	2.44	122	46.0	1.52	Secondary
Yalcuk, 2009	0.50	0.08	0.01	8.0	0.03	2.44	122	62.0	1.20	Secondary
Matthys et al, 2002	0.07	0.01	0.00	3.5	0.01	5.44	156	112	1.52	Secondary
Jia et al, 2010	0.24	0.06	0.02	3.0	0.10	4.31	43.1	2.65	3.51	Secondary
Green et al, 1997	0.10	0.03	0.02	1.0	9.77	9.38	40.0	1.20	11.6	Secondary
Green et al, 1997	0.10	0.03	0.02	1.0	9.77	23.4	100	33.0	20.1	Secondary
Fan et al, 2013	0.04	0.01	0.01	3	0.21	8.4	40.1	27.0	1.1	Secondary
Fan et al, 2013	2.00	0.60	0.06	10.0	0.03	1.11	37.0	28.2	0.26	Secondary
MODIFIED/INTEGRATED										
Cooper et al, 1997	32.0	9.0	15.0	0.6	0.47	5.16	11.0	1.00	4.69	Tertiary
Johansen and Brix, 1996	527.0	147.6	20.0	7.4	0.04	1.84	48.5	6.60	1.59	Secondary
O'Hogain, 2003	184	44.0	8.0	5.5	0.03	1.96	45.0	7.0	1.65	Secondary
Zhao et al, 2011	4.40	1.50	1.28	2.0	0.29	31.0	107	31.0	12.9	Secondary
Curia et al,2011	1.00	0.35	2.00	3.0	3.20	12.8	6.40	3.00	0.40	Tertiary

Green et al,1997	0.10	0.02	0.03	0.8	0.25	9.84	40.0	2.30	9.28	Secondary
Green et al,1997	0.10	0.02	0.02	1.3	0.16	16.0	100	32.5	10.8	Secondary
Sun et al, 2005	0.03	0.008	0.01	0.2	0.42	43.7	104	76.0	39.2	Secondary
Babatunde et al, 2010	0.03	0.01	0.04	0.2	1.27	169	133	35.6	117	Secondary
Zhao et al, 2004	0.14	0.05	0.05	0.9	0.34	34.0	100	60.0	13.6	Secondary
Zhao et al, 2004	0.03	0.003	0.01	0.2	0.43	52.0	121	75.0	19.8	Secondary
Sun et al, 1999	0.03	0.008	0.01	0.4	0.42	50.8	121	63.0	40.6	Secondary
Ong et al, 2010	0.03	0.01	0.00	3.0	16.5	1.95	32.3	4.96	1.88	Secondary
Ong et al, 2010	0.03	0.01	0.00	3.0	34.0	0.95	32.3	10.2	1.52	Secondary
Ong et al, 2010	0.03	0.01	0.00	3.0	34.0	0.95	32.3	13.4	1.30	Secondary
AERATED										
Cottingham et al,1999	150	36.00	14.4	3.00	0.10	5.37	55.9	17.9	3.04	Primary
STW Site 1, (unpublished)	390	93.60	144	0.65	0.37	2.03	5.50	0.50	1.85	Tertiary
STW Site 2, (unpublished)	100	24.00	28.0	0.86	0.28	3.33	11.9	0.40	3.22	Tertiary
STW Site 3, (unpublished)	520	124.8	179	0.70	0.34	0.63	1.84	0.14	0.59	Tertiary
STW Site 4, (unpublished)	200	48.00	50	0.96	0.25	7.25	29.00	1.00	7.00	Secondary
STW Site 5, (unpublished)	600	144.0	155	0.93	0.26	2.56	9.90	1.00	2.30	Tertiary
Nivala et al, 2007	93.0	22.32	0.40	55.8	0.004	0.87	203	9.53	0.83	Secondary
Ouellet-Plamondon et al,2006	0.96	0.12	0.03	5.00	0.03	0.07	2.33	0.03	0.06	Secondary
Zhang et al, 2010	2.10	0.84	0.13	6.46	0.06	2.21	35.7	4.00	1.96	Secondary
Matthys et al, 2001	0.07	0.01	0.003	3.50	0.03	5.44	156	27.3	4.49	Secondary
Ong et al, 2010	0.03	0.00	0.002	3.00	0.06	1.90	32.3	0.69	1.86	Secondary
Ong et al, 2010	0.03	0.01	0.003	0.00	0.00	0.95	32.3	0.75	0.95	Secondary
Palmer, 2008	0.13	0.05	0.006	10.0	0.02	0.41	9.50	0.57	0.33	Tertiary

Palmer, 2008	0.13	0.02	0.011	5.00	0.04	0.82	9.50	1.62	0.29	Tertiary
Fan et al, 2013	0.04	0.01	0.01	3.0	0.21	8.42	40.1	0.3	3.45	Secondary
Fan et al, 2013	2.00	0.60	0.06	10.0	0.03	1.11	37.0	1.7	1.06	Secondary
Foladori et al, 2013	2.25	1.35	0.36	3.8	0.16	9.16	58.0	19.5	6.08	Secondary
Dong et al, 2012	0.02	0.01	0.00	4.2	0.19	1.34	7.05	1.8	1.00	Secondary
Dong et al, 2012	0.02	0.01	0.01	2.1	0.38	2.68	7.05	1.7	2.03	Secondary
Dong et al, 2012	0.02	0.01	0.01	1.1	0.76	5.36	7.05	3	3.08	Secondary
Dong et al, 2012	0.02	0.01	0.00	4.2	0.19	1.34	7.05	1	1.15	Secondary
Dong et al, 2012	0.02	0.01	0.01	2.1	0.38	2.68	7.05	0.9	2.34	Secondary
Dong et al, 2012	0.02	0.01	0.01	1.1	0.76	5.36	7.05	2.2	3.69	Secondary

NB: Percentage removal is commonly quoted to describe ammonia removal efficiencies in specific CWs but is inadequate for comparing the performance of different systems as it takes no account of the characteristics of the bed; ammonium removal rate (AR) is considered a better performance parameter as the area, concentration, and hydraulic retention time are acknowledged, allowing different systems to be comparable. Equation (1) was used to calculate nitrification rates as it enables the use of hydraulic residence times (HRT) obtained from tracer studies. Associated assumptions for the calculated rates were: a) where the porosity of the bed was not stated a value of 0.4 has been used (Kadlec and Wallace, 2009), and b) where HRT values have been measured as part of the study these have been used in preference to theoretical values

$$AR = Ve * (Cin - Cout) * 1/A * 1/HRT$$

Where: AR =ammonium removal rate, V_e = Effective volume (m^3) = (area of bed * depth of water *porosity of bed media), C_{in} = NH_4^+-N concentration at inlet (mg/L), C_{out} = NH_4^+-N concentration at outlet (mg/L), HRT = Hydraulic retention time (d) = V_e/Q

Chapter 3

Effect of artificial aeration on tertiary nitrification in a full-scale sub-surface horizontal flow constructed wetland

Effect of artificial aeration on tertiary nitrification in a full-scale sub-surface horizontal flow constructed wetland

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3.1 Abstract

A full scale comparison of a newly commissioned artificially aerated horizontal sub-surface flow constructed wetland and a non-aerated bed of identical design was conducted to determine the efficacy of artificial aeration to upgrade treatment performance for ammonium removal. The works serves a population equivalent of 400; each bed has a mean hydraulic loading rate of 0.27 $m^3/m^2/d$ and during the first 9 months of operation received inlet loadings of: ammonium (NH_4^+-N): 3.1±2.4 gNH_4^+-N/m^2/d; carbonaceous biochemical oxygen demand: 2.8±2.0 $gO_2/m^2/d$; chemical oxygen demand: 19.4±11.2 $g/m^2/d$ and total suspended solids: 6.6±5.0 $g/m^2/d$ (mean ± standard deviation, n=17). Results demonstrated enhanced nitrification in the aerated bed with 99 % mass ammonium removal up to the maximum tested loading rate of 10.1 gNH_4^+-N/m^2/d. In comparison, an ammonia removal of 59 % was observed in the non-aerated bed up to a loading rate of 1.6 gNH_4^+-N/m^2/d beyond which performance deteriorated. Carbonaceous

biochemical oxygen demand and suspended solids removal were seen to be statistically similar between the beds while a significant difference was observed in terms of mixing pattern, quantity and characteristics of the accumulated solids and hydraulic conductivity. The suitability of the technology was also assessed through comparison of cost, carbon footprint and land area relative to alternative upgrading options. Retrofitting existing horizontal sub-surface flow wetlands was shown to be the most cost effective solution delivering the required treatment at 38 % of the cost of the least expensive alternatives.

Keywords: Aeration; constructed wetlands; carbon; cost; dissolved oxygen; nitrification

3.2 Introduction

Horizontal sub-surface flow constructed wetlands (HSSF CWs) are passive wastewater treatment systems commonly used for tertiary suspended solids, organic matter, and nitrate removal. In addition, the systems are increasingly required to remove ammonical nitrogen (NH_4^+-N) which principally occurs through the aerobic biological process of nitrification. Oxygen transfer in conventional HSSF CWs is poor and variable, occurring primarily via convection and diffusion from the air to the surface water with estimated oxygen transfer rates of 0.3-3.2 $gO_2/m^2/d$ (Kadlec and Wallace, 2009; Tyroller et al, 2011). Consequently, dissolved oxygen (DO) within the bed is generally below 0.5 mg/L such that the low oxygen availability in HSSF CWs is cited as the primary cause of their poor nitrification capabilities (Cottingham et al, 1999, Vymazal and

Kröpfelová, 2008). Vertical flow systems are one alternative that allows oxygen transfer via intermittent loading achieving oxygen transfer rates of 28-100 $gO_2/m^2/d$, thus enabling consistent ammonia removal (Cooper, 1999, Brix et al, 2002, Weedon, 2003). Indeed, vertical flow systems are commonly used for secondary treatment with subsequent polishing for solids and denitrification in a HSSF CW (Kadlec and Wallace, 2009). Another option is introducing artificial aeration of HSSF wetlands via a blower and coarse bubble air diffusers (Davies and Hart, 1990; Cottingham et al, 1999; Ouellet-Plamondon et al, 2006; Wallace, 2001).

Artificial aeration (AA) has been proposed as a solution to create an aerobic environment conducive of nitrification with the additional benefit of the ability to be retrofitted into existing HSSF CW systems. The efficacy of the approach has been shown at laboratory and pilot scale wetlands (Ouellet-Plamondon et al, 2006; Zhang et al, 2010), where an increase in ammonia removal of between 21 % and 63 % was observed compared to a non-aerated bed. Illustrations of the use of AA have also been presented at full-scale in terms of tertiary treatment of landfill leachate (Nivala et al, 2007), industrial wastewater (Wallace and Kadlec, 2005), and secondary treatment of domestic wastewater. For the latter, a study of 17 artificially aerated HSSF showed loading rates of 0.001 to 0.049 m/d and 0.2 – 6 $gTKN/m^2/d$ for hydraulic and total Kjeldhal nitrogen (TKN), respectively (Wallace et al, 2010). Based on the statistical study of these wetlands in the US, Wallace et al, (2010) found that above TKN loading rates of 1 $gTKN/m^2/d$ little reduction of nitrogen was achieved. However, for tertiary applications, hydraulic loading rates are considerably higher with, for instance, the UK range being 0.2 to 0.9 $m^3/m^2/d$

(Knowles et al, 2011). In small works not originally designed to nitrify, influent ammonia concentrations to the tertiary wetlands can be similar to secondary wetland influents (i.e. up to 40 mgNH_4-N/L) which under these higher hydraulic loading rates would result in a much greater ammonia (or TKN) loading rate than has been tested to date at full scale. Thus, the application of aerated wetlands for tertiary nitrification of domestic wastewaters, whilst attractive in principle for upgrading small works, has yet to be proved.

Additional benefits of artificial aeration have been postulated in terms of increased solids degradation rates and its associated impact on the lifetime of the bed (Chazarenc et al, 2009; Nivala et al, 2012). However, the only evidence to date relates to a change in accumulated total suspended solids a laboratory-scale study (Chazarenc et al, 2009). Further research is required to characterise the systems hydraulically and quantify the benefits from a clogging minimisation perspective.

This study aims to determine the performance of a full-scale aerated HSSF CW system for tertiary nitrification of domestic wastewater against a conventional HSSF control. To the authors' knowledge, this paper is the first to describe the efficacy of artificial aeration under low organic matter, high ammonia and hydraulic loading rates typically found in tertiary systems and includes an assessment of the impact of aeration on ammonia removal, solids accumulation and characteristics, hydraulic conductivity, and a cost comparison against common alternatives employed for nitrification on small works.

3.3 Materials and methods

3.3.1 Site description

The trial site (Figure 3.1) serves a population equivalent of approximately 400, designed to treat a dry weather and maximum flow of 61 and 363 m^3/d, respectively. The works is gravity fed and comprises of an initial primary settlement tank and an integral rotating biological contactor (RBC) unit for secondary treatment. The flow from the RBC is split between two parallel tertiary HSSF wetlands, 100 m^2 each (approximately 0.5 m^2/pe). In the event of the works receiving six times the dry weather flow, an overflow operates and diverts excess untreated sewage to a combined sewer overflow (CSO) HSSF wetland (180m^2). Both tertiary beds were refurbished due to clogging in October 2010. This involved the excavation and washing of the existing gravel, which was later returned to the bed. During the refurbishment, air lines (12 mm diameter perforated piping; perforations 2 mm diameter at 300 mm spacings) were placed on the floor of the beds, as per a USA- patented aeration system (Wallace, 2001). Both beds were un-aerated until March 2011. On the 3rd of March, aeration was switched on in the test bed (i.e. aerated) and left off in the control bed. Aeration in the test bed received continuous aeration from a 1.6 kW blower delivering approximately 150 m^3/h.

The beds contain 0.6 m of 6-12 mm gravel media giving a measured porosity of 0.4, and are planted with *Phragmites australis* (P. *australis*) at 4 seedlings/m^2. The mean flow to each bed is 27.2 m^3/d, resulting in mean hydraulic loadings of 0.27 $m^3/m^2/d$. Inlet loadings recorded since the onset of aeration are NH_4^+-N: 3.1±2.4 gNH_4^+-N/m^2/d, 5 day carbonaceous biochemical oxygen demand ($CBOD_5$) 2.8±2.0

$gO_2/m^2/d$, chemical oxygen demand (COD): 19.4±11.2 $gO_2/m^2/d$ and total suspended solids (TSS): 6.6±5.0 $gTSS/m^2/d$ (mean ± standard deviation, n=17).

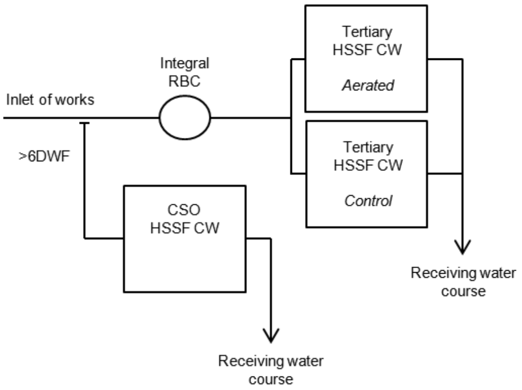


Figure 3.1 Process flow sheet of the trial site (left) and image of the trial site June 2011, showing the aerated bed left and control bed right (right)

3.3.2 Standard wastewater parameters

Composite samples were taken once every 2 weeks and consisted of an amalgamation of samples taken every 15 minutes over 24 hours using auto-samplers (ISCO 3700, Teledyno Isco Inc, USA) placed at the inlet and outlet of the individual beds over 9 months (March-November 2011; n=17). Samples were collected in 1L plastic sampling bottles and stored in a cool box with ice blocks during transition to the laboratory for same day testing. Where same day testing was not feasible samples were stored at 4 °C and brought to room temperature prior to analysis.

Hach-Lange test kits were employed for determining NH_4^+-N (detection limits – low range, medium range and high range test kits dependent on site conditions: 0.015-2; 1-12; 2-47 mg/L NH_4^+-N), NO_3-N (0.23-13.5, 5-35 mg/L NO_3-N) and TN

(1-16; 5-40, 20-100 mg/L TN), according to Hach-Lange procedures and read via a Hach-Lange DR2800 spectrophotometer. Standard methods (APHA, 2005) were followed to determine the $CBOD_5$ of the solutions, using Hach-Lange nutrient buffer pillows for preparation of the dilution water and Hach-Lange nitrification inhibitor. Dissolved oxygen concentration was measured using an LBOD with auto-stirrer (Hach-Lange). Total suspended solids were quantified following standard procedures using a three-piece filter funnel with 70 mm filter diameter and 1.2 μm pore size (APHA, 2005). Portable meters and probes were used to determine pH (Fisher Scientific), redox potential, dissolved oxygen and temperature (Hach-Lange HQ40D). Probes were checked weekly with standard pH solutions of 4.01 and 7.0; Zobell's solution and hydrogen sulphite solution for pH, redox potential and DO, respectively. Hach standards were used periodically to ensure accuracy of test kits and triplicate tests of all standard wastewater parameters indicated results precise to ±0.8 mg/L.

Statistical analyses were carried out using Graphpad Prism v.5. Data was tested for normality by calculating the t-statistic that describes the number of sample standard deviations that the sample was above or below the sample mean. In a normal distribution 68 % of values will lie within one standard deviation, 95 % within two deviations and 99 % within three. The data did not follow a normal distribution and therefore statistical significance between contaminant removal in the aerated and control beds was tested using the non-parametric Wilcoxon matched-pairs signed rank two-tailed test at ($P<0.05$) as an alternative to the paired students t-test.

3.3.3 Tracer Tests

Triplicate tracer tests were conducted to establish transport and mixing behaviour during July and August 2011. The tests involved the addition of a 0.135 g impulse of 20 % Rhodamine Water Tracer Liquid (Keystone Europe Ltd.) at the inlet points of both wetlands (De Novio, 2004). The rhodamine outlet concentration was measured using a fluorescence spectrophotomer (YSI sondes 6 series, fitted with Rhodamine sensor) with a sensitivity of up to 0.001 µg detection. Calibrations were first conducted with deionised water and on site before each test with wetland samples for background readings. Tracer recovery percentages were calculated by carrying out mass balance analyses on effluent data. Retention times, index modal retention time, Morrill dispersion index (MDI), and dispersion were determined in accordance with standard methods (Metcalf and Eddy, 2003).

3.3.4 Solids quantification and characterisation

Triplicate grab samples were collected using a metal sample can (0.1 m diameter, 0.2 m length) per each of the 9 test locations (Figure 3.2) 11 months after refurbishment (i.e., five months after aeration was turned on) for the quantification of accumulated solids. The collected solids were separated from the gravel by shaking vigorously with 250 mL effluent wastewater and passed through a 5 mm sieve. The solids solution was analysed for suspended and volatile solids to determine differences in the amount and type of within-bed solids present in each bed and characterised in terms of sludge volume index (SVI), specific resistance to filtration (SRF) and capillary suction testing (CST). The sludge volume index (SVI) is the volume in millilitres occupied by 1 g of a suspension after 30 min settling and distinguishes between settling characteristics of the solids in each of the beds (APHA, 2005). Cake filtration consisted of passing 50 mL of the sample through a 0.45 μm filter membrane and analysis to determine the SRF and the compressibility of the cakes according to standard procedures (Tiller and Lloyd, 1972).

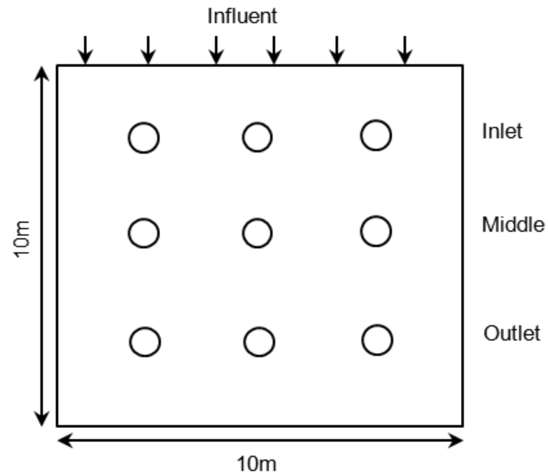


Figure 3.2 Hydraulic conductivity test / solids sampling locations

3.3.5 Hydraulic conductivity

In-situ saturated hydraulic conductivity measurements were taken using a steel pipe perforated at the base and a model 3001 Solinst levelogger following the falling head methodology described by Pedescoll et al, (2009). Measurements were taken in triplicate along three transect points spanning the length of each of the wetlands (Figure 3.2). This measurement gives approximate saturated hydraulic conductivity values, as vertical conductivity is measured, and does not take into account the horizontal flow. In addition, a certain degree of compaction occurs when inserting the pipe into the bed and thus presents a source of error but these errors have been evaluated and are considered acceptable (Pedescoll et al, 2009).

3.3.6 Feasibility

To assess the feasibility of employing artificial aeration, a cost and footprint comparison was performed against conventional treatment options capable of achieving effluent concentrations of 10/15/3 ($CBOD_5$/TSS/NH_4^+-N). Options to upgrade the works included upgrading the secondary treatment with the addition of a rotating biological contactor (RBC), mineral media trickling filter (TF), submerged aerated filter (SAF) in parallel to the existing RBC; the addition of a tertiary TF or a nitrifying SAF in series, or retrofitting the HSSF CW with artificial aeration during standard refurbishment (Figure 3.3).

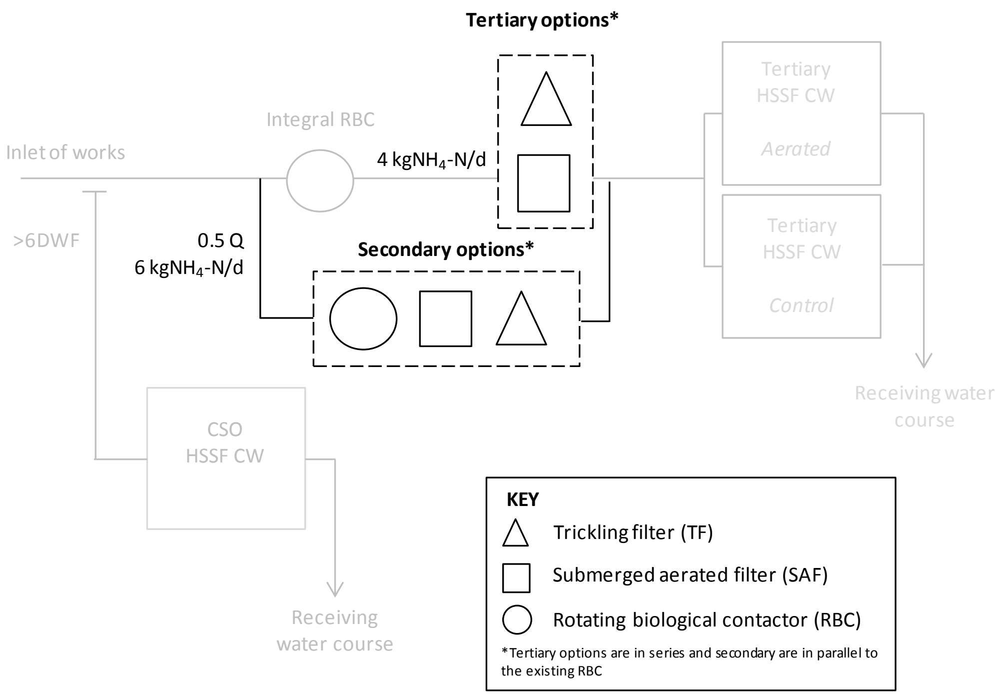


Figure 3.3 Flowsheets tested in the feasibility study

Sizing and costs of the different options were calculated according to standard literature, manufacturer's specifications and internal Severn Trent design standards as appropriate (Table 1). Capital costs were based on a Severn Trent database of past projects and inflated to current worth. As no historic costs existed for AHFCWs, costing was based on quotes for the test site. All data was normalised to this technology to provide direct comparison and translation to other geographical regions. Conventional options assume a 25 year design life. Traditional CWs for tertiary application are expected to require refurbishment due to clogging of the media approximately every 8-15 years (Griffin et al, 2008; Cooper et al, 1996). Thus, the retrofit option included the cost of one additional refurbishment. To simplify the costing, common items to all options were excluded such as primary settling and flow metering.

In addition to initial construction costs (based on cost curves), the cost of operation was calculated assuming electrical equipment (Table 3.1) to run 24 h/d, with additional energy used for de-sludging of the RBC, TFs and SAFs averaged at running twice a week for two hours (£0.09/kWh). Maintenance was estimated at 1 h/month for the CWs and 2.5 h/month for all other options, based on Severn Trent experience. The land footprint includes the area of the process and associated humus tanks, balancing tanks, pumps, blower units, and control panels. Carbon footprint was taken to consist of operational carbon and excludes process emissions, with a conversion from grid electricity to carbon dioxide (CO_2) equivalents calculated as 0.544 $kgCO_2e/kWh$ (UKWIR, 2008).

Table 3.1 Design parameters used for sizing comparative technologies

Process	Design basis	Design depth (m)	Power requirements (kW)
RBC	20 m^2/p.e.	0.0	1.1 (Drive unit)[a]
TF (2^{nd})	0.04 $kgCBOD_5/m^3/d$	2.0	1.7 (Pump)[b]
TF (3^{ry})	0.06 $kgNH_3/m^3/d$	2.5	1.0 (Pump)[b]
SAF (2^{nd})	0.6 $kg/m^3/d$	3.5	4.0 (Blower)[c]
SAF (3^{ry})	0.4 $kgNH_3/m^3/d$	3.0	4.0 (Blower)[c]
AHFCW	0.7 m^2 / p.e.	0.6	1.6 (Blower)[d]
Retrofit AHFCW	N/A	0.0	1.6 (Blower)[d]

[a]Size of drive unit based on manufacturers details (Tuke and Bell Ltd.)
[b]Pumping requirements based on a wetting rate of $1.4m^3/m^2/h$ (Severn Trent)
[c]Blower sizing based on manufacturers details (WPL)
[d]Blower sizing based on installed unit on site (NB: This is considered oversized)

3.4 Results and analyses

3.4.1 Contaminant removal

Ammonium concentrations (quoted as the mean ± standard deviation, n=17) at the outlet of the aerated bed were significantly lower than the control demonstrating the superior NH_4^+-N removal occurring in the aerated bed across the full range of loading rates experienced (Figure 3.4 left and right). In contrast, effluent ammonia in the non-aerated bed increased in proportion to the loading rate (Figure 3.4, right).

Mean effluent ammonium concentrations were 0.1±0.05 mg NH_4^+-N /L for the aerated and 8.6±6.4 mg/L for the control, corresponding to mass removals of 99 % and 13 % respectively. While both systems removed ammonia, a significant difference (W=118) observed between the aerated and control system confirms the effectiveness of the artificial aeration for use in highly loaded CWs. Notably, ammonia removal in the control bed was substantial, at 59 %, up to a loading rate of 1.6 gNH_4^+-N/m^2/d, above which efficiencies deteriorated (Figure 3.4, right). In agreement with other HSSF wetlands (Kadlec and Wallace, 2009), higher NH_4^+-N loading rates in the control bed correspond to higher NH_4^+-N outlet concentrations, exhibiting a linear relationship (R^2=0.74). Nitrification rates (NR) in the aerated bed increased linearly from 1.6 to 6.4 gNH_4^+-N/m^2/d with increasing NH_4^+-N loading rates, demonstrating a strong causal relationship.

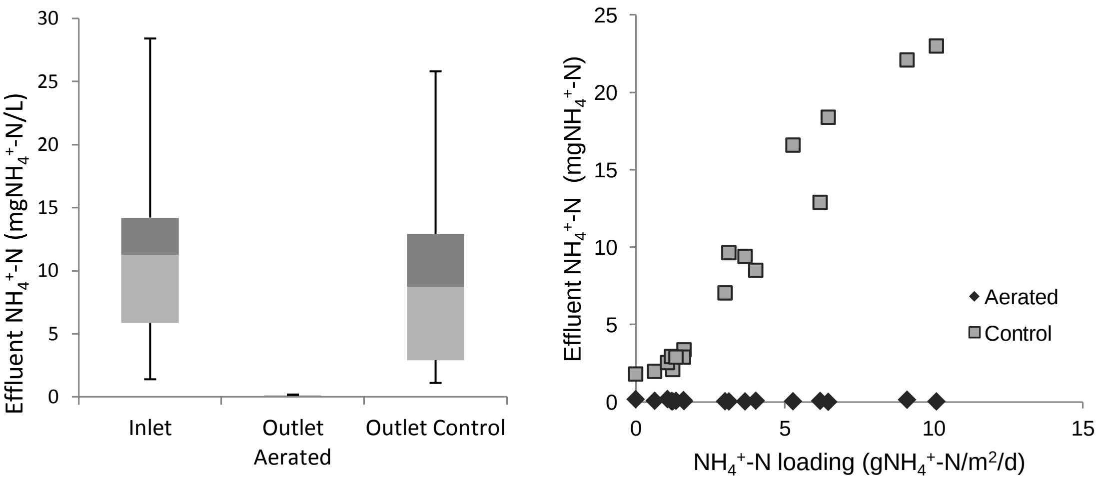


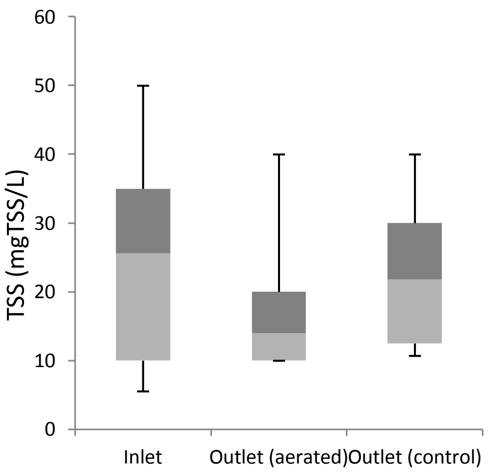
Figure 3.4 Box and whisker plot of NH_4-N, concentrations (n=17) at the inlet and the outlet of the control and aerated beds. The box represents the interquartile range; the line indicates the mean and the whiskers the 25th and 75th percentiles

(left). NH_4^+-N loading rate and associated effluent concentrations for the aerated and control beds (n=17) (right)

Effluent nitrate concentrations of 1.2-16.9 mgNO_3-N/L, with a mean value of 5.2 mgNO_3-N/L, were recorded for the aerated bed whilst the control bed recorded a lower mean value of 2.0 mgNO_3-N/L and range of 0.1-8.6 mgNO_3-N/L. Total nitrogen was not routinely measured (n=3) as the investigation was primarily focussed on ammonia removal, but it is interesting to note that percent reductions of 77 % and 55 % (corresponding to mean outlet concentrations of 25.3 mgTN/L and 18.1 mgTN/L) were observed for the aerated and control bed respectively.

Effluent TSS and $CBOD_5$ samples produced comparable measurements in the aerated and control beds (Figure 3.5 left and right). Effluent TSS concentrations were 14.0 ± 9.4 mgTSS/L for the aerated and 21.7 ± 11.0 mgTSS/L for the control, and populations did not differ statistically (W=77). The effluent $CBOD_5$ was 4.3±3.1 mgO_2/L for the aerated and 7.0±5.6 mgO_2/L for the control. Mean $CBOD_5$ concentrations (Figure 3.5 right) in the effluent of both beds were significantly lower than the inlet, highlighting successful removal, but did not differ significantly from each other (W=77) highlighting treatment capacity with regard to $CBOD_5$ was not improved by artificial aeration. This result could be expected due to the low $CBOD_5$ loading rates and expected background concentrations for HSSF constructed wetlands (Kadlec and Wallace, 2009).

Within-bed dissolved oxygen concentrations were less than 1.0 mgO_2/L across the control bed, with 80 % below 0.5 mgO_2/L. These low DO levels have been associated with low nitrification rates in both suspended growth and biofilm reactors (Dotro et al, 2011). In contrast, the DO in the equivalent aerated bed was 8 to 11 mgO_2/L, equating to saturated DO levels. However, it is well documented that nitrification is achievable in other biofilm systems, such as rotating biological contactors (RBCs), at a DO of 2 mgO_2/L or above (Nowak, 2000), indicating the bed is potentially being over-aerated.



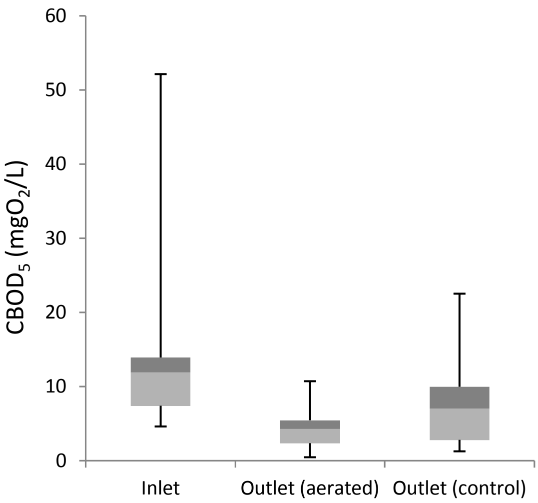


Figure 3.5 Box and whisker plots of TSS (left) and $CBOD_5$ (right), concentrations (n=17) at the inlet and the outlets of the control and aerated beds

3.4.2 Hydraulic characterisation

The aerated beds had similar residence time distributions (RTD) at all flow rates, with a sharp rise to peak followed by an exponential decay indicating a trend towards an ideal continuous stirred-tank reactor (CSTR) system. In contrast, the control displayed a double peaked curve, characteristic of channelling conditions and a classical representation of non-ideal CSTR system (Figure 3.6).

Detention times obtained from the aerated bed varied greatly between tests and correlated with the prevailing flow rate during each trial. Values of 22, 6 and 57 hours corresponded with average flow rates of 1.9, 3.5 and 0.9 m^3/h respectively. In contrast, the mean detention times in the control were more consistent at 31, 31 and 21 hours irrespective of flow rate. Associated nominal residence times (τ_n) were 12.6, 6.9 and 26.7 hours for flow rates 1.9, 3.5 and 0.9 m^3/h respectively such that the measured detention times were larger than expected. Analysis of the data and the resultant diagnostic of the hydraulic characteristics of the beds needs to reflect respective tracer recoveries of 77 % in the aerated and 35 % in the control congruent with of 77 % in the aerated and 35 % in the control may in part be due to adsorption of the rhodamine to sediment within the beds (Lin et al, 2003). Further analysis of the tracer detention time (τ) in comparison with the associated nominal residence time (τ_n) indicated stagnation / delay due to higher air:water ratio associated during low flow conditions.

More detailed analysis of the RTDs in terms of the index of modal retention time ($\tau p/\tau_n$) indicated a non-uniform flow distribution in both beds as expected in packed bed mixing (Metcalf and Eddy, 2003). Under low flow conditions, the aerated bed switched from more CSTR towards a plug flow (PF) mixing pattern as evidenced by an increase in $\tau p/\tau n$ from 0.12-0.14 to 0.82 which was similar to those obtained for the non-aerated beds which tended towards a PF reactor system (Figure 3.6). Similar conclusions can be drawn from comparison of the Morrill dispersion indexes (MDI) for normal flow conditions where the MDI of the aerated bed is higher and more consistent with CSTR mixing patterns (12.7 and 14.2) than those for the control system (7.3, 5.4 and 6.8). In the event of low flow, the MDI value of the aerated bed decreased to 4.3 suggesting considerable plug flow mixing behaviour.

As mixing was presumed to follow a convective dispersion mechanism, the dispersion coefficient (D) represented the spreading process within the beds. Analysis of the data indicated that under high flow rates the aerated bed presented significantly higher D values than during the other trials suggesting an increased risk of advective spreading during aeration which is not observed in the control. In contrast, during low flow when the water is below the media line, high dispersive/turbulent mixing is more apparent in the aerated bed compared to the control as signified by the higher dimensionless dispersion number of 0.63 compared to 0.23 in the case of the control bed.

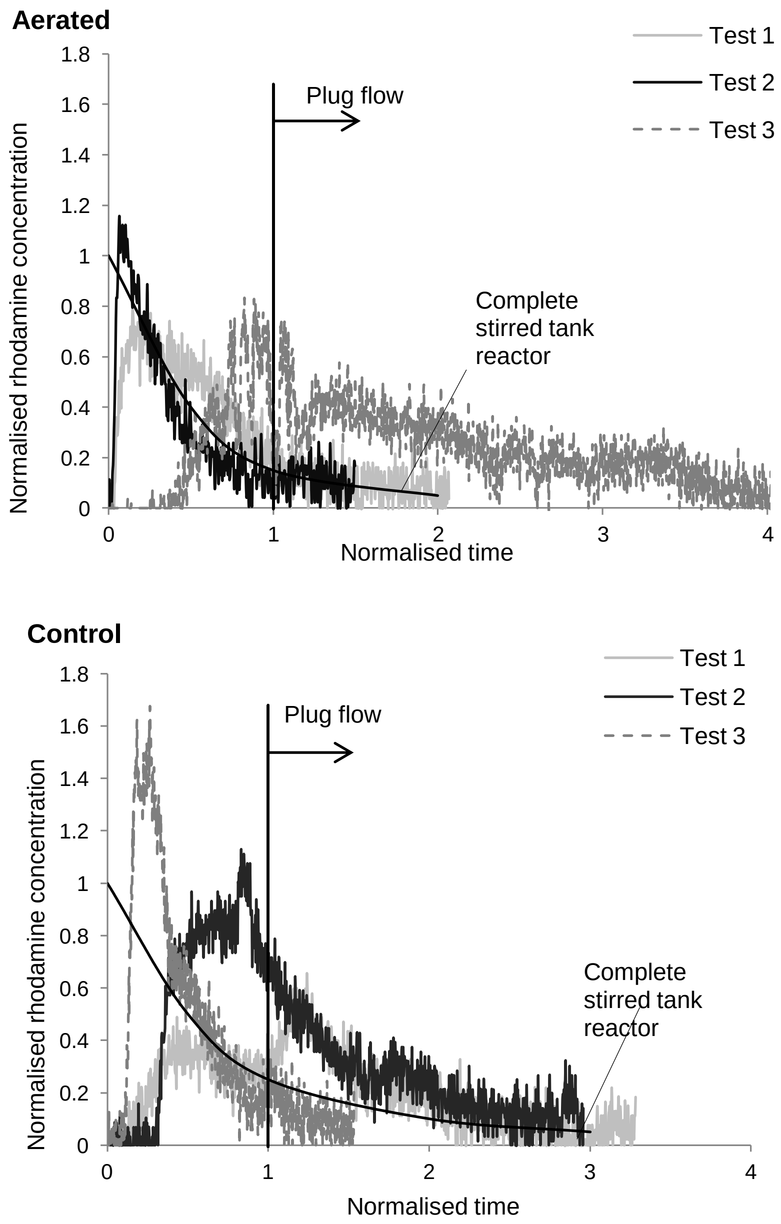


Figure 3.6 Residence time distributions for the aerated (top) and control (bottom) beds

3.4.3 Solids quantification, characterisation and hydraulic conductivity

Analysis of the solids across the surface of both beds indicated aeration generated significant differences in both the quantity and character of the solids compared to the non-aerated bed (Table 3.2). To illustrate, within-bed solids were 12.9; 6.2 and

5.4 kgTSS/m^2 for the control and 2.5, 0.5 and 0.7 kgTSS/m^2 for the aerated bed at the inlet, middle and outlet respectively. These translate to an average accumulation rate of 8.8 and 1.3 kgTSS/m^2/year for the control and aerated bed, respectively.

The nature of the solids was also observed to be different between beds (Table 3.2). In the case of the control bed, the characteristics of the solids are consistent throughout the bed with minimal variation in SVI, SRF or compressibility. The low SVI (28-39 mL/g), relatively high specific resistance to filtration (4-5x10^{13} mg/kg) and moderate compressibility (0.4-0.5) suggest a compact solids mass through which water transport is heavily restricted. For instance, the SRF values are representative of sludge material that is very difficult to dewater and is typical of raw sludge (Wakeman and Tarleton, 2005). In comparison, in the case of the aerated bed the SVI and SRF change through the bed suggesting the solids are being transformed in character. The reduction in SVI from 78 mL/g to 14 mL/g and the respective change in SRF from 5x10^{13} to 6x10^{11} mg/kg, indicate that the solids are becoming more compact as they travel through the bed and that water can transport more easily through the solid cake material. The SRF of the material at the end of the aerated bed is indicative of a well-conditioned sludge that is moderately easy to dewater (Wakeman and Tarleton, 2005). To illustrate the impact of these changes, the SRF experiments took around 10 hours to complete with the samples taken from the control bed compared to minutes for the samples taken from the end of the aerated bed.

Table 3.2 Solids characteristics in the inlet, middle and outlet of the aerated and non-aerated control

		Inlet	Middle	Outlet
TSS (mg/L)	*Aerated*	4183	780	1220
	Control	21560	10320	8920
VSS (mg/L)	*Aerated*	1543	250	315
	Control	14271	5328	4910
SVI (mL/g)	*Aerated*	77.7	21.8	13.9
	Control	38.9	36.3	28.0
CST (s/g)	*Aerated*	9.7	11.2	6.9
	Control	6.3	3.5	4.3
SRF (0.6 bar) mg/kg	*Aerated*	6×10^{13}	2×10^{12}	6×10^{11}
	Control	5×10^{13}	4×10^{13}	5×10^{13}
Compressibility	*Aerated*	0.5	0.8	<0.1
factor	*Control*	0.5	0.4	0.5
DO (mgO_2/L)	*Aerated*	9.4	10.5	9.8
	Control	0.3	0.5	0.7

TSS = total suspended solids; VSS = volatile suspended solids; SVI = sludge volume index; CST = capillary suction tests; SRF = specific resistance to filtration

Higher hydraulic conductivities (k) were observed in the aerated compared to the control bed at all measurement locations, with values of 20 to 41 m/d and 2 to 28 m/d in the aerated and control, respectively (Figure 3.7). Similar k values in the aerated bed were observed at all measurement locations. In the control bed, k values were lowest at the inlet and highest at the outlet suggesting improved hydraulic conductivity with distance from the inlet.

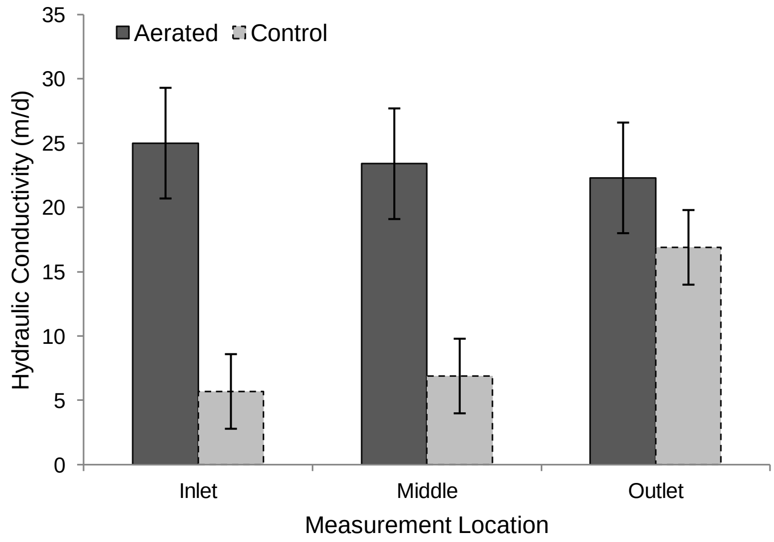


Figure 3.7 Comparison of hydraulic conductivity at the inlet, middle and outlet of the aerated and control beds. Each location is the mean of the lateral three readings. Error bars represent average standard deviation

3.4.4 Feasibility

Comparison of the treatment technologies reveals that the artificial aeration option delivers a lower total monetary cost than the alternatives (Figure 3.8). The total cost ratio ranged from 7.8 times for the RBC to 2.8 for the tertiary trickling filter (TF 3ry). The retrofitted artificial aeration system (AHFCW) was significantly lower in total cost than conventional options and required no additional area aside from housing of the blower unit, making its impact minimal and demonstrating the overall attractiveness of the use of this system to upgrade existing HSSF CWs.

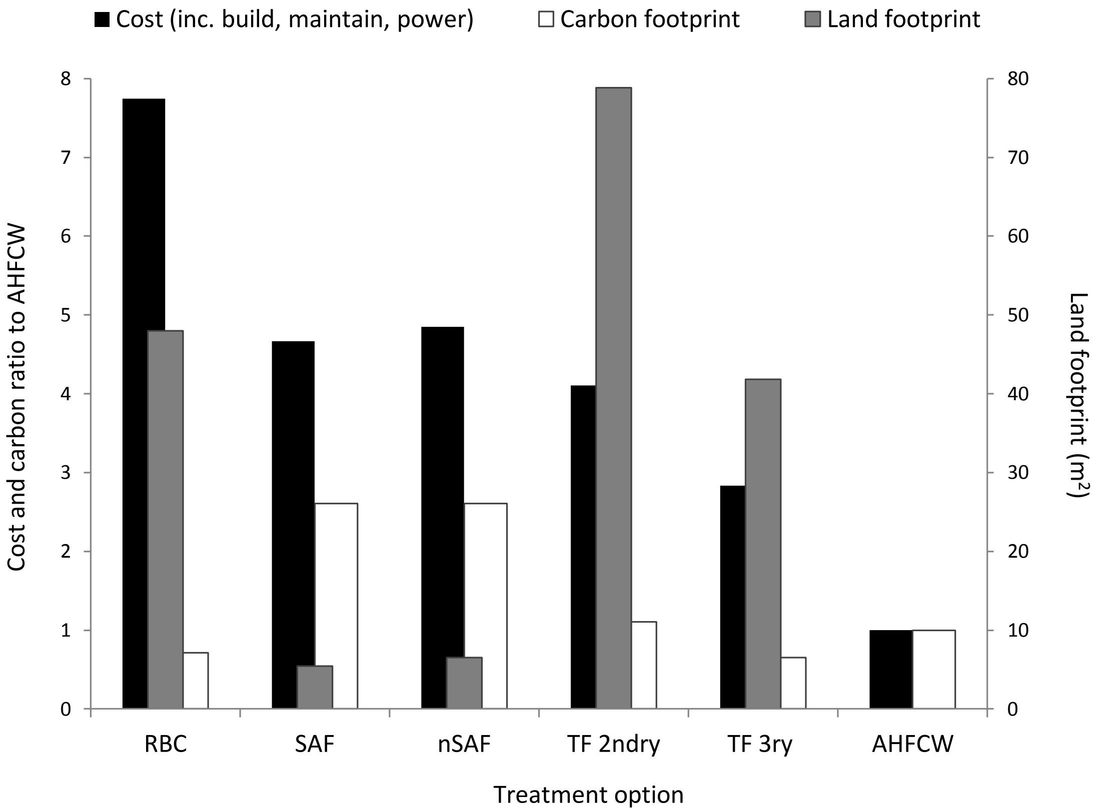


Figure 3.8 Comparative ratio of upgrading options to meet tightening ammonium consents on a small wastewater treatment plant

In relation to power consumption and the associated carbon footprint, both the RBC and TF 3ry provided lower solutions at 28 % and 35 % lower than the AHFCW option due to the efficiency of aeration. However, given that the DO in the beds was saturated, a reduction in power usage would be possible whilst still providing sufficient treatment therefore reducing this difference.

3.5 Discussion

Key observations from initial results of this full scale comparative study have demonstrated that artificially aerating HSSF CWs enables complete ammonia removal without deterioration in solids and $CBOD_5$ removal capabilities of un-aerated HSSF CWs.

Effluent $CBOD_5$ concentrations were not affected by organics loading in either the aerated or control bed, highlighting adequate oxygen is present for removal without the addition of AA. Ammonia removal in the traditional non-aerated bed was seen to correlate to loading rates in terms of both ammonia (Figure 3.4 right) and organics with a limiting loading rate of up to 1.6 $gCBOD_5/m^2/d$ (Figure 3.9) beyond which ammonia removal became less reliable. This suggests that during periods of higher $CBOD_5$ loading, the limited oxygen available in conventional systems is utilised by the faster growing heterotrophic bacteria that effectively out compete the ammonia oxidising bacteria for available DO. In contrast, ammonia removal was consistent in the aerated bed irrespective of loading rates confirming oxygen to be the rate limiting component over other mechanisms. In the current study, effective nitrification was observed across all the ammonia loading rates experienced (up to 10.1 $gNH_4^+\text{-}N/m^2/d$) indicating that the system had not become ammonia load limited. This observation offers the potential to increase loading rates further, and demonstrates the efficacy of the technology for highly loaded CWs, extending previous findings with landfill leachate at a loading rate of 0.9 $gNH_4^+\text{-}N/m^2/d$ (Nivala et al, 2007) and domestic sewage at a loading rate of 5.4 $gNH_4^+\text{-}N/m^2/d$ (Cottingham et al, 1999).

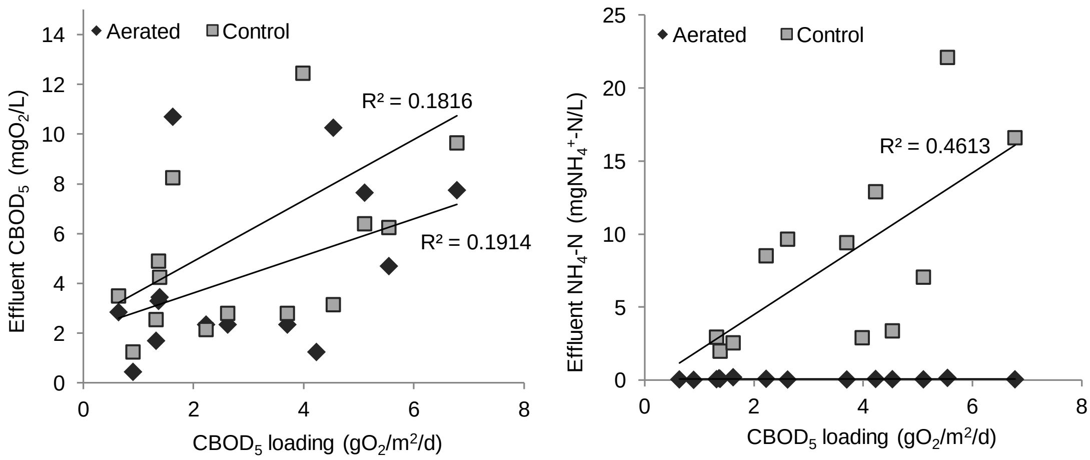


Figure 3.9 $CBOD_5$ loading rates vs. Effluent $CBOD_5$ concentrations (left) and effluent ammonium concentrations (right)

Nitrogen loss and retention is variable within CWs and is largely a result of nitrification/denitrification; plant uptake; sediment storage and nitrous oxide (N_2O) production via nitrification and incomplete denitrification (Maltais-Landry et al, 2009a). Total nitrogen removals in this study of 77 % and 55 % were recorded for the aerated and control beds respectively (n=3). A comparison of N species measured in the aerated and control beds (n=17) highlights the superior nitrification in the aerated system, with low ammonium and high nitrate recorded in the effluent (Table 3). Together with the knowledge that the presence of oxygen inhibits both synthesis and activity of denitrification enzymes (Kampschreur et al, 2009) the results suggest denitrification in the aerated bed is minimal. Evidence that partial denitrification is occurring in the control bed can be inferred from a 50 % reduction in nitrate levels.

Table 3.3 Influent and effluent N species loadings

N species	Influent	Effluent	
		Aerated	Control
	($g/m^2/d$)	($g/m^2/d$)	($g/m^2/d$)
NH_4-N	3.6	0.03	3.0
NO_3-N	4.7	7.2	2.4
NO_2-N	0.2	0.02	0.06
Total inorganic N	**8.5**	**7.3**	**5.5**
(TN	9.8	8.0	5.4)

Nitrogen species in the effluent of the non-aerated bed suggest 3.0 $g/m^2/d$ are unaccounted for compared to 1.2 $g/m^2/d$ in the aerated bed. Sediment storage has been measured to account for 27-63 % of N retention (Maltais-Landry et al, 2009a) and N retention via this method is likely to be greater in the control bed, due to the higher accumulated solids recorded herein. Nitrogen is also 'lost' from CWs via nitrous oxide emission. The most important parameters affecting N_2O emissions in wastewater treatment plants are cited as low DO in the nitrification and denitrification stages; increased NO_2 concentrations in both the nitrification and denitrification stages and limited availability of biodegradable organic carbon during denitrification (Kampschreur et al, 2009). As denitrification is considered to be minimal in the aerated bed it is expected the control bed is emitting higher levels of N_2O. In support of this reasoning, results of a mesocosm planted with P. *australis* recorded 0.4 % of TN emitted as N_2O in a non-aerated system compared to 0.1 % in an aerated (Maltais-Landry et al, 2009b). At O_2 concentrations < 1 mg O_2/L (as measured in the control bed) N_2O production can correspond to as much as 10 % of the N load (Kampschreur et al, 2009) and has been recorded at a maximum of 1.9 $gN_2O/m^2/d$ in CWs (Huang et al, 2012).

Total nitrogen removal of up to 96 % has been recorded in laboratory-scale aerated systems through configurational differences to the current study where only the front end of the bed is aerated, thus providing post anoxic zones (Maltais-Landry et al, 2009a,b). The similarity in organics removal between the two beds in the current study and the ability to denitrify at the back end of the bed in the Maltais-Landry et al, (2009a,b) study indicate that availability of organic matter is not limiting the extent of denitrification seen in these systems. Thus, should NO_3^- or TN removal be required in the future, this could be achieved by limiting the aeration of the bed to a fraction of the area.

Nitrifier activity is known to be susceptible to cold temperatures. In conventional HSSF CWs, temperature effects are typically quantifiable where readings fall below 15-10 °C (Vymazal, 1999; Delatolla et al, 2009). Results from this study relate primarily to operation where water temperatures ranged between 10-20 °C; consequently, the effect of colder temperatures in these full-scale systems have yet to be quantified. Historical records for this particular site indicate water temperatures can average 8 °C over the winter and indeed drop to 4 °C on extreme days. It is unclear if effluents at 8 °C would result in significantly lower nitrification efficiency. Even so, complete nitrification as achieved in this study is not commonly required; indeed, this particular site only has to produce an effluent of 4 $mgNH_4^+$-N/L for compliance purposes although it is currently discharging 0.2 $mgNH_4^+$-N/L

(95th percentile value). In addition, discharge permits are typically caveated for cold temperature conditions. In the UK, ammonia consents are mandatory with the exception of the unusual occurrences where water temperature falls below 5 °C. Therefore, whilst the nitrification rate of systems may be affected once lower temperatures ensue and should be monitored to determine the design envelope for this technology, current performance suggests the systems should continue to deliver a compliant effluent throughout the year.

Additional benefits of aeration were further seen with respect to the rate of accumulation and characteristics of the solids within the beds (Table 2), with the aerated bed having a lower quantity of solids and less compact than the control, which would favour water transport through the accumulated sludge. These results agree with higher hydraulic conductivity values found in the aerated bed when compared against the control. Other conventional HSSF CWs studied with the same hydraulic conductivity protocol have shown values of 300 m/d when the gravel is clean and 4-6 m/d when the gravel was deemed "clogged" (Pedescoll et al, 2010). Thus, whilst the aerated bed has higher conductivity values than the control, the potential extension of bed life cannot yet be quantified. Although the effect on system age requires long-term monitoring, this study provides initial information for identical, side-by-side full-scale systems with and without aeration.

The hydraulic conductivity results agree with a lower actual mean residence time recorded in the control bed, demonstrating an increase in preferential flow paths within the bed and increased overland flow. Suspended solids concentrations in the effluent wastewater (Figure 3.4, left) show the aerated bed contains similar solids in the effluent as the control bed, suggesting the decrease in solids within the aerated bed is not due to increased solids in the effluent caused by mixing. A comparison of the volatile suspended solids to the total suspended solids gives values of 52-66 % and 26-37 % VSS in the control and aerated beds, respectively; supportive of the theory that the abundance of oxygen is allowing increased degradation of organic matter. The full scale system results are in general agreement with mesocosm studies conducted by Ouellet-Plamondon et al, (2006) and Chazarenc et al, (2009), in that greater suspended solids removal (as opposed to storage) is achieved in aerated systems due to increased biological activity and more available pore space.

The fluctuation in MDI and the index of modal retention time observed with the tracer studies in the aerated bed at high and low flows suggests the hydraulic performance of the aerated bed is significantly influenced by the water level, where true sub-surface operation favours a more efficient system and a few millimetres of overland flow can result in a change from a CSTR to a more plug-flow reactor. As this was not the case for the control bed, the introduction of aeration into a conventional bed should take into account the potential for variable performance when the water level changes (e.g., due to bed clogging) from sub-surface to partial surface flow. In this study, the majority of the time the water level was above the gravel due to site operational issues and still achieved complete nitrification, similar $CBOD_5$ and TSS removal efficiency, and better hydraulic conductivity of the bed under the same conditions as the control, non-aerated wetland.

The feasibility assessment highlighted the advantages and weaknesses the different biofilm technologies can offer against an aerated constructed wetland built as tested in this full-scale study. Whilst the calculations required a number of assumptions and hence provide an initial guidance only, the overall comparison does show the efficacy of using the retrofit option on sites that currently contain HSSF CWs but need to enhance nitrification performance. In the case where conventional CWs did not exist on-site, the major trade off would become land requirements; where the non-wetland options would require between 2 % (SAF) and 19 % (TF 3ry) of the aerated wetland land footprint. In cases of limited land availability or land prices becoming prohibitive, TF 3ry or SAF options would become more suitable, providing a balance of small footprint and cost compared to an additional RBC.

3.6 Conclusions

The performance of a full scale aerated HSSF CW was analysed to determine contaminant removal, hydraulic behaviour and solids characterisation in comparison to a non-aerated control. Observations from the full scale trial have demonstrated that artificially aerating HSSF CWs results in complete nitrification of the secondary effluent without negatively impacting on solids and $CBOD_5$ removal efficiencies. Aeration also results in different solids accumulation quantities and characteristics, with less compact solids, increased hydraulic conductivity, and better hydraulic efficiency in the aerated bed under sub-surface flow, resulting in a more efficient reactor. A cost comparison of implementing the technology against conventional biofilm options suggests artificial aeration of tertiary treatment horizontal flow constructed wetlands is competitive in terms of cost, treatment performance, and carbon footprint where nitrification is required on a small works site. Long-term studies are required to assess the lifespan of aerated HSSF CWs and to determine the sustainability of the trends observed in this study.

3.7 Acknowledgements

The authors would like to express their gratitude for the financial and resource support of the work from Severn Trent Water Limited and to the EPSRC for their allocation of funding and support of the project.

3.8 References

American Public Health Association (APHA) (2005) *Standard methods for the examination of water and wastewater (21st edn)*. Port City Press, Baltimore, Maryland

Brix H., Arias C., Johansen, N. (2002) BOD and nitrogen removal from municipal wastewater in an experimental two-stage vertical flow constructed wetland system with recycling. Proceedings of the *8th International Conference on Wetland Systems for Water Pollution Control*, 16th-19th September. Arusha, Tanzania

Chazarenc F., Gagnon V., Comeau Y., Brisson J. (2009) Effect of plant and artificial aeration on solids accumulation and biological activities in constructed wetlands. *Ecological Engineering* 35(6):1005–1010

Cooper P. (1999) A review of the design and performance of vertical-flow and hybrid reed bed treatment systems. *Water Science and Technology* 40(3):1-9

Cooper P., Job J., Green B., Shutes R. (1996) *Reed Beds and Constructed Wetlands for Wastewater Treatment*. WRc Publications, Swindon, UK

Cottingham P., Davies T., Hart B. (1999) Aeration to promote nitrification in constructed wetlands. *Environmental Technology* 20(1):69–75

Davies H., Hart B. (1990) Use of aeration to promote nitrification in reed beds treating wastewater. In: Cooper, P.F., Findlater, B.C. (Eds.), *Constructed Wetlands in Water Pollution Control.* Pergamon Press, Oxford, UK

Delatolla R., Tufenkji N., Comeau Y., Gadbois A., Lamarre D., Berk D. (2009) Kinetic analysis of attached growth nitrification in cold climates**.** *Water Science and Technology* 60(5):1173-84

De Novio N., Saiers J., Ryan J. (2004) Colloid movement in unsaturated porous media: Recent advances and future directions. *Vadose Zone Journal* 3(2):338-351

Dotro G., Jefferson B., Jones M., Cartmell E., Vale P., Stephenson T. (2011) A review of the impact and potential of intermittent aeration on continuous flow nitrifying activated sludge. *Environmental Technology* 32(15):1-13

Griffin P., Wilson L., Cooper D. (2008) Changes in the use, operation and design of sub-surface flow constructed wetlands in a major UK water utility. Proceedings of the *11th International Conference on Wetland Systems for Water Pollution Control*, 1st–7th November. Vikram University and IWA, Indore, India

Kadlec R., Wallace S. (2009) *Treatment Wetlands (2nd edn.)* Taylor & Francis Group, FL

Knowles P., Dotro G., Nivala J., García J. (2011) Clogging in sub-surface flow treatment wetlands: Occurrence and contributing factors. *Ecological Engineering* 37(2):99-112

Langergraber G. (2012) Are constructed treatment wetlands sustainable sanitation solutions? Proceedings of the *13th IWA Specialized Group Conference on Wetland Systems for Water Pollution Control*, 25th-29th November. Perth, Australia

Maltais-Landry G., Maranger R., Chazarenc F. (2009a) Nitrogen transformations and retention in planted and artificially aerated constructed wetlands. *Water Research* 43(2):535-545

Maltais-Landry G., Maranger R., Brisson J. (2009b) Effect of artificial aeration and macrophyte species on nitrogen cycling and gas flux in CWs. *Ecological Engineering* 35(2):221-229

Metcalf & Eddy., Tchobanoglous G., Burton F., Stensel, H. *(*2003) *Wastewater engineering: treatment and reuse (4th edn)*, Metcalf Eddy Inc., McGraw-Hill, New York

Nivala J., Hoos M., Cross C., Wallace S., Parkin G. (2007) Treatment of landfill leachate using an aerated, horizontal subsurface-flow constructed wetland. *Science of the Total Environment* 380(1-3):19–27

Nivala J., Rousseau D. (2009) Reversing clogging in subsurface-flow constructed wetlands by hydrogen peroxide treatment: two case studies. *Water Science and Technology* 59(10):2037–2046

Nowak O. (2000) Upgrading of wastewater treatment plants equipped with rotating biological contactors to nitrification and P removal. *Water Science and Technology* 41(1):145-153

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

Pedescoll A., Uggetti E., Llorens E., Granés F., García D., García J. (2009) Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. *Ecological Engineering* 35(8):1216-1224

Pedescoll A., Samso R., Romero E., Puigagut J., Garcia J. (2011) Reliability, repeatability and accuracy of the falling head method for hydraulic conductivity measurements under laboratory conditions. *Ecological Engineering* 37(5): 754-757

Tiller F., Lloyd P. (1972) *Theory and practice of solid-liquid separation (4th edn)*. University of Houston, US

Tyroller L., Rousseau D., Santa S., García, J. (2011) Application of the gas tracer method for measuring oxygen transfer rates in subsurface flow constructed wetlands. *Water Research* 44(14):4217-4225

UKWIR. (2008). *Carbon Accounting in the UK Water Industry: Operational Emissions*. Report Reference No. 08/CL/01/5. UK Water Industry Research Limited, London

Vymazal J., Kröpfelová L. (2008) *Wastewater treatment in Constructed Wetlands with Horizontal Sub-Surface Flow*. Springer

Wakeman R., Tarleton S. (2005) *Solid/Liquid Separation: Principles of industrial filtration*. Elsevier Ltd, Oxford

Wallace S. (2001). System for removing pollutants from water. US Patent: US 6,200,469 B1

Wallace S., Kadlec R. (2005) BTEX degradation in a cold-climate wetland system. *Water Science and Technology* 51(9):165-171

Wallace S., Higgins J., Crolla A., Kinsley C., Bachand A., Verkuijl, S. (2010) *High rate ammonia removal in engineered wetlands*. Proceedings of the *10th International Conference on Wetland Systems for Water Pollution Control*, 4th-8th October. Venice, Italy

Weedon C. (2003). Compact vertical flow constructed wetland systems: first two years performance. *Water Science and Technology* 48(5):15-23

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

Chapter 4

Effect of media on oxygen transfer in packed beds

Effect of media on oxygen transfer in packed beds

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4.1 Abstract

The aim of this study was to determine the difference in oxygen transfer rates in a densely media packed bed vs. a non-media system using a non-porous diffuser. Various configurations of diffuser hole diameter size and air flow rate were tested to determine their affect on associated aeration efficiency and supporting bubble visualisation experiments were designed to provide further insight and explanation of the mechanisms occurring within the system. The findings of this study indicate enhanced mass transfer quantified by higher K_La_{20s} of 3.2-24.7 h^{-1} recorded in the media compared to 0.4-6.1 h^{-1} in the non-media tests. Smaller bubble diameters were recorded in the media systems ranging from 0.5-9.0 mm to 2.0-15.0 mm in the media and non-media systems respectively over all tested orifice sizes and air flow rates. Lower bubble velocities of 48-202 mm/s were recorded in the media systems compared to 202-330 mm/s in the non-media. In addition, increased gas hold up caused by tortuous bubble pathways and retardation in the void spaces, observed in the bubble visualisation column, increased the residence time of the bubbles; increasing the time for oxygen transfer to occur. In terms of overall aeration efficiency, the packed bed systems produced an α^M factor of 3.2 improvement compared to non-media systems.

Results provide the basis for energy reduction on small wastewater treatment plants, introducing the possibility for renewable energy sources to adequately run these systems.

Keywords: Oxygen transfer, packed beds, constructed wetlands

4.2 Introduction

Aeration in the form of the addition of rising bubbles to the base of wastewater treatment reactors is commonly used to facilitate biological nutrient removal processes such as nitrification. Air bubble size and gas-hold up control the oxygen transfer rate, as these factors determine the gas-liquid interfacial area available for transfer (Fujie et al, 1992). Efforts to enhance oxygen transfer by the addition of rising bubbles include manipulation of the depth of the reactor, variation of air flow rates (Ashley et al, 1991; Ashley et al, 1992; Cornel, 2003), the diffuser type, placement and density (Hasanen et al, 2006; Mueller et al, 2002) and re-direction of the bubble flow path via inclined bubble aeration (Kim et al, 2013). The optimal system aims to maximise the residence time of the bubble in the reactor and minimise bubble size which will enhance transfer with minimal energy input.

In traditional systems the air bubble size is controlled by the combination of the orifice size used and the air flow rate. Previous research has shown that the average bubble size increases as air flow rate through each diffuser increases (Polli et al, 2002), decreasing the benefits of using smaller orifice hole diameters (Hasanen et al, 2006). At lower air flow rates bubbles form more slowly and the oxygen transferred to the wastewater during formation increases (Ashley et al, 1991) and at higher flow rates space between bubbles is decreased so lateral diffusion is limited. Increasing diffuser density can offset some of these impacts and has been shown to increase oxygen transfer efficiency (OTE) with typical diffuser densities in the range of 0.2-0.5 m^2/ diffuser (ASCE, 1988). Typical diffuser depths of 3-6 m at air flow rates of 7-68 m^3/h/unit in non-porous diffuser systems correspond to clean water standard oxygen transfer efficiencies (SOTEs) of 2-4 %/m. Similar depths are used for porous systems with lower air flow rates per unit (0.6-5.5 m^3/h) resulting in higher SOTEs of 5-7 %/m (Mueller et al, 2002). In packed bed systems e.g. aerated constructed wetlands, biological aerated filters (BAFs), and submerged aerated filters (SAFs), optimised design of the aeration system is complicated due to media presence negating some of the traditional methods for improved oxygen transfer. As such, accepted methods to enhance transfer in non-media systems such as activated sludge plants (ASPs) are not always applicable. Increased transfer has been recorded in media filled tanks compared to those with no media (Harris et al, 1996) and oxygen transfer in a SAF was reported to be higher than the same system with no media operating at the same flow rate, resulting in a power economy of 2-3 kgO_2/kWh; twice that of an ASP (Fujie et al, 1992; Fujie and Kubota, 1986). The increase in transfer has been attributed to increasing bubble hold up time and controlling bubble size (Fujie et al, 1992, Harris et al, 1996).

Non-porous diffusers are also termed coarse bubble aerators, referring to typical orifice sizes of 4.8-9.5 mm producing bubbles 6-10 mm (Mueller et al, 2002) whilst porous diffusers represent fine bubble systems producing 2-5 mm bubbles (Mueller et al, 2002). The presence of the media can also cause the coalescence of fine bubbles, decreasing bubble surface area, resulting in a lower OTE and/or break-up of coarse bubbles which will increase OTE; indicating that the benefit of porous (fine bubble) diffusers may be reduced in such systems compared to traditional coarse bubble diffuser systems (ASCE, 1988). For instance, relative transfer in an integrated fixed-film activated sludge (IFAS) configuration saw little difference in SOTE between fine bubbles with media and without: 8-16 % and 9-17 % respectively. The impact of the media on transfer was more pronounced in the coarse bubble system, which recorded an SOTE of 6-7 % with media compared to 4-6 % without (Collignon, 2006). The SOTE of the fine bubble aeration decreased with increased airflow rate (16 % at 5 m/h compared to 8 % at 35 m/h); whilst the SOTEs for the coarse bubble system was relatively similar at all tested air flow rates (6 % at 5 m/h and 7 % at 35 m/h). Break-up and coalescence was attributed to the size, shape and density of the media. Consequently it is standard practice to fit packed bed systems with non-porous, coarse bubble diffusers.

Aerated constructed wetlands (CWs) are a relatively new form of packed bed system created to enhance oxygen delivery into this traditionally passive treatment solution used on small sewage treatment works. The addition of aeration has been demonstrated to provide an effective solution for tertiary nitrification which is easily adaptable into existing constructed wetland assets and hence is gaining consideration when treatment upgrading is required with respect to ammonia discharges (Chapter 3, Butterworth et al, 2012; Fan et al, 2013).

Commercial systems utilise coarse bubble aeration from 2 mm orifice diffusers below a 0.6 m bed of gravel (6-12 mm) with a measured porosity of 0.4. The packing density is comparable to traditional media systems such as BAFs (0.4 - Mann and Stephenson, 1997) and less than trickling filters using reported porosities of 0.8 (Dermou et al, 2007) that are typically employed for tertiary nitrification on small works. Operational experience indicates that the beds are typically over aerated resulting in saturated oxygen levels leading to poor energy utilisation. For instance, a recent energy survey at a small works including an aerated wetland, revealed energy usage akin to a main stage activated sludge process at around 2.45 Wh/PE (Pearce, 2013). Consequently, it has been suggested that such an approach is irreconcilable with the low energy / passive preferences for small works operation. However, the author posit that a better understanding of oxygen transfer in such systems will enable a more rational design basis and a significant reduction in overall energy demand enabling the utilisation of aeration in a manner more compatible with the wider operational aspirations.

Accordingly, the aim of this study was to determine oxygen transfer rates in a densely media packed bed using non-porous diffusers of various configurations of diffuser orifice diameter size and air flow rate to determine their effect on associated aeration efficiency. Supporting bubble visualisation experiments were designed to provide further insight and explanation of the mechanisms occurring within the system. Corresponding non-media systems for all experiments were conducted to serv as a control.

4.3 Materials and methods

4.3.1 Oxygen transfer theory

The rate of oxygen mass transfer is commonly described according to two-film theory (Lewis and Whitman, 1924); that envisages stagnant gas and liquid films at the gas-liquid interface. As oxygen solubility in water is low, the liquid side becomes rate limiting such that the rate of oxygen transferred is principally considered dependant on the thickness of the liquid film. The oxygen transfer rate is then described by the liquid film transfer coefficient (K_L); the ratio of bubble surface area to water volume (a); the volume of the reactor (V) and the oxygen deficit (C*-C). As the gas-liquid interfacial area (a) is not easily measured, it is amalgamated with K_L to produce the volumetric mass transfer coefficient (K_La) that can be experimentally determined. The final result of the test is expressed as the standard oxygen transfer rate (SOTR) (Equation 1), a hypothetical mass of oxygen transferred per unit of time at zero dissolved oxygen concentration, water temperature of 20 °C and barometric pressure of 1 atm under specified gas rate and power conditions (Strenstrom et al, 2006).

$$SOTR = K_La_{20} \cdot V \cdot (C^*-C) \quad (1)$$

SOTR = Standard oxygen transfer rate in liquid volume V (gO_2/d)
K_La_{20} = Volumetric mass transfer coefficient at 20°C (h)
V = Volume of water (m^3)
C* = Oxygen saturation concentration (gO_2/m^3)
C = Oxygen concentration in tank (gO_2/m^3)

The SOTR is then compared to the mass of oxygen delivered to derive the standard oxygen transfer efficiency (SOTE) (Equation 2):

$SOTE = SOTR/w_0$ (2)

SOTE = Standard oxygen transfer efficiency (%)
SOTR = Standard oxygen transfer rate in liquid volume V (gO_2/d)
w_0 = Mass flow of oxygen in the air stream (gO_2/d) (Equation 3)

$w_0 = \rho Q_a$ (Mueller et al, 2002) (3)

w_0= Mass flow of oxygen in the air stream (gO_2/d)
ρ = Oxygen density (gO_2/m^3)
Q_a = Air flow rate at standard conditions (g/d)

SOTEs are commonly reported per m depth of reactor to enable comparison to similar technologies and this approach has been adopted in the current study.

4.3.2 Affecting factors

Aeration system performance is often reported for clean water conditions at standard temperature and pressure. In reality, the oxygen transfer rate is affected by process conditions including variations in temperature, pressure and process water characteristics. Increasing temperature causes a reduction in oxygen solubility and can be corrected for variation through inclusion of an Arrhenius term (Equation 4).

$K_La_{(T)} = K_La_{(20)}\theta^{T-20}$ (4)

$K_La_{(T)}$ = Volumetric mass transfer coefficient at temperature T (°C)
$K_La_{(20)}$ = Volumetric mass transfer coefficient at 20 °C
θ = Effect of temperature on mass transfer coefficient, typically given the value 1.024 for diffused and mechanical aeration devices (Metcalf and Eddy, 2003)

Correction for differences in barometric pressure (Ω) in tanks less than 6 m depth are estimated by calculating the ratio of atmospheric pressure to the standard total pressure (ASCE, 1998). In non clean systems additional corrections are required based on the impact of surface active agents reducing the mass transfer coefficient (α) (Wagner and Pöpel, 1996) and presence of salts on equilibrium solubility (β) (ASCE, 1988). However, clean water testing provides the baseline for aeration system performance as it is comparatively reproducible regardless of geographical location (Mueller et al, 2002) and is useful in deriving mechanisms of mass transfer when analysing new systems and hence represents the basis of the current investigation.

4.3.3 Aeration efficiency

The standard aeration efficiency (SAE) is a useful term to quote whilst comparing processes as it expresses the quantity of oxygen supplied (at standard conditions) per unit of energy consumed (Equation 5).

$$SAE = SOTR/P \qquad (5)$$

SAE = Standard aeration efficiency (kgO_2/kWh)

SOTR = Standard oxygen transfer rate in liquid volume V (gO_2/d)

P= Power (Equation 6)

The power requirement for the delivery of the required air based on adiabatic compression is given by Metcalf and Eddy (2003):

$$P_w = wRT_1/29.7.n.e[(p_2/p_1)^{0.283})-1] \qquad (6)$$

P_w = Power requirement of blower (kW)

w = Weight of flow of air (kg/s)

R = Gas constant for air (8.314 kJ/k mol K)

T_1 = Absolute temperature (K)

p_1 = Absolute inlet pressure (atm)

p_2 = Absolute outlet pressure (atm)

n = (k-1)/k = 0.238 for air

k = 1.395 for air

29.7 = Constant for SI unit conversion

e =Blower efficiency

A detailed example calculation can be found in supplementary information 4.1.

4.3.4 Oxygen transfer tests with and without media in tank

Oxygen transfer tests were carried out in a 1.5 m x 1.5 m x 1.0 m depth tank (Broomhill Composites Ltd.). The diffuser was made from 12 mm (inner diameter) LDPE pipe fixed to a plastic grid and connected to the air supply through a vertical supply pipe (Figure 4.1). Perforations were then drilled into lateral pipe to form a grid of holes spaced 300 mm apart. The plastic grid was lowered onto the tank floor on top of a layer of gravel to ensure it was level. When a new orifice size was required the plastic grid was removed, amended and replaced.

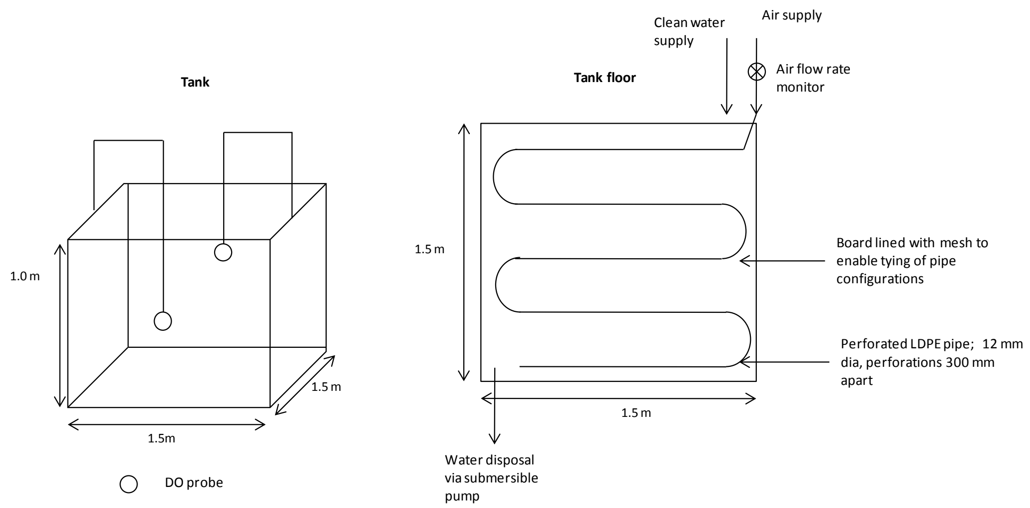

Figure 4.1 Tank and diffuser set up

Orifice sizes of 0.5, 0.8, 1.0, 2.0 and 3.0 mm were tested at air flow rates of 10, 15, 20, 40, 60, 100 L/min. The design was based on UK aerated CWs that typically consist of a 12 mm aeration pipe and 2 mm hole diameters spaced 300 mm apart.

Oxygen was removed from a known volume of water in a separate mixing tank using a well mixed solution of sodium sulphite (Na_2SO_3) at 7.88 mg/L per 1.0 mg/L dissolved oxygen (DO) combined with a cobalt catalyst (0.3 mg/L) until a DO of less than 0.5 mg/L was attained. The test tank was then filled with de-oxygenated water to a depth of 0.6 m from the mixing tank via a tap and pipe. The air supply was switched on and DO recorded periodically at 0.1 m and 0.4 m below the water/gravel surface using DO probes (Hach-Lange HQ40D). Freshly prepared water was used for each individual test. For tests including media, the tank was filled with 6-12 mm gravel as used as standard in Severn Trent Water constructed wetlands with a tested porosity of 0.4. The terms k_La and C* were analysed using the two film theory mass transfer model (Equation 1).

The affect of the media was quantified by creating media factors (α^M); calculated by plotting media versus no media values against each other for the comparative variables.

4.3.5 Bubble visualisation column

Bubble behaviour was visualised using a Perspex column (Figure 4.2). Pipe was inserted into the bottom of the tank with one central orifice and linked to an air supply. The same pipe, orifice sizes and air flow rates were realised as used in the test tank. The column was filled with clean water or water and media to a depth of 0.6 m. A metal graticule (mm) was attached across the column to allow post experiment bubble diameter analysis. The air was switched on and video footage taken at 25 frames/second (Sony digital camera model SLT-A55V).

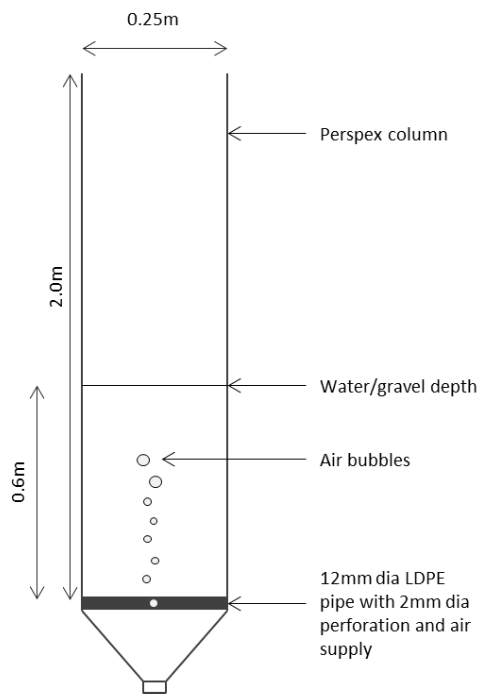

Figure 4.2 Bubble visualisation column set up

Still images were captured from the video footage using VLC media player (VideoLan, 2013) and sized relative to the graticule. Velocities were measured as vertical equivalence by measuring the number of frames required for bubbles to travel a fixed vertical distance. One hundred bubbles were counted for each experiment (hole size and flow rate) to provide a representative sample as previous experiments demonstrated that the standard error of the distribution remained stable once more than 80 bubbles were analysed (data not shown).

4.4 Results

4.4.1 Oxygen transfer tests with and without media in tank

Oxygen transfer was enhanced due to the presence of media across all experiments. For example, in the case of a 3 mm orifice hole size operating at an air flow rate of 20 L/min the time required to produce an increment rise in DO per unit tank volume was 116 s compared to 515 s in the media and non-media systems respectively (Figure 4.3). Correspondingly, the time required to reach a DO set point was shorter in the presence of media (Figure 4.3).

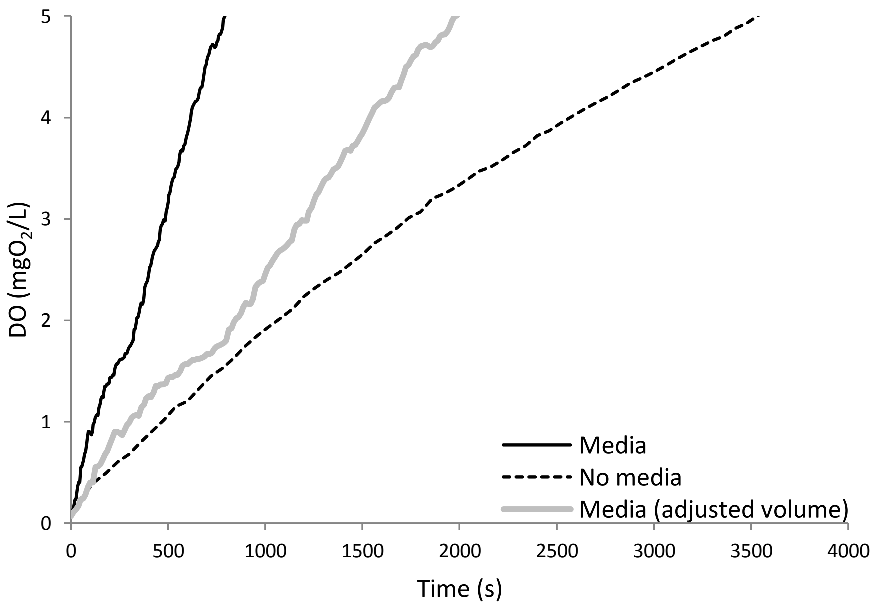

Figure 4.3 Relative rates of dissolved oxygen concentration in media vs. non-media filled systems (3 mm orifice size; 20 L/min air flow rate)

To illustrate, based on a 5 mgO_2/L target (set for ease of measurement) the non - media systems required on average 6.6 times as long (range 2.2-13.0). The impact of the media on the rate of DO increase was reduced when normalising for the total volume of water such that the DO set point was reached only 2.6 times faster on average than when no media was present.

The mass transfer coefficient (K_La_{20}) in the media ranged between 3.2-24.7 compared to 0.4-6.1 in the non-media tests (Figure 4.4). A general trend of increased K_La_{20} with increasing air flow rates was observed across all conditions with, for example, an increase in the average K_La_{20} from 6.2 h^{-1} at 10 L/min to 22.8 h^{-1} at 100 L/min in the case of the 1 mm hole size during the media trials (Figure 4.4). The average standard deviation for each orifice size in the non-media tests ranged from 0.04-0.09 compared to a much larger deviation recorded in the media tests of 2.8-4.6 (error bars on Figure 4.4).

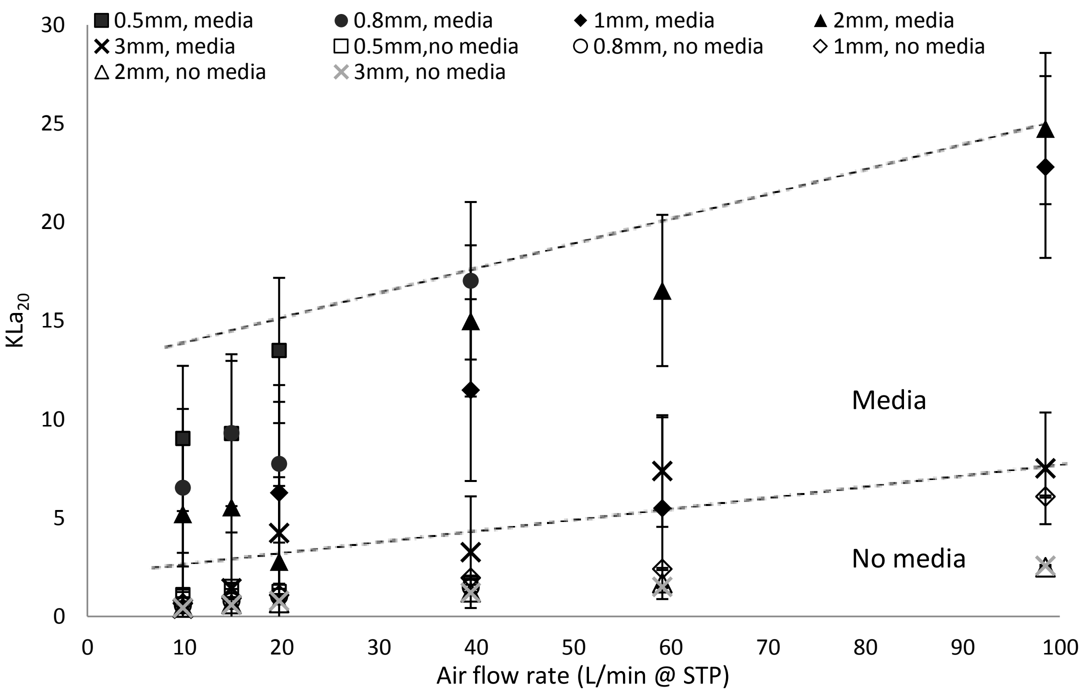

Figure 4.4 Variation in volumetric mass transfer coefficient at tested orifice size and air flow rates

The mass transfer coefficient increased as a function of air flow rate in both systems. In the case of the non-media system small orifice sizes (0.5-1.0 mm) resulted in higher K_Las than coarse orifice sizes (2.0-3.0 mm) congruent with standard aeration knowledge (Mueller et al, 2002). The same pattern held during the media case but much greater variation was observed between trials, reducing the clarity of the relationship which is indicative that other mechanisms were operating over and above those when no media was present. The SOTE/m in the non-media systems was generally less than half than that recorded in corresponding media tests below air flow rates of 60 L/min (Figure 4.5).

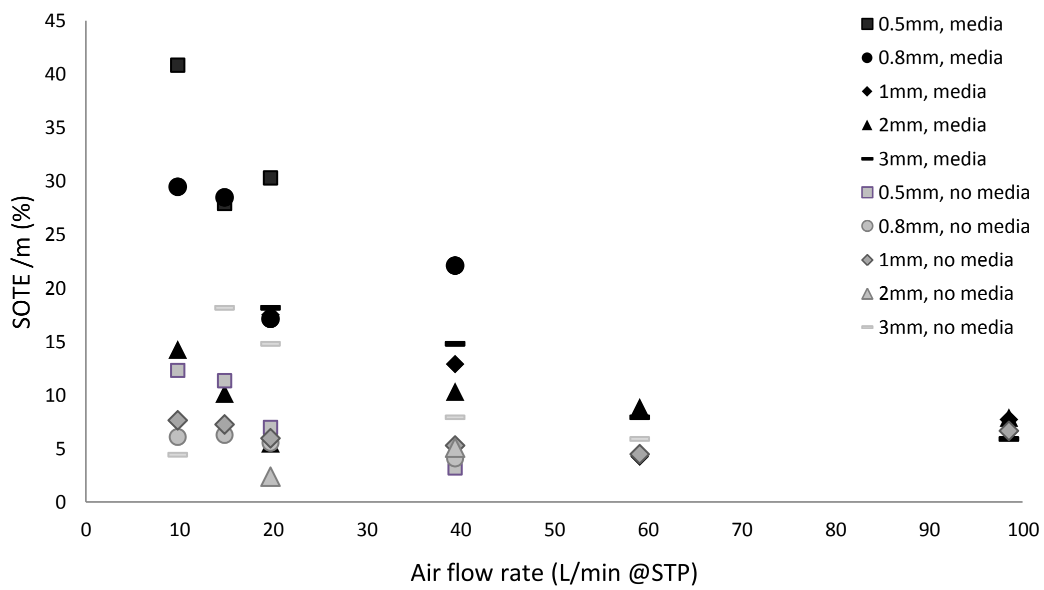

Figure 4.5 Average SOTEs (%/m) for all tested air flow rates and orifice sizes with and without media

The difference became less apparent as air flow rate increased such that at an air flow rate of 100 L/min no discernible difference was observed. The lowest efficiencies (2-8 %/m) relate to tests conducted with no media and remain comparatively low despite changes in air flow rate. At the low flow rates the 2 mm and 3 mm orifice sizes in the media tests produced results in a similar range of 0.5, 0.8 and 1 mm with no media. The high K_La values observed in the media systems at 100 L/min (Figure 4.4) did not translate into corresponding high SOTEs as the porosity of the reactor does not influence the measured K_La and is only accounted for during the conversion of the data into SOTR (Equation 1).

4.4.2 Bubble visualisation column

To elucidate the mechanisms responsible for the enhanced oxygen mass transfer a series of bubble visualisation experiments were undertaken (Figure 4.2). The bubble size in the media systems tended to be smaller and distributed over a narrower range compared to the non-media systems indicated by size ranges of 0.5-9.0 mm and 2.0-15.0 mm respectively. For instance, at an air flow rate of 20 L/min through a 0.5 mm orifice (Figure 4.6), the majority of the bubble diameters in the non-media system fell between 3.0-5.0 mm (85 %) compared to 51 % in the media system suggesting the presence of media was reducing the bubble size and hence increasing the total surface area for mass transfer per unit volume of air. The percentage of bubbles between 3-5 mm decreased with larger orifice sizes with levels of 73 %, 83 % and 54 % in the case of the non media trials with orifice sizes of 0.8 mm, 1.0 mm and 2.0 mm respectively, indicating significant coalescence when using smaller orifices. Corresponding figures in the case of the media trials were 38 %, 63 % and 55 % respectively indicating that the impact of media was less significant as the orifice size increased and confirmed a difference in underlying mechanisms. Although some differences between media vs. non-media systems were apparent, the data suggest orifice size was not the controlling factor on bubble diameter in either system as demonstrated by median bubble sizes of 4.0 mm, 4.0 mm, 3.0 mm and 4.0 mm for the 0.5 mm, 0. 8 mm, 1.0 mm and 2.0 mm holes sizes respectively in the case of media and 4.0 mm, 6.0 mm, 4.0 mm and 5.0 mm bubbles in the case with no media.

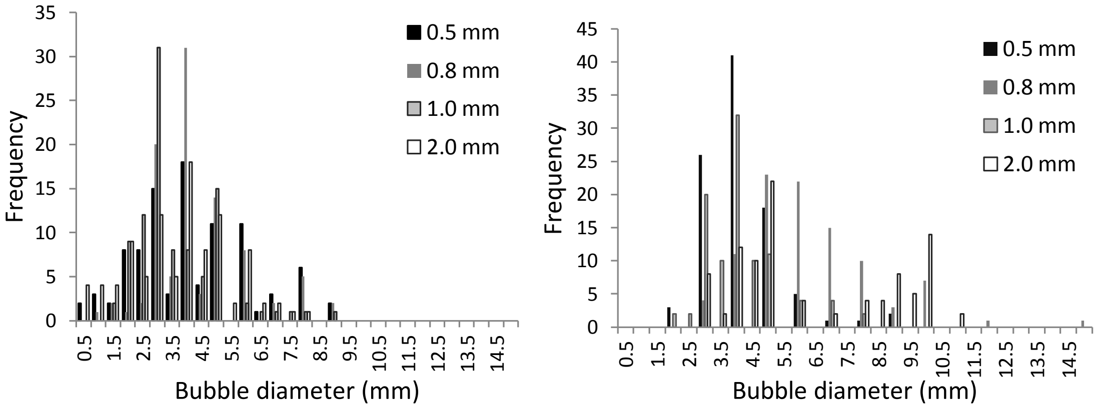

Figure 4.6 Bubble diameter frequency in the media column (left) and the non-media column (right) at air flow rate 20 L/min

The variation of bubble diameters was analysed in relation to air flow rate (Figure 4.6). Larger and more variable bubble diameters were found in the non-media systems compared to those with media (Figure 4.7). Bubble sizes increased in the non-media systems in relation to increasing flow rates at all orifice sizes. For example, in the case of a 0.8 mm orifice size upper quartile values of 6.0, 7.0 and 8.0 mm were observed for flow rates of 15, 20 and 40 L/min respectively. The media systems showed no relationship between air flow rate and bubble diameter with corresponding upper quartile values of 4.0, 5.0 and 4.0 mm. These data suggest flow rate had an impact on bubble size in the non-media systems but not on the media systems.

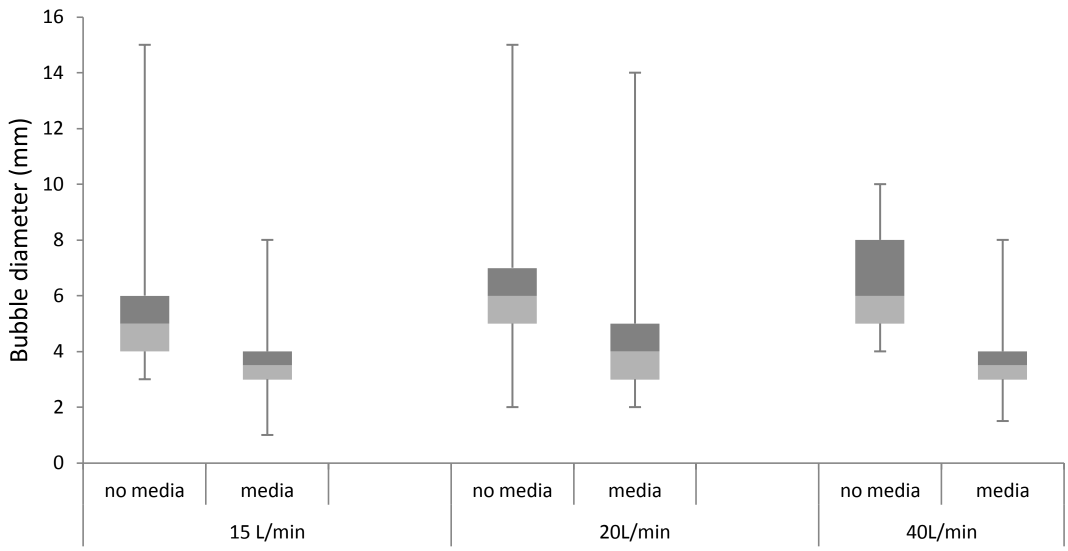

Figure 4.7 Box and whisker plots of bubble diameters recorded at 0.8 mm diffuser size in media and non-media systems at various air flow rates. The box represents the interquartile range; the line indicates the median and the whiskers the maximum and minimum

Lower velocities were measured in the media systems compared to the corresponding non-media systems at all orifice sizes (Figure 4.8). To illustrate, over all flow rates, velocities ranged from 48-202 mm/s in the media systems compared to 202-330 mm/s in the non-media. The impact of air flow rate was minimal with respect to the rise velocities of similar bubble sizes when using an orifice hole size above 0.5 mm and air flow rates above 20 L/min. For example, in the case of a 0.8 mm orifice hole size, the bubble rise velocity was 129±24, 133±30 and 115±15 mm/s for 15, 20 and 40 L/min respectively compared to 295±49, 259±37 and 287±53 mm/s in the non-media system (Figure 4.8).

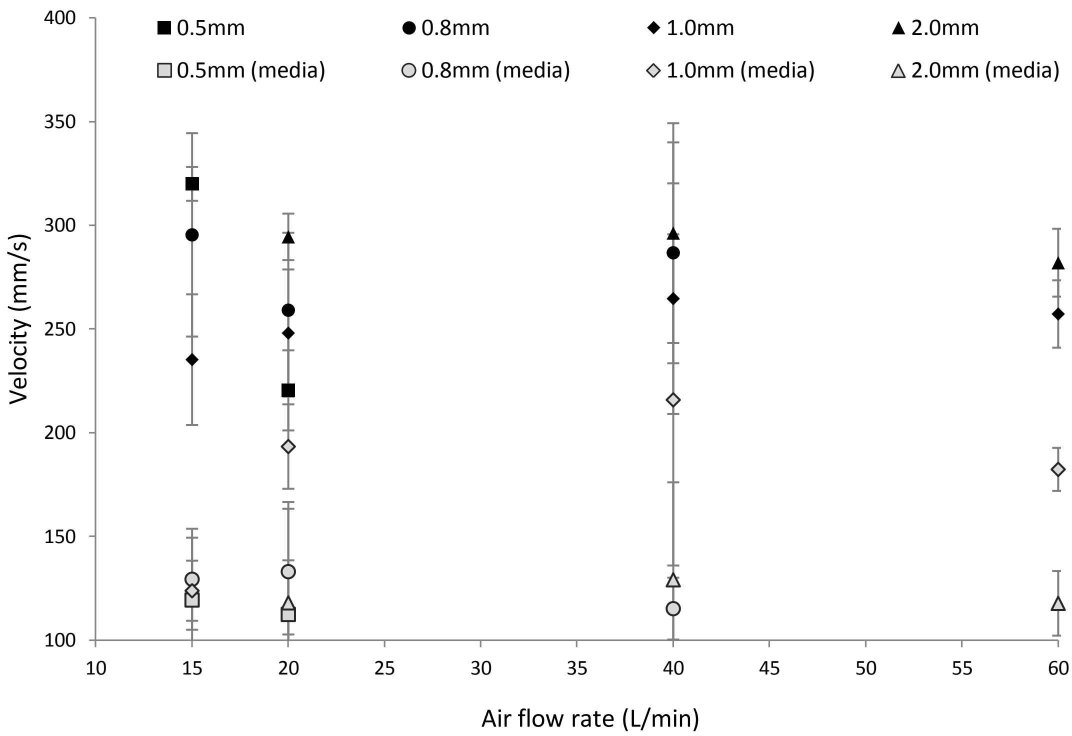

Figure 4.8 Average bubble velocities (mm/s) at various orifice size and air flow rates with and without media

Video footage at the slowest flow rate in the non-media system confirmed individual bubbles were formed constantly with little mixing and coalescence supported by a narrow and consistent bubble pathway (Figure 4.9, left). In contrast, bubbles observed at higher flow rates resulted in coalescence leading to the formation of larger bubbles and increased mixing (Figure 4.9, middle). Comparatively, in the media systems, bubbles travelled via various pathways; tortuous in comparison to the non-media and therefore spending a comparatively longer time in the system (Figure 4.9, right). In addition, a number of bubbles were trapped in the voids between the gravel until the next flow of bubbles took the same pathway and dislodged them, increasing gas hold up and further extending the residence time of the bubble. At higher flow rates this phenomenon was not observed as prevalently as the speed and force of the pathways was too strong to allow extra duration in the void. For example, at 1 mm orifice size this phenomenon accounted for approximately 15-20 % of the gas bubbles, increasing the time spent in the reactor by 1-10 s per bubble at 15 L/min and fractions of a second-2 s at 60 L/min.

Figure 4.9 Bubble pathways in non-media system (0.8 mm), 15 L/min (left) and 60 L/min (middle) and media system (right)

Visual inspection of the video footage also revealed the zone of influence (ZOI) per orifice to be in the range 150-300 mm from the centre point of the orifice hole. For instance in the case of the 0.5 mm orifice, the majority of the bubbles were observed within the first 150 mm with progressively fewer bubbles travelling as the distance extended away from the centre point of the orifice. The ZOI increased with bubble size and flow rate such than at 100 L/min and 2.0 mm orifice size the

ZOI recorded a larger range with a more even distribution of bubbles consistently around 300 mm compared to the small orifice size and low flow rate.

4.5 Discussion

The findings of this study indicate enhanced mass transfer in the packed bed system if the applied air flow rate is sufficiently low. Both orifice size and air flow rate influenced the overall mass transfer rates even though the bubble size was effectively constant, indicating that another mechanism was controlling the overall transfer rate. Accordingly, the data suggest that the increased bubble hold up and more tortuous bubble pathways observed in the media systems results in the enhanced transfer.

The impact of the observed enhanced mass transfer is illustrated through calculation of the standard aeration efficiency (SAE) to express the quantity of oxygen supplied per unit of energy consumed (supplementary information 4.1). SAEs decreased with increasing air flow rate and orifice size for both media and non-media systems with a more pronounced affect in the media systems. To illustrate, SAEs varied from 1.1-24.5 kgO_2/kWh in media systems and 1.7-7.4 kgO_2/kWh in the non-media. This compares to typical levels of 3.1-7.2 kgO_2/kWh for various modern diffuser types and air flow rates tested in clean water with no media (Kember, 2007). Comparing the SAEs in media vs. non-media systems indicates an overall enhancement factor (α^M) of 3.2 (not including variations due to orifice size or air flow rates) quantifying the extent the media presence improves relative SAE between the systems (Figure 4.10).

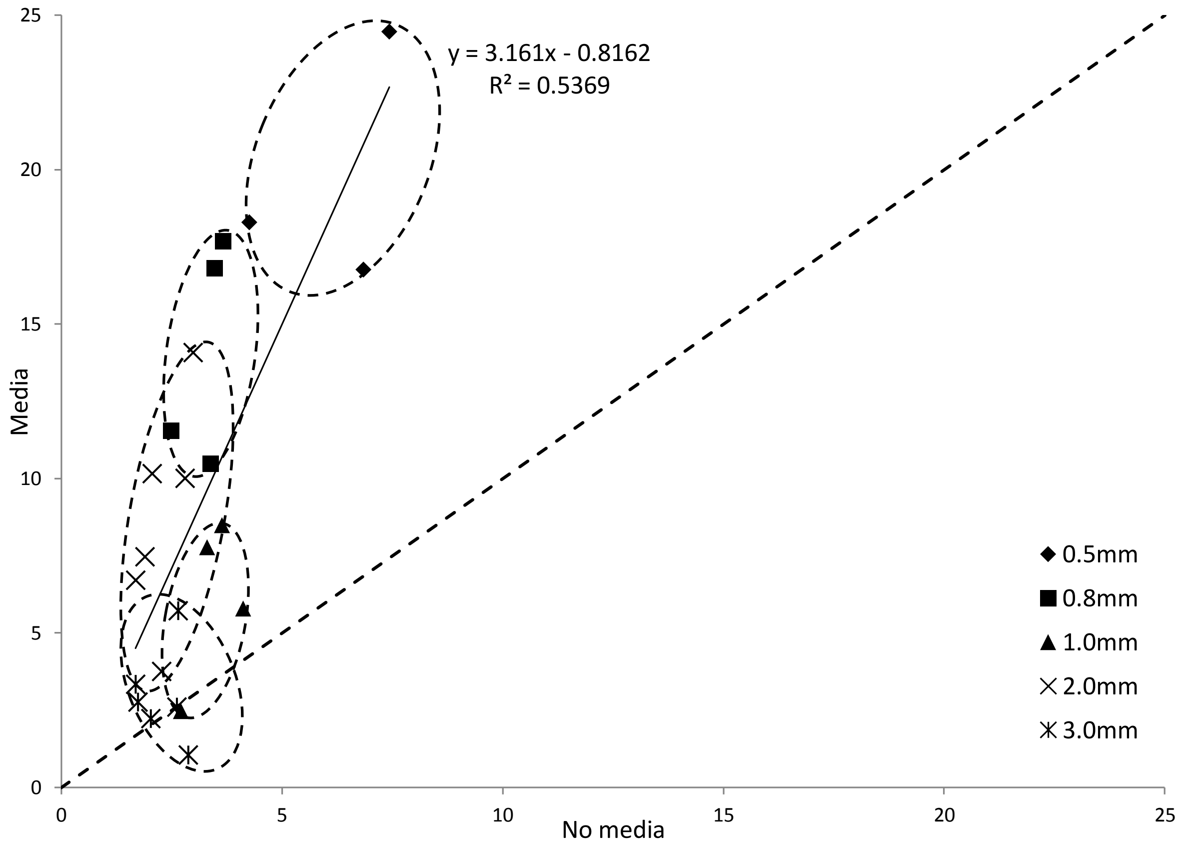

Figure 4.10 Relationship between SAE in media vs. non-media systems as a function of orifice size

Non-media to media ratios were calculated to assess the relationship between bubble specific surface areas, velocity and aeration efficiency. Interestingly, the recorded bubble size distributions when using a 0.5 mm orifice were smaller in the non-media system compared to the media one; with respective ranges of 1.5-8.5 mm and 0.5-11.0 mm) suggesting that the presence of media had reduced the specific surface area for transfer yet provided the greatest enhancement in SAE. This is congruent with the dominant mechanism being related to contact time with the presence of media retarding the rise velocity by up to 2.5 times that of the non-media system. Additionally, gas was seen to reside within the available pore spaces and grow as additional gas entered until it grew to a size when the buoyancy force exceeded the drag forces exerted by the gravel surfaces. Accordingly, the gas hold up rate is considered to be largely due to the size of the voids present in the media system controlling the growth of the bubble. The rate of gas flow will also influence this by reducing the percentage of total gas held pseudo statically within the pore spaces reflected by the highest difference in SAE seen with small orifice sizes and low air flow rates.

The SAE results reflect the balance between the power required to deliver a higher flow rate and the higher flow rate not delivering a favourable increase in OTE. A similar pattern was seen for orifice size i.e. bigger holes require more air and therefore power to deliver and as an increase in OTE is not associated with larger hole sizes it follows that media systems with small orifice sizes and low flow rates will provide the most efficient solution for aerobic treatment on small works in aerated constructed wetlands.

To illustrate the impact of the findings above, the data was used to size blower requirements for an existing operational aerated CW serving a population equivalent of 400 (Chapter 3, Butterworth et al, 2013). The site had an 80th percentile flow of 117 m^3/d and 95th percentiles of 5.1 kg COD/d and 2.5 kgNH_4^+-N/d being removed across the system. The calculations are based on standard approaches (Austin and Nivala, 2009) and assume a respiration load equivalent to that of the COD (to provide conservative estimates of the SAE) and assumed process water conversion factors of α = 0.3-0.7, β = 0.9 and τ = 1.2 (based on an average temperature of 11 °C). The range in α was calculated to assess the potential impact that enhanced contact time may have on mass transfer in real wastewater through additional accumulation of surface active agents. The power required was based on a blower efficiency of 70 % and the OTEs measured in from this study that varied from a minimum of 2.6 % to a maximum of 24.5 % with a median value of 13 %, relating to a 0.6 m depth reactor.

Corresponding blower sizes necessary to deliver the air flow rate requirements to meet the oxygen demand, ranged from 0.13 to 0.31 kW for α factors of 0.7 and 0.3 respectively, based on the median OTE. The minimum blower size, based on the maximum OTE and an α of 0.7, was 0.07 kW. Using a blower appropriately sized for the required transfer (0.3 kW, using median OTE and α of 0.3) translates to a power usage of 0.8 Wh/PE, considerably less than previous quoted values of 2.43 Wh/PE (Pearce, 2013). Assuming favourable conditions, this usage reduces to as low as 0.2-0.3 Wh/PE suggesting that aerated wetlands can be operated with an energy demand around 8 % of a standard activated sludge plant and those currently in operation. To account for uncertainty, realistic prediction of potential reductions are likely to be close to 30 % of the current level, although further reductions are possible, as recent evidence has shown that only the initial third of the bed requires aeration to maintain nitrification performance (Chapter 6).

Overall, the findings demonstrate that the use of aerated wetlands does not have to challenge the preferences for low energy operation at small works. Whilst additional energy is required compared to traditional passive systems, the potential low levels are in line with appropriately sized renewable energy technologies currently available (energysavingtrust, 2012; Vasili, 2013) and ultimately leads to the possibility of off grid operation whilst maintaining robust nitrification performance.

4.6 Conclusions

The findings of this study indicate that there is enhanced mass transfer in packed bed systems compared to non-media systems if the applied air flow rate is sufficiently low. Smaller bubble diameters, lower velocities and increased gas hold up caused by tortuous bubble pathways and retardation in the void spaces increased the residence time of the bubbles increasing the time for oxygen transfer to occur. In terms of overall aeration efficiency, the packed bed systems produced an average enhancement (α) of 3.2 compared to non-media systems. Consequently, results provide the basis for energy reduction on small wastewater treatment plants, introducing the possibility for renewable energy sources to adequately run tertiary nitrification systems such as aerated constructed wetlands enabling the aspiration of small works operators to be secured whilst delivering robustly against challenging ammonia discharge consents.

4.7 Acknowledgements

The authors would like to express their gratitude for the financial and resource support of the work from Severn Trent Water Limited and to the EPSRC for their allocation of funding and support of the project.

4.8 References

American Society of Civil Engineers (ASCE). (1988) Manuals and reports on engineering practise No.68 *Aeration*. WEF Manual of practise FD13. USA

Ashley K., Hall K., Mavinic D. (1991) Factors influencing oxygen transfer in fine pore diffused aeration. *Water Research* 25(12):1479-1486

Ashley K., Mavinic D., Hall K. (1992) Bench-scale study of oxygen transfer in coarse bubble diffused aeration. *Water Research* 26(10):1289-1295

Austin D., Nivala J. (2009) Energy requirements for nitrification and biological removal in engineered wetlands. *Ecological Engineering* 35(2):184-192

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

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

Cornel P., Wagner M., Krause S. (2003) Investigation of oxygen transfer rates in full scale membrane bioreactors. *Water Science and Technology* 47(11): 313-319

Dermou E., Velissariou A., Xenos D., Vayenas D. (2007) Biological removal of hexavalent chromium in trickling filters operating with different filter media types. *Desalination* 211:156-163

Energy Saving Trust. (2012) A buyer's guide to buying solar electricity panels. Available on-line at http://www.energysavingtrust.org.uk/. [Accessed04/03/2014]

Fan J., Wang W., Zhang B., Guo Y., Ngo H-H., Guo W., Zhang J., Wu H. (2013) Nitrogen removal in intermittently aerated vertical flow constructed wetlands: Impact of influent COD/N ratios. *Bioresource Technology* 143:461–466

Fujie K., Kubota H. (1986) Power economy characteristics of an activated sludge aeration tank. *Water Research* 20(8):1051-1058

Fujie K., Hu H-Y., Ikeda Y., Urano K. (1992) Gas-liquid oxygen transfer characteristics in an aerobic submerged biofilter for wastewater treatment. *Chemical Engineering Science* 47(13/14):3745-3752

Harris S., Stephenson T., Pearce P. (1996) Aeration investigation of biological aerated filters using off-gas analysis. *Water Science and Technology* 34(3-4):307-314

Hasanen A., Orivuori P., Aittamaa J. (2006) Measurements of local bubble size distributions from various flexible membrane diffusers. *Chemical Engineering and Processing* 45:291-302

Kember S. (2007) *Evaluation of aeration technology in activated sludge. MTech Thesis*. Cranfield University, Cranfield, UK

Kim J., Park M., Lee J. (2013) *Diffuser for aeration*. US Patent: US 8556238 B2

Lewis W., Whitman W. (1924) Principles of gas absorption. Industrial and *Engineering Chemistry* 16(12):1215-1220

Metcalf & Eddy., Tchobanoglous G., Burton F., Stensel, H. (*2003) Wastewater engineering: treatment and reuse (4th edn),* Metcalf Eddy Inc., McGraw-Hill, New York

Mann A., Stephenson T. (1997) Modelling biological aerated filters for wastewater treatment. *Water Research* 31(10):2443-2448

Mueller J., Boyle W., Pöpel H. (2002) *Aeration: principles and practice*. CRC Press: London

Strenstrom M., Leu S-Y., Jiang P. (2006) *Theory to Practice: Oxygen Transfer and the New ASCE Standard.* WEFTEC

Pearce P. (2013) Sustainable processes to meet low ammonia consents – the small works challenge. Proceedings of the *2nd Conference on low energy wastewater treatment systems*, 24th September. Solihull, UK

Polli M., Stanislao M., Bagatin R., Bakra E., Masi M. (2002) Bubble size distribution in the sparger region of bubble columns. *Chemical Engineering Science* 57(1):197-205

Vasili A. (2013) How Much Electricity Does the Average Solar Panel System Generate? Available online at: http://www.theecoexperts.co.uk/how-much-electricity-does-average-solar-panel-system-generate [Accessed 05/03/2014]

Wagner M., Pöpel H. (1996) Surface active agents and their influence on oxygen transfer *Water Science and Technology* 34(3-4):249–256

Supplementary information 4.1

Example SAE calculation

Nomenclature
C_0= oxygen concentration at beginning of test (ppm)
C^* = oxygen saturation concentration (ppm)
C^*_{20} = oxygen saturation concentration at standard temperature (20°C) = 9.08 (ppm)
DO = Dissolved oxygen (ppm)
kLa = oxygen transfer coefficient (h^{-1})
$kLa_{(20)}$ = standard oxygen transfer coefficient
P_1 = Absolute inlet pressure (bar)
P_2 = Absolute outlet pressure (bar)
P_w = Power (kW)
P_{act} =Actual pressure (bar)
P_{atm} = Atmospheric pressure (bar)
Q_{atm} = Air flow at atmospheric conditions (m^3/h)
SOTE = Standard oxygen transfer efficiency
SOTR = Standard oxygen transfer rate
V = volume of tank (m^3)
V_a = Active volume of the tank (m^3)
WO_2 = mass flow of oxygen in the air stream (kgO_2/h)

Measured values: Temperature 11.9°C; kLa from graph = 0.908h^{-1}; C_0 =0.03ppm, C^*= 10.68ppm

Pre-calculations:

1. Correct kLa to standard temperature (20°C):

   $kLa_{(20)}$= $kLa/1.024^{(T-20)}$ =1.101 h^{-1}

2. Calculate active volume of the tank (V_a):

   With media presence the active volume (V_a) is the porosity x volume of filled tank = 0.4 x 1.35 = 0.54m^3

3. Correct air flow to standard pressure (Q_{atm}):

$Q_{atm} = Q_{act} \times (P_{act} / P_{atm})$

Q_{act} = reading @ flowmeter 10 L/min = 0.60 m^3/h

P_{atm} = Atmospheric pressure = 1.01325 bar

P_{act} =Actual pressure = 0.6m depth + 0.1 as estimate for friction (0.0686 bar) + P_{atm}

Q_{atm} = 0.641 m^3/h

4. Calculate mass flow of oxygen in the air stream (WO_2) = air flow (Q_{atm}) x density of air (*at temperature and humidity at time of test)* x mass ratio of oxygen in air

Density of air @ 11.9C = 1.234 kg/m^3

Mass ratio for O_2 in air = 0.232 kg/kg

WO_2 = 0.641 x 1.234 x 0.232

=0.183 kg/h

SAE Main calculation

SOTR= $kLa_{(20)}$ x $C^*_{(20)}$ x V_a

=1.101 x 9.08 x 0.54

= 5.4004 g/h

= <u>0.0054 kg/h</u>

SOTE = SOTR/WO_2

=0.0054 / 0.183

= 2.9 %

= <u>4.9 %/m</u>

Power

P_w = $wRT_1/29.7.n.e[(P_2/P_1)^{0.283})-1]$

P_w = Power (kW)

w = Weight of flow of air (Kg/s) = Q_{act} (m^3/h) x 1.231 (Kg/m^3) = (0.79 Kg/h) = 0.000219 (Kg/s)

Absolute inlet pressure (P_1) = 0.98692 bar

Absolute outlet pressure (P_2) = 1.05374 bar

Blower efficiency (e) = 0.7

n = 0.238

R = Gas constant for air = 8.314 (kJ/k mol K)

T_1 = Absolute temperature = 285 K

wRT = 0.520

29.7ne = 5.88

wRT/29.7n.e. = 0.884

$(P_2/P_1)^{0.283})$-1 = 0.020

P_w = 0.0018kW

SAE = SOTR/P_w

=0.0054/0.002

=2.99 kg/kWh

Chapter 5

Impact of aeration on macrophyte establishment in sub-surface constructed wetlands used for tertiary treatment of sewage

Impact of aeration on macrophyte establishment in sub-surface constructed wetlands used for tertiary treatment of sewage

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5.1 Abstract

The effect of artificial aeration on plant growth in constructed wetlands in terms of above and below ground biomass and nutrient uptake of two macrophyte species *Phragmites australis* and *Typha latifolia* was carried out to provide quantitative, mechanistic evidence to support any differences between the plant species establishment. Pilot scale systems were built and supplied with different intensities of aeration and corresponding controls with supporting evidence from two full scale operational sites. Results observed a decrease in terms of height and yellowing of leaves in both plant species and a decrease in biomass of the *T. latifolia* compared to the control. Decreased manganese concentrations were recorded for all plant fractions and a decrease in iron observed in the root systems. This was thought to be caused by a neutral to slightly basic pH (7-8) and DO (>2 mgO_2/L) causing these micronutrients to be present in an unavailable form.

Keywords: Aeration, plant development, *Phragmites australis*, *Typha latifolia*

5.2 Introduction

Constructed wetlands (CWs) by definition contain vegetation adapted to saturated conditions (Mitsch and Gosselink, 2000). The specific function of this vegetation in CWs has been much discussed in the literature (Brix, 1997; Stottmeister et al, 2003; Langergraber, 2005) with their importance primarily being attributed to seasonal storage of nutrients; extra surfaces for microbial growth (although this is small in comparison to the surface area of the gravel); insulation in cold and temperate climates, blocking wind and shading out algae that can lower re-aeration; carbon content of plant litter supplying energy for heterotrophic denitrifiers; promotion of wildlife / biodiversity; aesthetics and prevention of unwanted species colonizing the bed (Brix, 1997; Kadlec and Wallace, 2009).

A recent innovation in horizontal sub-surface flow (HSSF) CWs has been the inclusion of forced aeration to promote aerobic conditions and ammonia removal (Ouellet-Plamondon et al, 2006; Zhang et al, 2010). This has proved successful in the US (Kadlec and Wallace, 2009) and testing is producing successful results in the UK (Butterworth et al, 2013) and Europe (Nivala, 2012). However, a negative impact has been observed at some sites in relation to plant growth where academic and commercial groups have reported yellowing of the leaves (chlorosis) and poor establishment of the commonly used plant *Phragmites australis* in artificially aerated and re-circulated effluent vertical-downflow CWs (Weedon, 2014). For instance, during full scale trials of aerated horizontal sub-surface flow (HSSF) CWs (Butterworth et al*,* 2013), poor establishment of *P. australis* was observed in an artificially aerated HSSF CW compared to excellent growth in an identical non-aerated control (Figure 5.1). Additionally, poor establishment, yellowing of leaves and invasion of other species/weeds outcompeting *P. autralis* was observed on two additional full-scale sites (UK) and poor growth/non-establishment of *P. australis* has been reported in the first growing season in the second half of a pilot aerated CW in Germany (Nivala, 2012).

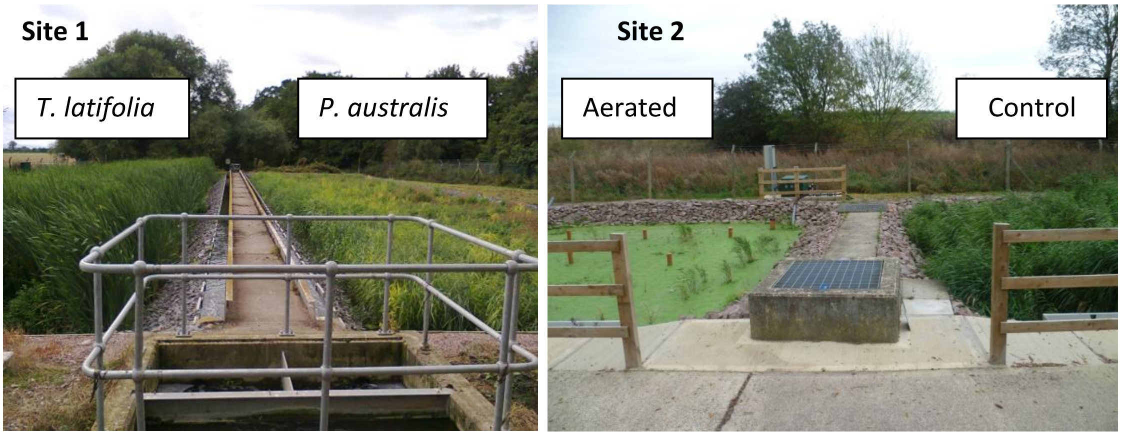


Figure 5.1 Full scale (Site 1) aerated beds planted with *Typha latifolia* (left) and *Phragmites australis* (right) 5 months after planting. Full scale (Site 2) aerated (left) and non-aerated control (right) 12 months after planting

Consequent literature searches revealed a gap in the knowledge with respect to the response of the plants to forced aeration. Discussions with experts within the wetland and botany communities (Dotro et al, pers. comm, 2012) generated two main potential mechanisms that could be responsible: macro or micro nutrient deficiency caused by the aerobic environments leading to changes in redox states that can affect their bio-availability, and turbulence caused by vigorous aeration causing root destabilisation resulting in stress induced growth inhibition and/or water ingress into the aerenchyma preventing effective oxygen transfer to the roots. Additionally, it was discussed that the problem is possibly species specific, as species other than *P. australis* have been able to colonize the aerated environment. It was further noted that the species *T. latifolia* is commonly used in the US due to its competitive and aggressive nature; ability to colonize inert substrates and adapt to diverse and not always optimal climate conditions (Jesperson et al, 1998). Consequently, full scale trials using *T. latifolia* were undertaken by a UK water company (Site 1, Severn Trent Water) in response to the poor establishment of *P. australis* in aerated CWs and following contractor recommendations.

To date, discussion has been largely qualitative such that to the authors' knowledge a paucity of direct experimental evidence exists to help populate the discussion, limiting the potential to define mitigation strategies going forward. Accordingly, the aim of the current investigation was to provide an experimental assessment of the impact of aeration on plant health in terms of above and below ground biomass and nutrient uptake of two macrophyte species *P. australis* and *T. latifolia* to enable the proposed mechanism to be further discussed.

5.3 Materials and Methods

5.3.1 Pilot studies

Individual test microcosms were constructed utilising water butts (height 83cm, width 46cm, depth 45cm - gardens4less.co.uk) fitted with an aeration pipe of 12 mm diameter irrigation pipe with a single 2 mm diameter orifice connected to the mains air supply to imitate conditions found in the aerated bed (Figure 5.2). The columns were filled 0.6 m deep with gravel of 6-12 mm diameter, and planted with a single plug plant (Reeds from Seeds, Denbighshire, UK; 10 cm pot size), as per standard Severn Trent Water reed bed design, and fed with partially treated wastewater (12.1-20.9 mg NH_4^+-N /L and 184-300 mg COD /L for ammonium and chemical oxygen demand respectively) to 0.1 m below the gravel surface. A tap fitted to the bottom of the water butt allowed drainage and re-filling which took place every other day for the duration of the experiment. The composition of the wastewater was tested three times throughout the study for ammonium (NH_4^+-N), nitrate (NO_3^-N) and nitrite (NO_2-N) and COD with Hach-Lange test kits post filling and draining.

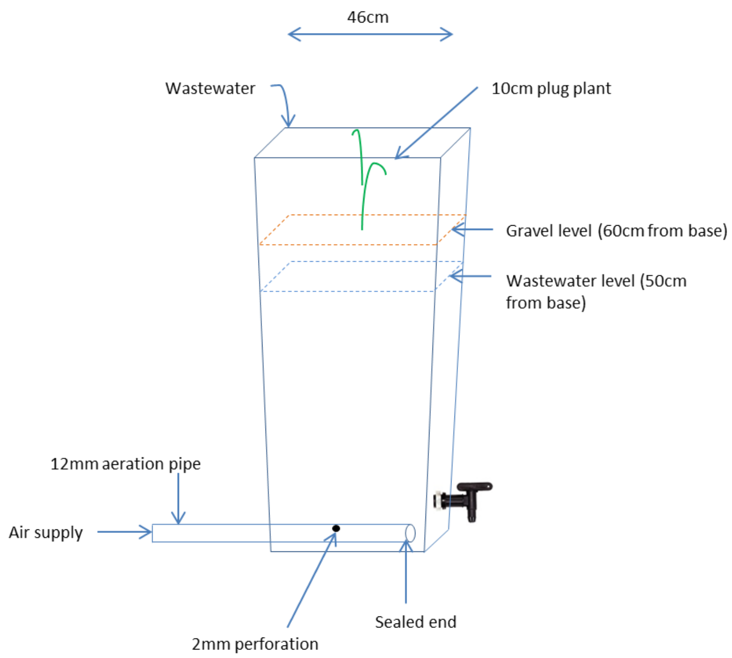


Figure 5.2 Experimental set-up

Two aeration rates were investigated. The first represented the equivalent rate used on an operational site (Site 2, Figure 5.1b) defined as 150 m^3/h of air delivered per 100 m^2 of bed, corresponding to 2.3 L/min per hole [defined as *high*]. The second was based on a separate study that determined the required air delivery rate to maintain aerobic environments as the operational rates above had been observed to exceed those actually required (Chapter 4). The reduced rate was 0.8 L/min per hole [defined as *low*] and thus provided insight into whether optimised aeration could mitigate any observed impacts. Tests were carried out in triplicate and include a non-aerated control for each plant species (Figure 5.3).

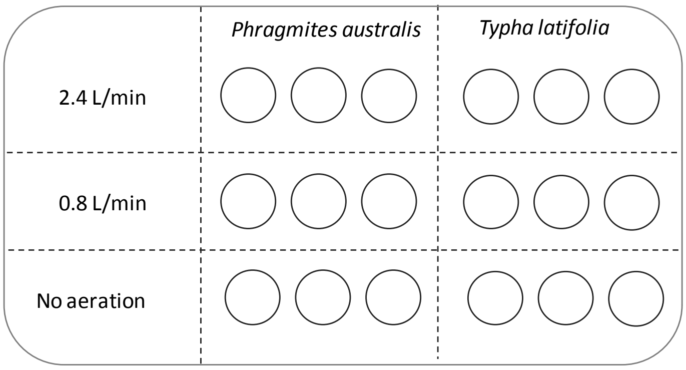


Figure 5.3 Variables tested for each plant species

Direct observations were recorded during the growing period including plant height (measured from gravel level to the tip of the tallest stem); number of stems, leaf length (measured from the base to the tip) and leaf width (measured at the widest part). Harvesting took place 4 months after planting, at the end of the growing season. The plants were removed from the butts and any loose soil washed away. Visual differences were recorded in the roots and rhizomes including the presence of primary and lateral fine side roots, their number, diameter and length. In addition, butts were turned on their side at the end of the experiment to enable direct visual observation of any differences. Plants were then fractionated into roots and rhizomes (below-ground biomass) and above-ground biomass and dried at 70 °C until constant weight and cooled in a dry environment. Primary productivity was determined via measurement of above and below-ground biomass (g/plant). Samples were then ground to 1 mm and dried further for use in nutrient analysis. Total nitrogen (N) was analyzed by gas chromatography after combustion of ground plant material (Vario EL III Elementar Analyzer). Concentrations of phosphorus (P), potassium (K), sodium (Na), calcium (Ca), magnesium (Mg), iron (Fe), manganese (Mn), copper (Cu), zinc (Zn) and nickel (Ni) were determined by spectrometry (atomic absorption for Na, Ca, Mg, Fe and Ni; inductively coupled plasma for K, Mn, Zn and Cu using a Perkin Elmer Analyst 800 and Perkin Elmer ELAN 9000 system respectively, and P using UV/ VIS spectrophotometer Nicolet Evolution 100E) after extraction by a nitric/hydrochloric acid mixture and microwave digestion of 0.25 g of ground plant material. Standards were run prior to running samples and every 10 samples thereafter for all machines, plus two blanks for quality assurance.

5.3.2 Full scale studies

5.3.2.1 Site 1

The site contains two equally sized tertiary HSSF CWs of 56 m x 12.5 m each. Artificial aeration was retrofitted in both beds during gravel cleaning and replacement (refurbishment) and one was planted with *T. latifolia* and one with *P. australis*.

5.3.2.2 Site 2

Tertiary treatment consists of 2 x HSSF CWs 10 m x 10 m each with a separate combined sewer overflow CW. This site serves as a full scale control site whereby both beds were fitted with aeration; one is switched on (3/3/2011) and the other left dormant to provide a control.

For determination of plant coverage, each full scale bed was divided into 9 sections and the total number of plants counted. For determining the percentage of yellowing plants, the proportion of yellow reeds within the sample section was counted and related to the total number of plants. A single plant was excavated from the middle of each bed (October, 2013) by gently easing a fork to loosen the root ball and a shovel to ease it out then taken to the lab and treated as for the pilot studies. Plants from Site 1 were excavated at the end of their first growing season and third growing season in Site 2.

The data did not follow a normal distribution therefore non-parametric statistical tests ($P<0.05$) were conducted to determine differences between datasets with the use of GraphpadPrism.5 (2013). When comparing datasets of greater than two, the Friedman test was used as a non-parametric alternative to analysis of variance (ANOVA). Where two datasets were compared, the Wilcoxon matched pairs test was used as an alternative to the paired student t-test. This test was used over the

Mann-Whitney test that assumes independent samples, after assessing the data for effective pairing by calculation of the spearman's rank correlation coefficient.

5.4 Results and Discussion

5.4.1 Growth

5.4.1.1 Comparison in pilot studies

The relative growth rate in terms of height for both species was slow during the first month before a greater difference between species and the control became apparent, 33 days after planting (Figure 5.4). In terms of overall growth, the *T. latifolia* grew much taller than the *P. australis* with a maximum height observed in the control of 1380 mm compared to 635 mm seen in the *P. australis* control. Plant growth was reduced in all trials and when normalised against the control revealed greater absolute difference in the case of *T. latifolia* compared to *P. australis* (Figure 5.5). To illustrate, the median decrease in height compared to the control at the end of the growth phase was 380±24 mm and 580±195 mm in the case of the high and low aerated *T. latifolia* compared to 240±11 mm and 130±95 mm respectively in the case of *P. australis*. The reduction in height equated to 35-53 % and 15-29 % for the *T. latifolia and P. australis* respectively indicating that despite lower overall height, the height of *P. australis* was less affected by the presence of the aeration than the *T. Latifolia*.

Significance testing of the average relative growth rates reported significant differences of both the low and high aeration cases in comparison to the control for both species (Friedman, matched-pairs signed rank test; Friedman statistic 13.29

for *P.australis* and for *T. latifolia* 26.14, P<0.0001). Further analysis of the impact of aeration intensity on the overall relative growth rates (as height) compared to the control revealed no significant difference between the median values of the low and high aeration for *P.australis* (Wilcoxon matched-pairs signed rank test, W=-27, P=0.4263). However, a significant difference was recorded between the high and low aeration for *T.latifolia* (Wilcoxon matched-pairs signed rank test, W=103, P=0.0002). The results are in contrast to reported observational evidence that has suggested less impact when using *T.latifolia*.

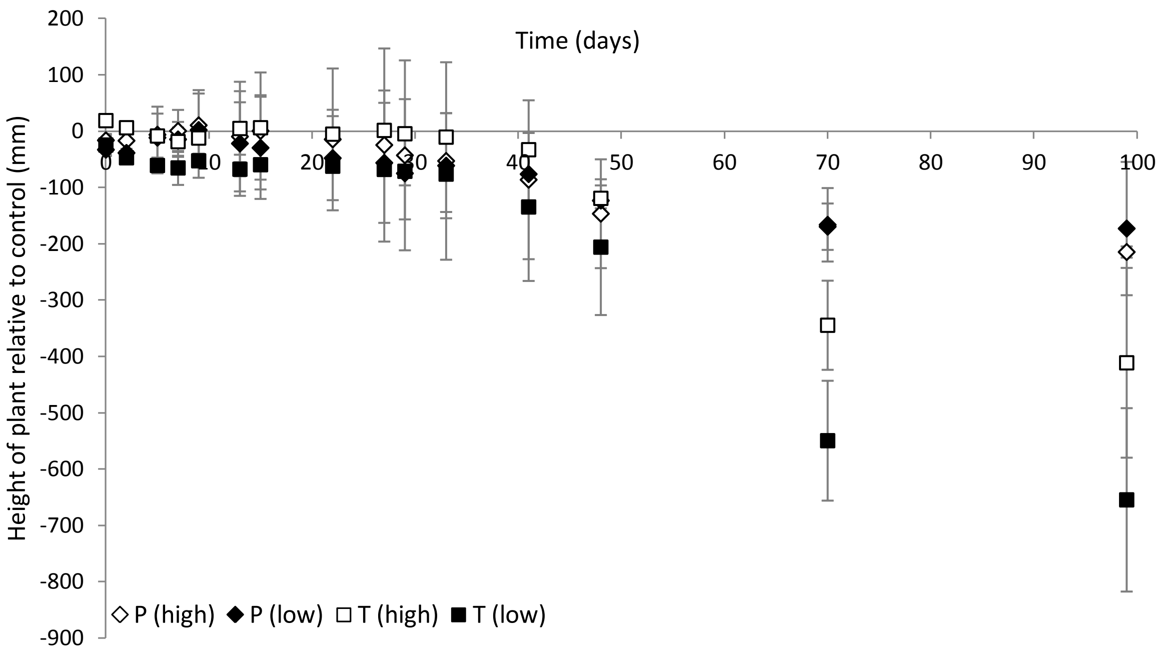
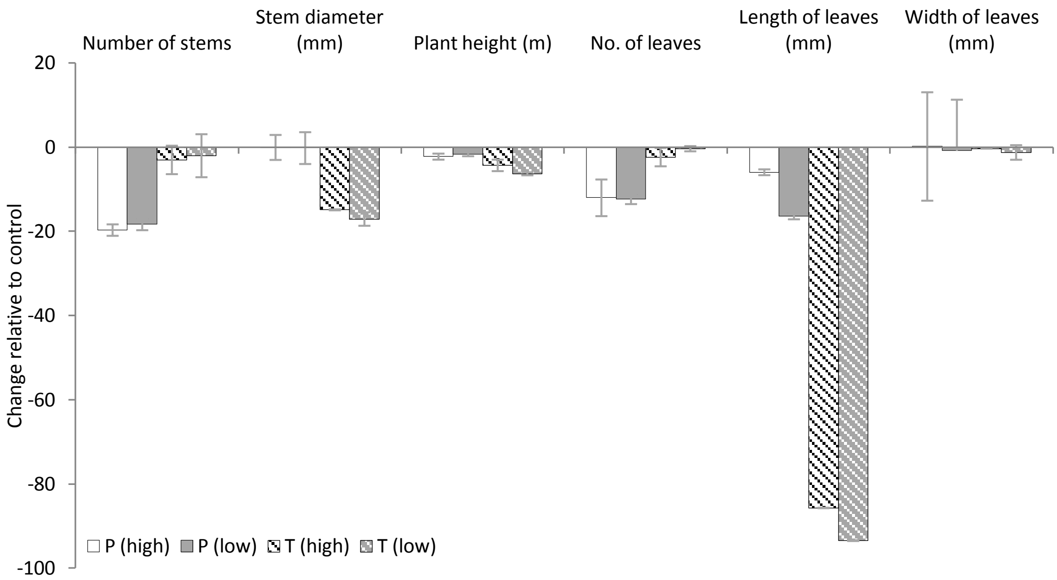


Figure 5.4 Relative plant growth compared to control (P = *P. australis;* T = *T. Latifolia*)

The impact of aeration on the above ground growth occurred predominately in different features for the two macrophyte species (Figure 5.5, top). In the case of *P. australis*, greater impact was seen in the number of stems and leaves. In comparison, the predominant impact for *T. latifolia* was observed in terms of the stem diameter and the length of leaves, with no impact on leaf width in either species. For example, at the end of the growing phase, aeration reduced the number of stems at the end of the growth period compared to the control by 18-19 stems and by 12 leaves in the case *P. australis* (Figure 5.5 top). Whereas the greatest absolute difference was observed in the reduction in leaf length of the *T. latifolia* which was 85-93 mm shorted due to aeration at the end of the growth period. Yellowing of the leaves was observed in all cases and more predominantly in the case of the aerated beds compared to their respective controls. In addition, slight yellowing in the control beds was more pronounced in the *P. australis* than the *T. latifolia*.

The below-ground measurements (Figure 5.5, bottom) indicate differences in growth between both species and between aerated and non-aerated systems, although the intensity of aeration does not appear substantial. The rhizome number, length and diameter and the root length in the aerated *T. latifolia* trials were considerably reduced compared to the control. Whereas the number of rhizomes observed in *P. australis* was also fewer than the control, although to a lesser extent than *T. latifolia*. For example, the number of rhizomes was reduced by an average of 10 in both the high (±3) and low (±5) systems in *T. latifolia,* compared to 2±1 (low) and 3±1 (high) in the *P. australis*. On first appearance, in the case of *P. australis*, the rhizome length and diameter and the root length is also reduced compared to the control but the large standard deviation recorded in these measurements does not allow any level of certainty. The root diameter does not seem to have been affected by the presence or intensity of aeration in either species and little effect is observed in the rhizome diameter of *P. australis*. When the standard deviations are taken into account no one below ground growth parameter appears to have been affected considerably more than another.



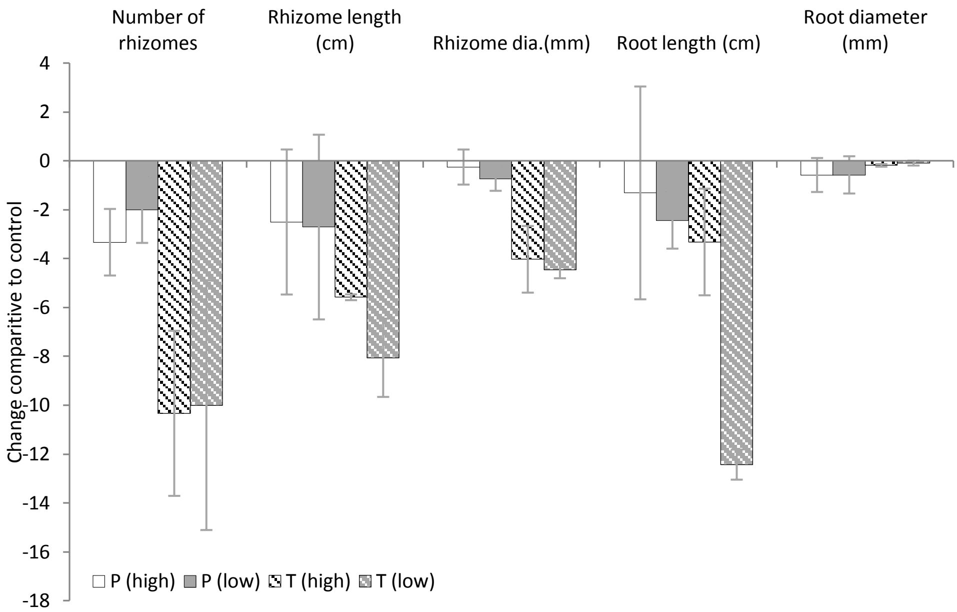


Figure 5.5 Difference in above ground biomass characteristics after 4 months (top). Difference in below ground biomass characteristics after 4 months relative to control (bottom) (P = *P. australis;* T = *T. Latifolia*)

Visual observation of *P. australis* showed an obvious difference in the size of the above ground biomass and a denser root structure with deeper reaching roots and increased number fine roots in the control (Figure 5.6). The *T. latifolia* control too displayed deep reaching roots with a couple of roots reaching the bottom of the water butt (0.6 m deep). Investigation of root integrity revealed no discernible difference in the nature of the roots between aerated and non-aerated samples. The reduced root intensity and size is indicative of stress derived growth inhibition due to root disturbance as seen in many terrestrial plants exposed to waves (Coops and Van der Velde, 1996). However, the lack of difference between aeration intensity suggests that aeration level is not a factor and that the minimum aeration levels required in aerated wetlands will exceed any potential threshold level.

Figure 5.6 Example of root systems in-situ (enhanced): *P. australis* control (left); *P. australis* high aeration (right)

5.4.1.2 Comparison on full scale beds

Both species grew well at the inlet of the beds of Site 1 (Figure 5.1) in their first growing season and were largely green, upright and healthy. *T. latifolia* averaged a height of 2.2 m±0.4 m, compared to the *P. australis* which averaged 1.4±0.3m, giving a *P. australis* to *T. latifolia* ratio of 1:1.57. The height ratio was similar in the pilot studies at 1:1.55 indicating commonality between the different scales of systems investigated. The plants gradually thinned in density, were shorter and showed signs of yellowing towards the outlet of both beds and invasive species had begun to take over. The effect was much more pronounced in the *P. australis* bed with approximately 40 % of the *P. australis* plants displaying significant yellowing as close as 0.3 m from the inlet and approximately half of the bed was taken over with weeds. In comparison, signs of stress began 3-4 m from the inlet in the *T. latifolia* bed, yellowing was apparent in approximately 20 % of the plants and invasive species took over approximately 25 % of the bed. The *P. australis* taken for analysis from site 1 displayed slight yellowing of the leaves, the plant was taken from the middle of the bed where plants were beginning to deteriorate it terms of size and colour. The *T. latifolia* also displayed slight yellowing on the outer leaves but to a visually different degree than the *P. australis*. The *P. australis* from the aerated bed at site 2 had yellowing leaves whilst the control was largely green. *T. latifolia* displayed thick (1.3±0.6 cm), shallow, long rhizomes (8.1±3.2cm) compared to the root system of *P. australis* that recorded an average of 0.6±0.3 cm

– half the diameter found in *T. latifolia*; a comparatively more dense root network, with a greater number of fine roots. Observational differences in above ground growth in *T. latifolia* and *P. australis* (Site 1) indicate *T. latifolia* was able to establish quickly in the aerated site and out-compete the invasive species more so than *P. australis*.

Site 2 allowed the comparison of the growth of *P. australis* at full scale with and without aeration. At the time of writing, the plants were coming to the end of their third growing season. This site in particular demonstrated a stark contrast between plant growth and establishment in an aerated vs. a non-aerated control (Figure 5.7). The control bed displayed excellent above ground growth (Figure 5.7a), covering 100 % of the bed and averaging a height of 1.6 m ± 0.3 m compared to a coverage of approximately 20 % of the bed and an average height of 1.1 m ± 0.5 m (Figure 5.7b). The control to aerated height ratio was 1:1.45, similar to the pilot which recorded 1:1.41 for the *P. australis*.

A visually apparent difference was also seen between the below ground growth of *P. australis* in the aerated bed (Figure 5.7b, d) compared to the control (Figure 5.7a, c). The quantitative differences were similar to those seen in the pilot, in that the control plants were comparatively larger than in the aerated bed, illustrated by 31 % taller plants in the control bed compared to 15-29 % in the pilot and 66 % more stems, comparable to 63-67 % measured in the pilot. The leaf number indicted 73 % more leaves on the control plants, more than the 37-38 % increase observed in the pilot, most likely related to the height of the stems being taller in the full scale bed after 3 growing seasons compared to the shorter stems in pilot study after only one growing season.

Figure 5.7 Above and below ground growth in a full scale aerated and non-aerated site a. Site 2 control, above ground; b. Site 2 aerated, above ground; c. Site 2 control, below ground; d. Site 2 aerated, below ground

5.4.2 Biomass allocation

Compared to their respective controls, there was a much greater decrease in plant biomass (g/plant less than the control) in the *T. latifolia* than the *P. australis* and the effect of low and high aeration did not appear to affect the distribution of the biomass between above ground, roots and rhizomes (Figure 5.8).

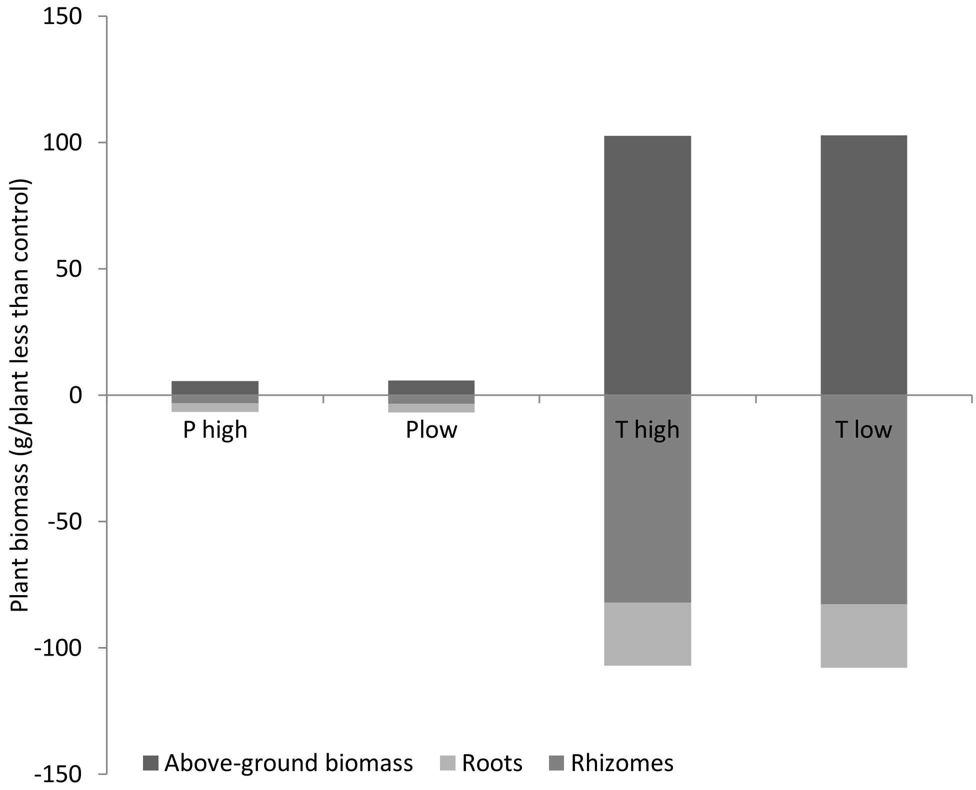


Figure 5.8 Plant biomass (g/plant compared to control). Minus numbers indicates below-ground biomass (P = *P. australis;* T = *T. latifolia*)

The ratio of below ground to above ground biomass (BGB:AGB) in the *P. australis* was 1:1.3-1.7 for all test conditions; comparable to reported literature values of 1.3 for standard HSSF conditions (Tanner, 1996) and 1.4-1.8 for various intensified CW systems (Nivala, 2012). The aerated tests recorded a mean of 29 % of the BG biomass allocated to the roots and 45 % in the control; higher than previously reported values of 29 % under non-aerated conditions (Tanner, 1996); suggesting more effort was put into rhizome growth in the aerated bed compared to the roots in the control. In all *T. Latifolia* test conditions the ratio of BGB:AGB biomass was 1.3, comparable to the *P. australis*. Allocation of biomass was similar for above ground biomass, roots and rhizomes in both the aerated tests and the non-aerated tests with 39-43 %, 12-13 % and 42-48 % respectively, indicating the presence of aeration did not affect biomass distribution (Figure 5.9). Of the BG biomass, 21 % was attributed to the roots in the aerated condition and 23 % in the control. This was less than observed in the *P. australis* and there was less difference between the aerated and control.

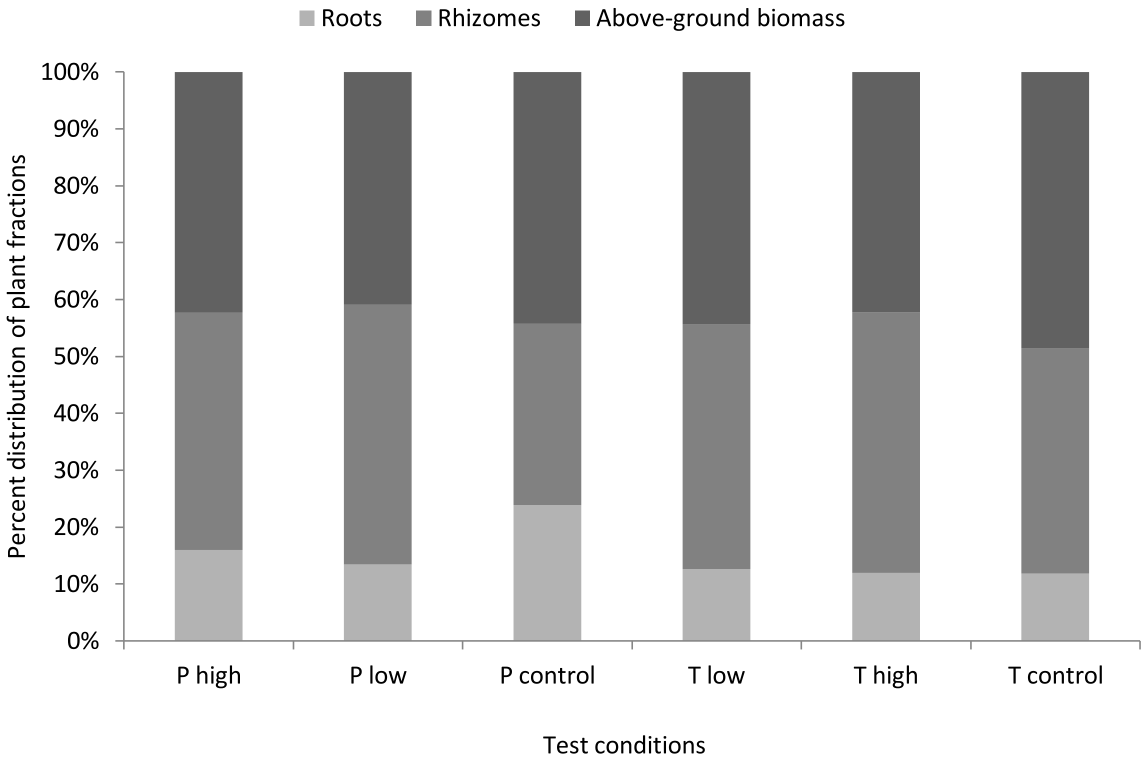


Figure 5.9 Percentage distribution of biomass in plant fractions in all test conditions (P = *P. australis; T* = T. *latifolia*)

In terms of plant size and biomass, the presence of aeration appeared to be detrimental to the growth of both species in the first growing season and to a greater degree in the *T. latifolia* compared to the *P. australis*. The data suggest aeration is affecting the growth of each species by different mechanisms as the most apparent differences were seen in terms of a reduction of the length of the leaves, stem diameter and number of rhizomes in *T. latifolia* compared to a reduction in the number of stems, the stem diameter and the number of leaves in *P. australis*.

5.4.3 Macro and micro nutrients/trace minerals

Macro and micro nutrient concentrations in the above ground biomass, rhizomes and roots revealed no consistent discernible differences between the non-aerated and aerated plants (Table A, supplementary information 5.1). In most cases, the concentrations observed in the control plants were within the same range as the plants exposed to low and high levels of aeration or were in between the two. Accordingly, no direct causal link could be identified between the observed growth differences due to aeration and changes in nutrient levels. To illustrate, in the case of copper in *P. australis*, the AGB contained a median level of 11.3 μgCu/g (7.9-14.6) in the non-aerated bed compared to 13.9 μgCu/g (9.3-18.3) in the aerated bed with the high level of aeration. Across all nutrients, similarity between the non-aerated and aerated samples was more apparent in the case of *P. australis,* with indicative differences (defined as the ranges not substantially overlapping) only identified in relation to calcium levels in the roots and rhizomes (reduced from control), manganese levels in the roots and AGB (reduced from the control) and an increase in sodium in the AGB. In contrast, indicative differences were observed with *T. latifolia* in relation to nitrogen, magnesium, calcium, zinc and copper in the AGB and manganese in the roots and rhizomes. The reported levels in the current study are within the ranges seen previously for reeds grown in a range of water types including municipal sewage (Nivala et al, 2012; Moises, 2012), salt marshes (Windham et al, 2003), dairy wastewater (Tanner, 1996), coal ash basin (Babcock et al, 1983) but lower than those from plants used for treatment of sewage sludge (Obarska et al, 2002). Although N concentrations were not found to be lacking in any of the tested species it is interesting to note that the nitrogen measured in the *T. latifolia* control was present at lower concentrations than in the aerated tests in all plant fractions e.g. 9.9 mgN/g (9.4-10.6) in the AGB of the control compared to 19.9 mgN/g (17.7-21.7) and 20.5 mgN/g (19.5-10.6) in the high and low aerated systems respectively (Table A, supplementary information 5.1); indicating the presence of aeration may be affecting N uptake rate, requiring further investigation to first repeat the trend and second to look into possible affecting mechanisms

Both species have been reported to function on an excluder strategy with regard to metal translocation (Windham et al, 2003) such that the majority of metals can be expected to reside in the roots in order to protect photosynthetic function in the leaves (MacFarlane and Burchett, 2000). The data from the current trial broadly supports this, with the exception of nickel that was generally more evenly distributed between the AGB and the roots. To illustrate, AGB:root ratio for nickel ranged between 1.03 and 1.54 indicating higher levels in the AGB in all cases. In contrast, the equivalent ratio for copper ranged between 0.43 for the *P. australis* non-aerated to 0.57 for the *T. latifolia* aerated with an equivalent range for zinc of 0.44-1.0. Similar measurements, but based on the ratio of shoots to roots indicates much greater partitioning with maximum levels of ~0.15 when examining *P. australis* grown in salt marshes (Windham et al, 2003) and 0.21 in lake water

(Schierop and Larsen, 1981). Whereas shoot to rhizome ratios of 1:1 for copper and zinc have previously been reported for *T. latifolia* treating coal ash basin wastewater (Babcock et al, 1983). This compares to ratios for *T. latifolia* in the current study of 0.7 and 1.9 for copper and 0.23 and 0.9 for zinc for the non-aerated and aerated plants respectively. Overall, analysis of the data revealed changes in partitioning between the plants grown in non-aerated and aerated conditions, and between species, but not in a consistent enough manner to discern an identifiable probable cause.

Previous discussions on the impact of aeration on nutrient uptake have focussed on iron deficiency as the potential cause of the stunted growth and chlorosis of leaves in aerated CWs (Weedon, 2014). This is consistent with the aerobic environments and neutral to slight alkaline pH leading to the majority of the iron existing in a precipitate iron hydroxide form such that truly dissolved levels in the water are likely to range between 0.1 and 0.5 μgFe/L based on iron equilibrium chemistry. In the current trials, average iron levels in the AGB in the case of *P. australis* were 0.5 mgFe/gDM (range: 0.1-1.2) in the control compared to 0.2 mg/gDM (range: 0.1-0.3) for the low aeration level and 0.4 mgFe/gDM (range 0.2-0.5) in the case of the high aeration level (Figure 5.10, top). Similar patterns were observed in terms of the roots and rhizomes, offering some support for the proposed mechanism. In contrast, in the full scale site with the test and control beds, iron levels were higher in the AGB in the aerated bed at 0.6 mgFe/gDM compared to 0.2 mgFe/gDM in the control bed (Figure 5.10, top). This compares to 0.45±0.058 mgFe/gDM for AGB from a non-aerated wetland treating secondary effluent at a municipal sewage works (Moises, 2012) suggesting no deficiency in the AGB, where the iron is utilised as part of the photosynthetic function. The reverse was observed in the rhizome and roots where substantially more iron was present in the plants in the control bed at 4.5 and 6.7 mgFe/gDM in the roots and rhizome compared to 1.0 and 0.6 mgFe/gDM respectively in the plants grown in the aerated bed. Aggregate iron content in the roots from plants used in a variety of different wetland configurations (including aerated beds) have been reported at 0.84-3.5 mgFe/gDM (Nivala et al, 2012). Further, in a study concerning the impact of elevated iron levels in water, mean root iron contents of 5.5±0.8 mgFe/gDM were observed once the water contained 0.5 mg/L of iron with a range of 0.2-2.2 mgFe/gDM when iron concentrations were 0.1 mgFe/L (Batty and Younger, 2003). Iron levels during the current trials were around 0.1-0.5 mgFe/L indicating greater iron accumulation in the current trial than previously reported.

Manganese was also reduced in the AGB for both the pilot trials and the full scale sites sampled in the aerated compared to the non-aerated beds. To illustrate, in the aerated beds planted with *P. australis*, manganese levels for the AGB, ranged between 27.7-138 μgMn/gDM compared to 120-196 μgMn/gDM in the case of the non-aerated samples. Required nutrient concentrations for healthy plant growth are very species specific but a general estimate of the required levels of Fe and Mn are 0.1 mgFe/gDM and 50 μgMn/gDM respectively (Barker and Pilbeam, 2007) indicating that the aeration was more likely to have influenced available manganese levels than iron.

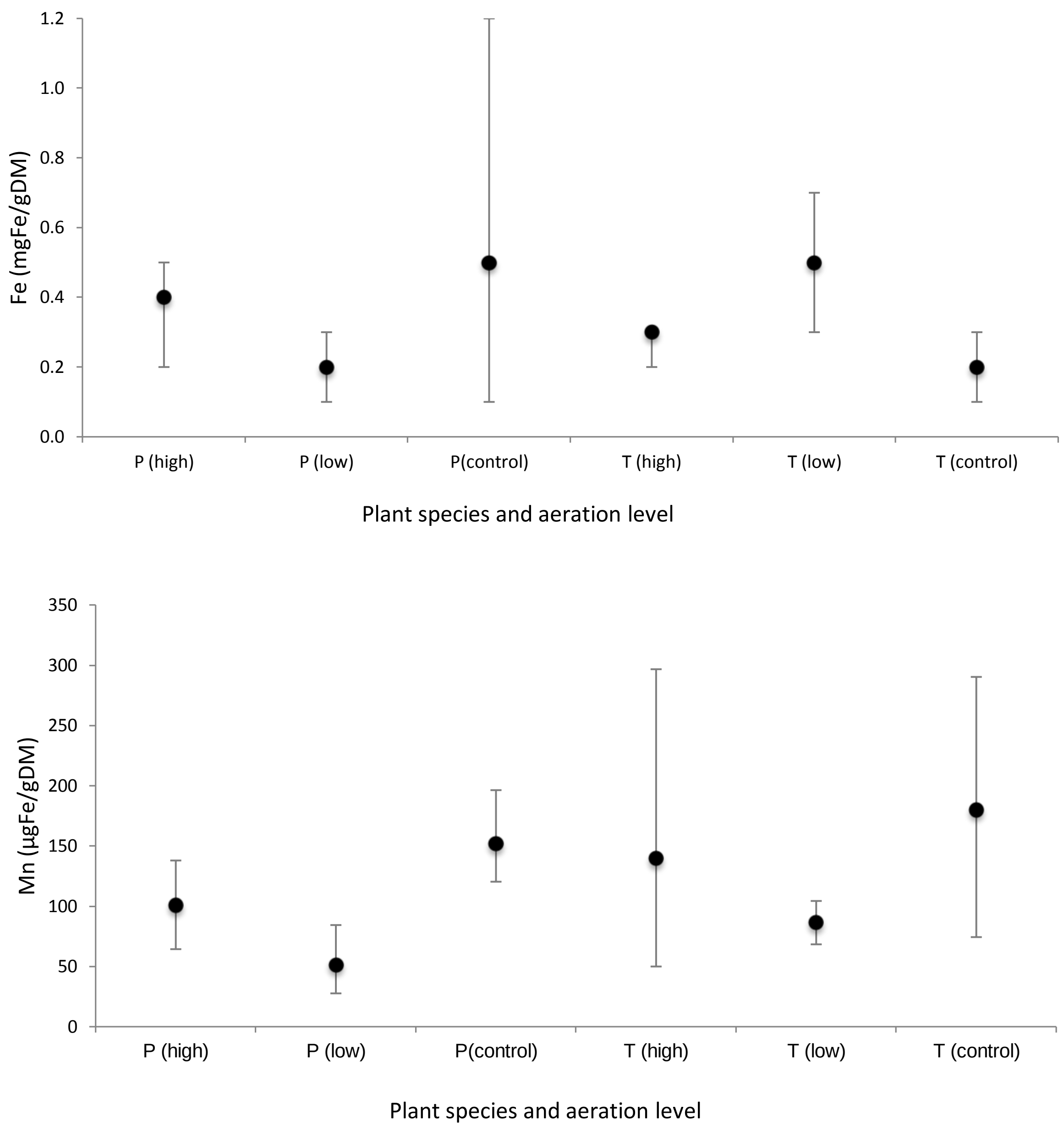


Figure 5.10 Iron (top) and Manganese (bottom) concentrations in the AGB of plants from the pilot study at the end of the growing season (P = *P. australis*; T = *T. Latifolia)*

Manganese, like iron is essential for chlorophyll production and its deficiency is often confused with, and occurs concurrently with iron deficiency, both displaying similar symptoms of interveinal chlorosis of the leaves. Mn^{2+} is the primary form in which Mn is absorbed by plants and is most available at pH 5-6.5 (Hong et al, 2010). Conditions that restrict availability are similar to those for iron based around neutral or slightly alkaline pH, aerobic conditions and the presence of relatively higher levels of iron, copper or zinc, consistent with aspects of the environments found within the aerated beds.

5.5 Conclusions

Overall, the results confirm reports of the negative impacts of aeration on plant growth in terms of visual differences in stunted growth and yellowing of leaves. The effects appear to be species specific, with aeration impacting more in terms of fresh growth in the case of *P. australis* (number of stems and leaves) and more with respect to the size of stems and leaves in the case of *T. Latifolia*. Whilst on an individual plant basis, *T. Latifolia* was more strongly influenced by aeration than *P. australis* compared to their respective controls; the impact on the overall growth was less significant; congruent with its faster growing rate. Consequently, *T. Latifolia* is recommended where rapid plant cover is required in an aerated wetland.

Metal content analyses and visual observations of the plants from both the pilot trials and full scale beds were inconclusive in providing a definitive mechanism for observed retarded growth. However, a potential synergistic impact is suggested in relation to both iron and manganese.

5.6 Acknowledgments

The authors would like to express their gratitude for the financial and resource support of the work from Severn Trent Water Limited and to the EPSRC for their allocation of funding and support of the project.

5.7 References

Babcock M., Evans D., Alberts J. (1983) Comparative uptake and translocation of trace elements from coal ash by *Typha latifolia*. *Science of the Total Environment* 28(1-3):203-214

Barker A., Pilbeam D. (2007). *Handbook of plant nutrition*. Taylor and Francis. New York

Batty L., Younger P (2003) Effects of External Iron Concentration upon Seedling Growth and Uptake of Fe and Phosphate by the Common Reed, *Phragmites australis* (Cav.) Trin ex. Steudel. *Annals of Botany* 92(6):801-806

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

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

Coops H., Van der Velde (1996) Effects of waves on helophyte stands: mechanical characteristics of stems of *Phragmites australis* and *Scirpus lacustris*. Aquatic Botany 53(3-4):175-185

Dotro G., Brix H., Wallace S., Nivala J., Comeau Y., Stein O., Rosseau D., Tyrell S., Gowing S., ARM Ltd. (2012) Personal communication

Hong E., Ketterings Q., McBride M. (2010) *Agronomy Fact Sheet 49: Manganese*. Available on-line at: http://nmsp.cals.cornell.edu/publications/factsheets/factsheet49.pdf [Accessed 3/3/2014]

Jesperson D., Sorrell B., Brix H. (1998) Growth and root oxygen release by *Typha latifolia* and its effects on sediment methanogenesis. *Aquatic Botany* 61(3):165-180

Kadlec R., Wallace S. (2009*) Treatment Wetlands (2nd edn)*. Taylor & Francis Group:Florida

Langergraber G. (2005) The role of plant uptake on the removal of organic matter and nutrients in subsurface flow constructed wetlands: a simulation study. *Water Science and Technology* 51(9):213-233

MacFarlane G., Burchett M. (2000) Cellular distribution of copper, lead and zinc in the grey mangrove Avicennia marina (Forsk.) Vierh. *Aquatic Botany* 68(1):45-59

Mitsch W., Gosselink J. (200) *Wetlands (3rd edn).* Van Nostrand Reinhold: New York

Moises P-M. (2012) Evaluating the application of harvested reeds in anaerobic digestion. MSc Thesis: Cranfield University

Nivala J. (2012) Effect of design on treatment performance, plant nutrition and clogging in subsurface flow treatment wetlands. PhD Thesis: Aarus University

Obarska-Pempkowiak H., Tuszynska A., Sobocinski Z. (2002) Polish experience with sewage sludge dewatering in reed systems. *Water Science and Technology* 48(5):111-117

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

Schierup H., Larsen V. (1981) Macrophyte cycling of zinc, copper, lead and calcium in the littoral zone of a polluted and non-polluted lake I. Availability, uptake and translocation of heavy metals in *Phragmites australis* (Cav.) Trin. *Aquatic Botany* 11(3):197-210

Stottmeister U., Wießner A., Kuschk P., Kappelmeyer U., Kästner M., Bederski O., Müller R., Moormann H. (2003) Effects of plants and microorganisms in constructed wetlands for wastewater treatment. *Biological Advances* 22(1-2):93-117

Tanner C. (1996) Plants for constructed wetland treatment systems- A comparison of the growth and nutrient uptake of eight emergent species. *Ecological Engineering* 7(1):59-83

Weedon C. (2014) Yellow *Phragmites*: Significance, Cause, and Remedies. *Sustainable sanitation practice* 18:37-42

Windham L., Weis J., Weis P. (2003) Uptake and distribution of metals in two dominant salt marsh macrophytes, *Spartina alterniflora* (cordgrass) and *Phragmites australis* (common reed). *Estuarine, Coastal and Shelf Science* 56(1):63-72

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

Supplementary information 5.1

Table A Nutrient concentrations in plant fractions

		P (high)		P (low)		P (control)		T (high)		T (low)		T (control)	
		Mean	Min-Max	Mean	Min-Max	Mean	Min-Max	Mean	Min-Max	Mean	Min-Max	Mean	Min-Max
AGB	N (mg/g DM)	22.7	17.1-26.0	25.5	19.7-31.4	24.5	21.7-26.5	19.9	17.7-21.7	20.5	19.5-21.9	9.9	9.4-10.6
	P (mg/g DM)	2.1	1.9-2.3	1.8	1.1-2.1	1.9	1.3-2.0	2.2	2.1-2.3	1.8	1.6-2.2	1.5	1.1-2.2
	K (mg/g DM)	15.4	11.1-20.5	7.0	3.9-9.1	15.4	11.9-18.5	19.4	15.4-22.9	8.5	15.7-21.4	25.4	13.5-34.9
	Mg (mg/g DM)	2.2	1.9-2.5	1.5	1.4-1.6	2.4	2.2-2.7	3.0	2.4-3.3	2.2	6.3-11.3	1.4	1.0-1.9
	Ca (mg/g DM)	5.6	4.0-6.6	4.3	3.4-6.2	4.2	3.8-4.7	25.6	15.7-40.1	17.8	15.7-21.4	8.0	1.6-13.1
	Na (mg/g DM	3.9	3.3-5.0	2.4	1.0-3.6	1.3	0.9-1.7	6.6	5.5-7.4	4.1	2.6-5.7	2.9	1.0-4.7
	Zn (µg/g DM)	82.0	43.2-125.2	47.4	29.4-72.0	98.8	66.8-160.0	116.4	60.4-177.2	91.1	49.6-149.2	26.5	22.6-31.4
	Cu (µg/g DM)	13.9	9.3-18.3	10.1	7.1-16.2	11.3	7.9-14.6	13.1	12.5-13.9	17.4	11.9-22.4	6.1	4.9-6.8
	Ni (µg/g DM)	18.3	5.0-44.4	26.3	4.6-67.2	17.7	4.0-42.4	8.5	3.6-13.6	7.9	3.7-15.6	9.3	6.6-10.8
	Mn (µg/g DM)	100.9	64.4-138.0	51.6	27.7-84.4	151.9	120.4-196.4	140.0	50.0-296.8	86.8	68.4-104.4	179.9	74.4-290.4
	Fe (mg/g DM)	0.4	0.2-0.5	0.2	0.1-0.3	0.5	0.1-1.2	0.3	0.2-0.3	0.5	0.3-0.7	0.2	0.1-0.3
Rhizomes	N (mg/g DM)	20.1	16.7-23.5	23.6	22.6-25.4	24.9	22.1-30.4	21.9	18.2-26.3	26.2	22.9-31.7	15.2	13.4-18.6
	P (mg/g DM)	1.8	1.7-2.0	1.7	1.5-2.0	2.3	2.0-2.6	1.8	1.1-2.4	1.7	0.7-2.3	2.2	2.0-2.5
	K (mg/g DM)	7.4	5.3-9.6	6.4	4.7-8.5	12.4	7.0-15.7	15.6	13.3-19.1	12.4	9.5-14.3	16.4	16.0-17.5
	Mg (mg/g DM)	0.7	0.6-0.8	0.8	0.6-1.2	1.2	0.6-2.1	1.5	0.9-2.1	2.0	1.7-2.4	1.3	1.1-1.6
	Ca (mg/g DM)	0.8	0.7-0.9	1.5	0.8-3.1	3.8	0.7-9.5	4.2	0.6-8.0	4.8	2.6-6.7	2.1	1.4-2.4
	Na (mg/g DM	0.9	0.6-1.2	2.3	0.9-5.0	2.4	0.7-4.7	4.5	1.1-8.4	3.8	3.5-4.1	2.8	1.6-4.2
	Zn (µg/g DM)	35.1	31.6-38.6	45.3	38.3-49.6	43.3	30.6-55.2	37.6	25.2-61.2	31.2	30.2-32.2	34.0	23.1-52.0
	Cu (µg/g DM)	8.4	7.3-9.6	12.1	10.1-15.5	10.3	8.8-12.6	6.8	5.4-8.0	10.1	5.2-16.6	8.0	6.5-10.6
	Ni (µg/g DM)	3.6	2.5-4.7	5.4	2.7-8.1	5.9	3.5-8.8	12.6	1.8-20.1	8.6	3.7-14.0	9.1	6.4-12.2

	Mn (µg/g DM)	27.8	26.8-28.8	35.9	32.8-38.6	47.2	20.7-72.8	14.5	13.6-15.4	24.8	9.1-47.2	36.8	18.0-61.2
	Fe (mg/g DM)	0.3	0.2-0.3	0.1	0.1-0.2	0.3	0.1-0.5	0.1	0.1-0.1	0.3	0.0-0.5	0.1	0.1-0.2
Roots	N (mg/g DM)	17.3	13.7-20.8	14.3	-	17.1	8.3-22.8	12.7	12.4-12.9	12.1	11.1-13.7	10.2	9.4-11.1
	P (mg/g DM)	0.8	0.7-1.0	1.5	-	0.9	1.6-1.2	0.9	0.8-1.4	0.8	0.7-0.8	1.6	1.6
	K (mg/g DM)	7.3	3.1-11.5	14.1	-	6.8	2.3-12.5	13.3	11.2-16.0	6.1	4.3-.2	10.3	7.0-13.7
	Mg (mg/g DM)	1.9	1.8-2.1	1.5	-	1.6	1.2-2.0	2.0	1.5-2.3	2.3	1.7-3.0	1.4	1.2-1.6
	Ca (mg/g DM)	5.7	3.0-8.4	2.7	-	9.4	6.0-15.0	24.3	20.5-30.7	11.1	7.6-12.9	14.0	4.9-23.1
	Na (mg/g DM	1.4	1.2-1.6	1.2	-	1.5	0.9-2.3	6.4	5.0-8.8	4.8	4.1-6.0	5.0	1.3-8.6
	Zn (µg/g DM)	184.4	84.0-284.8	75.6	-	97.1	72.0-130.4	165.6	156.8-174.4	127.6	52.8-232.8	52.4	38.0-66.8
	Cu (µg/g DM)	26.7	19.8-33.6	26.0	-	26.5	20.4-30.6	22.9	21.2-25.7	20.6	16.6-28.6	12.7	10.3-15.1
	Ni (µg/g DM)	11.9	3.6-20.2	13.0	-	16.3	8.6-27.9	7.2	4.8-10.6	7.3	6.3-7.8	9.0	6.1-12.0
	Mn (µg/g DM)	148.0	80.4-215.6	134.4	-	212.8	198.8-233.6	150.4	117.6-185.2	74.5	48.4-120.8	284.4	-
	Fe (mg/g DM)	1.6	0.4-2.7	0.6	-	1.9	1.2-2.6	1.5	1.3-1.7	0.8	0.3-1.3	2.2	0.5-3.9

		Site 1 (Typha)	Site 1 (Phrag)	Site 2 (aerated)	Site 2 (control)
AGB	N (mg/g DM)	15.6	22.2	20.5	15.3
	P (mg/g DM)	2.4	1.1	1.3	-
	K (mg/g DM)	17.2	11.0	7.1	20.3
	Mg (mg/g DM)	1.2	1.3	1.3	1.4
	Ca (mg/g DM)	2.2	3.0	0.2	10.8
	Na (mg/g DM	2.3	0.9	1.0	7.6
	Zn (µg/g DM)	50.0	35.0	80.8	26.7
	Cu (µg/g DM)	19.2	11.8	13.1	5.1
	Ni (µg/g DM)	9.5	3.3	11.8	10.8
	Mn (µg/g DM)	170.4	97.2	45.6	299.6
	Fe (mg/g DM)	0.3	0.1	0.6	0.2
Rhizomes	N (mg/g DM)	24.9	24.3	25.1	19.2
	P (mg/g DM)	3.1	2.1	1.5	1.5
	K (mg/g DM)	18.4	19.4	9.5	10.0

	Mg (mg/g DM)	1.5	2.3	0.9	2.5
	Ca (mg/g DM)	3.7	2.9	2.5	0.3
	Na (mg/g DM	1.5	7.2	0.4	2.1
	Zn (µg/g DM)	79.6	84.4	130.0	144.0
	Cu (µg/g DM)	51.6	87.6	27.9	53.6
	Ni (µg/g DM)	13.2	15.0	10.7	19.8
	Mn (µg/g DM)	157.6	130.4	97.2	69.6
	Fe (mg/g DM)	0.9	0.1	0.6	6.7
Roots	N (mg/g DM)	15.6	22.3	22.5	17.4
	P (mg/g DM)	3.3	3.2	1.5	2.5
	K (mg/g DM)	13.8	11.0	10.9	10.9
	Mg (mg/g DM)	3.5	1.2	1.8	2.8
	Ca (mg/g DM)	14.0	0.8	2.9	3.6
	Na (mg/g DM	1.5	1.5	0.9	2.4
	Zn (µg/g DM)	198.0	161.6	240.8	448.0
	Cu (µg/g DM)	259.6	290.8	45.6	139.2
	Ni (µg/g DM)	20.3	14.3	48.4	27.7
	Mn (µg/g DM)	760.0	385.6	157.6	140.0
	Fe (mg/g DM)	5.0	6.8	1.0	4.5

Chapter 6

Ammonia removal at four full-scale artificially aerated horizontal flow constructed wetlands

Ammonia removal at four full-scale artificially aerated horizontal flow constructed wetlands

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6.1 Abstract

A comparison of the performance of four full scale aerated constructed wetland systems was conducted to determine the efficacy of the technology on sites receiving high and variable ammonia loading rates not yet reported in the literature; building and expanding on an initial one year study conducted at a fully aerated tertiary site with a non-aerated control (Chapter 3, Butterworth et al, 2013). Performance was assessed in terms of ammonia and solids removal, hydraulic conductivity and mixing patterns. The capability of systems to produce ammonium effluent concentrations < 3 mgNH_4^+-N/L was observed across all sites in systems receiving variable loadings between 0.1-13.0 gNH_4^+-N/m^2/d. Potential resilience issues were observed in relation to response to spike loadings posited to be due to the development of an insufficient abundant nitrifying population within the beds. Hydraulic conductivity and mixing pattern data suggested deterioration

over time, refuting initial patterns of the improvement between the aerated and non-aerated control in the initial study.

Keywords: Aeration, constructed wetlands, dissolved oxygen, nitrification

6.2 Introduction

Horizontal sub-surface flow constructed wetlands (HSSF CWs) are passive wastewater treatment systems commonly used for tertiary suspended solids, associated organic matter and nitrate removal. The relatively large footprint associated with the technology (0.7-5 m^2/p.e., Kadlec and Wallace 2009) focuses applications towards rural small works (sub 2000 population equivalence) which typically consist of aerobic biofilm processes such as trickling filters or rotating biological contactors upfront of the wetland to deliver against the required treatment goals. In response to the challenges associated with achieving “good ecological status” in European rivers as part of the Water Framework Directive, 2000, a large number of these rural domestic wastewater treatment plants are requiring upgrading in order to discharge lower levels of organic matter and ammonia into the receiving waters. To achieve this on such sites the flowsheet is upgraded by replacement / enhancement of the secondary biological processes or inclusion of additional aerobic biological processes such as submerged aerated filters (Ishikawa et al, 2003).

Sub-surface flow wetlands can also be considered to be a form of biofilm-based bioreactors and hence functionally have the capability to meet the future needs. However, traditional HSSF beds are often hydraulically loaded at rates that cause anoxic/anaerobic conditions to predominate with the associated sub-surface oxygen limitation (sub 0.5 mg/L) restricting observed nitrification rates in full scale tertiary HSSF CWs to 0.05-0.22 gNH_4^+-$N/m^2/d$ (Chapter 2); generally insufficient to meet future discharge targets.

Across the full spectrum of applications for constructed wetlands a number of innovative adaptations have been developed to overcome the oxygen limitation including decreasing the depth of the bed (Garcia et al, 2005); recirculation of the treated effluent (Gross et al, 2008); batch loading including traditional vertical flow systems (Molle et al, 2006) and adaption to horizontal flow beds, i.e. tidal flow (Wu et al, 2011); forced (artificial) aeration of traditional vertical flow systems (Foladori et al, 2013), flooded vertical flow systems (Nivala et al, 2013), or horizontal flow beds (Butterworth et al, 2013; Maltais-Landry et al, 2009). In the context of tertiary treatment, many sites already include HSSF CWs and so the last option offers additional favourable attributes as it can be easily retrofitted into existing systems during routine gravel cleaning and replacement (refurbishment).

The general efficacy of the approach towards enhancing nitrification has been demonstrated predominately at pilot scale for a range of feed water types including synthetic wastewater (Fan et al, 2013; Matthys et al, 2000), heavily polluted river water (Dong et al, 2012), and municipal secondary sewage (Nivala et al, 2013). Illustrations of the efficacy of the technology have also been presented at full scale in terms of the treatment of landfill leachate (Nivala et al, 2007), industrial wastewater (Wallace and Kadlec, 2005), secondary domestic wastewater and tertiary treatment of municipal sewage (Chapter 3, Butterworth et al, 2013). In the later case hydraulic loading rates are considerably higher than those utilised elsewhere with typical rates of 0.2-0.9 m/d reported in the UK (Knowles et al, 2011) compared to 0.001-0.049 m/d for the sites treating industrial wastewaters (Wallace and Kadlec, 2005). Ammonia concentrations feeding the tertiary beds can reach as high as 40 mgNH_4^+-N/L on sites that previously had no requirement to remove ammonia and as such ammonia loading rates can significantly exceed those previously considered. To illustrate, non tertiary cases have reported loading rates in the range 0.9 gNH_4^+-N/m^2/d to 6.9 gNH_4^+-N/m^2/d (Nivala et al, 2007, 2013) which compares to a maximum rate of 10.1 mgNH_4^+-N/m^2/d observed during an assessment of the initial stages of operation of a full scale tertiary aerated wetland (Chapter 3, Butterworth et al, 2013). The same study demonstrated near complete nitrification (99 % removal) without negatively impacting on solids and $CBOD_5$ removal efficiencies. Aeration also appeared to increase hydraulic conductivity, and improve hydraulic efficiency in the aerated bed under sub-surface flow, resulting in a more efficient reactor (Chapter 3, Butterworth et al, 2013).

A paucity of information remains around full scale tertiary systems treating variable flows and loads including storm flows and the inherent seasonal variations in wastewater characteristics and temperatures. Coupled to the short duration of the majority of previous investigation at both pilot and full scale the appropriateness of the technology has yet to proven sufficiently to enhance confidence, particularly in terms of the performance consistency across various applications of the technology.

This study aims to respond to this knowledge gap through assessment of four full-scale wetlands retrofitted with artificial aeration in small domestic wastewater treatment works in the UK. The current study extends beyond the initial study carried out on a fully aerated tertiary system with separate storm over-flow during its first year post commissioning (Chapter 3, Butterworth et al, 2013), to assess suitability of the artificial aeration across various situations including intermittent aeration, tertiary combined storm flow and a secondary bed. Performance was assessed in terms of ammonia and solids removal, hydraulic conductivity and mixing patterns and in terms of process robustness to determine whether the use of aerated HSSF CWs is viable as a reliable tertiary process for ammonium removal on small sites.

6.3 Materials and Methods

6.3.1 Site details

Aeration systems were retrofitted into existing HSSF CWs during refurbishment on three Severn Trent Water sites with tightening ammonium consents (Site A, B, and C) and one secondary treatment system with a descriptive consent that required upgrading for improved response with regards to odour emissions on the site (Site

D; Table 1). In all cases, the aeration system consists of an air blower (1.6 kW, model 2RB510H26), a distribution header and loops of perforated LDPE 12 mm piping with 2 mm holes drilled into it at 300 mm intervals placed on top of the impermeable liner covering the surface area of the bed floor with an approximate spacing of 300 mm. The beds contained 0.6 m of 6-12 mm gravel media giving a measured porosity of 0.4, and are planted with *Phragmites australis* at 4 seedlings/m^2 with the exception of Site C which was planted with *Typha latifolia*.

Table 6.1 Site details and current and future consents for aerated HSSF CWs

Site	Treatment	PE	Area m^2	Pre-treatment	Aeration on	Power w/m^3	HRT *d*	Previous consents *DWF*	*BOD*	*TSS*	NH_4-N	Future consents *DWF*	*BOD*	*TSS*	NH_4-N
A	Tertiary	393	100	Integral RBC	03/03/2011	26.7	2.0	61	30	45	n/a	80	21	n/a	4
B	Combined	396	390	Integral RBC	25/11/2010	6.3	1.5	90.2	25	45	10	110	14	45	3
C	Combined	599	600	2 x RBC	26/01/2012	6.1	1.1	135	30	50	n/a	166	30	50	12
D	Secondary	58	200	Septic tank	04/05/2011	18.3	1.9	5.8	D	D	D	5.8	D	D	D

PE = population equivalent; DWF = Dry weather flow (m^3/d); BOD = 5-Day Carbonaceous Biochemical Oxygen Demand (mgO$_2$/L); TSS =Total Suspended Solids (mgTSS/L); NH_4^+-N = ammonium nitrogen (mg NH_4^+-N/L); D = descriptive consent; Power based on the blower sized used per m^3 of bed.

Treatment at Site A consists of a primary storage tank (PST) followed by a semi-integral RBC. Tertiary treatment is via two HSSF CWs (100 m^2 each – approximately 0.5 m^2/p.e.) with a separate combined sewer overflow (CSO) HSSF CW that receives the wastewater exceeding six times the dry weather flow (Figure 6.1). Site A serves as a control site with side by side wetlands of equal size. Aeration of the test bed began 03/03/2011 and was left dormant in the control bed. The beds at this site were dormant for 5 months (December 2011-April 2012) due to mechanical maintenance of the RBC during which time the flow was tankered off-site. Temporary secondary treatment was installed in the form of a

SAF and flow was returned to the reed beds. The mean flow to each bed is 46 m^3/d, resulting in mean hydraulic loadings of 0.5 $m^3/m^2/d$. Inlet loadings to the bed during the trial were 9.1±7.6 gNH_4^+-$N/m^2/d$, 11.9±5.1 $gCBOD_5/m^2/d$ and 25.0±19 $gTSS/m^2/d$.

Treatment at Site B is via an integral RBC followed by a combined wetland (360 m^2) (Figure 6.1). The bed was retrofitted with aeration in October 2010 to provide a failsafe for occasional ammonia peaks observed in the conventional flow sheet. Since its retrofit, the bed has been operating with intermittent aeration, with the blowers only being active between 8 am – 8 pm. The mean flow to the bed is 45 m^3/d, resulting in mean hydraulic loadings of 0.1 $m^3/m^2/d$. Inlet loadings to the bed during the trial were 1.6±8.6 gNH_4^+-$N/m^2/d$, 10.6±2.6 $gCBOD_5/m^2/d$ and 1.3±8.9 $gTSS/m^2/d$.

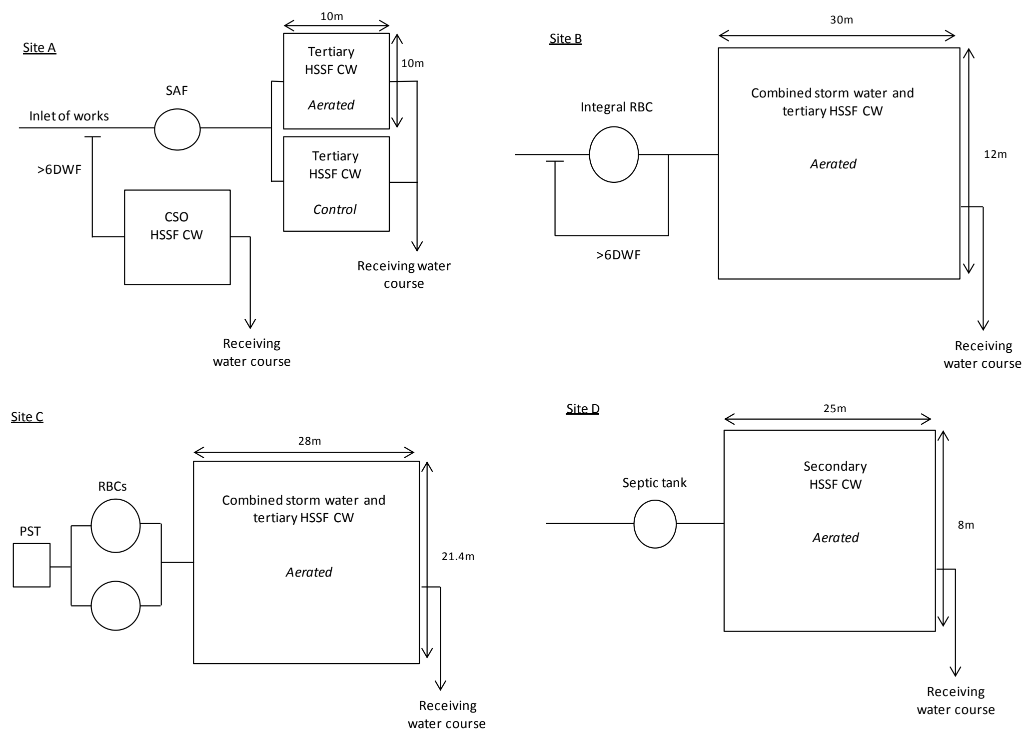

Figure 6.1 Site process flow sheets of aerated HSSF CW sites

Site C consists of two integral RBCs and a combined tertiary reed bed (Figure 6.1). The aeration was activated on 26/01/2012. The driver for aeration at this site was the addition of an ammonia effluent consent at a site that was not originally designed to nitrify. The mean flow to the bed is 248 m^3/d, resulting in mean hydraulic loadings of 0.4 $m^3/m^2/d$. Inlet loadings to the bed during the trial were 8.9±4.5 gNH_4^+-$N/m^2/d$, 7.6±2.1 $gCBOD_5/m^2/d$ and 16.7±19.0 $gTSS/m^2/d$.

Site D is the only secondary bed in the trial (Figure 6.1). Treatment consists of a septic tank followed by a combined reed bed (200 m^2). Consents here are descriptive, as set out by the Environment Agency, and are based on visual inspection of the effluent into the watercourse. The bed was refurbished in March 2010 and fitted with aeration on 30/03/2011. The mean flow to the bed is 50 m^3/d, resulting in mean hydraulic loadings of 0.3 $m^3/m^2/d$. Inlet loadings to the bed during the trial were 29.4±14.7 gNH_4^+-$N/m^2/d$, 79.4±4.3 $gCBOD_5/m^2/d$ and 57.0±32.6 $gTSS/m^2/d$.

6.3.2 Sampling and analysis

Composite samples (every 15 minutes over 24 hours) were collected fortnightly during the first year and monthly thereafter using ISCO auto-samplers at the inlet and outlets of the beds. Samples were collected in 1 L plastic sampling bottles and stored in a cool box with ice blocks during transition to the laboratory for same-day testing. Where same-day testing was not feasible samples were stored at 4 °C and allowed to reach room temperature prior to analysis. Sampling was conducted from the onset of aeration at respective sites to February 2013.

Hach-Lange test kits were employed for determining NH_4^+-N according to Hach-Lange procedures and read via a Hach-Lange DR2800 spectrophotometer. Portable meters were used to determine dissolved oxygen and temperature (Hach-Lange HQ40D). Probes were checked weekly with hydrogen sulphite solution and Hach standards were used periodically to ensure accuracy of test kits. Total suspended solids were quantified following standard procedures using a three piece filter funnel with 70 mm filter diameter and 1.2 µm pore size (APHA, 2005).

6.3.3 Tracer Tests

Tracer tests were carried out as in Chapter 3, (Butterworth et al, 2013) and involved the addition of a 0.135 g impulse of 20 % Rhodamine Water Tracer Liquid (Keystone Europe Ltd.) at the inlet points of both wetlands. Concentrations were monitored at the outlet over time using a fluorescence spectrophotometer (YSI 6 series sonde fitted with Rhodamine sensor). Comparative tracer studies were only carried out at Site A.

6.3.4 Hydraulic conductivity

In-situ saturated hydraulic conductivity measurements were taken using a steel pipe perforated at the base and a model 3001 Solinst levelogger; following the falling head methodology as described by Pedescoll et al, (2009). Measurements were taken in triplicate along three transect points spanning the length of each of the wetlands. This measurement gives approximate saturated hydraulic conductivity values, as vertical conductivity is measured, and does not take into account the horizontal flow. In addition, a certain degree of compaction occurs when inserting the pipe into the bed presents a source of error but these errors have been evaluated and are considered acceptable (Pedescoll et al, 2009).

6.3.5 Robustness analysis

Robustness curves were generated by plotting the percentile distribution against the effluent values. In this context, robustness is described as the ability of a treatment unit to produce consistent effluent quality under varying influent characteristics and differentiates from resilience which is defined as the ability to return to normal after a dynamic event (Hartshorn et al, 2014).

A robustness index (RI) was calculated with respect to overall ammonium removal performance against a treatment goal (T_{goal}) and the percentage time spent under the treatment goal (Equation 1, Hartshorn et al, 2014). As the goal term heavily influences the outcome, RI was calculated for a range of treatment goals from 0.1-5 $mgNH_4^+$-N/L. A lower robustness score indicates a more robust process.

$$RI = \left[\left(1 - \frac{G\%}{100}\right) * \frac{T_{90}}{T_{50}}\right] + \left[\frac{T_{50}}{T_{goal}} * \frac{G\%}{100}\right] \quad (1)$$

RI = Robustness index

G% = percentage time spent under T_{goal}

T_{90} = 90th percentile value ($mgNH_4^+$-N/L)

T_{50} = 50th percentile value ($mgNH_4^+$-N/L)

T_{goal} = treatment goal ($mgNH_4^+$-N/L)

In accordance with Hartshorn et al, (2014) the approach was used to assess overall robustness rather than directly assess the ability of the sites to meet regulatory standards as RI is more effectively used as an indicator of variation rather than directly about meeting a fixed goal. Accordingly the 90th percentile was used to avoid confusion with regulatory compliance at the 95th percentile.

Statistical analyses were carried out using Graphpad Prism v.5. Data was tested for normality by calculating the t-statistic that describes the number of sample standard deviations that the sample was above or below the sample mean. In a normal distribution 68 % of values will lie within one standard deviation, 95 % within two deviations and 99 % within three. The data did not follow a normal distribution and consequently statistical significance between datasets was tested using the non-parametric Mann-Whitney unpaired, two-tailed test ($P<0.05$) as an alternative to the paired students t-test. Where multiple comparisons were required the Kruskal-Wallis test was used ($P<0.05$) as an alternative to the one-way analysis of variance (ANOVA).

6.4 Results and Discussion

6.4.1 Ammonium removal

Low ammonium effluent (< 3 mgNH_4^+-N/L) was achieved in all aerated sites up to the maximum measured ammonia loading rate of 13.0 gNH_4^+-N/m^2/d (Figure 6.2). The efficacy of nitrification was consistent across the full range of variable loading rates observed indicating no deterioration in effluent quality at higher loadings. Consequently, associated ammonium removal rates in the aerated beds increased linearly with increased loadings from 0.05 to 13.0 gNH_4^+-N/m^2/d demonstrating a strong causal relationship suggesting ammonia loadings were not a rate limiting factor. In comparison, a non-linear range of ammonium removal between 0.04-7.0 gNH_4^+-N/m^2/d was observed in the non-aerated wetland at Site A (Figure A, supplementary information 6.1). The current findings extend the reported maximum loading rates for effective nitrification in aerated wetlands (Nivala et al, 2007, Zhang et al, 2010, Fan et al, 2013, Matthys et al, 2001, Nivala et al, 2013) even extending above the 10.1 gNH_4^+-N/m^2/d previously reported during the initial months of the trial (Chapter 3, Butterworth et al, 2013). In comparison, the control bed (Site A) exceeded 3 $mgNH_4^+$-N/L once the loading rate rose above 1.9 gNH_4^+-N/m^2/d; consistent with previous findings concerning sub-surface oxygen deficiency limiting nitrification as loading increased due to the commensurate increase in organic loading resulting in the heterotrophic community out-competing the nitrifying microbial community (Chapter 3, Butterworth et al, 2013). The median mass removal was 23 % in the control bed across all loading rates tested but increased to 72 % below a loading rate of 1.9 gNH_4^+-N/m^2/d suggesting adequate removal can be achieved in HSSF CWs at sufficiently low loading rates.

Comparison of the data across the different sites revealed median effluent ammonia concentrations ($mgNH_4^+$-N/L±standard deviation) of 0.1±0.3 (Site A), 0.2±3.0 (Site B), 0.2±0.2 (Site B not including upstream failure), 0.2±1.2 (Site C) and 0.6±0.8 (Site D) (Figure 6.3) equating to median mass removal rates of 98.8 %, 93.9 %, 94.7% 94.7 % and 97.5 % for the sites respectively. Overall, median outlet concentrations of 0.1-0.2 $mgNH_4^+$-N/L were recorded for the tertiary aerated beds from inlet concentrations of 1.6-9.1 $mgNH_4^+$-N/L and an effluent of 0.6 $mgNH_4^+$-N/L for the secondary bed from an inlet of 29.6 $mgNH_4^+$-N/L. This compares to a median effluent concentration of 6.7 $mgNH_4^+$-N/L in the non-aerated control.

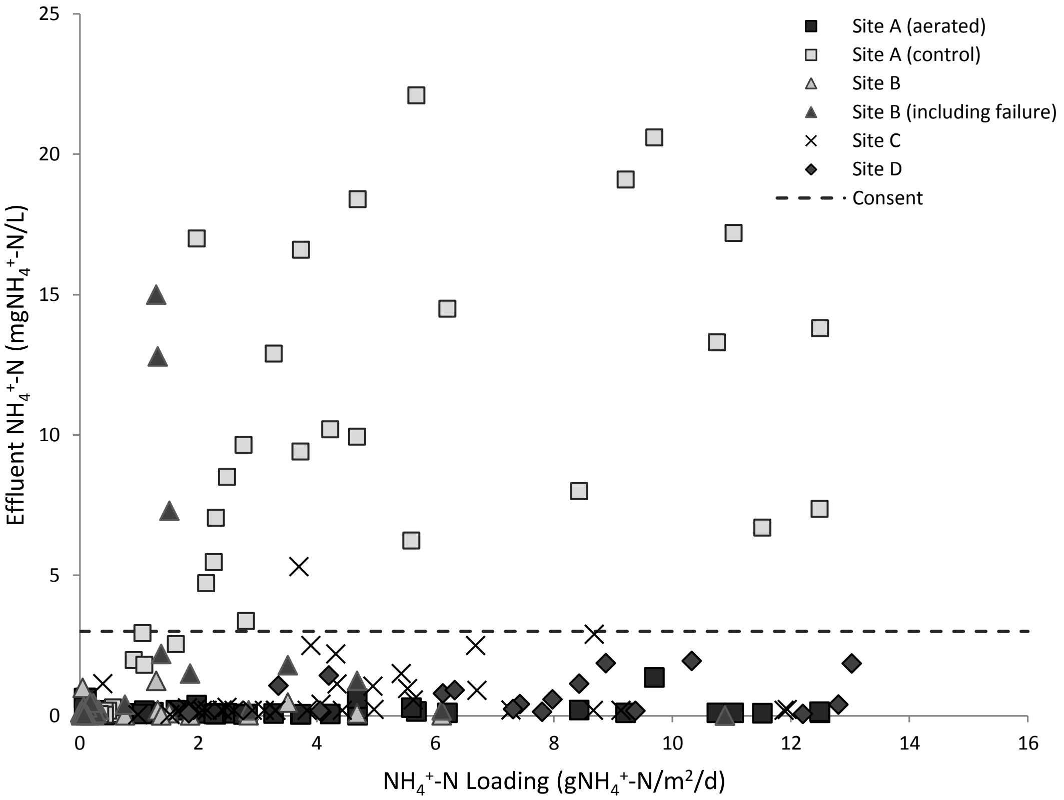

Figure 6.2 Ammonium loading rates and corresponding effluent ammonium concentrations at all sites

For context, typical literature values report influent concentrations of 22-54 $mgNH_4^+$-N/L for tertiary HSSF CWs, corresponding to effluents of 5-31 $mgNH_4^+$-N/L (Llorens, 2009; O'Luanaigh, 2010; Albuquerque et al 2009; Yousefi and Mohseni-Bandpei, 2010). Whereas reported secondary systems achieve concentrations of 3-61 $mgNH_4^+$-N/L from inlets of 15-225 $mgNH_4^+$-N/L

(Matamoros, 2008; Cottingham et al, 1999; Vymazal, 2011; Vymazal, 2005; O'Luanaigh, 2010).

More detailed profiling at Site A revealed NH_4^+-N concentrations of 0.8 , 0.7 and 0.2 mgNH_4^+-N/L with corresponding DOs of 2-3, 3-4 and >5 mgO_2/L at 0.7, 1.2 and 10 m from the inlet of the bed, with the final value corresponding to the final effluent from the bed compared to an influent level of 9.1±7.6 mgNH_4^+-N/L. Accordingly, only an initial fraction of the bed appeared to be utilised for nitrification, indicating that the majority the bed was unnecessarily aerated supporting previous laboratory investigations that limited aeration to the front portion of the bed to allow a reduced DO environment conducive of denitrification thereafter (Maltais-Landry et al, 2009).

Greater variability in the effluent ammonia concentrations was observed in Sites C and D compared to A and B. Site C (tertiary combined storm flow, continuously aerated) showed the greatest variation in effluent concentration with effluent ammonium concentrations over 1.0 mgNH_4^+-N /L recorded on 11 occasions out of 43 (25 %). This compares to 2 % and 0 % for the other tertiary sites A and B (excluding upstream failure) suggesting the observed instability was not directly due to the inclusion of storm flows, increased loading rates or inadequate level of aeration. Diagnostic analysis of the bed confirmed no link to aeration as DO profiles ranged between 3.3 and 9.5 mgO_2/L across the length and breadth of the bed. Further diagnostics concerning alkalinity revealed no definitive explanation and further investigation is required to establish a coherent diagnosis. However, the results show that process instability in aerated wetlands is possible, congruent with other tertiary nitrification systems (Boller and Gujer, 1985).

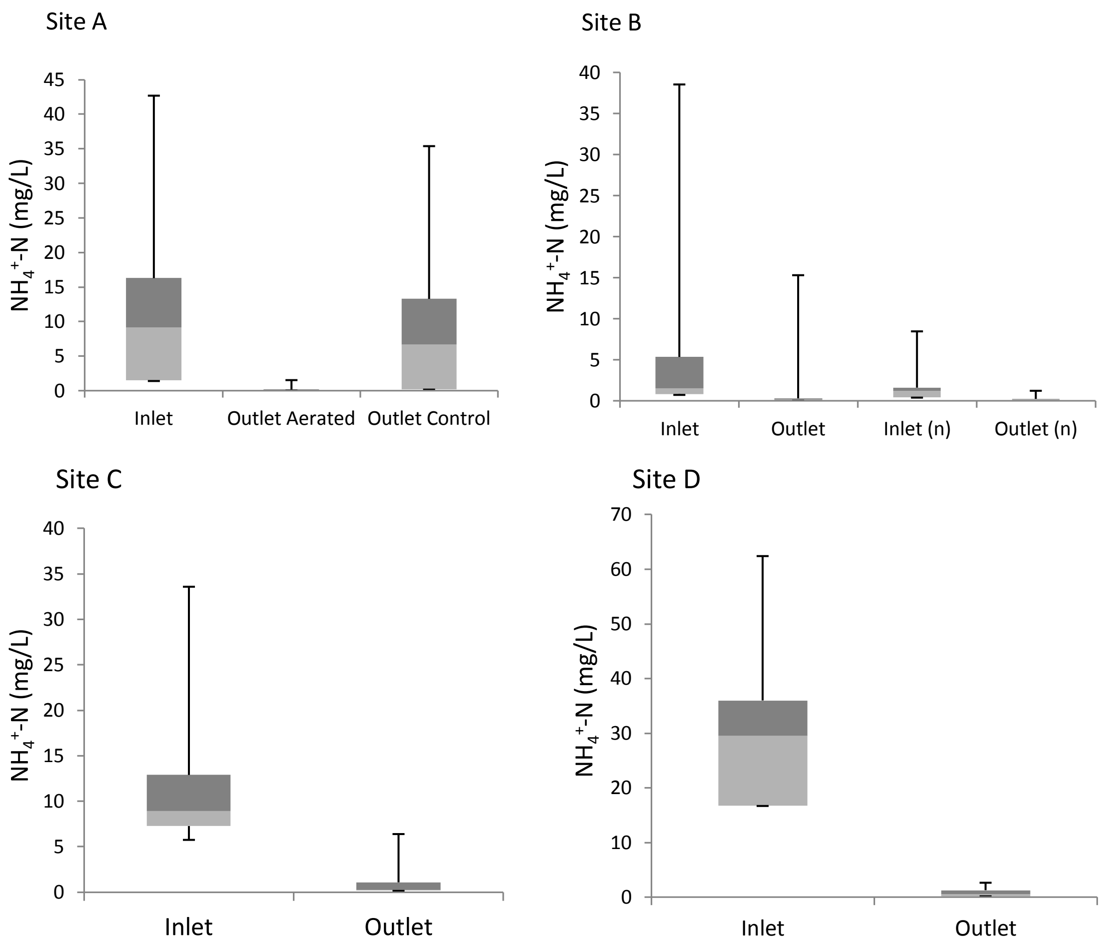

Figure 6.3 Box and whisker plot of NH_4^+-N, concentrations at the inlet and the outlet of the control and aerated beds. (n = Site B data excluding the upstream process failure)

Site B (tertiary combined storm flow, intermittently aerated) experienced an operational issue with the upstream secondary biological process such that a substantial increase in ammonia concentration was loaded on to the bed (Figure 6.4). During the resolution of the upstream event, the influent ammonium concentration to the wetland increased from 1.2±4.0 mgNH_4^+-N/L to 16.0±8.4 mgNH_4^+-N/L with a peak concentration of 33.2 mgNH_4^+-N/L during the winter period when the effluent temperature ranged between 5.2-9.3 °C. Effluent ammonia peaked at 15 mgNH_4^+-N/L, 7 days post failure, and began to decrease 10 days after the event started. Ultimately, the effluent returned to below 3 mgNH_4^+-N/L after 30 days (water temperature 9.5-13 °C), despite consistently high loadings comparable to those received from the secondary bed (Site D), and confirmed the ability of the aerated system to treat high ammonia concentrations under steady state conditions.

Further consideration of response time (resilience) was possible at Site A (test and control site, tertiary combined storm flow) as the beds were taken off line for a period of 5 months, during which time no flow was run through the beds. Upon re-commencement of the flow to the wetlands, the effluent concentration decreased to below 1 mgNH_4^+-N/L after seven days of operation whilst the control bed provided only minimal removal (Figure 6.5). The response of the systems relates to the abundance and activity of the nitrifying population. Once the maximum capable rate of ammonia oxidation per cell is exceeded, the systems will start to fail until sufficient growth of the population occurs to meet the demand (Pickering, 2008). No direct quantification of the nitrifier community was conducted during the current investigation, but it is posited that the consistently low ammonium loading experienced prohibited establishment of large populations such that the system was incapable of responding to concentration spikes due to a lack of abundance of active nitrifiers and hence the system lacks a degree of inherent resilience. This has been observed in other tertiary nitrification systems, where an increase in effluent ammonia has been recorded during the autumn when the inlet ammonium increases concurrent with a temperature derived decrease in nitrification activity in the main secondary process (Pearce, 2013). Effluent from such processes then improves as the nitrification population increases commensurate with the increased available substrate.

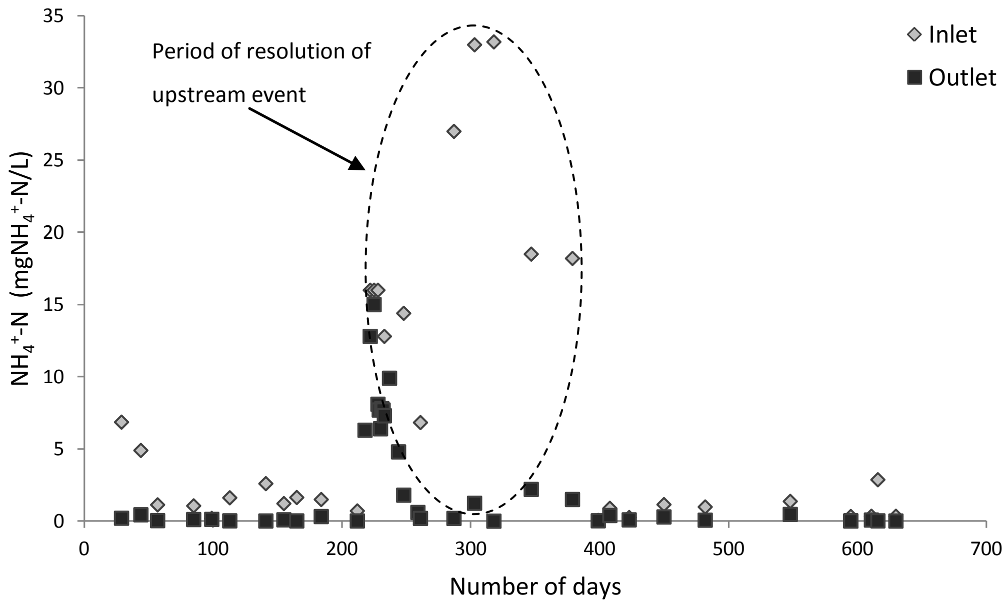

Figure 6.4 Influent and effluent concentrations of ammonium at Site C showing the period of increased loadings and time to reach steady state (30 days)

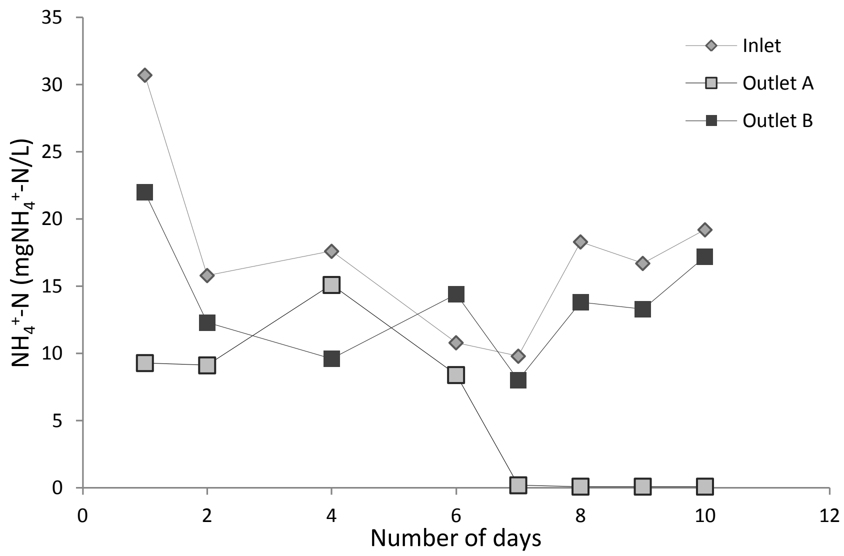

Figure 6.5 Site A restart profiles

In the case of the aerated wetlands, analysis of seasonal impacts revealed a slight increase in effluent ammonium concurrent with increased hydraulic loading posited to be due to heavier rainfall and consequently less residence time in the bed. For instance, sites C and D recorded an increase in the median outlet concentrations in the aerated beds during winter of 0.9 and 1.6 mgNH_4^+-N/L compared to 0.2 and 0.5 mg/L in the summer. In comparison, no difference was observed at Site A with the aerated bed where the mean effluent ammonium remained at 0.2 mgNH_4^+-N/L during both summer and winter periods. In contrast, higher mean outlet concentrations of 8.4 mgNH_4^+-N/L were observed in winter compared to 6.5 mgNH_4^+-N/L when the temperature decreased to 4.1°C.

In fact, no direct correlation was observed between effluent ammonia and temperature or between increased ammonium in the inlet and the outlet during the reduction in activity of the upstream processes. Whilst this suggests that the combination of aeration and long residence time affords a degree of robustness, the response to shock loads suggests that further enhancement of the nitrifying population would be beneficial in terms of resilience. Given that it appears only a small proportion of the bed is active, and only the aerated portions of the bed can sustain an active nitrifying population, it is posited that sequencing the portion of the bed that receives air in a pre defined cycle will enable a larger total community to be maintained and when required utilised by aerating all sections of the bed. Previous work on the integrated fixed film activated sludge process (IFAS) has shown that when the nitrifying microbial community occupy non competitive niches (such as the biofilm carriers in IFAS) the nitrification rate can be controlled by aeration levels as it correlates with DO up to 5 mg/L (Kappel, 2009). Combined, these features offer the potential to enhance resilience of the system and offer a degree of turn up/ turn down control with regards to nitrification through changing aeration rates.

Table 6.2 Summer and winter ammonium inlet loadings and effluent concentrations

		Inlet (gNH_4^+-N/m^2/d)			Outlet ($mgNH_4^+$-N/L)			
		Median	Min	Max	Median	Min	Max	*n*
Summer	Site A (aerated)	2.0	0.3	11.5	0.1	0.02	1.4	19
	Site A (control)	2.0	0.3	11.5	6.5	0.03	20.6	19
	Site B	0.08	0.02	1.9	0.1	0.01	1.5	20
	Site C	0.04	0.01	0.07	0.2	0.04	0.3	5
	Site D	7.4	1.8	12.8	0.5	0.1	2.6	11
Winter	Site A (aerated)	2.8	0.07	12.5	0.1	0.02	0.6	23
	Site A (control)	2.8	0.07	12.5	8.4	0.1	22.1	23
	Site B	0.2	0.02	10.9	0.2	0.02	15.0	22
	Site C	4.3	0.4	11.9	0.9	0.2	5.3	29

Site D	4.1	0.1	13.0	1.2	0.1	13.3	20

6.4.1.1 Process robustness

Excluding the variation due to the upstream events, statistical analysis revealed no significant difference between the effluent concentrations between the tertiary beds A and B (Mann-Whitney U=1434, P=0.7687). Whilst still achieving low effluent concentrations, the increased spread in the data at Site C compared to consistently extremely low concentrations in Site A and B meant the effluent data could not be categorized as statistically the same. However, effluent concentrations were statistically similar in Site C and the secondary bed D (Mann-Whitney U=388, P=0.7545). Accordingly, the efficacy of artificial aeration appears irrespective of configuration of the wetland in terms of the type of flow treated (inclusion of storm flows, secondary or tertiary) across the range of concentrations and loading rates observed in the current study. This leads to the suggestion that nitrification rate is not the controlling factor in the design and operation of such systems.

Further analysis of the data was conducted through calculation of the robustness index (RI) across a range of target goals (Hartshorn et al, 2014). The robustness profile for the control bed at Site A exhibited a curved shape across the majority of the data sets indicative of an un-robust system (Figure 6.6). This is commensurate with the outlet ammonium concentrations that were above the future consent of 4 mgNH_4^+-N/L 63 % of the time. In contrast, comparative data from the aerated beds revealed relatively robust systems with regards to ammonium removal as illustrated by steep near vertical lines across the majority of the data (Figure 6.6) and commensurate with the fact that the outlet concentrations were below respective future consents 100 % of the time, with the exception of the failure recorded at Site B. The use of RI enabled consideration of the impact of tightening consents through changing the target goal level and hence an analysis of the link between tightening discharge consents and robustness of the systems. A similar level of robustness was observed across all sites down to a target effluent level of 3 $mgNH_4^+$-N/L below which the RI of the different sites diverged (Figure 6.7). This extended down to a target concentration of 2 $mgNH_4^+$-N/L including the secondary bed (Site D) when Site C was excluded, the site that was identified to have some general nitrification issues. Sites A and B remained robust down to target concentrations of 0.5 $mgNH_4^+$-N/L indicating that artificial aeration can remain robust even when considered in relation to the most challenging consents discussed for small works in the near future (0.5 $mgNH_4^+$-N/L) (Pearce, 2013).

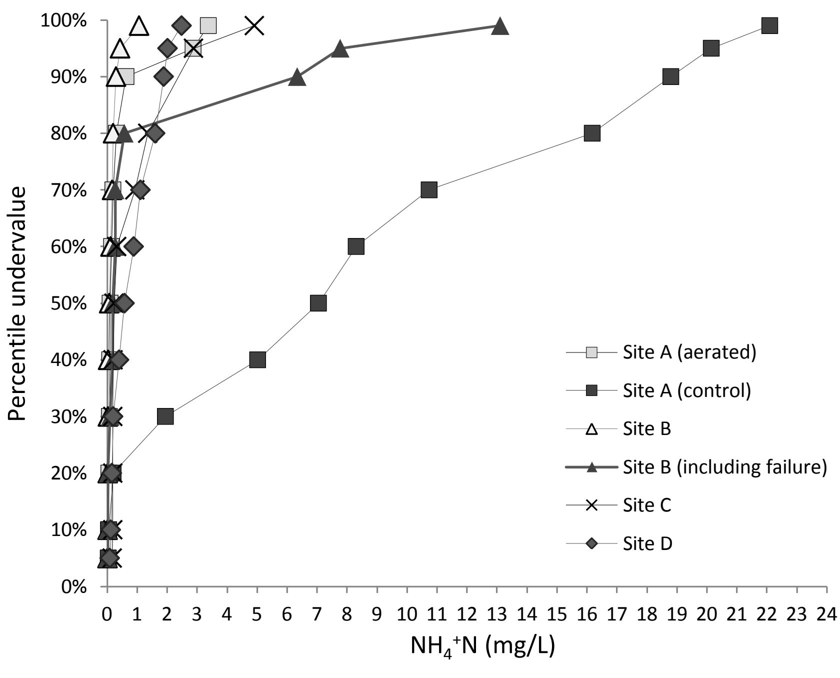

Figure 6.6 Ammonium removal resilience curves for all sites

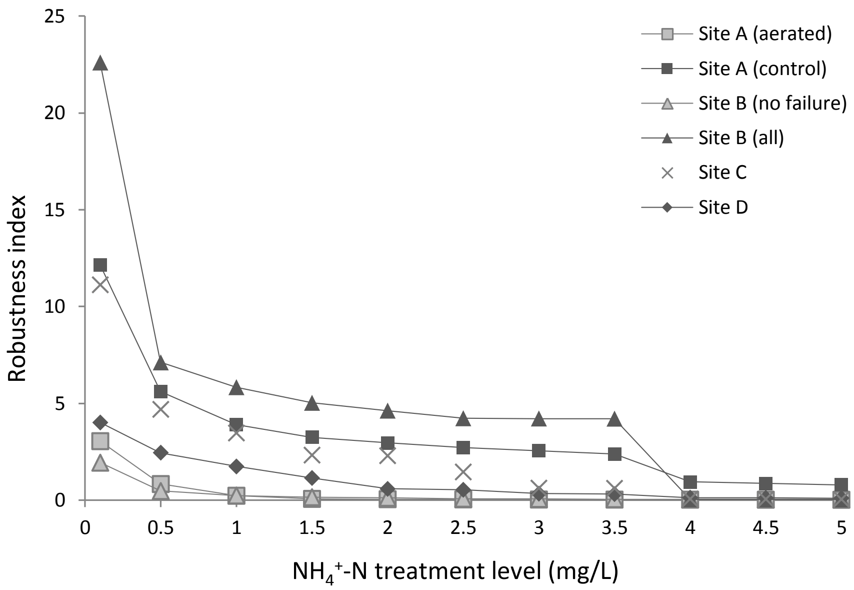

Figure 6.7 Robustness index (RI) of all sites against a range of treatment goals

6.4.2 Solids and organics removal

A significant reduction in effluent total suspended solids concentrations was observed in comparison to inlet concentrations at sites B-D (Mann-Whitney U=172.5, 364, 45.5, P<0.0001) recording median effluent concentrations of 0.1, 2.7 and 20 mgTSS/L respectively (Figure 6.8). This equated to median removal levels of 92 %, 90 % and 65 % for sites B, C and D respectively and corresponding median loading rates of 0.1 (0.1-10.9); 6.3 (0.7-14.1) and 12.5 (2.5-37.5) gTSS/m^2/d (Table A, supplementary information 6.1). In the case of Site A, TSS concentrations were not significantly different between the inlet and outlets of both the aerated and non-aerated beds (Kruskal-Wallis statistic 2.194, P=0.3338). Investigation into the cause revealed a site specific sampling issue reducing confidence in the significance of any findings related to the solids data at that site. No relationship was apparent between TSS loading rate and effluent concentration at any individual site or across all sites combined with the effluent TSS remaining below 40 mgTSS/L up to a maximum loading rate of 25 gTSS/m^2/d with the exception of one measurement (Figure B, supplementary information 6.1). However, comparison of the robustness profiles (Figure 6.9) revealed a decrease in the robustness of the systems as a function of the median TSS loading rate across sites B-D.

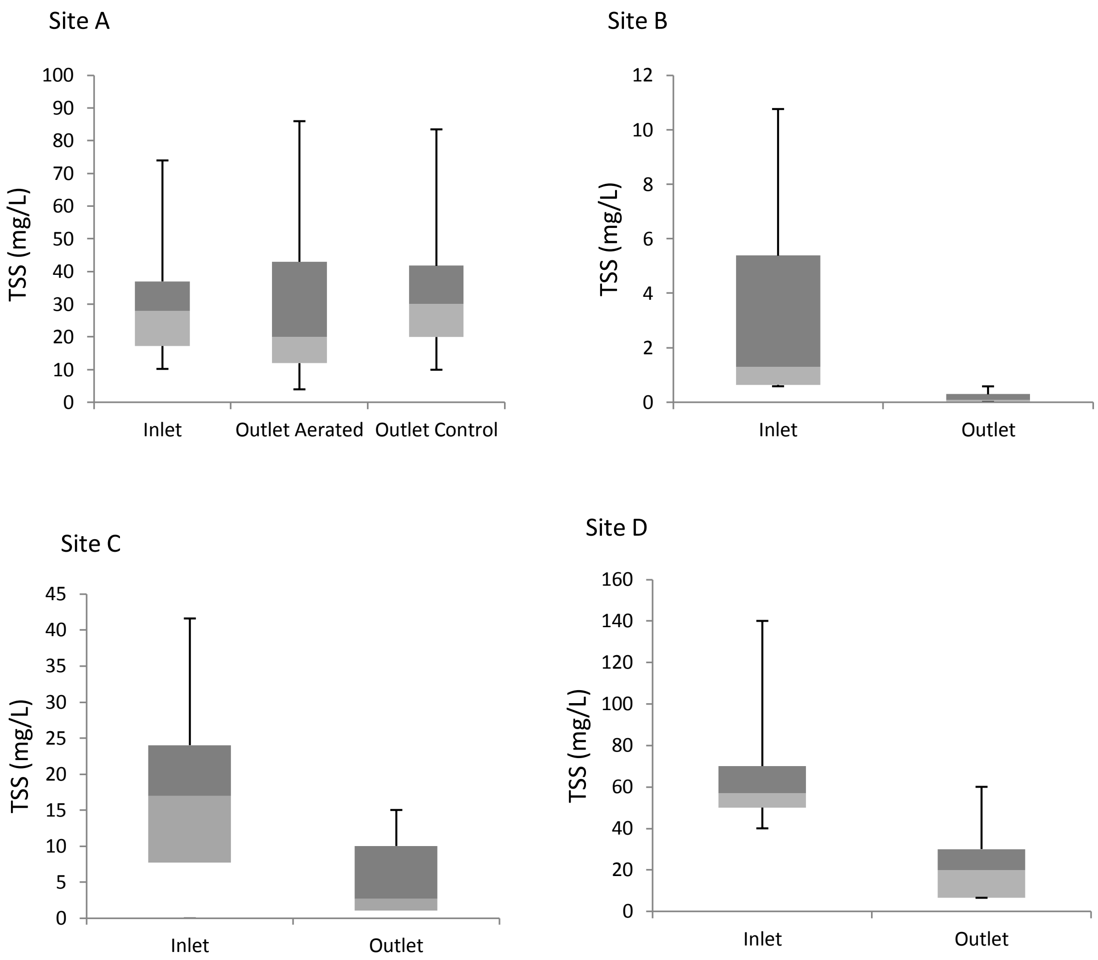

Figure 6.8 Box and whisker plot of TSS, concentrations at the inlet and the outlet of the control and aerated beds. The box represents the interquartile range; the line indicates the mean and the whiskers the 25th and 75th percentiles

For instance, Site B exhibited a near vertical line, consistent with a very robust system only deviating from its near vertical slope beyond the 80th percentile and then only to a solids concentration of 5.5 mgTSS/L at the 99th percentile level and well below the future consent of 45 mgTSS/L 100% of the time (Figure 6.9). However, inlet concentrations onto this site were very low ranging between 0.01-7.3 mgTSS/L with a corresponding median loading rate of 0.1 gTSS/m^2/d. Less robust systems were observed at sites C and D although effluent solids remained below the consent level 100% of the time. To illustrate, Site D (secondary) represented the least robust system with respect to solids with a shape similar to the non-aerated bed with respect to ammonia which is characteristic of a system with little or no inherent robustness to the treatment objective. However, the site achieved a 45 mgTSS/L up to the 95th percentile equivalent to consents on a number of small works and hence shows that aeration of secondary wetlands can provide effective treatment even with regards to solids.

Effluent $CBOD_5$ concentrations remained below respective future treatment goals for all sites (Table A supplementary information 6.1). No significant difference was found between the medians of effluent concentrations from the aerated and control bed at Site A (Kruskal-Wallis statistic 5.428, P=0.2461). Furthermore, $CBOD_5$ loadings did not affect effluent concentrations (Figure C, supplementary information 6.1) and together results suggest the presence of aeration did not further enhance $CBOD_5$ removal. Across all aerated sites effluent $CBOD_5$ remained below 14 mg$CBOD_5$/L up to the maximum tested loading of 25 g$CBOD_5$/m^2/d and confirms previous findings with regards to organic removal in aerated wetlands (Chapter 3, Butterworth et al, 2013)

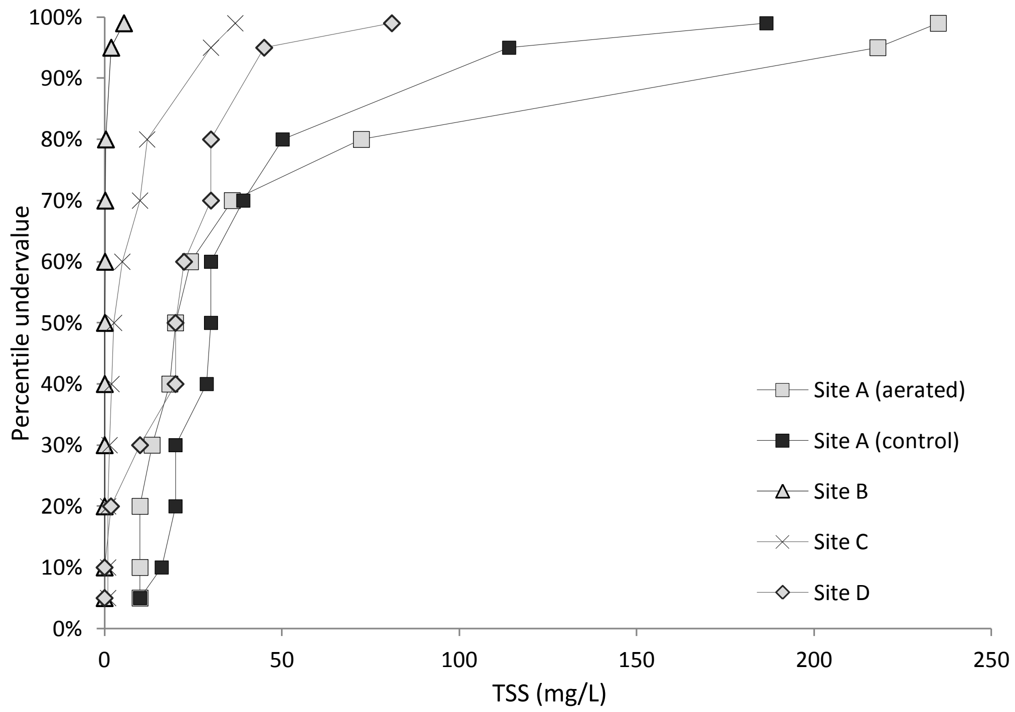

Figure 6.9 TSS removal resilience curves for all sites

6.4.3 Hydraulic conductivity

Conductivity was compared across the majority of sites in year two of the study quantified by a range of 11.5-15.4 m/d and 18.8-25.9 m/d for the inlets and outlets respectively of Sites A, C and D (no significant variation between medians, Kruskal-Wallis, $P<0.05$) and generally showed a decrease at the inlet of the bed compared to the outlet (Figure 6.10), indicating improved hydraulic conductivity with distance from the inlet. Site B differed with comparatively lower values of 3.4-6.0 m/d and no decrease from inlet to outlet. For context, this is comparable to studies that report 350 m/d for clean systems and 4-6 m/d for clogged systems (Pedescoll et al, 2011).

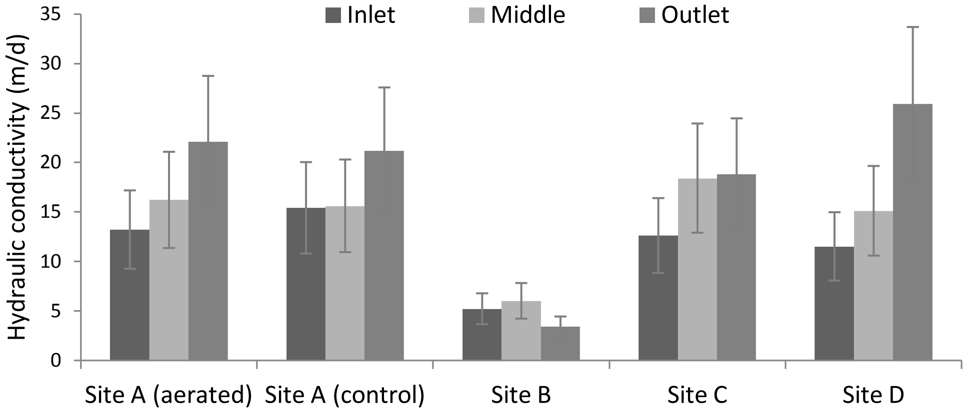

Figure 6.10 Hydraulic conductivity (m/d) for all sites measured during year 2 of the study. Age of sites (months): A =16, B =21, C = 6, D =16

In terms of deterioration in hydraulic conductivity over time, k was higher in year one in the aerated beds at Site A and B followed by a decrease in subsequent years indicating that the initial trend that suggested aeration may improve hydraulic conductivity (Chapter 3, Butterworth et al, 2013) and solids removal (Chazarenc et al, 2009) is not supported by our data. No pattern was obvious in values measured from the inlet, middle and outlet of the beds at Site A, but an increase from inlet to outlet was observed in site B in both years (Figure 6.11).

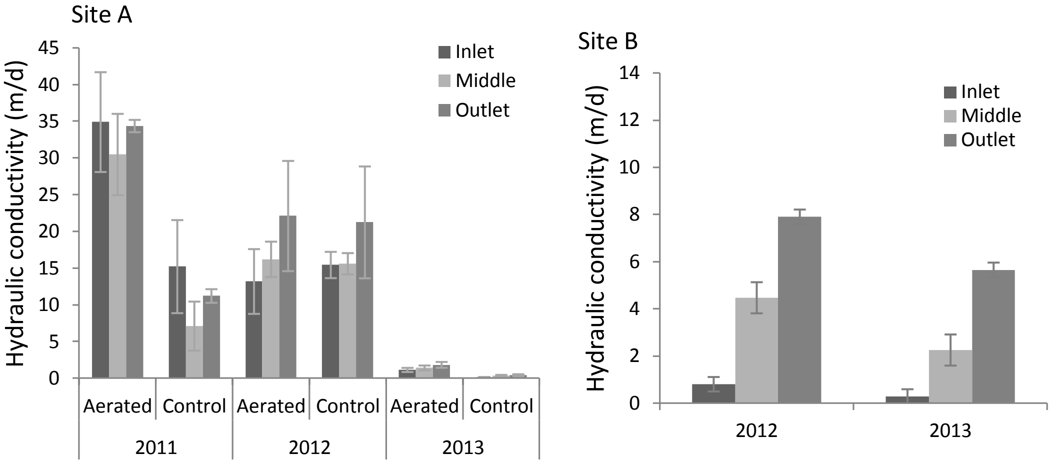

Figure 6.11 Yearly hydraulic conductivity (m/d) for Site A (left) and Site B (right)

Further analysis of the impact of aeration on mixing pattern was conducted by means of tracer studies in years one and two at Site A (aerated v non-aerated)(Figure 6.12).

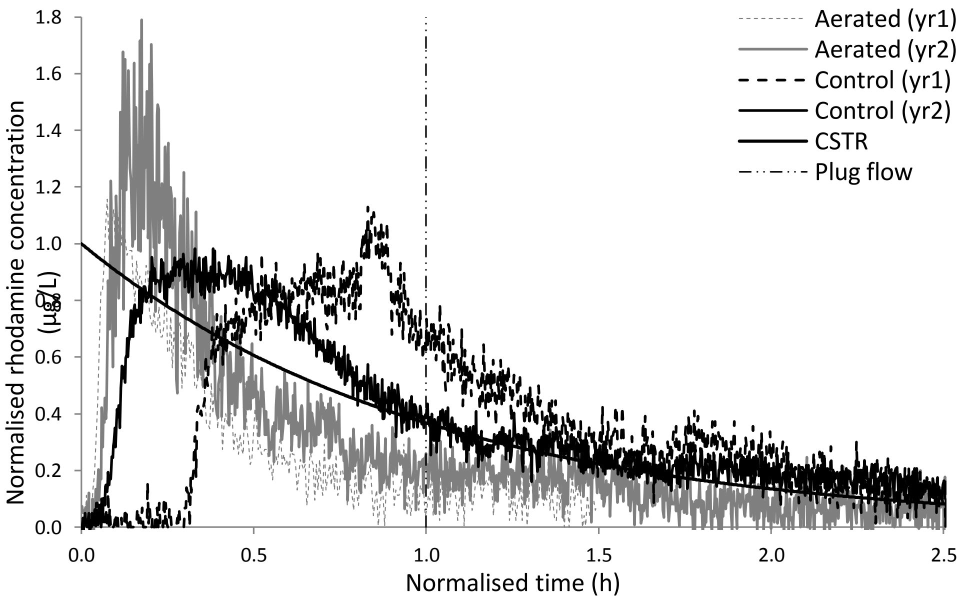

Figure 6.12 Year one and two residence time distributions for the aerated and control beds at Site A

Both beds displayed non-uniform flow, with a greater tendency towards plug flow in the control bed compared to the aerated bed, indicated by the higher index of modal retention time (τ_p/τ_n) of 0.33-0.56 in the control compared to 0.01-0.17 in the aerated bed (Figure 6.12), similar to the patterns observed in year one (Chapter 3, Butterworth et al, 2013). In support of the difference in mixing patterns, the Morrill dispersion index (MDI) was higher in the aerated bed than the control in both years, more consistent with CSTR mixing patterns; calculated as 14.9-31.7 and 4.5-9.0 in the aerated and control beds respectively (Table 6.3). In addition, the MDI approximately doubled in both beds in year two compared to year one consistent with increased back mixing. Support for this is provided through analysis of convective dispersion which increased in both beds as a function of age rising from 0.37 to 0.59 in the aerated and 0.15 to 0.26 in the control beds (Table 6.3). The volumetric efficiency was calculated as 3.2-6.7 % in the aerated bed compared to 11.1-22.3 % in the control and decreased by 52 % in the aerated bed between year one and year two and 50 % in the control. Overall, analysis of the change in RTDs indicates that the beds are progressively deviating from plug flow consistent with the beds becoming clogged and operated with a slight free water level. Additionally the change in hydraulic conductivity and mixing pattern in both beds appears to progressive at a similar relative rate suggesting that aeration is unlikely to positively influence long term hydraulic operation.

Table 6.3

	Q (m^3/h)	Recovery (%)	τ_n (h)	τ (h)	τ_p/τ_n	D	MDI	e_v (%)
Aerated yr 1	1.8	77	14.2	6.2	0.01	0.37	14.9	6.7
Aerated yr 2	1.8	72	14.2	24.5	0.17	0.59	31.7	3.2
Control yr 1	1.1	35	34.2	28.1	0.56	0.15	4.5	22.3
Control yr 2	1.2	67	21.2	14.7	0.33	0.26	9.0	11.1

Q = flow rate; τ_n = nominal residence time; τ = residence time; $\sigma = \tau_p/\tau_n$ = index of modal retention time; D = dispersion number; MDI =Morrill dispersion index; e_v = volumetric efficiency

6.6 Conclusions

Ammonium removal performance of aerated horizontal flow sub-surface flow constructed wetlands was assessed at four full-scale small sewage treatment works of various configurations. Based on the results of this study, the technology was shown to be capable of delivering nitrified effluents down to 3 $mgNH_4^+$-N/L in both a secondary system and tertiary treatment applications including combined storm flow beds with the potential to deliver sub 1 $mgNH_4^+$-N/L in the case of tertiary systems. The system was observed to remain robust in systems receiving variable loadings between 0.1-13.0 NH_4^+-N/m^2/d. However, the systems showed limited resilience to spike loads posited to be due to insufficient abundance of the nitrifying community within the bed which could be ameliorated through cyclic operation of the aeration to sequenced parts of the bed.

Further investigation into hydraulic characterisation recorded hydraulic conductivity values in a similar range over the sites and comparison of values over time suggested the presence of aeration did not reduce the clogging effect commonly reported in CWs. Furthermore, mixing patterns in the bed demonstrated a decrease in volumetric efficiency over time. Overall, the study demonstrates the efficacy of the technology where ammonium removal is required on small sites receiving high and variable flow rates, with adequate removal of $CBOD_5$ and TSS, but has no significant benefit to the long term hydraulics of the system.

6.7 Acknowledgments

The authors would like to express their gratitude for the financial and resource support of the work from Severn Trent Water Limited and to the EPSRC for their allocation of funding and support of the project.

6.8 References

Alburquerque A., Oliveira J., Semitela S., Amaral L. (2009) Influence of bed media characteristics on ammonia and nitrate removal in shallow horizontal subsurface flow constructed wetlands. *Bioresource Technology* 100(24):6269–6277

Boller B., Gujer W. (1985) Nitrification in tertiary trickling filters followed by deep-bed filters. *Water Research* 20(11):1363-1373

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

Chazarenc F., Gagnon V., Comeua Y., Brisson J. (2009) Effect of plant and artificial aeration on solids accumulation and biological activities in constructed wetlands. *Ecological Engineering* 35(6):1005-1010

Cottingham P., Davies T., Hart B. (1999) Aeration to promote nitrification in constructed wetland. *Environmental Technology* 20(1):69–75

Dong H., Qiang Z., Li T., Jin H., Chen W. (2012) Effect of artificial aeration on the performance of vertical-flow constructed wetland treating heavily polluted river water. *Journal of Environmental Sciences* 24(4):596-601

Garcia J., Aguirre P., Barragan J., Mujeriego R., Matamoros V., Bayona J. (2005) Effect of key design parameters on the efficiency of horizontal subsurface flow constructed wetlands. *Ecological Engineering* 25(4):405-418

Gross A., Sklarz M., Yakirevich A., Soares I. (2008) Small scale recirculating vertical flow constructed wetland (RVFCW) for the treatment and reuse of wastewater. *Water Science and Technology* 58(2):487–494

Fan J., Wang W., Zhang B., Ngo H., Guo W., Zhang J., Wu H. (2013) Nitrogen removal in intermittently aerated vertical flow constructed wetlands: Impact of influent Cod/N ratios. *Bioresource Technology* 143:461-466

Foldori P., Ruaben J., Ortigara A. (2013) Recirculation or artificial aeration in vertical flow constructed wetlands: A comparative study for treating high load wastewater. *Bioresource Technology* 149:398-405

Hartshorn A., Prpich G., Upton A., Macadam J, Jefferson B., Jarvis P (2014)

Assessing filter robustness at drinking water treatment plants. *Water and Environment Journal* Submitted

Hwang J., Oleszkiewicz J. (2007). Effect of cold-temperature shock on nitrification. *Water Environment Research* 79(9):964–968

Ishikawa S., Suzuki T., Nikata T., Kakii K., Shirakashi T. (2003) Tertiary nitrification in a new dual-medium aerated filter. *Water Science and Technology* 47(1):219-227

Kadlec R., Wallace S. (2009) *Treatment Wetlands* (2^{nd} edn) Taylor & Francis Group: Florida, US

Kappel C. (2009) Simplified Evaluation of the Operational Oxygen Transfer Efficiency of a novel dispersed integrated fixed film activated sludge process. MSc Thesis Cranfield University, Cranfield UK

Knowles P., Dotro G., Nivala J., Garcia J. (2011) Clogging in subsurface-flow treatment wetlands: occurrence and contributing factors. Ecological Engineering 37(2):99-112

Llorens E., Matamoros V., Domingo V., Bayona J., García J. (2009) Water quality improvement in a full-scale tertiary constructed wetland: Effects on conventional and specific organic contaminants. *Science of the Total Environment* 407(8):2517-2524

Maltais-Landry G., Maranger R., Brisson J. (2009) Effect of artificial aeration and macrophyte species on nitrogen cycling and gas flux in constructed wetlands. *Ecological Engineering* 35(2):221-229

Matamoros V., García J., Bayona J. (2008) Organic micropollutant removal in a full-scale surface flow constructed wetland fed with secondary effluent. *Water Research* 42(3):653-660

Matthys A., Parkin G., Wallace S. (2000) A comparison of constructed wetlands used to treat domestic wastes: conventional, drawdown and aerated systems. Proceedings of the *7th International Conference on Wetland Systems for Water Pollution Control*; *International Water Association,* 11th-16th November. Colchester, UK

Molle P., Lienard A., Grasmick A., Iwema A. (2006) Effect of reeds and feeding operations on hydraulic behaviour of vertical flow constructed wetlands under hydraulic overloads. *Water Research* 40(3):606-612

Nivala J., Headley T., Wallace S., Bernhard K., Brix H., van Afferden M.,Muller R. (2013) Comparative analysis of constructed wetlands: The design and construction of the ecotechnology research facility in Langenreichenback, Germany. *Ecological Engineering* 61:527-543

Nivala J., Hoos M., Cross C., Wallace S., Parkin G. (2007) Treatment of landfill leachate using an aerated, horizontal subsurface-flow constructed wetland. *Science of the Total Environment* 380(1-3):19–27

O'Luanaigh N., Goodhue R., Gill L. (2010) Nutrient removal from on-site domestic wastewater in horizontal subsurface flow reed beds in Ireland. *Ecological Engineering* 36(10):1266–1276

Pedescoll A., Uggetti E., Llorens E., Granés F., García D., García J. (2009) Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. *Ecological Engineering* 35(8):1216-1224

Pedescoll A., Samsó R., Romero E., Puigagut J., Garcia J. (2011) Reliability, repeatability and accuracy of the falling head method for hydraulic conductivity measurements under laboratory conditions. *Ecological Engineering* 37(5): 754-757

Pickering R. (2008) *How hard is the biomass working?* PhD thesis Newcastle University, Newcastle upon Tyne, UK

Wallace S., Kadlec R. (2005) BTEX degradation in a cold-climate wetland system. *Water Science and Technology* 51(9):165-171

Wu S., Zhang D., Austin D., Dong R., Pang C. (2011) Evaluation of a lab scale tidal flow constructed wetland performance: oxygen transfer capacity, organic matter and ammonium removal. *Ecological Engineering* 37(11):1789-1795

Vymazal J. (2005) Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. *Ecological Engineering* 25(5):478–490

Vymazal J. (2011) Long-term performance of constructed wetlands with horizontal sub-surface flow: Ten case studies from the Czech Republic. *Ecological Engineering* 37(1):54-63

Yousefi Z., Mohseni-Bandpei A. (2010) Nitrogen and phosphorus removal from wastewater by subsurface wetlands planted with *Iris pseudacorus*. *Ecological Engineering* 36(6):777–782

Supplementary information 6.1

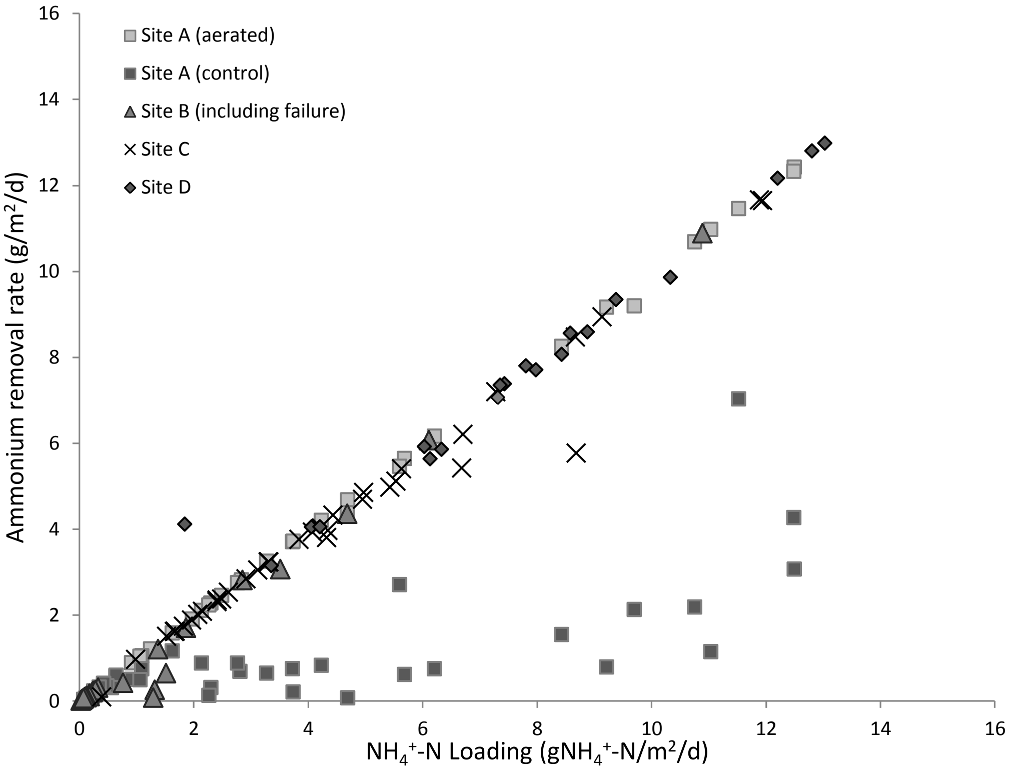

Figure A Ammonium loading and associated nitrification rate Sites A-D

Table A Inlet and outlet concentrations of $CBOD_5$, TSS and NH_4^+-N

		Inlet (mg/L)			Outlet (mg/L)			
		Median	Min	Max	Median	Min	Max	*n*
$CBOD_5$	Site A (aerated)	11.9	4.6	67.2	4.7	0.5	31.0	32
	Site A (control)	11.9	4.6	67.2	6.2	1.3	33.1	32
	Site B	10.6	0.3	74.4	3.7	0.1	27.3	45
	Site C	7.6	1.1	31.2	4.2	1.2	21.8	16
	Site D	79.4	39.9	98.3	4.7	1.8	20.8	19
TSS	Site A (aerated)	25.0	3.7	87.0	20.0	1.5	237.0	40
	Site A (control)	25.0	3.7	87.0	25.0	2.0	212.0	40
	Site B	1.3	0.1	5.5	0.1	0.0	7.3	40
	Site C	16.7	2.0	80.0	2.7	1.0	40.0	37
	Site D	57.0	10.0	150.0	20.0	0.0	90.0	17
NH_4^+-N	Site A (aerated)	9.1	0.1	26.4	0.1	0.02	1.4	43
	Site A (control)	9.1	0.1	26.4	6.7	0.03	22.1	43
	Site B	1.6	0.1	33.2	0.2	0.01	15.0	90
	Site C	8.9	1.5	20.7	0.2	0.04	14.0	40
	Site D	29.4	1.0	59.6	0.8	0.1	2.6	16

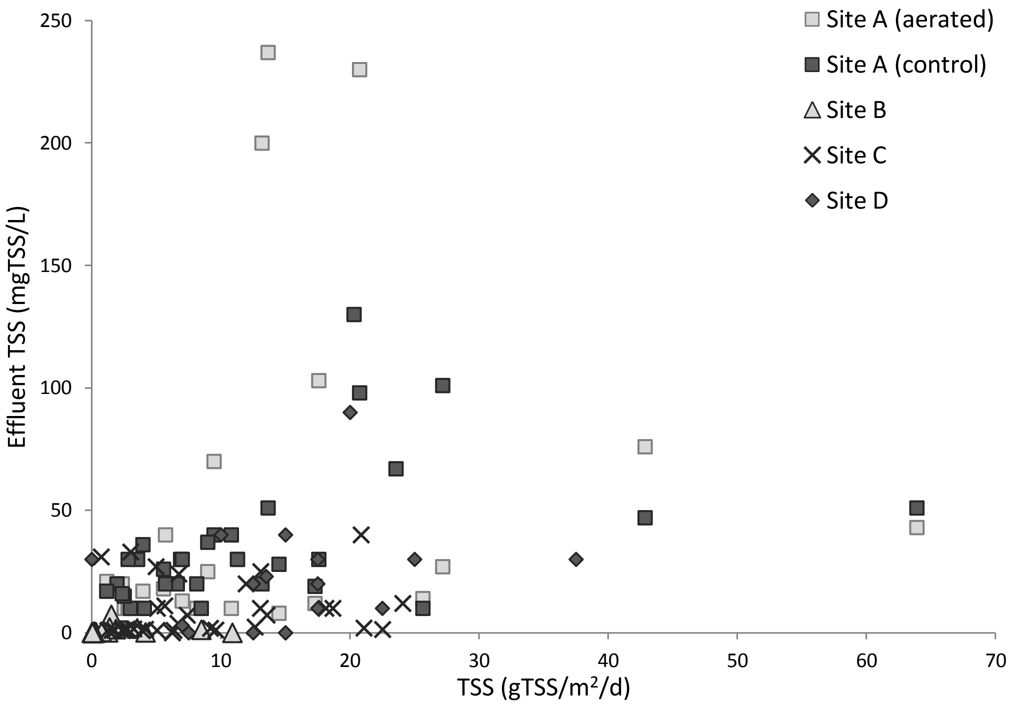

Figure B TSS loadings and effluent concentrations for all sites

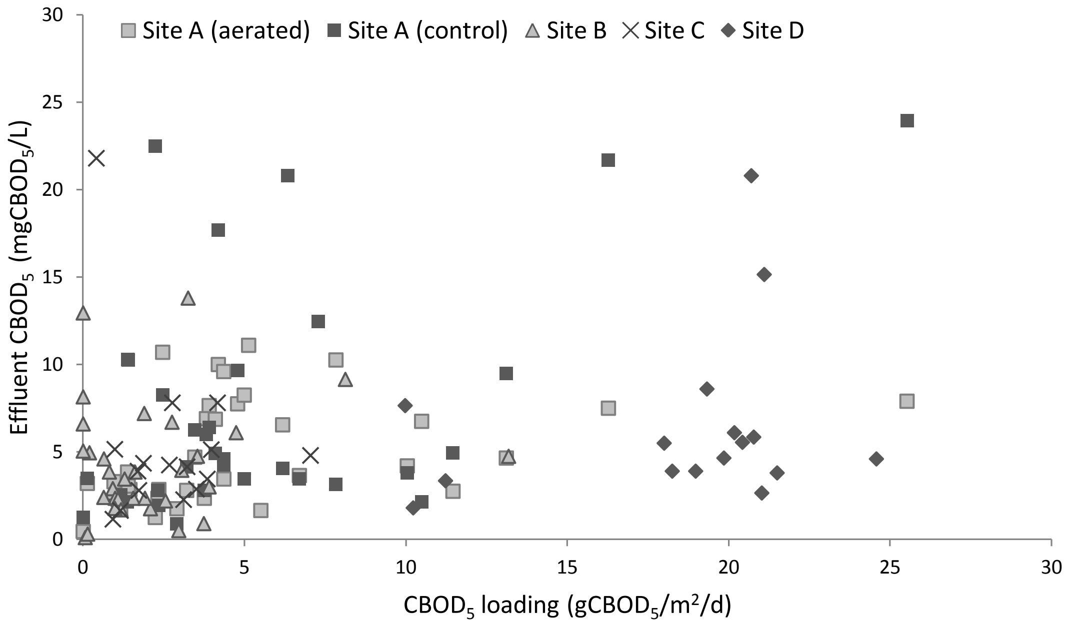

Figure C $CBOD_5$ loadings and effluent concentrations for all sites

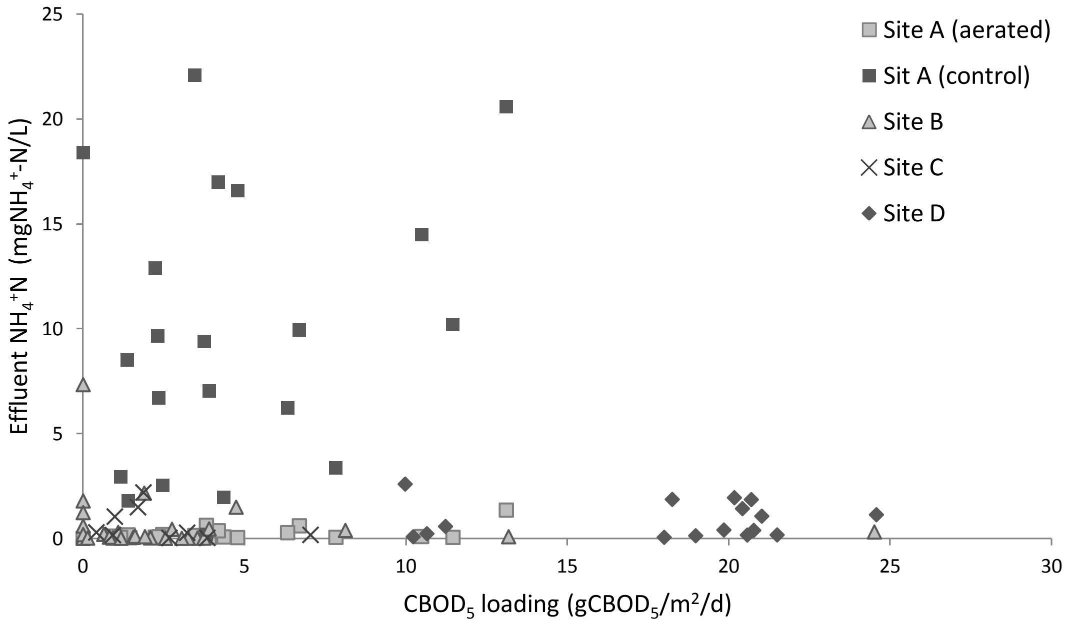

Figure D $CBOD_5$ loadings and NH_4^+-N effluent concentrations for all sites

Chapter 7

Implications of the work

Implications of the work

The work presented in this thesis has covered a range of topics in order to understand the efficacy of artificial aeration of HSSF wetlands towards tertiary nitrification and consider future best practice in terms of design and operation.

7.1 Efficacy of artificial aeration

The work has demonstrated that inclusion of an aeration system to a standard design of a HSSF wetland (based on Severn Trent practice) is an effective approach to upgrade the technology and enable effective nitrification to be delivered. This is demonstrated across the range of current operational situations where HSSF wetlands are used within the sponsoring water company including tertiary treatment, combined storm flow tertiary treatment and secondary treatment (Chapter 6). Comparison across the sites revealed that a sub 3 $mgNH_4^+$-N/L could be robustly delivered and that robust treatment could potentially be maintained down to below 0.5 $mgNH_4^+$-N/L. This compares well with alternative options such as nTFs, RBCs, nSAFs (Boller et al, 1994) and nitrifying depth filters (Barter and Smith, 2007) and even against upgrading to activated sludge or membrane bioreactors options as the main secondary processes where sub 1.0 $mgNH_4^+$-N/L can be routinely achieved (Judd, 2011). Consequently, the current work provides confidence that the inclusion of aeration into HSSF CWs is an effective option against even the most restrictive consents currently experienced on small works (0.5 $mgNH_4^+$-N/L) (Pearce, 2013). One site (Site C) showed a reduced level of robustness indicating potential risk to compliance, especially if more restrictive consents are targeted. Initial diagnostics eliminated oxygen deficiency and alkalinity as probable causes such that ongoing investigations are focussed towards limited establishment of the nitrifying microbial population within the bed (see section 7.3)

Comparison to alternative tertiary treatment options revealed that retrofitting of existing HSSF CWs was an attractive option in terms of cost offering a reduction compared to more traditional options of between 2.6 (nTF) to 7.2 (RBC) times. This was balanced against an increase in energy associated with aeration by a factor of three to four times that of the nTF and RBC respectively (Chapter 3). This is consistent with reports from other case studies which have reported the energy demand to be around 2.43 Wh/PE which is similar to that of an activated sludge plant at 2.45 Wh/PE (Pearce, 2013). This challenges the validity of artificial aeration of HSSF CWs with respect to the wider philosophy of small rural works as low energy systems. However, subsequent experimental testing confirmed the hypothesis that such systems are currently sub-optimally designed (Chapter 4) such that significant reduction in the energy demand associated to aeration can be achieved through alternative design (see section 7.2). This has the potential to reduce the equivalent energy demand down to as low as 0.2-0.3 Wh/PE with a confident expectation that equivalence to the other low energy treatment option can be routinely delivered (c0.6-0.8 Wh/PE) (based on energy calculations from Chapter 3). This realigns the attributes of the technology to that of a low energy treatment system where choice between alternatives become more focussed on footprint versus total cost and hence where HSSF CWs already exist, artificial aeration appears the most suitable option.

7.2 Design of artificially aerated horizontal flow wetlands

High ammonium loading rates were shown not to be the rate limiting factor for effective nitrification up to the maximum nitrification loading rate observed of 14.9 gNH_4^+-$N/m^2/d$. Therefore, across the range of applications considered in the current work (secondary and tertiary municipal sewage treatment) aerated CWs can be considered to be hydraulically and solids loading limited. Further, whilst aeration was seen to influence solids build up, their character and hydraulic conductivity during the first year of operation (Chapter 3), comparison across the different sites indicated no long term benefit (Chapter 6). Consequently, overall sizing of the wetland should adhere to current design specifications but with the inclusion of aeration. Accordingly, sizing should be based on the current asset standards of 0.5 m^2/PE (storm only), 0.7 m^2/PE (tertairy), 1.0 m^2/PE (combined strom and tertiary), 5 m^2/PE (secondary) (Cooper et al, 1993). The bed is then filled with distribution and collection media of 20-200 mm diameter at the inlets and outlets of the bed and the remainder filled with river gravel of 6-12 mm diameter to a depth of 600 mm. The influent wastewater is then fed to the bed via a 'v' notched distribution trough and travels through the bed to the effluent collection pipe and discharged to the receiving watercourse. An overflow pipe and a freeboard of 500-1000 mm are advised to mitigate against flooding events (Figure 7.1).

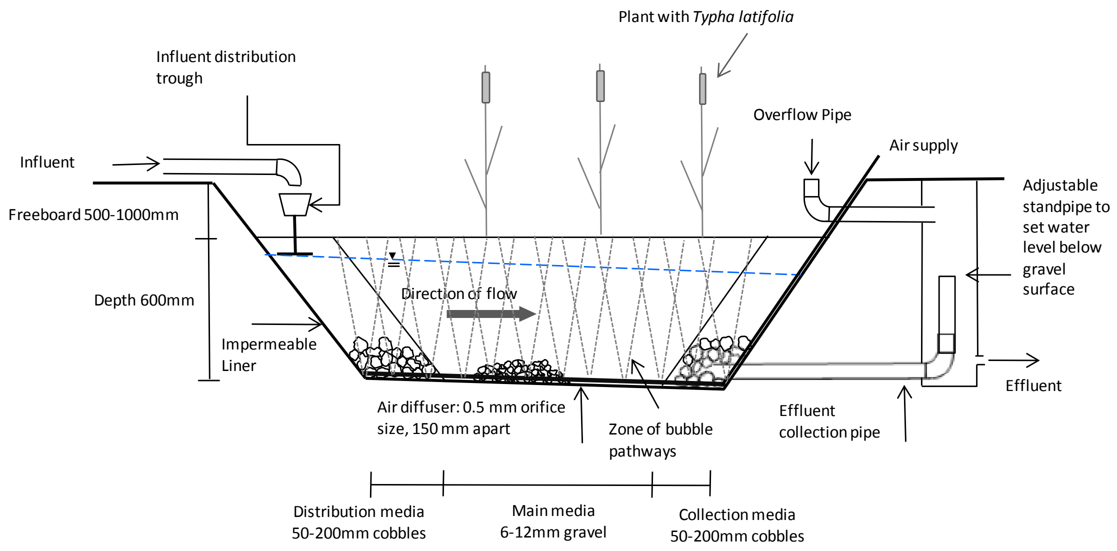

Figure 7.1 Amended design of an HSSW CW for aeration

Three specific amendments are proposed based on the findings of this work:

(1) Redesign of the aeration system;

(2) Switch of the preferred plant species to *Typha latifolia*;

(3) Operation as truly sub surface

7.2.1 Aeration system

Current design involves the use of a perforated pipe with 2 mm orifice holes spaced roughly 300 mm apart delivered from a fixed speed blower that keeps the beds approximately saturated with DO and hence constitutes over aeration. An investigation into mass transfer in packed beds in this work demonstrated enhanced oxygen transfer over that achieved in open tanks (Chapter 4). Both orifice size and air flow rates were observed to influence overall transfer efficiency whilst bubble size remained effectively constant. Accordingly, it is suggested that the mass transfer enhancement mechanism concerns retarded bubble rise velocity and pseudo stagnant gas storage. Consequently, enhanced mass transfer is partially related to pore space replenishment rate and as such is maximised when smaller orifice holes and lower gas flow rates are utilised. This is illustrated through comparison of the SAE (Table 7.1) whereby reducing the orifice size increased SAE by 1.8 times when comparing a 2 mm orifice to a 0.5 mm orifice at a gas flowrate of 20 L/min.

Table 7.1 Comparison of SAE

Orifice size (mm)	Air flow rate (L/min)	SAE media (kgO_2/kW)
2	100	6.7±2.2
2	20	10.1±4.8
0.8	20	10.5±3.4
0.5	20	18.3±7.0

Bubble visualisation experiments, using a sub 1.0 mm orifice, observed a zone of influence of the bubbles in the media systems of around 150-300 mm with the majority of the bubble within the first 150 mm radius from the centre point of the orifice. This compares to recent observations from Site C (Chapter 6), during the ongoing diagnostic investigations (Dotro, 2014) that indicated that the zone of influence extended 250 mm with the 2.0 mm orifice size with DO saturated at the centre point of the plume and reduced down to 2.5 mg/L at the midpoint between plumes (i.e. 125 mm from the centre point of the plume).

Based on the above it is recommended in future systems that orifice size is reduced to sub 1.0 mm configured in a grid with 150 mm spacing between orifices. Illustration of the benefit of this is provided based on one of the existing sites (site A, Chapter 4, 6) where even when conservative design assumptions are employed (α=0.3, minimum OTE observed - 11%) the required air flow rate to meet the OD was 3378 m^3/d which equates to 0.5 L/min/hole based on the 10x10 m site and holes spaced in a 15 cm grid. This is lower than the suggested rate of 0.8 L/min/hole from the OTE and hence the system has a degree of buffer to ensure appropriate delivery. However, this is substantially lower than the current rate of 2.3 L/min/hole which lowers enhancement of mass transfer through replenishing the pore spaces too frequently. At a rate of 0.8 L/min/hole the estimated blower power is 0.36 kW equating to approximately 0.8 Wh/PE.

The low α was used to take into account the potential impact that enhanced contact time may have on mass transfer in real wastewater through additional accumulation of surface active agents. Consideration from the literature concerning cases directly considering the role of surfactants (Asselin et al, 1989) or elevated biomass concentrations (Germain et al, 2007) as well as the comparison of fine versus coarse bubble systems (Painmanakul and Hebrard, 2008) indicates that alpha factors around 0.5 would be a more realistic minimum level likely to be experienced in aerated wetlands. Re-calculation utilising the more realistic alpha factor and a median OTE reduces energy usage down to a predicted 0.4-0.5 Wh/PE demonstrating that artificial aeration can be utilised at rural small works without challenging the underlying philosophy of low energy treatment.

7.2.2 Vegetation

Artificial aeration of wetlands was observed to have a negative impact on plant growth (Chapter 6) consistent with previous reports from commercial groups who have reported yellowing of the leaves (chlorosis) and poor plant establishment. This is known to occur in other aerobic wetland configurations such as tertiary vertical flow systems (Weedon, 2014) and has been suggested to be species specific (Dotro et al, 2012). Work presented here (Chapter 5) confirmed the negative impact of aeration on plant growth for both *Phragmites australis* and *Typha latifolia.* On an individual plant basis, this thesis indicated *T. Latifolia* was influenced more heavily by aeration than *P. australis* compared to their respective controls. However, the impact on the absolute growth was less significant; congruent with its faster growing rate such that more effective bed coverage remained possible.

The importance of full vegetative coverage in HSSF CWs has been linked to a number of attributes but in temperate and cold climates, its role in insulating the bed is considered one of the most important (Mander and Jenssen, 2002) along with seasonal nutrient storage. Consequently, it is desirable to use plants that have high above ground biomass and quick establishment to create a dense cover soon after planting (Vymazal and Kröpfelová, 2005). High potential productivity, deep rhizome and root system, ready propagation and wide distribution of *P. australis* have made it the most common plant used in constructed wetlands in Europe; where it is specified as the desired choice in design guidelines (Cooper, 1993). However, it has been documented that *P. australis* usually takes three to four growing seasons to reach the maximum standing crop but in some systems it may take even longer (Vymazal and Kröpfelová, 2005). In comparison, *T. Latifolia* is competitive and aggressive in nature; colonizing inert substrates and adapting to diverse and not always optimal climate conditions (Jespersen et al, 1998). Consequently, *T. Latifolia* is recommended where rapid plant cover is required in an aerated wetland. However, this should be balanced against the impact of the additional above ground biomass likely to be generated by switching from *P. australis* to *T. Latifolia,* even considering the reduced growth rates of the latter due to aeration. The consequence is an increased surface sludge accumulation rate as current practice is not to harvest at the end of the growing season with the potential for a commensurate reduction in the cycle time between refurbishments. Given that switching to *T. Latifolia* ameliorates but does not totally resolve the negative impacts of aeration on plant coverage the net balance may not justify the change unless maintenance practice changes to include biomass harvesting (see section 7.3).

7.2.3 Water level

Artificial aeration was observed to alter the mixing pattern within the bed as a function of the operating water level (Chapters 3, 6). When the water level exceeded the top of the gravel layer, the overall mixing pattern in the aerated bed deviates from that of the non-aerated bed to be progressively more associated to that of a continuously stirred tank reactor (CSTR). When the water level was below the gravel, both beds exhibited similar mixing patterns to that of a non ideal plug flow reactor (PFR).

The importance of this is related to advective spreading as indicated by the increased dispersion number (0.63 in the aerated compared to 0.23 in the non aerated, Chapter 3). Further, re-examination after a longer operating period (Chapter 6) indicated that solids accumulation had increased the dispersion number of both aerated and non aerated beds roughly equally, suggesting that back mixing and spreading will increase over the life of the bed. The importance of this in relation to aerated wetlands is that material is being lifted into the free water layer above the bed and consequently provides a potential short residence bypass in the system and hence constitutes a conceivable process risk. Whilst this has not been realised during the full scale site monitoring, it represents a future risk which is most likely to be an issue once solids accumulation has progressed and so is unlikely to occur during the first few years of operation and will become observable at different time periods on different sites based on solids accumulation rates.

The risk can be minimised through controlled operation of the bed so that the water level remains below the gravel at all times where practically feasible. This will ensure the aerated bed still operates under plug flow characteristics. It is posited that this will also maximise the effectiveness of the systems and potentially enhance its robustness as in any biofilm based biological process. Additionally, in cases of risk of restrictive odour emissions (secondary beds) the influent feed systems should be delivered sub-surface, preferably near the bottom of the bed to maximise oxygen transfer.

7.3 Alternative operating strategies to enhance resilience

One site (Site C) revealed reduced robustness (normal operation variation) and all sites showed a slow response in terms of resilience (response to shock load) (Chapter 6). This is consistent with general understanding of nitrifying processes that are recognized as being difficult to maintain in practical wastewater treatment plants (Satoh et al, 2003) requiring a long start-up period and demonstrating slow recovery from failure events due to the slow growth rate and low growth yield of nitrifying bacteria (Rittmann and Whiteman, 1994). Nitrification performance is known to be dependent on the activity, composition and diversity of the ammonia oxidising bacteria (AOB) and nitrite oxidising bacteria (NOB) (Knapp and Graham, 2007; Wittebolle et al, 2005; Bellucci et al, 2013). Once the maximum capable rate of ammonia oxidation per cell is exceeded, the systems will start to fail until sufficient growth of the population occurs to meet the demand and has been demonstrated for activated sludge systems (Pickering, 2008). Two principal situations can cause this lack of abundance in the aerated HSSF CWs: (a) low overall feed ammonia load and (b) out competition from heterotrophic organisms.

The first case is considered the most common where the systems are being used to deliver against tightening standards and where the upfront works already provides a significant level of nitrification. Examples in the current work include sites A and B and here it is suggested that the systems will be generally robust but not resilient as they should maintain a sufficient nitrifying population to cope under normal ammonia load situations. For instance, observations at Site A, revealed that the majority of ammonia was removed (< 1 mgNH_4^+-N/L) within the first metre from the inlet. Aeration beyond this point is unlikely to have developed or been able to sustain any level of abundance of nitrifiers and hence the system effectively acts as a very short residence time bioreactor for nitrification which can quickly become overloaded. It follows therefore that increasing the number of active cells may enhance the resilience of the bed to variable ammonium loadings by ensuring the maximum nitrifying potential is not exceeded. Quantitative studies based on real-time qPCR and FISH demonstrated that efficient nitrification can be achieved in WWTPs when the AOB concentration is higher than 10^5 cells/mL (Knapp and Graham 2007; Pickering 2008; Wittebolle et al. 2008) and as such this represents a target level. Given that only aerated sections of the bed will nitrify it is proposed that sequencing the section of the bed that receives air in a pre defined cycle will enable a larger total community to be maintained. By doing this each section of the bed will receive ammonia feed and oxygen at a sufficient frequency to keep the nitrifying population present and active, whilst maintaining low overall ammonium effluents. During periods of elevated load all sections of the bed can be then simultaneously aerated enhancing the capacity of the system and thus improving its resilience. In addition the non-aerated bed will become anoxic such that partial denitrification can be expected, offering the ability to meet total nitrogen in the future. This approach has, in part, been demonstrated at pilot scale by only aerating (continuously) the front portion of a bed (Maltais-Landry et al, 2009).

The second case represents sites where the upfront works was not originally designed to nitrify and/or is potentially undersized. In such situations the system may exhibit reduced robustness as well as poor resilience and consequently it is suggested that this is part of the explanation for the reduced robustness of Site C (Chapter 6) currently being investigated. In these cases, heterotrophic bacteria will predominate due to their faster growth rate congruent with traditional plug flow biofilm based bioreactors such as TFs and SAFs where organic load must be reduced before the nitrifying population will start to proliferate (Boller et al, 1994). Consequently, sizing and aeration rates must be appropriate to such cases to ensure a proportion of the bed can develop and sustain a nitrifying population. In practice this represents sizing such beds at a rate between traditional combined storm and tertiary beds (1 m^2/PE) and those of true secondary beds (5 m^2/PE) as well as increasing the aeration rate to maximise heterotrophic activity.

It is postulated that the situation can be exacerbated in the case of aerated wetlands due to their additional function concerning sludge storage. As sludge builds up in the bed it will exert an oxygen demand and load that will further inhibit the ability of a nitrifying population to develop over and above that seen in the other processes (TFs, nSAFs etc.). Consequently it may be beneficial to alter current solids management strategies to incorporate in-situ management approaches akin to filter flushing as used in tricking filters (Albertson, 1995). This could be relatively easily delivered due to the changing mixing pattern aeration generates when the water level exceeds that of the gravel (see above). A temporary increase in aeration rate coupled with increase flow would enable the bed to be flushed and hence limit the impacts of excessive solids build up. Additionally, above ground biomass is currently retained on the bed, at a rate of approximately 2 cm/year (Dotro et al, 2014) contributing to the above issue and representing a case where reed harvesting at the end of the growing season would be a beneficial maintenance strategy. The harvest material has been demonstrated to have value as a bioenergy crop and hence could generate additional benefit and enhance the overall sustainability of small works.

Further research and experimentation is required to test the validity of these ideas in terms of characterising the existing microbial community along the bed and to determine a sensible cycle time in terms of switching on and off nitrification activity including how long the community can stay dormant before becoming non viable (Chapter 9). Additionally, care is required during switch over to ensure continuous nitrification. Previous investigations on intermittent aeration in wetlands have shown DO residual remain above 1 mgO_2/L for one hour after cessation of aeration with effective nitrification lasting for 90 minutes (Fan et al, 2003). A paucity of information exists concerning restart times after commencement of aeration so that it is unclear whether the available 90 minutes is sufficient such that overlapping of the cycles may be necessary.

Delivery of the suggested cyclic aerating operating approach requires a more flexible aeration system that enables only specific portions of the bed to be aerated at any given time (Figure 7.2). The exact complexity of this depends on the required sequencing frequency but is expected to be able to be delivered manually (during routine site visits) or on a pre-defined time sequence established relative to historical knowledge of the wastewater profiles. In situations where control is required against inherently variable, chaotic loading, then an automatic system could be implemented using on-line ammonia detection to control how many sections of the bed are aerated. This constitutes an equivalence of the IFAS process where aeration rate is controlled to manage nitrification activity within the biofilm contained on the media, linked to ammonia detection (Tordera, 2009).

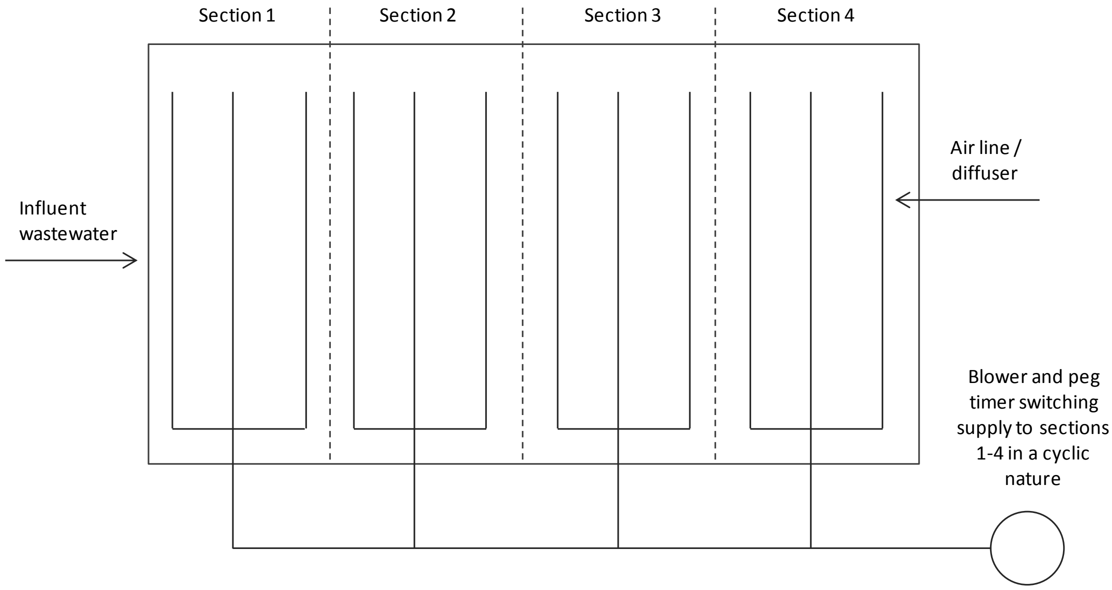

Figure 7.2 Plan of an aerated wetland with sectioned air supply

7.4 References

Albertson O. (1995) Excess biofilm control by distributor-speed modulation. *Journal of Environmental Engineering* 121(4):330-336

Asselin C., Comeau Y., Ton-That Q. (1989) Alpha correction factors for static aerators and fine bubble diffusers used in municipal facultative aerated lagoons. *Water Science and Technology* 38(3):79-85

Barter P., Smith J. (2007) Using tertiary aerated sand filters for ammonia removal. Proceedings of the *European Water and Watstewater management conference* 24th-26th September. Newcastle:UK

Bellucci M., Ofiţeru I., Curtis T., Graham W. (2013) Nitrification in hybrid bioreactors treating simulated domestic wastewater. *Journal of Applied Microbiology* 115(2):621-630

Boller M., Gujer W., Tschui M. (1994) Parameters affecting nitrifying biofilm reactors. *Water Science and Technology* 29(10–11):1–11

Cooper, P. (1993) The use of reed bed systems to treat domestic sewage: The European design and operational guidelines for reed bed treatment systems. In: G.A. Moshiri (Ed.), *Constructed Wetlands for Water Quality Improvement*. Lewis Publishers, Boca Raton, FL, pp. 203-217

Dotro G, Brix H., Wallace S., Nivala J., Comeau Y., Stein O., Rosseau D., Tyrell S., Gowing S., ARM Ltd. (2012) Personal communication

Dotro G, (2014) Personal communication

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

Germain E., Nelles F., Drews A., Pearce P., Kraume M., Reid E., Judd S., Stephenson T. (2007) Biomass effects on oxygen transfer in membrane bioreactors. *Water Research* 41(5):1038-1044

Jespersen D., Sorrell B., Brix H. (1998) Growth and root oxygen release by *Typha latifolia* and its effects on sediment methanogenesis. *Aquatic Botany* 61(3):165-180

Judd S. (2011) *The MBR book (2nd edn).* Butterworth-Heinemann

Knapp C., Graham, D. (2007) Nitrite-oxidizing bacteria guild ecology associated with nitrification failure in a continuous-flow reactor. *FEMS Microbiology Ecology* 62(2):195–201

Mander U., Jenssen P. (2002) *Constructed Wetlands for Waste Water Treatment in Cold Climates.* WIT Press Publishing

Maltais-Landry G., Maranger R., Brisson J. (2009) Effect of artificial aeration and macrophyte species on nitrogen cycling and gas flux in constructed wetlands. *Ecological Engineering* 35(2):221-229

Painmanakul P., Hebrard G. (2008) Effect of different contaminants on the α-factor: Local experimental method and modeling. *Chemical Engineering Research and Design* 86(11):1207-1215

Pearce P. (2013) Sustainable processes to meet low ammonia consents – the small works challenge. Proceedings of the *2nd Conference on low energy wastewater treatment systems*, 24th September. Solihull:UK

Pickering R. (2008) *How hard is the biomass working?* PhD thesis Newcastle University, Newcastle upon Tyne, UK

Rittmann B., Whiteman R. (1994) Bioaugmentation: a coming of age. *Water Quality International* (1):12–6

Satoh H., Okabe S., Yamaguchi Y., Watanabe Y. (2003) Evaluation of the impact of bioaugmentation and biostimulation by in situ hybridization and microelectrode. *Water Research* 37(9):2206-2216

Tordera R. (2009) *Improve the energy efficiency of a full scale integrated fixed film activated sludge (IFAS) process at Thames water oxford WWTW*. MSc thesis Cranfield University, Cranfield, UK

Vymazal J., Kröpfelová L. (2005) Growth of *Phragmites australis* and *Phalaris arundinacea* in constructed wetlands for wastewater treatment in the Czech Republic. *Ecological Engineering* 25(5):606-621

Weedon C. (2014) Yellow *Phragmites*: Significance, Cause, and Remedies. *Sustainable sanitation practice* 18:37-42

Wittebolle L., Boon N., Vanparys B., Heylen K., De Vos P., Verstraete W. (2005) Failure of the ammonia oxidation process in two pharmaceutical wastewater treatment plants is linked to shifts in the bacterial communities. *Journal of Applied Microbiology* 99(5):997–1006

Chapter 8

Conclusions

8.1 Conclusions

The major conclusion is that artificial aeration of horizontal sub-surface flow constructed wetlands (HSSF CWs) is an effective option with regards to enhanced nitrification on tertiary small works sites and provides comparative treatment to higher energy technologies with the benefit of being the ability to retrofit into existing HSSF CWS sites.

Specific conclusions were as follows:

1. A review of the literature related to nitrification potential in existing sub-surface flow constructed wetlands revealed that nitrification rate was commensurate with oxygen availability. Consequently, the sub-surface oxygen deficiency encountered in traditional HSSF CWs limits nitrification unless operated at very low loading rates. Higher nitrification rates are possible when the design of the wetland ensures adequate sub-surface oxygen availability such as in vertical sub-surface flow and artificially aerated CWs. Available data indicates that both configurations are capable of meeting the requirements for tertiary nitrification (Chapter 2) (Objective 1).

2. Artificially aerated HSSF CWs were found to be capable of robustly delivering nitrified effluents sub 3 mgNH_4^+-N/L in both secondary and tertiary treatment applications including combined storm flow beds with the potential to robustly deliver sub 1 mgNH_4^+-N/L in the case of tertiary systems. (Chapter 3, 6) (Objective 2).

3. The operation of artificially aerated HSSF CWs for tertiary nitrification is not limited by the ammonia loading rate and can thus work effectively when receiving high and variable flow rates. No difference was observed compared to a non-aerated bed with respect to removal of $CBOD_5$ and TSS and long term hydraulics of the system such that future design should be based on current sizing based around hydraulic and solids loading rates (Chapter 3, 6) (Objective 2,5).

4. Artificial aeration of tertiary HSSF CWs was shown to robustly deliver nitrification but had limited resilience to shock loads (Chapter 6) (Objective 2). It is suggested that this is commensurate with the systems being able to sustain only limited populations of nitrifiers through either limited available ammonia or out competition from heterotrophic organisms. Accordingly, enhancement could be achieved by increasing the population of active nitrifiers and thereby increasing resilience. This can be achieved by sequencing aeration to sections of the bed (Chapter 7, 8) (Objective 5).

5. The presence of media enhances the oxygen transfer efficiency in a packed system compared to a comparative control through a combination of retarded bubble rise velocities and pseudo stagnant residence within the pore spaces. The enhancement is maximised when the replenishment rate of the pores is decreased and so occurs when small orifice sizes and low air flow rates are utilised (Chapter 4) (Objective 3).

6. The presence of artificial aeration was detrimental to the growth of *Phragmites autralis* and *Typha latifolia* in terms of stunted growth and yellowing of leaves. However, the impact on the absolute growth was less significant with *T. Latifolia*, congruent with its faster growing rate. (Chapter 5) (Objective 4).

7. A combination of the findings from the three years investigation on the impact on the design basis of an aerated HSSF CW resulted in recommendation of orifice sizes below 1.0 mm in a grid 150 mm apart at a flow rate of 0.8 L/min/hole. Planting with *T. Latifolia* is recommended for use in aerated wetlands and the system should be run sub-surface flow for effective treatment (Chapter 7) (Objective 5).

Chapter 9

Further work

9.1 Enhancing the resilience of tertiary nitrification

Findings from the thesis demonstrated the poor response to shock loads of a lowly loaded system, suggested to be due to the limited size and activity of the nitrifying population.

To assess the validity of the proposal (section 7.3) a custom built pilot plant is recommended, fed with synthetic wastewater to enable control of variation and ammonia spikes. The system set-up would allow development of a natural biofilm with the system operating with constant aeration in all sections and ammonia feed around 5 gNH_4^+-N/m^2/d based on the receiving load of successful full-scale sites. Regular monitoring would consist of DO, temperature, pH, COD, ammonium, nitrite and nitrate concentrations in the inlet and effluent and at 0.2 m intervals across the length and breadth of the system. Supporting microbial analyses would be conducted to quantify abundance and activity of ammonia-oxidising and nitrite oxidising-bacteria. This would require collection of the biofilm for DNA extraction and quantitative polymerase chain reaction (qPCR). The increasingly common use of these techniques means target organisms (total bacteria, AOB, *Nitrobacter*, *Nitrospira*) and associated primer/probes and sequences are already known and can be found in the literature (Belluci et al, 2013, Kowalchuk et al, 1997, Graham et al, 2007, Harms et al, 2003). Fluorescence in situ hybridisation (FISH) can then be employed for visualisation of the AOB and NOB and DGGE for diversity analysis. Results can then be compared to studies that have correlated process stability with diversity and composition of the nitrifying community to establish the required levels in aerated HSSF CWs. For example, quantitative studies based on real-time qPCR and FISH demonstrated that efficient nitrification can be achieved in WWTPs when the AOB concentration is higher than 10^5 cells/mL (Coskuner et al. 2005; Graham et al, 2007; Knapp and Graham 2007; Pickering 2008; Wittebolle et al. 2008).

The next phase would test the length and frequency of aeration cycles to establish start up rate linked to the duration of dominancy (when not aerated). Once these had been established the pilot plant should be used to validate the findings and establish a link between shock loading profile (duration and peak concentration) and the amount of active bed required.

In addition, experiments should be undertaken to establish the validity of bioaugmentation as a means of rapid start up and remedial action. Some evidence of the successes of bioaugmentation has been published with regards to nitrification start-up (Sahtoh et al, 2003, Rittmann and Whiteman, 1994) however further investigation is required to build confidence in its ability and reliability. The work would take the form of comparative sequence analysis of 16S rRNA genes and FISH to identify and monitor specific microorganisms in complex communities supported by high spatial resolution microelectrode measurements (Satoh, 2003, Okabe et al, 1999, De Beer et al, 1997). This will enable direct monitoring of whether the initial population dosed onto the bed subsequently proliferates and whether all of the community is equally able to act as a bioaugmentation agent.

9.2 Optimising the design

Process water is known to affect the oxygen transfer rate. A low α of 0.3 was used to take into account the potential impact that enhanced contact time may have on mass transfer in real wastewater through additional accumulation of surface active agents suggested. However direct measurement would enable more accurate predictions and so the mass transfer study is recommended to be repeated with real wastewater as well as site commissioning of the aeration systems.

9.3 References

Bellucci M., Ofiţeru I., Curtis T., Graham W. (2013) Nitrification in hybrid bioreactors treating simulated domestic wastewater. *Journal of Applied Microbiology* 115(2):621-630

Coskuner G., Ballinger S., Davenport R.., Pickering R., Solera R., Head I., Curtis, T. (2005) Agreement between theory and measurement in quantification of ammonia-oxidizing bacteria. *Applied and Environmental Microbiology* 71(10):6325–6334

De Beer D., Schramm A., Santegoeds C., Kuhl M. (1997 A nitrite microsensor for profiling environmental biofilms. Applied Environmental Microbiology 63(3):973-977

Graham D., Knapp C., Van Vleck E., Bloor K., Lane T., Graham, C. (2007) Experimental demonstration of chaotic instability in biological nitrification. *The ISME Journal* 1:385–393

Harms G., Layton A., Dionisi H., Gregory I., Garrett V., Hawkins S., Robinson K., Sayler G. (2003) Real-time PCR quantification of nitrifying bacteria in a municipal wastewater treatment plant. *Environmental Science and Technology* 37(2):343–351

Knapp C., Graham, D. (2007) Nitrite-oxidizing bacteria guild ecology associated with nitrification failure in a continuous-flow reactor. FEMS Microbiology Ecology 62(2):195–201

Kowalchuk G., Stephen J., De Boer W., Prosser J., Embley T., Woldendorp J. (1997) Analysis of ammonia-oxidizing bacteria of the beta subdivision of the class Proteobacteria in coastal sand dunes by denaturing gradient gel electrophoresis and sequencing of PCRamplified 16S ribosomal DNA fragments. *Applied Environmental Microbiology* 63(4):1489-1497

Okabe S, Satoh H, Watanabe Y. (1999) In situ analysis of nitrifying biofilms as determined by in situ hybridization and the use of microelectrodes. *Applied Environmental Microbiology* 65(7):3182–91.

Pickering R. (2008) *How hard is the biomass working?* PhD thesis Newcastle University, Newcastle upon Tyne, UK

Rittmann BE, Whiteman R. Bioaugmentation: a coming of age. Water Quality International 1994(1):12–6

Satoh H., Okabe S., Yamaguchi Y., Watanabe Y. (2003) Evaluation of the impact of bioaugmentation and biostimulation by in situ hybridization and microelectrode. Water Research 37(9):2206-2216

Wittebolle, L., Vervaeren, H., Verstraete, W. and Boon, N. (2008) Quantifying community dynamics of nitrifiers in functionally stable reactors. *Applied and Environmental Microbiology* 74(1):286–293