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

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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

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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

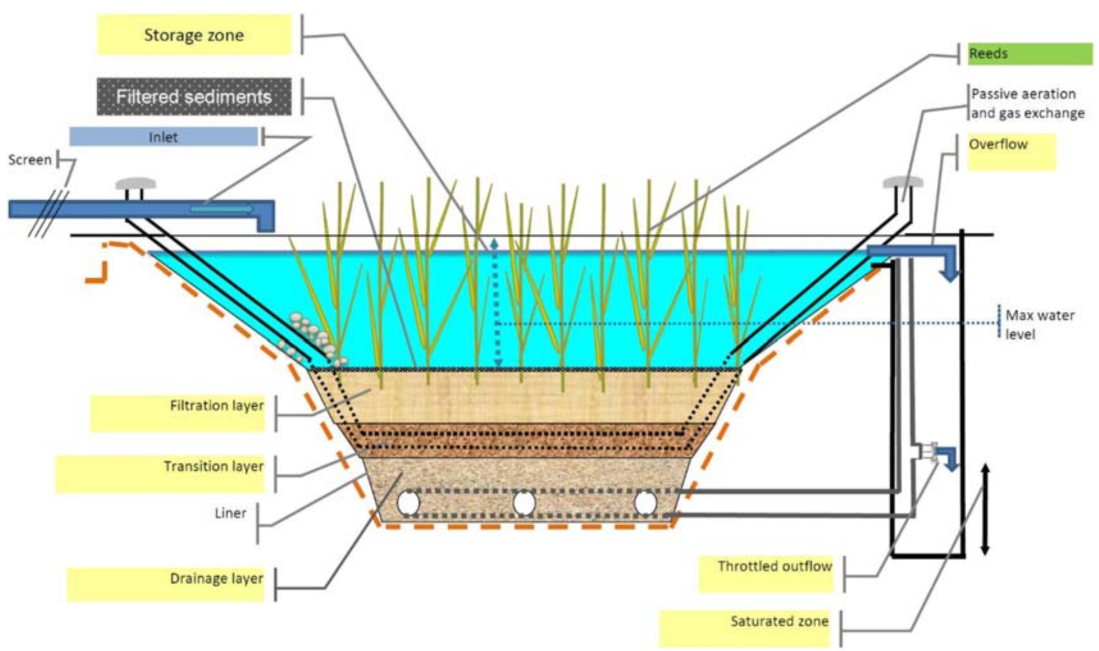

Figure 4.1 Cross section of VF wetland with storage volume on top of the filter surface and throttled outflow, as used for stormwater and CSO storage and treatment.

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 · 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 · 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 · 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

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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

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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

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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

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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

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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

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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 / Wetland Type	Caffeine	Ibuprofen	Naproxen	Benzotriazole	Diclofenac	Acesulfame	Carbamazepine	Triclosan
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

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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

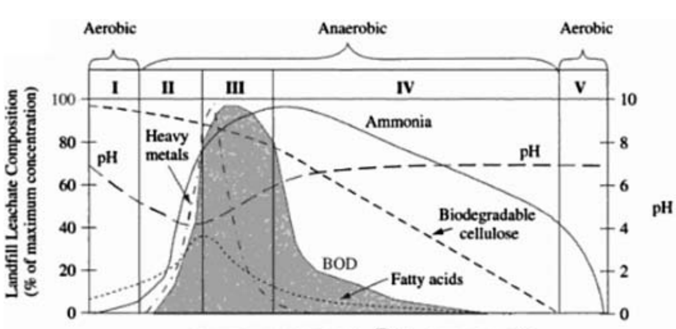

Figure 4.2 Changes in the composition of leachate with aging of the landfill (from DoE, 1995).

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

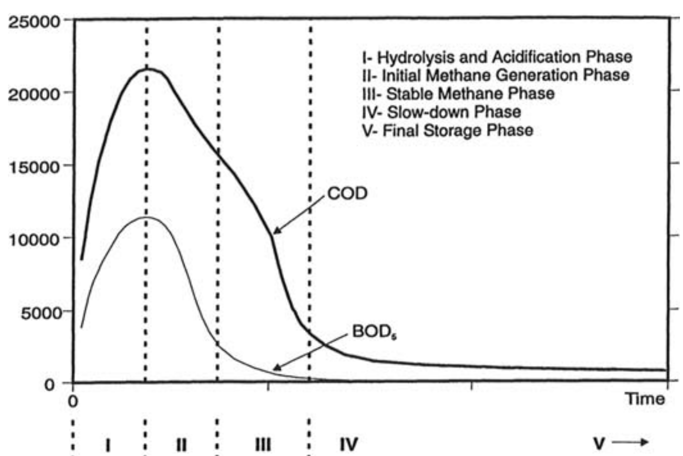

Figure 4.3 Changes of COD and BOD_5 concentrations with aging of the landfill (adapted from McBean & Rovers, 1999).

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

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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

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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

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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

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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

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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

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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-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

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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

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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

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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

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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).

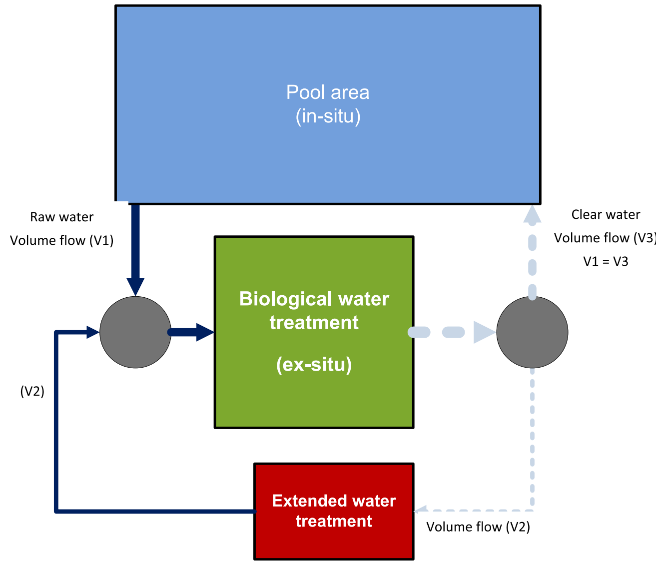

Figure 4.4 Circuit diagram for integrating the physical–chemical water treatment into the biological processes.

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

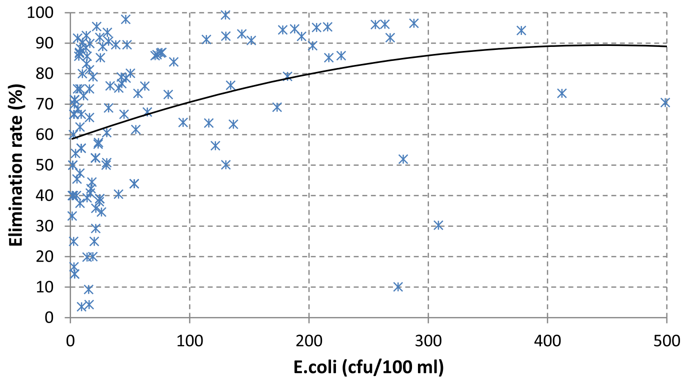

Figure 4.5 Elimination performance of *Escherichia coli* in freely drained vertical-flow filters under field conditions (from monitoring data collected by the DANA database developed by POLYPLAN on public swimming pools from 2005 to 2018.

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).

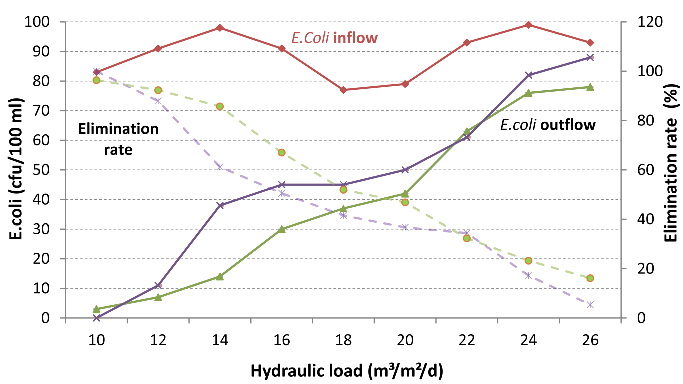

Figure 4.6 Elimination performance of *Escherichia coli* as a function of the hydraulic loading of two unsaturated filter columns (laboratory conditions) (adapted from Scholz & Frehse, 2004).

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

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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.