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Practice

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- Effective Sanitation in Developing Regions
- Solids Accumulation and Clogging
- Intensified and Modified Wetland Designs
- Constructed Wetlands for Combined Sewer Overflow Treatment
- Microbiology in Treatment Wetlands
- Modelling of Treatment Wetlands
- Yellow Phragmites: Significance, Cause, and Remedies

Outcomes from the UFZ wetland workshop

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Sustainable Sanitation Practice (SSP) aims to make available high quality information on practical experiences with sustainable sanitation systems. For SSP a sanitation system is sustainable when it is not only economically viable, socially acceptable and technically and institutionally appropriate, but it should also protect the environment and the natural resources. SSP is therefore fully in line with SuSanA, the Sustainable Sanitation Alliance (www.susana.org). • SSP targets people that are interested in sustainable sanitation systems and the practical approach to it. • Articles are published after blind review only. • Sustainable Sanitation Practice is published quarterly. It is available for free on www.ecosan.at/ssp.

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Cover Photo / ***Titelbild***

Editorial

From 12-14 June 2013, the Helmholtz Center for Environmental Research (Helmholtz-Zentrum fuer Umweltforschung – UFZ) hosted a three-day workshop in Leipzig, Germany. The purpose of the workshop was to foster discussion and collaboration between research institutions, private sector, and other supporting organizations working in the field of treatment wetlands. More general information on treatment wetlands can be found in Issue 12 of the SSP journal.

The format of the UFZ Wetland Workshop was different from that of a traditional conference. The first and third days were broken into 90-minute blocks of time (eight blocks in total over the two days) where we had the opportunity for a 30-minute presentation followed by a 60-minute discussion session on a specific topic. Each session was chaired by 2 – 3 people who were asked to give an overview of the current state-of-the-art for the specified topic and facilitate fruitful discussion. The second day of the workshop contained technical visits to two UFZ treatment wetland research sites. The first site visit was to Langenreichenbach, a facility which contains 15 small scale wetlands for the treatment of domestic wastewater. The second site visit was the Leuna industrial facility where the UFZ has a pilot-scale industrial treatment wetland. The system at Leuna was designed to effectively treat the local groundwater contaminated with organic compounds (BTEX, MTBE).

The first session of the workshop was dedicated to the topic “Research & Industry”, where successful case studies and “lessons learned” from cooperation between research organizations and industry partners were presented. The second session focused on “Industrial Applications” and included presentations on groundwater remediation and treatment of process water generated by the oil industry. Summaries of the following six sessions on various aspects of treatment wetland research and implementation as well as the report on discussions and observations of yellowing reeds in treatment wetlands are included in Issue 18 of SSP.

Issue 18 of the Sustainable Sanitation Practice (SSP) on the “Outcomes from the UFZ Wetland Workshop” therefore includes 7 contributions:

- Effective Sanitation in Developing Regions
- Solids Accumulation and Clogging
- Intensified and Modified Wetland Designs
- Constructed Wetlands for Combined Sewer Overflow Treatment
- Microbiology in Treatment Wetlands
- Modelling of Treatment Wetlands
- Yellow Phragmites: Significance, cause, and remedies

We would like to mention that for each session pdf-files of the introductory presentations and transcription of discussions are available at the WaterWIKI page of the IWA Specialist Group on “Wetland Systems for Water Pollution Control”:
www.iwawaterwiki.org/xwiki/bin/view/WorkGroup_SG+on+The+Use+of+Macrophytes+in+Water+Pollution+Control/WebHome

SSP is available online from the journal homepage at the EcoSan Club website (www.ecosan.at/SSP) for free. We also invite you to visit SSP and EcoSan Club on facebook (www.facebook.com/SustainableSanitationPractice and www.facebook.com/EcoSanClubAustria, respectively).

Jaime Nivala (Guest editor), Günter Langergraber (Editor)

EcoSan Club Austria (www.ecosan.at/ssp)

Content

Effective Sanitation in Developing Regions

Effective sanitation in developing regions requires a solution that fits the local context.

Authors: Chris Sullivan and Fabio Masi

Abstract

Lack of suitable sanitation infrastructure is having a significant impact on environmental, human health and economic measures. Many different individuals and organisations have tried to respond to this need by implementing different sanitation approaches. Previous attempts to implement large scale centralised infrastructure with little local consultation or consideration of the local conditions has now been largely discarded due to a lack of long term effectiveness.

Practitioners and researchers working in the constructed wetlands field have often identified constructed wetlands and ecotechnologies as appropriate for developing regions due to the fact that they lend themselves to the utilisation of local materials, they have low capital and ongoing costs and can be relatively simple to operate. While these benefits as well as others may exist, it is important that wetland practitioners do not follow the previous model unsuccessful model of implementing technologies without considering the requirements of the local context. A number of different skills sets and experiences were identified by practitioners as requirements for the industry to improve the sustainability of sanitation systems in developing regions.

Introduction

It is commonly accepted that the approach of applying large scale centralised sanitation systems into low income and developing regions is not meeting the needs of the region. Instead the focus of individuals and groups working in these regions is to find approaches that meet the local social, economic and environmental context and can continue to perform appropriately over extended periods of time.

Constructed wetlands have been suggested as an appropriate technology for remote, developing and low cost regions due to perceived benefits around operating simplicity, capital and ongoing costs and energy and chemical demands (e.g. Kivaisi, 2001; Whitney et al., 2003; Zhang et al, 2012).

While these advantages may be offered by constructed wetlands it is important that as designers and practitioners we do not revert to the previous model of delivering solutions based on pre-conceived ideas. We must retain flexibility in our responses and access or develop skills in a number of different areas in order to maximise the sustainability of sanitation systems in developing regions.

Current background / status

Individuals and organisations working in the sanitation sector in developing regions come from a number of different backgrounds. These differing backgrounds can tend to lead to design bias based on previous experiences and beliefs. This may also be the case for practitioners working in the constructed wetland field. As a result it is important to identify and find ways to acquire the missing skills required for the identification and implementation of appropriate sanitation responses in developing regions.

An initial assessment of the potential skills sets and considerations required for the implementation of sanitation systems in developing regions. The skills and items requiring consideration have been divided into 5 different groups (Figure 1): Environment, Social, Institutional, Technical and Economic/Financial.

Main outcomes of the session:

- The difficulty of providing sustainable sanitation solutions in developing, low income and remote areas was identified; and
- There was strong recognition of the need to include a number of different skill sets from social, economic, technical and environmental backgrounds. It was also noted that it is very difficult to access these skill sets on small short term projects.

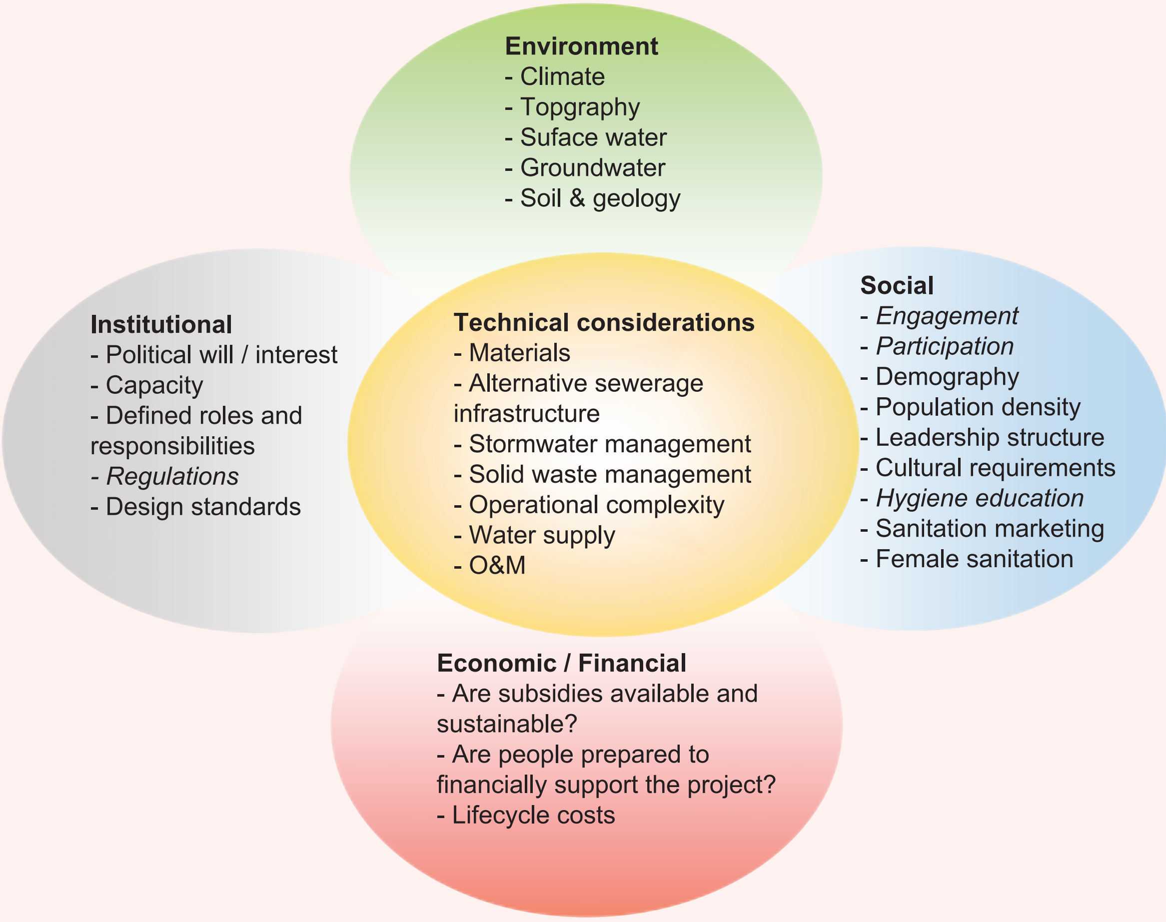

Figure 1 Potential considerations for the implementation of sanitation systems in developing regions Challenges / opportunities

Results

During the session participants were asked to identify the challenges, opportunities and major considerations for effective implementation of sanitation approaches in developing regions. This information is presented below as presented by the 6 Groups.

Group 1

Participation and communication with end users
Awareness of environmental and health risks
Legislation and sanitation planning
Cultural factors
Costs (investment and running costs)

Group 2

Lack of sufficient background data for design and implementation
Understanding local needs, conditions and the historical perspective
Do local regulations exist and are they appropriate?
Operation and maintenance
  Who is responsible
  Who will pay?
  Training
  Local political framework
How to make systems financially self sustainable?
  Financing models
  Incentives via reuse of effluent

Group 3

Technical
Infrastructure – Distances and lack of materials
  Different conditions – Topography and climate
  Maintenance
  Missing data
Social / Cultural
  Communication
  Lack of scientific knowledge
  Bad reputation of wetland systems
  Corruption
  Political situation
  Over expectation of the system
Economic
  Lack of incentives
  Sustainability
  Cost

Group 4

How to build local capacity?
What are the 'columns' for implementation?
Technology
Economy
Capacity development
How cheap can technologies be without loosing treatment efficiency?
How to get the big investment / regional scale projects.

Group 5

Pre-consultation with local community – do you have an additional budget? Who pays?
Long term operation, who is paying?

Group 6

What systems are acceptable to the local community?
Build consensus.
What materials are available? What can be used?
Operation and maintenance
Who is doing the work? Who is paying for it?
Knowledge transfer and capacity building.

Conclusion

The discussion associated with Session 3 identified a number of different questions, issues and challenges. Social and economic sustainability were sighted by several different groups as significant issues associated with the implementation of sanitation systems. The need to engage with the community early in the process to ensure that solutions will meet the needs of the local context was identified. This includes meeting local regulations as well as gaining an understanding of environmental and social conditions.

Participants also identified the need to utilise local materials in the construction of appropriate technologies. Questions around the financing of ongoing costs were also raised on several occasions along with concerns over the local government and non government organisations capacity to maintain and support treatment approaches over the long term.

As practitioners working in remote, low income and developing regions it is important that these skills are recognised in project development and involved as soon as possible in project design.

As a group the requirement for a number of different skills was identified and one of the major challenges presenting the industry is finding a way to access and integrate these skills into our projects.

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Solids Accumulation and Clogging

Solids accumulation (and production) occurs in all types of treatment wetlands and with time will lead to media clogging; the time it takes to clog depends on design and operation practises, with the effects of clogging will be more severe in systems where oxygen transfer is essential for their functioning.

Authors: Gabriela Dotro and Florent Chazarenc

Abstract

Solids accumulation and clogging are known processes in constructed treatment wetlands, which can be mitigated by organic and solids loading management and resting periods. Unless there is solids withdrawal, the inorganic and refractory matter stored within the wetland will result in clogging, regardless of treatment type.

In French vertical flow systems, a freeboard enables the accumulation of solids for up to 10-15 years and this surface layer plays an important role in the treatment performance of mature systems. Indeed, temporary surface ponding in the first stage is critical for the distribution of flow and load across the surface of the bed. In horizontal flow wetlands clogging results in overland flow but not necessarily in changes in the effluent quality.

Clogging has detrimental effects on systems that depend on passive oxygen transfer for their functioning, such as secondary vertical flow wetlands. Where claims of "no clogging" have been made, these are typically because not enough time has elapsed to match the storage capacity of the system (e.g., due to very low loading rates) or other mechanisms of solids release have occurred (e.g., periodic loss of solids immediately following resting periods in vertical flow systems). Wetland failure due to clogging is rare as most designers understand the balance to be struck between loading rates, solids management, and overall asset management.

Main outcomes of the session:

- There are examples of clogging of wetland systems from across the world, but these represent a very small percentage of all systems built
- French first stage systems are designed to "pond", which does not mean they are clogged or incorrectly operated or designed. The accumulation of solids is key in distributing the pollutant load across the surface of the bed and as this surface layer matures, the system performs better. This works because of the resting periods built in between feeding stages.
- There are instances where "sacrificial" horizontal flow beds have been designed and used to deal with high solids loading and high hydraulic loading rates, where refurbishing the HSSF wetland every 10 – 15 years is more economical than building the beds bigger to last longer
- The main cause of clogging in secondary vertical flow wetlands (generally filled with fine materials such as sands 0.1-3 mm) is typically poor solids management in the primary stage. These can include lack of routine emptying of septic tanks, poor septic tank design, and/or solids flushing from the upstream process. Clogging has a negative impact on these systems as it restricts oxygen transfer in a process designed to be aerobic to meet its treatment objectives.
- Clogging in systems that rely on chemical precipitation for functioning is also problematic and will require intervention. Design criteria for these are being developed.
- There is limited full scale data of the influent characteristics to the wetlands before they clog; the practitioner is usually called to assist after the systems have clogged and with limited information on what caused clogging.

Introduction

Clogging development in subsurface flow wetlands is the result of physical (settling, filtration), chemical (precipitation), and biological (plant detritus and biofilm accretion) processes within the treatment system. These can be affected by design, operation and maintenance practices, such as pore size of the selected media when compared against loading rates applied to the bed or degradability of the solids under the conditions within the bed. When retention rates exceed degradation rates, solids accumulate both on top of the surface of the bed and within the bed media, reducing infiltration rates and eventually leading to permanent ponding on top of the bed. Further information on the contributing factors to clogging can be found in the review articles of Knowles et al. (2011) and Nivala et al. (2012). In this session, whilst types of clogging were discussed, the focus was primarily on first stage vertical flow beds in French-type systems (VF CWs; Chazarenc and Merlin, 2005) and tertiary horizontal flow beds (HSSF) used in the UK for domestic wastewater treatment. The rationale for this is that both types of systems have similar media (gravel), and receive similar hydraulic loading rates and solids loading rates (Knowles et al., 2011), about 50 g TSS m^2/d for the 1st stage of the French VF CWs (considering the entire surface of the first stage, it generally represents 150 g TSS m^2/d on the fed bed) and up to 35 g TSS m^2/d in HSSF wetlands.

The available quantification tools, remediation strategies and their cost comparison are provided in Nivala et al. (2012). Briefly, options to quantify clogging include on-site permeability tests for horizontal flow wetlands and drain tests for vertical flow systems, plus the conventional solids characterisation both as surface sludge and sludge within the bed. To manage clogging on a proactive basis, the only strategies available are either lowering the loading rates or managing the resting or recovery periods for the beds. In practice, all of these are mitigation strategies and, unless a periodic removal of solids occurs either intentionally or unintentionally, the system will clog as it matures. The balance is then in designing a system that can cope with the loading rates, retains solids within the bed, and is the lowest whole life cost when considering the frequency needed for refurbishment or sludge removal.

Solids release from subsurface wetlands can occur under: (a) high flow conditions, when an overflow system is in place (Figure 1), (b) immediately following a drying or rest period on the first stage of a French system (Molle, 2003) , and (c) when operating a horizontal subsurface flow system with surface flow, as it enables less dense particles to travel through the top of the bed until they reach the effluent point, where it is only filtered by a thin column of gravel (Dotro et. al., unpublished data). The extent of solids loss through these routes remains to be quantified.

Accumulation of water on top of subsurface flow wetlands is only problematic for some HSSF, as it can enable solids carryover across the length of the bed, and in VF wetlands if ponding remains between batches. Temporary surface water accumulation in VF CWs is part of normal operation (Figure 2) and, as such, fundamentally different from clogging. Surface water in HSSF can also occur due to poor operational practices where water level is poorly regulated (i.e., kept above the surface on purpose) and in some cases, unrelated to clogging as well. In both of these cases, there were only positive effects (VF CWs) or no effect (HSSF) on overall performance of the beds.

Figure 1: Example of subsurface flow wetlands with surface flow during high flow conditions. The water flows preferentially over the top of the bed and will result in partially treated water exiting through the overflow pipe and blending with the treated effluent at the final collection point.

Figure 2: Example of 1st stage VF Wetland with ponding during feeding. The water flow preferentially over the top of the bed around inlet points and the percolates.

Experiences with Positive and No Treatment Impact

In French VFCWs, solid accumulation expressed in terms of dry matter accumulation rate can range between 5 -10 kg DM/m^2/year on the surface of the first stage (Prigent et al., 2013; Chazarenc and Merlin, 2005). The accumulation rate is variable and linked to several operating and seasonal parameters, with the amount of sludge accumulated being higher during fall and summer than in winter time (Chazarenc and Merlin, 2005). The quantity of solids accumulated is more homogenous in older system with a surface layer covering the entire beds 2-3 year after the commission period (Figure 3), which results in improved treatment performance in terms of organic matter and ammonia removal.

An example of ponding on secondary horizontal flow wetlands with no impact on treatment performance was reported by Nivala and Rousseau (2009) from the USA, showing persistent ponding in the inlet area four years after installation. The wetland was designed to treat the wastewater from a residential development to produce an effluent compliant with BOD and TSS concentrations of 30 mg/L each, prior to discharge into soil infiltration systems. The treated water exiting the wetland was well below the consented values, with effluent BOD and TSS at <5 mgO_2/l and <15 mg/l, respectively. However, areas of open water where the water is partially treated sewage were considered a health risk, which meant the system required intervention to rectify this clogging.

There are over 640 tertiary horizontal flow wetlands in the UK owned and operated by Severn Trent Water, a major utility company which pioneered this particular use of the technology. The main purpose of the tertiary beds is to trap any residual solids and particulate organic matter remaining from upstream settling and biofilm processes, and provide flow attenuation during storm events. As such, the wetlands are designed to operate as subsurface flow but have 25 cm on top of the gravel

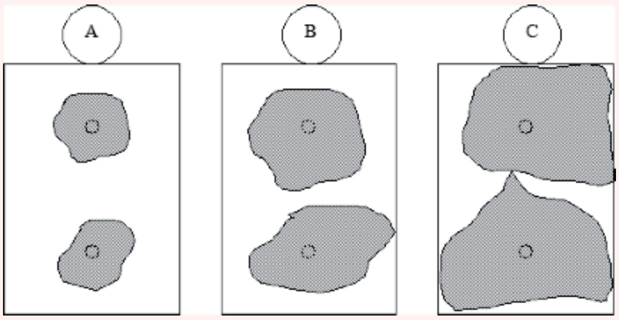

Figure 3: Plan view of surface layer area evolution in a bed of Glandieu during 12 day assay. Circles represent feeding points, from no surface layer at day 0 to (a) day 2; (b) day 4; and (c) day 12. Adapted from Chazarenc and Merlin (2005).

for additional storage of water and an additional 25 cm above that before reaching the top end of the bed (Knowles et al., 2011). The rationale is that, in the event of extreme flows (or severe clogging), partially treated water will bypass the wetland system and will blend with the tertiary treated effluent at the wetland exit point. Because the systems are for a tertiary application, the hydraulic loading rates can be high ranging from 0.2 to 0.8 m/d (Knowles et al., 2011). This, combined with the surface feeding strategy favoured by UK operators results in surface sludge accumulation over the top of the gravel, which will restrict permeability and eventually lead to ponding. One such case was observed by the authors at a small sewage works serving 13,700 pe, and fed with effluent from a trickling filter. The site was designed to meet a consent of 20, 25, and 5 mg/L of BOD, TSS and ammonia, respectively. The wetlands, however, were severely undersized, as they were based on available land and influent biochemical oxygen demand (BOD) concentrations rather than hydraulic load. When the site was visited in early 2013, after five years of operation, the three wetlands were fully flooded to the extent that they were hydraulically linked (Figure 4). The effluent quality, however, was again well below the consent discharging < 2, 4, and 1 mg/L for BOD, suspended solids, and ammonia representing 75%, 80% and 95% removal

Figure 4: Examples of clogged horizontal flow wetlands (a) Hydraulically linked UK tertiary wetland cells and (b) Surface ponding and hydrogen sulfide generation in secondary wetland in Italy. Picture (b) courtesy of IRIDRA Srl.

efficiencies, respectively. Like in the USA example, the site was refurbished due to health and safety concerns, that is, to ensure the water remained within the wetlands rather than due to deterioration in treatment performance.

Two examples of clogging of horizontal flow wetlands were supplied by IRIDRA Srl. In their 20 years of experience, these are the only systems that have resulted in clogging and required intervention. The first one, reported by Masi et al. (2013), consists of a system located in the municipality of Dicomano, Italy, and has been in operation for over 12 years. It comprises a primary settling stage in the form of an Inhoff tank, followed by a horizontal flow wetland (secondary treatment), a vertical flow wetland (tertiary treatment), another horizontal flow system and a final stage with a surface flow wetland for polishing and habitat creation purposes, for a total surface area of 6,080 m^2 for 3,500 pe. The secondary HF wetland was designed as a sacrificial bed, designed to trap the majority of the solids exiting the primary treatment stage and protect the VF wetland where most of the organics and ammonia removal would take place. The wetland started to show ponding in the first 3 meters of the influent after 4 years, and within 9 years ponding covered half the surface of the bed, which had a distinct white colouring (sulphide production). The hydraulic loading rate used in this secondary HF was 0.52 m/d with an organic loading rate of 0.105 kgCOD/m^2/d, which was an order of magnitude greater than the organic loading applied to the subsequent VF wetland. There was no deterioration in effluent quality from the overall treatment flowsheet but there were concerns of odour production once new housing was developed within 200 m of the treatment plant (F. Masi, pers comm.). The use of “sacrificial” HSSF wetlands, i.e., systems that will require solids removal or refurbishing within 8 – 10 years of operation, has been shown to be more cost-effective in some cases, such as this case study, and in tertiary HSSF wetlands in small treatment works in the UK (Figure 5).

A second example of a multi-stage or hybrid system that resulted in clogging of secondary horizontal flow wetlands was reported by IRIDRA during the workshop. The system was designed to treat 35 m^3/d of winery wastewater in 2001 but by 2007, the production at the winery had increased and was resulting in 100 m^3/d of water to be treated by the wetlands. The system comprised an Imhoff tank, followed by a horizontal flow wetland and a surface flow wetland. Ponding in the HF wetland started to appear in the middle of the bed and reed growth shifted so that they only grew in the perimeter (Figure 6). Sulphide formation was evidenced by a white colour in the wastewater inside the HF bed and exiting the bed, with odour being generated. The treatment performance was, again, within the consents in spite of the system overload but the smell issues resulted in the entire facility being re-built in 2009. The treatment flowsheet now comprises an equalisation tank, a French type first stage VF, followed by a refurbished HF wetland, a refurbished surface flow wetland and a final filter stage.

Experiences with Negative Impact on Treatment

Examples of wetlands that have suffered from clogging and this has impacted effluent water quality are rare. The workshop was a good place for practitioners to share their experiences and as such, a total of six systems with data and two anecdotical reports were compiled. In general, the issues were associated with wetlands that relied on oxygen transfer for achieving their treatment objectives where surface clogging developed and prohibited this. This could be the result of hydraulic, organic or solids overloads; uneven flow and load distribution; and media that did not meet the design specifications when the systems were actually constructed. As a result, the clogging in these wetlands impacted their treatment ability and resulted in the necesary interventions to rectify the issues. An additional special case of clogging

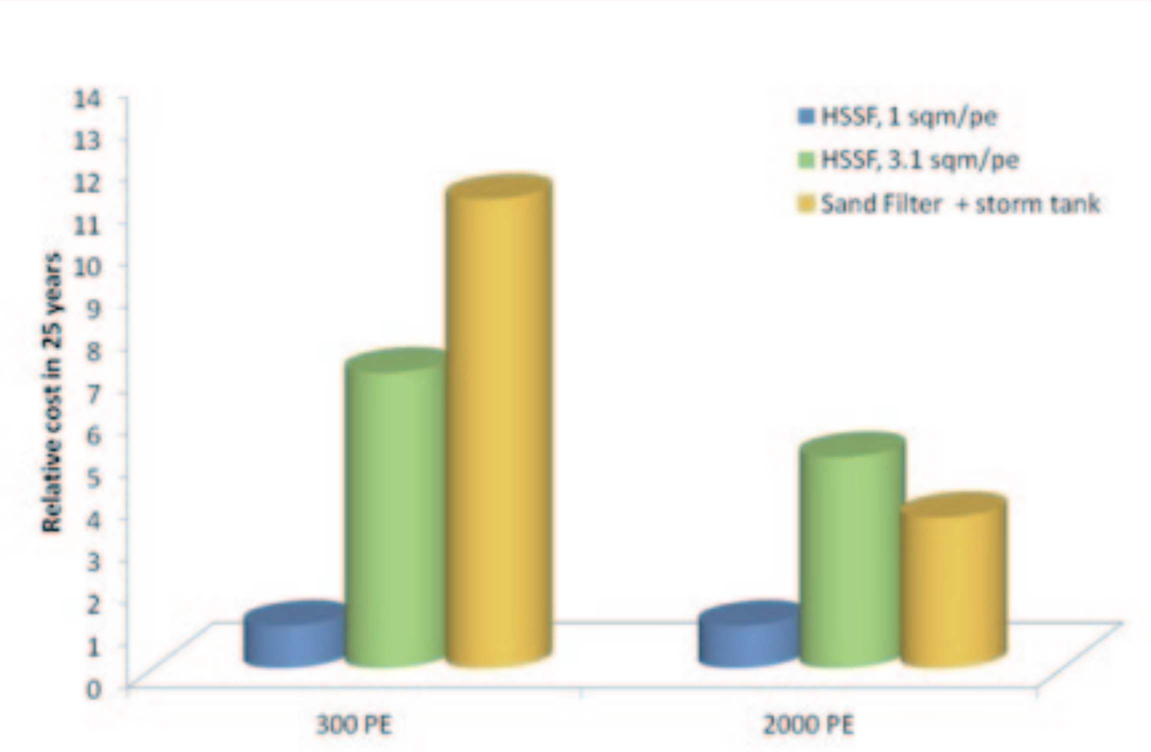

Figure 5: Relative cost comparison for a conventional tertiary HSSF sized at 1 m²/pe (assumed asset life is 8 years), a wetland designed at 3.1 m²/pe (designed for 18 years) and a conventional sand filter plus storm tank for an example small works serving 300 pe and 2000 pe. The difference between the two wetlands are the refurbishment intervals vs capital cost of a bigger wetland. All costs calculated based on Severn Trent Water's standards for 2010-2015. Adapted from Dotro et al. (2012).

Figure 6: Winery horizontal flow wetland (a) before and (b) after clogging developed. The loading rate onto the wetlands was three times the design load by the time clogging became permanent. Images courtesy of IRIDRA Srl.

ocurs in systems where chemical reactions are the main mechanism for the removal of phosphorus. Examples of clogging were shared where steel slag resulted in clogging at laboratory scale (France) and mixing of iron particules with the sand in a VF wetland resulted in clogging at full scale (Belgium).

From over 2,000 VF CWs in France used for domestic wastewater treatment, there have only been a few reports on clogging and hydraulic failures of the first stage beds. However, several cases can be highlighted, as the main indicator of a clogging of the first stage is the permanent ponding leading to reduced conditions and plants die-off (Figure 7).

The bed fed in the first stage can receive up to 300 g COD/m²/d during one week, before 2 weeks of rest enabling the accumulated solids to be mineralized. When the amount of COD and TSS is above this load, the risk of clogging is higher especially during non-vegetative seasons. If a system is properly designed (Molle et al., 2005) clogging appears mostly as a consequence of inappropriate operating conditions. In most of the cases, the COD overload can be linked to sewer cleaning operations, unusual rainfall patterns (very low flows followed by very high flows, leadings to release of accumulated solids), or the discharge of low biodegradability compounds (xenobiotics, mineral oils, etc).

Figure 7: Example of 1st stage clogging in a VF CWs system.

Second stage clogging in the French VFCW system is more frequent as finer material are employed (coarse sand). The failure are mostly linked to problems including poor treatment in the first stage which bring more solids on the second stage, poor distribution and drainage systems, and poor organic solids degradation. These are similar to issues encountered in conventional secondary VF CWs and present a problem as they limit oxygen transfer into the bed matrix.

A secondary vertical flow wetland that clogged and resulted in deterioration of the effluent quality was reported by Nivala and Rousseau (2009), in Geel, Belgium. The system was built in 1996 on a milk farm and was designed to treat a mixture of primary settled domestic wastewater and settled rinse water from the milking parlour. With time, the farm operation grew and so did the load onto the beds. The wetland received, in addition to the design loads, a number of shock loads coming from the milking operations during its history. This resulted in permanent ponding on the surface of the VF wetland, which did not percolate between feeding batches. When the site was visited in 2007, there were between 2 and 8 cm of sludge on top of the media and permanent ponding. As a result, the effluent quality markedly deteriorated, moving from the normal average effluent BOD, COD, TSS and NH_4-N of 5, 41, 6 and 1 mg/L to 865, 1200, 105 and 62 mg/L, respectively. The clogging was so severe and the wetland so undersized under the new farm operation that, in the end, the system was abandoned and a new, larger wetland was built instead. The main factor believed to have influenced effluent quality was the reduced oxygenation of the bed matrix, thus shifting the microbial metabolism from aerobic to anaerobic, which meant slower degradation rates and an inability to treat the waste under the loading rates applied with the area and design configuration available.

Two examples of VF systems that clogged as a direct result of inadequate solids handling in the primary treatment and incorrect media used in the construction of the wetlands were reported by Kilian Water, from Denmark. They reported that out of 450 secondary VF systems designed for onsite tretment two presented clogging. The first system had an old septic tank which allowed too many solids to pass through to the VF bed, which, in addition, had sand that was too fine (i.e., non compliant with the Danish guidelines published by Brix and Arias in 2005) and uneven flow distribution. The combination of sand with low permeability and organic and solids loading in excess of the assumptions made for the design resulted in premature clogging of the surface of the VF wetland. In the second system, the effluent from a septic tank was connected to the VF wetland through a 160 mm pipe without slope and a tipping bucket. When the pipe was full, the tipping bucket discharged a pulse of septic tank effluent onto the VF bed. Unfortunately, this arrangement also meant that solids would accumulate in the pipe and be washed out in the tipping bucket pulse directly onto the surface of the VF wetland. This accumulation of solids on the surface was unplanned in the design, and so oxygen transfer was insufficient to provide the treatment quality required of the system with the designed footprint.

A separate case of clogging due to accumulation of solids from chemical reactions (i.e., precipitates) is found in some wetlands used for phosphorus removal such as slag filters or VF wetlands where the sand has been mixed reactive media (slag or iron). In the case of slag filters, the effect of calcium phosphate precipitates was assessed in column experiments. These showed clogging was mostly linked to the reduction of void space and an increase of dispersion within the filter; however, the full scale slag filters presented normal conductivities (i.e., no clogging) in the first two years of operation (Barca et al., 2013). In terms of iron-containing media, an example was reported at the workshop by Rietland, from Belgium. When phosphorus removal is required, a layer of approximately 30 cm of sand is mixed with iron scaling from steel manufacturing. When the ratio of iron scaling to sand by mass is less than 5%, the systems have presented no issues; if the ratio is above this value, or if the sand is not throughouly mixed with the iron media, clogging can develop. In this case, clogging is a result of high concentrations of iron hydroxides which created a crust that encouraged the accumulation of organic matter (black sludge) and lead to ponding. The issue was rectified by breaking the impermeable layer with an excavator and re-mixing (Figure 8).

Opportunities

A few items were highlighted during the workshop that offer opportunities for future research:

- More cost-effective solutions to clogging management than digging out the sludge layer every decade or so when the surface of the wetland ponds permanently and negatively impacts treatment performance.

Figure 8: Chemical precipitation within wetland media leading to clogging (a) sand and iron clusters from a VF wetland, (b) media cementation from a steel slag pilot wetland in the UK. Images courtesy of (a) D. Van Oirschot and (b) Flint Walters.

- Better design/ failsafe for oxygen input to the - bed when the system clogs
- Resting periods in horizontal flow beds

Earthworms

- Better records of operating conditions that could have lead to clogging.
- Improved design (and proof) for ventilation pipes
- Simple indicators of early signs of clogging to enable operators to report issues at an early stage.
- Design criteria for wetlands with reactive media to ensure good treatment performance and permeability is maintained for the life of the asset.

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Intensified and Modified Wetland Designs

This paper summarizes recent developments in intensified and modified treatment wetland designs, with specific examples from France, the UK, and Germany.

Authors: Jaime Nivala, Clodagh Murphy, Stéphane Troesch, Scott Wallace, Dirk Esser

Abstract

This paper summarizes recent developments in intensified and modified treatment wetland designs, giving a brief overview of the current status of the technological advancements, with experiences from both the private consulting and research sectors. Current challenges are also discussed, such as optimizing treatment performance, and accurately estimating energy consumption. Finally, a summary of the discussion session is presented, which includes: global nitrogen cycling, the necessity for plants in intensified wetlands, and a surprisingly widespread observation of yellowing Phragmites in systems treating highly-nitrified effluents.

Introduction

Standard constructed wetland designs have been well-established for decades and successfully implemented throughout the world (Vymazal & Kröpfelová 2008; Kadlec & Wallace 2009) As the use of treatment wetlands has become more widespread, alternative designs have been developed in order to overcome the limitations of standard designs. Many of the advancements in treatment wetland design originate from consultants working in the private sector. Because economics and treatment efficiency are highly important in the private sector, modifications tend to be funded through the internal research and development efforts of private

Main outcomes of the session :

- There is overall interest in the mechanisms of nitrogen removal in intensified wetland systems.
  - Questions were raised about gaseous (N_2O) emissions from aerated wetlands, which is currently being investigated by Gabriela Dotro (Cranfield University, UK).
  - The extent that alternate nitrogen removal pathways, such as ANAMMOX, play a role in nitrogen removal in these systems is largely unknown.
- When using an industrial by-product as wetland media, with an anticipated end-use as a fertilizer (such as steel slag), care must be taken to ensure that it meets the allowable limits for soil application. This is important to characterize because slags vary greatly between producers.
  - A quick test that can be performed on large piles of slag should be developed, in order to ensure that all slag used in a wetland is suitable for soil application.
  - If the material is not suitable for soil application (due to metal content higher than what is allowed for agricultural reuse, for example) the slag could be repurposed for use in road construction.
- Yellowing Phragmites has been observed in many wetland treatment systems, typically in tertiary treatment wetlands, or wetlands with recirculation or active aeration. Observations from Canada, Denmark, France, Germany, and the UK, and other countries were reported.
  - For consultants, clients think yellowing plants look bad, even if the treatment system is performing well.
  - Reasons for yellowing may be due to iron deficiency and lack of preferred form of nitrogen (NH_4-N).
  - The use of other plants such as *Phalaris* and Iris, which might stay green under such conditions, is being investigated.
  - Further detail on this topic is presented in a separate report in this issue of SSP Journal.

companies and quickly implemented upon proof of concept (Nivala et al. 2013). One recurring theme in the development of intensified and modified treatment wetland designs is the overarching aim to improve subsurface oxygen availability – and thus, better removal of compounds such as ammonium nitrogen and carbonaceous compounds.

Current background / status

Common design modifications include the use of multi-stage or hybrid wetlands (Molle et al. 2008), recirculation (Gross et al. 2008; Troesch & Esser 2012, Prost-Boucle & Molle 2012), batch loading (Stein et al. 2003; Corzo et al. 2008) or tidal flow operation (Behrends 1999; Wu et al. 2011), and/or active aeration of the water column (Wallace 2001; Ouellet-Plamondon et al. 2006; Murphy & Cooper 2011). While most modifications involve some incremental increase in energy input to the treatment system, some modifications can be implemented without external energy inputs if the site topography is favourable (Austin & Nivala 2009). Another vein of intensified and modified wetland designs focusses on the use of active filter materials, which through their physico-chemical properties can increase removal of pollutants such as phosphorus (Molle et al. 2005; Vohla et al. 2011), ammonium (Austin 2006), and/or heavy metals (Sheoran & Sheoran 2006).

Experiences / examples

Experiences from Epur Nature (France), ARM (UK), and UFZ (Germany) were presented and at the workshop.

France

The standard French design for treating raw wastewater consists of a first stage of three alternately loaded, gravel-filled vertical flow beds and a second stage of two sand-filled vertical flow beds (Troesch and Esser, 2012). The system is capable of high levels of nitrification, is able to accept a relatively high hydraulic load, and also copes well with seasonal load variations. However, this standard design requires approximately 3 – 5 m^2/PE. Other challenges have been encountered as well, such as availability of a suitable sand substrate for the second stage beds. As a result, the standard French design is not always economically competitive. Recent work at Epur Nature (France) has investigated ways to decrease system footprint and improve ammonia removal at the same time. This has been achieved through the development of a single-stage recirculating wetland. Recirculation has proven to increase nitrification up to a recirculation rate of 100%. A decrease in nitrification was observed for recirculation rates higher than 100%.

A second design that has both unsaturated and saturated zones in one stage has also been developed in France (Figure 1). This approach includes a deeper bed depth, in order to “stack” the unsaturated zone (100 cm) on top of the saturated zone (40 – 60 cm deep). At the interface between the unsaturated and saturated zone are aeration pipes that facilitate transfer of oxygen to the subsurface. This “stacked” French design has shown high removal efficiencies for COD, BOD_5, and TKN. which makes it able to guarantee an outlet limit of 70 / 15 / 15 /25 mg/l of COD / BOD_5 / SS / KN respectively and reduces the global footprint.

A third design that has been developed by Epur Nature (France) consists of a first stage wetland followed by a trickling filter and settling zone. This combination of technologies has also resulted in high levels of treatment performance and reduced costs compared to the standard French design. Table 1 summarizes area requirements, costs, and outlet TKN concentrations for the standard French system compared to the “stacked” design and wetland-trickling filter combination design.

Investigations into phosphorus removal have also been conducted in France, specifically, the use of alternate media, such as apatite or slag. The use of a phosphorus-sorbing media offers an alternative to the classical approach, which involves the use of chemicals (FeCl), dosing and mixing devices, and sludge management.

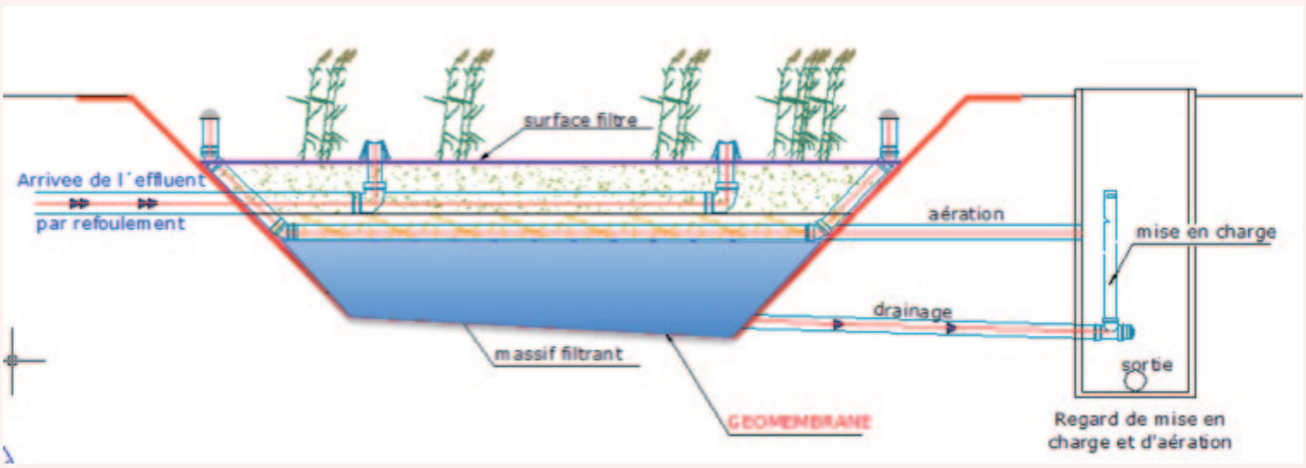

Figure 1: Modified French wetland design with unsaturated and saturated zones (Epur Nature).

Table 1: Area requirements, costs and expected effluent concentrations for various French treatment wetland designs. Data for a 1000 PE capacity (1 PE= 150L/PE.d and 120 g COD/PE.d).

	Standard Design	Stacked Design	Wetland + Trickling Filter	Recirculating Design
Global Footprint (m^2/PE)	4 – 5	1.5 – 2	1.5 – 2.5	1.5 – 2.5
Investment Cost (€/PE)	450 – 550	350 – 380	370 – 400	300 – 350
Operating Cost (€/PE.year)	6 – 9	6.5	6.5	6
Energy Cost (€/PE.year)	0 – 0.3	0.3	0.5	0.3
TKN Effluent Concentration (mg/L)	10 – 20 (or less)	25	15	25
TN Effluent Concentration (mg/L)	-	50	-	-

United Kingdom

In the UK, one of the main drivers behind the development and implementation of intensified wetland systems is the increasingly strict discharge consents. Many community wastewater treatment systems (less than 10,000 PE) in the UK must meet an effluent ammonium-nitrogen concentration of less than 5 mg/L. In the next AMP phase (2015 – 2020), the number of additional treatment systems that will be subject to this limit will increase by nearly 30% (Pearce, 2012; Koodie et al, 2012). Many existing treatment works will struggle to meet these tighter consents. Population growth, increasing land costs, and the need to enhance existing assets have put a demand on the water industry to find appropriate effective solutions. Aerated wetland systems have been identified as a viable solution for both community wastewater treatment systems owned by the Water Companies, and privately owned wastewater treatment works in the UK, both as a retro-fit solution as well as new build systems. Aerated systems are deeper than conventional passive subsurface flow systems and therefore have a smaller footprint making them suitable for sites where the available land space is at a premium.

ARM Ltd has installed 32 systems in the UK over the last 3 years which vary in size from 10 m^2 to 2.1 ha. 53% of these have been retrofitted into existing constructed wetlands, thereby improving the treatment capability of the site whilst making use of the existing infrastructure. 87% of aerated systems were designed for sewage treatment which include secondary and tertiary treatment as final polishing to achieve < 5 mg/L of ammonia and one to treat effluent from a CSO. The systems designed to treat industrial effluents include a new build system to treat brewery effluent and a retrofit system to treat run off from airport winter deicing activities which have high levels of BOD, and ammonia and sulphide removal from landfill leachates (Figure 2).

Retrofitting aeration into an existing wetland can be implemented during a refurbishment. Gravel is removed from the bed and the airlines installed at the base of the bed before the gravel is replaced. On some sites such as the wetland at Mayfield Farm, the airlines can also be ploughed directly into the gravel (Figure 3). Figure 3 also shows photos from Hounslow, UK, where an aeration system is being retrofit into an existing treatment wetland.

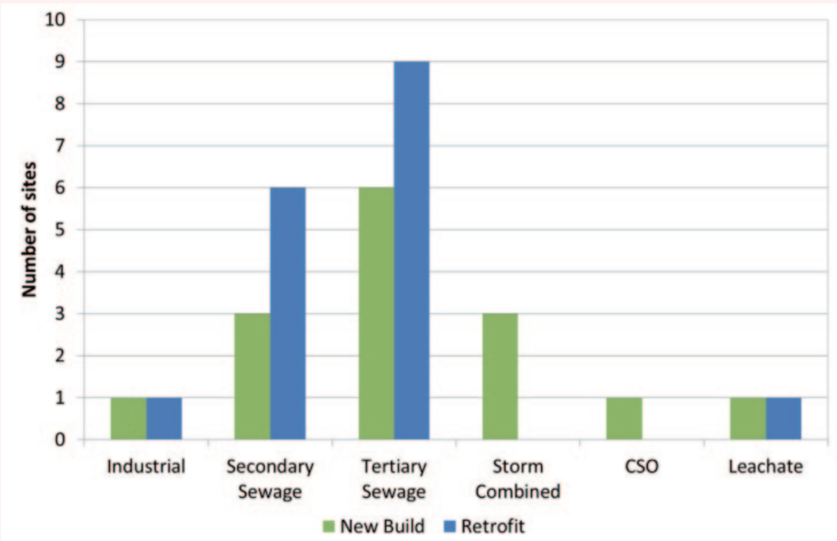

Figure 2. Number of aerated systems and associated effluent type installed in the UK (as of September 2013)

Figure 3: Retrofit of an aeration system into an existing treatment wetland at Mayfield Farm, UK.

ARM Ltd.'s pilot system at Rugeley, UK and the full scale system at Wolseley Bridge have been monitored for the past 3 years and provide first insights into how quickly aerated wetlands achieve high levels of nitrification from the date of commission (start-up) (Figure 4). Results from their investigation indicate that nitrification is reached at approximately 4 weeks for a summer start-up and 6 weeks for a winter start. Water temperatures for the Period of Record (POR) for Wolseley was 17.6 – 21.6˚C and for Rugeley 5.4 – 12.7˚C. Further studies on this pilot system aim to optimise aeration in order to increase TN removal capabilities and to characterize how aerated systems respond if the air pump is turned off; and how quickly the system recovers when the air supply is restored.

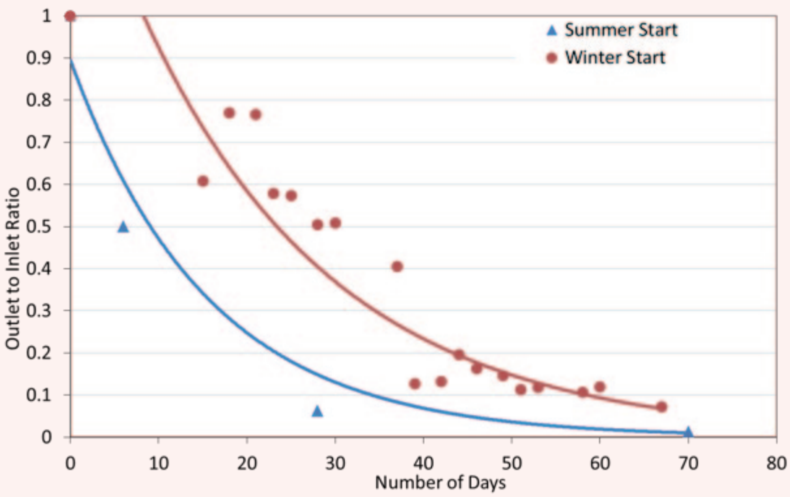

Figure 4: Outlet to inlet ammonium concentration ratios during the start-up phase for two aerated wetland systems in the UK (ARM, Ltd).

Germany

Located approximately 50 km northeast of Leipzig, Germany, the UFZ Ecotechnology Research Facility at Langenreichenbach contains traditional and innovative treatment wetland designs in order to compare the relative merits of various systems in terms of treatment performance and nutrient cycling, the role of plants, water use efficiency, and energy efficiency.

The research facility is unique in the fact that it is located adjacent to the wastewater treatment plant for the nearby villages, enabling all of the pilot-scale systems to receive the same domestic wastewater. The wastewater has no industrial inputs. Raw wastewater for the research site receives primary treatment in a large septic tank before being dosed to the wetland systems. Details of the 15 individual pilot-scale systems are given in Table 2.

Table 2: Details for the 15 pilot-scale treatment systems at Langenreichenbach, Germany (adapted from Nivala et al. 2013).

System Abbreviation[1]	System Type	Effective Depth (m)	Saturation Status	Main Media	Surface Area (m^2)	Inflow (L/d)
Horizontal Flow						
H25, H25p	HF	0.25	Saturated	8 – 16 mm gravel	5.6	100
H50, H50p	HF	0.50	Saturated	8 – 16 mm gravel	5.6	200
Vertical Flow						
VS1, VS1p	VF	0.85	Unsaturated	1 – 3 mm sand	6.2	600
VS2, VS2p	VF	0.85	Unsaturated	1 – 3 mm sand	6.2	600
VG, VGp	VF	0.85	Unsaturated	4 – 8 mm gravel	6.2	590
Intensified						
VA, VAp	VF + Aeration	0.85	Saturated	8 – 16 mm gravel	6.2	590
HA, HAp	VF + Aeration	1.00	Saturated	8 – 16 mm gravel	5.6	730
R	Reciprocating	0.85	Alternating	8 – 16 mm gravel	13.2	1770

[1]Systems planted with P. australis are denoted with “p” in the system abbreviation; other systems are unplanted.

First results from Langenreichenbach provide insight into the treatment performance of the 15 individual treatment systems. Results for common wastewater parameters ($CBOD_5$, TSS, TOC, TN, NH_4-N, NOx-N) are summarized in Nivala et al. (2013) and E. coli results are provided in Headley et al. (Headley et al. 2013). Of particular interest is the observed E. coli removal in the horizontal flow beds with aeration (HA), which showed upwards of 4.5 log10 unit E. coli removal at a hydraulic retention time of 2.9 days (Figure 6).

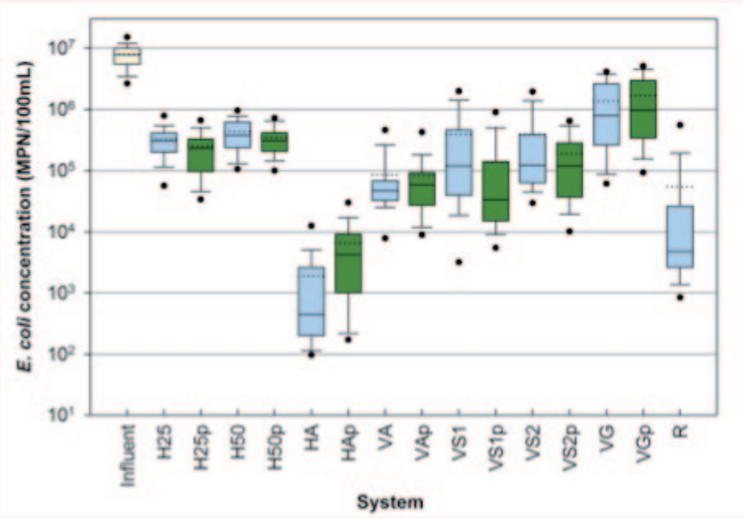

Figure 6: Box and whisker plot showing effluent E. coli concentrations from each treatment system (Headley et al. 2013).

Current research at the research facility in Langenreichenbach is now focused on assessing and optimizing the energy efficiency of the various designs, and aiming to further improve removal of priority contaminants such as total nitrogen and E. coli. Additional research into the microbiological community function and structure in standard and intensified treatment wetlands is currently underway.

Summary of the discussion

Main topics discussed and questions raised include:

- How do intensified / modified systems compare to other conventional wastewater treatment technologies (cost, footprint, etc.)?
- How efficiently are emerging pollutants and organic compounds removed in intensified/ modified wetlands (compared to standard designs)?
- To what extent to alternate nitrogen pathways (e.g. Anammox) play a role in nitrogen removal in intensified / modified treatment wetland sytsems?
- Denitrification in aerated wetland systems
  - Can increased denitrification be achieved through different aeration techniques and/or different orientation of the air distribution lines in the bed?
  - Can recirculation in aerated systems improve TN removal?
  - What can be done in the case of stoichiometric carbon limitation?
- Longevity of aerated wetland systems
  - How is sludge handled over the long term?
  - Does aerating a wetland bed help maintain hydraulic conductivity?

- What mechanism is responsible for high levels of E. coli removal in the horiziontal flow aerated wetlands at Langenreichenbach?
  - It is suspected that microbial predation plays a key role (Headley et al. 2013), but further research on this topic is only in the beginning stages.
- Bacterial shifts in aerated wetlands
  - How does a system react if the aeration is turned off?
  - How long does it take for the system to recover?

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Constructed Wetlands for Combined Sewer Overflow Treatment

CSO-CWs are generally agreed to be efficient in overflow treatment, but national approaches differ widely in their design and operation.

Authors: Daniel Meyer, Pascal Molle, Dirk Esser, Stéphane Troesch, Fabio Masi, Katharina Tondera, Johannes Pinnekamp and Ulrich Dittmer

Abstract

Combined sewer systems are designed to transport stormwater surface runoff in addition to the dry weather flows up to defined limits. In most European countries, hydraulic loads greater than the design flow of wastewater treatment plants are discharged directly into the receiving water bodies with minimal treatment (screening, sedimentation) or with no treatment at all. One feasible solution to reduce negative impacts on receiving waters is the application of vertical flow constructed wetlands. In Germany, first attempts to use this ecological technology were made in early 1990's. Since then, the further development let to a high level of treatment performance. During recent years, the national 'state-of-the-art' (defined in 2005) was adapted in other European countries including France and Italy. Against the background of differing national requirements in combined sewer system design, substantial development steps were taken. The use of coarser filter media in combination with alternating loadings of separated filter beds allows direct feedings with untreated combined runoff. Permanent water storage in deep layers of the wetland improves the system's robustness against extended dry periods, but contains operational risks. The constructions show similar functions despite different designs and layouts, but the correct dimensioning of all approaches (as well as inside sewer system simulation tools) suffer from uncertainties (e.g. impermeable surfaces, parasitic runoff and dry weather flow) in long-term runoff predictions. Current research studies aim to improve predictions of the system performance under varying conditions – both for classical wastewater parameters and emerging pollutants.

Introduction

Annual combined sewer overflow (CSO) pollutant loads can exceed those of WWTP effluent due to the enormous discharge volumes. This kind of pollution can lead to a high impact on the receiving water body over differing periods of time (e.g. short-term: acute oxygen demand, release of fish toxic NH_3-N, re-suspension of solids / long-term: sludge accumulation, eutrophication). To reach the requirements of the EC Water Framework Directive, CSOs need not only to be managed, but also require the effluent to be treated in many locations. One of the most (economically and ecologically) feasible solutions seem to be vertical flow constructed wetlands (VFCWs), specified as constructed wetlands for CSO treatment (CSO-CWs) in general and "retention soil filters" (RSFs) in Germany.

Compared to dry weather flows, flows from CSOs are usually diluted in terms of classical parameters like COD or NH_4-N. This correlation results in lower inlet concentrations for treatment facilities compared to WWTPs, but the hydraulic loads of single overflow discharges can exceed the typical loading of a VFCW due to enormous volumes of water.

Background

RSFs, as described in Uhl and Dittmer (2005), are able to retain numerous pollutants (Frechen et al., 2006; Dittmer & Schmitt, 2011; Tondera et al., 2013a,b). In German combined sewer systems, RSFs are generally located in series with stormwater tanks (Fig. 1-A). In the Federal State of North Rhine-Westphalia, about 1,870 stormwater tanks for CSO are operated – approximately 120 of them are combined with RSFs.

Main outcome of the session:

The main outcome of the session was the opportunity to demonstrate and to explain the differences between CSO-CWs and other types of vertical flow constructed wetlands both in general and in detail for (a) operation, (b) design requirements, and (c) research studies.

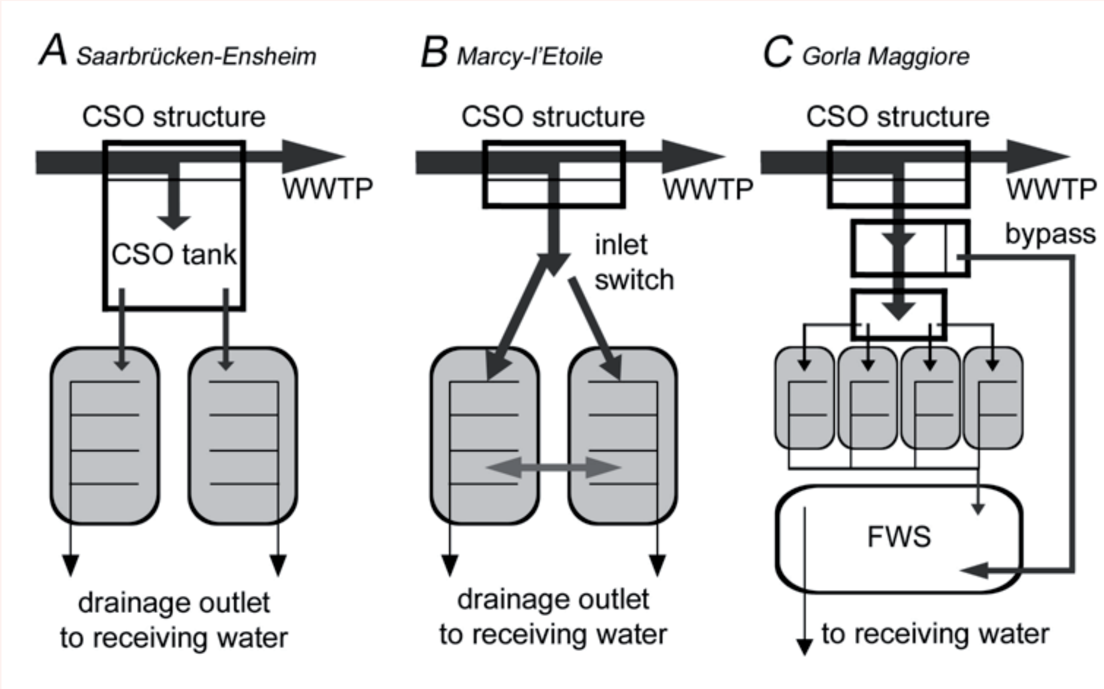

Figure 1: Simplified system sketch for (A) RSFs in Germany, and CSO-CWs in (B) France and (C) Italy (Meyer et al., 2013)

The French solution avoids the need for concrete basins (Fig. 1-B) if a treatment demand was noticed by sewer simulations showing overflows on a regular basis. This direct discharge results in heavier loads of particulates. In order to reduce clogging risks for small and concentrated as well as regular events, an alternating loading of two filter beds is necessary. In "French design" VFCWs (treating domestic wastewater), one bed is operated for about 3 - 4 days, while two other beds can regenerate sludge load abilities via mineralization (Molle et al. 2005). This kind of process control cannot be transferred to CSO-CWs directly, because system feedings are only corresponding to rain fall events. Experiences from the currently running research projects "SEGTEUP" and "ADEPTE" (full-scale Marcy-l'Etoile) will indicate adapted operation strategies for switching the feedings between the two beds.

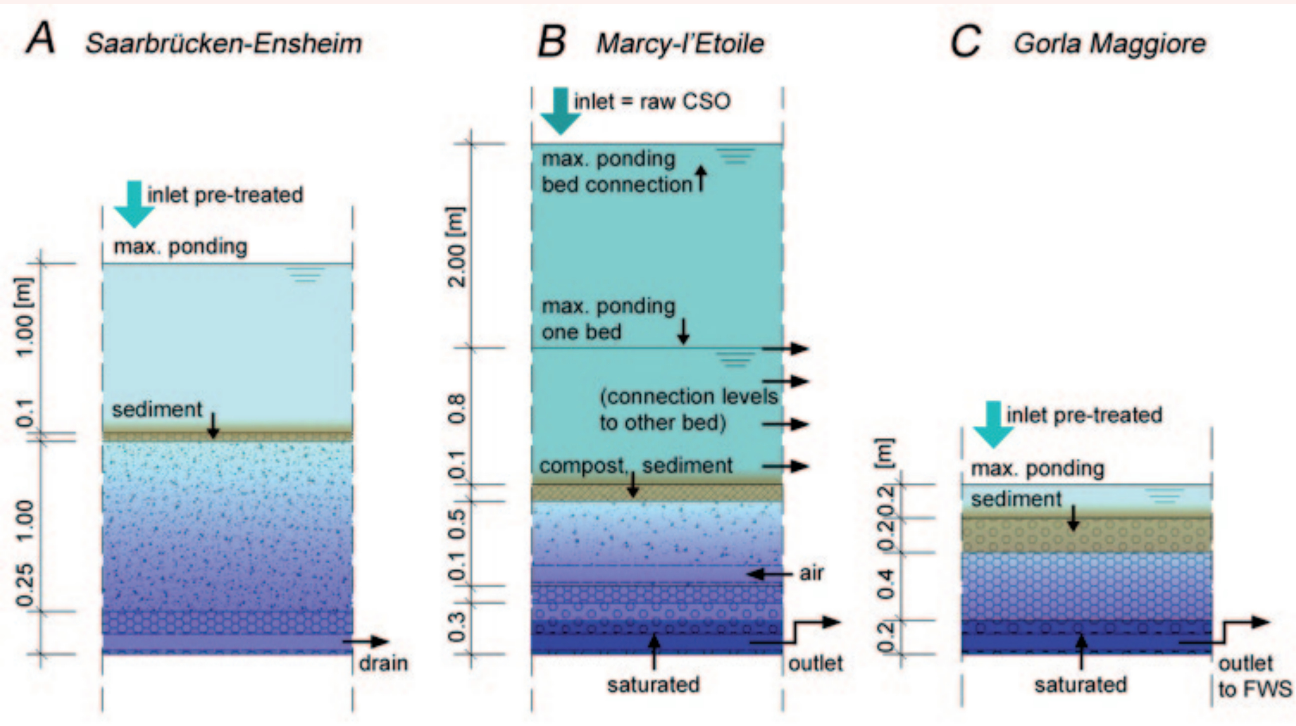

Figure 2: Simplified cross-sections of CSO-CWs in (A) Germany, (B) France and (C) Italy (Meyer et al., 2013)

Table 1: Comparison of characteristic design criteria (Meyer et al., 2013)

Criterion	RSF Germany (DWA-M 178, 2005)	CSO-CW France (full-scale Marcy-l'Etoile)	CSO-CW Italy (full-scale Gorla Maggiore)
Inlet water	CSO tank overflow	raw CSO	raw CSO (pre-treated before infiltrated)
Filter beds	1 or more in parallel	2 alternated loaded, in parallel for extreme events	4 alternated loaded, in parallel for extreme events
Retention layer depth	not defined (usually ~ 1.0 m)	flexible (0.1, 0.35, 0.6 or 0.8 m each bed), 2.0 m for connected beds	minimum 0.2 m
Filtration layer	0.75 m minimum (sand 0/2 mm, carbonate content > 10%)	minimum 0.5 m (one bed sand + zeolite, one bed pozzolana)	0.2 m (gravel 10 mm) + 0.4 m (gravel 2/6 mm)
Saturated layer	none (drainage layer 0.25 m gravel 2/8 mm)	flexible, minmum 0.2 m (0.3m gravel 10/20 mm, 0.1m gravel 3/8 mm)	0.2 m (gravel 40/80 mm)
Outflow limitation	0.02 L/(m²/s)	0.02 L/(m²/s)	0.004 L/(m²/s)
Max. hydraulic loads	40 m³/m² in annual average (maximum 60 m³/m² per year)	~ 40-80 m³/m² per year	35-40 m³/m² in annual average (maximum 50 m³/m² per year)
Design tool	long-term hydraulic sewer simulation	long-term hydraulic sewer simulation	long-term hydraulic sewer simulation

In Italy, in a first flush concept the needs of treatment are separated from those of hydraulic retention. The prototype located in Gorla Maggiore consists of two inlet splitters, four filter beds in parallel as well as an extended retention basin for the second flush flow (Figure 1-C). A maximum first flush of up to 640 L/s, generated by a rainfall event of 10 mm/h, can be retained. Exceeding volumes are bypassed into to the additional free water surface wetland (FWS).The filter inlet has to pass through an automatic screen and a grit separation tank (volume 110 m3) as a rough mechanical pre-treatment. Flow from the filter bed outlets are also fed into the FWS for secondary treatment. The FWS water level can be raised inside its artificial basin in order to release a maximum flow of 700 L/s to the River Olona due to flood protection. The water flow values given are representing a system in which almost all CSO will be treated by passing through the filter beds. Only peaks of maximum events with a return period of 10 years will pass by.

The cross-sections of the differing national approaches also show variations due to the specific needs (Fig. 2, Tab. 1): The French and the Italian CSO-CWs both provide permanent water layers. This is not particularly correlated to treatment processes – the design provides water for the reeds during extended dry periods, especially during hot and dry summers. In order to improve re-aeration after feedings an additional set of pipes was implemented (in RSFs the two functions of drainage and aeration are given by the same pipe system). Earlier German experiences with permanent saturation showed negative effects: Treatment efficiencies were decreased, low pH-values led to a release of carbonates, and anaerobic conditions caused odors. In comparison, the Italian CSO-CW will show shorter ponding periods due to the smaller retention space.

Until now, the filter design in all given countries is based on similar annual hydraulic loads (Tab. 1). The permanently increasing database – in combination with simulation tools – may lead to increased maximum annual and single event loads in future. This could be achieved by strategies to take higher benefits from divided filter beds. Pollution load criteria as design parameters could increase the adaptability to specific treatment needs.

Current research topics

In opposite to the French and Italian approaches, RSFs in Germany are under operation for more than 20 years. The national design guideline is currently under revision in order to implement the technical progress since 2005, e. g. for decreased CSO tank volumes. As an example, experiences from a completed research project on large-scale plants after several years of operation in Germany (Tondera, 2013a) can be summarised as follows:

- Micropollutants were investigated in a one year sampling phase. The reduction rates for substances like diclofenac, metoprolol and bisphenol a - up to a median of almost 75% - is at a level which makes further investigation worthwhile.
- Substances like carbamazepine, 1-h-benzotriazole and sulfamethoxazole were retained with a median between 26 and 39%. However, it was not possible to set up an inlet/outlet mass balance. To determine long-term retention or degradation, further investigations should be conducted in bench-scale.

In France and Italy, the basic function of the systems needs to be proven first. First monitoring results from 2013 show retention performances for C, N and P comparable to the well-known German approach. A special focus is given on a simplified modeling tool called RSF_Sim (Meyer, 2011), which allows to estimate the treatment performances in a long-term view. The following topics are currently investigated in detail:

- Retention limits, e.g. NH_4-N adsorption limits.
- Effects of the permanently saturated water layer.

- Risks of shortcut flows due to the coarser filter media.
- Maximum ponding time spans.

Summary of the discussion

The discussion was focussed on detailed questions about the main design ideas in France and Italy. The attended responsible persons (F. Masi, D. Esser, P. Molle & S. Troesch) could explain how they adapted their approaches starting from the German design guideline due to national requirements. In addition, the aims of current research projects could be explained in detail by the presenting authors (D. Meyer, K. Tondera, F. Masi).

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Microbiology in Treatment Wetlands

Methodological advances have allowed for the direct measurement, comparison, and characterization of microbial community structure and function in wetlands, which will now help researchers perform more microbiologically comprehensive studies ultimately moving the field closer to a complete understanding of the quantitative and functional role of microbial communities in treatment wetlands.

Authors: Kela P. Weber and Vincent Gagnon

Abstract

Microbial communities play a central role in treatment wetland systems, contributing to both contaminant removal and hydrological development. Given that both of these parameters are key to the proper operation of a treatment wetland system, consultants and researchers alike have identified microbiological research in the field treatment wetlands as a priority area. Through many studies, ranging from 1988 onward, researchers and design consultants have gained a general understanding of the importance of microbial communities in treatment wetland systems. Earlier studies focused on microbial enumeration or activity quantifications. Recent advances in both functional and structural characterization methods and equipment have provided researchers the opportunity to adapt and develop these methods for treatment wetland systems. The field of treatment wetlands is now at a point where studies can assess both microbial communities and water treatment simultaneously. Future work investigating both spatial and temporal microbial community dynamics in treatment wetland systems is expected to uncover the quantitative role of microbial communities and create connectivity with water treatment performance. It is expected that microbiological research will soon assist in optimizing overall performance and treatment wetland design. Several recent studies have moved the field forward in this fashion; however because of the large number of unique treatment wetland designs operating under a large variety of conditions throughout the world, significant effort uncovering the role and contribution of microbial communities in treatment wetland systems is still required.

Introduction

Microbial communities play an important role in wetlands designed for water pollution control (Kadlec and Wallace, 2009; Truu et al., 2009; Faulwetter et al., 2009; Garcia et al., 2010). Microbial communities 1) directly influence and contribute to contaminant removal, 2) develop biofilms which can affect hydrological development, 3) have a close interaction with plant roots within the rhizospheric region, and 4) can contribute to other beneficial or negative ancillary effects related to treatment wetland operations. Treatment wetlands (TWs) house many different microenvironments within a single system. Each microenvironment can have varying conditions, such as oxygen concentration, redox potential, ionic strength, pH, nutrient availability, or pollutant concentration, to name a

Main outcome of the session:

- Direct evaluations and linkages between the roles of microbial communities and water treatment would assist in our understanding and eventual design improvements for treatment wetlands.
  - Direct measurements evaluating both spatial and temporal variations of microbial community structure, function and activity within systems over several seasons are needed to advance our understanding of the roles of microbial communities in treatment wetland operations.
- Recent methodological developments have given researchers the tools to better characterize and understand both the structure and function of microbial communities in treatment wetland systems.
  - Not all research groups have the equipment or expertise on staff for microbial community investigations but several groups have developed the expertise and attained the instrumentation required. It is expected that collaborative studies between groups will increase over time allowing for more comprehensive studies looking at both water treatment and microbial communities in conjunction.

few. These variations allow for the development of diverse microbial communities within different microenvironments of a treatment wetland. Figure 1 presents a simplified depiction of microbial community interactions with plant roots, and the bed media.

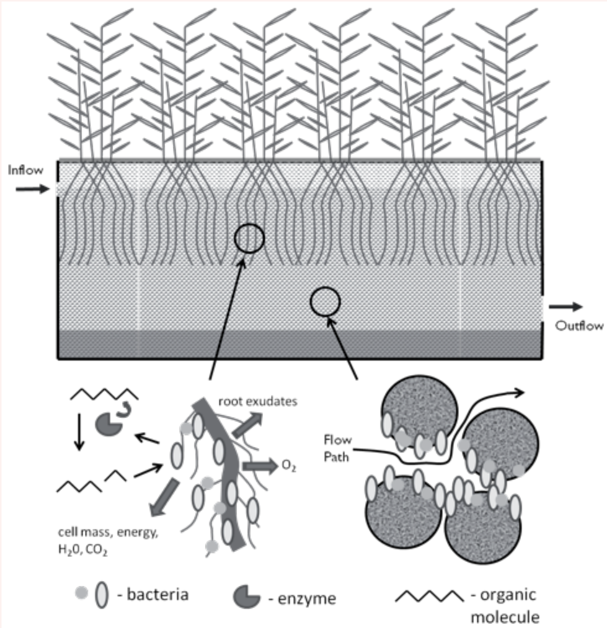

Figure 1: Simplified depiction of microbial community interactions with bed media, plant roots, and organic wastewater components in a horizontal subsurface flow treatment wetland system. (Diagram not to scale)

Microbial communities can exist as free-floating microorganisms within the interstitial spaces of the bed media or as anchored/attached colonies surrounding either the bed media or integrated within the rhizosphere or roots of the plants. It is generally accepted that interstitial microbial communities, although present, play a relatively small role in contaminant removal when compared to rhizospheric or other biofilm-bound microbial communities. Weber and Legge (2013) reported quantitative microbial activity observations supporting this view. Nonetheless, the interstitial water contains free enzymes excreted by fixed bacteria which will assist in the degradation of organics and other contaminants. Depending on the oxygen concentrations and redox potential in a specific region within a TW, different microbial communities will develop and therefore different metabolic pathways will be responsible for the removal of pollutants from the water (Figure 2). For example, microenvironments within the near-root zone (within 1 mm of a root) of horizontal subsurface flow wetlands can be largely aerobic (redox potential +250 to +700 mV), even though the rest of the bed is dominated by anaerobic processes (redox potential +250 to -400 mV, Truu et al., 2009). The potential for localized conditions is one feature of TWs that has allowed for unique and sometimes improved contaminant removal capabilities over more conventional, high-energy input, water treatment technologies.

Microbial communities play a role in organic matter degradation, nitrogen transformations, phosphorus cycling, and other more specific processes such as methanogenisis, sulphate reduction, dehalogenation, iron oxidation/reduction, or the degradation of specific contaminants such as MTBE and BTEX, among others. Significant interest has also been given to the role of microbial communities in the treatment of pathogens and emerging contaminants (pharmaceuticals, personal care products, antibiotics, nanomaterials, synthetic hormones, etc.), in addition to the possible generation of antibiotic resistant microorganisms within wetland systems receiving antibiotics at low concentrations (ng/L).

Besides directly treating, utilizing, mineralizing or transforming pollutants in TWs, microbial communities also play a role in terms of contaminant retention through the creation of biofilms. The attachment or anchorage of microorganisms in TWs depends on the capsule or slime layer surrounding the specific microbial communities developing in the TW, the grain size of the bed media, the availability of roots or root hairs, and the local water velocity in the immediate region. Microbial attachment/detachment occurs readily, with extracellular polymeric substances (EPS) excreted into the slime layer or capsule region assisting attachment, and shear stress working to detach the same microorganisms. These EPS's are made up largely of polysaccharides, as well as proteins that give a sticky exterior. This sticky exterior also allows for the adsorption of contaminants from the interstitial waters. This biofilm adsorption aids the physicochemical removal processes and also provides non-motile microorganisms entrapped within biofilms access to a carbon and energy source. Water

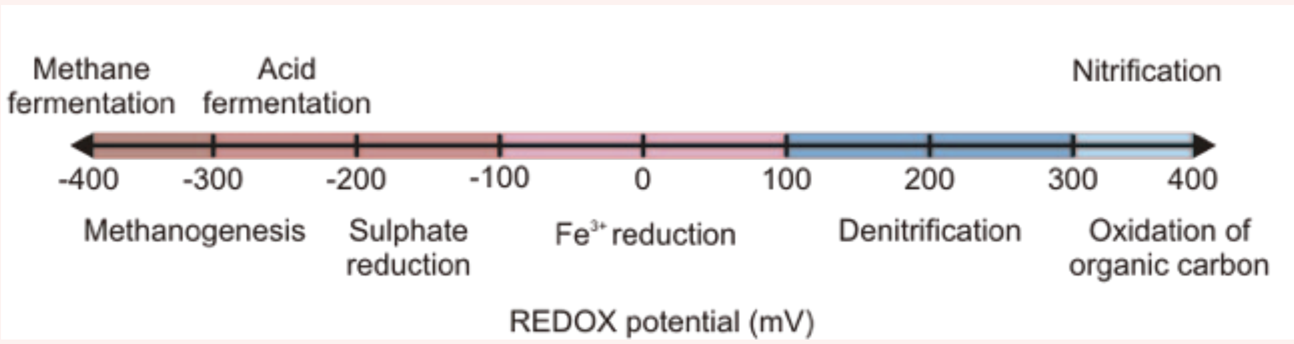

Figure 2: Relationship between microbial process and REDOX potential. Adapted from (Gagnon et al., 2007).

velocity and the associated shear stress will have an effect on microbiological development, which may lead to the selection of specific groups or even microbiological species developing within a system. Biologically based biofilm development has been documented in the literature and can have a significant effect on overall system porosity (Weber and Legge, 2011). Porosity reduction based on microbiological development also affects dispersivity (mixing) characteristics, and can lead to preferential flow paths (short-circuiting), and even eventual clogging given specific conditions (unpublished personal observations discussed during the workshop).

Current background / status

The field of microbial community characterization has been through an immense growth period within the last 30 years. Figure 3 summarizes the main categories of microbial community characterization in a simplified fashion.

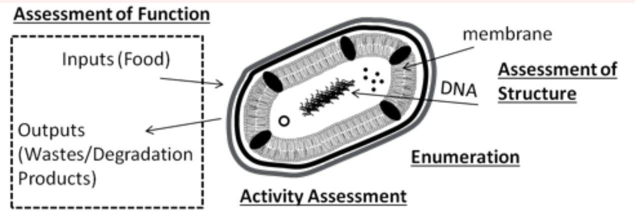

Figure 3: Microbial community characterization techniques. (Cellular components not to scale)

Enumeration Techniques

Enumeration was one of the first characterization techniques utilized in TWs. Originally this involved plate cultures and the subsequent counting of colony forming units, filtering and dry weight measurements of total organic matter, and direct counting and/or identification under a microscope (e.g. Petroff-Hauser counting). Later developments included microbial staining techniques, flow cytometry, and eventually real-time polymerase chain reaction (RT-PCR) (also known as quantitative PCR – qPCR).

Quantification of Microbial Activity

Microbial activity methods were also developed and utilized very early in the field of TWs (although not always expressly described as microbial activity). For example measurements of soil respiration have been used and described as far back as the 1980s. Respiration rates have generally been measured in aerobic systems or using samples from aerobic regimes and have most often tracked either O2 utilization rates, or CO2 production rates. Other activity measurements include the direct or indirect quantification of adenosine triphosphate (ATP - the main coenzyme used in cellular metabolism) or nicotinamide adenine dinucleotide (NADH - coenzyme involved in cellular metabolism), and the quantification of extracellular enzyme activities (eg. fluoresceine diacetate method).

Structural Characterization of Microbial Communities

Some of the first methods available for microbial community structure comparisons were fatty acid methyl ester (FAME), and phospholipid-derived fatty acid (PLFA) analysis. Although not used for direct identification of microorganisms they give the ability to compare or differentiate complex microbial communities based on the specific make-up of the plasma membrane of prokaryote cells.

A number of methods have been developed based on the characterization of PCR amplified DNA segments from a mixed microbial community. Most methods utilize primers that amplify a highly conserved region of DNA encoding for the 16s ribosomal unit to gain an understanding of all prokaryotes in a sample; however other regions or specific genes can be targeted to gain more specific information. Some of these methods include denaturing gradient gel electrophoresis (DGGE), temperature gradient gel electrophoresis (TGGE), and single-strand conformation polymorphism (SSCP), each of which yield patterns of bands embedded within a gel which can then be excised and sequenced. To gain a full understanding of microbial community structure, sequencing is required; however useful information regarding structural diversity can also be gained without sequencing.

Other methods that allow for community comparisons include terminal restriction fragment length polymorphism (TRFLP), amplified rDNA (Ribosomal DNA) restriction analysis, ribosomal intergenic spacer analysis (RISA), length heterogeneity PCR (LH-PCR), and random amplification of polymorphic DNA (RAPD). Although all methods mentioned can give useful information, perhaps the most powerful methods to be developed are the high-throughput sequencing methods. Several different methods/platforms have been developed by various companies and research groups throughout the world including but not limited to, pyrosequencing, ion torrent semiconductor sequencing, sequencing by ligation, and reversible dye-terminator sequencing. These high throughput sequencing methods allow for the simultaneous relative quantification and sequencing of all targeted genes within a sample. These methods hold great potential as they give a complete snapshot of a sample's microbial community structure in one simple method, but they are currently the most costly microbial community characterization methods available, which can be prohibitive.

Functional Characterization of Microbial Communities

Microbial community function looks to gain an understanding of exactly what types and in what quantities the microbial community is utilizing and excreting different compounds. It is through these basic functions that microbial communities interact with different trophic levels, participate in different nutrient cycles in the environment, and offer pollutant removal capabilities in TWs. Rather than quantifying and identifying DNA fragments within a sample, primers and probes can be developed for mRNA segments. Although mRNA is more difficult to work with, it gives an actual indication of gene expression and therefore an indication of a specific active function, rather than the potential for a specific function when assessing DNA. qPCR and fluorescence in-situ hybridization have been used to this end.

Community level physiological profiling (CLPP) is another functional characterization method where the metabolic activity of a community sample is measured with relation to 31 to 95 different carbon sources on a microtitre plate. With this method, both a relative activity and total metabolic potential for degrading a range of carbon sources is obtained.

The last functional approach is the use of microarrays, such as the Geochip 3.0, to assess the presence of anywhere from 20,000 to 60,000 genes via RNA (or DNA) segments using specified probes on a small microscope slide. Although in its infancy this methodology also holds great potential. With the expression of so many genes being assessed in a single sample, full enzymatic pathways can begin to be assembled and assessed, giving a more thorough (although not directly measured) indication of overall function. Microarrays can also be costly, which can be a prohibitive factor in its common usage.

Historical View of Microbial Community Characterization in Treatment Wetlands

We completed a literature review to offer an understanding of the efforts invested into microbiological studies for wetland systems to date (Figure 4). Studies included here were for the most part not solely focused on microbiological processes or characterizing the communities, but rather had a characterization or microbiological component to complement other concurrent investigations (see Figure 4 for keywords). The search was also not restricted to treatment wetlands alone, but included natural wetland systems as well. Of the 564 papers identified, 235 were removed as they were not related to wetlands, with another 97 removed as there were no microbiological methods used, leaving a total of 232 papers. Through a review of the remaining papers, a comprehensive understanding of the history of microbiological characterization in wetland systems was gained.

The first studies in the late 1980s used general enumeration or activity techniques. Use of both enumeration and activity measurements continue to date and each year accounts for the majority of publications. In the early 2000's TRFLP, DGGE and more directed genetic sequencing was introduced into several studies along with functional characterization including CLPP. From 2006 onwards structural studies have become more frequent, most likely because of the greater accessibility of the instrumentation and lower cost of materials. In 2010 through 2012 studies have begun to be multiphasic in nature, with combinations of activity, enumeration, structure and function methodologies applied to the same systems. Of the 232 papers reviewed, 49 investigated nitrogen transformations, 40 focused on pathogen removal and/or characterization, and the remaining papers investigated other microbiological processes or focussed on specific microbial community characterization techniques.

In 2013 (up to June when this review was completed) 11 studies that included a microbiological assessment component were published. Three studies used qPCR, four studies used some type of enzymatic activity measurement, two studies were completed using pyrosequencing, one study utilized CLPP, and one study (conference proceeding) used the GeoChip 3.0 microarray.

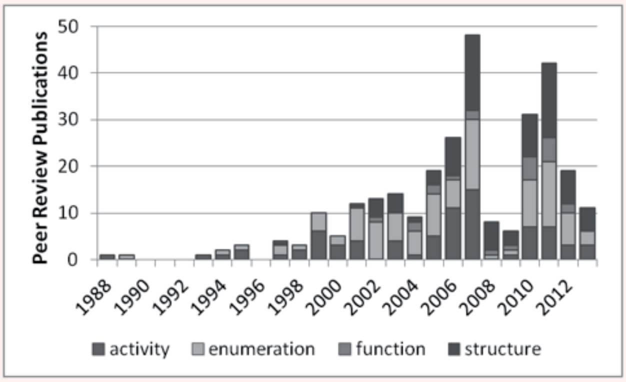

Figure 4: Summary of microbial community characterization publications in the field of treatment wetlands. Keywords: wetland, constructed wetland, treatment wetland, microbiology, microbial, microbiological (with all combinations). Databases: Compendex, Referex, Inspec, GEOBASE, GeoRef, Scifinder, Web of Science.

Challenges / opportunities

Spatial Microbial Community Dynamics

Spatial variances in microbiological density, activity, and structural or function diversity are not completely understood. There have been a limited number of studies evaluating spatial dynamics. In the majority of cases significant variations have been observed. Understanding these spatial dynamics and why they occur is key to further understanding how design and environmental factors can affect the role of microbial communities in TWs.

Most comprehensive studies have been done at the meso or micro scale, but in moving forward an understanding of spatial dynamics at the pilot or full scale is needed. In doing this, microenvironments within close proximity (for example rhizosphere-region samples taken directly from the root, within 1 mm from the root, and then progressively outwards) would be of use in truly understanding the spatial variances found within what is assumed to be a highly diverse microbiological region. To this end, it is also important that a greater understanding of anaerobic microbial community functions are evaluated as these processes are of importance in TW systems.

Also noted at the UFZ workshop is the need to accurately evaluate the microbial communities contributing to water treatment during microbiological characterization studies. Many mesoscale studies have been able to compare and in some cases make use of interstitial water for characterization due to the specific operational design and configuration. However in larger scale operations characterization of the microbial communities in outlet water samples may not be (and most likely, in the opinion of the authors, is not) a microbial community sample representative of the communities contributing to water treatment in TW systems. Further study comparing biofilm, interstitial, and outlet water samples for microbial community characterization for several TW system scales should be completed to better understand this aspect.

Temporal Microbial Community Dynamics

Much like the spatial studies previously mentioned, temporal variations in microbial community density, activity, structure and function have been recorded. Even fewer temporal studies (compared to spatial) have been completed to date, likely because of project/ funding timelines and costs associated with such studies. Temporal variations have been noted with either season or differing input water parameters affecting the microbial community in some way. Again, understanding temporal variations and possibly how external factors can be modified to enhance the microbial community function with TWs could lead to improved water treatment performance.

Additional fundamental research is also required investigating temporal variations in microbial activity and how this can affect results and interpretation. In treatment wetland operations it is suggested that the temporal dynamics of microbial activity in a set spatial area could change based on differing non-continuous wastewater inlet parameters such as organic content, nutrient ratios, and loading. Past research has been completed through the use of a consistent time point for activity measurements throughout studies (eg. a specified time following a bulk wastewater inlet loading), however it is rarely understood if a chosen reference time point is the most appropriate for the study being conducted.

Connectivity between Microbial Community Characterization and Treatment Wetland Operations

Creating connectivity between microbial characterization studies and treatment wetland operations is the end goal for all researchers, but this connectivity was identified as a weakness at the UFZ workshop. This is most likely due to the fact that many recent studies focused on method verification or first time trials using specific microbiological methods in TW systems. Microbiological methodologies are not easily transferred between system types (soil to sediment for example) and require significant effort to adapt and optimize for any one system type. The field of TWs will always be working to adapt new methodologies to TW systems, although the field as a whole is at a point where a large breadth of methods have been adapted and are ready to be applied in new research studies.

As noted previously, both temporal and spatial studies are important stepping stones in moving forward. Nevertheless, there are many other basic questions still requiring further research. Many opportunities were discussed at the UFZ workshop including but not limited to the quantitative role of ANAMMOX pathways in nitrogen transformations; heterotrophic nitrification and aerobic denitrification; the role of microbial communities in industrial effluent treatment (both the identification and possible isolation of specific species or groups of bacteria); the quantitative split between catabolism and anabolism for microbial communities in TWs; the effect of COD:N:P ratios on the structure and function of microbial communities in TWs; the generation of antibiotic resistant bacteria in different system types of varying sizes; the identification of microbial species directly transforming emerging contaminants in TWs; identification of both protozoa and bacteria involved in pathogen removal; the effect of biofilm on hydrological parameters; and verifying several microbiological parameters for use in TW numerical models.

Summary

For many decades, researchers and industry leaders alike have made gains in understanding, while also posing many questions regarding the role of microbial communities in wetland systems. Future research frontiers include both spatial and temporal analyses. At present there are many tools available for microbial community characterization and the future holds many great discoveries.

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Modelling of Treatment Wetlands

Modelling of the complex system constructed wetland requires a number of sub-models and input parameters which are strongly related to the type of the wetland and the objectives of the simulation study.

Authors: Günter Langergraber and Ania Morvannou

Abstract

Due to the complexity of constructed wetland (CW) processes numerical models have to comprise a number of sub-models to describe all relevant processes. Only few tools based on process-based models are available for modelling the pollutant transport and degradation in subsurface flow CWs. The paper describes briefly the current background and status of CW modelling, experiences from applying existing simulation tools and limitations of existing tools, and some challenges identified. These challenges include the correct description of water flow, the incorporation of a model that allows predicting clogging, the determination of biokinetic model parameters, and the needs for developing a new design tool for CWs that is based on process based models. Finally, the results of the discussion in the session are summarised for the following points: modelling of water flow, data needed for a simulation study, and requirements of CW designers for a design tool.

Introduction

In constructed wetlands (CWs) a large number of physical, chemical, and biological processes are active in parallel and mutually influence each other (Kadlec and Wallace, 2009)). Therefore wetlands are complex systems and for a long time have been often considered as „black boxes". When developing a wetland model a number of different processes have to be considered (Langergraber et al., 2009b):

- The flow model (describing water flow)
- The transport model (describing transport of constituents as well as adsorption and desorption processes)
- The biokinetic model (describing biochemical transformation and degradation processes)
- The influence of plants (growth, decay, decomposition, nutrient uptake, root oxygen release, etc.)
- The description of clogging processes
- Physical re-aeration

Still today most models for wetlands are using a „black box" approach, i.e. they do not consider processes in wetlands in detail. Data from experiments are needed to derive model equations for „black box" models. In process-based models the mathematical model equations are based on processes in wetlands and include balance equations for energy, mass, charge, etc. Data from experiments are used for calibration and validation of models. A better prediction should be possible using these models (Langergraber, 2008; Langergraber et al., 2009a).

Main outcome of the session:

- The amount and nature of data needed for the calibration depend on the CW type and the objectives of the simulation study.
  - Water flow: more data required for vertical flow CWs compared to horizontal flow CWs.
  - Pollutant concentrations: depends on biokinetic model parameters (influent fractionations) and dynamics in the influent of the system.
- Designers of CWs have the following requirements for a design tool:
  - Design tools need to be simple to use and predict reliable effluent concentrations.
  - It should be possible to change individual elements of design (e.g. grain size of filter media; order of beds in a multi stage system, etc.) and show the impact of these factors on effluent concentrations.
  - They should be able to predict failure of the system, e.g. which loads are acceptable over which time.

Current background / status

The available simulation tools describing transformation and degradation process in CWs are described by Langergraber (2011). Horizontal flow (HF) systems can be simulated when only water flow saturated conditions are considered. For modelling vertical flow (VF) CWs with intermittent loading, transient variably-saturated flow models are required. Due to the intermittent loading, these systems are highly dynamic, adding to the complexity needed to model the overall system. Available models can be grouped into the following categories (Langergraber, 2011):

- Reactive transport models for saturated flow conditions
  - applicable only for constant flow rates
  - with a tanks-in-series approach for water flow
  - applicable to variable flow rates (incl. changing water table level in the HF bed)
  - coupled to a complex groundwater flow model
- Reactive transport models for variably saturated flow conditions
  - with simplified approach for simulating the variably-saturated water flow
  - coupled with flow models that use the Richards equation to describe variably-saturated water flow.

Recent developments, especially toward implementation of the CWM1 biokinetic model (Langergraber et al., 2009b) include the works of Langergraber and Šimůnek (2012), Samsó and Garcia (2013), and Mburu et al. (2012). Table 1 compares the 3 tools regarding the considered sub-models required for a wetland model.

Experiences / examples

Experiences from applying existing simulation tools can be summaries as follows (Langergraber, 2011):

- Simulation results (effluent concentrations) match the measured data when the hydraulic behaviour of the system is well described, i.e. the influence of the parameters of the hydraulic properties of the filter material is much higher compared to the influence of the parameters of the biokinetic model.

  For water flow simulations in VF beds it is advised to measure:
  - at least the porosity and saturated hydraulic conductivity of the filter material, and
  - if possible the volumetric effluent flow rate between loadings
- not all measured data acquired from experiments are useful for simulation purposes (e.g. sampling frequency, analysed parameters, dynamic behaviour
- information gained from experiments and/or measurements can be of too much detail compared to the needs of the simulation tools and their underlying numerical methods
- modern biotechnological tools help to gain new insights in the functioning of CWs (e.g. data obtained from these experiments are usually not in a form and/or have the appropriate units to be used directly for modelling purposes)
- a common language and understanding is needed between modellers and specialists from other fields (e.g. microbiologists, plant physiologists, hydrologists, CW designers, etc.) to produce useful data for modelling purposes

Limitations of existing simulation tools are (Langergraber, 2011):

- One of the main obstacles for the simulation tools available is that they are rather complicated and difficult to run. Meyer (2011) developed a simplified but robust and reliable model for design purposes for CWs treating combined sewer overflow based on experiences from simulations with a complex simulation tool.

Table 1: Comparison of recent simulation tools for constructed wetlands.

Simulation tool Reference	**HYDRUS wetland module** Langergraber and Šimůnek (2012)	**BIO_PORE** Samsó and Garcia (2013)	**AQUASIM** Mburu et al. (2012)
Flow model	Richards equation (variably saturated flow)	Variable water table (saturated flow)	No flow considered
Transport model	Advection, dispersion, adsorption	Advection, dispersion, adsorption	No transport model
Biokinetic model	CW2D + CWM1	CWM1	CWM1
Influence of plants	Evapotranspiration, uptake and release of substances	Evapotranspiration, uptake and release of substances	Evapotranspiration, uptake and release of substances
Clogging model	Not considered	Included	Not considered
Re-aeration	Considered	Considered	Considered

- Additions required for a CW design tool - 2 simple models for pre- and post-treatment units would be needed :
  - a simple model for prediction of TSS and COD removal based on the design of the mechanical pre-treatment unit, and
  - a model for pre- and/or post-treatment of phosphorus with pre-precipitation in the mechanical pre-treatment unit and/or adsorption filters after the filter beds, respectively.

Challenges / opportunities

Correct description of water flow

Morvannou et al. (2012) showed that the porosity of the layer of a VF CW may serve as preferential flow paths through which water can bypass most of the soil porous matrix in a largely unpredictable way. This is especially true for the sludge layer in French-style VF CWs (Troesch and Esser, 2012). Water flow in such a system cannot be modelled with uniform flow models (such as the van Genuchten-Mualem function in HYDRUS, Šimůnek et al, 2011). The comparison between measured and simulated tracer breakthrough curves indicates that the non-equilibrium approach (i.e. using a model to separately describe flow and transport in preferred flow paths and slow or stagnant pore regions) seem to be the most appropriate for simulating preferential flow paths. Such a dual-porosity model therefore also need to be incorporated in the software tools for accurately describe water flow and solute transport in French VF CWs.

Clogging model

Clogging models need to be able to describe i) the transport and deposition of suspended particulate matter, and ii) the deposition of particulate matter, bacterial and plant growth that may reduce the hydraulic capacity/conductivity of the filter medium. This is of importance for the simulation of the long-term performance and to predict the potential failure of CWs due to clogging.

Values of the biokinetic model parameters

One of the basic assumptions of Langergraber (2001) was that bacteria in CWs are and behave similar to those in activated sludge systems. Therefore the parameters of the biokinetic models developed for activated sludge systems should be applicable also to describe processes in CWs.

This assumption has been confirmed as experience showed that a good match between measured and simulated concentrations can be achieved when the hydraulic behaviour of the system is well described (see above). Additionally, Morvannou et al. (2011) found good agreement between measured and calculated volumetric nitrification rates (Table 2)

We therefore advise not to change parameters of the biokinetic model unless for good reasons. However, parameters describing the inflow wastewater that are related to the biokinetic model chosen have high impact on the simulation results and need to be adapted for each simulation study. These parameters include i) the fractionation of influent COD (i.e. estimation of the different COD model fractions from measured total COD) and 2) the calculation of the organic N content of the different COD fractions.

Table 2: Comparison measured and calculated volumetric nitrification rates (adapted from Morvannou et al., 2011).

Method	Measured with solid respirometry	Calculated from simulation results *
Results [mg $O_2/L_{sample}/h$]	32-50 (mean = 41, SD = 9; 2 values)	30.5

* from simulations using parameters for the biokinetic model from activated sludge systems.

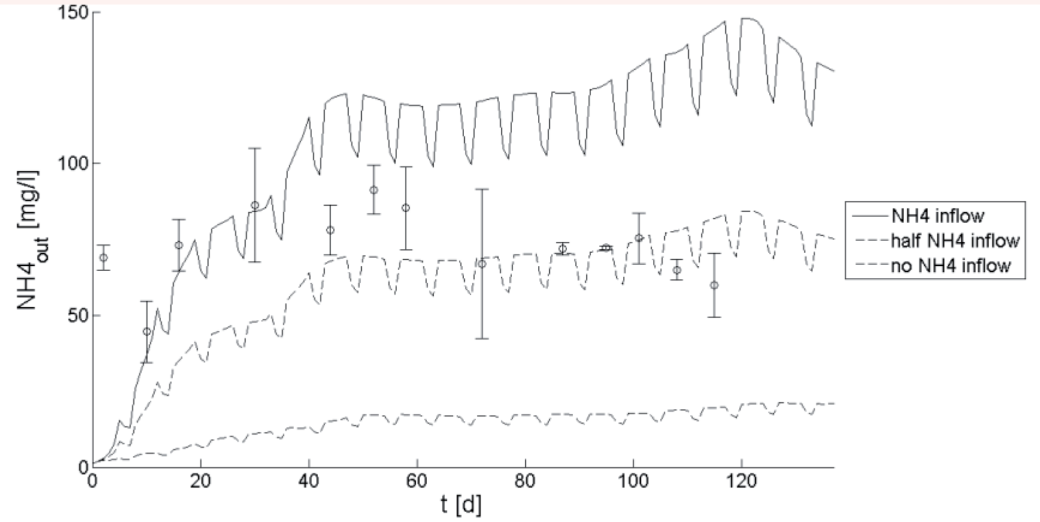

Figure 1: Influence of inflow NH_4-N concentrations on effluent NH_4-N concentrations.

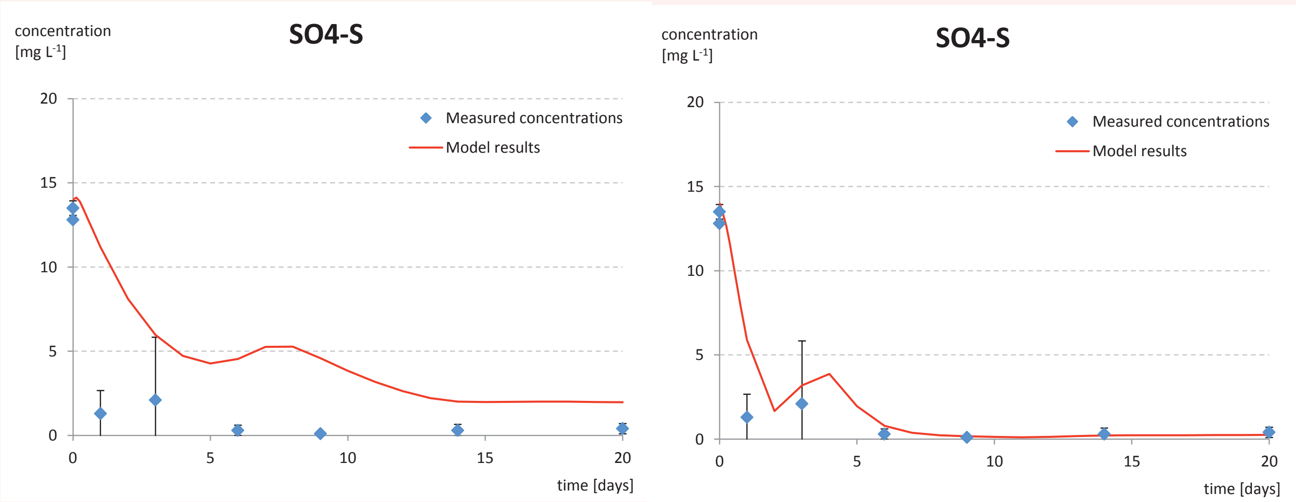

Figure 2: Measured and simulated sulphate concentrations for a batch-fed column at 24°C planted with Carex (left: using the standard parameter set of the biokinetic model; right: after adaptation of inhibition and half-saturation coefficients to allow anaerobic, anoxic and aerobic processes to occur in parallel; Pálfy, 2013)

Rizzo et al. (2013) describe the set-up of a model to simulate experimental data from a horizontal flow CWs fed with artificial wastewater. During the experiments the only nitrogen parameter measured was TKN. As simulation requires influent ammonia nitrogen concentrations, the organic N content of the different COD fractions had to be adapted from standard values for the type of artificial wastewater used. Figure 1 shows the effect of the different influent ammonia nitrogen concentrations on simulated effluent concentrations.

Pálfy (2013) simulated experimental results from batch-fed column experiments. He described the need to adjust some parameters of the biokinetic model to be able to simulate anaerobic, anoxic and aerobic processes to occur in parallel. These phenomena occurred in practice and can be explained by the local effect of root zone re-aeration. Figure 2 shows measured and simulated sulphate concentrations before and after adaptation of parameters of the biokinetic model. Batch experiments can be a way to calibrate biokinetic model parameters as there is no impact of water flow on the treatment performance.

Design tool for CWs

Langergraber (2011) concluded that to make numerical simulation a useful and applicable tool for CW design further developments of the existing models are needed. A simplified computer-based CW design tool based on process-based numerical models shall be developed that

- can be used with knowledge on CW design but do not require special knowledge on numerical modelling,
- allows designing CWs for different boundary conditions (such as climatic conditions, wastewater characterization, filter material, etc.), and
- makes the description of the dynamic behaviour of the designed CW possible thus allowing to show the higher robustness of CW treatment systems e.g. against fluctuating inflows and peak loads

Summary of the discussion

The discussion was organised according to the 4 topics listed in the previous chapter. The main points raised and discussed are summarized below.

Application of models, water flow models

- The models have been developed and mainly used for domestic wastewater right now. It should be however possible to use the models also for wastewaters with similar characteristics as domestic wastewater such as wastewaters from food industries. For synthetic wastewater recent experiences are available as well.
- Alain Petitjean (France) pointed out that the next step for the development of di-phasic flow models is to take into account the biofilm growth and its impact on water flow. Kela Weber (Canada) pointed out that they developed a model based on COMSOL that also considers biofilm growth and links that to changes of the flow pattern.
- Also for the HYDRUS wetland module it is planned to include preferential flow for bio-kinetic models. Assuming biofilm is an immobile part of the water, and to simulate this.

Which data needed for a simulation study?

- The data needed for the simulation study in general and for the calibration of the water flow model in particular depend a lot on the objectives of the simulation study. E.g. if nitrification and denitrification are occurring close together your hydraulic model has to be really, really accurate.

- Hydraulics of HF CWs:
  - Tracer studies
- Hydraulics of VF CWs:
  - Minimum requirement: porosity and saturated hydraulic conductivity of the filter material (allows, according to the experience, a moderate good fit of water flow simulations to measured data)
  - if possible: volumetric effluent flow rate between loadings (allows determination of parameters of the water flow model by inverse simulation)
  - additionally, if available: tracer studies and measurements of water content(s) and pressure potential(s) within the filter bed (will allow more accurate calibration of the water flow model)
- Concentrations:
  - Measured concentration according to the parameters of the biokinetic model used and that allow estimation of COD fractionation and organic N content of these fractions.
  - If the dynamic behaviour of the system should be modelled also the data need to show the dynamics, e.g. diurnal variations, and have to be collected in the respective frequency.

<u>Do companies need design tools for constructed wetlands? What are your requirements, what would companies like to have?</u>

- Models have to be applicable for people who have to design CWs.
  - For this objective the models need to predict outflow concentration with high accuracy. When designing CWs outflow concentration need to be guaranteed, models should also present uncertainties with results (e.g., 95% confidence interval).
  - Models also need to have a simplified interface, where an engineer can learn in a week's time how to use the model.
- Most companies that build CWs do have design tools, but they would like to have better design tools. They basically have two types of tools to design wetlands: rules of thumb (m2/PE, based on the collective experience in e.g. France, Austria or Germany), or using something like the P-k-C* model. But then it's important to remember that the k-rates in text books (e.g. Kadlec and Wallace, 2009) are lumped parameters. Those rates are affected by wastewater type, type and gravel size of media, plant and microbial community maturity – all those go a single lumped parameter.
- Where designers would like to go is to say: If we could change individual elements of design, how would that change the performance of the system? Examples are:
  - If there is experience with using 0 – 2 mm sand and then one has to use a 5 – 15 mm gravel because that's what's available, how does that affect design and in the end, the treatment performance of the system?
  - How much can I stress my system? That is a main topic including the forecast of the lifespan of the system. What is the maximum concentration/ load for certain periods that can be accepted? (knowing such concentrations/loads in the long term will kill the system).
  - For multi stage systems: How small can I make the first stage and make it still survive?

  None of the tools available now allows changing one element of design and tells the impacts on treatment performance. Moving beyond lumped parameters into discrete design elements, to look how changes in the physical design will change the effluent water quality.
- To enhance the prediction capability of models also data from stressed and failed systems are needed during calibration. However, CW designers usually present only working systems. For these experimental CW systems such as the LRB site can be of great help. However, even under controlled conditions sometime it is difficult to assess why systems collapsed.
- Resources are required for developing a design tool based on process-based models.
  - Who wants to pay for such a development? If an international project could be launched this goal could be reached sooner but still some year's development work would be needed.
  - For a small company it's not acceptable to wait for 3 or 4 years. They must have some information even if it is not the most accurate; to have something is better than nothing at all. Rough and approximate is good compared to having nothing at all (and waiting years and years for the perfect model).

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Yellow Phragmites: Significance, Cause, and Remedies

Preliminary observations of chlorosis of the leaves of Phragmites australis growing in some aerobic treatment wetlands, including discussion of significance, cause and remedies.

Author: Chris Weedon

Abstract

Summary of observations of profound chlorosis of the leaves of Phragmites australis, deployed in treatment wetlands receiving high redox potential influent, including likely cause (lack of bio-available iron), reasons for cause (low ammonia, high redox potential), impact on growth and wetland performance, and possible remedies (e.g. use of alternative plant species or foliar application of iron solution).

Introduction

Although not widely broadcast before this meeting, within the discussion period following Clodagh Murphy's presentation concerning performance of forced aeration constructed wetlands, observations were verbally reported from many groups of profound yellowing of the leaves of Phragmites australis when used in particular treatment situations, characterised by well aerated sewage feeding the bed. Forced aeration beds and recirculated-effluent vertical down-flow systems were reported to be especially prone to chlorosis.

Importance

Generally regarded as a trivial or "cosmetic" issue, greater importance has arisen due to chronic susceptibility of the plants to diseases, such as fungal rusts, and insect infestation, as well as reduced biomass – occasionally catastrophic – allowing disproportionate weed intrusion. The combined effect can lead to an unhealthy and unacceptable appearance to owners and operators, requiring substantially increased maintenance labour in order to achieve an acceptable bed appearance.

Observations

General

Distinct yellowing, usually in combination with green & yellow longitudinal striation (Figure 1), of the majority of the leaves of the majority of Phragmites australis plants was reported to be typical, in:

- Free-draining vertical down-flow reed beds operated with effluent recirculation and/or receiving effluent of a rotating biological contactor

Main outcome of the discussion:

- Many academic and commercial groups verbally reported chlorosis of the leaves of Phragmites australis growing in either free-draining vertical flow (VF) treatment wetlands or forced aeration treatment wetlands
- Plants were reported with the majority of leaves predominantly yellow, and/or showing distinct green and yellow striation, over wide areas or throughout the entire bed
- Whilst producing an undesirable "unhealthy" appearance, more important consequences include reduced stem height and reduced biomass density allowing excessive weed intrusion, and susceptibility to disease and infestation, increasing undesirable bed appearance and increasing maintenance labour requirements
- Effluent treatment performance has not been reported to have been affected
- Informal interdisciplinary discussion during the course of the three day meeting has revealed the likely cause of Phragmites australis leaf chlorosis
- Low ammonia and high redox potential in the wastewater feeding the Phragmites is believed to result in the iron content taking the ferric form with insufficient solubility to meet the nutrient demands of the growing plants, resulting directly in leaf chlorosis (insufficient synthesis of chlorophyll) caused by lack of iron within the leaf
- Remedies include use of substitute plant species and foliar application of ferrous sulphate.

Table 1: RBC effluent concentrations feeding 2 compact VF treatment wetlands, operating conditions and leaf colour of reed.

Location	Growing season	Leaf colour	Recirculation	BOD_5 (mg/L) [1]	NH_4-N (mg/L)	NO_3-N (mg/L)
Bodiam Castle (UK)	1	G	x	22	64	26
	2-3	Y	+	8	8	15
Pont Abraham (UK)	1-4	Y	x	18	3	28
	5-6 [2]	G	x	146	85	9

[1] Leaf colour (G = green, Y = yellow) was established by June in each growing season. BOD_5, NH_4-N and NO_3-N figures are average inlet concentrations of monthly spot samples March-June

[2] RBC switched off, i.e. bed treating settled sewage

(RBC) or receiving first stage French vertical down-flow reed bed effluent; and in

- Forced aeration reed beds, both horizontal and vertical flow.

Table 1 shows the influent concentrations, and operational parameters from 2 CW systems in UK, i.e. Bodiam Castle (Figure 2) and Pont Abraham (Figure 3), as well as the leaf colour of reed in different growing seasons.

Second Stage "French System":

Of the many two stage vertical down-flow French systems in existence (first stage receives raw sewage; second stage receives effluent from first stage), a small proportion exhibit strong chlorosis of the second stage reeds (Figure 4) [JP], the most extreme resulting in almost complete disappearance of reed plants, replaced by terrestrial "weed" species (Figure 4d).

Forced Bed Aeration:

Langenreichenbach horizontal subsurface flow forced aeration reed bed (Figure 5), from early development, exhibited a striking plant morphology pattern, correlating with distance from the point of settled effluent entry to the bed: tall, green plants at the proximal end with progressively yellower and shorter plants along the bed.

Ferrous Sulphate Foliar Application

One-off foliar spray application of fresh ferrous sulphate (3.82g/l aqueous Fe2+SO42-) early in the growing season resulted in sustained green Phragmites australis plants that had grown chlorotically throughout the previous season(s) [CA, RK], in vertical down-flow beds treating settled effluent with partial recirculation to the pump chamber feeding the beds.

This approach was taken at Bodiam Castle, with iron application on 16 July 2013 (i.e. mid-growing season; following the Leipzig meeting). Close inspection of Figure 1a reveals droplets of FeSO4 solution on the leaf surface, suggesting poor Fe uptake into the leaves. However, surface droplet appearance is consistent with leaf iron uptake [CA]. Eight weeks after application, no discernible decrease in chlorosis was evident. However, their was an improvement in plant health, the leaves appearing more turgid than at the time of iron spraying.

Alternative Plant Species

Other aquatic plants species growing in the same bed as chlorotic Phragmites australis exhibited normal growth appearance, notably Phalaris arundinacea (Bodiam Castle) and Iris pseudacorus [RK].

Figure 1a: Chlorotic Phragmites australis leaves showing yellow & green striation (Bodiam Castle, 16Jul13); note droplets of FeSO4 on leaf surface, following spraying

Figure 1b: Chlorotic Phragmites australis leaves showing increased susceptibility to rust attack (Bodiam Castle, 16Jul13)

Figure 2a: Bodiam Castle reed bed, during first growing season, with no effluent recirculation, showing dense, green reeds throughout (12Sep11)

Figure 2b: Bodiam Castle reed bed, third season, with effluent recirculation, showing less dense, yellow reeds throughout (even following ferrous sulphate foliar spray on 16Jul13) (22Aug13)

Figure 3a: Pont Abraham reed bed, during first growing season, receiving well aerated RBC effluent (diluted with groundwater infiltration), showing yellow reeds throughout (13Aug08)

Figure 3b: Pont Abraham reed bed, fifth season, receiving settled sewage (undiluted), showing dense, green reeds throughout (21Aug12)

Figure 4a: Floirac first and second stage French system reed bed (left & right, respectively), showing yellowing of leaves in second stage, compared to vigorous green leaves in first stage (courtesy Joëlle Paing)

Figure 4b: Hauterive second stage French system reed bed, May 2011, receiving vertical down-flow reed bed effluent treating raw sewage, showing some yellowing throughout bed (courtesy Joëlle Paing)

Figure 4c: Hauterive second stage French system reed bed, July 2012, showing poor health and disappearance of reeds creating significant reed-free areas (courtesy Joëlle Paing)

Figure 4d: Hauterive second stage French system reed bed, May 2013, almost complete disappearance of reeds within three years of first yellowing, with replacement by "weeds" (courtesy Joëlle Paing)

Figure 5a: Langenreichenbach horizontal flow forced aeration reed bed, receiving settled effluent: tall, green, dense reeds near inlet; short, yellow reeds further from inlet (1Jul11)

Figure 5b: Langenreichenbach horizontal flow forced aeration reed bed, receiving settled effluent: progressively shorter, yellower reeds further from inlet (13Jun13)

Discussion

Cause of Chlorosis

The specific characteristics of the leaf appearance reported are typical of iron deficiency [BS, CF]. Lack of iron is a well documented prime cause of chlorosis.

The observations of chlorosis correlate with *Phragmites* supplied with well oxygenated sewage effluent, likely to be high in redox potential, pH 7 or slightly above, with a relatively low BOD and low ammonia content.

The same plants growing in seasons both before and after periods of chlorosis (Bodiam Castle & Pont Abraham compact vertical flow reed beds) but fed with poorly oxygenated sewage effluent, with low redox potential, and relatively high ammonia yielded "normal" chlorophyll levels and were green throughout.

Ferrous iron (Fe^{2+}) is the predominant form of Fe in low redox potential situations and is readily soluble in water. Fe^{2+}, is, therefore, invariably available for biological uptake – eg into plant roots or leaves. Ferric iron (Fe^{3+}) predominates in higher redox potential environments (eg as a result of oxidation of Fe^{2+}) and is relatively insoluble, when it is not available for take up into plant cells.

The presence of protons (H^+) – enhances Fe^{3+} solubility.

Nitrification results in a release of H+. Therefore, the presence of ammonia in oxygen-rich conditions, which allow nitrification, tends to increase the bioavailability of iron, even when the iron has been oxidised to Fe^{3+} in the aerobic conditions.

The observations of Langenreichenbach (Figure 5) can, therefore, be explained (BOD_5, NH_4-N, NO_3--N & DO data from Tom Headley & Jaime Nivala, 2011, pers comm):

1. Settled effluent flows from one end of the bed to the other, at a depth of approx 1000mm
2. Aeration is provided evenly throughout the length of the bed (via subsurface compressed air supply)
3. Growing medium (ie sewage effluent) at the inflow end is relatively high in BOD_5 and NH_4-N, the microbial degradation of which sustains a relatively low DO and redox potential, despite the continuous oxygen supply, combined with H^+ release from nitrification – consistent with conserving Fe^{2+}, allowing sufficient iron uptake to ensure normal chlorophyll synthesis, and the observed green plants
4. As effluent proceeds along the bed, it becomes progressively lower in BOD_5 and NH_4-N, allowing a higher DO to pertain
5. In addition, NO^{3-}, produced from nitrification, is taken up by the *Phragmites* as a N source, with concomitant uptake of H^+
6. Therefore, passage along the bed results in the effluent iron being oxidised from Fe^{2+} to Fe^{3+}, which additionally encounters conditions of decreasing $[H^+]$, so falling out of solution and depleting the Phragmites of an essential nutrient; hence chlorophyll synthesis becomes progressively less, correlating with decreased root iron uptake, resulting in the observed decreasing plant height and increased chlorosis with distance from sewage inlet.

Explanation of the observed growth pattern, during a site visit in 2011 (Figure 5a), favoured progressive N source depletion, with N removal by denitrification suspected in the proximal (green plant) zone. This explanation has now been superseded. The symptoms of N-limitation in *Phragmites* are green but shorter plants, rather than chlorosis [BS].

Avoiding Chlorotic Phragmites

Dissolved Oxygen Concentration in Influent

The higher the DO concentration of the influent water, the more likely that iron will be present in the insoluble ferric form. One approach to preventing chlorosis due to iron deprivation is, therefore, to minimise the DO concentration of the influent. However, this is often undesirable, the high oxygen content being prioritised as an essential ingredient for effective microbial sewage treatment; or unfeasible, without negating other more important benefits, such as nitrate return to achieve denitrification. For instance, part of the rationale for final effluent recirculation at Bodiam Castle was to introduce DO to an earlier stage, thereby eliminating foul odour as well as enhancing treatment, including increased TN removal. Nevertheless, optimisation of the recirculation ratio should take account of the risk of encouraging downstream iron deprivation.

Ammonia Influent Concentration in

Phragmites australis is able to utilise both NH_4-N and NO^{3-}-N as nitrogen sources for growth [BS]. However, the presence of some ammonia is important, as although Phragmites can survive on nitrate alone, the pH of environments devoid of NH_4-N can remain consistently too high to allow iron solubility, most of the iron occurring as insoluble $Fe^{3+}(OH)^{3-}$, leading to chlorosis [CF]. Because nitrification releases H^+, positively influencing iron solubility, the presence of some ammonia in the influent feeding an aerobic reed bed is likely to prevent iron-limited chlorosis. A ratio of NH_4-N/NO_3-–N of 2:1 is ideal, although 1:1 is adequate to prevent chlorosis through iron depletion [BS].

The Bodiam Castle and Pont Abraham data suggest that influent NH_4-N >60mg/l will prevent chlorosis in vertical down-flow reed beds; while NH_4-N <10mg/l tends to allow iron depletion and chlorosis.

However, this simple relationship does not apply in all situations because of the effects of other solutes contributing to the redox potential, which strongly influences which iron species dominates and iron solubility. For example, high ammonia, high redox potential conditions may still prevent iron remaining in solution, leading to chlorosis.

However, chlorosis has been observed when *Phragmites* was grown in recirculated (ie low ammonia) low DO influent (eg at Vesterskovej, <5% DO [CA]). Thus, the most influential factor on chlorosis in reed beds may be low influent ammonia concentration (and associated raised pH), rather than high DO, or high redox potential.

Alternative Plant Species

Although *Phragmites australis* naturally inhabits a wide range of ecological and geographical situations, the favoured rhizosphere condition is anoxic. Therefore, selection of a plant evolved to thrive in low redox potential environments is at odds with serving wastewater treatment needs in consistently high redox potential situations.

The growth characteristics of *Phragmites australis* are ideally suited to VF beds, their relatively thin, evenly spaced stems allowing easy flow of effluent across the bed surface during intermittent distribution pulses, while growing at sufficient density to out-compete terrestrial plants, which would otherwise thrive in such beds. Wind-blown stem movement at the stem-sand interface has also been observed to enhance effluent permeation into the bed.

Phragmites australis roots grow to relatively great depth, as part of the propensity to inhabit deep anoxic mud. However, this unique characteristic has not been conclusively demonstrated to be a decisive advantage in VF beds and, especially when fed with high redox potential effluent, it is difficult to ascribe a treatment benefit to deep roots. Moreover, the species' rhizome and root system is severely morphologically compromised when grown in forced aeration reed beds.

From the above emerges a strong argument for using alternative species of emergent macrophytes in place of Phragmites australis. This approach has already been followed by installers of forced aeration beds, where Phragmites is avoided, commonly being replaced with *Typha* spp. and *Phalaris arundinacea*, among others.

Phalaris arundinacea has been growing well alongside *Phragmites* at Bodiam Castle as has *Iris pseudacorus* in many Danish vertical flow reed beds [RK]. Under controlled nutrient feed in the lab, DO 6-7mg/l, Phragmites australis experienced chlorosis when Phalaris arundinacea grew normally [KW]. It seems that Phragmites has a comparatively greater requirement for iron to enable healthy growth than many other emergent aquatics.

Despite their greater tolerance of iron-depleted feed water, however, neither Phalaris arundinacea nor Iris pseudacorus are ideally suited to the physical requirements of a vertical flow treatment bed, both forming clumps of stems, prone to obstructing the surface flow of effluent during intermittent distribution.

Alternative species under investigation include *Glyceria* maxima and *Schoenoplectus lacustris* (both inter-planted at Bodiam Castle in July 2013). *Schoenoplectus tabernaemontani* has been successfully used in New Zealand vertical flow treatment beds for some years [BS].

Foliar Application of Ferrous Sulphate

For existing treatment beds, replacing Phragmites with one or more alternative species may be unfeasible or disproportionately costly. The reported successful application of ferrous sulphate early in the growing season [CA, RK], resulting in green plant colouration throughout that growing season, offers a remedy for chlorosis in cases where Phragmites will continue to be deployed in low ammonia, high redox potential conditions (although this solution requires annual repetition).

Further Studies needed

- Characterise NH_4-N concentration, DO and redox potentials of wastewaters that result in chlorosis due to iron deprivation in Phragmites australis
- Identify optimum plant species for low ammonia and high DO, high redox potential influent situations – especially for vertical down-flow beds
- Fully characterise method of foliar application of bio-available iron to Phragmites australis, particularly ensuring optimal leaf take-up.

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Notes

Next issue:

Issue 19, April 2014: **„The CLARA project“**

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