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Assessing the potential for tertiary nitrification in sub-surface flow constructed wetlands

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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, 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. Constructed wetlands (CWs) offer a low energy alternative to meet these challenges. This review explores oxygen transfer theory; nitrification performance of existing CW systems, and the key affecting factors to be considered when implementing the technology for tertiary treatment upgrades. 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: reed beds; ammonia; wastewater; sewage; review

Introduction

One of the most important challenges facing the management of wastewater discharges around the world concerns the maintenance or improvement of treatment performance, without causing an excessive increase in energy usage or costs. This is perhaps most evident at small works serving less than 2000 population equivalents (PE), as they employ predominately low energy technologies with low operation and maintenance requirements. Whilst initially designed to meet discharge standards on organic matter (5-day carbonaceous biochemical oxygen demand; BOD_5) and suspended solids (TSS), they are increasingly required to deliver higher quality effluents including ammonia concentrations down to discharge levels of $10\,mgNH_4^+\text{-}N\,L^{-1}$ in Austria,[1] $5\,mgNH_4^+\text{-}N\,L^{-1}$ or even $0.5\,mgNH_4^+\text{-}N\,L^{-1}$ in the UK.[2] Accordingly, upgrades have translated into the inclusion of tertiary aerobic biological processes such as submerged aerated filter and trickling filters or replacement of the secondary treatment process, with the potential to switch to activated sludge plants or membrane bioreactors where meeting discharge consents is particularly challenging. This transformation 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 an established low energy technology utilized on small wastewater treatment plants. CWs are traditionally passive 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, reviews of which are available elsewhere.[3,4] The simplest classification of the technology is based on the direction of flow: horizontal flow (HF) or vertical flow (VF) systems, the majority of which are operated as sub-surface systems. Passive HF CWs are typically anoxic to anaerobic whilst VF CWs are operated intermittently to enable aerobic conditions to develop within the bed. Tidal flow and reciprocating wetlands are classifications of flood and drain systems based on the VF 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, by supplying air via the addition of blowers and diffusers placed on the wetland bed.[5–7] In addition to these classifications, numerous integrated or hybrid

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systems have been developed that combine variations of wetland design. Traditionally, VF beds are used for nitrification followed by a horizontal bed for denitrification and solids removal.

The rapidly increasing challenge of achieving complete nitrification at the minimum cost, footprint, energy and carbon emissions represents a new context for application of low energy treatment systems such as CWs. The current paper aims to review the available literature in order to understand nitrification potential in sub-surface flow CWs by assessing performance achieved at laboratory, pilot and full scale systems around the world. The work first presents an overview of oxygen transfer theory and the factors affecting it, which is followed by a review of nitrification performance from the literature and a discussion of the influences affecting nitrification rates. The paper then concludes with a discussion of the outlook and challenges with regard to tertiary nitrification on small sewage treatment works.

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 diffusion 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,[8] whilst the latter is known to be rate limiting once the bulk dissolved oxygen (DO) falls below 2 $mgO_2\,L^{-1}$.[9] The rate and efficiency of oxygen transfer is described in different ways including; the mass of oxygen transferred per unit time (oxygenation capacity, $kgO_2\,h^{-1}$); the percentage of the oxygen transfer compared to that available (oxygen transfer efficiency – OTE), commonly measured per metre of submergence to normalize against different studies; mass of oxygen transfer per unit of energy consumed (oxygenation efficiency – OE) and the aeration efficiency, both measured in $kgO_2\,kWh^{-1}$.

The rate of oxygen transfer is proportional to the area of contact between the liquid and gas phases.[10] Consequently, aerobic processes are designed to maximize this feature in one of two ways: falling films or rising bubbles (Figure 1). 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 1(a)). Typical examples of this technology include trickling filters, rotating biological contactors and VF 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

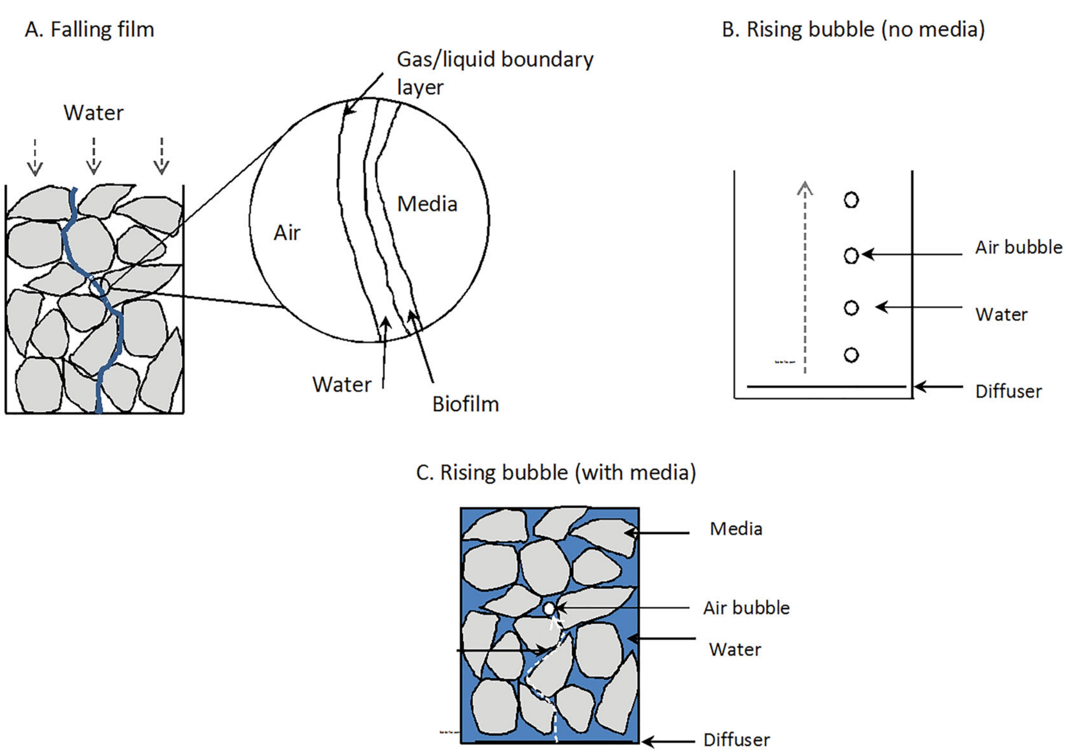

Figure 1. Common methods of air delivery to wastewater: (a) Falling films; (b) rising bubbles (non-media system) and (c) rising bubbles (media system).

effective use.[11] Reported oxygen transfer efficiencies for such systems are in the region of 5% m^{-1}.[12,13]

In contrast, rising bubble systems (Figure 1(b)) 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 sequencing batch reactors as well as biofilm processes such as submerged aerated filters and artificially aerated (AA) 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; [14]) are preferred over coarse bubble (6–10 mm bubble size; [14]) 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 [10] with, for example, typical oxygen transfer efficiencies in the range of 8–12% m^{-1} for activated sludge systems.[13]

The importance of initial bubble size is less clear in fixed-film rising bubble systems (Figure 1(c)) 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.[15,16] 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 process where rates remained around 4–7% m^{-1}.[17] 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.[18] 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.[18]

Oxygen transfer in CWs

In CWs the oxygen consumption rates are commonly based on a mass balance using water quality data, which has led to overestimation of oxygen transfer rates depending on the assumptions made on oxygen required to treat the organic matter and/or nitrogenous compounds.[19] To illustrate, direct measurement of the OTR by gas tracer methods has shown OTRs of around 0.3–3.2 $gO_2\ m^{-2}\cdot d^{-1}$ in HF CWs – lower than previous estimates of 5.8– 22 $gO_2\ m^{-2}\ d^{-1}$ based on mass balances and theoretical calculations.[20] Respirometry techniques have also been adapted from use in activated sludge and applied to measure biological kinetics in VF CW systems.[21] 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 2.5–5 $gO_2\ m^{-3}\ h^{-1}$, which translates to a maximum of 72 $gO_2\ m^{-2}\ d^{-1}$.[21,22] This is significantly higher than the observed chemical oxygen demand (COD) removal rates in VF systems.

Oxygen supply in conventional HF CWs 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 $gO_2\ m^{-2}\ d^{-1}$ [20] compared to required consumption rates of 2.4–11.6 $gO_2\ m^{-2}\ d^{-1}$.[19] Oxygen transfer from plants in excess of plant respiration requirements are uncertain, but considered insignificant.[23,24] Such low oxygen transfer rates lead to a residual DO of around 0.1–0.9 $mgO_2\ L^{-1}$ [25,26]; insufficient for nitrification [9] and consequently, complete nitrification is only considered achievable in lowly loaded systems of up to 2 gNH_4^+-N $m^2\ d^{-1}$.[26]

Vertical

sub-surface flow CWs are falling film systems (Figure 1) reported to consume 5.7–156 $gO_2\ m^{-2}\ d^{-1}$ [19] and maintain a residual DO of 4.3–6.5 $mgO_2\ L^{-1}$.[27,28] Part of the oxygen demand is met via nitrate utilization during the flooded phases that is subsequently released into the flow during the drain phase.[29,30] Effective delivery of the aerobic environments occurs when the sand is not saturated; thus good drainage and distribution is critical.[31] 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.[31] Variation in loading approaches has led to a range of estimated average OTRs between 50 and 90 $gO_2\ m^{-2}\ d^{-1}$.[32] Tidal flow [33] and reciprocating operating strategies [34] are based on VF systems, with several flood and drain cycles occurring daily, designed to enhance oxygen transfer and therefore increase nitrification.[29] To illustrate, a laboratory study run with a 3 h fill : 3 h drain cycle demonstrated that the oxygen demand in the tidal flow system was fully met with OTRs reaching 450 $gO_2\ m^{-2}\ d^{-1}$.[35] The systems have been reported to deliver the required oxygen quickly with saturation of the biofilm occurring in less than 1 minute.[34]

AA systems are fixed-film, rising bubble systems (Figure 1(c)) reported to achieve residual DOs of 3.3–7.0 $mgO_2\ L^{-1}$ [24,36] and meet oxygen demand rates of 50–1027 $gO_2\ m^{-2}\ d^{-1}$.[19] Studies have been increasing in this area since 1999 [37] and generally relate to aerated HF systems, although more recently include flooded VF systems.[38,39] 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 $L\ min^{-1}\ m^{-3}$ bed [5]; a 90 mm slotted PVC pipe across the width of the bed at two locations along a 30 m bed, delivering 4.2 $L\ min^{-1}\ m^{-3}$ of bed [37] 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^{-1} m^{-3} bed 12 h a day.[40] The latter system used an orifice size of 3 mm indicating coarse bubble aeration akin to those used in submerged aerated filters whereas the former systems use fine bubble aeration as used in activated sludge plants.

Pilot investigations into the impact of hole size and air flow rate have revealed low air flow rates (10–20 L min^{-1}) and small hole sizes (0.5–0.8 mm) produced higher SOTEs m^{-1} than high flow rates (40–100 L min^{-1}) and larger hole sizes (2–3 mm).[18] The time required to reach the maximum DO took around 5 min which compared to 20 min reported in a pilot aerated flooded VF systems used for secondary treatment.[41] The system was intermittently aerated at a range of 1 ± 0.5 L min^{-1} m^{-3} and stopped once the DO reached 3.5 mgO_2 L^{-1}. The DO decline took 60 min to drop below 1.0 mgO_2 L^{-1} enabling total nitrification and partial denitrification to occur.

Nitrification performance

Comparison of data from the collated studies of different wetland configurations revealed median nitrification rates of 2.8 gNH_4^+-N m^{-2} d^{-1} for VF, 2.0 gNH_4^+-N m^{-2} d^{-1} for AA systems and 0.5 gNH_4^+-N m^{-2} d^{-1} for the HF, with integrated systems operating at higher nitrification rates of 9.3 gNH_4^+-N m^{-2} d^{-1} (Table A, *supplementary information*). Variation of reported NRs was highest in the integrated (0.4–12.8 gNH_4^+-N m^{-2} d^{-1}) and VF systems (0.1–79.3 gNH_4^+-N m^{-2} d^{-1}) 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 VF 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 NRs for HFs was 0.03–7.2 gNH_4^+-N m^{-2} d^{-1} 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.

Median nitrification rates in secondary treatment systems were higher than those reported from tertiary systems for HF (0.92 vs. 0.25 gNH_4^+-N m^{-2} d^{-1}), AA (1.99 vs. 1.22 gNH_4^+-N m^{-2} d^{-1}) and modified systems (10.8 vs. 2.54 gNH_4^+-N m^{-2} d^{-1}), with only VF systems having a reported lower median NR for tertiary systems (2.84 vs. 5.08 gNH_4^+-N m^{-2} d^{-1}). Although the tertiary systems dataset was significantly smaller than the secondary treatment systems for all wetland types, the fact higher NR were achieved under competitive conditions with heterotrophic organisms (i.e. during secondary wastewater treatment) suggests tertiary wetlands could, in theory, achieve greater nitrification rates if operated differently.

Comparison of the tested systems revealed full scale studies utilizing bed depths of 0.6–0.7 m and areas of 4–10,000 m^2 whilst pilot and laboratory 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 a general underestimation of NRs in small systems compared to full scale (Table 1). For instance, underestimations appeared in the cases of VF (31%) and HF (27%) wetlands although a much closer translation between scales of operation appears to exist in the case of aerated HF systems (17%). However, in the case of integrated systems NRs 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 and laboratory scale integrated systems compared to those at full scale (Table A, *supplementary information*). As such, only

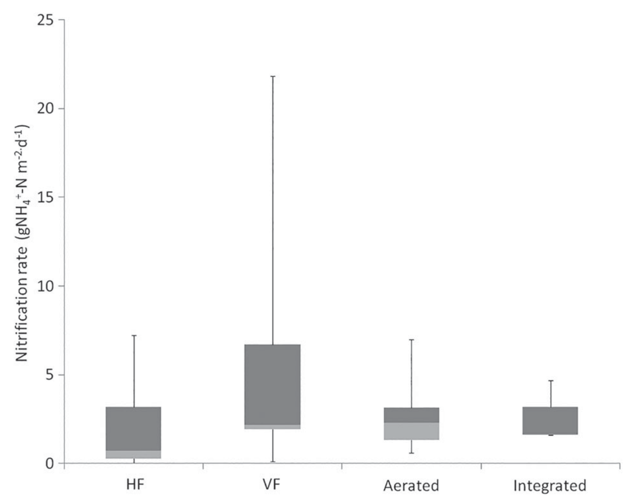

Figure 2. Comparison of nitrification rates calculated for various full scale wetland systems. The ‘box’ represents the 50th percentile range, the line the median and the ‘whiskers’ the upper 5th and 95th percentiles.

Table 1. Comparison of nitrification rates (NR) in various wetland systems at full and pilot scale.

	Full scale		Pilot/lab scale			
System	Number	NR ($g\,m^{-2}\,d^{-1}$)	Number	NR ($g\,m^{-2}$ d)	Total number	ΔNR ($g\,m^{-2}$ d)
HF	16	0.71	15	0.52	31	− 0.19
VF	13	2.19	6	1.52	21	− 0.67
Integrated	3	1.65	12	11.9	15	+ 10.3
Aerated HF	7	2.30	17	1.91	13	− 0.39

data relating to full scale beds and outdoor systems were taken into account for the remainder of the data analyses.

For full scale systems, median nitrification rates of 2.3 gNH_4^+-N m^{-2} d^{-1} for the AA systems; 2.2 gNH_4^+-N m^{-2} d^{-1} for VF and 0.7 gNH_4^+-N m^{-2} d^{-1} for the HF were calculated; with integrated systems decreasing from 9.3 to 1.7 gNH_4^+-N m^{-2} d^{-1} (Figure 2).

Influences on nitrification rates

Loading rates

A strong correlation between NH_4^+-N loading and nitrification rate was found in all systems (Figure 3) in agreement with previous findings showing rates of NH_4^+-N removal increased with mass loading for an HF CW.[42] Strong correlations were also observed for each of the individual configurations; for instance increasing ammonium loading rates between 0.8 and 8.0 gNH_4^+-N m^{-2} d^{-1} corresponded to NRs of 0.7–7.2 gNH_4^+-N $m^{-2}{\cdot}d^{-1}$ ($R^2 = 0.99$) for HF systems older than two years (Figure 3). 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 NR of 1.8 gNH_4^+-N m^{-2} d^{-1} at a loading rate of 7.7 gNH_4^+-N m^{-2} d^{-1}. This suggests that mature HF systems, albeit typically oxygen limited, can deliver nitrification provided they are given sufficient time to establish a nitrifying population. The capacity to nitrify also appears related to hydraulic residence time as a study revealed a reduction in removal between 0% and 91% for a fixed inlet concentration when changing the hydraulic loading rate from 31 to 146 mm d^{-1} corresponding to N loading rates of 1.5 and 6.9 gNH_4^+-N m^{-2} d^{-1}.[43]

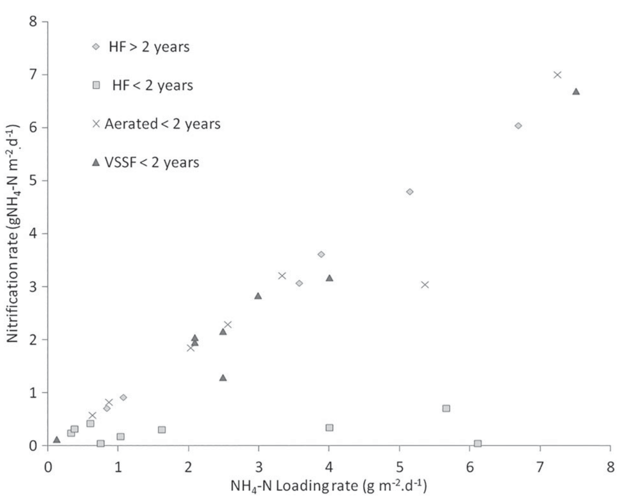

Figure 3. Relationship between nitrification rate and NH_4-N loading rate from long- and short-term performance data from full scale HF CWs, and full scale aerated and VF. All NRs and loading rates relate to data from the full scale studies taken reported in Table 2. HF systems older than 2 years old 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 m^{-2} d^{-1}.

The majority of studies report on CWs used for secondary treatment, whilst full scale tertiary systems appear under-represented in the literature (Table A, *supplementary information*). Typical removal efficiencies in full scale secondary HF CWs varied from 6.5% to 93.4% corresponding to effluent ammonia concentrations predominately above 5 mg L^{-1} with a range between 3 and 61 $mgNH_4^+$-N L^{-1}. In comparison, effluent ammonia in the aerobic secondary CWs was lower and ranged between 1.5 and 12.0 $mgNH_4^+$-N L^{-1} for VF and 1.0–9.5 $mgNH_4^+$-N L^{-1} for AA HF systems, corresponding to removal efficiencies of 51.8–97.5% and 95.3–96.7% respectively.

Lower NRs were reported for full scale tertiary HF CWs (0.05–0.22 gNH_4^+-N m^{-2} d^{-1}) with corresponding hydraulic loading rates of 0.01–0.28 m^3 m^{-2} d^{-1} (Table A, *supplementary information*). 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.[44] Further investigation showed the NH_4^+-N:COD ratio was low (1:9), suggesting poor performance could be due to competition from the faster growing heterotrophic bacteria utilizing available oxygen to degrade organic matter, leaving insufficient oxygen 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 mineralization, immobilization and plant uptake. Equivalent loading rates in tertiary VF and AA HF CWs ranged from between 0.03 and 0.53 m^3 m^{-2} d^{-1} treating ammonia concentrations up to 50.5 $mgNH_4^+$-N L^{-1} resulting in effluent ammonia concentrations of 0.2–29.2 $mgNH_4^+$-N L^{-1} (Table A, *supplementary information*).

Operation

Operational practice has also been shown to influence capacity at laboratory and 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 h fill:24 h drain) VF wetland.[45] Alteration of the dosing frequency has been shown to influence treatment through its impact on hydraulic retention time in the bed.[46] 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.[31] 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.[47]

A study of the effect of the flood: drain ratio on performance of VF 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.[48] The study also documented decreased nitrification rates during a period of high BOD_5 ($330\,g\,m^{-2}\,d^{-1}$) 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 consistently shown 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 [40] recorded 95% ammonium removal efficiency (average loading $81\,gNH_4^+\text{-}N\,d^{-1}$) compared to a yearly average of 32% removal in the system during periods of no aeration (average loading $29\,gNH_4^+\text{-}N\,d^{-1}$). The same has been reported for municipal sewage treatment where a full scale system operating at a loading rate of $5.5\,gNH_4^+\text{-}N\,m^{-2}\,d^{-1}$ enabled 68% removal during aeration compared to 15% without.[37] Further, [5] 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) $gNH_4^+\text{-}N\,m^{-2}\,d^{-1}$. 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 $0.1 \pm 0.05\,mgNH_4^+\text{-}N\,L^{-1}$ in the aerated bed compared to $8.6 \pm 6.4\,mgNH_4^+\text{-}N\,L^{-1}$ in the non-aerated bed.[26]

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 commonplace at small rural works and onsite at single houses in parts of Europe.[49–51] In both cases, vertical or AA horizontal beds are used to ensure sufficient oxygen transfer to drive nitrification. Both types of systems are shown to be able to reduce ammonia to below $5\,mgNH_4^+\text{-}N\,L^{-1}$ at a 95th percentile when operated as a secondary process (Table A, *supplementary information*) unless high hydraulic loading rates ($0.53\,m^3\,m^{-2}\,d^{-1}$) 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.

A relative paucity of data exists for aerobic CWs used for tertiary nitrification but in both VF and AA CWs where studies exist, the data indicate effective treatment. Indeed, data from both real and synthetic trials suggest effluent ammonia concentrations of less than $1\,mgNH_4^+\text{-}N\,L^{-1}$ are possible whilst maintaining treatment levels in terms of solids and BOD_5 compared to non-aerated systems by just aerating standard designs of HF CWs [26] or operating VF CWs in series.[52] 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 HF 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 HF CW.[26] Upgrading with artificial aeration was a more viable option in terms of cost, land and footprint than the traditional options of rotating biological contactors, submerged aerated filters or trickling filters.

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 AA wetlands provide enhanced resilience against cold temperatures compared to high rate equivalents.[2] 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 $5\,mgNH_4^+\text{-}N\,L^{-1}$ and are often substantially lower [26] (Table A, *supplementary information*). This prohibits establishment of large communities of ammonia oxidizing bacteria within the beds such that during periods of increased load, available substrate may exceed the cell specific ammonia-oxidation rate of the community ($4\text{–}10\,fmol\,cell^{-1}\,h^{-1}$, [53]). No direct studies on bacterial abundance or community profile have been reported for aerobic wetlands used for tertiary nitrification systems but investigations on established HF CWs revealed 1–3% of the total community being related to ammonia oxidizing bacteria [54] which increases to around 16% when assessing aerated secondary VF beds.[41] These represent the limits between which tertiary systems will likely sit and research is required to understand how to increase the active ammonia oxidizing bacteria 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.[2]

A recent report based around a 700 population equivalent site revealed the use of a 1.7 kW blower for a tertiary aerated wetland treating $250\,m^3\,d^{-1}$.[2] Whilst this generates a very small energy cost (less than £1000 yr^{-1}

based on UK prices) it represents a high relative daily energy demand per person at around 58 Wh PE^{-1} d^{-1}, comparable to typical levels for activated sludge of 59 Wh PE^{-1} d^{-1}.[2] When all energy use on the site was compared, the aeration system accounted for between 40 and 50% of the total energy demand, with 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 would be required to maintain an adequate DO and so the potential for optimization exists.[18] 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. The equivalent energy use for a conventional VF wetland for this site would be 5 Wh PE^{-1} d^{-1}, using a 4.5 kW pump to deliver 15 batches of three minutes of duration throughout the day.[31] This makes tertiary aerobic wetland systems an attractive, low energy option for polishing effluents. Reduction in the actual energy demand also enhances the opportunities for using localized renewable energy sources providing a route for future off-grid operation of such sites,[55] 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 [56] and enhances hydraulic conductivity during the initial years of operation at full scale.[26] 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.[57] The equivalent information has not been reported for VF 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.

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 HF CWs or increased oxygen transfer through the use of aeration in the form of passive (VF) or artificial (AA HF) 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 is currently limited. Existing sites provide evidence that CWs can be an effective choice, thus, the challenge for growth increasingly relates to comparing whole life costs of 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 vertical flow and aerated horizontal flow CWs as they can potentially deliver the nitrification requirements in a smaller footprint than conventional HSSF systems under a very low energy demand.

Disclosure statement

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Supplementary material

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