Scientific and Technical Report No. 27

Wetland Technology

Practical Information on the Design and Application of Treatment Wetlands

Edited by Günter Langergraber, Gabriela Dotro, Jaime Nivala, Anacleto Rizzo and Otto R. Stein

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Contents

List of Abbreviations

BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
CSO	Combined Sewer Overflow
CSR	Corporate Social Responsibility
EDC	Endocrine Disrupting Chemicals
EO	Essential oil
ET	Evapotranspiration
FWS	Free Water Surface
GW	Greywater
HF	Horizontal Flow
HRT	Hydraulic Retention Time
HSE	Health, Safety and Environment
IWA	International Water Association
LW	Living Wall
MTBE	Methyl tert-butyl ether
MWR	Multi-functional Water Reservoir
NBS	Nature-Based Solution
NH_4-N	Ammonium Nitrogen
O&M	Operation and Maintenance
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Poly-Chlorinated Biphenyl
PE	Person Equivalent
PFAS	Poly-Fluorinated Alkyl Substances
PPCP	Pharmaceuticals and Personal Care Products

SAPS	Successive Alkalinity Producing System
STR	Scientific and Technical Report
STRB	Sludge Treatment Reed Bed
TKN	Total Kjeldahl Nitrogen
THC	Total Hydrocarbon Content
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
TW	Treatment Wetland
VDD	Vegetated Drainage Ditch
VF	Vertical Flow
WWTP	Wastewater Treatment Plant

Members of the IWA Task Group on Mainstreaming Wetland Technology

Gabriela Dotro, Cranfield University, United Kingdom (chair)
Günter Langergraber, BOKU University Vienna, Austria (co-chair)
Jaime Nivala, Helmholtz Center for Environmental Research (UFZ), Germany
Jaume Puigagut, UPC Barcelona, Spain
Otto R. Stein, Montana State University, USA

Editors and Authors

Editors

Name	Affiliation
Langergraber, Günter	BOKU University Vienna, Austria
Dotro, Gabriela	Cranfield University, United Kingdom
Nivala, Jaime	Helmholtz Center for Environmental Research (UFZ), Germany
Rizzo, Anacleto	Iridra S.r.l., Florence, Italy
Stein, Otto	Montana State University, USA

Authors

Name	Affiliation	Section Numbers
Anconelli, Stefano	Consorzio di Bonifica Canale Emiliano Romagnolo, Bologna, Italy	6.3
Arias, Carlos	Aarhus University, Denmark	5.10, 5.11,
Austin, David	Jacobs, USA	5.8
Breuer, Roman	Bauer Nimr LLC, Muscat, Oman	6.5
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Di Luca, Gisela A.	Universidad Nacional del Litoral (UNL) & CONICET, Argentina	6.11

(*Continued*)

Name	Affiliation	Section Numbers
Dotro, Gabriela	Cranfield University, UK	1.1–1.4, 5.4, 5.11, 6.1
Eregno, Fasil Ejigu	The Arctic University of Norway (UiT), Faculty of Engineering Science and Technology, Narvik, Norway & Norwegian Institute of Bioeconomy Research (NIBIO), Ås, Norway	3.1–3.4, 4.8
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Headley, Tom	Wetland & Ecological Treatment Systems, Australia	4.10, 5.5, 5.9
Heistad, Arve	Norwegian University of Life Sciences (NMBU), Ås, Norway	3.1–3.4
Istenič, Darja	University of Ljubljana, Slovenia	4.7, 4.13, 5.10, 6.9
Jefferson, Bruce	Cranfield University, UK	5.11
Kõiv-Vainik, Margit	University of Tartu, Estonia	5.11
Krivograd Klemenčič, Aleksandra	University of Ljubljana, Slovenia	4.13, 6.9
Langergraber, Günter	BOKU University Vienna, Austria	1.1–1.4, 2.1–2.3, 3.1–3.4, 5.2, 6.1
Lavrnić, Stevo	University of Bologna, Italy	4.5, 6.3
Lecciones, Aaron Julius M.	University of the Philippines Los Baños & Society for the Conservation of Philippine Wetlands, Inc., The Philippines	6.10
Lecciones, Amy M.	Society for the Conservation of Philippine Wetlands, Inc., The Philippines	6.10
Maine, María A.	Universidad Nacional del Litoral (UNL) & CONICET, Argentina	6.11
Marzo, Alessia	University of Catania, Italy	4.7, 4.11.4, 4.11.5, 5.4
Masi, Fabio	Iridra Srl, Florence, Italy	2.1–2.3, 4.11.6, 4.16, 5.2, 5.12, 6.6, 6.7, 6.8
Mæhlum, Trond	Norwegian Institute of Bioeconomy Research (NIBIO), Ås, Norway	4.8
Meney, Kathy	Syrinx Environmental, Australia	6.4
Milani, Mirco	University of Catania, Italy	4.7, 4.11.4, 4.11.5, 5.4
Molle, Pascal	IRSTEA, Villeurbanne, France	5.3, 5.11
Mufarrege, María M.	Universidad Nacional del Litoral (UNL) & CONICET, Argentina	6.11
Nielsen, Steen	Orbicon, Denmark	4.6, 5.6

(*Continued*)

Name	Affiliation	Section Numbers
Nivala, Jaime	Helmholtz Center for Environmental Research – UFZ, Germany	1.1–1.4, 4.8, 4.9, 6.1
Pálfy, Tamás Gábor	University of Sopron, Hungary & IRSTEA, Villeurbanne, France	4.4
Pantelic, Ljiljana	Syrinx Environmental, Australia	6.4
Paruch, Adam M.	Norwegian Institute of Bioeconomy Research (NIBIO), Ås, Norway	4.8
Platzer, Christoph	Rotaria do Brasil, Brazil	4.2
Prigent, Stephane	Bauer Nimr LLC, Muscat, Oman	6.5
Prudente, Ma. Cheryl F.	Society for the Conservation of Philippine Wetlands, Inc., The Philippines	6.10
Pucher, Bernhard	BOKU University Vienna, Austria	5.2
Quintos, Jose Carlo H.	Society for the Conservation of Philippine Wetlands, Inc., The Philippines	6.10
Regelsberger, Martin	Technisches Büro Regelsberger, Gleisdorf, Austria	2.1–2.3, 3.1–3.4, 4.16
Rizzo, Anacleto	Iridra S.r.l., Florence, Italy	4.4, 4.11.6, 4.16, 5.12, 6.6, 6.7, 6.8
Rous, Vit	Czech University of Life Sciences, Prague, Czech Republic	4.11.2, 5.2
Sánchez, Gabriela C.	Universidad Nacional del Litoral (UNL) & CONICET, Argentina	6.11
Schwarzer, Claudia	Bio Piscinas Lda., Aljezur, Portugal	4.15
Schwarzer, Udo	Bio Piscinas Lda., Aljezur, Portugal	4.15
Solimando, Domenico	Consorzio di Bonifica Canale Emiliano Romagnolo, Bologna, Italy	6.3
Stefanakis, Alexandros	Bauer Nimr LLC, Muscat, Oman	4.6, 4.11.1, 4.11.3, 4.12, 5.2, 5.5, 5.6, 5.7, 6.5
Stein, Otto	Montana State University, USA	1.1–1.4, 6.1
Tondera, Katharina	IMT Atlantique, Nantes, France	4.3, 4.4, 5.9, 6.2
Toscano, Attilio	University of Bologna, Italy	4.5, 6.3
Troesch, Stéphane	Eco Bird, Chaponost, France	4.3
van Oirschot, Dion	Rietland bvba, Belgium	5.7
von Sperling, Marcos	Federal University of Minas Gerais, Brazil	4.2
Vymazal, Jan	Czech University of Life Sciences, Prague, Czech Republic	5.4
Wallace, Scott	Naturally Wallace Consulting, USA	5.7, 5.8
Weber, Kela P.	Royal Military College of Canada, Kingston, Canada	4.9
Yang, Lei	National Sun Yat-sen University, Kaohsiung, Taiwan	4.14

Preface

The IWA Task Group on Mainstreaming Wetland Technology

The IWA Task Group (TG) on Mainstreaming the Use of Treatment Wetlands was initiated in 2013 by Gabriela Dotro, Günter Langergraber (at that time Secretary of the IWA Wetland Systems Specialist Group) and Fabio Masi (at that time Chair of the IWA Wetland Systems Specialist Group). Initial discussions showed that the topic of treatment wetlands was not well covered in traditional wastewater treatment courses and that there was a lack of proper teaching material on treatment wetlands. The main objectives of the Wetlands TG were defined as:

1. Updating and enhancing of the *IWA Scientific and Technical Report (STR)* on *Wetland Technology* that had been published in 2000;
2. Developing a new textbook on wetland technology for the "Biological Wastewater Treatment in Warm Climate Regions" series; and
3. Organising workshops to increase collaboration with closely related IWA groups.

Already in the proposal stage, the aim was to publish the work of the Wetlands TG as Open Access material.

The TG was approved by IWA's Strategic Council at their meeting prior to the IWA World Water Congress 2014 in Lisbon and given a starting date of 1 July 2015.

Activities

The first meeting of the Wetlands TG took place in September 2015 in York, UK. During that first meeting, we decided to first focus our work on the textbook and only after finishing it would we start on the STR.

For the preparation of the textbook, we held three working meetings (December 2015 in Vienna, Austria; March 2016 in Lisbon, Portugal; and July 2016 in Leipzig, Germany) and had a final draft version ready for review at the IWA Wetland Systems conference in October 2016 in Gdansk, Poland. After review from 10 wetland experts and revision, we published the textbook as an Open Access e-Book in 2017:

> Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O.R. and von Sperling, M. (2017). Treatment Wetlands. *Biological Wastewater Treatment Series, Volume 7*, IWA Publishing, London, UK, 172p. eISBN: 9781780408774. Available for download at https://www.iwapublishing.com/open-access-ebooks/3567.

The target audience for the textbook is bachelor-level students with basic knowledge of biological wastewater treatment, as well as practitioners seeking general information on the use of treatment wetlands. The chapters focus on the main types of treatment wetlands for domestic wastewater applications.

In addition to the e-Book, the Wetlands TG was involved in organising the following workshop:

> "Role of nature-based systems in decentralised approaches for linking sanitation to energy and food security" at the 2nd IWA Resource Recovery Conference held in New York City in August 2017.

As an outcome of the workshop the following mini review was published:

> Langergraber, G. and Masi, F. (2018). Treatment wetlands in decentralised approaches for linking sanitation to energy and food security. *Water Science and Technology* **77**, 859–860. https://iwaponline.com/wst/article/77/4/859-860/39083.

The Scientific and Technical Report

The work on the STR started with a survey among the Wetland Systems SG members (about 140 SG members took part in the survey). In February 2018, a three-day workshop with approximately 20 participants was organised at BOKU University Vienna in Austria to kick off the STR work. During the workshop the main structure of the book and responsibilities for chapters were defined.

The main outcomes were that the textbook should be the basis of the STR, and that the content of the STR should include useful information for practitioners and researchers aiming to design treatment wetlands.

After the IWA Wetland Systems conference in October 2018 in Valencia, Spain, a meeting on the STR was organised in which the status of the work was reviewed and responsibilities for the remaining work were refined.

The editors had the task to compile the new *Scientific and Technical Report* on *Wetland Technology.* The STR includes contributions from more than 50 wetland colleagues from academia and practice. The editors are very thankful for all the enthusiasm and effort put into this work and we hope that the STR will be a useful addition to the treatment wetland literature.

We hope you will enjoy this *Scientific and Technical Report* on *Wetland Technology*.

Günter Langergraber
Gabriela Dotro
Jaime Nivala
Anacleto Rizzo
Otto R. Stein

Chapter 1

Introduction

Günter Langergraber[1], Gabriela Dotro[2], Jaime Nivala[3] and Otto R. Stein[4]

[1]Institute for Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Muthgasse 18, A-1190 Vienna, Austria
[2]Cranfield Water Science Institute, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK
[3]Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany
[4]Civil Engineering Department, Montana State University, Bozeman, MT, USA

1.1 RATIONALE

Treatment wetlands (TWs) are natural treatment technologies that efficiently treat many different types of water. They are used worldwide and have gained increasing popularity during recent decades as they require less operational effort compared with other solutions for wastewater treatment.

In the textbook volume *Treatment Wetlands* (Dotro *et al.*, 2017) the main types of treatment wetlands for domestic wastewater applications were described. Bachelor students with a basic knowledge on biological wastewater treatment, as well as practitioners seeking general information on the use of treatment wetlands were the main target audience for this work. In this new Wetland Technology STR the information already presented in the *Treatment Wetlands* textbook will not be repeated.

The "old" wetlands STR (Kadlec *et al.*, 2000) was structured like a textbook. After producing the above-mentioned textbook, the Wetlands TG did not want to simply update the previous STR and make another textbook. Thus, the focus of this new *Wetland Technology* STR is to provide **practical information on design of treatment wetlands that is simple to use**.

The main content, i.e. the technical parts, is framed in a wetland design approach considering that:

- Treatment wetlands are designed for improving water quality for a specific purpose;
- Treatment wetlands are designed within a productive system; and
- Treatment wetlands are designed as multi-purpose systems.

1.2 WHO SHOULD READ THIS STR?

The primary target audiences for this STR are **engineers focusing on wetland design** (including graduate students as future designers) as well as academics. Secondary target audiences include decision-makers and people from a non-water technical background who have an interest in wetland technology and its potential.

1.3 STRUCTURE OF THIS STR

After this *Introduction*, the STR continues with:

Chapter 2: *Why use treatment wetlands?, w*hich outlines the new approach to water management and the roles of wetlands within this new approach.

Chapter 3: *Design approach for treatment wetlands,* which outlines the treatment wetland design approach in which, as a first step, the treatment objectives are defined. In a second step, the processes that are required to reach the treatment objectives are identified. The third and final step helps to choose the TW type(s) with which the treatment objectives can be achieved. Besides selecting the right TW type, other important considerations need to be made in the design process that are summarised in this chapter.

Chapter 4: *Designing wetlands for specific applications,* which outlines the design of TWs following this approach for 15 different applications (e.g., stormwater treatment) and/or treatment objectives (e.g., removal of pathogens).

Chapter 5: *Practical information on design of specific wetland types and typical pitfalls,* which includes practical information related to treatment wetland design for 11 TW types.

Chapter 6: *Case studies,* which includes a checklist for reporting treatment wetland data (related to the information required on the TW type and reporting experimental data) and presents 10 case studies of treatment wetlands for various applications.

References: Includes the complete list of references used in the STR.

1.4 HOW TO USE THIS STR

As mentioned before, the content of the STR builds upon the content of the *Treatment Wetlands* textbook. Consequently, we also use the notation that was introduced by Dotro *et al.* (2017) for TW main types:

- **VF wetlands** (for vertical-flow wetlands),
- **French VF wetlands** (for the variant of VF wetland developed in France for treating raw wastewater),
- **HF wetlands** (for horizontal-flow wetlands), and
- **FWS wetlands** (for free water surface wetlands).

General information on treatment wetlands is not provided in this STR. For this, the user is referred to the *Treatment Wetlands* textbook. This *Wetland Technology* STR provides information on design of treatment wetlands that should be useful in practice.

If the reader is interested in using a treatment wetland for a specific application and/or treatment objective, he/she is referred to Chapter 4 in which the design of wetlands for 15 such applications and/or treatment objectives is described.

If the reader aims to get more information on a specific TW type, he/she is referred to Chapter 5. In that chapter detailed information on designing TWs in practice is presented for 11 TW types, including information on the four TW main types that is beyond the information that was presented in the *Treatment Wetlands* textbook chapter.

Last but not least, 10 case studies of full-scale treatment wetlands in Chapter 6 highlight different applications and sizes of treatment wetlands.

Chapter 2

Why use treatment wetlands?

Martin Regelsberger[1], Fabio Masi[2] and Günter Langergraber[3]

[1]Technisches Büro Regelsberger, Marburger Gasse 11, 8200, Gleisdorf, Austria
[2]Iridra Srl, via La Marmora 51, 50121, Florence, Italy
[3]Institute for Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Muthgasse 18, A-1190 Vienna, Austria

2.1 NEW APPROACH TO WATER MANAGEMENT

Wastewater is a historical development. Current approaches to wastewater treatment result from a combination of a need to protect public health (limiting human contact with waste) and the belief that we can dispose of things on this planet. It is also based on the idea that we can taint things and fix them later. In the case of wastewater this means mixing together whatever comes along, only to separate it at the end of a long pipe in a treatment plant, or at least separate water from everything else in order to release the water back into the natural environment, causing "limited" negative impact, where the definition of limited is entirely dependent on what is accepted at any given time and place.

This concept of disposal of treated water into the aquatic environment is the main goal of wastewater systems and, with few exceptions, all regulations have this goal in mind, even if it is not explicitly mentioned. The approach worked as long as we considered the planet as boundless for us. With the growing number of human beings and their influence on the surface of Earth this is no longer true. We are increasingly realising that we cannot get rid of substances which are not metabolised and reintegrated into natural cycles harmlessly. Simultaneously we have discovered that extracting resources and discarding them after a single use has become too inefficient for our needs and the available offer on Earth. Both aspects are illustrated by footprint or Earth overshoot day calculations, which show that our present behaviour needs more space than is available on this one Earth or, expressed in time, that the resources available per year fall far short of lasting until the end of the year at present rates of consumption. We are therefore in search of a new way to use resources, not least those related to water, which comprise water itself, plant nutrients, carbon and energy.

In a first step the conventional boundaries between different aspects of water – water supply, wastewater disposal and urban drainage – are expected to disappear. “The complex water issues are intertwined and cannot be sustainably solved by the traditional siloed water management approaches” (Ma *et al.*, 2015). Thus, for any given water need the best and most effortlessly available water source can be used. Treatments will be applied to various types of waters and for different purposes, each with its own requirements, making the particular water source fit for the next purpose.

In the future, treatment of water will always involve the definition of a further use of that water, determining the treatment needs. While all wastewater has to be treated, the reflection on the supply side will also need a water balance and an examination of all available water streams beginning at the source. To optimise the reuse potential it may be useful to segregate such streams and treat them separately. At the same time that may lead to scale considerations to find the best size of collection, treatment and distribution systems for a particular reuse option. This may result in systems of very different scales simultaneously: a water supply scheme for a metropolitan area, domestic and industrial wastewater treatments of various sizes from municipal to one particular production process down to greywater (i.e. all the wastewater except those from toilets) treatment for one building producing service water for toilet flushing, garden irrigation and even laundry in that same building. The Water Supply and Sanitation Collaborative Council postulated in 2000 at its Bellagio meeting that the household is the basic unit at which to start examining water issues, with the aim of solving every issue at the smallest possible scale, from household to entire country, optimising the possible solutions in repeated cycles. This was named the “household-centred approach” (EAWAG-SANDEC & WSSCC, 2000). It was initially conceived for developing countries, but is applicable everywhere.

In an additional step, water use optimisation will be achieved by considering the entire urban metabolism. That would mean including all water aspects and all related substances into an integrated urban material flow management. The key characteristic is to consider all material and energy flows as a system in order to optimise that system as a whole, and to proceed according to the general principles of material flow management (Figure 2.1) or the classical three Rs: Reduce, Reuse, Recycle. The shift from supply, drainage and treatment of water to a material flow management approach will open entirely new possibilities in terms of reduction, its first and most important element, far beyond conventional water saving and efficiency increases. This will be achieved by considering all water sources, but also other collection and transport options beyond water. Reduction of water use will become an integral part of a green economy, based on the three key aspects of sufficiency, i.e. what is really needed, consistency with nature of all steps involved, and efficiency as the last element, once the first two have been consecutively completed.

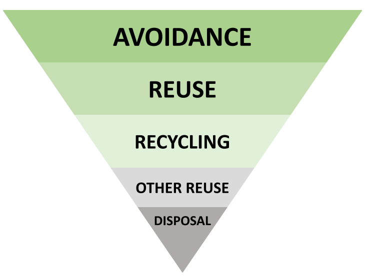

Figure 2.1 Hierarchy of measures for material flow management.

Additionally, a city does not consist of material and energy flows alone. It is built infrastructure, arranged around people and their needs. Jan Gehl therefore requests "Cities for People" (Gehl, 2010), respecting a "human scale".

Cities are complex systems of people, physical fabric and functions. While the present urban system works, neither the cities nor their inhabitants are sustainable. However, "achieving the vision of lively, safe, sustainable and healthy cities has become a general and urgent desire" (Ma *et al.*, 2015). We could add that cities have to become sustainable to thrive within the known planetary boundaries (Steffen *et al.*, 2015). This will only be achievable if the system is addressed in its complexity. The relations between its elements have to be examined and optimised, and the resource flows balanced, in a systemic approach.

With respect to water this means it has to be seen in connection with "urban green" to lead to blue–green solutions. The built environment in combination with these blue and green features should allow the characteristics of the natural water balance to be kept, in terms of infiltration, retention, evapotranspiration and run-off.

The blue–green nature-based "infrastructure" must be linked to urban space use and green infrastructure planning. Urban green will host urban food production in a future with green mobility, linking water professionals to agriculture and traffic, while at the same time providing for biodiversity and nature-based solutions (NBS) for urban services instead of grey infrastructure. Simultaneously the needs and potential of the people living in the cities and using the water and the blue and green infrastructure must be considered, which means co-development of solutions by all major actors with the assistance of sociologists and experts in participatory processes.

2.2 ROLE OF WETLANDS IN THE NEW APPROACH

Treatment wetlands are nowadays a well accepted technology for the treatment of different types of wastewater. Additionally, TWs are increasingly used for other purposes. The new approach in dealing with water, however, with respect to all the issues detailed in the preceding section, is introducing entirely new applications and new requirements for TW design. The need to produce water from any of a range of different possible sources that is fit for a particular purpose will require different treatment targets rather than just discharging a mixed treated wastewater stream into a final sink (freshwater or soil). TWs also must fit into the urban fabric and provide additional ecosystem services and benefits beyond producing water. Thus, the following main urban applications can be identified (Masi *et al.*, 2018):

- Water reuse:
  - Greywater treatment (outdoor, indoor) for local reuse and recreational purposes, possibly as the only liquid treatment, while excreta are collected and processed separately (Masi *et al.*, 2010, 2016);
  - Rainwater (including first flush) treatment and storage (Nolde, 2007) for domestic or industrial purposes, or irrigation of urban green, including food production;
  - Combined Sewer Overflow (CSO) treatment and storage, also to prevent spreading of persistent organic pollutants (Meyer *et al.*, 2013);
  - Treatment of persistent organic molecules in low concentrations for water reuse (Matamoros *et al.*, 2016; Verlicchi & Zambello, 2014);
  - Polishing of secondary treated WW, as long as these still exist, for reuse (Ayaz, 2008; Rousseau *et al.*, 2008).

- Nutrient recovery:
    - TWs as pre-treatment for fertigation (disease vector reduction, separation of liquid and solid phase);
    - Biomass production from secondary sludge (as long as such sludge is still produced), digestate or primary sludge;
    - Biomass production by harvesting TW vegetation, further used as pelletized slow-releasing soil amendment/fertiliser.
- Energy production:
    - Anaerobic reactor (biogas) + TW as polishing stage;
    - TWs as biomass production plots (Avellán & Gremillon, 2019).
- Ecosystem services:
    - Multi-purpose TWs for rainwater buffering or storage, recreation and wetland ecosystems;
    - Re-adaptation of ornamental green areas in terms of ecosystem services (green roofs, green walls, indoor green areas, roundabouts, sidewalks, parks, permaculture productive areas) comprising organic food production in integrated habitats.

A very interesting factor to be noted is that for the above-mentioned targets there are specific configurations of TW systems and combinations of TWs with other technologies available that can perform better or be more efficient in economic terms than others. This will be given particular attention in Chapters 4 and 5.

This list highlights the fact that TWs can help to close loops or at least use substances in cascades in various ways. They can treat water for a certain next purpose, e.g. domestic, industrial, irrigation of urban green or crops. They can be used to recover other substances for their further use or to extract and trap hazardous or recalcitrant substances, thus increasing the possible usages of the treated water and control of the spread of harmful substances around the planet (see global distillation theory). Finally, they are productive systems in themselves, producing biomass, providing organic matter (especially TWs for or comprising sludge treatment), cooling through evapotranspiration, providing habitats, etc.

The integration of nature-based water retention and treatment systems in the urban fabric is enormously enlarging the potential number of applications of TWs, even more so if the concept of "retention" is not only thought of in terms of flood risk reduction, but also considering the trapping of nutrients and organic compounds, in particular the emergent and more persistent and hazardous ones. This sector is the most obvious for the need to involve a large variety of competences to take optimal advantage of the multiple potential benefits of the installations. Such advantages comprise increasing the water retention capacity of a city, to locally bolster biodiversity by offering habitats for wildlife, to work as a last barrier and interface between settlements and water bodies (i.e. adsorbing persistent organic pollutants), to create enjoyable spaces and recreation areas, to reduce air pollution and to contribute to climate change adaptation or even mitigation. Other applications of TWs can offer perhaps not the full panoply but, in each case, at least some of the additional benefits of nature-based solutions, if designed properly and taking these benefits into consideration, both with regard to the necessary competences in the team involved as well as to the outcome. The concept of nature-based solutions should therefore become a core principle in every urban planning process, spreading multi-purpose green infrastructure in our cities (Liquete *et al.*, 2016; Masi *et al.*, 2018) in order to make their benefits available everywhere.

Thus, beyond the already well established designs of treatment wetlands at the downstream fringe of settlements and their wastewater pipes or CSOs, they can be implemented at many other places.

- In buildings
- On buildings (roofs, facades)
- Next to buildings in backyards or gardens,

- Along streets, as additional green areas storing and treating water
- In parks
- Along rivers and other natural features
- Downstream of agricultural areas, including urban agricultural land as buffer strips
- Integrated into existing treatment plants, as polishing stages, the main treatment stage, or for sludge treatment.

A few particular advantages of TWs are their flexibility in size, with little economy of scale, their simple maintenance requirements, demanding skills very similar to widespread irrigation systems, and the very limited to no disturbance that most applications cause in their immediate vicinity if properly designed and operated. This combination of characteristics allows a high flexibility in size, location and vicinity of their implantation and makes them particularly appropriate for urban applications.

2.3 THE NEW DESIGN APPROACH FOR WETLANDS

Based on the previous chapter, we propose that when designing a treatment wetland, the following steps shall be followed:

(1) Define the treatment objective(s).
(2) Define the processes required to reach the treatment objective.
(3) Choose the proper treatment wetland type, or a combination of different types, that allows to reach the treatment objective.

This new design approach will be elaborated in more detail in Chapter 3.

Chapter 3

Design approach for treatment wetlands

Günter Langergraber[1], Martin Regelsberger[2], Fasil Ejigu Eregno[3,4] and Arve Heistad[5]

[1]Institute for Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Muthgasse 18, A-1190 Vienna, Austria
[2]Technisches Büro Regelsberger, Marburger Gasse 11, 8200, Gleisdorf, Austria
[3]The Arctic University of Norway (UiT), Faculty of Engineering Science and Technology, Narvik, Norway
[4]Norwegian Institute of Bioeconomy Research (NIBIO), Division of Environment and Natural Resources, Ås, Norway
[5]Norwegian University of Life Sciences (NMBU), Faculty of Science and Technology, Ås, Norway

3.1 DEFINE THE TREATMENT OBJECTIVES

Treatment wetlands have one main objective, i.e., treating water to make it suitable for a certain purpose. Other objectives, besides treating water can be:

- Retaining water to store it to later evapotranspire it or attenuate flood waves;
- Evapotranspirating water, which is key for sludge treatment wetlands, but also for cooling and reducing urban heat island effects;
- Producing biomass;
- Harvesting nutrients;
- Creating a nice landscape, including for recreational purposes;
- Enhancing ecosystem services (mainly for FWS wetlands);
- Fostering biodiversity, directly or by creating habitats.

This list is not exhaustive but shows some of the additional benefits. TWs can be designed for a single objective, which then would be just to treat water, or with multiple objectives, whereby treating water is always included. Engineers should seek multi-objective solutions.

The purpose for which treated water should be utilised defines the treatment objective. For example, if treated water is to be used for irrigation purposes, it makes less sense to remove nutrients that are beneficial for crop fertigation.

However, restrictive regulations in various countries often obstruct the producing of effluent with a desired quality for a particular purpose. The full potential of circular management of water and substances will therefore only be possible after a revision of the respective guidelines. Such a revision should aim at protecting water users and the consumers of products that have come into contact with the "reused" water, but also eliminating unnecessary obstacles. A zero-risk approach, as applied e.g. in Italy for treated wastewater for irrigation, leads to difficulties in spreading this practice. A different view of the same concern is offered by the World Health Organization, which proposed a pragmatic approach based on microbial risk assessment, evaluating case by case the pathogen reduction for treated wastewater to be used in agriculture, and how to achieve this (Licciardello *et al.*, 2018).

3.2 WHICH PROCESSES DO WE NEED TO REACH THE DESIGN OBJECTIVES?

Once the design objectives are defined, the designer needs to identify the processes that are required to deliver them. Table 3.1 summarises the most significant processes required to reach typical treatment objectives. As the main treatment objective is improving water quality, most processes are related to this aspect.

3.3 WHICH TW TYPE CAN BE USED TO REACH THE SPECIFIED OBJECTIVES?

Table 3.2 summarises the processes occurring in the main TW types. A '++' indicates that this process is a primary process in this TW type, meaning that the TW type is primarily designed in a way that this process occurs. For instance, if nitrification is required, only TW types with vertical flow (VF) and intermittent loading can be used, i.e. classical VF wetlands and French VF wetlands. A '+' or 'o' indicates that the process occurs to some extent, but that the TW type is not primarily designed for this process.

3.4 OTHER IMPORTANT DESIGN ASPECTS

During the design of TWs additional important aspects have to be taken into account. These are:

- *Considering malfunctioning*. Designers have to consider situations in which the system is not working in the way it was designed, e.g. when pumps break or when filter beds become clogged. A major challenge that has to be considered is that inflow water still needs to pass through the system without causing severe damage. Two typical strategies are bypasses and redundant structures: overflows within pump sumps or wetland beds could be one way, or planning several treatment lines in parallel so that if one is offline, the wastewater can still be treated by the other lines. Risk considerations must be given particular attention when there is not only a treatment but also a supply commitment, either in terms of quantity or quality or both, that must be complied with.

Table 3.1 Processes required to reach specific design objectives.

Objective	Processes
Improve water quality	
Removal of solids	Filtration Sedimentation
Removal of dissolved organic matter	Aerobic degradation Anaerobic degradation
Removal of ammonia	Nitrification Adsorption
Removal of nitrogen	Denitrification after nitrification Plant uptake
Removal of phosphorus	Adsorption Precipitation Plant uptake
Removal of microbial contamination	Filtration Disinfection
Removal of organic micropollutants	Biological degradation Adsorption
Removal of metals	Sorption Plant uptake Precipitation
Remove water/reduce water content	Evaporation Evapotranspiration
Recover energy from biomass	Biomass production
Enhance biodiversity	Creation of habitats

Table 3.2 Processes in TW main types.

TW Type/Processes	Sedimentation	Filtration	Aerobic Degradation	Anaerobic Degradation	Nitrification	Denitrification	Adsorption	Sorption	Precipitation	Plant Uptake	Evaporation	Biomass Production	Creation of Habitats
VF wetland		++	++		++		+	+				+	+
French VF wetland	+	++	++		++		+					+	+
HF wetland		++	o	++		o	+	+	o		+	+	+
FWS wetland	++	+	+	+	+	+			o	+	o	+	++
Sludge treatment wetland	+	++	++								++		+
Aerated wetlands		++	++		++		+		o			o	o

- *Operation and maintenance*. Operation and maintenance of the system must be considered during the planning phase. These considerations include:
  - o Requirements for removing the sludge from the primary treatment unit (e.g. frequency, method for sludge or solid waste transport, treatment and reuse/disposal);
  - o The required maintenance for the wetland plants (e.g. frequency and timing of harvesting/cutting of vegetation, further use);
  - o General responsibilities and tasks for routine operation, monitoring and maintaining of the wetland system, including the preparation of a user-friendly operation manual and operational materials including (but not limited to) checklists and logbooks;
  - o The expected running time before major intervention will be required (e.g., removal of accumulated sludge from wetland surface) and the type of intervention it will require (e.g., digging and cleaning media, surface scrapping, replanting); as well as
  - o Access to the facility for major maintenance and repair work if required.
- *Monitoring of treatment wetlands*. Considerations for future monitoring of the TW should ensure that:
  - o Sampling locations must be present and easy to access;
  - o Sampling and analysis required for routine monitoring to ensure the proper operation of the system is clearly defined (frequency, location and parameters); and
  - o External requirements for sampling and analysis to fulfil legal obligations are met.
- *Construction phase*. Considerations important for the construction phase include, e.g.,
  - o The shape of the terrain and possible constraints such as the presence of power lines, gas pipes, railways, roads, riverine buffer zones, etc.;
  - o The local availability of sand/gravel required for the filter bed in the physical and chemical quality and granulometry required,
  - o The capacity of local workers available for welding plastic polymer liners,
  - o The availability of wetland plants (amount, species, etc.),
  - o The proper planning of the time schedule so that all materials are available on site when needed.
- *Health, Safety and Environment (HSE)*. HSE means a systematic process of identifying the impact of wetland technology projects related to health, safety and the environmental conditions that may occur during the construction and operational phases, along with recommendations for their management. Potential risks occur in different phases of the project:
  - o *The construction phase*. The Construction Design and Management Regulations 2015 (CDM, 2015) offer guidelines that broadly prescribe the general duties for employers, employees and the self-employed, and is useful for wetland technology construction sites. The fundamental principles that have been adopted in many countries around the world include (Aboagye-Nimo *et al*., 2018; CDM, 2015):
    - – Proper planning and coordination need to be undertaken from the beginning of the project
    - – Safety and health must be considered throughout the project
    - – All persons who contribute to the health and safety of a wetland technology project need to be included
    - – Those in charge of the provision of health and safety need to be professionally competent
    - – Communication and sharing of information between all parties must be undertaken
    - – A record of safety information for future use must be made.

    Early implementation of HSE principles is essential to the success of a construction project and can prevent negative consequences. All stakeholders, including the owners, have a duty to ensure works and activities are carried out under safe conditions (Aboagye-Nimo *et al*., 2018)

- *The operational phase*. Operation, maintenance and water reuse require planned strategies that incorporate multiple measures to minimise risks to public health and the environment. The WHO *Sanitation Safety Planning Manual* (WHO, 2015) can be used as a reference to identify potential hazards and define measures to prevent these.
- *Workers' safety*. Workers at TWs are exposed to hazardous chemical constituents and biological agents contained within the wastewater and in the biofilm during their work. Appropriate design of facilities, training of workers, proper use of personal protective equipment, and careful attention to personal hygiene can all greatly reduce the likelihood of exposure to hazardous chemicals, biological agents, wastewater and injury (Brown, 1997; NIOSH, 2002). These include:
  - Avoiding direct contact with wastewater – carefully wash the hands and face with soap and water after contact with wastewater and before eating, drinking or smoking
  - Avoiding touching face, mouth, eyes, nose, genitalia, or open sores and cuts, or nail-biting with dirty hands while working
  - Use of appropriate protective clothing (coveralls) and personal protective equipment (e.g. boots, gloves) and wearing respiratory protective equipment
  - Thoroughly cleansing all exposed injuries with soap and water and keeping them covered with a bandage (preferably waterproof) while at work, and seeking medical attention immediately after suffering cuts or penetrating injuries
  - Removing personal protective clothing and footwear at the end of shift, changing out of work clothes and taking a shower before leaving work and contact with other people.

- *Decommissioning of the TW system*. Each treatment system has a specific lifetime. Considerations on what to do once the lifetime is reached or the treatment system is no longer needed and is to be taken out of operation should be included.

Chapter 4

Designing wetlands for specific applications

4.1 INTRODUCTION

In this chapter the design approach, as was presented in Chapter 3, is used for 15 different applications or treatment objectives. Wetlands treating domestic wastewater are not described in this chapter, as this main application is already described in various textbooks (e.g., Dotro *et al*., 2017; Kadlec & Wallace, 2009).

The general structure of the sub-chapters is as follows:

(1) The design objective(s) are defined.
(2) The processes required to reach the design objective(s) are discussed, and based on this the selection of the TW type is discussed.
(3) Specific considerations during design and construction for each application are additionally mentioned.

4.2 TREATMENT WETLANDS IN DEVELOPING REGIONS

Marcos von Sperling[1] *and Christoph Platzer*[2]

[1]*Federal University of Minas Gerais, Brazil*
[2]*Rotaria do Brasil, Brazil*

4.2.1 Introduction

In the present context, developing countries or regions are those that are characterized by low income, and as such have limited resources for infrastructure implementation, operation and maintenance. Several developing countries show high regional economic contrasts, with technically developed areas coexisting with poor regions, but the focus here remains only on those with limited financial resources.

Even though the working principles of TWs in developing regions remain the same as for those in developed areas, there are several aspects that should be taken into account in the planning, design and operation of the treatment systems.

Another relevant aspect to be considered here is that many developing regions in the world are in warm-climate areas. The climatic factor needs to be taken into account in the design of wetlands in these regions. Again, the working principles of the treatment system will be the same, but specific characteristics need to be taken into account.

In this section, the development status and climatic factors are in many cases intertwined. However, of course in the world there are developed areas in warm regions, and also developing regions in temperate or cold areas. Whenever possible, mention of the influence of development level and climate will be made clear here.

4.2.2 Specific considerations during planning, design, construction and operation

The following aspects should be taken into account when planning, designing, constructing and operating treatment wetlands in developing regions, with additional considerations regarding the possible warm-climate conditions.

(a) Aspects related to regional development status

- *Need for low capital and operational costs (CAPEX and OPEX, respectively).* In regions with limited financial resources, it is essential that construction costs are small, so that the implementation of the treatment systems becomes viable. Additionally, operation and maintenance (O&M) costs must also be low, in order to guarantee that the plant will be sustainable in the long run, and not become neglected because of lack of funds. In many cases in developing countries funding for the implementation of the treatment plant comes from a state or international agency (frequently with financing at low interest rates), but O&M costs are taken over by the operator or service provider, and this may be affected by the tariff structure (if at all existent), which must be sufficient to cover all costs related to the good functioning of the treatment plant. Treatment wetlands are very competitive in terms of construction costs and are frequently very advantageous in terms of O&M costs, compared with other treatment systems. Thus, it is important to guarantee adequate routine O&M, since wetlands are systems which are very robust for a long time until they fail completely, needing large sums to recover the efficiency.
- *Need for simplicity*. In most applications in developing regions, conceptual simplicity is a must. Lack of skilled manpower for undertaking even basic operational duties is frequent, and this

reinforces the suitability of natural systems such as treatment wetlands. Unless aiming at specific applications, the level of mechanization should be kept to a minimum. Pump and valve operation is often the limit of knowledge in rural areas. Of course, in developing countries there may be well developed areas, and the operational level can be raised and justify a slight increase in the level of mechanization, if this leads to a reduction in the land requirements or an improvement in the effluent quality.

- *Risks associated with excessive overstatement of the concept of simplicity*. The fact that treatment wetlands are very simple systems to operate must not become an excuse to neglect the basic duties associated with the running of the treatment plant. It is observed that there is a tendency in many developing countries to abandon maintenance and operation rather than undertaking routine basic low-cost maintenance and operation. It is important to note that every system fails without proper O&M, and this is also the case with wetlands. Typical failures in the performance of wetlands due to inadequate O&M are:
    - Failure of the pre-treatment stage (e.g., septic tanks) due to lack of desludging, which may cause overflow of sludge to the wetlands. This sludge may lead to quick clogging of the wetlands and subsequent failure. Preventative measures of desludging the pre-treatment units at the correct frequency are much cheaper than the corrective action of unclogging a wetland, which is laborious and expensive.
    - Failure of the distribution system, especially in vertical-flow wetlands, where there is a need for a uniform distribution of the liquid over the whole surface of the bed. When pumps or siphons fail, or part of the distribution system becomes full of sludge, this leads to overloading and ultimately clogging some areas of the bed. The clogging spreads out and leads to failure of the system in the end. At an early stage it is possible to control the clogging process in vertical-flow wetlands.
    - Wetlands are extensive systems and, as such, most of them work well at the beginning. This may induce a relaxation that will conceal problems in the system performance associated with inadequacies in the design or in the operational practices, which will appear only later on. The critical point is that in some cases this may be too late for solving the problem, whereas a correction in early days could have been done with much less effort.
- *Differences in influent wastewater characteristics*. When planning and designing treatment systems in developing regions, including wetlands, the following aspects need to be taken into account (von Sperling, 2007; von Sperling & Chernicharo, 2005):
    - Population growth rates may be different from developed countries. It is common to see higher population growth rates in urban areas in developing countries, due to higher fertility rates and rural exodus, compared with developed nations. On the other hand, it is also common to see negligible or even negative growth rates in small towns in rural areas, owing to migration to larger cities. Treatment plants are designed for future populations (with planning horizons between around 20–30 years), and the population forecasts face the challenge of being representative of the future trends in the specific region to be covered.
    - Per capita sewage flows may be different from those considered typical in developed countries. In water-scarce areas the per capita water consumption in household activities tends to be small, and so is the wastewater production. A similar comment can be made for low-income areas, in which per capita water consumption tends to be lower than in affluent areas. However, it is observed that in urban settlements in which there is no household metering of water consumption, wastage of water can occur, thus leading to a

higher sewage production. Another aspect that needs to be taken into account is the value of the return coefficient (the ratio between sewage production and water consumption) in small towns and in rural areas: it might be different from the traditional value of 80%, because of the common practice of discharging greywater in the backyard for plant watering for household agriculture. Yet another factor that needs to be taken into account when computing the wastewater flow to be treated relates to the fact that in places where a separate sewerage system (sewage and stormwater in separate networks) has been implemented, there are households that practice illegal connections, discharging stormwater into the sewerage system, which may cause hydraulic overloads in the treatment system during storm events. Fortunately, the extensive nature of treatment wetlands makes them more robust to this type of instability.

- Per capita mass pollutant loads may be different from those considered typical in developed countries. For instance, typical per capita BOD loads used in the design of treatment plants in developed countries lie in the vicinity of 60 g/pe · pd, whereas in developing regions these values may be lower, from 40 to 60 g/ppe · pd. Also, wastewater composition may be different, as a result of feeding habits and household activities, and nitrogen and phosphorus concentrations may also be variable. In regions with low living standards, pathogen load is likely to be high, even though coliform concentrations, as expected, will not differ from those in developed areas.
- The variations in flow and sewage composition will have an impact in the design of the treatment wetlands. Instead of simply using the international literature, frequently based on the experience of developed countries, the designer should have the aim of using local or regional data and experience, which will reflect in a much better way the real characteristics of the wastewater to be treated.

- *Differences in treatment objectives and effluent requirements*. The legislation in developing countries may be different from that in developed nations regarding requirements for effluent quality for discharge into water bodies or for planned reuse. In general, more stringent requirements are found in developed countries, although this may not be true in several developing nations, which sometimes simply copy standards from high-income countries, without adaptations to their specific reality and needs (von Sperling & Fattal, 2001). If one considers a stepwise temporal evolution in the requirements for pollutant removal in developing countries, priority should be given to organic matter (BOD and COD) removal, for which treatment wetlands are very well suited. Another important objective, especially if water reuse is desired, is pathogen removal, with special consideration to helminth eggs. This is easy to achieve in TWs given their filtration capability. Nutrient (nitrogen and phosphorus) removal should be included if there is a real local need, and it should be remembered that wetlands designed with traditional criteria are not specifically efficient for nutrient removal. Monitoring practice related to verification of compliance with the legislation needs to be well planned in order to have realistic demands without incurring unnecessary costs in an already financially deprived area.

(b) Aspects related to favourable climatic conditions (warm-climate regions)

- *Differences in ambient temperature*. As mentioned before, most of the developing countries are located in warm-climate areas. Of course, there are low-income populations in temperate and cold areas, and for these the traditional design guidelines described in this book, subject to the special considerations listed above, may apply. However, in warm-climate regions, with a higher temperature of the wastewater, biochemical reactions and some physical processes take place at a faster rate, which can be considered advantageous in terms of the following two aspects: (i) for a given effluent quality, land requirements are likely to be smaller under warmer climatic conditions; (ii) for a given surface area allocated for wetlands, removal efficiencies are expected to be higher at more elevated temperatures. Therefore, under the prevalence of warm conditions, it is possible to adopt higher loading rates for the design of treatment wetlands and thus save in area (Hoffmann *et al*., 2011). Also, fewer stages or units in parallel may be applied in some specific processes, such as in the French VF wetlands, in which only the first stage may suffice in some applications, and further savings can be adopted by implementing only two units in parallel, instead of the usual three (Lombard-Latune & Molle, 2017). There may be a compromise between land savings and reductions in removal efficiencies, and the designer must find a good balance that suits well the requirements in each specific application.

- *Differences in rainfall regime*. Hydrological behaviour of treatment wetlands may be influenced by rainfall regime. In arid areas, evapotranspiration is likely to play an important role, leading to water losses and concentration of constituents in the effluent. Also, in arid areas, it is common to have a wide amplitude of temperature variations between day and night. On the other hand, in regions of intense rainfall events, stormwater flows may enter the sewerage system and sharply increase the influent flow to the wetlands. Fortunately, because wetlands are extensive systems, they tend to be more robust in coping with these peak hydraulic loads in comparison with compact systems. Finally, in regions that experience prolonged heavy rainfall, such as monsoon areas, this fact needs to be taken into account in the design of the system (Lombard-Latune *et al*., 2018).
- *Limitations in terms of the availability of regional design guidelines*. Most of the wetland literature emanates from developed countries under temperate or cold climate, in which there is a considerable accumulated experience as a result of thousands of units in operation. However, as highlighted in this section, developing areas and warm-climate regions have specificities that need to be taken into account. There should be a strong incentive to develop regional design guidelines for treatment wetlands based on actual experience in low-income and warm areas, so that future designs are really well suited to the local conditions.

4.2.3 Specific considerations for applications in developing regions

This section covers aspects of some specific applications of treatment wetlands in developing regions. The applications that are similar to the others covered in this book are not repeated here.

- *Rural areas in low-income regions*. A typical design for these areas should aim at simplicity and cut down operation and maintenance costs to a minimum. Whenever planning the solution, the simple concept of "what can fail, will fail" should be incorporated, and the systems need to be as robust as possible. Electromechanical equipment should be restricted to pumps. French VF wetlands, which may comprise only a first stage, could be a good solution due to their inherent simplicity, with no need for pre-treatment (grit removal and septic tanks), no need for separate sludge treatment, simple construction and possible compliance with effluent quality requirements.
- *Rural areas in low-income regions – effluent for reuse*. If an enhanced quality is needed, a French VF wetland could be applied, as it safely eliminates helminth eggs. Alternatively, the second stage of treatment can be performed by a HF wetland.
- *Housing areas*. Wetlands are a very promising possibility for housing areas in developing regions. Assuming that land availability may be scarce, the treatment plant must have a relatively small footprint. Compact solutions involving sophisticated technical processes, such as activated sludge

variants, have frequently failed due to inadequate operation and maintenance. Whilst still keeping some of the simplicity of traditional wetland systems, aerated wetlands offer a suitable possibility for a somewhat compact system, with only a small increment in terms of O&M requirements. They are robust to variations in influent flow and load, an important attribute for this type of application.

- *Touristic areas*. Many developing countries have touristic areas which are subjected to an alternation of periods with high peak loads followed by longer periods with only minor occupation. In contrast to compact technical treatment plants, various wetland configurations show robustness in handling such wide variations in influent flow and load. Short overload times may pose no problem when they are followed by underloaded periods. Under warm-climate conditions, this may be valid for weekend periods, as well as for summer overload periods (periods of up to three months). For treatments with only weekend occupation, the wetland can be designed as if the occupation was distributed along the week, multiplied by a safety factor. For touristic seasons of up to three months, a safety factor can also be included to the average typical daily load.
- *Decentralized systems up to 10,000 PE*. For this application, several wetland configurations can be applied. Important factors in the decision process are land availability and requirements for the final effluent quality (discharge in water bodies or reuse).
- *Sludge handling*. Sludge is one of the main reasons for failure or malfunctioning of treatment systems in developing countries. Desludging is frequently not done due to the lack of treatment facilities or due to the costs involved in transport or in constant handling. In this case, wetlands variants specifically conceived for receiving sludge (sludge reed-bed systems, planted sludge drying beds or sludge mineralization beds) are a very effective possibility for stabilizing and dewatering excess sludge generated in other treatment processes. The system is simple, with low O&M costs compared with other sludge handling alternatives, is able to store sludge for long periods of time and produces a safe sludge for agricultural applications.
- *Treatment of faecal sludge*. In many areas and cities in developing countries there is no piped sewerage system, and faecal matter is stored in pits, latrines and septic tanks. Septage or faecal sludge needs to be removed periodically from each individual system, and adequate treatment and disposal is very important. Wetlands are also a very convenient alternative, and they operate in a similar way to the sludge reed-bed systems, planted sludge drying beds or sludge mineralization beds mentioned above (Strande *et al*., 2014).

4.3 STORMWATER TREATMENT

Katharina Tondera[1], Dirk Esser[2] and Stéphane Troesch[3]

[1]IMT Atlantique, GEPEA, UBL, F-44307 Nantes, France
[2]Société d'Ingénierie Nature &Technique (SINT), Chef-Lieu, F-73370 La Chapelle du Mont du Chat, France
[3]Eco Bird, 3 route du Dôme, 69630 Chaponost, France

4.3.1 Introduction

Runoff caused by rainfall events is extremely variable in both pollutant loads and water volumes. Highway runoff transports comparably high loads of heavy metals and hydrocarbons, whereas runoff from urban areas can be characterised by a significant microbial contamination, e.g., caused by animal faeces, and sometimes high organic loads due to littering and cleaning of roads and market places (Table 4.1). Illicit connections in separate sewer systems contribute to nutrient and additional microbial loads in stormwater.

Stormwater has a high load of very fine particles which do not necessarily settle, flock or precipitate even over longer periods due to their size, electrostatic charge and low organic load (Table 4.2). A majority of the pollutant loads are attached to these fine particles (Boogaard *et al.*, 2014; Xanthopoulos, 1990). In addition, the organic pollutants in stormwater runoff, especially those originating from highway runoff, are not easily biodegradable.

The primary aims of stormwater treatment are are protecting surface waters from alterations to bed morphology, increased turbidity, deoxygenation, eutrophication, toxic heavy metal concentrations and, in some cases, microbial contamination. Additionally, treatment wetlands are also used as hydraulic buffers and reservoirs, in order to protect downstream areas from flooding. Due to the stochastic nature of

Table 4.1 Concentrations of stormwater runoff (Reproduced with permission from Chocat *et al.*, 2007).

Type of Urban Catchment	**Residential and Commercial**		**Highway and Road with Heavy Traffic**	
Parameter	**Mean**	**Min–Max**	**Mean**	**Min–Max**
TSS (mg/L)	190	1–4,582	261	110–5,700
BOD_5 (mg/L)	11	0.7–220	24	12.2–32
COD (mg/L)	85	20–365		128–171
NH_4-N (mg/L)	1.45	0.2–4.6		0.02–2.1
TN (mg/L)	3.2	0.4–20		
P tot (mg/L)	0.34	0.02–14.3		
Pb tot (µg/L)	210	10–3,100	960	241–34,000
Zn tot (µg/L)	300	10–3,680	410	170–355
THC (mg/L)	1.9	0.04–25.9	28	2.5–400
PAH (µg/L)	0.01	<0.01–3.2	–	0.03–6
Glyphosate (µg/L)	<1.5	<0.1–4.72	0.72	0–1,750
Diuron (µg/L)	<1	<0.05–13	0.05	0–2
Total coliforms (MPN/100 mL)	6430	40–500,000		10–1,000

Table 4.2 Particulate fraction of pollutants in stormwater runoff (Reproduced with permission from Chocat *et al*., 2007).

Pollutant	Particulate Fraction
COD	80–90%
BOD_5	75–95%
TKN	48–80%
Pb	80–98%
Zn	15–40%
Cu	35–60%
Cd	20–60%
THC	80–90%
PAH	75–97%

rainfall, the required storage and treatment capacity is extremely variable. Pollutant concentrations often show first flush patterns. Treatment wetlands offer the possibility to equip a great number of decentralized sites with an efficient passive treatment system which can can become an asset of the landscape with low operational requirements. Various designs for treatment wetlands are currently being used, depending on local space availability and intended co-benefits as well as treatment goals. Most commonly used are all variations of FWS wetlands, but also different variations of VF wetlands, which can either treat the outflow of stormwater sewers, or, as very small decentralised systems, directly treat street runoff. The latter systems are referred to as Sustainable Drainage Systems (SuDS – Woods-Ballard *et al*., 2015), Water Sensitive Urban Design (WSUD – Wong & Brown, 2009), Low Impact Development (LID – Dietz, 2007), or Sponge Cities (Li *et al*., 2017).

4.3.2 Design objectives

Treatment wetlands for stormwater runoff treatment need to have a double function:

- *A storage function*. The water to be treated must be stored in or on the treatment wetland, which requires an adequate storage volume and a throttled outflow. This is necessary to have retention times (in the case of FWS) or filtration velocities (in case of subsurface flow systems) compatible with a good treatment efficiency. However, the storage function can also be a target by itself, in order to assure flood protection of downstream areas. In some cases, legal limitations of the outflow can exceed the technical requirements for treatment and can thus become the key parameter for dimensioning. Bioretention filters do not have a throttled outflow, but a finer and less permeable filter layer.
- *A treatment function*. The primary targets are solids, especially fine suspended solids, and, to a lesser extent, dissolved substances. Treatment wetlands can also be designed in order to allow for the biodegradation and oxidation of dissolved organics during dry weather periods, if the dissolved pollutants are retained by sorption on plants and sediments (in FWS systems) or on the filter matrix (in subsurface flow systems) during the storm event. The treatment efficiency is, thus, at its best if the treatment wetland works in two phases: a first phase, during the storm event, in which the pollutants are retained by filtration or sorption, and a second phase of varying duration during the resting period for biodegradation of the organic pollutants.

4.3.3 Processes required and TW type to be used

Settleable and suspended solids can settle and/or be filtered. However, owing to the high amount of very fine particles, the effectiveness of sedimentation is limited and filtration and/or sorption is required.

Organic matter can be removed aerobically or anaerobically. Since quantitatively large water volumes occur in relatively short periods, pollutants from first flush loads need to be captured and treated subsequently during dry periods. Dissolved heavy metals, if necessary, need to be retained through adsorption on reactive filter media.

Suitable designs are FWS wetlands with emergent or submerged vegetation or VF wetlands (Table 4.3). Both systems can quite easily combine the storage function in the wetland itself (in the case of FWS wetlands) or on top of the filter surface, if the freeboard is high enough (for VF wetlands). If heavy metals need to be removed, either Floating Treatment Wetlands or VF wetlands with specific reactive media should be used (Borne *et al*., 2013; Fassmann *et al*., 2013; Hatt *et al*., 2008).

FWS wetlands have the advantage that they are less expensive to construct, usually have higher biodiversity than subsurface-flow wetlands and can be designed as a recreational amenity. They also provide a higher long-term storage capacity. However, mosquito breeding can be a problem.

VF wetlands tend to be more compact and efficient, as filtration and sorption on the filter matrix is more effective than sedimentation and biosorption on sediments and plants. In case of the larger systems with filter surfaces of several hundred square metres and storage capacities for complete settlements, integration into the landscape has so far not been a key consideration in their design, but it is possible to integrate them as an asset in open-access public areas. Smaller systems such as bioretention filters can be integrated as streetscapes into urban settlements.

In few cases a combination of VF wetlands and FWS wetlands has been successfully applied, combining the increased storage capacity, the biodiversity and the recreational value of a surface-flow wetland with the efficiency of the passage through a filter media (Jost *et al*., 2018)

Larger VF wetlands, especially for the treatment of highway runoff, are usually preceded by settling tanks equipped with scum baffles which remove coarse solids and more importantly protect the filter against

Table 4.3 Treatment efficiency of treatment wetlands for stormwater runoff (data from Blecken *et al*., 2018; Branchu, 2018; Giroud *et al*., 2007; Grotehusmann *et al*., 2016b; Stott *et al*., 2018; Tondera *et al*., 2018a, b, c).

Parameter	FWS Wetlands	VF Wetlands (Incl. Bioretention Filters)
TSS	From –97% to +89%	95–97%
Fine SS (<0.063 mm)	Not investigated	95%
P	10–90%	50–80%[1]
Indicator bacteria	0.1–2.1 $\log_{10}$	1–3 $\log_{10}$
TKN	2–60%	50–60%
Zn	30–95%	75–90%
Pb	80–90%	80–95%
Glyphosate	Dissipation/dilution of pesticides observed	58–80%
Sum of the 16 PAH of US EPA	Inconclusive	>86%

[1]Only with special active filter media.

accidental pollution, especially from hydrocarbons or oil. Bioretention filters are not equipped with primary settling tanks.

4.3.4 Specific considerations during design and for construction

The storage capacity of the wetland should be determined by hydraulic modelling, based on the maximum of tolerated overflows in a given time. This gives the storm event to be stored and treated, e.g., the monthly, annual or decennial event, the stochastic occurrence of events and their intensity, the runoff patterns generated by these events and the throttled outflow of the wetlands. The treatment capacity needs to be adapted to the pollution loads and runoff patterns specific to the catchment, considering first flush effects.

In some cases, an additional storage volume can be provided for water which does not have to undergo full treatment, and in all cases, at some point excess water has to be evacuated by overflows. The design should, however, always ensure that the most polluted part of the runoff (often, but not always the first flush) is properly treated.

Conditions for nitrogen transformation in FWS wetlands are more effective when the permanent water level is shallow enough (approx. 0.3 m depth) to allow sufficient oxygen exchange. Floating Treatment Wetlands can be used in zones with higher depths, e.g., to retrofit ponds or in concrete tanks, or when space is limited. It is, however, important that the design favours hydraulic conditions without shortcuts. Local plant species with extensive root growth into the water column should be used which remove fine particles and dissolved substances by sorption on biofilm forming on the roots.

Bigger, "end of the pipe" VF wetlands for stormwater runoff treatment can be designed like those for combined sewer overflow treatment (Chapter 4.4), as shown in Figure 4.1 below, although removal of

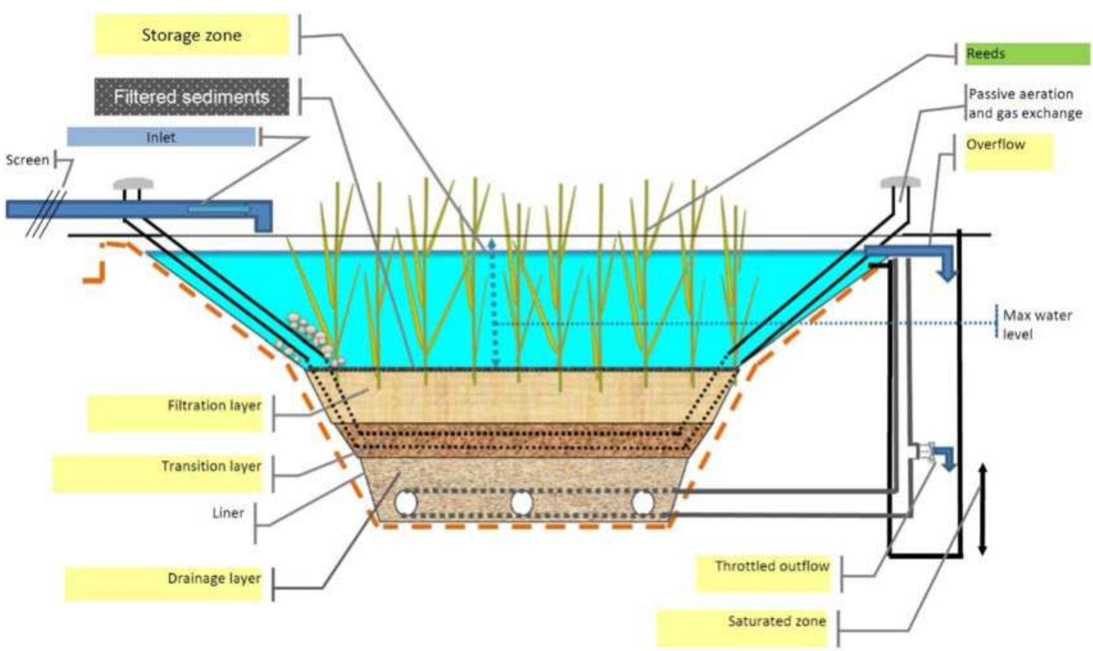

Figure 4.1 Cross section of VF wetland with storage volume on top of the filter surface and throttled outflow, as used for stormwater and CSO storage and treatment.

filtered sediments can play a minor role due to lower organic loading. Storage volumes on top of the filter level can be designed between 0.3 m and 1.0 m or even higher for less frequent immersions (once per month or less). Bioretention filters usually have a shallow freeboard of less than 0.4 m.

Recommended filtration velocities compatible with a good treatment efficiency can be up to 5×10^{-5} m/s, which means throttling the outflow at 0.05 $L/(s \cdot m^2)$ (Grotehusmann *et al.*, 2016b; Molle *et al.*, 2013). They should be 1×10^{-5} m/s if pathogen removal is required, but placing an UV-lamp for pathogen removal at the outflow of the VF wetland is often preferred instead of the slower filtration velocity.

Recommended filter material for VF wetlands treating stormwater runoff is fine to coarse sand (d_{10} between 0.2 and 0.5 mm). Finer sand is more efficient, especially for ammonia removal, but coarse sand is less prone to clogging. For bioretention filters, not throttling the outflow, a sandy loam is the recommended filter material (e.g., Woods-Ballard *et al.*, 2015). Over time, a secondary filter layer forms on top of the surface layer from the retained solids which provides additional sorption capacity, and which will increase the filtering efficiency. Phosphorous removal can be enhanced by reactive media, but it has to be considered that the reactive media will be saturated at some point and the efficiency of P-abatement will, therefore, decrease over time, limiting the lifespan of the reactive media. In Germany, it is considered that the addition of a few percent of iron hydroxide to the mass of the filter material can allow for a lifespan of 50 years (Grotehusmann *et al.*, 2016b).

As most of the treatment efficiency is based on filtration and sorption on fixed biofilms, the depth of the filter material is of lesser importance, and a depth of 30 cm of sand layer can be considered satisfactory in most cases (Molle *et al.*, 2013). A depth of 0.5 m to 0.75 m is recommended in Germany (Grotehusmann *et al.*, 2016b). Deeper filters can have a higher adsorption capacity for ammonia and, if reactive filter material is used, for phosphorous.

Generally, the required filter area is between 0.5 and 2% of the impervious catchment area for bigger VF wetlands with sandy filter material and a throttled outflow, 4–8% for FWS wetlands, and up to 6% for bioretention filters. Too frequent flooding of the filter surface and/or too long periods to drain down the filter after a rain event can result in a lack of oxygen for the aerobic degradation of the pollutant load during the dry period, resulting in a reduced treatment efficiency, and, more importantly, possible clogging of the filter. Hence, dimensioning of the filter can be based:

- On the annual load of fine solids: Grotehusmann *et al.* (2016b) recommend a maximum annual load of 7 kg/m^2 fine solids (<0.063 mm)/$(m^2 \cdot yr)$;
- On the time the filter needs to drain after the storm event (24–48 h; see Grotehusmann *et al.*, 2016b; Molle *et al.*, 2013);
- On the cumulative annual load which is used in older German guidelines, such as DWA-M 178 (2005), which recommends dimensioning VF wetland for stormwater runoff on the basis of a cumulative hydraulic load of 40–50 m ($=m^3/(m^2 \cdot yr)$) and a maximum of 70 m/yr. However, Grotehusmann *et al.* (2016b) only recommend having a minimum filter surface of 100 m^2 per ha of active catchment area if the annual rainfall exceeds 1000 mm.

In climates with frequent rainfall, it should be considered to divide the filter surface into two parts, which would be used alternately on a weekly basis for the more frequent, but less important rain events. In case of the less frequent but important rain events, the design should allow the entire filter surface and the entire storage volume to be used.

In climates with long dry periods, the treatment design needs to be functional even after extensive phases without rainfall. This can be partly overcome by a saturated layer in the lower parts of the filter, which provides a hydraulic reserve for the plants. In that case, intermediate passive ventilation is required above the saturated layer to allow for gas exchange when the surface of the filter becomes quickly flooded. However, the biofilm in the unsaturated layer degrades during long dry phases, thus reducing the treatment efficiency for dissolved pollution.

As in the case of CSO systems, plant species used for VF wetlands must be able to cope with low nutrient supply and long-lasting phases without loading, followed by hydraulic shock loading.

4.4 TREATMENT OF COMBINED SEWER OVERFLOWS

Katharina Tondera[1], Anacleto Rizzo[2] and Tamás Gábor Pálfy[3,4]

[1]IMT Atlantique, GEPEA, UBL, F-44307 Nantes, France
[2]Iridra Srl, via La Marmora 51, 50121, Florence, Italy
[3]University of Sopron, Institute of Geomatics and Civil Engineering, H-9400 Sopron, Hungary
[4]Department of Ecotechnologies, Irstea, Villeurbanne, France

4.4.1 Introduction

Combined sewer overflows (CSOs) from urban areas are an underestimated source of water pollution. They occur during wet weather events, when surface runoff entering a combined sewer system adds to the dry weather flow and exceeds the capacity of downward sewer sections or the treatment plant. The point of discharge is often constructed as a simple overflow barrier. A settling tank can be installed after these points to provide additional storage and primary sedimentation before the diluted wastewater enters the receiving water body. Due to high flow rates, CSOs discharge enormous pollutant and pathogen loads in comparison to the average flux projected for a year. The discharged volumes can have severe impacts on the surface water ecology and health-related ramifications, especially when people use the receiving surface waters for recreational purposes.

A primary target of CSO treatment is to retain solids and oxygen-depleting pollutants such as organic matter and ammonium. Furthermore, the removal of pathogens is required, especially in surface waters in densely populated areas. Compared to wastewater treatment with continuous flow, the necessary storage capacity is defined by the statistical reoccurrence of different flow volumes. The maximum flow volumes to be treated depend on the discharge requirements and the sensitivity of the receiving surface water body.

Over the last 25 years, treatment wetlands have proven to provide the most integrated treatment of CSOs currently. Most CSO wetlands have been implemented in Germany (Dittmer *et al*., 2016; Grotehusmann *et al*., 2016a; Tondera, 2017), but first sytems also have been constructed in France, Italy (Meyer *et al*., 2013) and the United States of America (Tao *et al*., 2014).

4.4.2 Design objectives

Treatment wetlands for CSOs are primarily targeting the removal of suspended solids and oxygen-depleting parameters (organics expressed as BOD or COD and ammonium). The main factors affecting the treatment performance of CSO wetlands are the number of load events per year and their stochastic occurrence, as they determine the regeneration time (often referred to as dry period). Possible issues related with a design not properly linked to stochastic nature of CSOs are (Pálfy *et al*., 2017a):

- Insufficient resting time can lead to clogging;
- Infrequent loads might harm the biofilm as the dry period results in literally dry pore spaces. this impacts on organic removal performance for the subsequent load and might cause washout of dead biofilm as well;
- Extensive phases without feeding or rainfall can lead to animal burrows, invasion of plant competitors, especially nettles, and plant decay

The treatment of CSOs requires additional storage capacity which can be provided either as external concrete tanks or on top of the filter layer. The latter has the advantage that no cleaning of settled particle is necessary as they mineralise on the filter layer and, over time, form a secondary filter layer that increases the overall adsorption capacity.

4.4.3 Processes required and TW type to be used

Settleable and suspended solids require sedimentation and/or filtration. For the oxidation of organic matter (organic N and BOD_5) and ammonium-N into nitrate-N, aerobic conditions are crucial. Since quantitatively large water volumes occur in relatively short periods, the oxidation mostly is a delayed process of adsorbed and absorbed substances. Thus, treatment wetland technologies with high sorption capacities and subsequent availability of oxygen are required.

Owing to the high organic load from the domestic wastewater, VF wetlands provide the most reliable design. Drainage pipes or separate aeration pipes provide passive aeration during dry periods for nitrification and further biological degradation. Therefore, access to the interior of such pipes for cleaning should be possible. Roots growing into the holes of the drainage pipes can be avoided by foil strips placed covering the drainage pipes (DWA-A 178, 2019). Frequent loads might limit regeneration time, especially at low filter bed temperatures where nitrification might be incomplete so that adsorption sites might saturate progressively (Pálfy *et al*., 2017b).

If total N removal is required, then the design will need to include the denitrification process to remove the nitrate generated from the upstream nitrification process. TW types suitable for denitrification include: FWS wetlands, in which the emergent vegetation provides a direct internal source of organic carbon for the process, and HF wetlands, which tend to promote anoxic conditions and can also return some organic carbon from the vegetation to the subsurface water.

CSO wetland systems are well suited to be designed for multiple purposes, providing ecosystem services additional to water quality improvement. Indeed, flood protection can be integrated, exploiting the water storage capacity of VF filters, as well as designing a second FWS stage also as detention basin (Rizzo *et al*., 2018a). Moreover, a second stage with FWS also can provide polishing due to a longer HRT (Masi *et al*., 2017a), increase the biodiversity value, and facilitate the inclusion of CSO wetlands in public parks, providing social benefits (Liquete *et al*., 2016).

4.4.4 Specific considerations during design and construction

The height of the filter bed and the filter material are critical for the treatment performance:

- Filter media should be sand or fine gravel with a steep sieve curve without organic supplement to avoid clogging;
- Special material can increase adsorption capacity (e.g., zeolite);
- Additions to the filter material such as limestone (as top layer or mixed with the filter material) can provide a buffer against acidification.

Infiltrating groundwater or other quasi-continuous flows, if led into the wetland, lead to permanent inundation and might cause biological clogging and for that, the filter area shall be sufficiently large to avoid clogging. However, oversizing of VF filters might lead to different problems, one of them being extensive dry periods. As result of a long-term simulation, at least 10 feedings per year should be targeted (DWA-A 178, 2019), although this recommendation stems from mild climates with regular rainfall events wetting the filter surface during dry periods. In dry climates, more frequent feedings should be targeted or watering of the filter surfaces during long dry periods should be maintained.

Plant species used for CSO wetlands must be able to cope with long-lasting phases without feeding followed by shock loading both hydraulically and in terms of pollutant loads. *Phragmites australis* (local genotype) has proven to be resilient under these circumstances. Owing to the overall low nutrient load, harvesting of the plants is not necessary and, in contrary, has shown negative effect on the growth of the subsequent spring season.

Similarly to wetlands for stormwater treatment described in Chapter 4.3, one key aspect in design of CSO wetlands is how to guarantee sufficiently slow infiltration rates for a proper treatment of CSO, which is solved by throttling the outflow. We suggest for reference values of throttled effluent flow rates and other design criteria those reviewed by Meyer *et al.* (2013) as well as German or French guidelines. The design-support tool *Orage* (Pálfy *et al.*, 2017b, 2018) is also available for a more detailed design of CSO wetlands.

4.5 AGRICULTURAL DRAINAGE WATER

Stevo Lavrnić and Attilio Toscano

Department of Agricultural and Food Sciences, Alma Mater Studiorum – University of Bologna, Viale Giuseppe Fanin 50, Bologna 40127, Italy

4.5.1 Design objectives

Agricultural practices have been reported to cause pollution of surface water bodies in different parts of the world (Blankenberg *et al*., 2008; Díaz *et al*., 2012; Dunne *et al*., 2005; Lenhart *et al*., 2016; Mendes *et al*., 2018). For example, nitrate has been recognised by the European Commission as one of the major agricultural pollutants and the Nitrate directive issued in 1991 aims to reduce such a pollution in the EU (EEC, 1991).

Nitrate losses from agriculture can be reduced through in-field (e.g., lowering usage of fertilisers or improving fertiliser uptake by crops) or edge-of field methods (e.g., treatment of agricultural drainage water) (Groh *et al*., 2015). Natural wetlands, small natural streams and vegetated stream banks have a certain capacity to purify water, but the loss of these systems has caused a drop in the quality of surface water bodies receiving agricultural drainage (Borin & Tocchetto, 2007). Therefore, there is a need for a more systematic approach to this problem. For example, grass strips were reported to be capable of successful treatment of agricultural drainage water, but their capacity for it is limited and is considerably lowered when the soil is saturated (Tournebize *et al.,* 2017). On the other hand, TWs are known to be able to treat wastewater through a technology that is sustainable and low cost (Li *et al*., 2018), can also successfully treat agricultural drainage water (Groh *et al*., 2015; Kasak *et al*., 2018; Vymazal & Březinová, 2015), and are more cost-effective for reducing non-point source pollution than other methods (Lavrnić *et al*., 2018). Their additional advantage lies in the fact that TWs can also provide several ecosystem services if managed well (Tournebize *et al*., 2017), an approach that lead to a development of the concept of integrated TWs, systems that combine water quality control and biodiversity enhancement (Scholz *et al*., 2007).

TWs for treatment of agricultural drainage water can be either on-stream or off-stream depending on whether they are located at the flow of drainage water or outside of it (Kasak *et al*., 2018). The first option is more suitable for nitrate removal, since concentration of nitrate is usually comparable during different periods. On the other hand, off-stream TWs are applied in cases when pesticide removal is a priority, since concentration of these substances is the highest in the first flow after their application. Therefore, the flow can be diverted towards TW only after pesticides application in order to increase HRT of the system and enable higher pesticide removal (Tournebize *et al*., 2017). Most of the TWs treating diffuse pollution are off-stream since in-stream systems cannot treat all drainage water or the area needed for them is too big (Kasak *et al*., 2018).

4.5.2 Processes required and type to be used

The type of TWs that is most often used for the treatment of agricultural drainage water is the FWS wetland (Dal Ferro *et al*., 2018; Vymazal & Dvořáková Březinová, 2018). Its advantage compared to other TW types is that it can cope with pulse flows and changing water levels (Kadlec & Wallace, 2009), both conditions typical in drainage water treatment. Except for wastewater treatment, FWS wetlands can also be used for flood attenuation, water retention and biodiversity enhancement (Dal Ferro *et al*., 2018; Díaz *et al*., 2012).

Although the removal performances vary, the majority of the studies that reported efficiency of TW systems treating agricultural drainage water showed improvement of water quality (Díaz *et al*., 2012). For example, these systems exhibit average removal of 1175 kg TN/ha/yr and 157 kg TP/ha/yr, the values that are comparable with those for various kinds of TWs treating different types of inflow (Vymazal & Dvořáková Březinová, 2018). However, most of the authors that deal with this topic focused on systems that were in operation for a short period of time, and not many report long-term effectiveness (Groh *et al*., 2015). Therefore, results obtained during the first few years should be taken with caution. It was suggested that TWs treating agricultural drainage water will achieve their maximum TN removal after a certain transition period (Borin & Tocchetto, 2007; Dal Ferro *et al*., 2018), which could be especially long in areas with cold climate since the vegetation period there is short (Kasak *et al*., 2018). On the other hand, TP removal might diminish over the years due to the saturation of the sorption sites and biomass storage (Dal Ferro *et al*., 2018). However, TWs could also be a long-term solution.

Hydraulic efficiency is an important characteristic of these systems and it affects pollutant removal processes. Structures such as dams or stones can increase hydraulic efficiency but can also improve aesthetics of the system and its attractiveness for a variety of wildlife (Braskerud, 2002; Kasak *et al*., 2018). Moreover, meanders or sinuous water paths can increase retention time, a factor that affects removal (Lenhart *et al*., 2016; Mendes *et al*., 2018).

Agricultural drainage water usually has a low C/N ratio and high concentration of nitrates (Li *et al*., 2018). Since denitrification is the dominant nitrate removal path in FWS wetlands (Groh *et al*., 2015; Tournebize *et al*., 2017), TN removal can be limited due to shortage of carbon. This problem could be overcome by addition of an extra carbon source that can be in liquid or solid form. Liquid carbon source has to be added constantly and could cause secondary pollution, difficulties that do not exist if a solid carbon source is used (Li *et al*., 2018). On the other hand, it has been reported that the retention of nitrogen in a FWS wetlands can be increased through addition of straw (Blankenberg *et al*., 2008) or non-removal of harvested biomass (Tournebize *et al*., 2017).

Apart from the cases when organic matter content is not enough to enable denitrification, TN removal through this process can be low when the system receives a medium–low yearly load, or when flooding and anaerobic conditions inside the system occur only for short periods of time (Borin & Tocchetto, 2007). Moreover, since denitrification decreases at low temperatures there is a certain variability in removal efficiency between different seasons (Tournebize *et al*., 2017), and it can be particularly low during the winter (Borin & Tocchetto, 2007). TN removal can also be hindered by stagnant water conditions, since oxygen can be depleted and therefore prevent complete nitrification (Díaz *et al*., 2012).

An especially important process in FWS wetlands is sedimentation of soil particles since phosphorus and other pollutants are generally attached to them (Braskerud, 2002). For that reason, the usual design of these systems is a deeper inflow section to facilitate sedimentation (1–2 m deep), followed by a vegetated bed (0.1–0.5 m deep) (Vymazal & Dvořáková Březinová, 2018). Factors that affect retention of soil particles are sedimentation velocity, flow rate and surface area. Since the soil particle concentration is high in the beginning of the rainfall event and the flow rate is low, sedimentation usually does not represent a problem in this phase. Resuspension of soil particles is undesirable, which can be mitigated by vegetation presence (Braskerud, 2002; Kasak *et al*., 2018). Moreover, vegetation in FWS wetlands can also improve removal efficiencies due to the provision of a carbon source for denitrification or passive transfer of oxygen from the atmosphere into the soil (Kasak *et al*., 2018).

Wetlands can remove phosphorus through biological (plant and microbial uptake), physical (sedimentation) and chemical pathways (sorption and precipitation) (Dunne *et al*., 2005), out of which the first two are the primary ones (Lenhart *et al*., 2016). The physicochemical characteristics of wetland soils and sediments are one of the main factors in these processes, since they affect inorganic P sorption dynamics (Dunne *et al*., 2005). Moreover, anaerobic conditions might cause release of phosphorus from the sediments and therefore the system should be in an oxic state (Kasak *et al*., 2018). Other factors that can affect long-term stability of phosphorus bound in the sediments are supply of phosphorus sorbents, sediment redox conditions and Fe_{tot}: P molar ratios (Mendes *et al*., 2018). FWS wetlands can experience a decrease in TP removal after a certain time due to the fact that sorption sites are saturated, and that initial vegetation growth has stabilised. Therefore, it is important to perform appropriate vegetation management and removal of sediments in order to enable the same or similar level of TP removal (Díaz *et al*., 2012).

Although pathogen concentration in agricultural drainage water is low unless there are animal farms in the catchment, it is still important to consider this parameter since TWs can act as their source, rather than a sink when inflow concentration of pathogens is relatively low ($\sim$100 CFU 100 mL^{-1} of faecal coliforms) (Beutel *et al*., 2013). For example, *Escherichia coli* removal might be lower in FWS wetlands that do not have a constant water flow and are often characterised by longer periods when water is in stagnant conditions. Stagnant water can have different environmental conditions (chemical and thermal properties) in the water column that can favour development of certain bacteria. Those conditions are often prevented by the constant water mixing that exists in systems with a constant flow (Díaz *et al*., 2012). Moreover, coliform bacteria could also be introduced into the systems by warm-blooded animals such as mammals or birds (Beutel *et al*., 2013; Díaz *et al*., 2012).

Similar phenomenon can also inhibit removal of pesticides, since they can be found accumulated in biofilm (Tournebize *et al*., 2017) and sedimentation is an important mechanism for pesticide removal (Díaz *et al*., 2012). Removal of pesticides therefore depends on the sediment characteristics (i.e., organic content, particle size, hydraulic conductivity), but also on the properties of pesticide itself (i.e., half-life, solubility, octanol–water partition coefficient, and distribution or sorption coefficient) (Mahabali & Spanoghe, 2014). Vegetation in the system can contribute to pesticide removal either by their uptake (Mahabali & Spanoghe, 2014) or by enabling development of biofilm in which pesticide biodegradation can occur (Tournebize *et al*., 2017).

4.5.3 Specific considerations during design and for construction

For wetlands treating agricultural drainage, specific considerations during design and for construction are:

- Predicted runoff should be taken into consideration when planning a TW in order to adjust the depth. This is particularly important when the area is limited (Blankenberg *et al*., 2008).
- Soil texture should be estimated before construction of a FWS wetland since infiltration can present an important component of water balance of non-waterproofed systems, and can cause high water losses to infiltration (Lavrnić *et al*., 2018).
- Systems should be designed to facilitate harvesting, a process that can increase permanent phosphorus removal and prevent its release (Lenhart *et al*., 2016).
- TW to catchment ratio is an important parameter to be considered during the design phase and to enable a HRT that is long enough to allow sufficient drainage water treatment; it should be at least 1%, or even higher in regions with cold climate (Tournebize *et al*., 2017).
- Sediment resuspension could be kept at minimal level if vegetation cover is approximately 50%. Therefore, plant requirements for optimal growth should be taken into account when designing the system (Braskerud, 2002).
- Existence of dead zones and short circuits should be avoided by a proper positioning of inlet and outlet points and by creation of dykes (Tournebize *et al*., 2017).
- Vegetation development should be encouraged before the system starts operation, since water level management can be controlled in that period and it can affect proper vegetation establishment (Lenhart *et al*., 2016).

4.6 SLUDGE TREATMENT WETLANDS

Steen Nielsen[1] and Alexandros Stefanakis[2]

[1]Orbicon, Linnés Allé 2, DK - 2630 Taastrup, Denmark
[2]Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

4.6.1 Design objectives

In conventional wastewater treatment plants (WWTPs), the treatment process results in large volumes of a sludge by-product. This excess sludge is produced at the various treatment stages of the WWTP, such as primary and secondary clarifiers and the biological treatment stage. Sludge contains high moisture content, nutrients and organic solids, and even heavy metals, synthetic organic compounds, pathogenic microorganisms and inorganic substances (Stefanakis *et al.*, 2014). Hence, disposal of sludge to the environment without proper quality or treatment is prohibited by regulations, while some compounds are considered valuable (e.g., organic carbon and nutrients) for reuse in agriculture.

Sludge management and handling is a main concern for WWTP operators owing to the large volume produced; for example, mean sludge production in Europe exceeds 0.09 kg dry mass/PE (Stefanakis *et al.*, 2014). Although sludge represents less than 1% of the wastewater volume, its management costs can reach up to 40–50% of total WWTP operation cost. Therefore, the main goal in sludge treatment is the reduction of the water content and an optimal solids content, along with substance degradation (Stefanakis *et al.*, 2014). Several methods are available for sludge dewatering and drying, such as mechanical systems (belt thickening, belt press, centrifuges, etc.), aerobic/anaerobic digestion, incineration, composting, among others. Mechanical systems can be expensive and problematic to run, owing to high energy consumption, use of chemicals and demanding maintenance. On the other hand, traditional low-cost methods such as drying beds, although cheaper, are mostly applicable under warm climates but they have high area demand, odour/nuisance issues and cannot provide a final dried sludge with high solids content. Therefore, Sludge Treatment Reed Bed (STRB) Systems or Sludge TWs appear as a dewatering technology with specific advantages (Nielsen, 2003; Nielsen & Bruun, 2015; Nielsen & Dam, 2016; Nielsen & Willoughby, 2005; Stefanakis & Tsihrintzis, 2012c; Stefanakis *et al.*, 2014).

The key objective of a STRB system is to provide a sustainable solution to excess sludge handling in WWTP. STRBs are designed to be able to receive and effectively dewater the daily excess sludge volume generated at a WWTP. One distinctive characteristic of STRBs is that there is no need for the regular (e.g., weekly or monthly) transport and disposal of dry sludge material. The STRBs are designed to continuously receive the daily excess sludge for 6–15 years (depending on the dimensioning and the loading rate), without any planned long-term intervals in their operation. This is achieved by having several beds in serial operation where a sludge feeding regime is applied that consists in feeding and resting periods, the extent of which mostly depends on the sludge quality and climatic conditions of the area (Nielsen & Cooper, 2011; Nielsen *et al.*, 2018; Stefanakis *et al.*, 2014). Ultimately, a properly designed and operated STRB facility can deliver a final dry sludge material, usually called biosolids, that has a high solids content and is well stabilized so that it can be reused, e.g., as fertilizer in agriculture (Nielsen & Bruun, 2015; Stefanakis *et al.*, 2011).

4.6.2 Processes required and TW type to be used

The general design of a STRB is more or less similar to that of a VF wetland: there is a substrate zone consisting of gravel layers with different grain sizes, an inlet distribution pipe network across the gravel surface and draining pipes at the bottom of the (lined) bed to collect the drained water. The overall system is divided into several beds (depending on the feeding/resting schedule). The difference here is that the applied sludge is not wastewater but a watery mixture with usually 0.5–4% dry solid, which also has different hydraulic properties. Additionally, the feeding strategy and operation regime differs from that of the VF wetlands for wastewater treatment, while dimensioning of the system follows a completely different approach (see Chapter 5).

In STRBs, two general mechanisms can be distinguished: (i) dewatering and (ii) mineralization. Dewatering in Sludge TWs occurs only through natural processes, i.e., draining and evapotranspiration. Sludge dewatering results in volume reduction through water removal, which is the first main goal of sludge treatment, and the solids content can increase up to 20–30% (Nielsen, 2003; Nielsen & Willoughby, 2005; Stefanakis & Tsihrintzis, 2011; Stefanakis *et al*., 2014).

Drainage appears to be the main dewatering process in STRBs. As in most other wetland systems, evapotranspiration (ET) also takes part in dewatering. ET consists of water evaporation from the sludge cake surface and plant transpiration. ET is affected by various parameters, such as the topography and geology of the area, the species and the plant growth, the local climatic conditions (i.e., solar radiation, temperature, humidity, wind speed etc.) and the total precipitation (Stefanakis & Tsihrintzis, 2011). It has been found that temperature values above 15–16°C could increase the ET rate in STRBs by 30% (Stefanakis & Tsihrintzis, 2011), while higher temperatures during summer months enhance sludge dewatering in STRBs by 40%. In STRBs, the sludge dewatering rate is enhanced by the presence of plants, which absorb water for their growth needs. Wetland plants absorb water through their root system and transfer it to the stems and leaves, where it is released to the atmosphere. Published literature indicates improved dewatering efficiency in planted rather than unplanted STRB beds due to higher recorded ET rates (Peruzzi *et al*., 2013; Stefanakis & Tsihrintzis, 2011).

Draining is the vertical gravitational movement of water through the porous media layers of the STRB bed. It usually occurs during the first few hours after sludge application onto the bed and after 15–24 hours the water flow returns to its initial lower values (Nielsen, 2011). Practically, after 2–7 days the water volume that leaves the bed is insignificant (Stefanakis *et al*., 2014). Draining removes a major portion of the sludge water volume in colder climates, which results in high solids content in the residual sludge layer (more than 30%; Nielsen, 2011). In moderate climates, such as the Mediterranean basin, draining can account for more than 40% of the water losses (Stefanakis & Tsihrintzis, 2011). The presence of plants also affects draining, since the movement of the plant stems creates cracks on the sludge layer, enhancing this way the water flow. However, it is reported that as the plants develop a deep and dense root system and increase their density with time, the draining rate is reduced (Stefanakis & Tsihrintzis, 2011).

The plants and their extensive root system affect the internal cohesion forces of the sludge layer, cleaving its colloidal stability and releasing part of the bound water, while they absorb water and nutrients from the sludge. This results in a dewatered and improved sludge quality. In STRBs, the top layer of the sludge cake having fresh sludge is usually black, due to iron sulfide, and of aqueous composition, while the lower parts of the accumulated sludge cake have a brown colour and soil texture, which indicates the presence of aerobic conditions and mineralized material (Stefanakis & Tsihrintzis, 2012c). A black colour of the deepest parts of the sludge layer implies that the mineralization is limited (anaerobic conditions). Generally, owing to its longer treatment and stay within the bed, the bottom sludge is more mature and stabilized than the top layer (Stefanakis & Tsihrintzis, 2011). Along the plant roots, the alternation of aerobic/anaerobic conditions enables various biochemical processes such as oxidation of organic matter and nitrogen, ammonification, nitrification, and denitrification. In general, the transformation and removal processes of organic matter and other constituents, e.g., nitrogen and phosphorus, are similar to those occurring in VF wetlands for wastewater treatment.

4.6.3 Specific considerations during design and for construction

There are some key parameters that should be considered in the design and construction phase of a STRB, to prevent problems during the operation of the system. These briefly are:

- *Sludge quality*. It is important to have a good understanding of sludge source, characteristics and composition (e.g., aerobic/anaerobic, viscosity, etc.) to select the appropriate loading rate;
- Climatic conditions, e.g., rainfall, solar radiation etc., are required prior to the design of the system;
- *Sludge loading rate*. Selected based on sludge quality and climate (avoid overload);
- *Operation cycle*. Selection of feeding/resting periods with appropriate duration to prevent stagnant water on the surface and insufficient dewatering;
- *Freeboard*. There should be enough free depth above the gravel layer to allow for residual sludge accumulation during the anticipated operational life time;
- *Pumps/piping*. Proper sizing and dimensioning for sludge material, i.e., mixture of water with solids, to prevent clogging;
- *Distribution pipes*. Proper dimensioning for uniform distribution of sludge across the surface
- Appropriate number of basins to allow for adequate feeding/resting periods duration;
- *Plants*: Selection of native plant species, adapted to the climate that can survive under the specific loading conditions;
- Commissioning of appropriate duration and with gradually increasing loadings to allow for plant growth and higher density values;
- Regular monitoring of accumulated sludge depth, sampling and analysis of different points across the sludge layer;
- Detailed and continuous sludge loading records;
- Consideration of the final resting phase duration for each basin before emptying the residual sludge layer.

4.7 BIOMASS PRODUCTION

Darja Istenič[1], Tjaša Griessler Bulc[1], Giuseppe Luigi Cirelli[2], Alessia Marzo[2] and Mirco Milani[2]

[1]Faculty of Health Sciences, University of Ljubljana, Zdravstvena pot 5, 1000 Ljubljana, Slovenia
[2]Department of Agriculture, Food and Environment (Di3A), University of Catania, Via Santa Sofia 100, Catania 95123, Italy

4.7.1 Perspectives for energy production from TW biomass

Traditional wastewater treatment plants are significant consumers of energy. Nevertheless, they can produce biogas in the sludge digestion process which is mainly used for heating the facilities at the treatment plant or is converted to electricity; however the net energy balance is still negative in the majority of cases (McCarty *et al*., 2011). Compared to traditional wastewater treatment plants, TW, owing to their design and operation, have lower energy demand per se.

The main objective of TW is to treat wastewater and thus protect natural ecosystems from pollution; however, TW have numerous additional functions, among which biomass production is getting increased attention. Biomass can be used for energy production, which is a growing area of research as a response to the global energy crisis and the effects on climate change. In this aspect, TW offer additional value compared to conventional cultivation of energy crops due to reuse of wastewater for production of biomass, i.e., the need for application of mineral fertilizers and irrigation to produce energy crops is significantly reduced or even eliminated.

TW are cost-efficient and often economically outcompete conventional systems which can become even more obvious when using the produced biomass as an energy source. Since TW are mostly used for decentralized wastewater treatment, centralized energy production of the produced biomass is a challenge due to transport and sustainability. Decentralized stations or individual systems for heat energy production are often not economically feasible in developed countries; the return on investment in the equipment for production and storage of wood chip and pellets is longer than a lifespan of TW. However, the situation is the opposite in developing countries where significant parts of the population rely on wood for cooking, which can be substituted with biomass from TW (Avellán & Gremillion, 2019).

There is a fast-growing number of TW for wastewater treatment, both in developed and developing countries, resulting in thousands of operating TW in the world. However, not many TW are used for energy production, even though there is great potential: Liu *et al.* (2012) found that TW even have greater greenhouse gas reduction than conventional systems for production of biofuel in a complete life-cycle. Despite this, currently in the majority of operating TW worldwide, the produced biomass is composted or combusted as waste.

4.7.2 Sources and production of bioenergy within or post TW

Biomass for energy production can be grown within the TW or by fertigation of energy crops with the TW's effluent. Pellets or woodchip can be produced already from the plants that are usually grown within the TW, e.g., *Phragmites* sp., *Typha* spp., *Phalaris* sp., *Cyperus* sp. etc. or from willow wood in case of willow systems. The pellets and woodchip can be directly used for heating in appropriate furnaces or wood stoves.

Willow systems are a type of TW that is planted with willows (see Chapter 5.10 Willow systems). Willows are energy crops commonly used in short rotation coppices where they can produce around 10 t DM ha^{-1} per year with the application of artificial fertilizers, while in willow systems, owing to high nutrient and water availability, biomass production can triple (Istenič *et al*., 2018). According to Gregersen and Brix (2001) the amount of nutrients that enter the system with wastewater is approximately the same as the amount of nutrients in willow biomass, i.e., the composition of the wastewater corresponds to the willows' nutrient requirements (Börjesson & Berndes, 2006).

According to Liu *et al.* (2012) wetlands can produce 1.1 to 184 $MJ/m^2/yr$. Energy production of TW is directly linked to biomass production (Table 4.4), which depends on nutrient availability or mass loading rate. Besides this, climate, latitude and elevation have to be considered. Because the primary function of TW is wastewater treatment, most TW remain at a low biomass productivity level. The latter can be scientifically increased by selecting productive plant species, optimizing the flow pattern and taking an advantage of using waste nutrients and water (Liu *et al*., 2012); moreover, harvesting and regrowth after it also affect the biomass yield. Designing a wetland to increase biomass production will also have a significant impact on evapotranspiration and thus on the amount of discharge from the system. In arid areas water availability might be a limiting factor for biomass production.

Phragmites australis is the most commonly used plant in TWs worldwide and its energy production is similar to other wetland plants (Table 4.4). Higher energy production per m^2 can be reached by *Cyperus papyrus* or by willow systems and the highest by *Arundo donax*, which is currently not often used in TW.

The energy produced from biomass grown in TW has to be compared against the energy input needed for TW operation. According to Liu *et al.* (2012), the net energy balance for vertical flow TW with pulse loading is positive, meaning that there is more energy produced than needed for operation. Moreover, the net energy balance is also higher compared to some other systems for production of energy crops (e.g., soybean, corn, microalgae).

TW can also contribute to production of bioenergy through reuse of treated wastewater for energy crops irrigation and fertilization. To achieve high productivity particularly in summer crops irrigation is generally necessary; in this context, treated wastewater presents an important water source. Post-wetland production of energy crops combines different advantages. Water fertilizing properties decrease the demand for

Table 4.4 Biomass production and energy yield for different plant species growing in TWs.

Type of Plant	Biomass Production in TW	Combustion Energy Yield	Energy Production	Methane Production
	kg DM m^{-2}/yr	MJ/kg · DM	MJ m^{-2}/yr	L/kg · DM
Phragmites spp.	1.9 ± 1.3[1] 3.3 ± 1.1[8]	18[1]	34 ± 24* 44 ± 31[4]	108–236[1]
Typha spp.	1.6 ± 0.9[1]	18[1]	29 ± 16* 37 ± 36[4]	NA
Arundo donax	6.1 ± 4.5[1] 2.1–4.9[7]	18[1] 17–24[7]	109 ± 81* 132 ± 34[4]	297[1]
Cyperus papyrus	3.6 ± 2.5[1]	18[1]	64 ± 44* 48 ± 6[4]	NA
Miscanthus sp.	0.6–3.8[7]	16–19[7]	22[4]	152[5]
Phalaris sp.	1.3 ± 0.5[8]	NA	23 ± 11[4]	185[9]
Salix spp.	3.3 ± 0.9[2]	19.8[3]	64 ± 18*	172[6]

*Calculation from production and combustion data: [1]Avellán and Gremillion (2019); [2]Istenič *et al.* (2018); [3]Keoleian and Volk (2005); [4]Liu *et al.* (2012); [5]Yang and Li (2014); [6]Triolo *et al.* (2012); [7]Ge *et al.* (2016); [8]Vymazal and Kröpfelová, (2005); [9]Lakaniemi *et al.* (2011).

Table 4.5 Biomass production and energy yield for different plant species irrigated with TW effluent (Barbagallo *et al*., 2014; Molari *et al*., 2014).

Type of plant	Biomass Yield	Combustion Energy yield	Energy Production
	kg DM m^{-2}/yr	MJ/kg · DM	MJ m^{-2}/yr
Arundo donax	2.6–7.9	21	55–166
Miscanthus giganteus	0.5–4.5	18	9–81

synthetic fertilizers and contribute to the reduction of nutrients loading in rivers; this practice increases the available agricultural water resources and it may lower treatment costs.

When using TW effluent for energy crops irrigation, the TW type can be simplified, i.e., to enable degradation of organic matter producing an outflow rich in nutrients which can be used for fertigation of energy crops such as herbaceous plant species (*Arundo spp., Myschantus spp.,* etc.) and short rotation coppices (willow, poplar, acacia).

Several research programmes were carried out in Italy (Barbagallo *et al*., 2014; Molari *et al*., 2014) highlighting the potential in the use of TW effluents for irrigation in order to reach high herbaceous biomass production. The perennial species, such as *Arundo donax* (L.) and *Miscanthus* × *giganteus* Greef et Deu., proved to be the most productive and with high heating values (Table 4.5). The two species are declared as "poor" crops due to the low economic value of their biomass; therefore, the use of conventional sources of water and chemical fertilizer is not feasible. However, where wastewater is readily available at low cost, *A. donax* and *M. giganteus* can be a very interesting option for wastewater reuse with benefits for the environment and farm income.

4.7.3 Design objectives

Wastewater with high concentrations of ammonium, sulphides, salts and metals may inhibit nutrient uptake and consequently the growth of wetland plants. Therefore, it is essential to know the quality of wastewater to be treated in order to select appropriate wetland plant species, which are known to have different capacity for nutrient uptake, different preferences for nitrogen forms and have evolved various adaptive mechanisms that protect them against the toxicity of inorganic substances.

Wastewaters with high concentrations of nutrients stimulate the growth of wetland plants that can accumulate, preferably on the above-ground tissues, more nutrients than that are needed for growth (so called 'luxury uptake'); however, the timing for biomass harvesting can influence the removal of nutrients from the TW:

- A single annual harvest performed in late summer, before the translocation of nutrients to the root system, allows removal of the maximum amount of nutrients from the TW. However, high concentrations of nutrients in the biomass can cause corrosive effects on the combustion plant. Furthermore, low concentrations of nutrients and carbohydrates in the roots could result in reduced plant regrowth in the next year. If the biomass is used for biogas production, a single harvest in late summer or two harvests at early growth stages have the advantages of lower lignin contents with better digestion kinetics and consequently higher methane yield.
- A single annual harvest performed in late autumn implies a reduction of biomass yield, due to loss of leaves, but ash and moisture contents decrease, creating a higher biomass quality for direct combustion.

Many metals such as Cu, Fe, Mn, Ni and Zn are involved in numerous plants' metabolic processes as constituents of enzymes and other proteins. However, they can become toxic if their concentration is higher than a specific critical point, as they can lead to a range of interactions at the cellular and molecular levels. In general, wetland plants are not hyper-accumulators; they store metals in below-ground tissues (Batty & Younger, 2004). Consequently, the health risks of above-ground wetland biomass as a solid fuel appear to be comparable to more traditional fuel sources.

In contrast, the low bulk density of biomass produced by herbaceous wetland plants can cause an incomplete combustion with a consequently poor air quality from cooking fumes and an increase of health risks (WHO, 2016).

4.7.4 Specific considerations during design, for construction and operation

There are some key parameters that should be considered in the design and construction phase of a TW for biomass production:

- In order to produce more biomass for energy purposes, the amount of nutrients in the supplied water has to be adjusted to the nutrient needs of the target crop. This leads to the situation when a complete elimination of nutrients in TW is not desired, therefore TW can be simplified or reduced in area.
- Appropriate TW technology has to be selected: FWS wetlands have lower energy production potential compared with subsurface flow TW owing to aquatic plants having lower biomass production per area unit compared with mesophytes.
- Appropriate plant species have to be selected in order to produce more biomass for energy purposes.
- Additional harvesting or thinning of the stand has to be considered in order to increase biomass production.
- From the perspective of plant regrowth and longevity, harvesting should not occur until plants are sufficiently mature that rhizomes have been resupplied with nutrients and carbohydrates.
- Appropriate ash disposal has to be considered, namely ash content of wetland biomass (usually 5–10% of dry mass) is higher compared with wood ($<1\%$) (Avellán & Gremillion, 2019).

4.8 TREATMENT FOR PATHOGEN REMOVAL

Fasil Ejigu Eregno[1,2]*, Adam M. Paruch*[2]*, Trond Mæhlum*[2] *and Jaime Nivala*[3]

[1]*Faculty of Engineering and Technology, The Arctic University of Norway (UiT), Narvik, Norway*
[2]*Division of Environment and Natural Resources, Norwegian Institute of Bioeconomy Research (NIBIO), PB 115, NO-1431 Ås, Norway*
[3]*Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany*

4.8.1 Introduction

Wastewater contains various pathogenic microorganisms that are a health risk to human beings. These can be divided into five categories: viruses, bacteria, protozoa, helminths and fungi. The diversity and magnitude of pathogens in wastewater vary with the level of endemic disease in the community, discharge sources, and seasonal factors. The removal of microbiological contamination is one of the targets for TWs. There have been several studies published on microbial water quality improvement using wetland systems (Vymazal, 2005; Wu *et al*., 2016). Pathogen treatment relies on complex mechanisms of multiple chemical (oxidation, UV radiation, exposure to plant biocides, unfavourable water chemistry, adsorption to organic matter and biofilm), physical (sedimentation, adsorption and filtration), and biological (predation, biolytic processes, antibiosis, natural die-off) factors, which often act in combination (Stefanakis & Akratos, 2016; Weber & Legge, 2008). The effectiveness of these treatment mechanisms is dependent on a synergistic effect of natural (environmental) and technical (design, operation and maintenance) features, which affect the various microbial pathogens differently. Pathogen removal in TWs varies depending on incoming wastewater characteristics, temperature, microbial activity, microbial ecology, plant type, substrate type, and biofilm interactions, among others (Vymazal, 2005; Wu *et al*., 2016). As TWs are complex in their chemistry, hydraulics, and distribution of specific removal mechanisms, at the time of writing, it is not possible to provide simplified design recommendations for pathogen removal. Principally, TWs are not designed solely for the removal of microbial contaminants.

4.8.2 Processes required and TW type to be used

The variety of pathogenic microorganisms and their diverging properties demand different technological processes for efficient removal. For instance, longer hydraulic retention time extends pathogens exposure to the specific removal processes, such as sedimentation, adsorption to organic matter and soil particles, predation, and the impact of toxins from microorganisms or plants, and UV radiation. Furthermore, most of these removal processes are directly and indirectly influenced by the different internal and external environmental conditions such as temperatures, pH, seasonal fluctuation, wastewater composition, availability of dissolved oxygen and organic carbon source.

Sedimentation, filtration, and adsorption phenomena play an important role in the removal of microbial pathogens. Sedimentation has been reported to be effective in removing some bacteria, such as coliforms, faecal streptococci, and helminth eggs, due to their higher settling velocities. However, protozoan (oo)cysts and some bacteria have much lower settling velocities, and these pathogens can only be effectively removed by sedimentation (in FWS wetlands) or filtration (in HF or VF wetlands) if they are attached to larger particles, in which case their removal correlates with particle removal. Viruses are generally stable in suspension and effectively removed by adsorption.

In FWS wetlands, a combination of densely vegetated and open-water zones in warm climate regions maximise pathogen removal. This is related to the fact that the rhizosphere and root zone of wetland vegetation play a substantial role in transporting contaminants, serving as pathways for gases, and moving grains into pore space (Scholz *et al.*, 2002), hence maximizing filtration and sedimentation of particles to which pathogens are adsorbed, while the open water zones maximise UV disinfection (Greenway, 2005). Furthermore, FWS wetlands can also act as a natural filter that holds particles and inhibits sediments against re-suspension by stabilising them within root zones. Rhizomes create a natural barrier for parasite eggs, thus they can be easily destroyed by antagonistic organisms (e.g., earthworms) settled in the wetland beds (El-Khateeb *et al.*, 2009; Reinoso *et al.*, 2008). Virus removal efficiencies in FWS wetlands are reported to range between 40% and 99% (Kadlec & Wallace, 2009).

In HF wetlands, a removal of up to 3 $\log_{10}$ units of faecal indicator organisms (such as *E. coli*) can be expected (Dotro *et al.*, 2017; Wu *et al.*, 2016), but the literature is replete with reports of removal rates of the order of 1–2 $\log_{10}$ unit removal (Caselles-Osorio *et al.*, 2011; García *et al.*, 2008; Neralla *et al.*, 2000; Nivala *et al.*, 2019a). Median removal rates of viruses in 53 HF wetland studies was reported to be 1.6 $\log_{10}$ units (Kadlec & Wallace, 2009).

In unsaturated VF wetlands, *E. coli* removal has been reported to be better in systems with a finer filter material. Tanner *et al.* (2012) report 3.2 $\log_{10}$ removal of *E. coli* in VF wetlands with coarse sand ($d_{10} =$ 0.64 mm) and 1.9 $\log_{10}$ removal for VF wetlands with fine gravel sand ($d_{10} = 1.1$ mm) as the main filter media. A similar trend is also reported in Headley *et al.* (2013), where unsaturated VF wetlands with coarse sand (1–4 mm) removed up to 2.1 $\log_{10}$ units, and unsaturated VF wetlands with gravel (4–8 mm) only removed 0.8 $\log_{10}$ units on average. Nivala *et al.* (2019b) also report the benefit of two unsaturated VF wetlands in series, with an improvement in *E. coli* removal from 1.7 $\log_{10}$ units to 3.3 $\log_{10}$ units with the addition of a second-stage cell.

E. coli removal in aerated wetlands depends on the internal hydraulics of the system. Horizontal-flow aerated wetlands with typical tanks-in-series hydraulics ($3 > N > 6$) have been shown to achieve up to 4.0 $\log_{10}$ unit removal in a single treatment cell (Headley *et al.*, 2013; Nivala *et al.*, 2019b). With annual mean effluent concentrations of *E. coli* below 700 MPN/100 mL, single-stage HF wetlands with aeration can meet the threshold value of 1,000 MPN/100 mL that is generally recommended for unrestricted use in irrigation (Mara, 2003). VF aerated wetlands, on the other hand, are reported to have very well mixed hydraulics (1.1 tanks-in-series; Boog *et al.*, 2014), and can only achieve on the order of 2.0 $\log_{10}$ unit removal of *E. coli* in one treatment cell (Headley *et al.*, 2013).

In general, combining different types of wetland systems can help to improve pathogen removal from wastewater (Wu *et al.*, 2016), but still not to the degree that would make effluent safe for unrestricted reuse. Therefore, disinfection units are generally required to fulfil quality obligations for reuse, as well as to comply with requirements set in certain directives, such as the Habitats Directive (92/43/EEC) (EEC, 1992) and Bathing Water Directive (2006/7/EC) (EC, 2006).

4.9 TREATMENT OF MICROPOLLUTANTS

Kela P. Weber[1] and Jaime Nivala[2]

[1]Environmental Sciences Group, Department of Chemistry and Chemical Engineering, Royal Military College of Canada, Kingston, ON K7 K 7B4, Canada
[2]Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany

4.9.1 Introduction

The definition of a micropollutant varies depending on the perspective and region, however in general it encompasses a substance, or residue, originating from synthetic products and anthropogenic activities which are found at concentrations in the low part per billion (ppb) and part per trillion range in the environment. They include classes such as pharmaceuticals and personal care products (PPCPs), industrial chemicals, pesticides, endocrine-disrupting chemicals (EDCs) including hormones, and nanomaterials. Micropollutants either originate from or are released during standard practices such as pesticide spreading on agricultural land, fighting fires, or via the usage of products such as pharmaceuticals, textiles, or electronics. Their inherent persistence due to their recalcitrant nature allows micropollutants to reach stormwater systems, sewer systems, wastewater treatment plants, and in some cases natural receiving waters. Complete biodegradation of micropollutants is challenging, and in most cases, does not readily occur under standard or natural environmental conditions. New micropollutants emerge each year as analytical techniques for detection are improved, and new substances are developed and incorporated into new processes or products. Conventional activated sludge treatment plants are not specifically designed to handle micropollutants, although the removal of micropollutants does occur (Grandclement *et al*., 2017). Over the last decade TWs have been evaluated and adapted for the removal of micropollutants.

4.9.2 The removal of micropollutants from water in treatment wetlands

TWs have been shown to remove micropollutants with varying degrees of success. Initial studies from the early 2000s were completed on HF systems (Matamoros & Bayona, 2006), with additional configuration types and intensified systems evaluated more recently (Nivala *et al*., 2019b). Although some of the first evaluations for specific micropollutants were completed at laboratory scale (micro-scale, meso-scale) a reasonable body of data at pilot and full scale is available. Table 4.6 summarizes selected and representative case studies evaluating the removal of micropollutants from different TW designs.

It is clear that TWs hold great promise for the removal of micropollutants. Removal efficiencies greater than 90% are seen for many micropollutants in several different TW configurations. In Table 4.6, for almost all cases where reasonable removal efficiencies were reported, adding aeration offered additional benefits. However, as in conventional water treatment, some micropollutants remain recalcitrant in TWs. Carbamazepine is a good example where perhaps additional innovation in design, operation, or even microbiological mediation/design is required before reasonable removal rates can be gained. Additional data compilations and recent reviews for the removal micropollutants in TWs can be found in Gorito *et al.* (2017) and Vymazal *et al.* (2017).

Table 4.6 Micropollutant removal efficiency (%) from selected pilot and full-scale treatment wetlands.

Parameter	**Caffeine**	**Ibuprofen**	**Naproxen**	**Benzotriazole**	**Diclofenac**	**Acesulfame**	**Carbamazepine**	**Triclosan**
Wetland Type								
HF	83[i],98[ii],84[iii]	28[i],80[ii],55[iii]	32[i], 90[ii]	25[i]	25[i],45[ii],41[iii]	5[i]	13[i]	65[iii]
HF + aeration	99[i]	99[i]	99[i]	85[i]	70[i]	62[i]	−4[i]	–
VF	96[i]	95[i], 95[iv]	90[i]	62[i]	53[i], 65[iv]	−5[i],	−9[i]	73[iv]
VF + aeration	99[i]	98[i], 99[iv]	94[i]	73[i]	74[i], 58[iv]	54[i]	−1[i]	86[iv]
Fill-and-drain	98[i]	93[i]	88[i]	61[i]	40[i]	59[i]	−1[i]	–

[i]Nivala *et al.* (2019b). Data from outdoor pilot-scale systems treating wastewater (5.6–13.2 m^2) in Germany. Removal efficiency is based on mass. All treatment wetland designs and operational details described in Nivala *et al.* (2013a).
[ii]Vymazal *et al.* (2017). Combined mean removal efficiency from four different full-scale systems (300–2100 m^2) in the Czech Republic. Removal efficiency is based on concentration.
[iii]Matamoros and Bayona (2006). Only data from a 0.27 m depth (54–56 m^2) pilot-scale system in May 2004 in Spain is shown. Removal efficiency is based on concentration.
[iv]Avila *et al.* (2014). Data from outdoor pilot-scale systems treating wastewater (5.6–13.2 m^2) in Germany. Only the gravel-based substrate systems are summarized. Removal efficiencies are concentration-based and were calculated from reported means. All treatment wetland designs and operational details are described in Nivala *et al.* (2013a).

4.9.3 Mechanisms involved in the removal of micropollutants in treatment wetlands

Mechanisms involved in the removal of micropollutants from TW influent may include microbiological degradation/transformation, plant uptake and metabolization, adsorption to biofilm or substrate, volatilization, abiotic degradation including hydrolysis or photocatalzyed oxidation, and other advanced reduction/oxidation reactions based on novel substrates or intensification schemes (Button *et al*., 2019). The majority of full-scale studies have not been able to evaluate the mechanisms involved in micropollutant removal, however some micro-scale and meso-scale studies have been able to lend some understand of mechanistic actions for specific micropollutants. For example, Matamoros *et al.* (2008) showed the pharmaceutical ibuprofen to be removed largely by aerobic microbial degradation processes. Button *et al.* (2016) and Auvinen *et al.* (2017) showed silver nanoparticles (AgNPs) to be removed mostly via adsorption to biofilm and settling into the sediment. Button *et al.* (2019) showed the antimicrobials sulfamethoxazole and triclosan to be initially removed via adsorption to biofilm, but later biodegraded within biofilms. Lv *et al.* (2016a) showed the pesticides imazalil and tebuconazole to be degraded by emergent wetland plants in hydroponic studies, attributing the majority of treatment to enantioselective degradation within the plants, however they did note that any microbial degradation in solution could not be differentiated.

Seasonal performance variations and microbial community adaptations have also been observed in TWs treating micropollutants. For example, Lv *et al.* (2016a) showed the pesticides imazalil and tebuconazole removal to be higher in the summer. The same research team also showed that the microbial communities of those TWs were adapting during those summer periods (Lv *et al*., 2016b), and that the biofilm microbial communities functional ability to utilize amine/amides and amino acids was positively related to the degradation of imazalil and tebuconazole (Lv *et al*., 2017).

Although the reported removal of micropollutants can be quite high in TWs, and other water treatment systems, the aspects of constituent transformation need to be accounted for. In many cases analytical methods for the detection of micropollutants are still developing, and these methods are often focused on gaining very low detection limits to better identify micropollutants in environmental media. However, if focused on looking for a specific micropollutant in its original form found in the influent, sometimes removal efficiency can be seen to be quite high, when in actuality the micropollutant is only partially augmented and not mineralized. In some cases this could mean the original micropollutant is transformed into a more toxic form (Escher & Fenner, 2011). For example, Matamoros *et al.* (2008) were able to show the partial transformation of ibuprofen to carboxylated and hydroxylated forms in HF pilot-scale systems through analytically scanning for compounds of similar molecular weight to ibuprofen. This additional level of analytical inquiry, with an added mass balance approach, allowed the authors to surmise that overall aerobic conditions were more conducive to the overall mineralization of ibuprofen in TWs.

Although challenging, the transformation of micropollutants can be studied in concert with mechanistic evaluations. Zhang *et al.* (2018) found microbial communities with an increased utilization of amines/amides and amino acids to be associated with improved ibuprofen removal. However, they further went on to identify co-metabolic processes involving L-arginine, L-phenyloalanine, and putrescine as potentially linked to ibuprofen transformations. In the same set of studies, Zhang *et al.* (2019) were able to also link the metabolic processing of the x-ray contrast agent iohexol to the TW microbial communities' use of putrescine in the summer and D-cellobiose, D,L-alpha-glycerol phosphate in the winter, suggesting co-metabolic processes to be important in the transformation and degradation of iohexol.

4.9.4 The resilience of treatment wetlands to the effects of micropollutants

Although TWs have been shown to remove micropollutants from water, there is still some concern over the effects micropollutants may have on the TWs themselves. The effects of pesticides and specific PPCPs such as antibiotics and antimicrobials on TW microbiological communities are of obvious concern (Lv *et al.*, 2016b; Weber *et al.*, 2011). Additional micropollutants such as silver nanoparticles, which are used in textiles for their antimicrobial properties (Button *et al.*, 2016), and per- and polyfluorinated alkyl substances (PFAS), which tend to concentrate at interfaces and are exceptionally resistant to degradation of any kind (Milley *et al.*, 2018), may also pose long-term risks to the efficacy and operational abilities of TWs. Despite the concern, antibiotics have generally been shown to cause little to no detrimental effects on TWs. Weber *et al.* (2011) showed that although exceptionally high levels of ciprofloxacin (2 ppm) had some effects on the microbiological regime of VF systems during start-up, the TWs recovered quite quickly. Button *et al.* (2019) showed that although TW microbial community activity was detrimentally affected by triclosan and sulfamethoxazole via benchtop assays at 100 ppb, no clear detrimental effects to water treatment capabilities (COD, N), hydrology, plants, or the microbial community was seen at the mesoscale. Similarly to the case for triclosan and sulfamethoxazole Button *et al.* (2016) showed that although clear detrimental effects to microbial communities could occur for citrate-coated AgNPs and ionic Ag at 500 ppb, no clear detrimental effects could be seen at similar concentrations in microcosms. Silver was however found to concentrate in the biofilm which contributed to the development of a more silver-resistant microbial community.

4.9.5 Summary

Treatment wetlands can remove micropollutants from water, and in many cases degrade these constituents over time. Adsorption to biofilm, microbial degradation and even plant degradation have been attributed to the removal of micropollutants in several studies. At present, TWs do not seem to be adversely affected (to a measurable degree) by micropollutants, including those with antimicrobial properties. TW removal rates are similar or in some cases superior to conventional activated sludge system performance, and TW micropollutant removal performance continues to be improved largely through intensification.

4.10 LANDFILL LEACHATE TREATMENT

Tom Headley[1] and Dirk Esser[2]

[1]Wetland & Ecological Treatment Systems, 84 Melbourne Street, East Maitland, NSW 2320, Australia
[2]Société d'Ingénierie Nature &Technique (SINT), Chef-Lieu, F-73370 La Chapelle du Mont du Chat, France

4.10.1 Introduction

Landfill leachate is the contaminated liquid which percolates through and drains from a solid-waste landfill. It is primarily derived from rainfall or groundwater entering the waste heap and from moisture contained in the waste material itself. As the liquid leaches through the heap, it dissolves and entrains soluble and particulate contaminants from the waste material, while promoting the decomposition and release of biodegradable substances. Landfill leachate normally contains relatively high concentrations of organic matter and ammonium nitrogen, while in some cases it may also contain significant levels of salts, metals and xenobiotic organic compounds. The specific composition varies significantly depending on the age and design of the landfill, the type of waste deposited in it, the climatic conditions and the practices applied for managing closed and active areas of the landfill. To prevent the excessive accumulation of this liquid inside the landfill, which can promote anaerobic conditions and impose a rising load on the landfill lining system (if it exists), the leachate is regularly extracted from the landfill or it will naturally flow out of the landfill. Thus, leachate needs to be managed accordingly, including appropriate treatment and disposal. Landfills tend to generate leachate for many decades, even after closure and capping of the landfill. A continuous production of leachate persists from all non-watertight landfills, which represents a large majority of existing old landfills. TWs are increasingly being integrated into leachate treatment systems, due to their robust performance and low operation and maintenance costs over the long-term. Low operation requirements are even more important at closed landfills, with no revenue-generating activities and staff on site to operate a treatment plant.

4.10.2 Design objectives

TWs for landfill leachate are most commonly designed with the objective of removing Total Kjeldahl Nitrogen (organic N plus ammonium-N) and organic matter (BOD_5 and COD), which are the most common contaminants of concern in leachate. In particular, ammonia is a persistent pollutant in leachate even decades after the closure of the landfill (Table 4.7).

Table 4.7 Nitrogen composition in leachate over time (Reproduced with permission from McBean & Rovers, 1999).

Parameter	Leachate 1–2 Years Old	Leachate 10 Years Old
Ammonia NH_3	1,000–2,000	500–1,000
Organic N	500–1,000	10–50
Nitrate NO_3	0	0–10

The required level of removal of these contaminants will depend on the fate of the final discharge or disposal method. Common disposal methods include:

- Discharge to sewer if nearby, typically requiring moderate reduction of TKN and BOD_5 or COD down to concentrations similar to raw sewage so as not to overload the sewage treatment plant. In some cases, other contaminants may need to be considered, such as salinity, phosphorus, heavy metals, or hydrocarbons.
- Land application or irrigation reuse, for which BOD_5 typically needs to be reduced to low concentrations, while the required level of nutrient removal will depend on the mass load that can be sustainably irrigated onto the available area of land, considering the crop uptake and harvesting rates, local climate and regulatory perspectives.
- Discharge to a nearby waterway, such as stream, lake or sea. This typically requires the highest level of treatment to satisfy stringent environmental standards and avoid eutrophication, nuisance issues and ecotoxicological impacts in the receiving environment.

Due to the typically long service life of landfill leachate treatment systems (outlasting the operational life of the landfill itself), a common design objective is to develop a system that will operate for several decades with low operating costs. Thus, it is preferable to minimise the number of electro-mechanical parts (e.g., pumps, mixers, blowers, control valves, mechanical screens, chemical dosing equipment), which tend to require regular servicing and replacement. Since landfills are usually not locations where a full-time wastewater treatment technician is available and after the landfill closes there may not be any operational staff at the site, it is also a goal to design systems which can provide robust treatment with minimal operator attention. In these regards, natural treatment technologies have many advantages over conventional processes, because even the most intensified and advanced treatment wetland systems require relatively little operator attention and utilize very few mechanical equipment for the process (e.g., one or two pumps or blowers).

4.10.3 Processes required and TW type to be used

Removal of TKN requires the mineralization of organic N and nitrification of ammonium-N into nitrate-N. Nitrification is an oxic process and mineralization of organic matter (organic N and BOD_5) typically occurs rapidly via aerobic pathways. Thus, treatment wetland technologies with relatively high oxygen transfer rates which promote conditions conducive for oxic processes are preferable for at least the initial stages of treatment. Commonly applied wetland technologies for such purposes include:

- VF wetlands (with the leachate intermittently loaded across the upper surface of an unsaturated bed of filter media)
- Aerated subsurface-flow wetlands (leachate flowing either vertically or horizontally through a submersed bed of actively aerated filter media)
- FWS wetlands (only applicable if influent concentrations are relatively low, as FWS wetlands can not provide fully aerated environment but require less O&M effort).

If appropriately sized and designed, such systems can achieve high levels of TKN reduction, while also removing BOD_5, hydrocarbons and some xenobiotic organic compounds.

If total N removal is required, then the design will need to include the denitrification process to remove the nitrate generated from the upstream nitrification process. Denitrification requires anoxic conditions and an available source of organic carbon for the denitrifying bacteria. Wetland technologies that are particularly suitable for denitrification include:

- FWS wetlands, in which the emergent vegetation provides a direct internal source of organic carbon for the process, but this can require very large surface areas
- HF wetlands, which tend to promote anoxic conditions and can also return limited amounts of organic carbon from the vegetation to the subsurface water (Zhai *et al.*, 2013). To boost organic carbon availability for denitrification, wood chips or other organic substrate are sometimes mixed with the filter media. Another option for denitrification is to add a liquid external carbon source into the inflow of the HF wetland (Rustige & Nolde, 2007).

Recirculation of the treated effluent back to the inlet of the system is sometimes employed to dilute the concentration of contaminants such as ammonium in the inflowing leachate (e.g., to alleviate toxicity issues), utilise the organic carbon that may be in the raw leachate for the purpose of denitrification and/or supply some of the alkalinity (derived from the denitrification process) needed for nitrification.

4.10.4 Specific considerations during design and construction

There are several key parameters that should be considered in the design and construction of a TW system to treat landfill leachate, including:

- *Leachate quality*. This varies from one landfill to another and usually varies over the life of the landfill (Figure 4.2). The type and concentration of contaminants depend on the type of wastes disposed in the landfill and the efficiency with which water is prevented from entering the landfill. Aside from the main parameters of concern, such as TKN, BOD_5, COD, TSS, TP, hydrocarbons and heavy metals, specific attention should be paid to the concentrations of Total Dissolved Solids (salinity), sodium, chloride, boron, iron, manganese, aluminium, strontium and zinc, which can sometimes be at high enough concentrations in leachates for toxicity symptoms to develop in the wetland vegetation.
- *Landfill characteristics*. Various characteristics of the landfill will have an influence on the likely contaminant concentrations and flow rates generated now and in the future. The age of the landfill and the types of wastes accepted have an influence over the concentration of ammonium-N and the biodegradability of organics in the leachate. Depending on how industrial, agricultural, medical and other hazardous waste materials have been received, segregated and contained within the

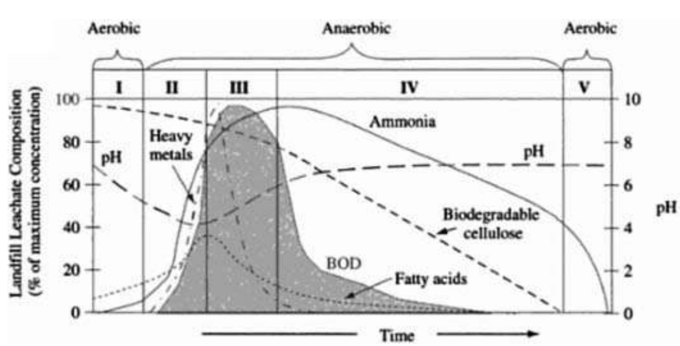

Figure 4.2 Changes in the composition of leachate with aging of the landfill (from DoE, 1995).

landfill, the leachate may contain a range of problematic organic compounds (e.g., hydrocarbons, pharmaceuticals, PCBs, PFAS and other xenobiotic compounds) or heavy metals, which need to be considered in the design. The life expectancy and management plan for the landfill will determine the required life span for the leachate treatment system (typically in the order of many decades) and the dynamics of leachate generation over that time as old landfill cells are capped while new cells may be created. Whether or not the landfill has been constructed and operated as a sanitary landfill (i.e., lined to isolate from groundwater, daily coverage of waste and minimization of stormwater generation) will influence the amount of groundwater and stormwater ingress into the landfill, thereby affecting the volume and concentration of the leachate.

- *BOD/COD ratios*. The BOD/COD ratio evolves with the ageing of the landfill, from around 0.8 in young landfills down to less than 0.1 in old landfills (Figure 4.3). This can become a problem in mature and old landfills, where relatively high COD outflow concentrations can persist while BOD concentrations are very low. These recalcitrant organics are very difficult to remove through any biological processes, be it treatment wetlands or more conventional processes, leading to persistent COD concentrations in the treated leachate. Still, biological systems with very long retention times will be more efficient to remove a fraction of this COD than compact systems with low retention times. Generally, at low BOD/COD ratios and if there are discharge limits on COD, pilot studies will be required to determine the kinetics of the degradation of this COD and whether it is possible to reach the required discharge standard with biological treatment. In many cases where there are discharge standards for COD, this may lead to the necessity to implement a non-biological polishing stage, such as an activated carbon filter unit.
- *Oxygen demand for treatment*. The specific oxygen demand for removal of BOD_5 and ammonium (nitrification) should be calculated and considered in the wetland sizing with reference to published oxygen transfer rates (see for example: Kadlec & Wallace, 2009; Nivala *et al.*, 2013b) for the wetland technology selected.
- *Ammonium concentrations, toxicity and inhibition*. Influent NH_4-N concentrations greater than about 300 mg/L (common in landfill leachate) may impose issues of toxicity on the wetland plants and inhibition on nitrifying bacteria. Selection of wetland plants with a high resilience to toxicity from

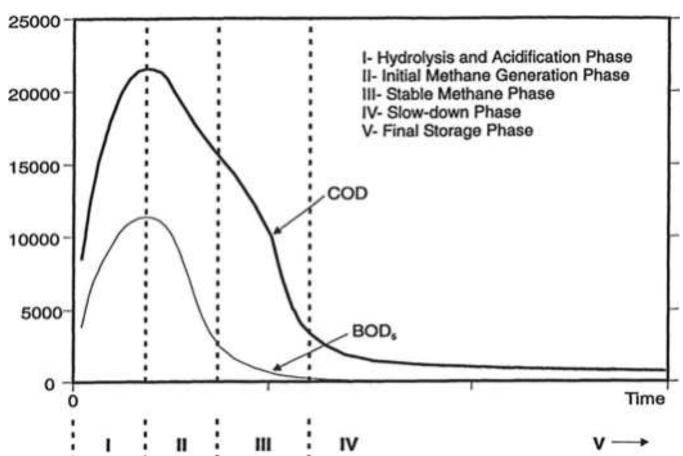

Figure 4.3 Changes of COD and BOD_5 concentrations with aging of the landfill (adapted from McBean & Rovers, 1999).

ammonia (and other elements) may be necessary. As previously mentioned, recirculation of treated effluent to the inlet where it can be mixed with the influent leachate is one strategy for reducing the concentration below toxic levels via dilution. Recirculation also helps to attenuate and stabilize the influent quality and flow rates. However, very high rates of recirculation are needed for high influent concentrations, which can significantly increase the required size of the wetland and hydraulic components (pipes and pumps).

- *Nitrification rates and alkalinity availability*. The nitrification process consumes approximately 7.1 g of alkalinity (as $CaCO_3$) per gram of ammonium-N nitrified to nitrate (Kadlec & Wallace, 2009). Thus, it is important to conduct an alkalinity balance, comparing the mass of alkalinity in the leachate against that required to remove the necessary mass of ammonium-N via nitrification. In some cases, there may be insufficient alkalinity in the leachate to supply the high rates of nitrification needed, imposing a limitation on the rate of nitrification possible without supplemental addition of alkalinity. Denitrification returns about 3 g of alkalinity ($CaCO_3$) per gram of nitrate-N reduced. Thus, as previously mentioned, recirculating treated leachate after the denitrification step back to the inlet of the nitrifying process can help to alleviate alkalinity limitations. Integration of alkalinity-rich media (e.g., limestone, $CaCO_3$) into the wetland substrate, can be considered as a means of supplementing the leachate alkalinity. Alternatively, dosing alkalinity (e.g., with caustic soda or lime slurry) into the leachate may be necessary. As highlighted below, careful consideration must also be given to the clogging risk posed by the presence of excessive calcium or magnesium carbonates.
- *Iron concentrations and potential clogging*. Some leachates can contain significant quantities of the reduced ferrous form of iron (Fe^{2+}) which will oxidise into the ferric form (Fe^{3+}) and precipitate as iron hydroxide ($Fe(OH)_3$) when exposed to the aerobic conditions provided for nitrification. This can increase the risk of media clogging in subsurface-flow wetland systems used for nitrification (e.g., VF or aerated wetlands) (Nivala *et al*., 2007). Thus, a preliminary treatment step may be needed to remove the bulk of this iron in a manner that will not pose a clogging risk, such as via aeration and sedimentation within a pond prior to the nitrification wetland step. In many landfills, ponds are used in any case to collect and store the leachate prior to treatment, so such design modifications may be relatively minor.
- *Precipitation of calcium carbonates and potential clogging*. This risk occurs mainly in younger landfills with still significant biological activity which are in contact with limestone substrates. Here the leachates can contain high concentrations of dissolved calcium or magnesium and the increase of the pH due to the stripping of carbon dioxide when agitating the leachate under atmospheric conditions can lead to substantial precipitation of calcium and/or magnesium carbonates and subsequent risks of clogging subsurface flow wetlands. Like for the removal of iron, a preliminary treatment step for stripping of carbon dioxide prior to treatment in subsurface flow wetlands may be required. However, it should be big enough not only to allow for stripping of the carbon dioxide but also for the sedimentation of the calcium carbonate or for the formation of non-clogging limestone deposits on specific contact surface areas. If this is not the case, then a limestone deposit will build up in the filter material of the subsurface flow wetland, ultimately leading to clogging. For example, despite of an aerated pond upstream of a VF wetland in France, calcium concentration in the filter media around the distribution points increased from 0.5% to 6% in one year (ADEME, 2013).
- *Climate conditions and the water balance*. Leachate production is partly a result of rainfall infiltration into the landfill and is therefore affected somewhat by the pattern of rainfall events at the location (although there is usually substantial attenuation of flows provided by the passage through and retention within the landfill itself). In climates with very cold winters, consideration

may need to be given to the wetland technology selection and provision of an insulation cover of mulch over the top of subsurface-flow wetland systems and other means of preventing the leachate from freezing in the wetland and associated pipework. Many biological treatment processes, such as nitrification and denitrification, tend to proceed more slowly at cold temperatures, which needs to be factored into the sizing calculations and process design. In extreme cases, with extended periods of less than $-10°C$ air temperatures, seasonal storage of the leachate through the winter and subsequent treatment and discharge during the warmer months, may be necessary (Mæhlum, 1999).

If the surface area of treatment wetlands required to achieve treatment is relatively large, then the water balance can become problematic under extreme climatic conditions. For example, in hot, arid climates, wetlands with a relatively low hydraulic loading rate (i.e., a relatively large area relative to the inflowing leachate volume), then evapotranspiration losses may represent a significant portion of the influent hydraulic loading rate during summer, leading to problematic salt concentrations at the outlet, or no outflow in the worst case (and subsequent salinity impacts on the wetland biota). In tropical, monsoonal climates, the amount of rain falling on the wetland catchment during the wet season may be several-fold higher than the leachate hydraulic loading rate, leading to a significant increase in the volume of treated leachate that needs to be managed or disposed of downstream. Therefore, it is important to compile a water balance (on a monthly time-step as a minimum) so any potential issues can be anticipated at the design stage. In some cases, it may be necessary to reconsider the wetland technology selection and look for avenues to reduce the footprint (e.g., by combining with more intensified treatment processes, whether they be wetland-based or more conventional).

- *Plant selection*. Selection is sought of a diverse range of locally occurring native plant species adapted to the climate that can thrive under the hydrologic conditions of the specific wetland type adopted. Care should be taken to identify plants that can tolerate the specific water quality characteristics, as some leachates contain considerable concentrations of salts, boron and other potentially toxic elements which can compromise the health and vigour of the wetland vegetation, especially in the medium to long term. On the long term (>10 years), plant growth can be limited by the low phosphorus concentration usually occurring in leachate.
- *Flammable and toxic gases*. Landfills and their leachates can emit significant quantities of flammable and potentially toxic gases, such as methane and hydrogen sulphide. Thus, appropriate risk assessments should be conducted during the design, construction and operational phases to minimize the risk of ignition, explosion and to identify hazardous areas of the site where such gases may accumulate to dangerous levels. While the risks are generally reduced by using natural treatment systems, due to inherently low use of electro-mechanical equipment (ignition sources) and predominance of extensive, open spaces (which tend to dissipate rather than accumulate gases), consideration may need to be given to the location and type (e.g., explosion-proof) of pumps and blowers if used. In some cases, pneumatic pumps operated by a remote air compressor located a safe distance from explosion hazards may be warranted. Manholes especially at the inlet and at intermediate treatment stages, or at the outflow of HF wetlands should be well ventilated in order to prevent accumulation of methane and highly toxic hydrogen sulphide gases. This can sometimes be in conflict with insulation issues in cold climates.

If, after consideration of the above design issues, several significant questions remain, then it may be advisable to incorporate a pilot study into the design development process, to evaluate key questions and minimize the design risk before proceeding to detailed design and construction.

4.11 INDUSTRIAL WASTEWATER TREATMENT

4.11.1 General considerations

Alexandros Stefanakis

Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

Introduction

The application of TWs for industrial wastewater can be a quite complicated and challenging task. Industrial wastewater can have a large variety of sources, physicochemical composition and pollutants nature, based on the industrial process and the raw materials/chemicals used. This is why industrial wastewaters are more complex than domestic and municipal ones, which makes it more difficult to develop an effective wetland design (Stefanakis, 2018). Characteristics that can be found in industrial wastewater are various and can include:

- High organic load, usually expressed as BOD_5 and/or COD
- Low biodegradability (i.e., low BOD/COD ratio)
- High solids content
- High nutrients concentration
- Presence of toxic compounds
- Pollutants variety, e.g., hydrocarbons, oil and grease, phenols, heavy metals etc.
- Intense colour, high turbidity, salinity, metals/metalloids, sulphate etc.
- Presence of emerging compounds – micropollutants
- Extreme pH values (acidic or alkaline)
- Fluctuations in flow rates, loads and even composition.

Over recent years, there has been an obvious increase in studies and applications of wetland systems for various industrial applications, indicating the new challenges arising from the industrial sector. Current results and experiences imply that there is indeed a high potential for wetland systems to be further applied in various industrial sectors. The various industrial sources include, but are not limited to, the following (Stefanakis, 2018; Sultana *et al.*, 2015; Vymazal, 2014; Wu *et al.*, 2015):

- *Petrochemical and chemical industry*. Oil and gas processing, refineries, coke plants.
- *Food and beverage industry*. Wineries, breweries, fish and shrimp aquaculture, sugarcane-mills, meat processing and slaughterhouses, vegetable processing, coffee and soft drinks processing, distilleries, starch and yeast processing, potato and molasses processing.
- *Agro-industry*. Olive mills, dairy farms, livestock farms, vinegar production, trout farms.
- *Wood and leather processing*. Tanneries, textile industries, pulp and paper mills, cork processing.
- *Drainage*. Mine drainage, landfill leachate, runoff and stormwater from industrial sites.
- *Others.* Cosmetics and pharmaceuticals industry, dewatering of industrial sludge, car-wash facilities, laundries, steel production.

Design objectives

As for domestic and/or municipal wastewaters, the ultimate goal of a wetland design is the effective treatment of the industrial effluent and the optimal reduction of pollutant load. Depending on the location of the industrial facility, e.g., if it is located within an industrial zone, the level of treatment can reach the legal limits for discharge in a centralized sewer network or even stricter limits for disposal to a surface water body. It is also common that specific standards are required for the treated effluent to allow for its reuse in the industrial process without creating any issues of re-contamination. Wetland systems are also often viewed by the industry as attractive alternatives to conventional treatment technologies, mainly due to the reduced operation costs, the minimum energy consumption and the minimum need for specialized staff.

The selection of a TW by an industrial entity also aims at covering the continuous need for sustainable solutions and processes (Nikolaou & Stefanakis, 2018). By adopting green practices in their wastewater treatment strategy, many industries can improve their green profile towards the society and the public, which is an essential tool for advancing operations and improving the corporate financial performance. The modern approach of proactive adoption of corporate social responsibility (CSR) and TWs by the industries presents multiple benefits such as increasing cashflow, enhancing their CSR performance and reputation. Thus, the increasing adoption of TWs in the different industrial sectors is also derived from ethical motivations to further contribute to environmental protection (i.e., to maintain a sustainable natural environment for future generations), and is not merely an essential tool to improve the financial position of industry.

Processes required and TW type to be used

Considering the above-mentioned issues, i.e., complexity of compositions, variety of origins, etc., there is no "rule of thumb" in the design of a wetland system for an industrial effluent. Each case is usually considered as unique, especially if there is no previous experience on a particular industrial effluent. A common practice is to first design and test pilot wetland beds and evaluate their performance, before the implementation of a full-scale wetland facility. This allows for a step-by-step approach to identify an effective design, optimize the treatment efficiency and minimize any financial and technical risks.

Practically all main TW types (i.e., FWS wetlands, HF wetlands, VF wetlands) have been tested and applied for industrial wastewaters (Stefanakis *et al.*, 2014; Wu *et al.*, 2015). Usually, hybrid systems are preferred in order to exploit a wider range of the required processes, depending on the nature of the pollutants present in each specific industrial effluent.

Specific considerations during design and for construction

The general considerations and/or requirements for the design of TW facilities for industrial wastewater treatment can be summarized as follows:

- Detailed information about the industrial process, raw materials and any chemicals used
- Detailed and full characterization of wastewater quality and composition
- Often a combination of aerobic/anaerobic processes is needed, i.e., transition areas from surface to subsurface wetland systems need careful design and construction
- *Heavy metals*. Their presence can affect the system performance; external carbon or an organic substrate may be required
- *Plant health*. Crucial for system efficiency; high loads or high salinity may restrain their growth; salt-tolerant species should be considered in this case
- *Clogging*: A common problem in such applications; usually a pre-treatment stage is required before the TW stage to limit clogging potential
- Higher loads and higher flows, which correspond to higher land area demands.
- Specific health and safety measures may be required if works are carried out within industrial areas and facilities

- A more frequent monitoring program may be required for the treated effluent
- Disposal/discharge strategy of the treated effluent should be considered in advance
- Limited access to the system is often required by industries – fencing may be needed

Examples of specific industrial wastewater applicatons

After this genral introduction to treatment wetland use for instustrial wastewater, the following chapters provide more details on the following applications: mine drainage, hydrocarbons removal, as well as citrus, winery and dairy wastewater.

4.11.2 Mine Drainage

Vit Rous

Faculty of Environmental Sciences, Czech University of Life Sciences Prague, Kamýcká 129, 165 21 Praha 6, Czech Republic

Design objectives

The key objective when using TWs treating mine drainage is to make water suitable for release into the environment. The main design objective for treatment wetlands in mine drainage remediation is the removal of (heavy) metals and sulphate, increasing alkalinity and pH so the water can be safely released to the environment.

Processes required and TW type to be used

The abiotic and pure physical and chemical processes are more important in the treatment of mine drainage than in more common uses of treatment wetlands (Table 4.8). Although most of the processes can occur abiotically some of them can be greatly enhanced by biotic structures in the wetlands (for example manganese oxidation catalyzed by manganese-oxidizing bacteria, bacterial sulphate reduction or physical filtration of suspended solids by plant roots).

Table 4.8 Design objectives for improving water quality of TWs treating mine drainage and required processes.

Design Objective for Improving Water Quality	Processes
Removal of metals	Abiotic and biotic oxidation and hydrolysis Metal reduction (metal sulphide formation) Precipitation Filtration Sedimentation Adsorption Plant uptake
Removal of sulphate	Bacterial sulphate reduction
Neutralize acidity	Limestone (calcite) dissolution Reductive precipitation of iron and sulphur

For mine drainage treatment, only FWS wetlands are being used from among the main types of wetlands as defined in this publication. HF and VF wetlands (both operated under saturated water flow conditions) are also used but mainly with special media such as compost, mulch and limestone to promote an anaerobic environment and to increase pH and alkalinity. These types of wetlands are often called successive alkalinity producing systems (SAPS).

Other components used in conjunction with TWs are sedimentation basins (deep ponds for settling precipitates), open or closed limestone channels for managing pH and alkalinity of the water, and aeration cascades for passive water oxidation (Ford, 2003; PIRAMID Consortium, 2003; Watzlaf *et al*., 2004).

Specific considerations during design and for construction

As mine drainage water has a wide range of chemical composition, there are only some basic rules of thumb for the design of these systems (PIRAMID Consortium, 2003; Sheridan *et al*., 2018). The designer should always know the chemical composition of the drainage water and the geochemical composition of the site.

Design assumptions specifically taking into account malfunctioning are:

- *O&M.* Overall non-adequate maintenance due to the basic misunderstanding that passive nature-like systems do not need any maintenance. Special attention must be paid to the amount of the sludge in the system (precipitates) because it can lead to clogging and short-circuiting. When any special media is used (limestone, organic substrate) there should be the possibility to easily replace them after the depletion.
- *Construction phase.* Proper lining and proper hydraulic parameters of media should always be checked.
- *Decommissioning of the TW system.* The precipitates (sludge) in the system can contain high quantities of heavy metals and radioactive compounds which can represent a hazard to the environment and must be appropriately disposed.

4.11.3 Hydrocarbons removal

Alexandros Stefanakis

Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

Introduction

Hydrocarbons are commonly found water contaminants with a large variety of compounds with different chemical and physical properties. They can be classified into three main categories; aromatic, aliphatic and alicyclic. Total petroleum hydrocarbons refer to compounds derived from petroleum sources and processing, e.g., diesel, petrol, kerosene and lubricating oils. Lighter hydrocarbon compounds (i.e., with less than 16 carbon atoms) include substances with higher solubility and volatility, e.g., benzene. Other substances (e.g., MTBE and alcohols) are highly soluble, while some (e.g., benzene, toluene, ethylbenzene, and xylenes) are soluble (Thullner *et al*., 2018).

Design objectives

Hydrocarbon contamination usually occurs in industrial areas, such as chemical-petrochemical industry, oil production and refineries, electricity generation plants, manufacture industry, plastics and steel production and water cooling plants, and is a common problem for groundwater or surface water quality in many regions around the world. Due to the importance and related risks of these compounds, the treatment of waters containing hydrocarbons is necessary. The goal of TW design is to effectively remove these compounds from water and reduce their load. Considering that common mechanical/chemical treatment technologies have high construction and operation costs, the use of wetland technology is viewed as an effective eco-tech treatment method with reduced construction costs, significantly reduced operation and maintenance costs and with multiple environmental, economic, and social benefits (Stefanakis *et al*., 2018; Thullner *et al*., 2018). This is the main driver for the oil and gas – petrochemical – chemical industries to invest in TW facilities.

Processes required and TW type to be used

All TW types have been tested for hydrocarbons-contaminated wastewater (Stefanakis & Thullner, 2016; Stefanakis *et al*., 2018; Thullner *et al*., 2018). The majority of the systems is subsurface systems with horizontal or vertical flow, with very good removal rates reported for compounds such as benzene, MTBE, phenols, and oil content. The main removal mechanism is biodegradation, with VF wetlands appearing as the preferred design due to their aerobic conditions. However, HF wetlands have also been proved successful, even when a variety of compounds is present in the water (Stefanakis *et al*., 2016). The FWS wetlands type is mostly applied for produced water treatment, i.e., a by-product produced during the exploration and production of oil and gas that is contaminated with residual hydrocarbons, salts, heavy metals, chemical additives and other organic and inorganic compounds (Ji *et al*., 2007; Stefanakis *et al*., 2018).

Specific considerations during design and for construction

Water contaminated with hydrocarbons is difficult to deal with, hence the selection of the proper TW type is crucial. First, good information is required on the source of the contaminated water, e.g., industrial facility, applied processes, raw materials and chemical additives used. It is important to identify the exact location in the industrial process line from which the water will be pumped and treated. A detailed characterization of the water quality and composition is also required. For this, the taking of more than one daily composite sample for chemical analyses is needed. This data will show the nature of the pollutants present in the water and their loads in order to select the appropriate wetland design, for example, if specific pollutants require aerobic or anaerobic conditions. The nature of hydrocarbons, i.e., dissolved or emulsified, also needs to be determined, as well as the presence of light and heavy oil fractions, since in some cases a pre-treatment stage may be necessary. If the treatment wetland is to be established in hot and arid climates (where the majority of produced water from oil and gas exploration occurs), then specific consideration should be taken to select plants with high productivity and high water use efficiency (to reduce evapotranspiration losses), to estimate the water losses through evapotranspiration and the area required to reach the treatment targets. Moreover, in cases where large daily volumes are to be treated, the design of the TW needs to consider in advance the available options for the disposal/reuse of the treated effluent.

4.11.4 Citrus wastewater

Alessia Marzo and Mirco Milani

Department of Agriculture, Food and Environment (Di3A), University of Catania, Via Santa Sofia 100, Catania 95123, Italy

Design objectives

The main design objectives of citrus wastewater treatment is to reduce the TSS, organic matter and essential oil concentrations. Citrus processing wastewater (water for fruit, plants, devices and floors washing, cooling, essential oil extraction and peel drying) is characterized by (Koppar & Pullammanappallil, 2013; Zema *et al*., 2012):

- Seasonal quantitative and qualitative variability;
- Low pH (generally <5);
- High organic matter (COD ranging from about 60–170,000 mg/L);
- High TSS (up to 70,000 mg/L);
- Lack of nutrients (nitrogen and phosphorus);
- High essential oil content (up to 600 mg/L).

Processes required and TW type to be used

Citrus wastewater is usually treated in intensive biological plants, mainly represented by activated sludge systems, which can suffer due to the lack of nutrients and presence of inhibiting compounds (essential oils, polyphenols, etc.). Treatment with a combination of aerobic–anaerobic aerated lagoons and multi-stage wetlands has proved to be a valid alternative to conventional plants thanks to their higher reliability and lower energy requirements.

In aerated lagoons, citrus wastewater is usually stored in large and deep basins with storage capacities of about 50% of the annual volume of produced wastewater and hydraulic retention times longer than 3–6 weeks. Processes in the lagooning treatment include (Andiloro *et al*., 2013):

- An equalization of quali-quantitative wastewater characteristics;
- A progressive increase of pH due to degradation of organic acids;
- A strong reduction of settleable and suspended solids due to flocculation and sedimentation processes; and
- A reduction of essential oils (EOs) concentration by the dilution effect within the lagoon and the biological degradation.

The treatment of lagoon effluent using a multi-stage wetland (HF–VF–FWS) is necessary to reduce the organic and TSS concentrations with filtration, sedimentation, mineralization and anaerobic degradation processes.

Specific considerations during design and for construction

- *Malfunctioning prevention.* High EO concentrations could inhibit biological processes. For this reason, it is advisable to treat wastewater with high EO concentrations in a separated lagoon to further improve efficiency and reliability through the whole cycle.
- *O&M.* Fertilizer may be applied on wetland surface area to promote macrophyte growth after planting.
- *Monitoring.* It is advisable to perform monitoring of pH values in the lagoon systems to evaluate a possible correction of low pH by lime addition or similar alkaline chemicals.

4.11.5 Winery wastewater

Alessia Marzo and Mirco Milani

Department of Agriculture, Food and Environment (Di3A), University of Catania, Via Santa Sofia 100, Catania 95123, Italy

Design objectives

The design objectives of winery wastewater treatment is usually based on the need to reduce the main pollutants, represented by the organic matter and solids, to limit the environmental pollution.

Wastewaters generated from wine production are characterized by: (1) large volumes (1.6–2.0 L of wastewater per litre of wine produced) and seasonal variability; (2) high concentrations of organic matter, with COD that varies from 340 to 49,103 mg/L and BOD_5 about 0.4–0.9 of the COD value; (3) variable amounts of TSS that range from 190 to 18,000 mg/L. The highest concentrations of organic matter and TSS are produced with the generation of the highest wastewater volumes (vintage and racking).

Processes required and TW type to be used

An equalization tank may be placed upstream of the treatment plant to reduce the qualitative and quantitative variability of wastewater.

The TSS and organic matter can be mainly removed by processes of filtration, sedimentation, mineralization and anaerobic degradation typical of subsurface-flow wetland systems.

Generally, in small wineries (<2,000 hL wine/year) the treatment plant consists of a septic or Imhoff tank, also with equalization function, followed by a single stage of HF or VF wetland. For medium-size and larger wineries different solutions are adopted (e.g., Masi *et al.*, 2015a): (1) multi-stage wetland (VF–HF–FWS; French VF–HF–FWS); (2) conventional technology combined with a TW (Upflow Anaerobic Sludge Blanket or Hydrolytic Upflow Sludge Blanket–VF–HF; Sequential Batch Reactor or Activated Sludge–French VF or VF).

Specific considerations during design and for construction

- *Malfunctioning prevention.* The feeding of HF wetland with high solids loading rates or with winery wastewater that has been poorly pre-treated leads to clogging phenomena and a reduction in performance in a short time. HF substrate clogging was observed with organic loading rates of about 500 g $COD/m^2/d$ (related to the surface area of the HF wetland).
- *O&M.* Low nutrient concentrations in raw winery wastewater can determine the need to use fertilizers to promote macrophyte growth in TWs. Fertilizer may be applied in the raw wastewater or on the wetland surface area after planting and at the beginning of each growing season.
- *Monitoring.* During the vintage period, it is advisable to monitor pH values in the raw winery wastewater to evaluate a possible correction of low pH by lime addition or similar alkaline chemicals.

4.11.6 Dairy wastewater

Anacleto Rizzo and Fabio Masi

Iridra Srl, via La Marmora 51, 50121, Florence, Italy

Design objectives

Dairy wastewater is usually produced by the cleaning and sterilization of the milking equipment and by the wash-down of the manure-spattered walls and floors of the milking parlour. These activities lead to the production of dairy wastewater characterized by high organic matter concentrations and wide fluctuations of pH. The organic compounds present in the wastewater are mainly carbohydrates, proteins and fats originating from the milk. A wide range of pH values (between 3.5 and 11) is encountered in the literature, due to use of both alkaline and acidic cleaners and sanitizers. The seasonality of typical dairy activities and the different products produced (milk, butter, yoghurt, ice cream, and cheese) lead to a wide range of dairy wastewater quality in the literature (BOD_5 1400–50,000 mg/L; COD 2000–90,000 mg/L; $N\text{-}NH_4^+$ 20–150 mg/L). On the other hand, dairy wastewater production is usually relatively low and the investment required for treating it is consequently has minimal impact on the business model; this therefore allows the design of CW systems with high HRTs, which are proven to provide optimal removal of high organic content wastewaters even with high fluctuations in their concentrations throughout the year. Treatment systems with a high retention time can also play a very favourable role in dealing with another relevant issue linked to dairy wastewater, which is the industrial production rhythm, including short and long pauses in producing effluents as most of weekends and seasonal holidays. The high volumes of wastewater that can be retained from the extensive treatment system are therefore properly buffering the variations in loads both from the quantitative and qualitative point of view.

Processes required and TW types to be used

Different primary treatment approaches have been adopted to remove suspended solids, greases and oils and eventually adjust the pH prior to treating dairy wastewater in wetland stages, including lagoons, three-chambered or Imhoff septic tanks, degreasers and settling basins or tanks. Wastewater from the milking parlour is strongly advised to be pretreated by a high volume degreaser (HRT $>$ 5 d); if built in concrete, the degreasers should be lined with HDPE or epoxylic coating liners to prevent the concrete being dissolved by lactic acids resulting from the biodegradation of milk.

Main pollutants are removed with typical processes of surface and subsurface TWs: TSS and organic matter by processes of filtration, absorption, sedimentation, mineralization and anaerobic degradation; nitrogen compounds by nitrification and denitrification biological processes as well as plant uptake and gas exchange; phosphorous through adsorption and sorption mechanisms as well as plant uptake and precipitation of insoluble salts.

Dairy wastewaters were successfully treated with different TW types, such as FWS, HF as well as VF wetlands, and hybrid schemes as well as intensified aerated wetlands. Subsurface-flow systems seem to be preferable in terms of removal efficiencies in comparison with free surface solutions.

Denitrification can be boosted adopting the recirculation of effluent towards either primary treatment or influent as well as by a tertiary pond or a FWS final stage. Limestone can be used as amendment to stabilize the pH and precipitate phosphorous, whose removal can also be enhanced by adding iron salts. Dutch experiences have shown a higher P removal using white limestone gravel “Jura marble” instead of broken seashells and grey limestone. Another option to remove TP is to add a post-treatment with high-adsorbing capacity material, such as apatite or to make use of a struvite reactor which precipitates struvite ($MgNH_4PO_4.6H_2O$) by adding Mg soluble salts. Particular caution has to be taken when considering the high organic loads, surely leading to fast clogging of the different subsurface-flow TWs if the issue is not considered properly.

Specific considerations during design and for construction

- *Considering of malfunctioning*. The use of a peat layer as carbon-source for denitrification is discouraged, since VF wetlands located in The Netherlands faced clogging issues in case of peat usage. Also the use of steel slag to improve TP removal is not suggested, since some experiences have shown a clogging tendency due to $CaCO_3$ formation; furthermore, slug ashes could release heavy metals at high pH and therefore the adverse effects linked to their usage could be higher than the positive ones.

  Attention should be paid to eventual extreme pH conditions, which could prevent a proper biofilm formation; for instance, the removal of highly acidic serum from the wastewater to be treated by CWs has shown to provide influent with more suitable pH values for wetland biological processes.
- *O&M*. The maintenance of a pH value between 5.5 and 8.5 is mandatory for a proper biofilm development and, consequently, the successful treatment of dairy wastewater. Data reported in the literature show a wide range of pH values for dairy wastewater, which can be highly acidic or highly basic (reported pH values ranging from as low as 3.5 to as high as 11). For this reason, a preliminary design of solutions aimed to optimize the pH is not possible without first analyzing the wastewater to be treated. Hence, it is important to consider some possible ways of managing pH during the design phase of the industrial cycle itself (for instance, the possibility of segregating the serum).
- *Monitoring*. A careful analysis of treatment performance during the start-up phase is always advisable, especially in terms of pH monitoring. In this way, prompt options can be adopted to neutralize the pH (e.g., serum segregation) and to guarantee a proper functioning of the CW for dairy wastewater treatment.
- *Construction phase*. Construction phase is similar to that of a CW for domestic or municipal wastewater. Many plants have been tested in CW for dairy wastewater treatment; among them, *Phragmites australis*, *Scirpus sylvaticus*, and *Urtica dioica* have shown to be able to grow and to not exhibit any toxicity effect from contact with dairy wastewater.
- *Decommissioning of the system*. The requirements for decommissioning of the TW system are similar to those adopted for domestic wastewater.

4.12 LARGE-SCALE WETLANDS

Alexandros Stefanakis
Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

4.12.1 Introduction

Large-scale TWs are considered as distinct applications of wetland technology due to their size. The term "large-scale" refers to wetland sizes much higher than the average wetland system, which is wetland beds with surface area starting from a few hectares up to a few thousands of hectares. Such facilities are built to deal with large flows, hence the higher area demand. As is easily understood, large-scale wetlands can be constructed only in areas where there is available land, e.g., in rural and/or remote areas, in the desert, etc. Since the main limitation of wetland technology is in any case the higher area demand compared with traditional/conventional treatment methods, the number of large-scale wetlands is small. The fact that the construction and operation/maintenance costs increase with increasing wetland size also contributes to the small number of large-scale wetlands, when compared with the several thousands of wetland plants operating around the world. However, some of these large facilities are unique and are even considered as milestones for wetland technology, demonstrating its treatment capacity and the scaling-up possibilities.

4.12.2 Design objectives

The main goal of large-scale wetlands is, as for all wetland plants, water quality improvement. The large size of such wetland systems allows for the receiving and treatment/polishing of high volumes up to hundreds of thousands of m^3 per day. Large-scale wetlands have been designed for the following main applications:

- The majority of large wetland systems (with a surface area of 40–2,600 hectares) receive stormwater and urban runoff, and function to control floods and to remove excess phosphorus from agricultural drainage (Dunne *et al*., 2012; Kadlec, 2016; Pietro & Ivanoff, 2015; Sim *et al*., 2008).
- Other systems (with a surface area of up to 900 hectares) have been designed as tertiary treatment stages, receiving and polishing secondary effluents from domestic/municipal and/or industrial wastewater treatment plants (Kadlec, 2016; Kadlec *et al*., 2010; Wu *et al*., 2017).
- Eutrophicated river or lake water treatment to remove nutrients (e.g., nitrogen, phosphorus) and improve the water quality of the final receiving water body is also another common application (Dunne *et al*., 2013).
- A TW with 2,400 hectares has also been designed to remove nitrate from the municipal drinking water supply in southern California, USA in order to protect human health and to reduce eutrophication and algal clogging in deep groundwater recharge ponds (Reilly *et al*., 2000).
- A large wetland system (360 hectares) has been designed to treat produced water contaminated with oil hydrocarbons from an oil field under desert climatic conditions (Stefanakis *et al*., 2018).
- A few applications also exist for wetland systems in secondary treatment of municipal wastewater, serving populations from 3,000 (Morvannou *et al*., 2015) up to 20,000 p.e. (Masi *et al*., 2017b). These figures are considered unusual for TWs for secondary treatment of municipal wastewater, since TWs are generally viewed as best choice for small and medium communities, but are indicative of the potential to design wetland systems even for thousands of inhabitants.

Large-scale wetlands also provide a series of additional ecosystem services, which are usually integrated in their function and operation. TWs with a surface area of several hectares are in practice a new habitat for wildlife that attracts birds, fish and reptiles. For example, it is reported that a large wetland system built in the desert of Oman is used by thousands of birds during their migration as a stopover to rest and feed (Stefanakis *et al*., 2018). The same is also observed in treatment wetlands in Florida, USA (Kadlec, 2016). Moreover, many of these systems are designed as polycultures, i.e., they are planted with more than one plant species, promoting in this way vegetation biodiversity. Additionally, considering that these systems are large vegetated areas they are also designed to provide an aesthetical upgrade of the site, while many systems are used for recreational and educational purposes.

4.12.3 Processes required and TW type to be used

Due to their large size and the associated high costs, the most frequent wetland type used for large-scale applications are FWS wetlands and only a few case studies of large-scale subsurface-flow wetlands exist (e.g., Masi *et al*., 2017b). FWS wetlands are simpler and easier (and, thus, cheaper) to build, compared to subsurface-flow systems filled with gravel media. The FWS type is widely used for stormwater and runoff treatment, to improve urban water quality and to polish effluents from wastewater treatment plants. The main target in these applications is nutrient (i.e., nitrogen and phosphorus) removal, hence biological processes are mostly required (such as microbial degradation), as well as physical/chemical processes (e.g., sedimentation, filtration, adsorption) and plant uptake/assimilation. Solids removal can also be a target (filtration). FWS wetlands are also used for produced water treatment at oilfields. In this case, oil hydrocarbons are the target pollutants and their removal mainly occurs through bacteria biodegradation.

4.12.4 Specific considerations during design and for construction

The design and construction of large-scale wetlands obviously includes a larger variety of technical and economic challenges, in order to successfully develop such a wetland project. The main issues that should be taken into account are as follow:

- Land availability is crucial for the financial sustainability of a large-scale wetland project. An area with relatively cheap (or even free) and adequate land should be selected for the wetland siting.
- There is an economy of scale for large-scale wetlands that for large-scale FWS wetlands reduces the cost per hectare in comparison to smaller systems. Use of large pumps to send water to a large wetland should, however, be avoided as it will offset this benefit.
- For large-scale FWS wetlands, installation of plastic impermeable liner is usually avoided due to cost implications. Natural minerals (e.g., clay) are often used to construct a sealing layer, but this is not always technically and financially feasible for large wetlands systems.
- Water flow path and depth variations may occur over time, owing to flow resistance by vegetation roots and stems, which could render it difficult to control water depth and could risk the stability of the embankments.
- Planting and establishing plants in large wetlands is an expensive task due the large number of seedlings and labour required and the potential initial need for nutrients supply.
- Maintaining a healthy vegetation cover can be a challenge; usually large wetlands are polyculture systems (i.e., with many different plant species) presenting changes with time. Although implemented in some cases, plant harvesting is usually unfeasible, as it can be expensive and technically challenging.
- Some large wetland systems (e.g., for stormwater treatment) may have seasons with no water inflow, which can result in complete dry-out and the subsequent risk of releasing pollutants stored in the

organic sediments of the bed. In such cases, the design should make provision to keep the wetland system saturated to prevent the drying of the vegetation.

- Short-circuiting, preferential flow, stagnant water or dead zones without vegetation within the wetland bed could all affect the transformation/removal processes and, thus, the treatment efficiency, as well as creating nuisance issues (mosquito breeding, odour). Vegetation management to maintain plant coverage and tracer tests to identify flow paths are often necessary.
- Longer start-up periods may be required for large wetlands.
- Multiple wetland cells, which can be isolated from the water flow, provide flexibility during the operation and maintenance period.

4.13 RIVER REHABILITATION AND RESTORATION

Tjaša Griessler Bulc[1], Darja Istenič[1] and Aleksandra Krivograd Klemenčič[2]

[1]Faculty of Health Sciences, University of Ljubljana, Zdravstvena pot 5, 1000 Ljubljana, Slovenia
[2]Faculty of Civil and Geodetic Engineering, University of Ljubljana, Hajdrihova 28, 1000 Ljubljana, Slovenia

4.13.1 Design objectives

Numerous watercourses and their surroundings have been changed due to the needs of agriculture, infrastructure, urbanization, flood protection and energy production. The lines of watercourses were straightened, the waterbeds were lowered, floodplains were dried out and bank vegetation was removed to speed up the water drainage from the area. These measures resulted in droughts in the upstream and floods in the downstream areas, degradation of habitats in watercourses and severe reduction in self-treatment capacity and biodiversity. The water quantity and quality were significantly altered.

In recent decades water quality in Europe has gradually improved due to wastewater treatment. Consequently, rivers and lakes have become increasingly important also in the cities through the planning of urban ecology, green infrastructure, green areas and climate change adaptation (EEA, 2016); therefore, restoration and rehabilitation techniques of waterbodies are getting increased attention.

Multifunctional solutions

Watercourse rehabilitation means to restore ecological equilibrium in the watercourse ecosystem, which increases self-treatment capacity and biodiversity and enables additional ecosystem functions. As a result, the watercourse gains higher ecological, environmental and social value.

River rehabilitation measures aim at habitat enhancement and reconnection of the watercourse with the floodplain, increasing the potential for natural water storage within the system, and thereby reduce the height of the flood peak (flood prevention) and extend the period of base flow within the channel (water retention). These measures also aim at removing the obstacles within the watercourse where possible to provide free movement of wildlife and gravel within the water ecosystem. Uninterrupted transport of gravel is important for maintenance of habitats and treatment processes. By habitat enhancement and increasing biodiversity, the self-cleaning capacity is increased, and potential water pollution is mitigated. The water that is retained in the restored watercourse can be used for different purposes such as irrigation in agriculture, groundwater recharge, or energy production in hydropower plants, thus contributing to an improved water management. Multifunctional benefits of river restoration reach social fields as well with establishment of recreational and educational possibilities.

Multifunctional river restoration measures can be of different dimensions according to available space and budget. In dense urban areas and intensive agricultural land restoration measures often take place inside existing water bodies; however, for better results interventions outside the watercourse is needed.

4.13.2 Processes required and TW type to be used

The measures of river rehabilitation are based on aquatic wetland as well as terrestrial ecosystems' characteristics, and should consider water management in a watershed, including flood prevention, water retention, biodiversity and specific physical, chemical and biological processes for reduction of pollutants. River restoration measures usually combine different design elements, of which some have its origin in TWs or technical river restoration measures. In all measures along with hydraulic, physical, chemical and microbiological processes, phytoremediation plays an important role (Griessler Bulc *et al.*, 2012). The implementation of different restoration measures significantly diversifies the watercourse. Diverse riverbed increases the number of microhabitats and thereby enhances the biodiversity and stability of the ecosystem (Wetzel, 2001). It provides better water aeration, retention of fine particles, aerobic and anaerobic processes, and higher nutrient intake by macrophytes, algae, and microorganisms (Griessler Bulc *et al.*, 2011, 2015). The most common measures are:

- *Anabranching*. Anabranching means diverting a part of watercourse in a separated channel which re-joins the main channel downstream. There can also be multiple channels, all separated by vegetated islands. The anabranch must be designed according to the characteristics of a natural watercourse in the corresponding area. At the beginning or at the end of an anabranch a gravel bed mimicking TW can be integrated which enables water filtration, growth of macrophytes and acts as media for development of microorganisms enabling treatment processes. Anabranching significantly increases water retention and enables flood mitigation, creates new habitats for wetland and aquatic plants, amphibians, birds and invertebrates.
- *TW and vegetated drainage ditches (VDDs)*. Relative to the location of a watercourse, TW and VDD can be positioned in-stream or off-stream. In a case of off-stream positioning, only part of the water is diverted and treated in a TW or VDD, while in a case of an in-stream system, all the water flows through and therefore they have to be levelled with the mean flow of a watercourse (Kadlec & Wallace, 2009; Kasak *et al.*, 2018). TW can be established as HF or FWS wetlands and can include inflow distribution pipes and an outflow pipe. VDDs are simple structures that usually do not include special piping systems as in the case of TW. To enable efficient filtration, the TW and VDD should consist of bigger fractions of gravel (>8 mm). Besides high treatment efficiency TW and VDD provide additional habitats for wildlife, act as a water reservoir during draughts and smaller water retention system during floods. Appropriate locations for their positioning are small tributaries or inflows of stormwater, melioration ditches etc. On such locations TW and VDD significantly contribute to the reduction of pollutant inflow from urban and agricultural areas into the watercourse.
- *Meanders*. Meanders lengthen the path of water flow, reduce the inclination, slow down the water flow, and increase the depth of water and the amount of water in the area and groundwater. Consequently, increased residence time enables better water treatment. With the meanders also the riparian area of the watercourse is lengthened as well as the hyporreic zone increased (Boano *et al.*, 2014). Riparian areas have high biodiversity; moreover, the contact between water and soil acts like a sponge, enabling water retention in the area which has multiple benefits (increasing low flows in summer, drought mitigation, groundwater recharge).
- *Pools*. Pools can be designed as self-sustaining systems by excavating sediment and placement of boulder arrangements to promote sediment scouring and maintain a self-sustaining mid-channel pool. Pool spacing would be based on the gradient and width of the channel using basic geomorphological principles.
- *Riffles*. Riffles consist of gravel and boulders that should not oversize the mean water level. On site of a riffle, the riverbed is narrowed, water flow concentrates and speeds up, and the water is mixed and aerated. Downflow a small pond is created. Riffles are also habitat for numerous invertebrates and a site for fish spawning.
- *Backwaters*. Backwaters are dead-end river branches with no or very little current. They enable water retention and a shelter for fish during high flows. A diverse wetland vegetation usually occurs in and around them. At the end of a backwater bay gravel beds or shallow water and low banks can be created which enables an easy access to water for animals and humans.

- *Gravel bed*. Gravel beds increase the self-treatment capacity of a watercourse and act similarly to a gravel filter. They can be installed at one or other bank or in the middle of a riverbed. The gravel bed should be higher than the main water level.
- *Reconnection with floodplain*. By lowering the berms of a watercourse, the frequency of flooding the surrounding areas is increased. Reconnection with the floodplain is important for increasing water storage capacity during higher flows and creating valuable semi-aquatic habitat. It can be done within meander bends to create smaller areas for flooding.
- *Water reservoirs*. Water reservoirs can be created in a floodplain as a deepening that enables retention of flood water for a longer period. Water reservoirs provide good water pollution mitigation as they enable retention of suspended and settleable solids; they provide groundwater recharge and create new habitat.
- *Measures for education and recreation*. With appropriate measures taken, the restored watercourse can also become an interesting educational site. Educational paths can be established including bird observation points, observation of self-cleaning elements of wetland and river, info-boards, leaflets, and apps can be prepared with educational contents. For recreational activities walking/running and biking trails can be provided, playgrounds for children, etc. (Griessler Bulc *et al.*, 2012, 2015).

4.13.3 Specific considerations during design and for construction

There are several critical aspects to be considered when planning restoration activities such as local planning, pollution prevention, flood risk management and climate change adaptation; however there are limitations to river restoration that include a lack of scientific knowledge of watershed-scale process dynamics, institutional structures that are poorly suited to large-scale adaptive management, and a lack of political support to re-establish the ecosystem amenities lost through river degradation (Wohl *et al.*, 2005). Existing river management practices should be improved by integrating ecosystems services and participatory approaches to enable decision makers and river managers to select and apply strategic planning approaches according to their needs. Where the term restoration is used, it is also important to aim for multiple benefits for different sectors helping to deliver synergies by implementing different policies, especially regarding ecosystem services (EEA, 2016).

Design components of the system, such as meander wavelength, riffle/pool spacing, sediment distribution, channel dimensions and sinuosity should be based on basic geomorphological/hydrological principles as well as studies of nearby meandering reference reaches/streams with similar boundary conditions, e.g., channel slope/dimensions.

There are some key challenges that should be considered in the design and construction of restoration measures in order to prevent problems during the operation of the system. These briefly are:

- Clogging of TW, VDD, gravel beds and similar filtration elements can occur due to high waters and the torrential nature of the watercourses, also causing damage to the plants. To minimise clogging, a barrier prior to sensitive structures and drainage pipes can be installed, including an adjustable barrier to control the flow of water into the system and thus protecting it against the intrusion of torrential waters. However, small deposits of silt are expected in the first TW/VDD segment.
- High waters can cause collapsing and sliding of river banks during periods of heavy rainfall. The reinforcement and successful overgrowth of banks with marsh plants are needed to avoid bank erosion. The velocity of water flow through meanders should remain below a critical velocity of 0.7 m/s to avoid erosion, alluvial deposit and plant and biofilm damage. Preferably, the velocity of water flow should remain close to 0.3 m/s.

- Stagnation of water, low water level, poor vegetative cover of banks and warming of water due to the exposure to solar radiation can enhance algae development. Sufficient shading by appropriate plants and higher water flow velocity can successfully reduce algae growth and the warming of water. Moreover, plants act as a buffer zone and enhance the self-cleaning capability of the watercourse.
- Knowledge on climate conditions, e.g., rainfall, high waters (10-year and 100-year flood events), water flow velocity, solar radiation etc., are required prior to the design of the system.
- Plants: selection of various native plant species is preferred to increase biodiversity and enable sufficient shading effect to reduce algae development and to enhance pollution reduction.
- Regular monitoring of pollution mitigation and regular maintenance is needed to avoid malfunction of river restoration elements.
- Space limitation: where available space is limited, river restoration can be possible by removing redundant structures and buildings to gain space for restoration activities.

4.14 SALINE TWs

Lei Yang

Department of Marine Environment and Engineering, National Sun Yat-sen University, Kaohsiung, Taiwan

4.14.1 Definition

Based on the salinity of water, natural wetland systems are divided into freshwater, brackish water, and saltwater types. Like natural wetlands, TWs can also be divided into those categories based on the salinity of wastewater treated. Brackish and saltwater types can be referred to as saltwater or saline TWs.

Saltwater TWs treat wastewater with salinity similar to seawater, i.e., >30‰, e.g., mariculture wastewater, seawater flush toilet water, and salt-curing food-processing industrial wastewater (e.g., soy sauce production), while brackish water ones are used to handle the wastewater having more salinity than fresh water, ranging between 5 and 20 ‰. So, brackish water types of saline TWs may result from mixing of seawater with freshwater types of wastewater, e.g., mariculture wastewater and sewage.

4.14.2 Design of saline treatment wetlands

The design of saline TWs is similar to freshwater types of TW. The selection of types includes free water surface (FWS), horizontal flow (HF), vertical flow (VF), and floating wetlands systems. Usually, FWS types are suggested for selecting saline TWs vegetated with salt-resistant woody plant species of mangroves, while both HF and VF saline TWs are of salt marsh type, e.g., *Spartina* sp. That is because FWS types are similar to natural habitats for mangrove swamp wetlands, and the soil type of substrate in FWS types is helpful to support the growth of woody plants of mangroves with deep root systems easily comparing to gravel substrates in subsurface-flow saline TWs. However, the herbaceous types of grass plant species in salt-marsh types of saline TWs can grow in gravel, as generally used in HF and VF types. However, most types of saline TWs designed and operated in Taiwan are FWS ones vegetated mainly with mangroves of different species, including *Kandelia candel*, *Avicennia marina*, *Rhizophora mucronata*, and *Lumnitzera racemosa*, which are the four mangrove species existing in Taiwan. The mangrove species of *Rhizophora mucronata* have been successfully restored in Taiwan and then applied as the vegetation in saline TWs.

The main benefit of aquatic plants applied in saline TWs is to penetrate the substrates, and to transport oxygen to the root zone. All aquatic plant species, including woody and herbaceous, selected in designing saline TWs should be salt resistant. However, generally there is less choice for salt-resistant aquatic plant species used in saline TWs. As mentioned previously, the woody plant species of mangroves are the first choice in designing saline TWs or saltwater types of wetland parks. But the mangrove species can only grow in tropical and subtropical areas, so it is necessary to think about some salt-resistant aquatic plant species other than mangroves that are able to grow in temperate zones of high-altitude areas. Some herbaceous aquatic plant species growing in the coastal and estuarine areas of natural salt marshes might be *Spartina alterniflora*, which is the same family (Gramineae) as *Phragmites sp*.

The process variables for saline TWs include hydraulic loading rates, hydraulic detention time (HRT), water depth in FWS systems, substrate depth in HF and VF and loading rates of pollutants to be removed, such as BOD, SS, N, and P. As with freshwater types of TWs, the selection of those variables for designing saline TWs depends on the performance expectations and design objectives. However, the main difference between freshwater and saline TWs is the salinity (high conductivity) in the systems, which might affect the reaction rates of biological, chemical, and physicochemical processes, such as plant uptake, microbial biodegradation, chemical precipitation, adsorption, and ion exchange. Thus, when the process variables are selected for designing saline TWs, the same processes used in freshwater TWs apply but require some weighting factors, which may be >1 or <1 depending on the salinity presenting either positive or negative effects on the processes. For example, salinity may depress the biodegradation rates for organic removals owing to salinity inhibition of freshwater microbial activities, so the weighting factor for degradation rate constant, *ks*, is <1 for BOD removed. Hence, acclimation for microbes is usually required for saline TWs. Thus, to achieve the same BOD removal efficiency under the same influent flow rate, the volume and HRT for saline TWs are generally larger and longer than those for freshwater TWs, respectively.

For nitrification–denitrification processes, Zhou (2011) found that nitrification was completely inhibited when the salt content was >25 g/L (salinity 25‰). Denitrifiers exhibited a better salt tolerance capability than nitrifiers, with only 49% inhibition present when salt content was increased to 40 g/L (salinity 40‰). However, Jonassen (2013) indicated that nitrifiers could be adapted to high saline environment after adequate acclimation and stepwise increase of salinity. But its weighting factors may be still <1. Thus, when we design saline TWs, it is suggested to stepwise increase salinity for microbial acclimation. In addition, salinity may interfere with physico-chemical processes occurring in TWs, such as phosphorus sorption that decreases with increasing salinity. So, the weighting factor for sorption coefficient of phosphorus is also <1 in designing saline TWs. However, the exact values of weighting factors for different reaction rate constants or process coefficients in different types of saline TWs may be obtained in the tests of microcosm, macrocosm, or pilot systems before designing full-scale saline TWs.

Due to high concentrations of electrolytes in high-salinity systems, there might be some interference for designing microbial fuel cell wetland and modular wetlands. In addition, some special industrial wastewaters containing very high salinity and high organic contents, *e.g.*, salt curing food processing industries in Mainland China and Taiwan, require either intensified saline TWs to treat the high organic and salinity wastewater or some pre-treatment processes for the original wastewater before it is discharged into conventional saline TWs.

4.14.3 Applications of saline treatment wetlands

As mentioned previously, saline TWs can treat salty and brackish wastewaters including aquarium, mariculture industry, and other industries including pharmaceutical, electroplating, printing and dyeing, fermentation, salt curing food processing, and seafood processing industries, etc. Besides, more and more wetland parks are built in coastal, bay, lagoon, and estuarine areas for functions of recreation, ecotourism, environmental education, and water purification for influents from natural seawater and brackish water in those areas. There are some case studies as following.

Dapong Bay is a coastal lagoon located in the southwest of Taiwan with only one entrance exchanging seawater with outer oceanic area. The lagoon is surrounded by many seawater fish (grouper) ponds, into which the mariculture wastewater was discharged. Thus, to prevent pollution to the lagoon, five saline TWs vegetated by mangrove (*Avicennia marina* mainly) around the lagoon were built to capture the saline wastewater for treatment with a total area of 52 ha, receiving a total flow rate of 42630 m^3/d of influents discharged into each of the saline TWs. In addition, some of the saline TWs also functioned as flood detention ponds and wetland parks for ecotourism and environmental education. The treatment units for these five saline TWs systems include sedimentation ponds and gravel filtration beds as pretreatment, FWS types of mangroves and deep ponds. The Dongbay mangrove treatment systems have been operated for 14 years with removal efficiencies for BOD, SS, TN and TP in the ranges 16–68%, 14–76%, 35–82%, and 10–87%, respectively (Yang & Chen, 2012). Since the operation period is over 10 years for these saline TWs, the removal efficiencies of water quality parameters have decreased, especially BOD and TP. It is suggested that the substratum media of saline TWs should be renewed and replaced to improve their treatment efficiencies. The mangroves growing inside the systems are also suggested to be thinned during yearly maintenance to increase the organic removal efficiencies. In addition, it was found that the saline TWs in Dapongbay achieved the carbon budget of -676 g CO_2 eq./m^2 yr revealing a carbon source effect due to N_2O (5.57 g N_2O/m^2 year, or 1,476 g CO_2 eq./m^2) emissions (Yang & Yuan, 2019).

Although there are very few cases of saline TWs vegetated with salt-marsh plant species applied in treating saline wastewaters, some studies have been conducted by using either microcosm or pilot scale of saline TWs vegetated with *Spartina alterniflora*. Sousa *et al.* (2011) used pilot-scale VF wetlands with and without macrophyte *Spartina alterniflora* to study the treatment efficiencies for mariculture effluent. According to their results, the saline TW with and without *S. alterniflora* were found producing reductions of 89 and 71% for inorganic solids, 82 and 96% for organic solids, 51 and 63% for total nitrogen, 82 and 92% for ammoniacal nitrogen, 64 and 59% for orthophosphate, and 81 and 89% for turbidity, respectively (Sousa *et al*., 2011). In addition, Sousa *et al.* (2011) found that the saline TW with *S. alterniflora* showed denitrification tendencies, while the one without *S. alterniflora* had higher oxygen levels leading to nitrification. Such findings agreed with the results of Chang's study (2018), in which microcosm-scale saline TWs vegetated with *S. alterniflora* were used to treat secondary treated effluents of saline sewage. It was found that the average removal efficiency for ammonia was 85% in the saline TW without vegetation, while the average ammonia removal efficiency was 62% for the one with vegetation (Chang, 2018). The results suggest that aerobic conditions are critical for controlling the purification processes as well as the potential for saline TWs vegetated with salt marsh plant species to treat saline wastewaters.

4.15 NATURAL SWIMMING POOLS

Stefan Bruns[1], Claudia Schwarzer[2], Udo Schwarzer[2] and Dirk Esser[3]

[1]Polyplan GmbH, Überseetor 14, D-28217 Bremen, Germany
[2]Bio Piscinas Lda., Ap. 1020, P-8671-909 Aljezur, Portugal
[3]Société d'Ingénierie Nature &Technique (SINT), Chef-Lieu, F-73370 La Chapelle du Mont du Chat, France

4.15.1 Introduction

Natural swimming pools are outdoor swimming pools with biological water treatment. They are separated from natural waters and sealed off from the groundwater. They are divided into bathing and treatment areas and must meet defined water quality requirements, especially in the case of pools open to the public.

In contrast to conventionally operated pools, the water in these baths is not treated by chemical disinfection (chlorination), but by means of biological, physical and physical–chemical processes. The biotechnological processes used to treat the water of these baths make use of the ability of living organisms to convert, degrade or incorporate water-polluting substances.

Natural swimming pools are therefore living systems in which the same processes take place as in natural waters. Technical facilities, such as treatment wetlands, support and control these processes with varying intensity.

TWs used for natural swimming pools work under conditions which are quite different from those in wastewater treatment:

- They usually only work in the vegetative season (which is the bathing season);
- Water is continuously treated in a closed-loop process: the treated water is reused for bathing and not released into the environment;
- The concentrations of organic matter and especially nutrients to be treated are very low (phosphorous is in the microgram and not in the milligram range) and so are the pollutant loads to be treated. Hydraulic loading however is high.

4.15.2 Design objectives

The aim of biological water treatment in natural swimming ponds is to provide bathers with hygienically safe and clear bathing water. Bathing and swimming should be safe and an aesthetic pleasure. The hygienic goals can be achieved on the one hand by a sufficient dilution and on the other hand by an appropriate water treatment. It is also important to achieve a very low trophic status so that the growth of planktonic algae and filamentous algae can be minimized by nutrient limitation, more precisely by limiting the concentration of phosphorous in the bathing water (Table 4.9).

4.15.3 Processes required and TW types to be used

Treatment wetlands for natural swimming pools must therefore primarily eliminate pathogens and reduce phosphorous concentrations. They also need to degrade different kinds of organic matter brought into the bath, and to accept high hydraulic loadings, as the water volume of the bath should be continuously treated in a closed-loop process.

Table 4.9 Trophic status and phosphorus concentration of lake water (adapted from Carlson & Simpson, 1996).

Trophic Class	Total P (μg/l)	Suspended Chlorophyl (μg/l)	Transparency (m)
Oligotrophic	<12	<2.6	>4
Mesotrophic	12–24	2.6–20	2–4
Eutrophic	24–96	20–56	0.5–2
Hypereutrophic	>96	>56	<0.25–0.5

Bathers bring pathogens, phosphorous and organic matter – such as grease from sun-tan products – into the pool, the filling water can be a source of phosphorous, and other external inputs such as leaves, dust, birds, etc. can also bring in pathogens, phosphorous and organic matter.

The German Guidelines for the design of public natural pools have established a "bather equivalent" based on an estimated 120,000 CFU/bather of *E. coli* and 75mg/bather for phosphorous (FLL 2011). As phosphorous and *E. coli* concentration are thought to be the two limiting parameters, these "bather equivalents" are used to dimension the treatment facilities for a specific bath.

For the user of the bath, hygiene is the most important issue, so that pathogen removal should be the main focus. As with conventional pools, the hygienic status of the bath is measured through the concentration of the indicator germs *Escherichia coli*, *Enterococcus* and *Pseudomonas aeruginosa*.

For the limnological system, however, what is relevant is essentially the input of phosphorus compounds or the phosphorus concentration, from which the trophic status of the bath is determined. The combined elements of the water treatment must therefore be able to keep the concentration of phosphorus very low (at 10 μg TP/L) in spite of temporarily high inputs. The same holds true for pathogens.

The water treatment facilities for natural swimming pools can either be based on biological or on physical–chemical processes. The physical–chemical treatment is usually a system that extracts dissolved phosphates from the water (such as a phosphate adsorber). Physical–chemical processes may only be used as a supplement to biological treatment. The water which has undergone such treatment must go through a biological treatment for hygienisation before it enters the bathing area (Figure 4.4).

Biological treatment units for natural swimming pools usually belong to one of the following categories:

(1) Planted Vertical Flow filters
(2) with saturated media
(3) freely drained (with unsaturated media)
(4) Unplanted Vertical Flow filters
(5) with saturated media
(6) freely drained (with unsaturated media)
(7) FWS Wetlands
(8) with submerged vegetation
(9) with emergent vegetation
(10) High-rate gravel or technical filters.

For the selection and combination of different water treatment units, their specific elimination rates related to the monitoring parameters *E. coli* and phosphorus as well as the maximum loading rate per square metre are of importance. The German Guidelines have established elimination and loading rates for the design of public natural pools (Table 4.10).

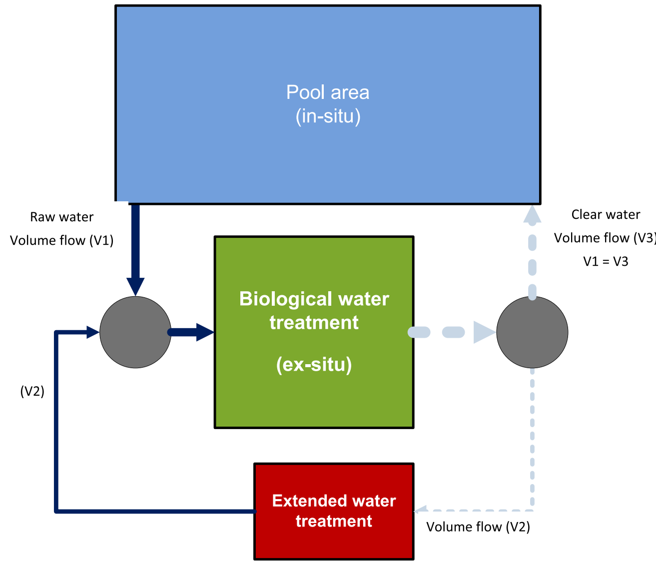

Figure 4.4 Circuit diagram for integrating the physical–chemical water treatment into the biological processes.

These elimination rates and maximum loading are empirical, based on observation from existing facilities. The reason why planted filters should only receive lower loading rates is that the root zone might reduce the volume of the voids in the filter and thus reduce hydraulic conductivity.

High-rate gravel filters or technical filters can have even higher loading rates, but they are not effective for pathogen removal. They do treat organic matter and are used especially for P elimination. Phosphorous is removed with the biological biofilm in the filters, which is often harvested at the end of the bathing season.

Figure 4.5 shows the elimination performance of freely drained vertical-flow filters for *E. coli* under field conditions. It should be noted that the quantification limit for *E. coli* is usually 15 CFU/100 ml. Values below this are given as <15 CFU/100 ml by the laboratories. In the evaluation on which Figure 4.5 is based, the value <15 is set to 15. An elimination of 90%, i.e., by one log level, can therefore only be

Table 4.10 Elimination rates of *E. coli* and phosphorous, and maximum hydraulic loading rates, for different treatment wetlands, according to the German guidelines for public natural swimming pools (adapted from FLL, 2011).

Type of Treatment Unit		Elimination Rate		Max. Loading m^3/day
		Phosphorus	*E. coli*	
Planted vertical flow	Saturated	20%	90%	3
	Freely drained	20%	90%	3
Unplanted vertical flow	Saturated	20%	85%	5
	Freely drained	20%	90%	10
Surface flow	Submerged vegetation	40%	10%	5
	Emerging vegetation	30%	10%	5

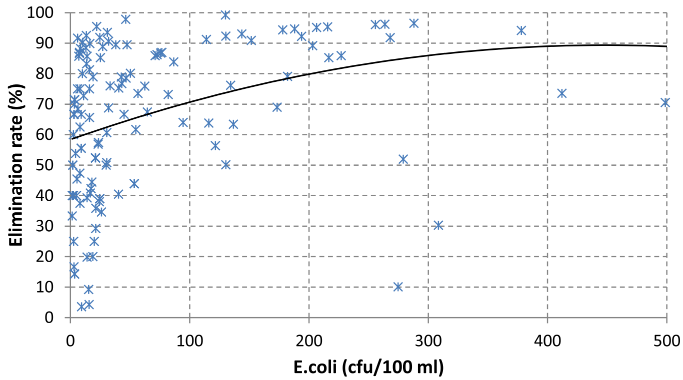

Figure 4.5 Elimination performance of *Escherichia coli* in freely drained vertical-flow filters under field conditions (from monitoring data collected by the DANA database developed by POLYPLAN on public swimming pools from 2005 to 2018.

mathematically proven for inflow concentrations of >150 CFU/100 ml. Since *E.coli* concentration >150 CFU per 100 ml rarely occurs, the evaluation of the monitoring data is of limited use. For this reason, supplementary studies were conducted under standardized laboratory conditions both by the German Federal Environment agency (Grunert *et al.*, 2009) and in the frame of a cooperative research project involving POLYPLAN (Scholz & Frehse, 2004).

Figure 4.6 shows the decrease in the elimination performance of *E. coli* with increasing hydraulic loading of filter columns under laboratory conditions. Elimination rates of 90% (one log level) of the tested unsaturated filter are only achieved for hydraulic loadings below 12 $m^3/m^2/d$ (Scholz & Frehse, 2004).

As far as the elimination of parasitic protozoan pathogens is concerned, the work of Redder *et al.* (2010) proved TWs for wastewater in pilot and field scale to achieve reduction rates of about 2 log for the protozoan pathogens *Cryptosporidium* oocysts and *Giardia* cysts. This is an important advantage for natural treatment systems as especially *Cryptosporidium* oocysts are resistant to chlorination concentrations found in conventional swimming pools (Korich *et al.*, 1990).

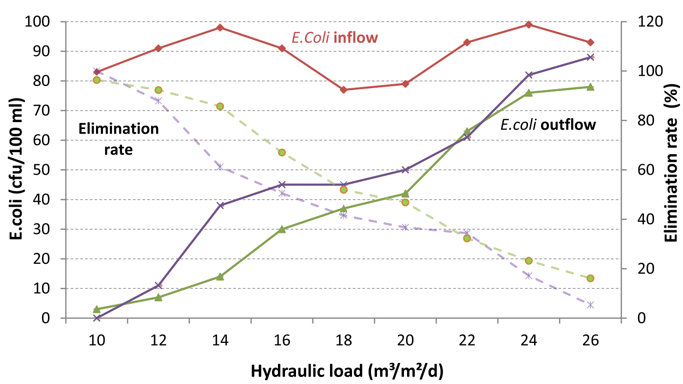

Figure 4.6 Elimination performance of *Escherichia coli* as a function of the hydraulic loading of two unsaturated filter columns (laboratory conditions) (adapted from Scholz & Frehse, 2004).

4.15.4 Specific considerations during design and for construction

As organic carbon concentrations to be treated are very low, so is the oxygen demand in the treatment wetlands. Usually, the dissolved oxygen in the water to be treated is higher than the BOD_5 concentrations, so that even saturated filters can work under aerobic conditions, as long as they are continuously fed with oxygen-rich water. Aquatic plants on the saturated filters further help to maintain oxidizing conditions around their root zones. If required, the nutrients bound in the plants can be finally exported from the bath by harvesting.

Further functions of the plants on the filters are shading and thus cooling the water. They also provide a habitat for a large number of aquatic invertebrates and amphibians. Helophytes with strong root or rhizome growth are often used.

Suitable plants are species of the genera *Carex, Juncus, Schoenoplectus, Bolboschoenus* and *Cyperus*. When choosing a species, it is important to consider whether it is a saturated or an unsaturated filter. Especially for the latter, with intermittent feeding, only very few species can be considered.

Depending on whether or not it is a more technically oriented natural swimming pool, submerged aquatic plants play a different role in *in situ* water purification. In technically oriented baths, submerged macrophytes are usually not used and their function of phosphorous removal is achieved by physical–chemical processes.

In calmer zones planted with submerged macrophytes there is increased sedimentation and thus the elimination or inactivation of nutrients and hygienically questionable bacteria. Furthermore, photosynthesis activity leads to temporarily increased oxygen concentrations in the area or above oxygen saturation.

But the most important role of the plants is their ability of to absorb nutrients such as – and especially – phosphorus. They thus compete with algae (phytoplankton and thread algae), which makes them a stabilizing factor in the ecosystem of a natural swimming pool. Well developed populations of thousand-leaf and pondweed species thus counteract the development of phytoplankton blooms.

Other functions of aquatic plants in natural swimming ponds are shading and cooling zones of relatively shallow water. Shading minimizes the spread of thread algae in shallow water areas, as thread algae compete with aquatic plants not only for nutrients but also for light. Submerged macrophytes form a habitat for many zooplankton species with the space-forming structures of their foliage. Emerged macrophytes provide mechanical protection for the shore areas, which prevents turbidity caused by swirling substrate.

The occurrence of aquatic plants and combinations of different species in their natural habitats are not random phenomena, but rather indicators of very specific living conditions. This basic knowledge of plant sociology is of great importance for the planting of natural swimming pool. In order to ensure good bathing water quality, nutrient-poor conditions should prevail there. Since these conditions can also occur in nature in a very similar way, it is obvious to orientate oneself on naturally formed plant associations.

Plant substrates should be chosen so that the plants can easily root but should not release phosphorous into the water. The substrate mixture and grain size allow a sufficient oxygen supply of the soil. It should be borne in mind that other factors (e.g., insufficient water depth or lack of light) may not allow the aquatic plants to thrive well.

Vascular plants have ecological preferences with regard to important growth factors, which Ellenberg *et al.* (1992) tried to determine with "pointer values" along a new scale. It should be emphasized that this is the ecological behaviour of the species under the natural conditions of socialization. Since the living conditions of the plants in natural swimming pools are very close to those in natural locations, there is no reason why Ellenberg's pointer values should not be used in natural swimming pool planning.

Schwarzer and Schwarzer (2008) see the quality of the filling water as the most important ecological framework condition for the development of the plant population in natural swimming pools. The plant species used in the natural swimming pools are selected in relation to the initial values of the filling water used. This is done using the information provided by Ellenberg *et al.* (1992) on ecological preference; in particular the *R*-value (according to Ellenberg reaction of the water) and the *P*-value (modified according to Ellenberg, called *N*-value there, nutrient supply). In this way – analogous to Stelzer (2003) in natural waters – a correlation is established between water values and plant selection for natural swimming pool projects. Since the plant species selected according to the *R* and *P* values also have a known plant sociological position, i.e., their natural association with other species, this can also be considered in the elaboration of planting plans, whereby the coexistence of the species in nature is taken into account as far as possible.

4.16 INDOOR WETLANDS FOR GREYWATER TREATMENT AND REUSE

Fabio Masi[1], Anacleto Rizzo[1] and Martin Regelsberger[2]

[1]Iridra Srl, via La Marmora 51, 50121, Florence, Italy
[2]Technisches Büro Regelsberger, Marburger Gasse 11, 8200, Gleisdorf, Austria

4.16.1 Introduction

The term greywater (GW) describes the particular kind of domestic effluent produced by all the water sources in buildings except sanitation (toilets or similar devices). Greywater is characterised by an easily biodegradable organic content (mainly tensides, greases, oils, proteins), a very low content of nutrients and considerably high densities of pathogens. As this effluent is constantly produced at household level, it constitutes a secure water source for different options of reuse. There are in fact several advantages in keeping GW segregated in the building, treating it on-site and then reusing it:

(1) Easy recovery of water with little treatment;
(2) Reduced potable water demand and therefore less energy spent for treatment and pumping for the water supply network;
(3) Little additional piping;
(4) Possibility of energy recovery which leads to highly positive energy balance (i.e., recovering heat);
(5) Widespread adoption would lead to higher concentrations of wastewater at the centralised treatment plants and reduced carbon load and sludge production; this factor theoretically results in a more favourable C:N:P ratio in the wastewater to be treated at the WWTP and therefore better operational conditions.

TWs are proven to be efficient in treatment and reuse of greywater (Arden & Ma, 2018; Scheumann *et al*., 2009) and can play a fundamental role in future circular economy approaches to wastewater treatment with nature-based solutions (NBS) (Masi *et al*., 2018). GW can be treated on-site by NBS like TWs located in the external available space, and in case of lack of such availability also by indoor treatment units.

Usage of indoor installations can obviously completely remove the common issue in the adoption of NBS, usually considered as extensive and soft engineering techniques (Weissenbacher & Müllegger, 2009), and is also offering further benefits, such as

(1) Humidity control,
(2) Provision of a safe source of water for irrigating indoor green and landscaping,
(3) Lower dependency of treatment performances on outdoor temperatures, as water stays at indoor temperature (reduced energy losses in cold climate),
(4) Lower risk of invasive weeds or pests,
(5) CO_2 storage and O_2 generation,
(6) Contribution to reducing indoor air pollution, and
(7) Minimisation of infrastructures aimed at reuse (fewer pipes and pumps needed).

Still relating to the footprint issue of the on-site GW treatment and closed-loop reuse, the best advantages are linked to the adoption of green walls making use of internal or, even better, external walls of the building, to hang the treatment units (Castellar da Cunha *et al*., 2018; Fowdar *et al*., 2017; Masi *et al*., 2016).

In terms of the biochemical processes involved in recycling greywater with an appropriate quality, if good segregation from the blackwater (or even just urine) is performed therefore obtaining the expected low values for ammonia and total nitrogen, then filtration, sedimentation and microbial degradation of the highly biodegradable organic content are sufficient for reaching the desired outputs for reuse. In fact, where standards exist for greywater reclamation, they are primarily focused upon microbial indicator organisms (total/faecal coliforms; *E. coli*), organic content (BOD_5), turbidity/suspended solids and pH (Avery *et al*., 2007).

4.16.2 Design consideration of indoor wetland systems

The following aspects should be considered when designing indoor treatment wetlands for greywater recycling:

- Primary treatment should include a screen with automatic backwash. After screening, a degreaser will be needed, if kitchen water is included in the greywater pipe. An inlet buffer tank should be constructed both for equalising the loads to the treatment unit and for recovering thermal energy through a heat exchanger, making use of the temperature gap between the cold water in the mains and the warm effluents discharged after home usage, for instance for preheating hot water.
- The adoption of subsurface-flow wetland systems is strongly advised in order to avoid any chance of mosquito breeding and odour diffusion inside the building.
- For the moment the semi-empirical kinetic constants or the statistical interpretation of the available databases on TWs performances are not providing specific and reliable values for use in the sizing equations, because of the very scarce peer-reviewed literature yet available on indoor TWs for greywater treatment and therefore the sizing is performed according with conventional methods. In future, a smaller theoretical footprint could be expected, designing by values collected specifically at similar full-scale applications, considering the much faster biodegradability of the typical greywater compared to the mixed grey + black domestic wastewater. Still during the design phase the two following steps have to be carefully minded:
    - *HF wetlands*. It is important to be conservative in cross-sectional organic load check, in order to avoid issues generated by biological clogging;
    - *VF wetlands*. The oxygen balance (oxygen inputs subtracted by the total oxygen demand) must be positive (see Chapter 5.2 on VF wetlands).
- Possible failures or lack of routine maintenance of the primary system can easily bring bad smell events with every flush event. As a consequence, keep the feeding system (distribution pipes) below the filling material surface in order to limit possible odour issues.
- The choice of water-tolerant plants according to availability of (preferably) natural light, or alternatively by lamps, and by the loading and operative mode of the treatment unit (i.e., in greenwalls, some pots can be kept saturated while some others fed by several flushes per day). Tropical plants are generally well adapted to the almost constant indoor temperature. Still referring to greenwalls, due to the efficient rooting linked to the small available volume (single pots) that the vegetation has for growing, the choice can also include terrestrial plants and be driven by aesthetic requirements.
- The treated effluent can be collected in a "service water" buffer tank, taking into consideration the following suggestions:
    - Design the accumulation volume as a function of water reuse demand; a simple water budget (supply availability–demand) for the different seasons in a year can optimise drastically the investment costs of the buffer tank;
    - Always consider backup feeding from either rainwater harvesting or water mains (malfunctioning, operation and maintenance of the treatment units);

   - Consider the possibility of integrating rainwater harvesting in case of same reuse of harvested rainwater and treated greywater (Leong *et al.*, 2017).
- A final disinfection, preferably by a UV lamp, is sometimes needed, depending on the type of planned reuse.

4.16.3 From horizontal to vertical: consideration on the use of indoor greenwalls for greywater treatment and reuse

The specific processes involved in greywater treatment by treatment wetlands (as explained before, these are mainly TSS and organic matter removal processes, i.e., sedimentation, filtration, adsorption, microbial degradation) have influenced the technical choices for the former full-scale designs of such application. The most common choice for external installations has been the simple passive HF wetland, more often gravel based, with a surface need of about 1–1.5 m^2/pe in a temperate climate. In arid climate conditions, though, this kind of technology presents an undesired consequence, the relevant reduction in the production of "new water" (the treated effluent) because of the high losses by evapotranspiration and evaporation.

When space outside of the building is not available for an external installation, there are still options for other NBS, quite comparable with the treatment wetlands existing typologies, such as Rooftop Wetlands or Greenwalls; both these solutions can provide several positive effects to the urban environment and enhance the possibility to valorise greywater (Masi *et al.*, 2018).

It must be put in evidence that this specific application is a novel technology and most of the published literature relates to studies only at pilot stage (Masi *et al.*, 2016) and commonly conducted with synthetic wastewater and not with real greywater (Prodanovic *et al.*, 2017, 2018). From these first studies some results and design considerations can already be highlighted:

- Indoor installations can play a role in making this technological choice suitable for reuse in developing countries, mainly because of the better climatic conditions ensured inside the building (Masi *et al.*, 2015b).
- Particularly for installations like rooftop wetlands or greenwalls (also named Living Walls – LWs) it is extremely important to make use of light material as filler, and porous material can be preferred for the higher provision of available surface for biofilm growth (Prodanovic *et al.*, 2017, 2018; Ramprasad *et al.*, 2017).
- There are already several proposals about how to implement a greywater treatment and reuse by NBS scheme integrating it into a multi-storey building (Castellar da Cunha *et al.*, 2018; Masi *et al.*, 2016); as an example, the treated effluent could be:
   - Accumulated and stored at the bottom of the building, mixing all the different apartments effluents and pumping them back to an upper store tank which is feeding all the flush toilets tanks by gravity;
   - Directly reused by gravity using as source for each apartment the upper apartment (reduced pumping).
- Another recent design suggestion is to include Hybrid Living Walls composed by VF and HF wetlands, presented in several combinations and even as stand-alone unit, designed for treating both lightly polluted wastewater, such as greywater, as also hydroponic growth effluents (rich in nutrients). This could contribute to a possible future development of urban farming or vertical farming, circular economy approaches implemented at urban scale with a particular focus on recovering precious nutrients such as phosphorus and ionic nitrogen (Castellar da Cunha *et al.*,

2018). The VF units are cylindrical pipes filled with three different layers with appropriate size for extending the HRT as much as possible without risks of superficial clogging, while the HF units are filled with P-reactive material mixed with some organic media (1:1).

- Shallow HF wetlands, filled with a 1:1:1 mix of gravel, sand and brick bats (with increasing size from 0.5 to less than 50 mm), operated with a HRT of about 1 day, a HLR of about 58 $L/m^2 \cdot d$ and an OLR of about 14 $gO_2/m^2 \cdot d$, are showing optimal performances for reuse in Indian climate (GROW = Green Rooftop Water Recycling System; Ramprasad *et al.*, 2017).
- In general terms the inclusion of water-saturated zones in the treatment reactor creates some interesting effects such as P adsorption, longer HRT, higher absorption of eventual persistent organic compounds, and the obvious denitrification process. In case the design is mainly aimed at nutrient recovery, though, nitrates can still be considered as valuable molecules and therefore unsaturated systems can be considered an efficient technical option. A proper selection of plants and filling reactive media can play a role in case of nutrient removal targets. While the influence of ornamental plants on the overall treatment has yet to be studied, the selection of plant species with a well developed underground root system can help in breaking the clogging layer and maintaining the bed porosity, with a desired infiltration capacity of about 200–400 mm/d. For suspended solids and organics removal, any sand-based LW system is able to provide excellent removal rates (>80% for TSS and >90% for BOD). Targets for reuse are usually obtained by a LW surface of 1–2 m^2/pe (Fowdar *et al.*, 2017).
- Outdoor systems can present variations, compared with the indoor installations, in the infiltration rate/permeability of the system during cold weather periods. During the design phase an issue that should be considered is that leaves could divert the water flow out of the pots at a certain time of their growth, with a high contamination risk, if the feeding system and the plants are not properly selected for preventing such occurrence.
- Aluminium-based pots can offer a high reduction of risk factor in a fire risk assessment compared to the currently more often used plastic polymers.

Chapter 5

Practical information on design of specific wetland types and typical pitfalls

5.1 INTRODUCTION

Design manuals and guidelines are available from a number of sources worldwide, providing recommendations on all aspects of wetland design, operation and maintenance. The purpose of this chapter is to move away from specific guidelines and provide a summary of collective practical experience with different TW types by practitioners and researchers from around the world. The information is organised based on TW type rather than treatment application, to highlight key elements relevant to each configuration. The TW types covered are:

- VF wetlands
- French VF wetlands
- HF wetlands
- FWS wetlands
- Sludge treatment wetlands
- Aerated wetlands
- Fill-and-drain wetlands
- Floating treatment wetlands
- Willow systems
- Use of reactive media for enhanced P removal
- Multi-stage wetlands.

5.2 VF WETLAND

Bernhard Pucher[1], Riccardo Bresciani[2], Fabio Masi[2], Vit Rous[3], Alexandros Stefanakis[4] and Günter Langergraber[1]

[1]Institute for Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Muthgasse 18, A-1190 Vienna, Austria
[2]Iridra Srl, via La Marmora 51, 50121, Florence, Italy
[3]Czech University of Life Sciences Prague, Faculty of Environmental Sciences, Kamýcká 129, 165 21 Praha 6, Czech Republic
[4]Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

5.2.1 Overview of existing design guidelines

The main design parameters of VF wetlands according to the design guides in Denmark, Germany and Austria are summarized in Dotro *et al.* (2017). When VF wetlands are designed according to the guidelines, legal requirements regarding organic matter and ammonia nitrogen removal in these countries can be achieved (Table 5.1). To achieve almost complete nitrification, the Austrian, Danish and German guidelines (Brix & Johansen, 2004; ÖNORM B 2505, 2009; and DWA-A 262E, 2017; Nivala *et al.*, 2018, respectively) require that sand is used for the main layer of the VF filter with a minimum depth of 50 cm. Since 2017, also the Czech wetland design guidelines include VF wetlands requiring a specific surface of 4 m^2 per person and 50 cm main layer of washed sand (0.06–4 mm).

5.2.2 Main factors affecting treatment performance

The main factors affecting the treatment performance of VF wetlands are (e.g. Stefanakis & Tsihrintzis, 2012a):

- Filter material of main layer (grain size of material, filter depth)
- Loading: loading interval, volume of single doses, resting periods
- Loading rate: hydraulic and organic loading rates
- Distribution pipes: number of holes in distribution pipes.

Table 5.1 Comparison of legal requirements for organic matter and ammonia in Austria, Denmark, Germany and Czech Republic.

Parameter	Requirement	Austria[1]	Denmark[2]	Germany[3]	Czech Republic[4]
BOD_5	Max. effluent concentration	25 mg/L	–	40 mg/L	40 mg/L
	Removal efficiency	–	95%	–	–
COD	Max. effluent concentration	90 mg/L	–	150 mg/L	150 mg/L
NH_4-N	Max. effluent concentration	10 mg/L*	5 mg/L	10 mg/L*	20 mg/L
	Removal efficiency	–	90%	–	–

[1]For wastewater treatment plants, i.e. ≤50 PE.
[2]For wastewater treatment plants, i.e. ≤30 PE.
[3]For wastewater treatment plants ≤1,000 PE (organic matter) and ≤10,000 PE (NH_4-N).
[4]For wastewater treatment plants ≤10 PE and infiltration into groundwater.
*For effluent water temperatures >12°C.

The effect of selected parameters on the treatment performance of a VF wetland treating domestic wastewater is shown in the following for identical systems with 50 cm main layer comprising of three different filter materials (based on Pucher & Langergraber, 2019):

(1) Sand, 0.06–4 mm
(2) Coarse sand, 1–4 mm
(3) Gravel, 4–8 mm.

Measured volumetric effluent flow rates for calibration of the water flow model as well as measured influent and effluent concentrations of COD and NH_4-N for calibration of the pollutant transport and degradation model were available. For the VF wetlands with filter materials 0.06–4 mm and 1–4 mm, the wetland models have been calibrated on data described by Canet Martí *et al.* (2018) and Pucher and Langergraber (2018), respectively. For the 4–8 mm gravel system, volumetric effluent flow data were measured at BOKU University Vienna whereas concentration data came from the system as described by Nivala *et al.* (2019a). Table 5.2 summarises the main operational parameters of the VF wetlands for which the wetland models were calibrated.

For each of the calibrated wetland models (i.e., filter materials) simulations for the following operational settings were run for:

- Organic loading rates of 20, 40 and 80 g COD/m^2/d;
- Loading intervals of 1, 3, 6 and 12 hours;
- Number of holes per m^2 in distribution pipes: 0.5, 1, 2, 4; and
- Water temperature: 5, 10, 15 and 20°C.

Thus for each filter material 192 simulations were run (in total 576 simulations for all three filter materials). It has to be noted that not all combinations of operational setting are applicable, i.e., for VF wetlands using sand as filter material in temperate climates an OLR of 40 g COD/m^2/d can only be applied when the systems are operated during the summer months, whereas an OLR of 80 g COD/m^2/d leads to clogging of the system.

The same influent concentrations have been used for all simulations and all VF wetlands (Table 5.3). Thus only the effect on effluent concentration is reported. In the case of changing influent concentrations the design parameters – of course – also influence removal efficiencies.

Table 5.4 show the simulated COD effluent concentrations for the different filter materials and different OLRs. VF wetlands in Table 5.4 were loaded every 6 hours with distribution pipes having 0.5 holes in distribution pipes per m^2 (these are standard design values for VF wetlands using sand with grain size 0.06–4 mm as filter material). Table 5.5 shows simulated NH_4-N effluent concentrations for the same settings.

Table 5.2 Main VF wetland operational parameters of the data sets used for calibration.

Filter Material (mm)	Loading Interval (h)	Organic Loading Rate (g COD/m^2/d)	Number of Openings per m^2	Data for Calibration
0.06–4	6	20	0.5	see Canet Martí *et al.* (2018)
1–4	3	80	1	see Pucher and Langergraber (2018)
4–8	1	80	1	Water flow: measurements at BOKU University Vienna; Concentrations: Nivala *et al.* (2019a)

Table 5.3 Influent concentrations (in mg/L) used for the simulation study (from Pucher & Langergraber, 2019).

Parameter	COD	CR	CS	CI	NH_4-N	NO_2-N	NO_3-N	PO_4-P
Concentration	495	325	163	7	65	0.015	0.4	11.9

CR = readily and slowly biodegradable COD; CS = slowly biodegradable COD; CI = inert COD.

Table 5.4 Median and maximum COD effluent concentrations in mg/L of VF wetlands loaded every 6 hours with 0.5 holes in distribution pipes per m^2 (maximum concentrations in brackets).

Filter Material	OLR	5°C		10°C		15°C		20°C	
(mm)	g COD/m^2/d	Median	(Max)	Median	(Max)	Median	(Max)	Median	(Max)
0.06–4	20	42	(45)	24	(25)	18	(18)	17	(17)
	40*	79	(86)	48	(55)	28	(33)	21	(23)
	80*	115	(136)	82	(108)	55	(81)	39	(58)
1–4	20	56	(65)	32	(38)	21	(24)	19	(20)
	40	97	(117)	63	(85)	40	(57)	27	(39)
	80	131	(149)	100	(130)	67	(107)	43	(86)
4–8	20	58	(67)	33	(40)	23	(26)	20	(22)
	40	99	(119)	65	(87)	43	(58)	27	(43)
	80	139	(148)	104	(127)	73	(106)	47	(86)

*OLR $>$20 g COD/m^2/d is not experimentally verified in temperate climates.

Table 5.5 Median and maximum NH_4-N effluent concentrations in mg/L of VF wetlands loaded every 6 hours with 0.5 holes in distribution pipes per m^2 (maximum concentrations in brackets).

Filter Material	OLR	5°C		10°C		15°C		20°C	
(mm)	g COD/m^2/d	Median	(Max)	Median	(Max)	Median	(Max)	Median	(Max)
0.06–4	20	4.0	(4.0)	1.1	(1.1)	0.4	(0.4)	0.1	(0.1)
	40*	3.1	(3.6)	1.3	(1.4)	0.4	(0.4)	0.1	(0.1)
	80*	1.4	(9.3)	0.2	(5.6)	0.1	(4.5)	0.1	(4.7)
1–4	20	29.8	(29.9)	8.0	(8.1)	2.9	(3.0)	0.9	(1.0)
	40	28.1	(30.3)	7.1	(8.9)	2.9	(3.2)	1.1	(2.1)
	80	34.9	(43.3)	15.6	(30.3)	14.1	(28.7)	16.7	(28.8)
4–8**	20	30.5	(30.6)	7.5	(7.5)	2.8	(2.9)	1.0	(1.0)
	40	27.0	(29.0)	6.1	(8.3)	2.7	(3.0)	1.0	(1.6)
	80	34.8	(41.7)	13.9	(27.8)	13.9	(26.3)	14.8	(26.2)

*OLR $>$ 20 g COD/m^2/d is not experimentally verified in temperate climates.
**Results for NH_4-N: the 4–8 mm main layer could not be fitted well (see Pucher & Langergraber, 2018a).

Table 5.4 and Table 5.5 clearly show the importance of the filter material used for the main layer on the achievable COD and NH_4-N effluent concentrations. The coarser the filter material of the main layer, the higher the effluent concentrations. If operated with the same loading interval, at higher OLR higher single doses are applied. When coarser filter material is used for the main layer this results in higher flow velocities in the filter and thus reduced removal efficiencies. The increase is even more significant for maximum COD and NH_4-N effluent concentrations. A higher increase of the maximum compared to the median COD and NH_4-N effluent concentrations can also be observed at lower temperatures.

Figure 5.1 and Figure 5.2 show the effect of different loading intervals and different numbers of holes in the distribution pipes on simulated COD and NH_4-N effluent concentrations, respectively. The example shows a VF filter with 50 cm main layer of coarse sand (1–4 mm) operated with an OLR of 80 g COD/m^2/d. The removal efficiencies can be increased if (a) the loading interval is decreased (i.e. more doses with less volume per single dose) or (b) the distribution network gets denser (i.e. more openings per m^2). Both measures lead to lower water flow velocities in the filter and thus to higher performance and lower effluent concentrations. The reduction is even more significant for maximum effluent concentrations (dashed lines in Figure 5.1 and Figure 5.2, respectively). If the same loading interval is applied, the difference between maximum and median effluent concentrations gets less by increasing the density of holes in the distribution pipes (i.e. less volume of water per opening and thus lower flow velocities).

5.2.3 Field tests for filter material

The previous section showed the importance of the filter material for the performance of the VF wetland. All design standards for VF wetlands include specifications for the filter material. Most of the time, these include the grain size distribution (e.g., sand, 0.06–4 mm), d_{10} and/or d_{60} (grain size under which 10% and 60%, respectively, of the grains pass [by weight]) and U (the uniformity coefficient). Filter material is usually purchased from gravel pits according to these requirements.

However, it is advisable to test the sand delivered for the main layer before filling the bed. The following field test according to EN 12566-2 has been proven to be adequate if sand will be used as filter material and full nitrification is the treatment target. To carry out the test only a few items are required, i.e., a measuring

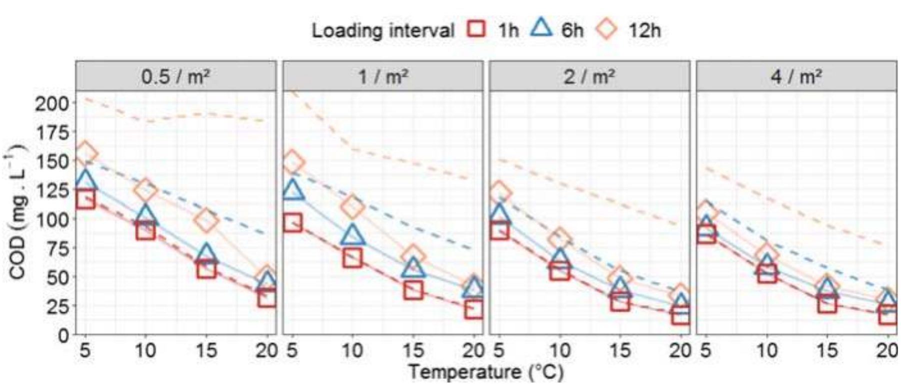

Figure 5.1 Simulated COD effluent concentrations of a VF filter with main layer of coarse sand (1–4 mm) for at an OLR of 80 g COD/m^2/d for different loading intervals and different number of holes in the distribution pipes (median concentrations: symbols; maximum concentrations: dashed lines).

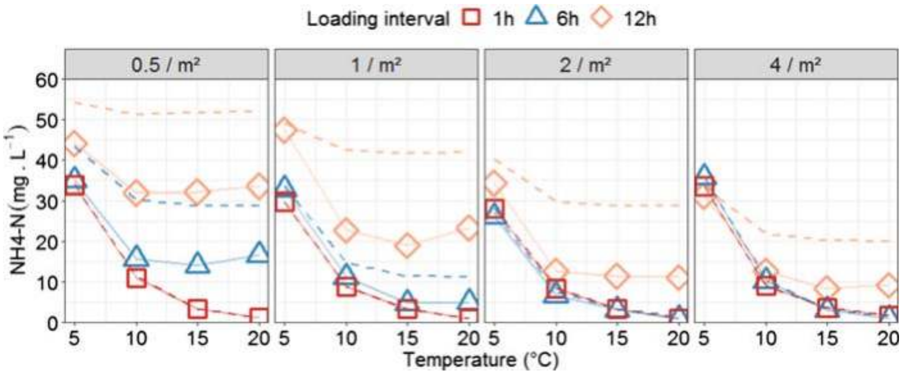

Figure 5.2 Simulated NH_4-N effluent concentrations of a VF filter with main layer of coarse sand (1–4 mm) for at an OLR of 80 g COD/m^2/d for different loading intervals and different number of holes in the distribution pipes (median concentrations: symbols; maximum concentrations: dashed lines).

cup, a stop watch, a metre stick and a thermometer (Figure 5.3). The instructions for the field test are as follows:

(1) Construct a test column of DN 100 mm, length 300 mm
(2) Put gravel (with grain size of 4–8 mm) in a bucket with draining holes on the bottom
(3) Put test column on top of the gravel
(4) Fill in 200 mm of filter sand to be tested, shake or knock on the pipe until filter column is medium densely packed
(5) Add 500 ml clean water without disturbing surface of the sand
(6) At the moment, when water has completely infiltrated start first test with stop watch
(7) At least 5 times in a row fill in 500 ml within 5 sec., stop time for infiltration

Figure 5.3 Items required for the field test of the infiltration capacity of filter material for VF wetlands according to EN 12566-2 (2005).

(8) When infiltration time stays nearly constant use average value out of 5 measurements.
(9) Calculate saturated hydraulic conductivity from:

$$k_s = \frac{l}{t} \ln \frac{h_1}{h_2}$$

where

k_s = saturated hydraulic conductivity (m/s);
l = length of the sand filter column = 0.2 m;
t = average value of 5 measurements of the infiltration time (s);
h_1 = head at the beginning of the infiltration test = 0.263 m (i.e. 0.2 m sand filter column + 0.063 m free water [500 ml] on top); and
h_2 = head at the end of the infiltration test = 0.2 m.

For sand 0.06–4 mm the target value for the infiltration time is I–30 s resulting in a targeted saturated hydraulic conductivity of $k_s = 10^{-3}$ m/s.

The obtained k_s at the air temperature during the test should be adapted to the climatic condition of the site. The final k_s value is acceptable if in the range 10^{-3}–10^{-4} m/s at $T = 10$°C. Values of k_s higher than 10^{-3} m/s limit the proper development of the biofilm and the nitrification processes, values lower than 10^{-4} m/s make the system very prone to clogging and favour too long saturated conditions in the sand layer.

5.2.4 Specific design considerations

Besides basic design recommendations presented by Dotro *et al.* (2017), the specific design considerations for VF wetlands are as follow:

- *Filter material*. The importance of the granularity of the filter material for the main layer has been shown in the previous sections. For sand-based VF wetlands, measuring the hydraulic conductivity onsite has been proven a valuable measure to ensure the hydraulic functioning of the system. Filter material should be mainly siliceous, with a low carbonate content. Besides using sand and gravel, also reactive media such as zeolite has been used to enhance the nitrification capacity of VF wetlands (e.g. Pucher *et al.*, 2017; Stefanakis & Tsihrintzis, 2012b).
- *Layer composition*. In the case 0.06–4 mm sand is used for the main layer, a transition layer of 4–8 mm gravel is required between the main and drainage layers to prevent the washout of the sand. To use different layers of filter materials with different grain size distributions does not have any advantage compared with using a main layer comprised of only one material.
- *Loading interval*. To achieve the maximum treatment efficiency, the time between two loadings must guarantee a complete percolation of the wastewater and complete aeration of the main layer of the VF wetland. This requires that for finer filter materials longer loading intervals are foreseen. For 0.06–4 mm sand the common design guidelines recommend less or equal than 6 loadings per day, i.e., a minimum loading interval of 4 hours.
- *Distribution system*:
  - *Loading with pumps*. In order to guarantee a complete distribution of the water on the sand layer and to ensure the cleaning of the distribution pipes, it is important to ensure brief and consistent loading periods. The single dose should be not less than 2 cm, whereas the pump flow should be decided on the basis of the diameter of the opening holes and in any case not less than 0.2 m^3/h per

m^2 of loaded sector. Minimal velocity in the pipes should be not less than 0.7 to 1 m/s to ensure their self-cleaning.

  - *Loading with siphons*. Similar considerations on flow and velocity are required when siphons are utilized to load the beds. Moreover, depending by the model of siphon, it is important to consider a minimum difference of level between the maximum water level in the siphon tank and the level of the surface of the VF bed. Siphon design has to ensure the maximum durability of the device, considering that this element is critical to allow adequate distribution of the water on the whole surface for years without requiring frequent maintenance interventions. Most of the siphons on the market are instead developed for self-cleaning operation of sewer with clean water and are not suitable for wastewater or frequent activations, resulting in improper functioning of loading operations. Therefore siphons should be tested for this specific use, ensuring a constant flow according with the design requirements and a fast emptying of the siphon tank
  - For *maintenance and operation* of the distribution system, it is advisable to allow their periodic cleaning every 1–2 years of operation, i.e., providing a removable plug at the end of each line.

- *Water-saturated zone at the bottom*. For VF wetlands with a main layer of coarse sand of 1–4 mm or gravel of 4–8 mm, a saturated zone on the bottom of the VF bed below the sand layer can be maintained to improve denitrification. It has been shown that denitrification cannot be enhanced with this measure for VF wetlands with a main layer of 0.06–4 mm sand (due to lack of organic matter for denitrification in the saturated water layer).
- *Shape of the VF beds*. VF beds are not subject to geometry constrains, therefore their shape can be chosen by interacting with landscaping architectural approaches and generate side benefits in terms of aesthetics, leisure and increased chances in finding available space for the realisation of extensive treatment solutions, making them a relevant component of the architectural design itself. When choosing the bed shape the only relevant factor which has to be kept in mind is to be able to make use of every m^2 of surface by an appropriate dosing of the influent over the whole surface, avoiding dead zones where the water will not flow properly. If the terrain where the beds are going to be realised is not too loamy, the side walls should be preferably designed with a 90° shape, in order to minimize the footprint of the system; otherwise a classic 1:1 ratio (45°) for the banks can be advised.

5.2.5 Considerations for the start-up phase

The following points shall be considered during the start-up phase of VF wetlands:

- Low initial applied loading rate, with gradual increase of the applied load in order to reach the design load by the end of the start-up phase.
- Ensure wastewater is distributed uniformly and reaches the most distant holes of the distribution pipes network.
- Secure plant establishment and growth and avoid open areas without vegetation: for this, the water level inside the VF bed can be set at a higher point to allow better growth of plants and gradually lower water level.
- Before start-up, to aid the establishment of the plants, the bed can be flooded with 5–10 cm of water above the surface, except when climate conditions favour algae formation that could partially clog the system at the beginning.
- During the first vegetative season a regular control and removal of weeds is very important for the growing of wetland plants; VFs are more prone than HFs to weeds intrusion.

5.3 FRENCH VF WETLANDS

Pascal Molle[1] and Dirk Esser[2]

[1]Department of Ecotechnologies, IRSTEA, Villeurbanne, France
[2]Société d'Ingénierie Nature & Technique (SINT), Chef-Lieu, F-73370 La Chapelle du Mont du Chat, France

5.3.1 Overview of existing design guidelines

French VF wetlands are two-stage and contain alternately operated cells for each stage. Sludge treatment and partial removal of organic matter takes place in the first stage, and nitrification and further removal of organic matter occurs in the second stage. The first stage, which is divided into three parallel filters, is fed with screened wastewater (Figure 5.4). The second stage is divided into two filters. The sludge from the first stage collects at a rate of approximately 2–3 cm per year and needs to be removed every 10–15 years. Sludge accumulation rates may be lower in systems that do not continuously receive the full design load. French VF wetlands are planted with *Phragmites australis* to ensure proper water

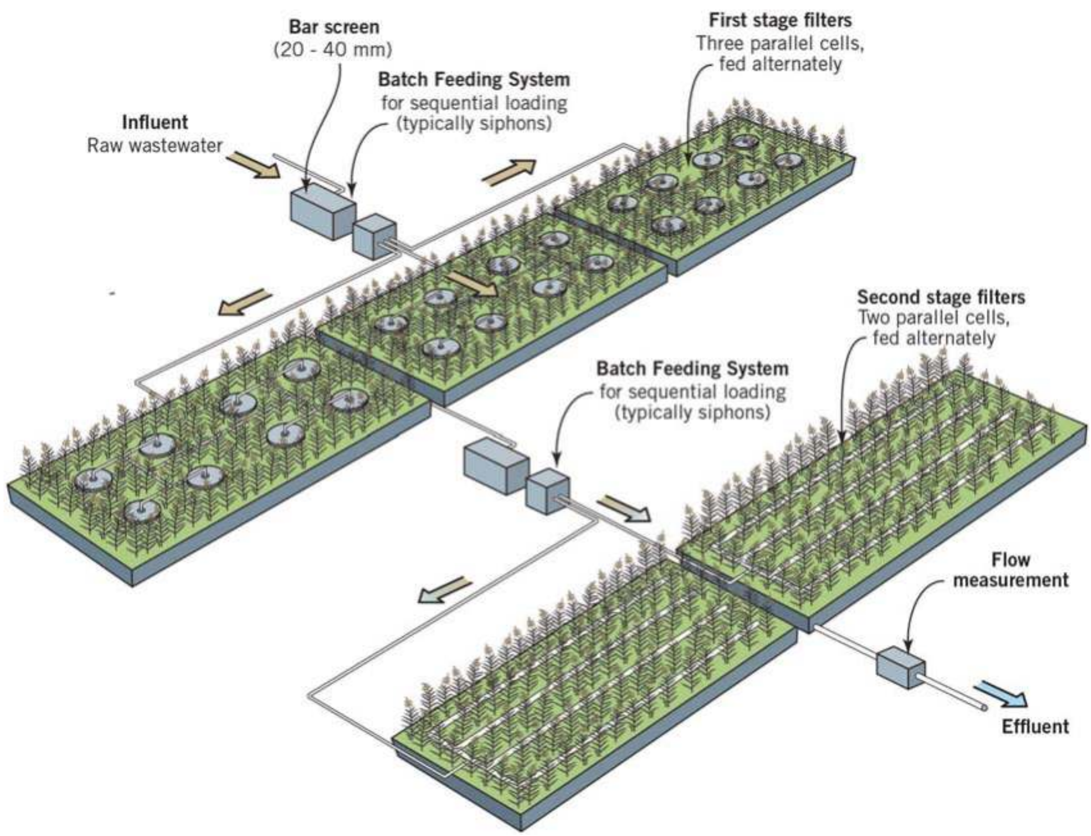

Figure 5.4 Schematic of the classical French VF design (Dotro *et al.,* 2017).

infiltration capacity and passive aeration of the filter. The *Phragmites* stems create small openings in the sludge layer that maintains the infiltration capacity of the filter. This is a critical component in proper functioning of the system. In other countries, other plants have successfully been used (Molle *et al.*, 2015) but it important to test whether other plant species can provide this function and also survive the resting periods without wastewater flow.

Over 4,000 classical two-stage French VF wetland systems have been built in France, with most systems serving populations less than 1,000 PE (Dotro *et al.,* 2017). The design has been adapted and implemented outside of France, specifically in tropical overseas French territories, South America, and other countries in the EU. The information in this technical report is restricted to the application of the classical two-stage French design in temperate European climates.

The maximum design loads for a classical two-stage French VF wetland system are given in Molle *et al.* (2005) and summarized in Table 5.6. For typical situations in France, this leads to a surface area of 1.2 m^2/PE for the first-stage cells and 0.8 m^2/PE for the second-stage cells. The anticipated effluent concentrations for French VF systems treating domestic wastewater are also provided in Table 5.6.

5.3.2 Hydraulic considerations

The cells of a VF French wetland are dosed on an alternating basis (e.g., one filter is dosed while the others are rested). The alternating dosing is a fundamental aspect of proper operation of the French VF system, because it (a) promotes mineralization and stabilization of the accumulated sludge (on the first-stage cells), (b) maintains aerobic conditions in the filter bed itself (both first- and section-stage cells), and (c) ensures that the plants in each cell receive water on a frequent basis (to avoid water stress). First-stage cells are typically loaded for three to four days and rested for seven days; second-stage cells are generally loaded for three to four days and rested for three to four days. This feeding schedule requires that the system operator visits the site twice a week to make these changes manually, unless dosing is performed by a programmable logic controller (PLC) system.

5.3.3 Specific design considerations

Influent distribution is different for first- and second-stage cells. First-stage cells generally use large distribution pipes to distribute the wastewater, with at least one feeding point per 50 m^2, and pipe diameters should be chosen for a flow velocity >0.7 m/s, in order to ensure self-curing. However, in order to avoid blockage, they should not be less than 90 mm in diameter. First-stage distribution pipes are suspended above the surface of the filter in order to allow for sludge accumulation, and a minimum flow of 0.5 $m^3/h \cdot m^2$ per batch is necessary to correctly distribute the water. Second-stage cells are fed

Table 5.6 Maximum design loads and expected effluent concentrations for classical French VF wetland design under dry weather conditions. Values are given per square metre of bed in operation (Dotro *et al.,* 2017).

	Hydraulic Load (m/d)	COD ($g/m^2 \cdot d$)	BOD_5 ($g/m^2 \cdot d$)	TSS ($g/m^2 \cdot d$)	TKN ($g/m^2 \cdot d$)
First stage	0.7	350	150	150	30
Second stage	0.7	70	20	30	15
Final effluent concentration	–	75 mg/L	15 mg/L	15 mg/L	15 mg/L

with pipes that are installed directly on the filter surface. Feeding points are drilled holes and there should at least be one hole for every 2 m^2 of filter surface. The diameters of the pipes and of the holes should be chosen in order to limit differences in flow between any two feeding points to less than 10%, which means minimizing the headloss in the pipes. The diameter of the holes should assure a squirt height of at least 25 cm at the outflow of each hole, but should be at least 8 mm, in order to avoid blockage. The squirt height at the outermost orifices in the distribution pipes on the second-stage cells must be >30 cm. In order to maintain aerobic conditions in the filter, passive oxygenation by the bottom of the filter is necessary. Drainage pipes (minimum diameter 125 mm) contain slots (with a length of one-third of the pipe circumference, and width greater than 8 mm) for at least every 25 cm of pipe length.

Different filter media are used in the first- and second-stage cells. The first-stage cells have a main layer of 2–6 mm gravel, which is coarse enough to avoid problematic clogging but fine enough to support the formation of a sludge layer on the surface of the filter. Below the main layer is a transition layer of larger gravel (5–15 mm) which prevents finer particles from being washed into the drainage layer. The drainage layer consists of a coarse gravel (20–60 mm) which is installed along with drainage pipes on the bottom of each cell.

Second-stage cells use sand for the main layer ($0.25 < d_{10} < 0.4$; uniformity coefficient <5; less than 3% fines). A deeper layer of sand must be used if the sand specifications in Table 5.7 cannot be met. The transition layer (3–12 mm gravel) and drainage layer (20–60 mm gravel) must adhere to the Terzaghi rule ($D_{15}/d_{85} \leq 4$) and permeability criterion ($D_{15}/d_{15} \geq 4$) to ensure that the interface between the filter layers does not produce a decrease in permeability by reducing the local porosity.

Construction of the cells is generally with a side slope of 1:1. The cells are lined with a combination of a plastic liner and geotextile membrane.

5.3.4 Considerations for the start-up phase

During the first year of start-up, excessive growth of weeds in the filter must be avoided. The only way to do this is to manually remove the weeds. It is possible to saturate an individual cell for one or two weeks during the first growing season to kill the weeds and favour establishment and growth of the *Phragmites*. However, do not saturate both first- and second-stage cells simultaneously because this will hinder the nitrification process.

If the system starts operation with a very low hydraulic load, the *Phragmites* that are not located near a feeding pipe can undergo water stress. This does not impact on the performance of the filter but can favour the growth of weeds. Weed removal is a tedious and time-consuming task.

Systems started at the nominal design load will form a sludge layer relatively quickly. If the *Phragmites* are still small, they cannot aid in water infiltration and mineralization of the sludge layer. The sludge

Table 5.7 Filter media specifications for French VF wetlands (Dotro *et al.,* 2017).

	First Stage		**Second Sctage**	
	Depth	**Material**	**Depth**	**Material**
Freeboard	>30 cm		>20 cm	
Main layer	30–80 cm	2–6 mm gravel	30–80 cm	sand $0.25 < d_{10} < 0.4$ and $d_{60}/d_{10} < 5$ and less than 3% fine particles
Transition layer	10–20 cm	5–15 mm gravel	10–20 cm	3–12 mm gravel
Drainage layer	20–30 cm	20–60 mm gravel	20–30 cm	20–60 mm gravel

deposits dry quickly, without mineralization, which can result in unwanted ponding on the surface of the bed. This problem ends when the *Phragmites* stand becomes established.

Storm events during the first year can result in ponding and/or surface clogging on the second-stage cells. This will end once the sludge layer is established on the first-stage cells. To accelerate the establishment of a sludge layer on the first-stage cells, a sludge or compost layer can be applied during start-up of the system.

5.3.5 Routine maintenance

Unless the filters are automatically dosed with a PLC system, the operator must visit the treatment system twice a week to alternate the feeding on the first- and second-stage cells. Feeding should be alternated every three to four days to maintain sufficient oxygen transfer into the bed. Weeds should be removed on a monthly basis and harvested once per year (if necessary). The height of the sludge layer should be checked once per year. The sludge deposit layer must be removed once it reaches a depth of 20–25 cm (generally every 8–15 years), otherwise problematic ponding will occur. Removal is conducted with mechanical machinery, and the sludge can be spread on fields (depending on local regulations). There is no need for a resting period before sludge removal (as opposed to sludge treatment wetlands), and the French VF wetlands can be put back into operation immediately after removal of the sludge layer. The sludge layer generally has a dry matter content of $>25\%$ and an organic matter content of less than 40%.

5.4 HF WETLANDS

Alessia Marzo[1], Giuseppe Luigi Cirelli[1], Mirco Milani[1], Gabriela Dotro[2] and Jan Vymazal[3]

[1]Department of Agricultural, Food and Environment (Di3A), University of Catania, Via Santa Sofia 100, Catania 95123, Italy
[2]Cranfield University, Cranfield, Bedfordshire, MK43 0AL, United Kingdom
[3]Czech University of Life Sciences Prague, Faculty of Environmental Sciences, Kamýcká 129, 165 21 Praha 6, Czech Republic

5.4.1 Introduction

Horizontal-flow wetlands have been used for a number of decades around the world. Current design, operation and maintenance guidelines have been summarised by Dotro *et al.* (2017). Many reviews exist in the literature on performance of these systems for treating municipal sewage, agricultural wastewater, industrial effluent, mine drainage, landfill leachate, polluted river and lake water, urban and highway runoff (Vymazal & Kröpfelová, 2008) The use of horizontal-flow wetlands has also been developed in various climate conditions such as cold climate (Wang *et al.* 2006) and tropical climate (Zhang *et al.* 2014). HF wetlands are most effective for removal of organic matter (measured as BOD and COD) and total suspended solids (TSS).

5.4.2 Design considerations

Most guidelines include recommendations on the main factors affecting the treatment performance of HF wetlands. This section summarises the rationale behind the recommendations made.

- *Filter material*. The selection of substrate is a key design parameter because it provides the area for biofilm attachment, rooting medium for the emergent plants, adequate hydraulic retention time and, if required, can react with specific pollutants such as phosphorus or metals. The hydraulic conductivity of the media is considered in current sizing criteria to balance the risk of clogging and contact time between the wastewater and the media (biofilm). Whilst media can be natural, industrial by-products or engineered products, typical material is gravel with sizes of 8–16 mm for the main layer and 50–200 mm for the inlet and outlet zones. Soil has proven to have too low a hydraulic conductivity for the loading rates typically applied and as such is no longer recommended. Whilst checking grading on delivery of media to site can be done with sieves, in reality this is not performed as the range of gravel sizing recommended is broad enough to be less critical if deviations occur, and easy to visually detect.
- *Distribution of wastewater*. Systems are typically loaded along the width of the bed, either with subsurface pipes (secondary treatment) or surface troughs (tertiary treatment). Cleaning access needs to be provided to either type of flow distribution structure as flow velocities from the upstream processes can vary daily and settling can occur within the pipes or troughs. Coarse stones are used to help flow distribution in depth.
- *Upstream treatment processes and loading rates*. The pollutant loads (g/m^2 d) are typically expressed in terms of plan area ($L \times W$), although for clogging considerations the cross-sectional area ($W \times D$) is a critical parameter. Rule of thumb sizing approaches assume typical influent quality and therefore loads applied. For example, areal loading rates of less than 10 g BOD m^{-2} d^{-1},

20 g COD m^{-2} d^{-1}, and 10 g TSS m^{-2} d^{-1} have been shown to enable secondary HF wetlands to operate without surface water ponding for 15 years of operation (Vymazal, 2018). Kadlec and Wallace (2009) suggested key design parameter for HF wetlands a design limit for the cross-sectional loading rate 250 g $BOD_5/m^2/d$. In tertiary systems, similar BOD and TSS areal loading rates have been employed in tertiary systems resulting in refurbishment intervals between 8 and 15 years. The main difference is the quantity of water that passes through the system as tertiary systems with hydraulic loading rates of 0.2–0.4 m/d, as opposed to 0.02–0.05 m/d in secondary systems (Knowles *et al.,* 2011). The capital and operational costs associated with sizing tertiary systems at these high hydraulic loading rates accept the fact that it will result in increased refurbishment intervals, as it is still a lower whole-life cost solution than building a significantly larger system that lasts longer between refurbishments (Dotro & Chazarenc, 2014).

Influent water quality can also affect the predominant wetland processes and require additional management allowed for in the design stage. Strongly anaerobic wetlands like secondary HF beds can generate sulphide and associated odours and a white discharge that will need management. Tertiary HF systems can be carbon limited for denitrification, resulting in low nitrate removal rates.

5.4.3 Potential design and operational issues

As HF systems are inherently passive (no mechanical parts for operation) and the media is fully saturated with water, they are less susceptible to critical failure than other wetland systems. Where systems have encountered major issues these are typically due to poor O&M or significant deviation from design guidelines. Experience from over twenty years of HF systems for secondary and tertiary treatment suggests, as well as following design guidelines (Dotro *et al.*, 2017), a few steps are recommended for operation. These include the protection of plant establishment in the first year of operation and management of preferential flow paths in mature systems.

Plant establishment may need to be protected based on site (region) specific risks. The most common strategies for protecting plant establishment from rodents that feed on reed plantings have included temporary flooding and temporary rabbit fencing. Flooding has resulted in unintentional incorrect operation of HF beds, with operators forgetting to lower the water level once plants have established. Fencing requires additional investment and can have a negative visual impact on the overall system. In many instances, no protection has been employed and the plants successfully established. Therefore, the risk of rodent access and damaging effects should be assessed during the design phase.

Preferential flow paths in mature systems will form as a result of accumulation of inert organics and the decay of biofilm within the bed media, as well as in the surface of HF systems that are surface loaded (Knowles *et al.*, 2011). Management strategies have included resting of HF beds (i.e., draining the water and leaving it to dry for a number of weeks) and altering the operational water level in the beds. Both of these solutions have implications for sites with only one operational bed as the ability of the system to continue to provide treatment is impaired during intervention.

5.5 FWS WETLANDS

Alexandros Stefanakis[1] and Tom Headley[2]

[1]Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman
[2]Wetland & Ecological Treatment Systems, 84 Melbourne Street, East Maitland, NSW 2320, Australia

5.5.1 Overview of existing design guidelines

FWS wetlands are probably the oldest TW type that has been tested and used, owing to its relative ease of construction, operational simplicity and considering that it resembles most closely a natural wetland. They are widely applied in North America (Kadlec & Wallace, 2009) and Australia (QDNR, 2000), but less so in Europe. FWS wetlands are typically applied as a polishing stage of secondary effluents, e.g., activated sludge, MBR, lagoons, but they have also been used in the treatment of various industrial and agro-industrial wastewaters and also surface runoff/stormwater.

There have been various design approaches over the years. Typically, sizing of FWS wetlands is based on area or volume. Design parameters that have been widely used in the volume-based design are the hydraulic retention time (e.g., 2–3 days per cell; USEPA, 2000), the vegetation porosity (0.65–0.75; Reed *et al.*, 1995), the water depth (0.1–0.6 m) and the length to width ratio (2:1 to 5:1; Economopoulou & Tsihrintzis, 2004). Area-based design considers the pollutant reduction using the overall wetland area. There is some merit to considering the sizing from an areal perspective, rather than volumetric, since increasing the area is the primary means to increase the amount of vegetation, and hence submersed stems supporting the biofilms responsible for treatment. Increasing the wetted depth (to gain more volume) beyond about 0.5 m will lead to a decline in vegetation health and a reduction in stem density. BOD and nitrogen removal rates are typically based on first-order kinetics and on the assumptions of plug flow (Crites and Tchobanoglous, 1998; Reed *et al.*, 1995; USEPA, 1988). The P–k–C^* first-order model is one of the latest models that can be effectively used to size the system and estimate its performance, e.g., for BOD_5 and/or ammonia reduction (Kadlec & Wallace, 2009).

5.5.2 Considerations for the start-up phase

For several reasons, a dense coverage of healthy and vigorously growing wetland vegetation is a particularly important component of a successfully functioning FWS wetland system, especially if the focus is on treatment. The vegetation provides a significant portion of the submersed surface areas for attached growth of biofilms and periphyton responsible for many biochemical treatment processes. For treatment processes such as denitrification, the designer often relies on the productivity and turnover of wetland vegetation to provide organic carbon via the internal photosynthetic conversion of CO_2 into biomass. Patchy vegetation growth can reduce the hydraulic efficiency of the wetland by creating short-circuiting flow paths. In many cases, gaps that develop in plant cover and occur during the initial planting and establishment phase can persist for many years and be difficult to close in an operating system, especially once bird populations establish in the wetland, which can have the effect of actively maintaining or expanding areas of open water. Thus, the initial planting and vegetation establishment period is a critical phase in the construction and commissioning of a FWS wetland.

Several factors need to be considered to optimize the chances for successful vegetation establishment, including appropriate plant species selection and diversity, use of good quality seedlings of an appropriate level of development (not too young, but not too old and root-bound), time of planting (spring is good, while late winter can be detrimental in cold climates that experience frost), suitable and well levelled topsoil, ensuring an adequate supply of suitable quality water immediately after planting, careful management of water levels and flow during the first three months after planting (keep soil moist and raise the water level as the young plants increase in height), management of algae which can quickly smother the soil surface and impose an oxygen stress on young plants in nutrient-rich waters, and management of waterfowl, aquatic wildlife and insect pests which may cause rapid damage to large areas of freshly planted seedlings.

Depending on the climate and nutrient status of the water, and how well the above considerations are managed, it may take anywhere between 6 and 24 months to achieve a dense coverage of well established vegetation. For some treatment processes (e.g., denitrification) which rely on the biofilms and organic matter turnover afforded by a healthy stand of vegetation, the duration of this start-up time should be considered in the project planning, especially if meeting specific treatment performance targets is critical.

5.5.3 Considerations for the construction

Despite the apparent simplicity of the FWS wetland system, there are a series of technical and economic challenges that should be considered in the construction stage, such as:

- Site selection, topography, geology and land availability are the first parameters that will define the economics and the feasibility of the project, considering that FWS wetlands tend to have higher area demands and typically operate via gravity flow.
- Good soil quality for the substrate and local availability are also important for the successful construction and operation of the system. Care should be taken that weed seeds are not introduced with the topsoil.
- Depending on the size of the system, a plastic impermeable liner or a clay soil layer can be used to seal the bottom. However, as the wetland size increases, the use of a plastic liner becomes prohibitive due to high cost, and a natural sealing layer is preferred. Hence, local availability of proper quality and quantity of the required materials is important.
- In terms of earthworks, adequate bund stability, including compaction and proper materials, hydraulic and geotechnical considerations, is also crucial to avoid any damage once the water level within the FWS wetland bed starts increasing. In general, earthworks should aim to balance excavation and filling to avoid buying surplus soil or discarding superfluous soil.
- FWS wetland systems designed for stormwater/runoff treatment may receive high volumes of water within short period time, thus the system should be able to accommodate these volumes and be constructed to withstand the expected flow velocities and erosional forces.
- For larger FWS wetland systems, large number of plants may be required for the plants establishment, which means that proper plant propagation schedule and logistics plans should be in place.

5.5.4 Design and dimensioning

For FWS wetlands designed for water quality improvement purposes, determining the size of the wetland to achieve certain pollutant reduction requirements is usually done using some form of first order concentration reduction model with reaction rates for each parameter of concern calibrated against performance datasets from existing systems. The current state-of-the-art approach for this is the *P–k–C** model, which is essentially a form of the retarded first-order tanks-in-series model derived for conventional wastewater treatment unit processes. An areal (rather than volumetric) approach is generally considered most appropriate for FWS wetlands (Kadlec & Wallace, 2009). Once the required area has been calculated from such a concentration reduction model, several cross-checks should be performed to verify if the predicted performance is in line with the experience base from other systems (e.g., by comparing areal mass removal rates), and to identify if any process limitations may exist which could slow the rate of pollutant reduction (e.g., alkalinity required for nitrification, or organic carbon required for denitrification). Such sanity-checks are particularly important for atypical wastewaters and contaminants or influent concentrations which are beyond the realm of the common performance experience (i.e., tertiary treatment of sewage). In some cases, a pilot study may be wise to gather information on performance rates and limitations.

After the FWS wetland area has been determined, the next critical design step is that of defining the number and configuration of individual wetland cells (in parallel and series) and their dimensioning (length and width). This is largely an iterative process to find the optimal solution with consideration of wetland hydraulics (headloss), site topography and slope, optimizing earthworks quantities (cutting versus filling) and operational considerations (e.g., ability to take cells off-line for maintenance). A key consideration here is the hydraulic design and calculation of headloss from inlet to outlet of a wetland cell, in order to define the maximum allowable length of any individual cell (for the given inflow rate and selected number of parallel cells). The vegetation imposes a resistance to flow through the wetland, which requires head (water elevation) to overcome this resistance. In large-scale wetland systems, the head-loss can be significant, resulting in significantly deeper water at the inlet end of the wetland (inhibiting plant growth) if the hydraulic design is not carefully considered. The power function calculation approach recommended by Kadlec and Wallace (2009), which includes a coefficient to account for the density of vegetation, is the most appropriate method currently available. To a certain extent, bed slope can be used to provide some of the head to overcome the vegetative resistance. However, achieving slight grades accurately during construction adds difficulty. Excessive difference in elevation between the inlet and outlet of a wetland cell, due to bed slope, also creates the risk that the front end of the cell will dry out (threatening vegetation) at low or no flow.

It is also important to consider the water balance for the wetland once the size is defined. In arid climates, evapotranspiration (ET) losses can be substantial, especially if relatively long residence times are required for treatment, leading to problematic salinity concentrations at the outlet or even no outflow during hot summer conditions in the worst case. Conversely, in tropical monsoonal climates which experience more than 3000 mm of rainfall per year, the wet season rainfall captured by the wetland can dominate the water balance, exceeding the influent hydraulic loading rate and resulting in a significant increase in the volume of water exiting the wetland which needs to be managed. As a minimum, a monthly water balance should be compiled to estimate the monthly outflow volumes, considering as a minimum the expected inflow rates, historical rainfall for the site (average and variability), an estimate of evapotranspiration (either from local Class A-pan data, reference ET from a weather station or monthly average potential ET maps that exist for many regions of the world), and assumptions about infiltration/exfiltration rates.

The vegetation selection and planting plans are also very important design considerations. The plant species should be selected based on the locally occurring flora, site conditions (e.g., climate, soil), the water quality (e.g., salinity, nutrient status and organic load), design water depth and considerations such as biodiversity and habitat creation. A high diversity of plant species is recommended to increase the ecological resilience of the wetland, especially with regards to pests and diseases which may threaten the health of the vegetation. The planting density needs to be defined, with consideration of cost (tending towards a lower density) and the desire to establish a dense cover of vegetation in the shortest timeframe and achieve the design treatment performance as soon as possible. Planting densities between 0.5 and 6 plants per m^2 have been used. In some FWS wetland projects, being constructed (or "reinstated") in former wetland sites that may have previously been drained, a sufficient seedbank may exist in the soil to achieve adequate revegetation without planting.

5.5.5 Main factors affecting treatment performance

In summary, the treatment performance of FWS wetlands can be affected by the following main parameters:

- Climatic conditions, i.e., rainfall, temperature variations, evapotranspiration or seepage, if not taken into account during the design stage;
- Inadequate hydraulic retention time and/or hydraulic design (e.g., length to width ratio);
- Higher applied pollutant loads than assumed in design, which exceed the oxygen transfer capacity of the wetland and result in anaerobic conditions, respective nuisance and decline in vegetation health;
- Monocultures, i.e., use of only one plant species, promote insects' development;
- Inadequate plant coverage and large open water areas, which could create algae blooms;
- Selection of plants species not adopted to the specific climate and water quality;
- Lack of vegetation management, overgrowth of plants and increased vegetation porosity, which may change the hydraulic flow patterns, create preferential flow within the system and, thus, affect the transformation/removal processes; and
- Variations in water depth and/or periods without inflow (e.g., in stormwater wetlands), which can result in dry-out and potential risk of releasing pollutants stored in the organic sediments of the bed.

5.6 SLUDGE TREATMENT WETLANDS

Steen Nielsen[1] and Alexandros Stefanakis[2]

[1]Orbicon, Linnés Allé 2, DK – 2630 Taastrup, Denmark
[2]Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

5.6.1 Overview of existing design guidelines

Sludge Treatment Reed Beds (STRBs) or Sludge TWs are designed to dewater and mineralize sludge from Wastewater Treatment Plants (WWTPs) and Water Works. The sludge is passively dewatered by drainage through the filter and by evapotranspiration. Plant and microbial activity contribute to the dewatering, aeration and mineralization, leaving the treated sludge residue layer on top of the filter. The process results in the production of a higher quality biosolid end-product, which can be safely reused and recycled as a fertilizer or soil enricher (Nielsen & Bruun, 2015; Stefanakis *et al.*, 2011).

The main design parameters of STRB according to the design guidelines in various countries have been summarized (Nielsen, 2003; Nielsen & Willoughby, 2005; Nielsen *et al.*, 2018; Stefanakis & Tsihrintzis, 2012c; Stefanakis *et al.*, 2014). Dimensioning of the STRB is based on sludge production (tons of dry solids per year), sludge origin and quality, and climate. Those dimensioning criteria define the process area, the area load (kg DS/m^2/yr), the number of basins, loading and resting periods and finally the capacity of the system and the basins during the emptying period (Table 5.8).

Loading must be planned in such a way as not to inhibit the development of the reeds and to prevent the sludge residue from staying permanently wet and growing so fast that could undermine the reeds growth. In order to achieve the necessary balance between loading and resting periods and meet the requirement for long-term treatment, it is recommended that the systems have a minimum of six to eight basins depending on the climate (Table 5.8). According to the guidelines and the operational strategy, a STRB commonly operates for around 30 years. During this period, two to three operational cycles of 10–15

Table 5.8 Design and dimensioning criteria (*dimensioning in hot climates).

General Guidelines	
Number of basins	8–14 (6–10)*
Area load (kg DS/m^2/yr) – Full scale	30–60 (50–100)*
Area load (kg organic solid/m^2/yr)	20–40
Loading days	3–8
Number of daily loads	1–3
Resting days (older systems)	40–50 (7–21)*
Operation cycle	10–15 years
Feed Sludge	
pH	6.5–8.5
Dry solid (%)	0.3–4%
Loss on ignition (%)	50–65%
Fat (mg/kg DS)	5,000
Oil (mg/kg DS)	2,000

years are completed. An operational cycle consists of four phases: (1) commissioning, (2) normal operation, (3) emptying and final disposal of the sludge residue, and (4) re-establishment of the system (Nielsen, 2003; Stefanakis *et al.*, 2014).

5.6.2 Considerations for the start-up phase

Before a new STRB can become fully operational or a newly emptied bed can be put back to operation, it must undergo a period of 1–2 years depending on the regrowth or replanting and the climate. During an operational cycle, the different beds in the STRB are emptied in shifts to avoid simultaneous emptying and/or commissioning. An operational cycle is completed when all beds have been emptied. When some of the beds are out of operation or receive a reduced sludge volume due to emptying or commissioning, the quota must be raised for the other beds. Therefore, when dimensioning a new STRB, the capacity of the individual beds during the emptying period should be taken into consideration. Some of the older Danish systems are now running with at least one basin out of operation each year.

5.6.3 Pilot systems

Before the design, dimensioning and construction of a system, it is important to determine the sludge quality; in particular, its dewatering characteristics and the ratio between organic and inorganic solids (phase 1). The main goal is to test whether the sludge would be suitable for further treatment in a STRB. Other goals in phase 1 are to find out the following:

- Is it possible to treat and drain the sludge in a STRB system?
- How will the sludge behave (dry/crack up) in a pilot bed?
- Is it possible to get the vegetation grow in the sludge?
- What will be the dewatering efficiency of the sludge (L/s/m^2)?
- Are there any adverse or undesirable effects on reed health/growth rates?

The main goal of the next phase is to test and ascertain the criteria for the dimensioning, number of basins and operation of a full-scale system. In this phase, different loading rates and loading/resting days are tested to define the following:

- What sludge loading (kg DS/m^2/yr) can the pilot bed treat?
- How big a load (m^3, kg DS) can a pilot bed receive?
- How many loads can one pilot bed receive daily and in one quota?
- What is the optimum load and rest program in relation to sludge quality?
- What is the measured dewatering efficiency (L/s/m^2)?

5.6.4 Design and dimensioning

The sludge quality and sludge capacity requirements are very important parameters for planning and dimensioning of a new STRB. Moreover, in order to ensure a sufficient resting period, the number of basins to be established typically should be six to eight or more, depending on the climate, the total annual sludge production and quality. Most STRBs are dimensioned for an area load of 30–60 kg DS/m^2/yr (depending on the climate and the sludge quality), but higher loads (up to 90 kg DS/m^2/yr) have also been successfully applied under warmer climates for aerobic sludge (Nielsen *et al.*, 2018; Stefanakis & Tsihrintzis, 2012c). For sludge with a large proportion of fat, oil, and/or organic matter or a low sludge age (<20 days), the recommended area load should be reduced (Nielsen, 2003; 2011). STRBs essentially comprise a series of gravel/sand basins that are planted with wetland plants/reeds and typically 6–12 beds, even up to 24 beds or more depending on the climate. It is very important to consider the climatic conditions as a part of the STRB dimensioning and operation. For this, it is recommended to measure sludge dewaterability and run a pilot STRB before the design and construction of the full-scale STRB.

5.6.5 Climate

The results from a pilot plan in Wacol (SE Queensland, Australia) demonstrated very high sludge dewatering capability and challenges from the implementation of the STRB in a hot, subtropical climate (Nielsen *et al.*, 2018). The climate is characterized by mild, dry winters with mean daily temperatures in the range of 20–25°C, lows of 10°C, and hot and humid summer conditions with daily temperatures above 30°C and even exceeding 40°C several times a year. The hot climate enabled faster and better drying of the sludge residue due to air drying and higher plant evapotranspiration rates allowed for increased total annual loading rates. A sustainable loading rate of 60–70 kg $DS/m^2/yr$ was established for the sludge quality from Wacol WWTP. A similar loading rate of 75 kg $DS/m^2/yr$ was also determined for a pilot STRB operating under the Mediterranean climate, with mild winters and warm summers (Stefanakis & Tsihrintzis, 2012c). However, the climate also imposes the challenge of maintaining plant health in hot and dry conditions. Extended resting periods without loadings lead to water stress conditions for the plants (Nielsen *et al.*, 2018; Stefanakis & Tsihrintzis, 2011). These indicate that water stress may need to be managed more actively in the first few years of operations, when a layer of sludge residue is building up. Once the beds are covered with a sufficient layer of sludge residue of approximately 20–25 cm, it is expected that this layer will retain more moisture than the filter layers, enabling the plants to use capillary-bound water in the sludge residue layer in periods of water depletion between the loading periods. Two main solutions were found, an adapted loading program and a design response to the problem where the basins were designed to fill up with water within the filter below the drying sludge residue (Nielsen *et al.*, 2018). The fast and more efficient drying of the sludge residue allows for shorter resting periods between basin rotations, which can respectively allow for a reduced required number of basins than for systems in cooler climates and, therefore, lower capital cost (Table 5.8).

5.6.6 Main factors affecting treatment performance

In Denmark, Germany, Sweden, France and other countries in Europe the design and dimensioning of STRB systems has been extremely variable during the last 20 years, even if they were treating the same sludge type: the number of basins in different systems was between 1 and 24 basins, basin areas between less than 100 m^2 and over 3,000 m^2 and the area load from 30 to over 100 kg $DS/m^2/year$. In spite of this variety, all STRBs were more or less designed and dimensioned for 10 years of operation. The overall experience showed that a great deal of the systems had run into operation problems with a low efficiency, i.e., a low dry solid content in the sludge residue. The problems were observed in the vegetation, the low dewatering degree and the fast development of the wet anaerobic residual sludge layer; vegetation becomes stressed, wilted and even non-vegetated areas occurred. The operation problems could be attributed to failure in one or all of the four main categories:

Category 1: Sludge Quality
Category 2: Design and dimensioning
Category 3: Construction
Category 4: Operation

Very often the design and dimensioning were not related to sludge quality. The process areas were too small resulting in high areal load and the system had a small number of basins. The filters were constructed with media with low or none capillarity and finally they were operated with a wrong ratio between loading and resting periods (Nielsen, 2011). On top of that, the sludge quality is a very important parameter, which is often neglected. Even systems with a large number of basins, low area load and long resting periods have been insufficient to ensure healthy reeds, proper dewatering and mineralisation of the sludge, if the sludge quality is not suitable for treatment in STRB. Higher concentrations of organic solids and/or fat and oil result in lower dry solid content and more pronounced anaerobic conditions in the sludge residue (Nielsen, 2011; Stefanakis *et al.*, 2014). Dimensioning, number of basins and system operation must be based on sludge quality analysis (Table 5.8), in particular the dewatering characteristics of the sludge and the ratio between organic and inorganic solids (Nielsen, 2011).

5.7 AERATED WETLANDS

Scott Wallace[1], Dion van Oirschot[2] and Alexandros Stefanakis[3]

[1]*Naturally Wallace Consulting, P.O. Box 37, 126 2nd Street S, Stillwater, Minnesota 55082, USA*
[2]*Rietland bvba, Van Aertselaerstraat 70, Minderhout 2322, Belgium*
[3]*Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman*

5.7.1 Introduction

Aerated wetlands are saturated, HF or VF wetlands that rely on a mechanical system (an air pump connected to a subsurface network of air distribution pipes) to introduce air bubbles into the water being treated (Wallace, 2001). The use of an artificial aeration system dramatically increases the oxygen transfer rate compared to passive wetlands (Table 5.9), enabling improved performance for treatment reactions that require oxygen (such as nitrification) or occur more rapidly under aerobic conditions. The aeration system can also be operated intermittently to promote nitrification/denitrification (van Oirschot & Wallace 2014). A simple schematic and description of the process was covered recently by Dotro *et al.* (2017) and the improved treatment performance through aeration of pilot scale systems fitted to the first-order kinetic (P–k–C^*) model by Nivala *et al.* (2019b).

5.7.2 Design considerations

Standard HF and VF wetland systems rely on passive diffusion of oxygen into the water column. This is a very slow process in saturated-flow wetlands (HF and FWS) and passively improved upon in unsaturated-flow wetlands (VF and French VF wetlands). Mechanically aerating the system allows the amount of air introduced to be independent of the surface area of the wetland, allowing aerated systems to be loaded up to maximum clogging limits, which greatly reduces the area required and associated

Table 5.9 Estimated oxygen consumption in g $O_2/m^2/d$ for different TW types (adapted from Wallace, 2014)

TW Type	Estimated O_2 Consumption	Notes
HF wetland[1]	6.3	50th percentile values from Kadlec and Wallace (2009) assuming aerobic BOD removal and conventional nitrification.
FWS wetland[1]	1.47	
VF wetland (unsaturated)[1]	24.7	
French VF wetland (1st stage)[2]	40–60	Data from France indicates that the first stage of a French VF wetland can sustainable operate at roughly 1.5 m^2/PE
Aerated (HF and VF)	250	Mechanically aerated wetlands can achieve higher oxygen transfer rates, but 250 g/m^2-d is considered an upper $CBOD_5$ limit for clogging; most sustainable designs operate at <100 g/m^2-d (Wallace, 2014).

Notes:
[1]50th percentile values from *Treatment Wetlands, Second Edition* (Kadlec & Wallace, 2009); assuming aerobic BOD removal and conventional nitrification.
[2]Data from France indicates that the first stage "French VF" process can sustainably operate at roughly 1.5 m^2/PE (Molle *et al.*, 2005).

Table 5.10 Typical design parameters for aerated wetlands.

Design Parameter	Recommendation	References
Pre-treatment	Primary treatment common (CSO systems typically do not have pre-treatment)	DWA-A262E (2017)
Influent loading (inlet cross-sectional area)	$<$250 g $CBOD_5/m^2/d$ (maximum)* $\leq$100 g $CBOD_5/m^2/d$ (recommended)	Wallace (2014)
Specific area	$\geq$0.5 m^2/PE $\leq$80 $g/m^2/d$ $CBOD_5$	Stefanakis and Prigent (2018)
Influent distribution	$\leq$50 m^2 per feed point (unless bed is permanently flooded)	Dotro *et al.* (2017)
Air flow rate	$\geq$0.6 $m^3/m^2/h$	DWA-A262E (2017)
Air distribution	30 cm $\times$ 30 cm	DWA-A262E (2017)
Media size	8–16 mm	DWA-A262E (2017)
Treatment kinetics	pilot testing	Nivala *et al.* (2019b)

*Mechanically aerated wetlands can achieve higher oxygen transfer rates, but 250 g/m^2-d is considered an upper $CBOD_5$ limit for clogging (Wallace and Knight, 2006); most sustainable designs operate at $<$100 g/m^2-d (Wallace, 2014).

capital cost. Aerated wetlands are generally dimensioned based on clogging, hydraulics, uniform air distribution, and first-order kinetics (Table 5.10).

Aeration of wetlands follows standard wastewater aeration design practices in terms of calculating oxygen demands and air flows based on actual/standard oxygen transfer rates (AOTR/SOTR) protocols (Metcalf and Eddy Inc., 2003). However, the hydrodynamic mixing of the water column induced by aeration is greatly reduced in gravel-bed systems compared to ponds or tanks (Wallace, 2014). This requires that the air distribution in wetland beds be very uniform (Wallace, 2014). Most air diffusers in mechanical treatment systems are high-flow/small-area devices that are poorly suited to uniform distribution, and successful wetland aeration designs have been based on alternative pipes or tubing that can distribute air uniformly. This generally requires empirical testing to determine the air flow vs. air pressure relationship for the product(s) under consideration.

Gravel media used in the system must have pore spaces large enough to allow the passage of air bubbles. Sand is too fine for aerated systems as the air collects and "blows out" in just a few locations. Air bubbles moving through the gravel media can combine and coalesce into larger bubbles (reducing oxygen transfer), however air bubbles follow a tortuous path through the media, slowing their transit time (increasing oxygen transfer). As a result, wetland aeration systems typically demonstrate an oxygen transfer efficiency intermediate between fine-bubble and coarse-bubble diffusers (von Sperling & Chernicharo, 2005; Wallace *et al.*, 2007).

5.7.3 Potential design and operational issues

Since aerated wetlands are high-rate treatment processes, they are sometimes designed very close to clogging limits, especially for HF; if overloaded, they can clog and require resting or refurbishment like other types of treatment wetlands.

During construction, testing of the aeration system to verify proper air delivery is essential. Since the air distribution lines are buried at the bottom of the wetland bed, replacing/repairing air lines after construction is difficult.

Fouling of the air distribution lines has been reported in isolated cases due to iron precipitates forming at the air distribution orifices. Using acid (HCl) to clean fouled air lines has been reported to be a successful quick and low-cost method (van Oirschot & Wallace, 2014).

Although the selection of the appropriate blower for the air distribution network should be based on air requirements, they can sometimes be limited by the smallest available size that a client can accept (based on rigorous health and safety requirements). To illustrate, four systems in the UK used the same size of blower to provide aeration to different size tertiary and secondary systems, resulting in a specific power allocation ranging from 4 W/m^3 of wetland to 26 W/m^3 wetland (Butterworth *et al.,* 2016a). In systems that are over aerated, venting of the air has been necessary resulting in wasted energy and noise complaints from adjacent residents. To minimise this, the selection of the correct aeration equipment should be emphasized to the client.

Stress of plants in both passive and artificially aerated wetlands has been reported in the literature, with chlorosis (yellowing of the leaves) being most predominant (Weedon, 2014) and a downward gradient observed in plant height from inlet to outlet in highly aerobic systems. In an assessment of four full-scale systems, one of the systems struggled to establish the common reed (*Phragmites australis*) whilst its twin bed under equal conditions but without aeration thrived with the same plants (Butterworth *et al.,* 2016a). The other three artificially aerated systems reported normal plant growth. The difficulty experienced with plant establishment in some UK systems did not affect treatment performance. A side-by-side full-scale trial comparing reeds (*P. australis*) to reedmace (*Typha latifolia*) plantings showed both plant species exhibited signs of stress (chlorosis and stunted growth) when grown with artificial aeration. Further controlled trials proved reedmace is proportionally more affected by aeration than the common reed but its higher natural growth rate can offset the true impact of aeration on biomass production (Butterworth *et al.,* 2016b). Plant stress has been attributed to iron deficiency and/or toxicity in aerobic systems. The fact it happens on some systems but not all suggests complex interactions between the biogeochemical conditions in the wetland subsurface and the plants. To illustrate, from 27 aerated wetlands built with expanded clay aggregates as their main media (instead of gravel), there have been no reports of plant stress to date. Recent research suggests observed iron-induced stress in reeds could be related to the plant's genetic code, with an iron foliar spray currently being assessed as mitigation strategy (Ren *et al.,* 2018). In practice, plant species selection for artificially aerated wetlands is typically done by the designer based on previous experience, and a variety of native wetland plants have been used to date.

5.8 FILL-AND-DRAIN WETLANDS

Scott Wallace[1] and David Austin[2]

[1]Naturally Wallace Consulting, P.O. Box 37, 126 2nd Street S, Stillwater, Minnesota 55082, USA
[2]Jacobs, 1295 Northland Drive, Suite 290, Mendota Heights, Minnesota 55120, USA

5.8.1 Introduction

Fill-and-drain wetlands are subsurface-flow wetlands that rely on the alternating filling and draining of wetland cells to move water (during the fill cycle) and air (during the drain cycle) in and out of the wetland cell. These systems are alternately called *tidal flow* or *reciprocating* wetlands (Austin, 2006; Behrends, 1999). Fill-and-drain wetlands are a further development of the contact bed systems developed in late 1800s (Kinnicutt *et al.,* 1910).

In the simplest configuration, a cell is filled using the influent flow and then drained so the next "batch" of water is then treated. Alternate configurations use recirculation so that the fill-and-drain cycle frequency can be adjusted independently of the influent flow rate.

Fill-and-drain wetlands are of interest because through the sequence of filling and draining, they cycle through aerobic and anoxic/anaerobic phases automatically. This makes them especially useful when removal of total nitrogen (TN) is a goal through nitrification/denitrification.

Fill-and-drain wetlands undergo cyclic changes in redox potential, ranging from aerobic (draining + empty phase) to anoxic/anaerobic (filling + full phases). These cycling changes result in distinct treatment mechanisms operating in different phases of the treatment cycle:

- *"Empty" Phase.* Air, having been drawn into the bed during the draining phase, allows rapid oxygenation of biofilms (Behrends *et al.*, 2001). Organic compounds having been previously adsorbed into biofilms are consumed by microorganisms under aerobic conditions, and positively charged ammonium ions (NH_4^+) are converted to negatively charged NO_3^- ions. Once the food supply is exhausted, further microbial activity results in the endogenous respiration, reducing the occurrence of clogging.
- *"Filling" Phase*. As the cell is filled, air is forced out of the system. Water enters first at the bottom of the treatment cell, and thus has the longest contact time. Chemical transformations of the "full" phase begin to occur.
- *"Full" Phase*. As the pore spaces fill with wastewater, oxygen is consumed and the redox potential decreases. Nitrate ions (NO_3^-), formed from previously oxidized NH_4^+, diffuse out of the biofilms into the bulk liquid. The presence of NH_4^+ (from the influent wastewater) and NO_3^- creates conditions suitable for alternate nitrogen transformations such as anammox. NO_3^- can also serve as an oxygen supply for degradation of organic matter and for conventional denitrification; approximately 80% removal of total nitrogen (TN) has been observed (Austin *et al.*, 2003).

  Positively charged NH_4^+ ions in the bulk liquid diffuse into the biofilms. This diffusion process and the overall total adsorption capacity of the bed is enhanced by the cation exchange capacity (CEC) of the bed media (Austin, 2006). Organic compounds are adsorbed into the biofilms, and this process is relatively rapid, taking approximately 5 minutes (Kinnicutt *et al.,* 1910). When NO_3^- and dissolved oxygen are fully consumed, biodegradation of organic compounds can continue under anaerobic conditions.

- *"Draining" Phase*. Water is released from the bed. Rapid drainage times of 30 minutes or less are recommended, as this aids in drawing air into the bed (Dunbar, 1908). Chemical transformations of the "empty" phase begin to occur as the bed is drained, and the cycle begins anew.

5.8.2 Design considerations

Fill-and-drain wetlands are generally dimensioned based on clogging, hydraulics, and the number of fill-and-drain cycles per day, as summarized in Table 5.11. Flow is typically rotated through multiple beds in parallel or series, often using internal pumping to achieve the desired number of fill-and-drain cycles.

Once the wetland cells are dimensioned to avoid clogging, the most important design parameter becomes the number of fill-and-drain cycles per day, as this relates to oxygen transfer (Table 5.12). In many designs, this is related to a "rule of thumb" oxygen consumption rate of approximately 7–10 g O_2/m^3 cycle (Wallace, 2014), with the number of cycles per day determined by the total oxygen demand (carbon + nitrogen) applied to the system. This "rule of thumb" is commonly used because it is simple, but it does not take into account the fact that oxygen is not limited during the "empty" phase, so ammonia removal is actually a function of the amount of ammonia adsorbed by the bed during the "full" phase.

Ammonia removal is related to the ammonia exchange capacity (AEC) of the bed materials, which determines the total amount of ammonia adsorbed during the "full" phase. The AEC is related to the presence of biofilms on the bed media and the cation exchange capacity (CEC) (ASTM D7503-18, 2018) of the material making up the bed media. Standard laboratory procedures for measuring AEC have yet to be developed, and designers typically devise tests specific to the project under consideration. However, ammonia removal in fill-and-drain wetlands clearly improves when materials with a high CEC are utilized (Austin, 2006). Fill-and-drain wetlands with very high rates of ammonia removal have been designed and constructed based on AEC methods.

Overall performance (inlet vs. outlet) of fill-and-drain systems can be described using first-order kinetic rate coefficients (k) (Nivala *et al.*, 2019b), as summarized in Table 5.13. However, the diffusion processes

Table 5.11 Typical design parameters for fill-and-drain wetlands.

Design Parameter	Recommendation	References
Pretreatment	Primary treatment required	Kinnicutt *et al.* (1910)
Influent loading (inlet cross-sectional area)		
BOD_5	$\leq$100 BOD_5 $g/m^2/d$	Wallace (2014)
TSS	$\leq$100 TSS $g/m^2/d$	
Influent distribution	$\leq$50 m^2 per feed point	Dotro *et al.* (2017)
Drainage system	$\leq$30 min to drain bed (generally by siphon)	Barwise (1899)
Fill-and-Drain cycles	6–24 per day (6–12 per day common)	Dotro *et al.* (2017) Kinnicutt *et al.* (1910), Austin *et al.* (2003)
Media size	8–16 mm	Kinnicutt *et al.* (1910), Nivala *et al.* (2014)
Number of beds	2–8	Nivala *et al.* (2013a); Austin *et al.* (2003)
Treatment kinetics	Table 5.13 or pilot testing	Nivala *et al.* (2019b)

Table 5.12 Estimated oxygen transfer rates for passive and fill-and-drain wetlands (Wallace, 2014).

Wetland Type	Estimated Oxygen Consumption (g $O_2/m^2/d$)
HF[1]	6.3
FWS[1]	1.47
VF (unsaturated)[1]	24.7
French VF (1st stage)[2]	40–60
Fill-and-Drain[3]	168–240

[1]50th percentile values from Kadlec and Wallace (2009) assuming aerobic BOD removal and conventional nitrification.
[2]Data from France indicates that the first stage of French VF wetlands can sustainably operate at roughly 1.5 m^2/PE (Molle *et al.*, 2005).
[3]Roughly estimated at 7–10 g O_2/m^3 per fill-and-drain cycle; at up to 24 cycles per day and a 1 m bed depth (Wallace, 2014). However, this also depends on the cation exchange capacity (CEC) of the media.

that occur during each phase of the treatment cycle appear to be rapid and occur more quickly than the beds can be physically filled and drained. As a result, each individual phase of the fill-and-drain treatment cycle have not yet been described using kinetics.

5.8.3 Potential design and operational issues

Loading fill-and-drain wetlands above recommended limits (Table 5.11) can result in clogging of the beds due to excess production of microbial biomass. Fill-and-drain wetlands are normally designed to receive primary-treated wastewater so that solids loadings on the beds are minimized. If primary treatment is problematic or is not provided, coarser bed materials (>75 mm) are required in the first treatment stage (Kinnicutt *et al.,* 1910). The use of coarser bed materials to reduce the potential for clogging also lowers treatment performance, so more than one treatment stage is employed (Barwise, 1899; Dunbar, 1908), with the first stage essentially acting as a roughing filter.

Fill-and-drain wetlands operate with a variable water level, and the gravel used does not have the same capillary action as fine-grained soils. This can be an issue during vegetation establishment, especially in arid climates. When plants are fully established, the root systems will extend throughout the bed down to the minimum water level. When plant root systems are still shallow, they can lose contact with the water,

Table 5.13 *P–k–C** model fit parameters for passive and fill-and-drain wetland systems at Langenreichenbach, Germany (Nivala *et al.*, 2019b).

Wetland Type	$CBOD_5$			NH_4-N			TN		
	P	C* mg/L	k_{A20} m/yr	P	C* mg/L	k_{A20} m/yr	P	C* mg/l	k_{A20} m/yr
HF	2.5	14.6	35	5	19	2.9	2.5	19.1	3.9
VF	2	0.6	315	2	1.5	176	2	11.5	40
Fill-and-Drain	2	0.3	672	2	0.1	450	2	4.4	123

increasing the risk of drought stress. This may require temporary irrigation systems during the plant establishment phase.

Treatment in the system is dependent on the number of fill-and-drain cycles per day. Designs that depend only on the influent flow rate to regulate the fill-and-drain cycle can have very slow cycling during low flows, consequently many designs use flow recirculation to increase the cycle frequency. Generally, it is best to design the system to support the maximum number of cycles desired, as it is much easier to slow down the cycle frequency than it is to speed it up.

5.9 FLOATING TREATMENT WETLANDS

Tom Headley[1] and Katharina Tondera[2]

[1]Wetland & Ecological Treatment Systems Ltd, 84 Melbourne Street, East Maitland, NSW 2320, Australia

[2]IMT Atlantique, GEPA, UBL, F-44307 Nantes, France

5.9.1 Introduction

Floating Treatment Wetlands consist of a pond or basin containing emergent macrophytes growing on a floating mat or raft, with the roots and associated biofilms hanging in the water column beneath (Figure 5.5). Water receives treatment as it passes through this hanging root-mat via biological, chemical and physical processes.

5.9.2 Overview of existing design guidelines

While some of the earliest full-scale deployments of Floating TWs for wastewater treatment were conducted in the late 1990s (see for example the acid mine drainage applications of Smith & Kalin, 2002), their broader application was relatively limited until the past 5–10 years. The recent and rapid development of the technology has also been across a broad range of applications, including urban stormwater, sewage effluent, eutrophic lakes and streams, airport de-icing runoff and various industrial applications. Thus, there is currently a lack of consistent and rigorous design guidelines for Floating Treatment Wetlands. It is fair to say that our understanding of the key treatment processes and functions that occur in Floating TWs is still in its relative infancy, albeit progressing rapidly with performance data from a growing number of studies each year. Pavlineri *et al.* (2017) provide a review of the performance data from published Floating TW studies. Based on results at that time, Headley and Tanner (2012) calibrated first-order areal removal rates (k) for BOD_5, NH_4-N, TN and TP within the P–k–C^* model structure of Kadlec and Wallace (2009). However, much of the available data is still from pilot or lab-scale systems

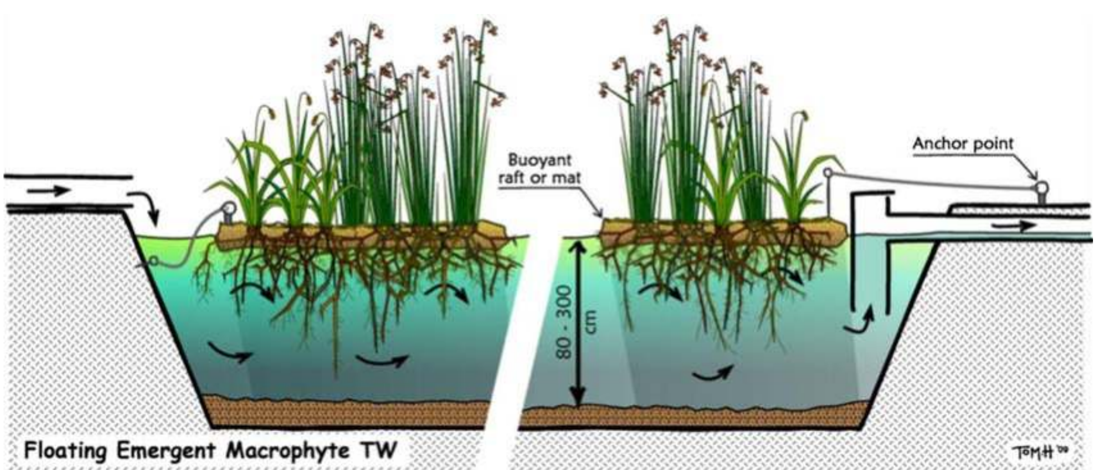

Figure 5.5 Schematic section through a typical Floating Treatment Wetland. Reprinted with permission from Headley and Tanner (2012).

and needs to be verified by long-term studies of full-scale systems. Arguments can be made as to whether it is more appropriate to take an areal or volumetric first-order approach to the sizing. However, for many contaminants (e.g. BOD, nitrogen species, fine suspended particles and associated metals) the rate of their removal will be fundamentally governed by the amount of hanging root-mat and associated biofilms that are present in the Floating TW. Given that the quantity of root-mat will be primarily a function of the area of floating mat, while the density of roots will decrease with depth, we advocate that it is more rational to consider the sizing of Floating TWs on an areal rather than volumetric basis.

5.9.3 Main factors affecting treatment performance

Aside from the usual universal biogeochemical processes which occur in other treatment wetland designs, the main factors affecting the treatment performance in Floating TWs are

- *Wetland vegetation*. especially the depth and extent of the hanging root-mat that develops beneath the Floating TW, which ultimately determines the specific surface area (m^2 root-mat per m^3 of water volume) of biofilm available for biochemical treatment processes and for trapping fine suspended particles.
- *Percentage areal coverage of the pond with floating mat*. This directly affects the amount of root-mat and associated attached-growth biofilm in the Floating TW reactor, versus suspended growth processes in open water. Also, dissolved oxygen tends to be consumed under a Floating TW, while the rate of passive oxygen transfer will be higher in open water areas due to diffusion, wind-induced turbulence and algal photosynthesis in the presence of sunlight.
- *Hydraulic Residence Time*. A minimum contact time may be required for some treatment processes.
- *Water depth and variability*. If the design water depth exceeds the depth of the hanging root-mat, then a certain portion of the flow will bypass beneath the root-zone with limited exposure to treatment. If the water depth is too shallow, the residence time may be compromised and there is a risk that the roots will embed in the benthic substrate causing operational problems, especially if the water level increases at some time.
- *Hydraulic efficiency and the configuration of the floating mats*. A given total area of floating mats on a Floating TW pond can be arranged in a multitude of ways to achieve the desired ratio of floating cover to open water (e.g. Walker *et al.,* 2017; Winston *et al.*, 2013). The configuration of the individual floating units can be used to manipulate the overall hydraulic efficiency of the Floating TW, with the aim of minimising the risk of short-circuiting paths, dead-zones and maximising the interaction between water and the hanging root-mats. Open water without root-mats will provide less resistance to areas with a dense root-mat beneath the floating rafts. Thus, water will preferentially flow through these open water zones. If possible, transverse bands of floating mats with complete connectivity from one side of the basin to the other, oriented perpendicular to the flow direction, are preferable.
- *Media used in the floating mat*. In some cases, media with specific properties have been trialled to promote certain processes, such as adsorption of compounds. However, the effectiveness and long-term feasibility of such approaches is yet to be verified at full-scale. In other cases, selection of inappropriate planting media in the Floating TW can leach nutrients or organic matter into the water column, leading to eutrophication or oxygen crash in the water column beneath. Care should also be taken to ensure that any plastic products used in floating mats are stable against UV degradation and do not degrade to release microplastic particles into the water body.
- *Aeration of the water column*. In some situations, where the oxygen demand of the water is relatively high, active aeration has been employed to overcome oxygen transfer limitations and enhance the efficiency of oxygen-consuming treatment processes.

5.9.4 Specific design considerations

Aside from the questions around sizing of the Floating TW to achieve given water quality improvement targets, there are several practical aspects to be considered in the design:

- *Techniques and materials for constructing the floating rafts*. Several options exist, with varying degrees of compromise between cost, longevity and convenience of deployment. Headley and Tanner (2012) provide a comprehensive overview of the main approaches adopted to date for construction of the floating structures.
- *Anchoring the floating structures* to the edges and/or bottom of the basin. The floating rafts should be secured sufficiently to prevent that they drift excessively with wind or wave action. Sufficient allowance should be made to for rising and falling of the floating rafts with changing water level in ponds with variable water depth, such as stormwater systems.
- *Minimum and maximum allowable water depths*. Careful consideration needs to be given in the design to the operating water depths of the Floating TW system. It is recommended that the minimum water depth be greater than the expected depth to which most of the plant roots develop, to avoid the roots imbedding in the benthic sediments or that the root-mat becomes damaged at low water levels. The design of outlet water control structures is often key in ensuring the desired water depths are achieved. In some cases, it may be important to consider design allowances to ensure a permanent pool and depth of water can be maintained during low or no-flow conditions.
- *Planting media*. Use of a planting substrate may be necessary to establish the plants in the floating rafts. Media that are lightweight, low in nutrients and will not impose a high oxygen demand when saturated with water, while providing a good substrate for root development are preferred. Coir fibre or peat moss materials are suitable options. In some cases, plants are established without growth media.
- *Hydraulic design*. Consideration should be given to the dimensioning of the system (length, width and depth) and configuration of the floating rafts from the perspective of flow velocities to avoid scouring of sediments or biofilms attached to the hanging root-mats.

5.9.5 Considerations for the start-up phase

Establishment of the wetland vegetation in Floating TWs can be approached in two ways:

- *In situ planting and establishment*. the rafts are planted while floating on the pond;
- *Onshore planting and establishment*. the rafts are planted out of the water (e.g., on the shore or at a remote location) and transferred onto the Floating TW at some stage thereafter, often when the plants have at least partially established.

The benefits of *in situ* planting are that the Floating TW rafts do not need to be relocated once the vegetation is established, with the risk of damaging the stems and roots in transport. It can also be physically challenging to transfer large floating rafts from the shore into a pond with the added weight of vegetation and waterlogged media. However, it can be difficult to control birds and conduct weed maintenance during plant establishment if the Floating TW rafts are floating on the pond. In this regard, the early care and maintenance of the vegetation can be easier with onshore planting, with the added benefit that the Floating TW commissioning and start-up period can be minimised if rafts are deployed already with established vegetation.

5.10 WILLOW SYSTEMS

Darja Istenič[1], Tjaša Griessler Bulc[1] and Carlos Arias[2]

[1]University of Ljubljana, Faculty of Health Sciences, Zdravstvena pot 5, 1000 Ljubljana, Slovenia
[2]Aarhus University, Department of Bioscience – Aquatic Biology, Ole Worms Alle 1, BLDG 1135, 8000 Aarhus C, Denmark

5.10.1 Introduction to willow systems and existing design guidelines

Willow systems are TWs dominated by willows. They are currently in use for *onsite* wastewater treatment and reuse in Scandinavian and Baltic countries, Ireland, England, China and Poland, and pilot systems are operating in France, Greece, Spain and Slovenia. Willow systems are mostly designed to treat all the influent water though evapotranspiration, so there is no outflow from the system. They are also called evapotranspirative systems as they combine two separate processes removing water from the soil surface by evaporation and from plants via transpiration.

Additionally, there are also willow systems that combine seepage with evapotranspiration or that are designed as flow-through systems that produce some outflow, which is used for the recovery of resources, such as nutrients and reclaimed water for irrigation.

National design guidelines for willow systems are only available in Denmark (Miljøstyrelsen, 2003a, b). The Guidelines comprise systems of up to 30 PE in two construction and operation versions:

- *Zero-discharge systems*. These systems are specifically designed and established to evapotranspire all the influent water (both wastewater and precipitation). Therefore, the bed must be lined using impermeable foil to prevent infiltration of water to groundwater. Since all the water is evapotranspired, these systems are intended to be built in locations with high water-quality discharge standards, where no effluent is an option. The treatment area demanded for such system will be the balance between the influent water and the plant's potential evapotranspiration in the geographical location.
- *Systems with infiltration*. For these systems the bed will not be lined, so some infiltration is possible. Such systems are intended for the locations with clayish soils where natural infiltration is already low. Calculating the area needed will also include the soil infiltration capacity.

According to the Danish national guidelines the surface area of the system depends on local climatic conditions and the amount of wastewater to be treated. In Danish climatic conditions, a single household willow system covers between 200 and 300 m^2. When properly dimensioned, willows should evapotranspire all the inflow wastewater and rainfall (Gregersen & Brix, 2001).

Willow systems enable wastewater treatment, evaporation of water and recycling of the nutrients through the willow biomass. They are most appropriate for the sites where standards for wastewater discharge are strict and where soil infiltration is not possible. In such areas, other treatment technologies like compact wastewater treatment plants or TWs may not reach the desired outflow concentrations, or the upgrading of the technology to meet the discharge limits is economically unfeasible.

Willow system produce a significant amount of biomass that can be used for energy purposes (see Chapter 4.7 Biomass production); therefore, the construction of willow systems should go along with establishment of a chain of biomass processing, combustion and end users of the produced heat.

5.10.2 Main factors affecting dimensioning and performance

In general, the amount of the wastewater that can be loaded into the system is equal to the difference between evapotranspiration of the system and precipitation; however not all falling precipitation enters the system on a yearly basis since a significant part of precipitation is captured in the tree canopies or system's surface from where it evaporates back to the atmosphere or is blown by the wind to adjacent areas (Istenič *et al.*, 2018). While evapotranspiration and precipitation vary throughout the year, the loading with wastewater is constant. According to the amount of wastewater to be treated per year, a net area of the system can be calculated (Brix & Arias, 2011). Existing willow systems have been designed in a way that the water loss in the system is at least twice the potential evapotranspiration. Potential evapotranspiration (ET_P) is calculated from local meteorological data and depends on the plant species. Namely, the ET_P is a product of reference evapotranspiration (ET_0) and crop coefficient (k_c). ET_0 data can be obtained from the nearest climate station and are given for a reference crop (short grass) (Penman–Monteith equation).

When designing a willow system its orientation on site is of crucial importance. To increase evapotranspiration, willow systems should be long and narrow (up to 50 m), placed in an open landscape and perpendicular to the prevailing wind direction, sheltered and shaded locations must be avoided (not surrounded by tall vegetation and/or buildings). When dimensioned correctly, the so-called clothesline and oasis effect significantly increase the evapotranspiration. Clothesline effect is caused by the fact that willow trees in the system are higher than surrounding vegetation acting as a line of drying clothes – the evapotranspiration is increased due to broadsiding of wind horizontally into the taller vegetation. The wind brings heat from the surroundings and increases the air turbulence in the canopy that enables transport of vapour away from the canopy. Similarly, the oasis effect is caused by the difference in temperature of the system compared to the surroundings. Namely, the vegetation has higher soil water availability than the surroundings. Solar radiation and heat provide the energy needed for transformation of water from liquid to vapour state. Due to this endothermic process, the air in the system is cooled causing a difference in air pressure and as a result, the warmer air from the surrounding blows into the system, increasing evapotranspiration. In this way, willow systems also affect the local microclimate.

The willow system must also be positioned to enable access for all the machinery during construction as well as for harvesting.

5.10.3 Operation and maintenance

The willows are harvested in one- to three-year cycles to maintain healthy vegetation with high biomass production. To maintain high evapotranspiration, not all the system is harvested at every cycle but only half or a third (depending on the length of the cycle), which in case of a three-year cycle results in three sections of the bed with harvested, one-year and two-year plants (Gregersen & Brix, 2001). To reach higher biomass production, the number of shoots has to be as high as possible; therefore, the first year all shoots are cut in order to stimulate their propagation. The harvest should be done during the dormant period.

After a certain period of operation of the evapotranspirative system, the media may become saturated with nutrients. The nutrient-rich media can be used as compost or fertilizer in agriculture (Gregersen & Brix, 2001), which enables the return of nutrient to the food chain.

5.10.4 Specific design considerations

The willow system consists of a septic tank as defined for other TWs, pumping well, and soil bed planted with willows (Figure 5.6). The bed is 1.8 m deep, in case of zero-discharge system, lined with high-density polyethylene or equivalent membrane and filled with original soil from the site up to 1.5 m level. A 0.3 m basin dike protects the basin against intrusion of water from the surroundings and enables accumulation of

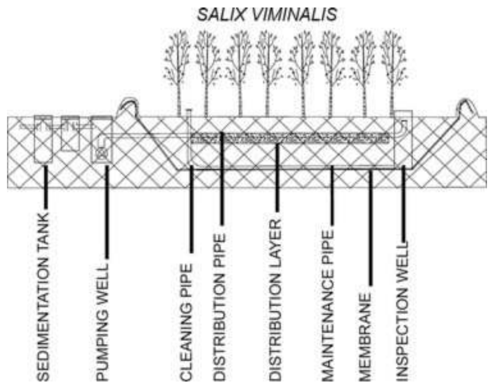

Figure 5.6 Cross section through a zero-discharge willow system. Reprinted with permission from Brix and Arias (2011).

the water on the surface during the winter or at high precipitation. The distribution pipes are placed on a 0.25 m thick and 0.20 m wide layer of 16–32 mm gravel or some other material, running in the middle of the bed through all its length to allow proper distribution of the influent wastewater. The distribution pipe must be covered with 0.6 m of soil or dug into the bed in order to prevent frost. The wastewater is pumped to the system according to the wastewater production.

Besides the distribution pipe, willow systems also include the maintenance pipe and inspection well. Maintenance pipe enables potential washing of the system with freshwater when the salinity is increased to the level to reduce the willow growth. The inspection well enables monitoring of water level in the bed and pumping out the water after the cleaning process.

Willow systems can be planted with species and clones of willows that are fast growing and have high biomass yield. Mainly selected clones of *Salix viminalis*, and *Salix alba* have been used (e.g. Curneen & Gill, 2014; Istenič *et al.*, 2017). Indigenous willows taken from nature are not appropriate due to lower biomass yield. Ideally different clones or varieties should be planted in parallel rows of a system to prevent spread of diseases though all the system. Willows are planted as 20–30 cm cuttings ideally in early spring after the last frost. Cuttings are gained from a year-old shoots during a dormant period (December–March in a temperate north latitude climate). If immediate plantation of cuttings is not possible, they should be stored at −2 to −4°C (can last for several weeks). Willows are planted in rows 1 m apart; the spacing along the rows is 0.5 m. Every three rows there is a wider gap of 1.5 m (Brix & Arias, 2011).

5.10.5 Considerations for the start-up phase

- The cuttings must be prepared correctly and planted at prescribed time of the year as described above.
- During the establishment year, weeds must be removed mechanically; willows must be controlled for general health and harmful pests must be eliminated.
- The winter after planting, willows must be cutback to 10 cm above ground which will induce multi-stem growth in the next season, resulting in a dense canopy that will significantly reduce the light penetrating to the ground and thus prevent weed growth.
- During the establishment year, evapotranspiration of the system is lower (according to Istenič *et al.*, 2018, half of the designed rate), thus it is recommended that the system is not fully loaded.

5.11 USE OF REACTIVE MEDIA FOR ENHANCED PHOSPHORUS REMOVAL

Gabriela Dotro[1], Bruce Jefferson[1], Margit Kõiv-Vainik[2], Carlos Arias[3] and Pascal Molle[4]

[1]Cranfield Water Science Institute, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK
[2]Institute of Ecology and Earth Sciences, University of Tartu, Vanemuise 46, 51003, Tartu, Estonia
[3]Aarhus University, Department of Bioscience – Aquatic Biology, Ole Worms Alle 1, BLDG 1135, 8000 Aarhus C, Denmark
[4]Department of Ecotechnologies, IRSTEA, Villeurbanne, France

5.11.1 Introduction

A common adaptation to both conventional HF and VF wetlands is the replacement of inert media (sand or gravel) by reactive media (e.g., steel slag, apatite), or the addition of a separate treatment step, to achieve sustained phosphorus removal. There are strong regulatory drivers for efficient removal (and recovery) of phosphorus from wastewater. During the last decades, a vast amount of research has been invested to tackle the problem (Dotro *et al.,* 2017). This section summarises the current knowledge and state of the art from collective experience on the design and operation of such systems from long-term studies (>2 years) at both pilot and full scale.

5.11.2 Overview of existing design guidelines

After decades of research on reactive media for P removal, the best available information on design and operation of reactive-media wetlands is based on long-term trials rather than national standards (Table 5.14). There is currently no literature reporting on full-scale operation of reactive-media wetlands reaching capacity and removal (or recovery) of the exhausted media. Lifetime predictions are based on retention capacities from various scales and operational conditions, with limited consistency between studies making it difficult to draw general design guidelines. On a recent report from three full-scale systems designed, built and operated with the same criteria in the UK the removal pattern was inconsistent among sites with no clear link to alkalinity or influent phosphorus concentration, the two commonly accepted key parameters to consider for the design of reactive-media systems for P removal (Fonseca, 2018).

5.11.3 Design considerations

The main factors affecting the treatment performance of reactive media wetlands are:

- *Reactive media type and size*. Phosphorus removal is mainly associated with the physical–chemical properties of the media. Chemical composition of the media is of importance, specifically its content of Ca, Al or Fe, three elements that can react with P under different environmental conditions. Media with small granulometry, higher porosity and larger specific surface area would be the best option. However, the smaller grain size is also associated with higher clogging risk (biological, physical and chemical clogging), low hydraulic conductivity and therefore, an optimal size, according to the special characteristics of the media and the expected water quality must be determined.
- *Reaction kinetics*. P retention kinetic has to be determined in real environment sampling water at different retention time within the filter. The use of simple models (i.e. P–k–C* or K–C*) is generally accurate enough to fit P concentration evolution within the media. As different

Table 5.14 Summary of existing design recommendations for reactive media.

Reference	Delgado *et al.* (2018)	Kõiv-Vainik *et al.* (2016)	de la Varga *et al.* (2017)	Fonseca (2018)
Target pollutant	o-PO_4	TP	TP	TP
Mode of operation	Saturated VF or HF	HF	VF	HF
Plants used	None or *Phragmites australis*	*NA*[1]	*NA*[1]	*Phragmites australis*
Influent wastewater range (mg/L)	5–15 (PO_4-P)	2–36 (PO_4) 4–36 (TP)	18–25 (TP)	4–7
Target effluent concentration (mg/L)	1.5–2.5	1–2 (TP)	1.5 (TP)	1–2
Media	Granulated apatite – Phosclean	Hydrated oil shale ash	Calcite based	Steel slag
Media size (mm)	2–6	4–16	4–8	10–14
Hydraulic residence time (hours)	6–48	144	48	24
Expected effluent pH	9–11 at the beginning then 7–8	8–13	7–9	10–12
Effluent pH management	Chemical acidification	Dilution (not solved)	Not needed	Blending (not solved)
Design life	5–6 years	Not determined	1 year	Not determined
Fate of material at breakthrough	Potential use as long-time release fertilizer	Potential use as slow release fertilizer (Kõiv *et al.*, 2012)	Fertilizer	Proposed as fertilizer but not currently done

[1]The systems in Denmark and Estonia are based on contactors complementing a wetland system rather than wetlands retrofitted with reactive media.

mechanisms can operate according to saturation state (adsorption–precipitation) and environmental conditions (pH, alkalinity), kinetics can evolve with time. The one measured in a commissioning period can differ from those after years of functioning. Designers should define kinetic evolution when long-term P retention is targeted.

- *Pollutants load to the reactive media*. All reactive filter materials are vulnerable to insufficient or lack of pre-treatment. As P retention relies on surface mechanisms, an excessive biomass growth will hinder access to the media surface. Therefore, it is recommended to locate the filter after effective biological treatment steps (i.e. tertiary treatment).
- *Alkalinity of influent wastewater*. When P retention mechanisms are linked to Ca–P bonds or precipitation, alkalinity of the influent can impact kinetics or the type of Ca–P retention (stable or not – competition with carbonates). In some cases, Ca addition can be necessary (by the use of calcite gabion) to increase alkalinity, pH and Ca concentration and favour P retention.
- *Temperature*. Some studies show that the seasonal variability in wastewater temperature affects the phosphorus removal efficiency in alkaline reactive media. In some cases, P removal efficiency

improved with increasing temperature, because this affected the rates of CaO-slag dissolution and Ca-phosphate precipitation (Barca *et al.*, 2013). However, the effect of the high temperatures can be opposite in case of higher influent organic pollutants content that can result with biofilm growth inside the media. The reaction kinetics must be defined at different temperatures prior to set up of the full-scale system to take into account possible seasonal variations in performance.

- *Hydraulic residence time*. Hydraulic residence time has to be set up, taking into account the porosity of the material, in accordance with outlet P concentration required and retention kinetic measured on the media. When long-term removal is targeted, kinetics with high saturation levels have to be used for design. Sorption capacity measured in batch tests decreases at real hydraulic residence time (Arias *et al.*, 2003). Removal performances will be higher at the commissioning period.

  The hydraulics of the filter (water distribution and collection) have to be carefully designed to avoid short-circuiting and dead zones that could impact on the efficiency of the filter. The residence time is calculated including the porosity of the media, which typically ranges from 0.35 to 0.5. For reactive media using calcium (e.g. steel slag, hydrated oil-shale ash, apatite), there is usually a direct link between HRT and effluent pH. Thus, the retention time must be carefully selected to avoid too high pH in the effluents (>9) as a result of excessive CaO-slag dissolution and rapid chemical saturation by secondary carbonate precipitates (Barca *et al.*, 2013; Liira *et al.*, 2009).
- *Necessity of pilot trials before full-scale application*. A pilot trial should be utilised before scaling up, testing the media with the target wastewater (i.e., no synthetics or surrogates) and operate it for at least a full year (preferably until media saturation with P) to enable results to be translated to full-scale systems. Particular consideration should be given to the hydraulics of the reactor on scaling up.

5.11.4 Potential operational issues

Secondary pollution

One of the main issues when using Ca-rich media is high pH of the effluent. One recent study dealing with high effluent pH of the slag filters shows that effluent neutralization with CO_2-enriched air from an upstream bioprocess could be a solution in some cases (Bove *et al.*, 2018). However, the addition of dosing strategies defeats one of the key benefits of using reactive media (i.e., no chemical dosing onsite). Other studies have suggested effluent dilution, polishing ponds, neutralising filters with acidic media (bark, peat, sand). Media-specific problems have also been identified on vanadium leaching from industrial by-products (steel slag; Fonseca, 2018), and chromium and radiation concerns from engineered media (apatite; Fonseca, 2018).

Role of vegetation

When alkaline reactive media are used in a separate treatment unit then the general recommendation is to avoid vegetation and any other biological activity inside of the filter. There have been mixed reports from planted reactive-media filters where plants (*Typha latifolia*) have established without issue or have struggled under similar pollutant loading conditions (Fonseca, 2018). However, because alkaline materials are vulnerable to air CO_2 (resulting in formation of Ca carbonates) the reactive filters are sometimes covered, with an insulation/cover media amenable to planting where root development can be contained above the reactive media.

Planning for maintenance and clogging management

In addition to the standard risks of clogging associated with the particular wetland configuration, chemical reactions within the media make clogging more likely to occur in these types of systems. Precipitates in alkaline media can result in cementing of media, which significantly limits porosity as the bed ages. Strategies to minimise this include isolating the media from air and choosing the right media size taking into consideration the removal mechanisms that will dominate in the system.

5.12 MULTI-STAGE WETLANDS

Anacleto Rizzo and Fabio Masi
Iridra Srl, via La Marmora 51, 50121, Florence, Italy

5.12.1 Overview

Multi-stage wetland systems (also known in literature as hybrid systems) can be defined as the combination of different TW types, in order to exploit the advantages of the different systems and to obtain better results in comparison with the use of a single stage. The definition is quite broad; indeed, every combination of HF, VF, or FWS wetlands can be considered a hybrid system. Due to the advantages and the provided removal processes, the hybrid systems are well suited for various applications, some of which are summarised in Table 5.15. A multi-stage approach can be valuable in case of particular wastewater to be treated. Two examples are the treatment of industrial wastewater with high influent organic loads (e.g., winery wastewater, see Chapter 4.11.5) and landfill leachates (see Chapter 4.10).

Strict guidelines on multi-stage systems are not available, due to diverse operating conditions and applications. Therefore, this section gives an overview of the possibilities using multi-stage wetland systems, thus allowing a proper consideration of hybrid instead of single-stage solutions when the use of TWs for wastewater treatment is under discussion. Moreover, this section provides suggestions for designers and practitioners to properly design multi-stage wetland systems as a function of different targets.

As a general observation, the design methods for sizing TWs which are based on datasets acquired at single-stage or single-typology treatment plants can be unsuitable for the design of "stages" where the performance in removing or retaining pollutants has to be decided in an often-tailored way for every specific realisation.

5.12.2 Nutrient removal

Multi-stage systems are often adopted when removal of nutrients is required.

Total nitrogen (TN) removal is a typical case in which the application of a hybrid TW leads to successful results (Gajewska *et al.*, 2015; Masi *et al.*, 2013). Indeed, the complete cycle of N removal is possible combining the efficient nitrification process of VF (aerobic conditions) with the denitrification promoted by either HF or FWS (anoxic/anaerobic conditions). The design of TN removal with hybrid wetlands can be done as follows:

- The VF nitrification stage needs to be sized for the desired effluent $NH4^+$-N concentration, according with methods and guidelines summarised in Chapter 4 of the *Treatment Wetlands* textbook (Dotro *et al.*, 2017), such as the oxygen balance method of Platzer (1999). It is important to fully nitrify the wastewater in order to provide an efficient denitrification in the next stage and/or meet legal discharge requirements on effluent $NH4^+$-N. The oxygen transfer rate for VF is reported in literature with a wide range, varying from 23 to 92 $gO_2/m^2/d$ (see Nivala *et al.*, 2013b). Therefore, it is better to be conservative in the selection of the oxygen transfer rate for the sizing of the VF for proper TN removal. In case of stringent limit for effluent $NH4^+$-N concentrations, a subsequent nitrifying stage can be added (e.g., VF + HF + VF) to ensure complete nitrification.
- The HF or FWS denitrification stage, or both (VF + HF; or VF + FWS; or VF + HF + FWS), can be designed according to well known kinetic formulations, which are summarised in Chapter 2 of the *Treatment Wetlands* textbook (Dotro *et al.*, 2017), such as the P–k–C* model of Kadlec and

Table 5.15 Examples of multi-stage systems for different applications.

Application	Multi-Stage Scheme	Size	Location	Reference
Municipal wastewater – Secondary treatment	HF + VF + HF + FWS	3,500 inhabitants	Dicomano (Italy)	Masi *et al.* (2013)
	HF + VF	190 pe	Sarbsk (Poland)	Gajewska *et al.* (2011)
	HF + VF + HF	150 pe	Wiklino (Poland)	Gajewska *et al.* (2011)
	HF + HF + VF + HF	600 pe	Darżlubie (Poland)	Gajewska *et al.* (2011)
	HF + VF + HF	500 inhabitants	Chorfech (Tunisia)	Masi *et al.* (2013)
	French VF (1st stage) + VF + FWS + Infiltration pond	1,000 pe	Castelluccio di Norcia (Italy)	Rizzo *et al.* (2018b)
Municipal wastewater – Tertiary treatment	HF + FWS	60,000 pe	Jesi (Italy)	Masi (2008)
Winery wastewater	HF + VF + HF + FWS + Pond + Sand filter	42 m^3/d	Bolgheri (Italy)	Masi *et al.* (2015a)
	French VF (1st stage) + HF + FWS + Sand filter	100 m^3/d	Castellina in Chianti (Italy)	Masi *et al.* (2015a)
	Hydrolytic upflow sludge bed + VF + HF	7 m^3/d	Pontevedra (Spain)	Serrano *et al.* (2011)
Landfill leachate	VF + HF	10 m^3/d	Liubljana (Slovenia)	Griessler Bulc (2006)
	Pond + FWS + Pond	473 m^3/d	Escambia County, FL (USA)	deBusk (1999)
Combined sewer overflow	VF + FWS	maximum treated flow 640 l/s	Gorla Maggiore (Italy)	Masi *et al.* (2017a)

Wallace (2009). Since the denitrification is planned to occur after a previous nitrifying stage, the risk of limited denitrification for carbon deficit needs to be checked. Kadlec and Wallace (2009) reports 0.7–1.1 g C/g N or a 5:1 C:N ratio for uninhibited denitrification in wetlands. The carbon deficit issue for denitrification is particularly relevant for influent wastewater showing strongly unbalanced C:N, such as swine wastewater (Masi *et al.*, 2017c). On the other hand, a carbon deficit can be overcome by nature-based solutions itself. Indeed, the more natural environment can provide endogenous C sources to fuel denitrification, for instance root exudation (Zhai *et al.*, 2013) and decayed plant biomass (Hang *et al.*, 2016). Indeed, FWS systems as tertiary stages for denitrification purposes (Masi, 2008) and multi-stage wetlands in general have shown the capability to efficiently remove TN with BOD_5/TN influent ratios (1.5–2.5) lower than other biological treatment methods (4–5) (Gajewska *et al.*, 2015). Alternatively, recirculation can be adopted in case of limited area, allowing effluent rich in nitrate and poor in carbon to be mixed in the first stage with an incoming wastewater rich in C (Saeed & Sun, 2012). For instance, Brix and Johansen (1999) propose a

HF+VF hybrid scheme with recirculation option to enhance TN removal. In case of recirculation, the HRT needs to be carefully checked, in order to avoid reducing too much the retention time due to the recirculation, compromising the denitrification process.
- Since no detailed guidelines are available for the design of multi-stage systems, it is always suggested to check the sizing with available similar systems and results reported in literature (e.g., Gajewska *et al.*, 2015; Masi & Martinuzzi, 2007; Vymazal, 2007).

Phosphorous is removed in TWs mainly via sorption processes (Kadlec & Wallace, 2009). Therefore, the adoption of multi-stage wetlands favours the removal of TP, since each stage removes TP as function of available sorption sites (i.e., sorption material). An option to improve the TP removal is to adopt a final "sacrificial" unplanted stage, which needs to be refurbished when the TP sorption capacity decays. The "sacrificial" filter need to be filled with highly adsorbing material, either natural or commercial (Kasprzyk *et al.*, 2018; Vohla *et al.*, 2011, see also Chapter 5.11 Use of reactive media for enhanced phosphorus removal). In the design phase of the sacrificial P filter, the reduction of adsorbing capability due to low-functioning HRTs must be considered (Arias *et al.*, 2003; Brix *et al.*, 2001). Alternatively, dosing of iron salts can enhance the TP precipitation (Dotro *et al.,* 2015; Kim *et al.*, 2015).

5.12.3 Enhanced disinfection with nature-based solutions and wastewater reuse

The use of tertiary treatment for disinfection purposes in multi-stage systems is a common practice, especially using FWS systems (Wu *et al.*, 2016). Typically, hybrid systems are used when treated wastewater reuse is a goal. The following points need to be checked in the design of multi-stage systems aimed to reuse treated wastewater:

- It is fundamental to check legislation limits for reuse, which can differ in different countries (Jeong *et al.*, 2016). If legislated limits are not available, an effluent safe pathogen water quality standard needs to be set by the designer as a function of differently aimed wastewater reuse, as suggested by WHO guidelines (Jeong *et al.*, 2016; Licciardello *et al.*, 2018).
- In case of particularly strict pathogen removal requirements, a cost–benefit analysis should be done to understand if it is better to oversize nature-based solutions only for water quality targets linked to disinfection or if it would be better to implement a technological disinfection unit (such as UV lamps, e.g. Álvarez *et al.*, 2017) as final disinfection stage. This is particularly true for applications in arid climates, in which the high evapotranspiration rate characteristic of extensive wetlands could reduce the amount of recovered water and decrease the dilution coefficient (i.e., higher effluent concentration with the same mass removal obtained in temperate countries).
- As function of the needed pathogen removal, it can be valuable to check the possibility to reduce the nitrogen removal to recover nutrients (Zurita & White, 2014), and build sustainable circular economy loops (Masi *et al.*, 2018).

5.12.4 Exploitation of different ecosystem services

Multi-stage design can be designed to exploit the additional ecosystem services provided by nature-based solutions (particularly FWS system) such as biodiversity increase (Hsu *et al.*, 2011), flood mitigation (Rizzo *et al.*, 2018a), and social benefits (Ghermandi & Fichtman, 2015; Liquete *et al.*, 2016). Therefore, hybrid TWs enhances the possibility to integrate ecosystem services in a multi-purpose design (e.g., Liquete *et al.*, 2016; Masi *et al.*, 2018).

From the point of view of urban runoff management, multi-stage wetlands have great potential to be integrated in green–blue infrastructures, following new city design concepts such as Sustainable Drainage Systems (SuDS – Woods-Ballard *et al.*, 2015), Water Sensitive Urban Design (WSUD – Wong *et al.*, 2009), Low Impact Development (LID – Dietz, 2007) or Sponge cities (Li *et al.*, 2017). To this aim, it is interesting to highlight that some nature-based solutions proposed in these approaches can coincide with TW classifications: bioretention systems (Liu *et al.*, 2014) can be considered an application of VF wetlands, while the wetlands reported in SuDS and LID manuals are what the TW experts call FWS wetlands. Following this analogy, it is interesting to observe how the "Treatment Chain" concept proposed by the SuDS Manual (Woods-Ballard *et al.*, 2015) is analogous with the concept of multi-stage wetlands, i.e., different stages in series, each one promoting different functions. An example is the use of bioretention systems for urban water quality improvement followed by a floodable wetland as detention basin, which is conceptually the same scheme (VF + FWS) proposed by Masi *et al.* (2017a) for CSO treatment. Therefore, a deep knowledge of the two "worlds", i.e., SuDS (or WSUD, sponge city, etc.) and TWs, is mandatory for a successful integration of multi-stage systems in green–blue infrastructures.

Chapter 6

Case studies

6.1 REPORTING TREATMENT WETLAND DATA

Günter Langergraber[1], Gabriela Dotro[2], Jaime Nivala[3] and Otto R Stein[4]

[1]Institute for Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Muthgasse 18, A-1190 Vienna, Austria
[2]Cranfield Water Science Institute, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK
[3]Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany
[4]Civil Engineering Department, Montana State University, Bozeman, MT, USA

Rational

When reporting data on TWs, a proper description of the wetland system under investigation is required. The experience from reviewing TW papers shows that a number of times, not all data related to the design of the system that are required to understand the system's functioning and/or all data-related sampling location/frequency and data evaluation are included. The following list should provide guidance on the minimum requirements of information on the wetland system that has to be provided.

Minimum information required on the TW system

- General information
  - Treatment capacity in PE, design flow and maximum flow to treatment
  - Dimensions of the system in m^2
  - Influent wastewater characterisation
  - Wetland plants used and harvesting frequency

  - Start of operation or length of operation before experimental data have been obtained.
  - Hydraulic loading rate (HLR) and pollutant loading rate.
- Specifically for VF beds
  - Depths and filter material of each layer of the VF bed
  - Characteristics of each filter material: granularity, d_{10}, U, etc.
  - Loading regime: intermittent or continuous
  - For intermittently loaded systems: loading interval, volume of a single dose, duration of the dose.
- Specifically for HF beds
  - Water level in relation to media depth
  - Flow distribution arrangement
  - Differentiate between plan area (length × width) and cross-sectional (depth × width) surface loading rates.

Reporting experimental data

- *Sampling*: description of location of sampling, sampling frequency and numbers of samples taken
- *Removal efficiencies* should be calculated from load data
- A minimum statistical evaluation of data is required.
- *Use of digits after decimal separator*: The way data are reported should reflect the accuracy of the measurement with which the data have been obtained, e.g. TSS is usually measured as integer number, also average values of TSS should be presented as integer (even MSExcel® presents data differently)
- Numbers on axes in figures should be integers, unnecessary digits after decimal separator shall be avoided.

6.2 CASE STUDY 1 – CSO TREATMENT WETLAND (GERMANY)

Katharina Tondera
IMT Atlantique, GEPA, UBL, F-44307 Nantes, France

Project Name:	Retentionsbodenfilter Kenten
Location:	Bergheim (Erft), Germany
Wastewater Type:	Combined sewage from retention tank overflow (pre-settled)
Design Flow:	Approx. 1,000 m^3/h (maximum capacity: approx. 4,200 m^3)
Completion Date:	2006
Technology:	VF wetland for the treatment of combined sewer overflows
Description of project need:	Requirements of EU Water Framework Directive makes further treatment of overflows from combined sewer systems necessary.
Description of project solution:	The TW is situated after two retention tanks and is only charged when the overflow from the sewer network exceeds their capacity. The filter has a surface of 2,200 m^2 and is designed to treat up to 4,200 m^3 with a filtration velocity of 0.025 $L/s/m^2$. The minimum interval between two events is 36 hours (Figure 6.1).
Special benefits of using TW technology compared to other solutions:	This technology is currently the only one available to provide biological, biochemical and mechanical treatment of combined sewer overflows. Retention of TSS (>90%), COD (60–85%), nitrification of ammonium (>60%) and indicator bacteria (1–3 $\log_{10}$) have been very well documented (Table 6.1).

Figure 6.1 Case study 1 – CSO treatment wetland (Germany).

Table 6.1 Performance data case study 1: mean influent and effluent concentrations in mg/L ten years after starting operation*.

Parameter	Influent Concentration	Effluent Concentration
TSS	53 (*n* = 4)	8 (*n* = 3)
COD (homogenized)	86 (*n* = 7)	24 (*n* = 6)
TOC (*n* = 4)	41.8	16.4
DOC (*n* = 3)	21.0	13.7
NH_4-N (*n* = 6)	5.3	2.4

*No values for P provided, as system was not enhanced for its removal.

6.3 CASE STUDY 2 – FWS WETLAND FOR TREATMENT OF AGRICULTURAL DRAINAGE WATER (ITALY)

Stevo Lavrnić[1], Stefano Anconelli[2], Domenico Solimando[2] and Attilio Toscano[1]

[1]Alma Mater Studiorum – University of Bologna, Department of Agricultural and Food Sciences, Viale Giuseppe Fanin 50, Bologna 40127, Italy
[2]Consorzio di Bonifica Canale Emiliano Romagnolo, Via Ernesto Masi 8, Bologna 40137, Italy

Project Name:	Green infrastructures for management and protection of water resources (Green4Water)
Location:	Bologna, Italy
Wastewater Type:	Agricultural drainage water
Completion Date:	Constructed in 2001 and operating since
Technology:	Free Water Surface (FWS) wetland
Description of project need:	A 12.5 ha experimental farm of Land Reclamation Consortium Canale Emiliano Romagnolo produces different crops throughout the year. In order to prevent pollution of surface water bodes with nutrient or chemical products, a low-cost and sustainable drainage water treatment solution, that could function with an intermittent inflow, was constructed.
Description of project solution:	The FWS wetland receives water from the main ditch to which is drained all the farm area. Two pumps convey water from the ditch towards the inlet once water in the ditch reaches a certain level. On the other hand, when the water level in the ditch is too high, excess water bypasses the system through a weir gate. The FWS wetland size represent 3% of the total farm area, and the system has a total volume of around 1500 m^3. It is divided into four meanders that create a 470 m long watercourse (Figure 16). The volume of water going in and out of the system is being constantly monitored by a central control station, as well as water level inside the ditch and the system itself. In addition, the control station has two refrigerated sampling units, one for influent and other for effluent, sampling being done on the basis of volume and time (Figure 6.2, Figure 6.3).
Special benefits of using TW technology compared to other solutions:	The water flow in the system is gravitational and therefore operating costs are low, especially since only occasional maintenance works are needed every few weeks. Long-term monitoring (2003–2017) showed that the system contributes to water quality in the area since it removes nutrients from the farm's drainage water and acts as a biofilter for different pollutants. Most importantly, the wetland technology applied was able to cope with different inflow volumes and pollution loads that are characteristic for agricultural drainage water (Lavrnić *et al.,* 2018). In addition, being located in the middle of arable land, vegetated and surrounded by trees, the FWS wetland provides ecosystem services and hosts various organisms such as birds, frogs or crayfish.

More information:

- https://site.unibo.it/green4water/en
- Lavrnić *et al.* (2018), *Water* 10(5), 644, https://doi.org/10.3390/w10050644.

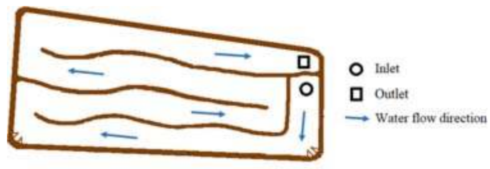

Figure 6.2 Case study 2 – Schematic of FWS wetland for treatment of agricultural drainage water (Italy).

Figure 6.3 Case study 2 – FWS wetland for treatment of agricultural drainage water (Italy).

6.4 CASE STUDY 3 – LANDFILL LEACHATE TREATMENT WETLAND SYSTEM (AUSTRALIA)

Kathy Meney and Ljiljana Pantelic

Syrinx Environmental, 12 Monger Street, Perth, WA, Australia

Project Name:	Burnie Landfill Leachate Treatment Wetland System
Location:	Burnie, Tasmania, Australia
Wastewater Type:	Landfill leachate
Design Flow:	Average flow of ~280 m^3/d, with a peak treatment capacity of 600 m^3/d
Completion Date:	January 2017 – construction completed February–July 2017 – commissioning & validation monitoring July 2017 – start of operation
Technology:	The system is comprised of vegetated surface-flow and subsurface-flow wetlands, followed by an evapotranspiration/infiltration forested wetland which further polishes effluent and mainly indirectly discharges the water to the creek via subsurface seepage.
The main drivers for the project:	This project was initiated because of (i) pressure to remove the leachate from the existing sewer network, (ii) impacts of leachate migrating off-site to the receiving environment, and (iii) changing community expectations due to urban encroachment. Environmental impacts were complicated by the fact that the treated leachate was to be discharged to a local creek used for irrigation and which is home to many nationally protected fauna species. These sensitivities invoked significant regulatory pressure to ensure any treated leachate discharge would need to be to a very high standard in order to protect environmental values.
Description of project need:	The wetland system needed to address the following key challenges: • Very complex hydrogeological setting (landfill is within a groundwater discharge valley catchment) with all surface/groundwater ultimately reporting to a nearby creek. • Unique leachate characteristics (high-volume, low-strength leachate), due to a complex interaction between leachate, groundwater and stormwater. • Very stringent discharge standards set to protect the sensitive receiving creek system. • Space limitations on site; apart from the landfill itself, very little available land surrounds the site.
Description of project solution:	The project solution is an integrated on-site leachate management (treatment and disposal) system that includes: (1) a treatment wetland system; (2) a separate stormwater treatment system; and (3) a raw leachate interception collection and phytoremediation treatment system to manage infrequent ponding events associated with seepages from the landfill during large rainfall events. Local species were used for planting of vegetated zones.

	The design makes provision for interpretive elements (boardwalks, signage) and educational/recreational engagement (school groups, researchers, tours, local residents). In time, as the urban zone advances, the landfill 'park' will become a key public open space element (Figure 6.4).
Special benefits of using TW technology compared to other solutions:	Use of wetland treatment technology for treatment of leachate in this project provided an effective, and relatively low-cost solution that goes beyond simply addressing an issue. In addition to a high-level treatment that enables leachate infiltration/discharge (primary project objective), the treatment wetland system provided a range of additional benefits which were not possible with other technologies, making it a showcase for sustainable remediation. These benefits are in line with sustainable triple bottom line principles and include environmental, social and economic benefits. Further to these benefits, delivery of this treatment wetland project provided a range of other important learnings for the different stakeholders involved in leachate management (designers, regulators, managers, operators). These learnings have already been adopted by other landfill managers to start the process of sustainable leachate management on their sites. The project has received wide recognition in Australia, including winning a number of state and national awards for sustainable remediation.
Performance:	Since commissioning was completed the removal efficiency for all key parameters (TN, TKN, ammonia, nitrate) has progressively increased, highlighting the importance of system maturation to overall performance.

More information: https://www.syrinx.net.au/portfolio/burnie/

Figure 6.4 Case study 3 – Landfill Leachate Treatment Wetland System (Australia).

6.5 CASE STUDY 4 – NIMR WATER TREATMENT PLANT (OMAN)

Alexandros Stefanakis, Stephane Prigent and Roman Breuer
Bauer Nimr LLC, PO Box 1186, PC114 Al Mina, Muscat, Oman

Project Name:	Nimr Water Treatment Plant (NWTP)
Location:	Nimr, Sultanate of Oman
Wastewater Type:	Produced water from oil exploration and production industry
Design Flow:	175,000 m^3/day
Completion Date:	Phases 1 & 2 (115,000 m^3/day) in operation since 2011, Phase 3 (additional 60,000 m^3/day to reach 175,000 m^3/day in total) under construction (completion: May 2019)
Technology:	The technology used is (i) passive hydro-cyclones for oil in water separation, (ii) FWS wetland for water polishing and hydrocarbons breakdown, and (iii) evaporation ponds for treated effluent disposal (zero-discharge system). Also, partial reuse of the treated effluent for irrigation of crops with market value has already started.
Description of project need:	Oil exploration and production in Nimr area is associated with large volumes of produced water, with a water-to-oil ratio as high as 1:10 after oil separation. A fraction of this produced water is injected to maintain the reservoir pressure and the remainder is disposed into deep aquifers. However, deep-well injection poses environmental risks and demands high energy consumption in a desert area with limited power supply. Thus, an alternative solution was required for produced water management that would be cost-effective, environmentally friendly and sustainable.
Description of project solution:	The NWTP is a hybrid system, incorporating elements of natural systems (green infrastructure) with traditional treatment technologies (grey infrastructure). First, separation and recovery of the majority of oil from the produced water takes place, using a series of passive hydro-cyclone oil separators. Then, the produced water is distributed to a FWS wetland of 360 hectares area via a long buffer pond. The treated water flows with gravity into a series of evaporation ponds (500 hectares), where evaporation results in salt formation, which can be processed into industrial grade salt as end-product. The NWTP currently treats 115,000 m^3/day of produced water, while an expansion with additional 60,000 m^3/day (130 hectares of wetlands to be added) is under construction. The size of the wetland facility makes this system one of the largest treatment wetlands in the world (Figure 6.5).
Special benefits of using TW technology compared to other solutions:	Due to the operation of the NWTP, five high-pressure deep-well disposal pumps have been shut down. Also, the whole NWTP is a gravity-based system with close-to-zero energy demand for the water treatment processes. This is a unique benefit of this technological solution, which translates to 98% reduction in energy consumption. The respective

estimated reduction in carbon emission is more than 1.5 million tons CO_2, or 99% compared to the other technological and disposal options. The NWTP alone contributes by approximately 4.26% to Oman's overall Intended Nationally Determined Contributions (according to Paris Agreement) to reduce emissions by 2% by 2030. This wetland facility is built in a previously arid desert. The large treatment wetlands and the series of evaporation ponds provide a valuable habitat for migratory birds and other wildlife. Given that the site is located in the middle of the East Asia/East Africa flyway, more than 120 different bird species have been identified in and around the wetlands and ponds, which utilize the facility as a comfortable stop-over during their migration. Furthermore, a large-scale three-year experiment will be completed at the end of 2018: it investigates the reuse of the treated effluent for irrigation of salt-tolerant plants with market value, e.g., biofuels, wood biomass etc. Ultimate goal is to make this facility a global example of circular economy with zero-waste production (Table 6.2).

Figure 6.5 Case study 4 – Nimr Water Treatment Plant (Oman). (Pictures reprinted with permission from Bauer Nimr LLC)

Table 6.2 Performance data case study 4: treatment performance (average values in mg/L).

Parameter	Inflow	Outflow
Total Dissolved Solids	7,000	12,000
Suspended solids	28	10
Oil in water	280	<0.5
BOD	15.7	<1
Total Nitrogen as N	2.5	<0.5
Ammonia Nitrogen as N	1.3	<0.1
Total Phosphorus as TP	0.03	<0.5
Boron as B (dissolved)	4.5	5.6

6.6 CASE STUDY 5 – CECCHI WINERY WASTEWATER TREATMENT PLANT (ITALY)

Fabio Masi, Riccardo Bresciani and Anacleto Rizzo

Iridra Srl, via La Marmora 51, 50121, Florence, Italy

Project Name:	Cecchi Winery Wastewater Treatment Plant
Location:	Castellina in Chianti, Italy
Wastewater Type:	Winery wastewater
Design Flow:	100 m^3/d (mean value during peak vintage season)
Completion Date:	In operation since 2001, upgraded in 2009
Technology:	The technology used is: 1st stage of a French VF wetland raw wastewater of 1,200 m^2; 2nd stage with 4 parallel HF Wetlands of 960 m^2 (240 m^2 each); 3rd stage a single-bed FWS wetland of 850 m^2; optional sand filter of 50 m^2 before discharge into freshwater (Gena River).
Description of project need:	The winery wastewater produced by the Casa Vinicola Luigi Cecchi & Sons (Castellina in Chianti, Siena) has been treated with a multi-stage wetland system since 2001. The system consisted of an Imhoff tank, followed by a single-stage HF wetland of 480 m^2 and then by an FWS of 850 m^2. The system was designed to treat 35 m^3/d, and starting from the year 2006 the production at the winery greatly improved and consequently flows to the wetland increased up to 70 m^3/d. A prolonged overload, for about 2–3 years, resulted in a severe clogging of the HF bed. Therefore, an upgrade of the TW was required in 2009.
Description of project solution:	The choice of a first stage of a French VF wetland for raw wastewater as first stage of a multi-stage TW system enhances the sustainability of the treatment plant, by the reduction of primary sludge production and sludge cycle management costs it is also providing more robustness to the treatment train, minimizing a big part of the problems observed in the above cited experiences with the older 'Imhoff + HF + FWS' configuration. The installation of the new first stage has resulted in removal of the old Imhoff tank, which was creating some problems in the HF due to frequent events of exceptionally high flows and linked wash-out events from the Imhoff tank itself, when unmeasured amounts of primary sludge reached the inlet section of the HF, surely contributing to its clogging (Figure 6.6).
Special benefits of using TW technology compared to other solutions:	Winery wastewater has proved to be difficult to treat with conventional technological solutions (e.g., activated sludge, anaerobic reactors), for the following reasons: (i) variable pH, usually ranging from 4 to 8 in the different periods of the year; (ii) low nutrient content and consequent unfavourable C/N ratio for the microbial growth;

(iii) high content of biodegradable compounds that often leads to difficulties in operating biological systems, for instance poor sludge settleability, floc disintegration and increased presence of solids in the treated effluent;
(iv) seasonality and load fluctuations;
(v) clogging in filtering reactors;
(vi) phytotoxicity and microbial inhibition by toxic organic and inorganic compounds, i.e. sulfur, phenols, tannins.

As well as being a low-cost, low-maintenance and energy-saving technology, the WWTP of Cecchi winery also shows that TWs are an effective solution to cope with winery wastewater issues. The TW of Cecchi winery also shows the potential of multi-stage systems in treatment of winery wastewater. In particular, the first stage of a French VF wetland as the system's first stage is providing more stable performance and no clogging signals have yet been noticed after 10 years of operation from the upgrading (Table 6.3).

Figure 6.6 Case study 5 – Cecchi Winery Wastewater Treatment Plant (Italy).

Table 6.3 Performance data case study 5: treatment performance (average values in mg/L).

Parameter	Inflow	Outflow
TSS	213	25
COD	3,800	67
BOD_5	1833	20
TN	19	7
TP	5	1

More information:

- Masi *et al.* (2015a), *Water Science and Technology* **71**, 1113–1127.

6.7 CASE STUDY 6 – DICOMANO WASTEWATER TREATMENT PLANT (ITALY)

Riccardo Bresciani, Anacleto Rizzo and Fabio Masi

Iridra Srl, via La Marmora 51, 50121, Florence, Italy

Project Name:	Multi-stage treatment wetland of Dicomano (Italy)
Location:	Dicomano, Italy
Wastewater Type:	Municipal wastewater
Design Flow:	525 m^3/day
Completion Date:	In operation since 2003
Technology:	The technology used is: first stage with two parallel HF wetlands of 1,000 m^2 (500 m^2 each); second stage with eight parallel VF wetlands of 1,680 m^2 (210 m^2 each); third stage with two parallel HF wetlands of 1,800 m^2 (900 m^2 each); fourth stage with single-bed FWS wetland of 1,600 m^2. Total surface of 6,080 m^2.
Description of project need:	Dicomano is a little settlement situated in the Florence countryside, about 160 m above sea level: before the new wastewater treatment plant (WWTP) the urban wastewater was discharged into the Sieve River, the most important Arno River tributary. Therefore, the settlement needed a WWTP suitable to treat the municipal wastewater according with the strict Italian law (especially in terms of nutrients), while achieving low operation and maintenance costs.
Description of project solution:	The concept design is based on the benefits given by multi-stage systems in terms of multiple water quality targets to be met. Therefore, a multi-stage wetland system has been realised with specific roles for each compartment: first, HF beds for organic and suspended solid removal; second, VF beds to obtain an enhanced nitrification; third, HF beds for denitrification; fourth, final FWS to improve pathogen removal and advanced denitrification (Figure 6.7, Figure 6.8).
Special benefits of using TW technology compared to other solutions:	The WWTP was able to meet specific limits set by Italian law (D. Lgs. 152/2006): BOD_5 (40 mg/L), COD (160 mg/L), TSS (80 mg/L), nitrogen compounds (35 mg/L), phosphorus (10 mg/L), and pathogens (5,000 cfu/100 mL). These strict limits, especially for nutrients, were met exploiting a multi-stage approach. In this way, the system was designed with a lower footprint in comparison to single-stage TW system, i.e. less than 2 m^2 p.e.$^{-1}$. A greater flexibility to influent variation in wastewater load was given adopting TW instead of conventional technology. Indeed, the multi-stage TW of Dicomano was able to respect Italian water quality standard even under severe influent fluctuation, due to the mixed nature of the municipal sewer system. Indeed, the sewer also drains in some periods 'parasite' rainwater from the ground and has been affected by a severe infiltration of water from a torrent for few years. The operation and maintenance costs were 20,000 € per year, significantly lower (0.1 €/m^3) than conventional technological solutions. Finally, the use of nature-based solutions has given the possibility to provide an additional ecosystem service in terms of biodiversity increase, since the FWS stage was planted with 16 different Tuscany's native macrophytes (Table 6.4).

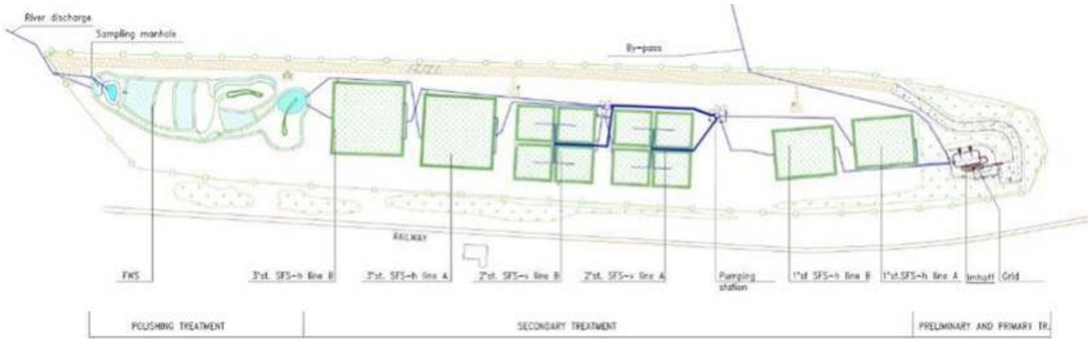

Figure 6.7 Case study 6 – Schematic of Dicomano Wastewater Treatment Plant (Italy).

Figure 6.8 Case study 6 – Dicomano Wastewater Treatment Plant (Italy).

Table 6.4 Performance data case study 6: treatment performance (average values in mg/L).

Parameter	Inflow	Outflow
TSS	51	5
BOD	66	4
COD	160	18
NH4+	31	7
TN	28	10
TP	2.7	1.6

More information:

- Masi *et al.* (2013), *Water Science and Technology* **67**, 1590–1598.

6.8 CASE STUDY 7 – ORHEI WASTEWATER TREATMENT PLANT (MOLDOVA)

Anacleto Rizzo, Riccardo Bresciani and Fabio Masi
Iridra Srl, via La Marmora 51, 50121, Florence, Italy

Project Name:	Orhei Wastewater Treatment Plant
Location:	Orhei, Moldova
Wastewater Type:	Municipal wastewater
Design Flow:	1,000 m^3/d (mean value)
Completion Date:	In operation since 2013
Technology:	The TW occupies a gross area of 50,000 m^2 and is designed as French VF wetland. Four independent two-stage treatment lines working in parallel are present, with first and second stage surface area for each line equal to 4,489 m^2 (three sectors of 1,496 m^2 each) and 4,248 m^2 (four sectors of 1,062 m^2 each), respectively.
Description of project need:	The Orhei municipality (20,000 PE) needed a new WWTP. The new plant was promoted and funded by the World Bank, who highlighted the need to minimize the operation costs according with the maximum affordable water tariff in the local economic situation.
Description of project solution:	In order to minimize the operation and maintenance costs, a French VF wetland was chosen to avoid the yearly cost given by classical primary treatment (septic or Imhoff tanks). The design followed the French VF wetland principles and guidelines. The first stage is fed with raw wastewater, designed for high removal of TSS, COD, and ammonia. The second stage is designed to refine the treatment and to complete the nitrification (Figure 6.9).
Special benefits of using TW technology compared to other solutions:	The Orhei WWTP design and supervision of the construction was promoted and funded by the World Bank. A TW treatment technology was chosen to minimize the operation costs. Indeed, the World Bank consultants have compared TW with other common systems (activated sludge plants, SBRs, and percolating filters), showing that TW would be the only financially feasible technology with the maximum affordable water tariff in the local economic situation. Moreover, the Orhei WWTP confirms that there are no upper limits, in terms of maximum treatable person equivalent, for the application of wetland systems for municipal wastewater treatment when land is available at a proper cost (Table 6.5)

Figure 6.9 Case study 7 – Orhei Wastewater Treatment Plant (Moldova).

Table 6.5 Performance data case study 7: treatment performance (average values in mg/L).

Parameter	Inflow	Outflow
TSS	583	23
COD	222	32
BOD_5	106	15
N-NH4+	47	16

More information:

- Masi *et al.* (2017b), *Water Science and Technology* **76**, 68–78.

6.9 CASE STUDY 8 – MULTIFUNCTIONAL WATER RESERVOIR IN LJUBLJANA (SLOVENIA)

Tjaša Griessler Bulc[1], Darja Istenič[1] and Aleksandra Krivograd Klemenčič[2]

[1]University of Ljubljana, Faculty of Health Sciences, Zdravstvena pot 5, 1000 Ljubljana, Slovenia
[2]University of Ljubljana, Faculty of Civil and Geodetic Engineering, Hajdrihova 28, 1000 Ljubljana, Slovenia

Project Name:	Multifunctional water reservoir (MWR) in Ljubljana
Location:	Ljubljana, Slovenia
Wastewater Type:	River water, urban stormwater runoff, septic tanks overflows
Design Flow:	173 m^3/day
Completion Date:	Phase 1 (construction of MWR) in operation since 2006, Phase 2 reconstruction of MWR, upgrading for several ecosystem functions (biodiversity, recreation, education) in operation since 2014.
Technology:	The basic design of MWR in Ljubljana consists of: (1) sedimentation basin, (2) vegetated drainage ditch (VDD) as a type of TW, and (3) a new river bed with meanders. The whole MWR is integrated in a swamp that was created by natural way in an engineered flood reservoir protecting west Ljubljana from floods.
Description of project need:	The City of Ljubljana has been dealing with flooding of rivers for many years, especially because settlements are gradually spreading to areas of periodic flooding. The flood reservoir was constructed in 1986 on Glinščica river to tackle the issue of floods, but later it was facing water quality problems, as it was affected by occasional overflows from septic tanks, polluted tributaries and urban stormwater runoff (gardens, parking places). The authorities have addressed the problem by constructing MWR in 2006 (Phase 1), but the 2010 flood event made the need for additional flood protection measures obvious (Phase 2). MWR was finally constructed to provide several functions regarding environmental protection, namely: (a) flood prevention; (b) water retention for irrigation purposes of nearby green areas; (c) water pollution mitigation from urban gardens and sewage overflows; (d) increased self-cleaning capacity of the ecosystem; (e) increased biodiversity; (f) establishment of recreation and education path. The hydraulic retention capacity of MWR was designed to 10-year flood events.
Description of project solution:	The first rehabilitation step prior to MWR construction was to redirect the flow of max 2 L/s from main river bed and deepen the first part forming a small retention basin (10 m^3), which was watertight, to slow down the water flow and enable efficient sedimentation of particles. After the sedimentation basin, the water

runs over a weir to the VDD, which functions as a horizontal flow TW. It is divided into three segments with a depth of 0.4 m and is lined with foil to ensure water tightness. Individual VDD segments are filled with sand and gravel of 60–80 mm (first segment), 30–60 mm (upper 10 cm layer of the second and third segment) and 16–32 mm (lower 30 cm layer of the second and third segment) and planted with common reed (*Phragmites australis*). For the purpose of water sampling and measurements of water level, there are piezometers installed at the beginning and at the end of each VDD segment. The treated water flows from the VDD into the newly established river bed with meanders. The banks of a riverbed are strengthened by in-built willow wattle fences; spurs, half logs and ripraps were also constructed. The MWR was planted with diverse indigenous wetland plants: at the banks of the riverbed broadleaf cattail (*Typha latifolia*), soft rush (*Juncus effusus*), sedge (*Carex* sp.) and yellow iris (*Iris pseudacorus*) were planted; while for greater distances from water woody plants were selected: willows (*Salix* spp.), common hazel (*Corylus avellana*), black alder (*Alnus glutinosa*) and pedunculate oak (*Quercus robur*). Maintenance on regular basis is required to avoid reduction of the retention capacity due to alluvial deposits and overgrowth of vegetation, including the establishment of safe operating conditions (Figure 6.10).

Special benefits of using TW technology compared to other solutions:

Flood prevention: MWR reduces hydraulic peaks by retaining water in the system and therefore prevents and mitigates floods and droughts in the nearby area.

Water treatment: Due to integration of VDD, MWR effectively treats the inflow water and increases the self-cleaning capacity of the area.

Energy savings: MWR can provide its services with very little or no energy input if designed appropriately.

Enhanced biodiversity: MWR creates a new habitat for wildlife and contributes to an increased biodiversity in a barren landscape (e.g. spawning ground for frogs and toads, breeding sites for birds etc.).

Recreation: MWR is designed with elements of landscape architecture (banks, walking path and bridge) and creates an attractive recreational place for the community.

Education: MWR is a tangible example (recognized as a good practice by European Environment Agency, 2016) of a measure aimed to achieve sustainable development. It is used by the City of Ljubljana, schools and universities to present the problems of pollution and its remediation in a natural way to different target groups. It offers new perspectives for future developments in water management and flood prevention.

Figure 6.10 Case study 8 – Multifunctional water reservoir in Ljubljana (Slovenia).

Performance data: Most of the inflow parameters were in low concentrations (TN 2.7 $\pm$ 1.2 mg/L, TP 0.3 $\pm$ 0.1 mg/L, BOD_5 6.9 $\pm$ 3.1 mg/L); therefore high removal rates were not expected. Average removal efficiencies for the MWR reached on average 68% for NO_3-N, 40% for TN, 7% for NH_4-N, 9% for BOD_5 and 3% for TP while SS and COD increased. The VDD was efficient in removal of NH_4-N (38%), and NO_3-N (63%), but these parameters than increased again in the new river bed with meanders, which was on the other hand efficient in removal of TP (10%). The performance of MWR should not be reviewed only through removal of stated parameters but also through the impact on ecosystem services. Concerning the biodiversity, marsh vegetation in this area and algae species are extremely diverse. Also, the area of the flood reservoir is potentially a suitable habitat of endangered animal species and rare birds, like the green woodpecker (*Picus viridis*), the presence of which has been confirmed in the area. With the appropriate arrangements, the flood reservoir is offering an interesting recreational path for local residents and an educational path (bird observation points, observation of self-cleaning elements of the wetland and the river) and a recreational place (walking, jogging) in dry periods.

6.10 CASE STUDY 9 – GREEN FILTERS PROJECT (THE PHILIPPINES)

Ma. Catriona E Devanadera[1,2], Aaron Julius M Lecciones[1,2], Amy M Lecciones[2], Ma. Cheryl F Prudente[2] and Jose Carlo H Quintos[2]

[1]Department of Community and Environmental Resource Planning, College of Human Ecology, University of the Philippines Los Baños, The Philippines
[2]Society for the Conservation of Philippine Wetlands, Inc., The Philippines

Project Name:	Green Filters Project
Partners:	LP4Y; Global Nature Fund; Kärcher; Sika; Holcim Philippines
Location:	Life Project for Youth (LP4Y) Green Village, Calauan, Laguna, Philippines
Wastewater Type:	Domestic Wastewater from LP4Y Green Village
Design Flow:	Target of 200–300 pax capacity when fully operational
Completion Date:	Construction completed (May 2017); not fully operational
Technology:	The Green Filters is composed of the following systems: (i) an Anaerobic Baffled Reactor which serves as the septic/holding tank receives all black wastewater from toilets and showers; wastewater (ii) flows to the VF wetland, then to the (iii) 2-stage HF wetland; and ends at the (iv) Polishing Pond. The final effluent is then released to the creek nearby. The plan is to reuse the treated domestic wastewater for Green Village's organic garden activities.
Description of project need:	Sanitation is a major issue in the Philippines as a result of the high cost of centralized domestic wastewater treatment system. Only about 5% of domestic wastewater is being treated in the Philippines and in mostly urban areas which can afford the expensive cost of centralized system. Rural areas remain a challenge when it comes to sanitation. If these issues are not addressed, waterbodies in the country will continue to degrade and will pose a threat to public health. This is the case in Manila Bay, Boracay, and Laguna Lake in the Philippines which have become "cesspools" due to untreated domestic wastewater discharge.
Description of project solution:	Treatment wetlands as a natural treatment system, harnesses the potential of plants, microbes, and filter materials to clean water. Local vegetation such as *Heliconia sp.*, canna lily, horsetail and sedges are used in the system. TWs are a low-cost system that can be easily adopted in communities and used even without connection to a central wastewater treatment system. This project will minimize household wastewater discharged directly into waterbodies which causes water pollution and health related diseases. Treated wastewater can also be reused for the purpose of irrigation and gardening.

The Green Filters project aims to demonstrate a method of treating domestic wastewater using a technology that employs natural and local resources. Specifically, it aims to: (1) provide an economically and ecologically sound alternative technology for treating domestic wastewater before discharge to waterbodies; (2) contribute to food and water security in the community; (3) establish a model for economically and ecologically sound alternative technology for treating domestic wastewater; and (4) increase the awareness of local communities and local government units on the problems caused by pollution and its effects on people and the natural ecosystems. The Green Filter project will achieve an overall design that blends with the natural environment in accordance with the landscape of the Green Village. Environmental awareness and protection, biodiversity enhancement, and social acceptance of the "natural wastewater treatment system'' will be realized through this project (Figure 6.11).

Figure 6.11 Case study 9 – Green Filters Project (The Philippines).

6.11 CASE STUDY 10 – BAHCO TREATMENT WETLAND FOR EFFLUENT FINAL POLISHING (ARGENTINA)

María A Maine, Hernán R Hadad, Gabriela C Sánchez, María M Mufarrege and Gisela A Di Luca

Química Analítica Ambiental, Instituto de Química Aplicada del Litoral (IQAL, CONICET-UNL), Facultad de Ingeniería Química, Universidad Nacional del Litoral (UNL), Santiago del Estero 2829, Santa Fe (3000), Argentina & Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Argentina

Project Name:	Bahco treatment wetland for effluent final polishing (Argentina)
Location:	Santo Tome, Santa Fe (Argentina)
Wastewater Type:	Metallurgical industry wastewater
Design Flow:	100 m^3/day
Completion Date:	In operation since 2002
Technology:	A FWS wetland of 2,000 m^2 was constructed. It is 50 m long, 40 m wide and 0.4–0.5 m deep. A central baffle was constructed, parallel to the flow direction, dividing the wetland into two sections of equal area and forcing the effluent to flow in "U" form, covering double the distance, resulting in a 5:1 length–width ratio. The wetland was rendered impermeable with 6 layers of compacted bentonite, in order to achieve a hydraulic conductivity of 10^{-7} m s^{-1}. A layer of 1 m of soil was placed on top of the bentonite layer. Several locally available macrophyte species were planted into the wetland. *Typha domingensis* became the dominant species, covering the total area of the wetland. Hydraulic residence time ranged from 7 to 10 days. The effluent, after passing through the wetland, was led to a 1.5 ha pond in the factory facilities. Phreatic water meters were placed around the wetland to monitor groundwater quality, as a security measure.
Description of project need:	Bahco metallurgical industry for toolmaking needed an effluent final-stage treatment. A large land area was available in the factory facilities and costs for maintance and operation of watewater treatment are limiting factors in Argentina. In addition, sewage from the factory also required a final treatment.
Description of project solution:	A FWS wetland was constructed. This type of TW was selected due to the efficiency in metal removal and the low costs for operation and maintenance. Although FWSs requires a large area, this is not a problem in this case. Industrial wastewater containing metals and sewage from the factory are treated together, both after a primary treatment (25 m^3 d^{-1} of sewage + 75 m^3 d^{-1} of industrial wastewater). Sewage improves the ability of macrophytes to take up heavy metals from wastewater (Figure 6.12).
Special benefits of using TW technology compared to other solutions:	The FWS wetland showed high removal efficiencies of Cr, Ni, Zn, Fe, COD and BOD. Treated effluent meets the Argentinian law limits for discharge. FWS performance improved with wetland maturity. Sediment and macrophyte roots were responsible for the metal removal. Metals were bound to sediment fractions that would not release them into water while the chemical and environmental

conditions of the system were maintained. Although this FWS wetland was faced with accidental events, it was capable of recovering its performance, demonstrating its robustness. FWS and the discharge pond provide an additional ecosystem service with a high diversity of macrophytes and have become the habitat for diverse wildlife, such as ducks, geese, coots, coypus, lizards, capybaras, turtles, etc (Table 6.6).

Figure 6.12 Case study 10 – Bahco treatment wetland for effluent final polishing (Argentina).

Table 6.6 Performance data case study 10 (Argentina). Ranges (minimum and maximum values in mg/L) of measured parameters at the inlet and outlet and removal efficiencies.

Parameter	Inlet	Outlet	% Removal
pH	10.4–12.2	7.9–9.3	–
Conductivity (µS/cm)	3890–8700	1400–2500	–
Fe (mg/L)	0.05–2.54	0.05–0.430	89.4
Cr (mg/L)	0.023–0.204	0.002–0.033	84.7
Zn (mg/L)	0.022–0.070	0.015–0.050	51.2
Ni (mg/L)	0.004–0.101	0.004–0.082	69.5
COD (mg/L)	27.9–154.0	13.9–42.9	74.6
BOD (mg/L)	9.8–30.9	3.0–20.1	73.2

More information:

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Index

Water quality standards across the world are being re-written to promote healthier ecosystems, ensure safe potable water sources, increased biodiversity, and enhanced ecological functions. Treatment wetlands are used for treating a variety of pollutant waters, including municipal wastewater, agricultural and urban runoff, industrial effluents, and combined sewer overflows, among others. Treatment wetlands are particularly well-suited for sustainable water management because they can cope with variable influent loads, can be constructed of local materials, have low operations and maintenance requirements compared to other treatment technologies, and they can provide additional ecosystem services. The technology has been successfully implemented in both developed and developing countries.

The first IWA Scientific and Technical Report (STR) on Wetland Technology was published in 2000. With the exponential development of the technology since then, the generation of a new STR was facilitated by the IWA Task Group on Mainstreaming Wetland Technology. This STR was conceptualized and written by leading experts in the field. The new report presents the latest technology applications within an innovative planning framework of multi-purpose wetland design. It also includes practical design information collected from over twenty years of experience from practitioners and academics, covering experiments at laboratory and pilot-scale up to full-scale applications.

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