

Jan Vymazal *Editor*

Natural and Constructed Wetlands

Nutrients, heavy metals and energy
cycling, and flow



Springer

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Preface

Wetlands are extremely diverse not only for their physical characteristics and geographical distribution but also due to the variable ecosystem services they provide. Wetlands provide many important services to human society but are at the same time ecologically sensitive and adaptive systems. The most important wetland ecological services are flood control, groundwater replenishment, shoreline stabilization and protection, sediment and nutrient retention, water purification, biodiversity maintenance, wetland products, cultural and recreational values, and climate change mitigation and adaptation. The ecosystem services are provided by natural wetlands but also by constructed wetlands. Constructed wetlands utilize all natural processes (physical, physicochemical, biological) that occur in natural wetlands but do so under more controlled conditions. The constructed wetlands have primarily been used to treat various types of wastewater, but water retention, enhanced biodiversity, and wildlife habitat creation are the important goals as well. The necessity of bridging knowledge on natural and constructed wetlands was the driving force behind the organization of the International Workshop on Nutrient Cycling and Retention in Natural and Constructed Wetlands which was first held at Třeboň, Czech Republic, in 1995. The workshop was very successful and naturally evolved in a continuation of this event in future years.

The ninth edition of the workshop was held at Třeboň on March 25–29, 2015. The workshop was attended by 36 participants from 15 countries of Europe, North America, Asia, and Australia. This volume contains a selection of papers presented during the conference. The papers dealing with natural wetlands are aimed at several important topics that include the role of riparian wetlands in retention and removal of nitrogen, decomposition of macrophytes in relation to water depth, and consequent potential sequestration of carbon in the sediment and a methodological discussion of an appropriate number of sampling for denitrification or occurrence of the genus *Potamogeton* in Slovenian watercourses. The topics dealing with the use constructed wetlands include among others removal of nutrients from various types of wastewater (agricultural, municipal, industrial, landfill leachate) on local as well as catchment scale and removal of heavy metals and trace organic compounds. Two

papers also deal with the effect of wetlands in the mitigation of global warming and the effect of drainage and deforestation in climate warming.

The organization of the workshop was partially supported by the program “Competence Centres” (project no. TE02000077 “Smart Regions – Buildings and Settlements Information Modelling, Technology and Infrastructure for Sustainable Development”) from the Technology Agency of the Czech Republic.

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March 2016

Jan Vymazal

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Chapter 1

Effects of Human Activity on the Processing of Nitrogen in Riparian Wetlands: Implications for Watershed Water Quality

Denice H. Wardrop, M. Siobhan Fennessy, Jessica Moon, and Aliana Britson

Abstract Wetlands are critical ecosystems that make substantial contributions to ecosystem services. In this study, we asked how the delivery of an ecosystem service of interest (N processing such as denitrification and mineralization) is impacted by anthropogenic activity (as evidenced by land cover change). We identify relevant factors (hydrology, nitrogen, and carbon variables), select headwater wetland sites in Ohio and Pennsylvania USA to represent a gradient of anthropogenic disturbance as indicated by land cover characteristics (represented by the Land Development Index, or LDI), and determine if there are differences in the selected variables as a function of this gradient by categorizing sites into two groups representing high and low disturbance. We utilized Classification and Regression Trees (CART) to determine which variables best separated high from low disturbance sites, for each spatial scale at which land cover patterns were determined (100 m, 200 m, 1 km radius circles surrounding a site), and within each category of water quality variable (hydrology, nitrogen and carbon). Thresholds of LDI were determined via the CART analyses that separated sites into two general classes of high and low disturbance wetlands, with associated differences in Total Nitrogen, NH_4^+ , Soil Accretion, C:N, Maximum Water Level, Minimum Water Level, and %Time in Upper 30 cm. Low Disturbance Sites represented forested settings, and exhibit relatively higher TN, lower NH_4^+ , lower Soil Accretion, higher C:N, higher Maximum Water Level, shallower Minimum Water Level, and higher %Time in Upper 30 cm than the remaining sites. LDIs at 100 m and 200 m were best separated into groups of high and low disturbance sites by factors expected to be proximal or local in nature, while LDIs at 1000 m predicted factors that could be related to larger scale land cover patterns that are more distal in nature. We would expect a water quality

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process such as denitrification to be relatively lower in forested settings, due to the low available nitrogen (associated with high C:N) and constant and saturated conditions; conditions for maximum denitrification may be found in agricultural settings, where high nitrate groundwater can interact with surface soils through a wetting and drying pattern. The use of land cover patterns, as expressed by LDI, provided useful proxies for nitrogen, carbon, and hydrology characteristics related to provision of water quality services, and should be taken into account when creating, restoring, or managing these systems on a watershed scale.

Keywords Headwater wetlands • Denitrification • Nitrogen processing • Disturbance • Land cover • Land Development Index (LDI)

1.1 Introduction

The need to manage landscapes for ecosystem services is essential if we are to find solutions to issues that are critical for humanity, including energy policy, food security, and water supply (Holdren 2008; Robertson et al. 2008). Wetlands are critical ecosystems that make substantial contributions to the most valued of these ecosystem services (Millennium Ecosystem Assessment 2003), and their common location between human activities (e.g., agriculture, development) and critical water resources (e.g., aquifers and rivers used as water supplies, streams for recreational use) adds to their importance. The recognition that wetlands provide valuable ecosystem services has led to the development of assessment protocols to estimate service levels across wetland types in a landscape, evaluate services in relation to the impact that human activities have on these systems, and provide guidelines for wetland restoration in terms of these services (e.g., Zedler 2003).

Human activities are known to alter the benefits that ecosystems provide (MEA 2003). However, human activities often occur within the wider surrounding landscape and may be spatially disconnected from the ecosystem services they impact. For example, activities such as agriculture, expressed on the landscape as land cover in row crops or pasture, create stressors/drivers such as sedimentation and modification of hydrological patterns, which may influence ecosystem processes and condition indicators such as soil biogeochemistry and plant community, thus influencing an ecosystem service such as denitrification. This complicates our ability to determine linkages between land use change and subsequent impacts on the ecosystems that are part of that landscape. Assessing impacts requires understanding how human activities generate stressors that alter wetland ecological condition, and ultimately affect the flow of these services. The many system connections between activities, stressors, condition, and the ultimate delivery of services render simple landscape predictions to ecosystem service impossible (Xiong et al. 2015). For example, to inform our understanding of biogeochemical processes in wetlands we must necessarily look at linkages at several intermediate scales, including landscape

to wetland scale linkages (e.g. how land use affects conditions within a site, such as water levels); and landscape to process scale linkages (how land use affects the delivery of materials that drive ecosystem processes, e.g. nitrogen inflow).

While all ecosystem services are important, some of the most valued ecosystem services that wetlands provide, and are managed for, are those associated with water quality improvement due to the biogeochemical processing and storage of nutrients and sediment. For example, denitrification is the primary process by which nitrate is transformed in wetlands, thereby removing a key waterborne pollutant. In the U.S., nitrate runoff is a significant problem, enriching surface waters (Carpenter et al. 1998; Verhoeven et al. 2006) and contributing to hypoxia in the Gulf of Mexico (Turner and Rabalais 1991; Rabalais et al. 2002). Because of their connectivity to lotic ecosystems, high C availability, and inflows of nitrate, denitrification tends to be greatest in riparian and floodplain wetlands (Fennessy and Cronk 1997; Hill 1996).

The ecosystem services related to nitrogen processing are potentially controlled by a number of factors that occur at a range of spatial and temporal scales (Fig. 1.1). For example, denitrification is a microbial process that is most directly affected by factors at the process scale (Groffman et al. 1988) such as the availability of nitrate (Seitzinger 1994), dissolved organic carbon (DOC) (Sirivedhin and Gray 2006), temperature (Sirivedhin and Gray 2006), pH (Simek and Cooper 2002), and levels of dissolved oxygen (Hochstein et al. 1984). These process scale factors are affected by the wetland-scale structures of vegetation and hydrology; vegetation can affect carbon availability and temperature while hydrology can affect nitrate loading and redox conditions (Prescott 2010;

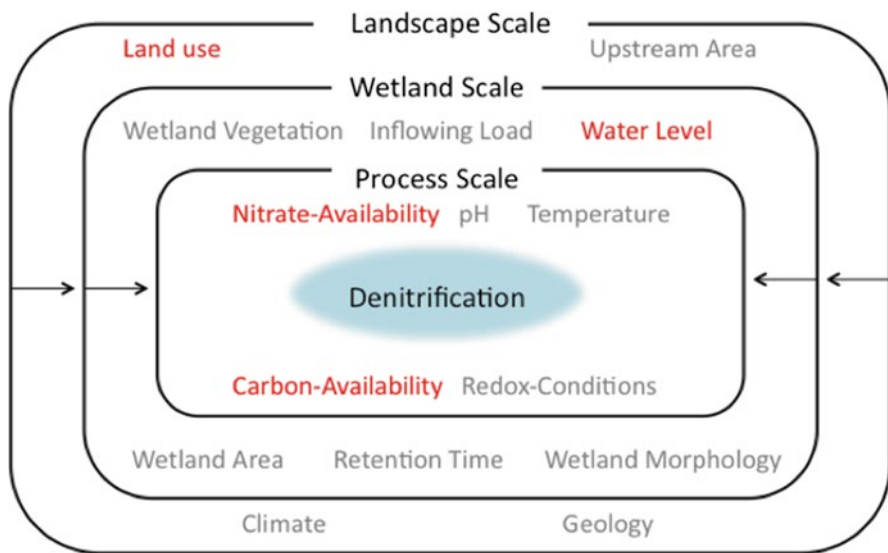


Fig. 1.1 Factors working at different spatial scales that affect the process of denitrification in wetlands. Factors shown in red were a focus of this study (Modified from Trepel and Palmeri 2002)

Adamus and Brandt 1990; Mitsch and Gosselink 2007). The wetland scale factors can also be affected by the landscape scale factors of land use, geology, and climate. Agricultural activities have been known to affect wetland hydrology, nitrate loading, and vegetative community, while climate and geology can affect wetland size and vegetation (Xiong et al. 2015; Groffman et al. 2002; Wardrop and Brooks 1998; Adamus and Brandt 1990; Mitsch and Gosselink 2007). This same dependence on both landscape- and wetland scale factors can be postulated for nitrogen mineralization, which is also affected by the process-scale factors of carbon and nitrogen availability, as well as temperature and pH.

Understanding the complexity of the interactions between an ecosystem and its landscape requires that the variables that drive ecosystem processes (shown in Fig. 1.1) be tested as a function of landscape characteristics, such as land cover pattern. Some variables serve dual roles; for example, hydrology can respond to land cover changes, but may also be a driver, affecting the microbial community present at a site, which is related to the denitrification potential. In this study, we asked how the delivery of ecosystem services is impacted by anthropogenic activity (as evidenced by land cover change), as described by the proposed conceptual model (Fig. 1.1). To investigate this we used the following approach: (1) identified the factors (variables) that affect the delivery of the ecosystem services of interest, in this case the soil characteristics that affect N processing (such as denitrification and mineralization); (2) selected sites to represent a gradient of anthropogenic disturbance as indicated by land cover characteristics, ranging from least impacted to heavily impacted land use conditions; and (3) determined if there are differences in the selected soil characteristics as a function of this gradient by categorizing sites into two groups representing high and low disturbance.

1.2 Methods

1.2.1 Wetland Study Sites

For this study, we selected 20 wetland sites in the Mid Atlantic Region, with 10 located in the Ridge and Valley region of Pennsylvania and 10 located in the Appalachian Plateau and Central Lowland of Ohio (Fig. 1.2). Riverine and depression wetland sites were selected within these regions to represent a range of surrounding land-uses and land covers (LULC) (Table 1.1), while keeping wetland Hydrogeomorphic Classification, climate, and geology similar. Floodplain and Headwater Floodplain designations represent similar wetland types (wetlands along headwater streams), located in Ohio and Pennsylvania, respectively. Depression and Riparian Depression represent similar wetland types (closed depressions in a floodplain setting of a headwater stream), located in Ohio and Pennsylvania, respectively.

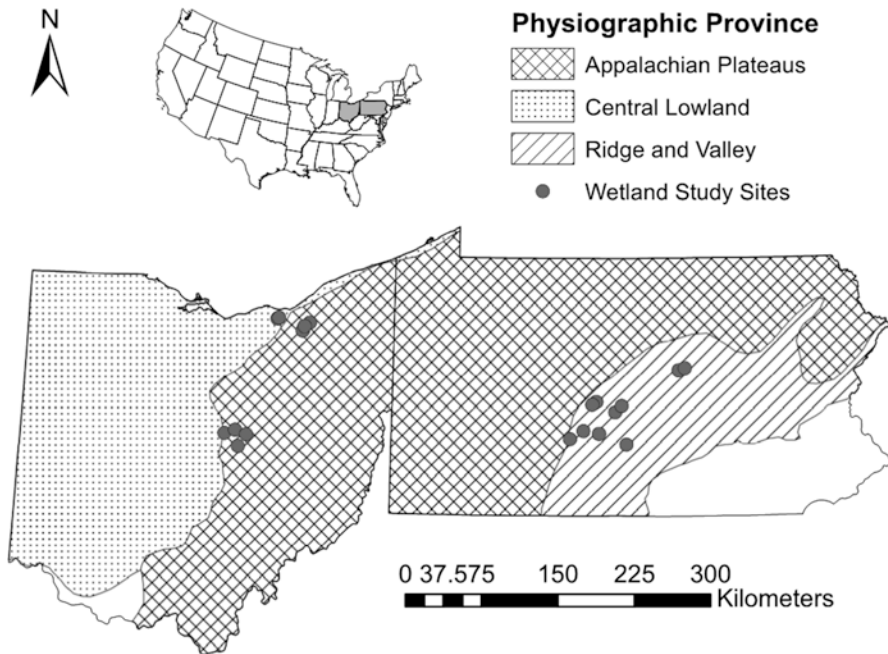


Fig. 1.2 Map of the study sites and the physiographic provinces in which they occur in Ohio and Pennsylvania

1.2.2 *Quantifying Anthropogenic Activity Surrounding Wetland Study Sites*

We used the Landscape Development Intensity (LDI) index, originally proposed by Brown and Vivas (2005), to assess the level of anthropogenic/human activity on wetland study sites. The LDI index estimates potential human impact to a study location by taking a weighted average of the intensity of land use (by LULC classifications) in a defined area surrounding the location. LDI index scores can range from 1 to 8.97, with a score of 1 indicating 100% natural land cover (e.g. forest, open water) and higher scores indicating increasingly more intensive land uses (e.g. agriculture, urban). The LDI scores are calculated based on assignment of land-use coefficients (Table 1.2). Coefficients were calculated as the normalized natural log of energy (embodied energy) per area per time (Brown and Vivas 2005), and defined as the non-renewable energy needed to sustain a given land use type. The LDI is calculated as a weighted average, such that:

$$LDI = \sum \%LU_i * LDI_i$$

where, LDI=the LDI score, %LU_i=percent of total area in that land use *i*, and LDI_i=landscape development intensity coefficient for land use *i* (Brown and Vivas

Table 1.1 Information on study site locations, soil series and the data that was available for use in the CART analyses

Wetland study site information				Data available for CART analyses						
Site name	State	Latitude	Longitude	HGM class ^a	Soil series ^b	N (n =14) ^c	C/S (n =20) ^d	Hyd. (n = 16) ^e	Hydrology data range (M-Y) ^f	No. wells
Laurel Run	PA	40.70264	-77.84851	HF	Atkins silt loam	Yes	Yes	Yes	6-2011 to 5-2013	3
Clarks Trail	PA	41.03170	-77.10830	RD	Udifluvents and fluvaquents, gravelly	Yes	Yes	Yes	7-2010 to 8-2011	2
McCall Dam	PA	41.01590	-77.18160	RD	Philo and atkins, Very stony soils	Yes	Yes	No	N/A	N/A
Secret Marsh	OH	41.31295	-81.58878	D	Sebring silt loam	No	Yes	Yes	7-2010 to 7-2011	2
Shavers Creek	PA	40.64521	-77.92506	HF	Atkins silt loam	Yes	Yes	Yes	7-2010 to 5-2013	3
Tuscarora	PA	40.35920	-77.79330	HF	Atkins silt loam	No	Yes	No	N/A	N/A
Mustang Sally	PA	40.73650	-78.14303	SP/HF	Holly silt loam	Yes	Yes	Yes	9-2011 to 05-2013	3
Kokosing	OH	40.37599	-82.44648	RD	Tioga fine sandy loam	Yes	Yes	Yes	7-2010 to 7-2012	4
Got Milk	PA	40.45390	-78.11040	HF/RD	Atkins silt loam	No	Yes	No	N/A	N/A
Cauldron	PA	40.47810	-78.28917	HF	Holly silt loam	Yes	Yes	Yes	7-2010 to 3-2012	3
Ballfield	OH	40.26875	-82.28320	D	Linwood much	No	Yes	Yes	7-2010 to 3-2012	3
Bat Nest	OH	40.37542	-82.19557	F	Shoals silt loam	Yes	Yes	No	N/A	N/A
Helbender	OH	40.40971	-82.32442	F	Tioga fine sandy loam	Yes	Yes	Yes	7-2010 to 11-2012	4

Cambaris	PA	40.40646	-78.44539	HF	Udifluent-Dystrochrepts complex	Yes	Yes	Yes	7-2011 to 5-2013	3
Skunk Forest	OH	41.38366	-81.50924	F	Tioga loam	No	Yes	Yes	7-2010 to 6-2011	3
R & R	PA	40.71338	-78.18883	HF	Udifluent-Dystrochrepts complex	Yes	Yes	Yes	9-2011 to 05-2013	3
Bee Rescue	OH	41.40832	-81.89191	D	Chagrin silt loam	Yes	Yes	Yes	7-2010 to 6-2011	3
Blackout	OH	41.35251	-81.56918	D	Mitiwanga silt loam	No	Yes	Yes	7-2010 to 7-2011	3
Lizard Tail	OH	41.41611	-81.87608	RD	Chagrin silt loam	Yes	Yes	Yes	7-2010 to 6-2011	2
Vernal Pool	OH	41.41833	-81.87273	RD	Chagrin silt loam	Yes	Yes	Yes	7-2010 to 6-2011	1

^aHydrogeomorphic (HGM) classifications include: *HF* headwater floodplain, *RD* riparian depression, *SP* slope, *F* floodplain, *D* depression

^bSoil series come from the SSURGO Web Soil Survey (accessed 10/15/2013)

^cSites used for categorical regression tree (CART) analyses based on nitrogen pools

^dSites used for CART analyses based on carbon pools and soil accretion rates

^eSites used for CART analyses based on hydrology metrics

^fThe ranges describe the temporal extents (Month-Year) of data collected across all wells at a wetland study site. Thus, all wells at a site might not have spanned the ranges denoted here

Table 1.2 Landscape development intensity coefficients used to calculate the LDI index scores (Brown and Vivas 2005). Land cover categories come from the National Land Cover Database (Homer et al., 2015)

Land cover categories	LDI weights
Water	1
Deciduous Forest	1
Evergreen Forest	1
Mixed Forest	1
Shrub/Scrub	1
Woody Wetlands	1
Emergent Herbaceous Wetlands	1
Grassland/Herbaceous	3.31
Pasture/Hay	3.31
Cultivated Crops	5.77
Developed, Open Spaces	7.18
Developed, Low Intensity	7.18
Barren Land	7.81
Developed, Medium Intensity	8.97
Developed, High Intensity	8.97

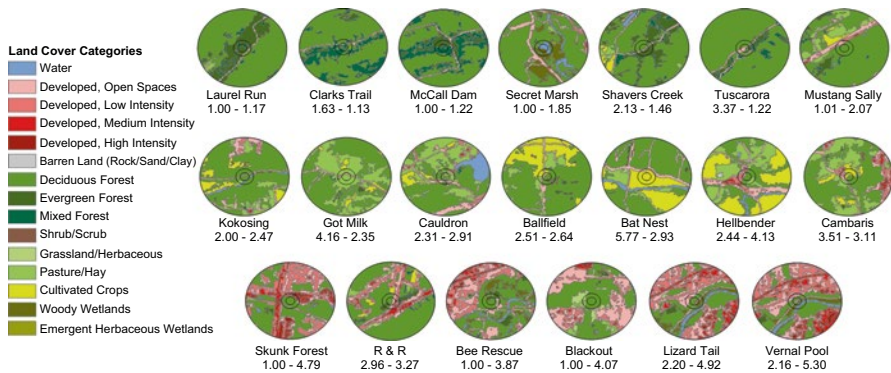
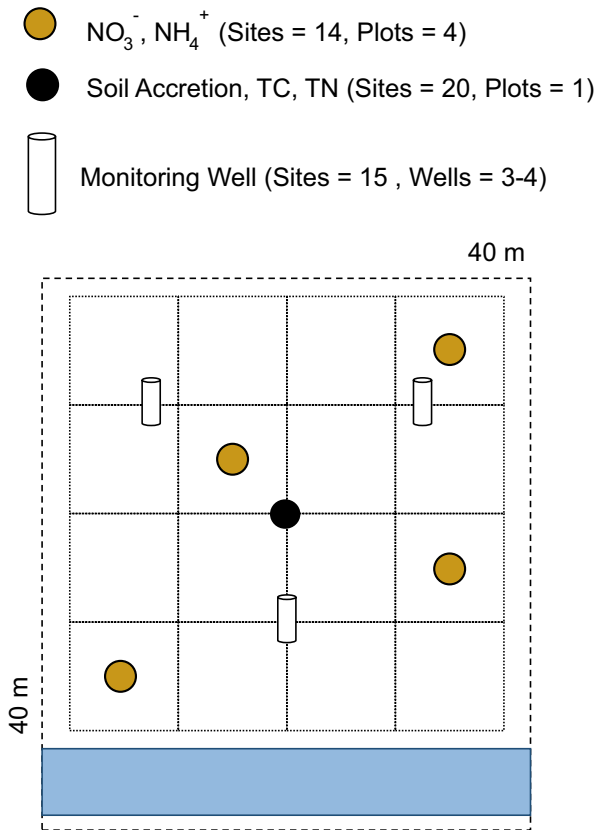


Fig. 1.3 Land cover in the 1000 m radius circles around each site included in this study. Sites are organized into rows according to their dominant land cover setting arranged, from *top* to *bottom*, by natural, agricultural, and urban/developed land use. Land Development Intensity Index (LDI) values at 100 m and 1000 m, respectively, are shown below each land cover *circle*

2005). This provides an integrative measure of land-use for a defined area around a site in a single score rather than looking at each land-use class separately.

The LDI was calculated using the 2011 National Land Cover dataset (NLCD) (Homer et al. 2015) for three landscape scale assessment areas in 100-m, 200-m and 1-km radius circles around the center of the wetland assessment area (Fig. 1.3). Percent area of LULC classifications for each wetland assessment area was extracted from the NLCD using ArcGIS (version 10.3, Esri, Inc.).

Fig. 1.4 Schematic of the sampling design used at each site



1.2.3 Field and Laboratory Measurements

Data were collected on ecosystem service measurements related to nitrogen cycling (e.g., denitrification and nitrogen mineralization), including related measures of soil carbon, and hydrologic variability. The generalized sampling design for each site is presented in Fig. 1.4.

1.2.3.1 Nitrogen Pools

Nitrogen processing in wetland ecosystems is spatially dynamic. As such, we implemented a spatial sampling regime to measure average site-level nitrogen pools in the fall of 2011 in 14 of our wetland study sites. Ten m by 10 m plots were established in a grid over a 40 m by 40 m wetland assessment area at each study site (Fig. 1.4), resulting in 16 plots. Four sampling plots were randomly selected from this pool of 16 plots for nitrogen pool analysis.

In each sampling plot, a soil core was collected at each of the four subplots using an 8-cm diameter PVC tube to a depth of 5 cm, placed into a re-sealable plastic bag and stored on ice for transport to the laboratory. The core was used to measure extractable pools of ammonium (NH_4^+) and nitrate (NO_3^-). All samples were stored at 4 °C until processing, which occurred within 48 h of field collection. Soils were weighed and subsequently homogenized by pushing through a 2-mm sieve. Approximately 20.0 g of wet mass soil was sampled in duplicate or triplicate for gravimetric water content. Samples were dried at 60 °C until a constant mass was reached.

A second subsample, consisting of 20.0 g of wet mass soil was weighed out in duplicate or triplicate for extraction of N species (i.e., NH_4^+ , NO_3^-), following Keeney and Bremner (1966). Samples were extracted using 2 M KCl, agitated for 1 h on an orbital shaker and left to settle for > 12 h, before the extractant was filtered through 1.5 μm binder-free glass fiber filters to remove any remaining soil particles. Filtrate was stored at -20 °C in polypropylene test tubes until colorimetric measurement were made using a Lachat QuikChem® 8500 Series 2 Flow Injection Analysis. Final soil extractable NH_4^+ and NO_3^- concentrations were calculated on a dry weight basis and corrections were made for small concentrations NH_4^+ and NO_3^- found on filters.

Approximately 1 month later (i.e., 24–29 days later), a second core, encapsulated by resin bags, was taken ~0.25 m away from the first core as part of a nitrogen mineralization incubation (methods based on Noe 2011). Soil extractable NH_4^+ and NO_3^- were significantly related at the site-level across sampling periods (extractable NH_4^+ adj. $R^2=0.65$, p-value < 0.001, extractable NO_3^- adj. $R^2=0.77$, p-value < 0.001) and as such, only initial soil core samples are used in the subsequent analyses.

1.2.3.2 Soil Accretion and Carbon Pools

A soil core, measuring 8.5 cm in diameter and 40–50 cm in depth (depending on site conditions), was collected from the center of each wetland assessment area in the summer of 2011 (Fig. 1.4). Cores were collected using a hand-operated stainless steel corer designed for use in freshwater wetland soils. Each core was extruded and sectioned into 2-cm increments for analysis. Increments were stored in re-sealable plastic bags and placed on ice while being transported to the laboratory for analysis. Increments were dried at 60 °C until a constant mass was reached.

Cesium-137 was measured on each increment by gamma spectroscopy of the 661.62 keV photopeak (Craft and Richardson 1998). The depth of the ^{137}Cs maximum in each core corresponds to the 1964 period of maximum deposition of radioactivity from aboveground nuclear weapons testing (Reddy and DeLaune 2008). This peak was used to calculate the medium-term (47-year) rate of vertical soil accretion. Only cores that contained interpretable ^{37}Cs profiles were used. Soil accretion rates ($\text{mm} \cdot \text{year}^{-1}$) were calculated as follows (Moshiri 1993):

$$\text{Soil accretion rate} (\text{mm year}^{-1}) = \frac{\text{Depth the } ^{137}\text{Cs peak}}{2011 - 1964}$$

Each core section was also analyzed for total carbon (TC %) and TN. A dried subsample was ground passed through a 0.25 mm sieve for analysis by dry combustion using a Perkin-Elmer 2400 Series CHNS/O elemental analyzer. TC and TN was averaged for the top 10 cm of soil and TC:TN ratios were calculated.

1.2.3.3 Hydrologic Metrics

Three to four Ecotone WM-1-m automatic water level monitoring wells (Remote Data Systems, Inc. Model #: WM16k1015) were established at 16 of the wetland study sites. When possible we positioned 3 of the wells in an equilateral triangle with the base of the triangle towards adjacent hillslopes. Water level recordings were collected at 3-h intervals between 2010 and 2013, with each site-well varying in its collection period.

Hydrology metrics were selected to provide insight into groundwater variation, biogeochemical processes, and environmental stress. To quantify these dynamics 11 hydrology metrics were calculated: average water level, relative to ground surface (cm), maximum water level, relative to ground surface (cm), minimum water level, relative to ground surface (cm), the 25th, 50th, and 75th percentile water level, relative to ground surface (cm) of all recorded water levels over the sampling period for an individual well, percent time the water level was above ground (%), percent time the water level was in the upper 10 cm of the soil profile (%), percent time water level was in the upper 30 cm of the soil profile, percent time the water level was between 10 and 30 cm (%), the mean water level difference over a 24-h period, and mean water level difference over a 7-day period.

Average water level provides a general measure of a site's hydroperiod during a given year. The metrics 'percent time the water level is in the upper 30 cm of the soil profile' and 'percent time above ground' provide information specifically relevant to water availability to vegetation and biogeochemical processes. The "percent time" metrics were calculated as the number of data points equal to or above the depths (i.e., ground level, 10 cm, and 30 cm) with respect to the total number of data points. Mean 7-day and 24-h differences provide insights into water level stability and temporal reaction rates. These two metrics were calculated once per day on a rolling basis. These hydrology metrics were calculated for individual wells, where duration was unique to each well. Final metrics were calculated as the average metrics across wells within a site; spatial variability of hydrology metrics within a given site was relatively low.

1.2.3.4 Data Analysis

We utilized classification and regression tree (CART) analysis to explore thresholds between LDI metrics at the three landscape assessment scales utilizing drivers of nitrogen processes (nitrogen, carbon, and hydrology metrics) as explanatory variables. CART is suited for this because it takes into account non-linear and

high-order interactions that might be missed in simple linear regression analyses. CART uses binary cluster trees to explain variation in a single response variable by one or more explanatory variable (De'ath and Fabricius 2000). For each landscape assessment scale, CARTs were further broken into three predictor groups, including predictors related to soil carbon (i.e., soil accretion, TC, and C:N ratios), those related to soil nitrogen (i.e., TN, extractable NH_4^+ , and extractable NO_3^-), and hydrology predictors (i.e., 10 metrics listed above in *Hydrologic Metrics*). CART analyses were performed in JMP® Pro (Version 12.0.1, SAS Institute, Inc.) Only the first split (i.e., strongest predictor) in each CART was used for discussion. CARTs were also used to examine threshold LDIs with the three landcover classes (forested, agriculture, and urban).

1.3 Results

Our collection of sites provided a diversity of land cover settings (Fig. 1.3); sites varied in their LDIs at our specified spatial scales of assessment (100 m, 200 m, and 1000 m), and in the change in LDI values with increasing distances from the site (i.e., the shape of the curves across this distance; Fig. 1.5). Sites basically fell into three groups based on the shape of the LDI curves: those that are in primarily low-LDI land cover settings from 100 to 1000 m (Clark's Trail, Shaver's Reference, Tuscarora, Laurel Run, McCall Dam in PA; Kokosing and Secret Marsh in OH); those that begin in moderate LDI settings and whose LDI decreases as one proceeds outward from the site (Got Milk, Cambaris, R&R, Cauldron, and Mustang Sally in

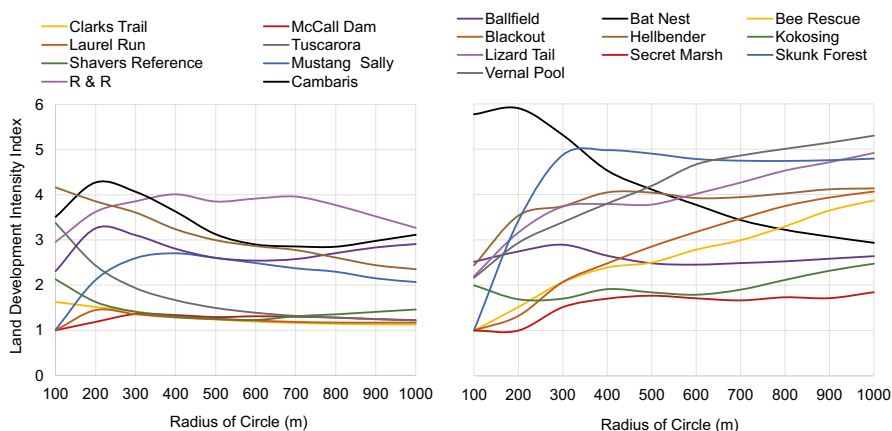


Fig. 1.5 Plots of LDI scores with increasing distance around each site. LDI scores were calculated at 100 m intervals across the distance from 100 to 1000 m. Pennsylvania sites are shown on the *left* and Ohio sites on the *right*

PA; Ballfield in OH); and those that begin in relatively high LDI settings and whose LDI increases as one proceeds outward from the site (Hellbender, Lizard Tail, Vernal Pool, Skunk Forest, Bee Rescue, Blackout in OH). One outlier (Bat Nest in OH) begins at the highest LDI and decreases to moderate LDI levels. These groups roughly correlate with the three groups of sites characterized as Forested, Agricultural, and Developed in Fig. 1.3, due to the direct relationship between predominant land cover (forested, agriculture, and urban) and the LDI weights (Table 1.2). There is also a difference in LDI pattern between the Pennsylvania and Ohio sites, which may be related to the two physiographic provinces in which the sites are located. The Pennsylvania sites are in the Ridge & Valley Physiographic province, characterized by long, unbroken forested ridges with limestone or shale valleys. A 1000 m radius circle will often just fit into the valley bottoms, and may trend toward the bottoms of the forested ridges at the far extent of the circle in which the LDI is calculated. In contrast, the Ohio sites are generally located in the Glaciated Allegheny Plateau, with a rolling hill topography and lacking the strict topographic constraints on land cover of the Ridge & Valley (i.e., activities such as agriculture and urban development are not constrained by the high slopes of ridges).

Our investigation of linkages between land cover patterns and wetland and process scale variables that are relevant for the provision of water quality services is initially organized by the spatial scale at which the land cover patterns are determined. For example, land cover patterns within a 1 km buffer around the wetland site may be predictive of soil accretion rates because of the increase in potentially erodible areas and the accumulation of the runoff volumes needed to transport it from the contributing watershed. In contrast, land cover patterns within a 100 m buffer around the wetland may be predictive of hydrologic characteristics such as median depth to groundwater, due to the importance of local variability in topography, soil, and vegetation characteristics. We utilized CART to determine which variables best separated high from low disturbance sites, for each spatial scale at which land cover patterns were determined (100 m, 200 m, 1 km), and within each category of water quality variable (hydrology, nitrogen and carbon). Results for 100 m, 200 m, and 1000 m are shown in Fig. 1.6. The results are described by each variable category, as follows.

1.3.1 Nitrogen and Carbon (Soil Properties Important to N Cycling)

We looked at several measures of soil carbon (C) and nitrogen (N) that are important to N cycling and relevant for the provision of water quality services in order to determine how they varied with anthropogenic activity, as represented by the LDI (Table 1.3). CART analysis identified high and low disturbance sites (Fig. 1.6), and indicated thresholds for the soil measures related to soil N and C. At the scale of 1000 m, the LDI index split the sites into two groups based on ammonium (N-related

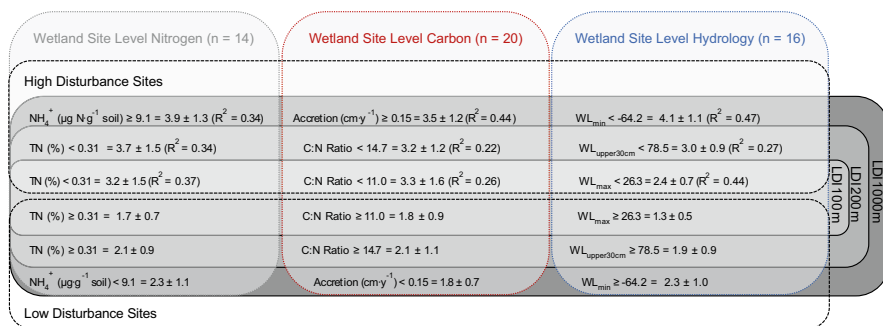


Fig. 1.6 Results of the classification and regression trees (CARTs) showing the relationship between the response variable of anthropogenic disturbance (LDI index scores at 100 m, 200 m, and 1000 m) and explanatory variables of nitrogen processing (soil nitrogen, soil carbon and hydrology). Based on the thresholds identified in the LDI index scores, sites were separated into High and Low Disturbance groups. Values are shown for the factors identified in the first split along with the LDI thresholds and R^2 values for relationships that are significant. LDI thresholds were consistent across scales with mean LDIs at High Disturbance Sites greater than 3 for all parameters except Maximum Water Level (Average LDI=2.4). Mean LDIs in Low Disturbance Sites ranged from 1.7 to 2.3

measures) and soil accretion rates (C-related measures) such that high disturbance sites were characterized by relatively high levels of extractable ammonium ($\geq 9.1 \mu\text{g g}^{-1}$ soil) and higher soil accretion rates ($\geq 0.15 \text{ cm y}^{-1}$). The high disturbance sites were defined as those with LDI values above 3.9, indicating that land use around the sites was, at minimum, as intensive as agricultural land. The strong links between high LDI scores and ammonium are illustrative of the excessive anthropogenic loading of nitrogen sources onto our landscapes.

Higher LDI scores in the 1000 m area around a site were also predictive of higher sediment accretion rates due to anthropogenic disturbance in the local watershed such as agricultural or construction activities. This is correlated with the flashy hydrology of the High Disturbance sites (see below) that increases the transport and deposition of sediment and organic matter as materials from upstream/up gradient accumulate over longer flow distances and are transported into the wetlands. Over the past 25–50 years, elevated sediment deposition rates have been observed in riparian wetlands with significant anthropogenic disturbances (Johnston et al. 1984; Hupp et al. 1993; Hupp and Bazemore 1993; Kleiss 1996; Wardrop and Brooks 1998). Studies have documented rates of sedimentation ranging from 0.07 to 5 cm year^{-1} in forested riparian wetlands affected by land use disturbance (Hupp et al. 1993; Hupp and Bazemore 1993; Kleiss 1996). In Central Pennsylvania, Wardrop and Brooks (1998) showed that sediment deposition ranged from 0 to 8 cm year^{-1} across four freshwater hydrogeomorphic subclasses with varying levels of land use disturbance. It is thought that this accelerated sedimentation overloads the assimilative capacity of these wetlands (Jurik et al. 1994; Wardrop and Brooks 1998; Freeland et al. 1999) and interferes with other ecosystem services wetlands provide.

Table 1.3 Mean values for measures of soil nitrogen, soil carbon, and hydrology by land cover category. Sites were categorized by dominant land cover in the 1000 m radius area surrounding each as shown in Fig. 1.3

Study parameters	Units	Surrounding landscape				All sites (n=20)
		Forested (n=7)	Agricultural (n=7)	Urban (n=6)		
<i>Soil properties</i>						
TC	%	9.27±7.74	6.96±6.98	12.28±15.29		9.36±10.03
TN	%	0.47±0.33	0.62±0.62	0.66±0.43		0.58±0.46
TC:TN	ratio	18.32±3.75	11.24±2.24	15.27±6.95		14.93±5.29
Extractable NO ₃ ^{-a}	µg N-NO ₂ ⁻ + NO ₃ ⁻ · g ⁻¹ soil	0.05±0.11	5.71±5.25	0.22±0.45		2.12±4.03
Extractable NH ₄ ⁺	µg N-NH ₄ ⁺ · g ⁻¹ soil	8.07±6.83	3.04±0.53	21.41±12.41		10.08±10.49
Soil accretion rate	cm · year ⁻¹	0.09±0.06	0.2±0.11	0.25±0.14		0.18±0.12
<i>Hydrologic properties^b</i>						
Mean water level	cm	-4.41±39.61	-47.25±25.85	-21.38±23.97		-24.16±33.15
Maximum water level	cm	41.05±24.87	17.98±10.7	33.59±32.79		31.04±25.43
Minimum water level	cm	-25.81±28.26	-73.83±20.22	-66.65±31.15		-56.13±33.18
Time water level above 30 cm	%	76.82±0.41	33.76±0.33	69.30±0.29		60.54±0.37
Time water level above 10 cm	%	58.60±0.47	16.70±0.24	50.11±0.27		42.32±0.36
Time water level above ground level	%	43.01±0.44	6.17±0.1	33.19±0.31		27.81±0.33
Time water level between 10 and 30 cm	%	18.30±0.26	17.23±0.15	19.36±0.12		18.36±0.17
Mean 24-h water level difference	cm	2.04±0.45	3.29±0.33	3.54±2.31		2.99±1.52
Mean 7-day water level difference	cm	7.63±0.88	13.1±2.43	13.92±9.64		11.7±6.39

^aExtractable N concentrations were averaged across 5 sites for “Forested” and “Agricultural” landscapes, and across 4 sites for “Urban” landscapes. Thus, 14 of the 20 sites were used in the “All Site” summary

^bHydrologic metrics were averaged across 5 sites for “Forested” and “Agricultural” landscapes, and across 4 sites for “Urban” landscapes. Thus, 16 of the 20 sites were used in the “All Site” summary

At smaller scales (LDIs for 100 and 200 m distances) the Low Disturbance group was defined by higher levels of soil TN (≥ 0.31 %TN) and higher soil C:N ratios, corresponding to the significantly higher mean soil C levels in these sites. TN and TC are lower overall in the High Disturbance sites, which is reflected in the lower C:N values. Lovette et al. (2002) reported that nitrate release from soils in forested watersheds is strongly affected by the C:N ratio of its soils; as C:N ratios increased, nitrate export decreased. These wetland sites may be behaving the same way; the high demand of heterotrophic bacteria for N when C:N ratios are high, leaves less N available (as evidenced by lower levels of extractable ammonium in the Low Disturbance sites) for processes such as nitrification and denitrification. This predisposes the more disturbed sites to perhaps act as biochemical “hot-spots” in the processing of N because of the convergence of the substrates and hydrological flows (see below) that are needed for biochemical reactions (McClain et al. 2003).

1.3.2 Hydrology

In general, the majority of hydrology metrics in this study are highly correlated with one another, as can be expected (Table 1.4). For example, a great number of the metrics are descriptive of general position of the water table (Average Water Level, Minimum Water Level, 25th Percentile, 50th Percentile, 75th Percentile, %Time Upper 30 cm, %Time in Upper 10 cm, %Time Above Ground), and they are highly correlated with each other (all $R^2 > 0.56$). In general, four metrics are remarkably poorly correlated with these general water table metrics: Maximum Water Level, %Time 10–30 cm, Mean 24-h Difference, and Mean 7-Day Difference. Of these four metrics, only Mean 24-h Difference and Mean 7-Day Difference are highly correlated ($R^2 = 0.95$). Based on this, we would propose that any general description of hydrologic character would include, at a minimum, a metric for average position of the water table, a metric to describe inundation or maximum water level, and a metric to describe flashiness. Other metrics could be added to indicate specific conditions: for example, %Time Above Ground as a metric of inundated and highly anaerobic conditions, and % Time 10–30 cm as a descriptor of optimal aerobic/anaerobic conditions within a zone of carbon availability.

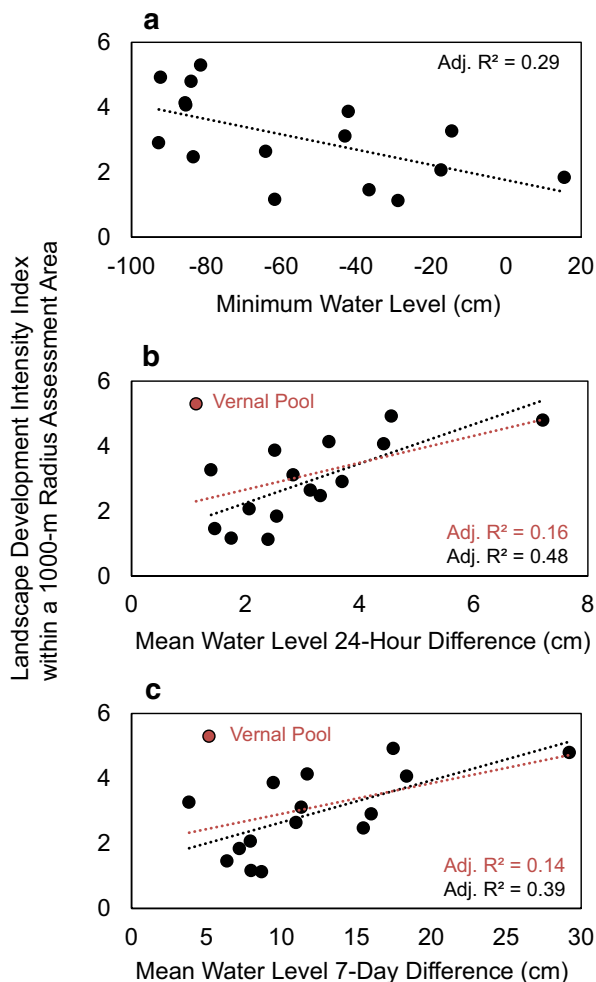
In general, the CART analyses utilizing hydrology metrics differentiated high from low disturbance sites better than nitrogen and carbon metrics, as evidenced by relatively high R^2 values for LDI values at 100 m and 1000 m (Fig. 1.6). The differentiation between high and low disturbance sites is most pronounced in terms of Minimum Water Level for LDI at 1000 m, where low disturbance sites with an average LDI of 4.1 are characterized by a Minimum Water Level that is within the upper 64 cm relative to ground surface, while high disturbance sites with an average LDI of 2.3 are characterized by a Minimum Water Level that goes below 64 cm from the ground surface. The relationship between Minimum Water Level and LDI at 1000 m was linear, with a significant negative slope (i.e., Fig. 1.7a, Adj $R^2 = 0.29$, $n = 16$, p -value = 0.0193). Urban and suburban land cover (expressed in this study as higher

Table 1.4 Correlation matrix of hydrologic metrics used in this study

Hydrologic metrics	WL _{max} (cm)	WL _{min} (cm)	25th percentile	50th percentile (median)	75th percentile	WL _{upper30cm} (%)	WL _{upper10cm} (%)	WL _{aboveground} (%)	WL _{10-30cm} (%)	WL _{24-HrDiff} (cm)	WL _{7- DayDiff} (cm)
WL _{avg} (cm)	0.26	0.81	0.91	0.97	0.92	0.90	0.90	0.87	0.03	-0.35	-0.45
WL _{max} (cm)		0.18	0.04	0.27	0.40	-0.04	0.24	0.40	-0.58	-0.20	-0.20
WL _{min} (cm)			0.88	0.69	0.58	0.68	0.67	0.75	0.03	-0.53	-0.62
25th percentile				0.81	0.70	0.82	0.80	0.77	0.07	-0.38	-0.47
50th percentile (median)					0.96	0.91	0.89	0.83	0.06	-0.37	-0.47
75th percentile						0.84	0.85	0.80	0.02	-0.26	-0.34
WL _{upper30cm} (%)							0.89	0.74	0.27	-0.35	-0.43
WL _{upper10cm} (%)								0.92	-0.19	-0.33	-0.43
WL _{aboveground} (%)									-0.34	-0.37	-0.47
WL _{10-30cm} (%)										-0.06	-0.03
WL _{24-HrDiff} (cm)											0.98

Numbers in bold indicate R values that are significant at $p \leq 0.05$. Numbers in bold indicate correlation coefficients (R) that are significant at $p \leq 0.05$. Metrics include: WL_{avg} average water level, relative to ground surface (cm), WL_{min} minimum water level, relative to ground surface (cm), WL_{max} maximum water level, relative to ground surface (cm), WL_{upper30cm} percent time water level was in the upper 30 cm of the soil profile, WL_{upper10cm} percent time the water level was in the upper 10 cm of the soil profile, WL_{aboveground} percent time the water level was above ground (%), WL_{10-30cm} percent time the water level was between 10 and 30 cm (%), WL_{24-HrDiff} the mean water level difference over a 24-h period (cm), and WL_{7-DayDiff} mean water level difference over a 7-day period (cm)

Fig. 1.7 Regression analyses showing the relationship between LDI index scores and (a) Minimum Groundwater Level, (b) Mean 24-h Difference in Water Level, and (c) Mean 7-day Difference in Water Levels. Note that the site Vernal Pool was an outlier in panels (b) and (c); the regression line including this site is shown in *red*, and excluding it is shown in *black*



LDI values) has been associated with stream incision and lowered water tables (Groffman et al. 2002), potentially inhibiting the interaction of nitrate-rich groundwater with near-surface soils, with an accompanying low probability of denitrification. However, some studies have noted relatively higher levels of denitrification in agricultural settings (Xiong et al. 2015), indicating that perhaps at intermediate LDI values, both nitrate rich groundwater from upgradient agricultural areas is able to interact with these surface soils. The LDI at 200 m split sites based on the percent of time the water level was within the top 30 cm, with high disturbance sites exhibiting lower percentages of time within this rooting zone, and again is reflective of lowered water tables. The LDI at 100 m split high and low disturbance sites fairly well, based on Maximum Water Level, with low disturbance sites having Maximum Water Levels of 26 cm or greater above ground surface. The correlation of Maximum Water Level

to LDI at 100 m may be indicative of connectivity to nearby streams or collection of precipitation and groundwater, since both floodplain and depression sites are combined in the analyses. In the case of floodplain settings, greater connectivity to streams has been associated with higher inputs of sediment, sediment-N, and ammonium, and greater soil net ammonification, N mineralization, and N turnover, but not denitrification potential (Wolf et al. 2013). These wetter sites may be too anaerobic (as evidenced by high Soil C values) for appreciable denitrification.

Although metrics describing water level stability did not characterize the initial splits in the CART analyses, on average Mean Water Level 24-h and 7-day differences were lower in forested sites compared to agricultural and urban landscape sites (Table 1.1). Using linear regression we see a positive relationship between both flashiness metrics for LDI at 1000 m (Fig. 1.7b, c). One outlier in this regression, Vernal Pool, had only one working well for metric calculations. Removing this outlier, the positive relationships between mean water level 24-h (p-value=0.0025) and 7-day differences (p-value=0.0079) and LDI at 1000 m were statistically significant. The link between relatively high LDIs and relatively large fluctuations in water levels is reflective of a hydrologic pattern that is expected to be controlled at a larger spatial scale, since flashiness should be an expression of a watershed wide characteristic. It is also reflective of the differences in sediment delivery as measured by soil accretion rates. The 'boom and bust' hydrology that characterizes flashy hydrology moves sediment and allochthonous carbon in times of higher flows, depositing it in wetlands as flows subside. However, lack of flashiness and constancy of saturation of the rooting zone may inhibit the adjacency of aerobic and anaerobic zones necessary for denitrification.

1.4 Summary and Conclusions

We determined differences in soil and hydrology characteristics important to water quality services as a function of a gradient of anthropogenic activity, as expressed by the LDI. Our study is unique in that it: (1) investigates anthropogenic activity, as expressed by LDI, as a gradient rather than a categorical variable (e.g., agricultural versus forested), thus allowing determination of potential thresholds of LDI where there are important differences in nitrogen, carbon, and hydrology characteristics; and (2) it seeks to identify the specific process and site-scale variables relevant to water quality improvement that can be inferred by land cover patterns at varying distances from the site. Thus, we can identify useful proxies for the level of water quality improvement services we can expect from a given site.

Thresholds of LDI were determined via the CART analyses that separated sites into two general classes of high and low disturbance wetlands, with associated differences in TN, NH_4^+ , Soil Accretion, C:N, Maximum Water Level, Minimum Water Level, and %Time in Upper 30 cm (Fig. 1.6). The LDI thresholds were remarkably consistent, with High Disturbance Sites characterized by average LDIs greater than 3 for all parameters except Maximum Water Level (Average LDI=2.4), indicating agricultural and urban landcovers. Low Disturbance Sites had average LDIs of

1.7–2.3, indicating forested conditions. Only 6 sites exhibit LDIs at both 100 m and 1000 m that are within this range (Laurel Run, Clarks Trail, McCall Dam, Secret Marsh, Shavers Creek, and Mustang Sally), and are consistent with an overall forested setting (Fig. 1.3). These Low Disturbance Sites exhibit relatively higher TN, lower NH_4^+ , lower Soil Accretion, higher C:N, higher Maximum Water Level, shallower Minimum Water Level, and higher %Time in Upper 30 cm than the remaining sites. Given the interaction of the carbon, nitrogen, and hydrology factors, we would expect a water quality process such as denitrification to be relatively lower in forested settings, due to the low available nitrogen (associated with high C:N) and constant and saturated conditions. Conditions for maximum denitrification may be found in agricultural settings, where high nitrate groundwater can interact with surface soils through a wetting and drying pattern.

The use of land cover patterns, as expressed by LDI, provided useful proxies for nitrogen, carbon, and hydrology characteristics related to provision of water quality services. LDIs at 100 m and 200 m were best separated into groups of high and low disturbance sites by factors expected to be proximal or local in nature, such as connectivity to streams or inundation from local runoff (Maximum Water Level, %Time Upper 30 cm) and levels of primary productivity and vegetation (C:N, TN). LDIs at 1000 m predicted factors that could be related to larger scale land cover patterns that are more distal in nature, such as Soil Accretion (reflecting erodible soils and flashy hydrology for transport), NH_4^+ (overall eutrophication), and Minimum Water Level (depression of water tables).

While all wetland types serve valuable roles in their watershed, headwater wetland/stream systems may contribute a disproportionate share to watershed functioning and the larger drainage areas and regional watersheds into which they drain. Headwater streams determine much of the biogeochemical state of downstream river networks (Brinson 1993), in part because, for example, in the U.S. low order streams account for 60–75 % of the total stream and river lengths, making their riparian communities of extreme importance for overall water quality (Leopold et al. 1964). We have demonstrated that anthropogenic activity surrounding these wetland systems leads to differences in the primary carbon, nitrogen, and hydrology drivers of the water quality ecosystem services that they are valued for. In addition, we have demonstrated the utility of the LDI as a proxy for these same drivers. Thus, land cover patterns, as expressed by the LDI, should be taken into account when creating, restoring, or managing these systems on a watershed scale.

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Chapter 2

Nutrients Tracking and Removal in Constructed Wetlands Treating Catchment Runoff in Norway

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Johannes Deelstra, and Ketil Haarstad

Abstract Water quality problems in Norway are caused mainly by high phosphorus (P) inputs from catchment areas. Multiple pollution sources contribute to P inputs into watercourses, and the two main sources in rural areas are agricultural runoff and discharge from on-site wastewater treatment systems (OWTSs). To reduce these inputs, Constructed wetlands (CWs) treating catchment runoff have been implemented in Norway since early 1990s. These CWs have been proven effective as supplements to agricultural best management practices for water quality improvements and therefore there are more than 1000 CWs established in Norway at present. This study aims to present some overall data on the present status of CWs treating catchment runoff in Norway, and in particular recent results of source tracking and retention of sediments and total phosphorus (TP) in a model, full-scale, long-term operated CW, which in practice treats runoff from a typical rural catchment with pollution from both point and diffuse sources. Nutrient contributions from agricultural runoff and OWTSs have been quantified in eight catchments, while the source tracking and retention of sediments and P has been studied in the model CW. P runoff in the catchments was largely affected by precipitation and runoff situation, and varied both throughout the year (every single year) and from one year to another. Annual TP contribution that originates from OWTSs was in general limited, and only 1 % in the catchment of the model CW. Monthly contribution, however, was higher than 30 % during warm/dry season, and cold months with frost season. For the purpose of source tracking study, faecal indicator bacteria (reported in terms of *Escherichia coli* - *E. coli*) and host-specific 16S rRNA gene markers *Bacteroidales* have been applied. High *E. coli* concentrations were well associated with high TP inputs into waterbodies during dry or/and cold season with little or no agriculture runoff, and further microbial source tracking (MST) tests proved human contribution. There are considerable variations in retention of sediments and TP in the

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CW between the years, and the annual yearly retention was about 38 % and 16 %, respectively. During the study period, the average monthly retention of sediments and TP was 54 % and 32 %, respectively. *E. coli* concentrations were also reduced in water passing the CW. The study confirmed that runoff from agricultural areas is the main P source in watercourses, however, discharges from OWTS can also be of great importance for the water quality, especially during warm/dry- and cold/frosty periods. Small CWs treating catchment runoff contribute substantially to the reduction of sediments, TP and faecal indicator bacteria transport into water recipients.

Keywords Catchment runoff • Agricultural runoff • Constructed wetlands • Phosphorus • Sediments • On-site wastewater treatment systems • Microbial source tracking

2.1 Introduction

Water quality problems in Norway are caused mainly by high phosphorus (P) inputs from catchment areas, as P is the main nutrient limiting eutrophication and algal blooms in waterbodies. In order to reduce the P transport into water recipients it is of great importance to track the pollution sources, find their localisation and contribution to the contamination input. Based on this, specific measures can be prioritized, planned and implemented.

It is a fact that maximal land use for maximal benefit is one of the goals in the modern agriculture, which has expansively extended areas through the removal of natural buffer systems such as wetlands, small streams, and vegetative buffer zones along streams. This has led to an increased erosion and losses of nutrients from agricultural areas into the watercourses. Geographically, Norway is located far north and has cold climate conditions. Due to the rough topography with mountains and large areas covered with forest, agricultural lands constitute only 3 % of the entire country. Despite these conditions, runoff and diffuse pollution from agricultural areas are one of the major anthropogenic sources of nitrogen, P, and sediment inputs to surface waters (Solheim et al. 2001; Selvik et al. 2006). In the rural catchments, there are also different kinds of point source pollution, e.g. landfill leachates, industrial sewages and domestic wastewater. Although the multiple pollution sources of nutrient inputs into watercourses, the two main sources are agricultural runoff and discharge from on-site wastewater treatment systems (OWTSs). In particular, the discharge of inadequately treated wastewater contributes with high P concentrations into watercourses.

Agriculture Best Management Practices are necessary but often insufficient measures to achieve acceptable water quality (e.g. as set by environmental goals related to the Water Framework Directive), (Direktoratsgruppa 2009). Measures such as vegetated buffer zones and small constructed wetlands (CWs) can be good supplements to improve the water quality.

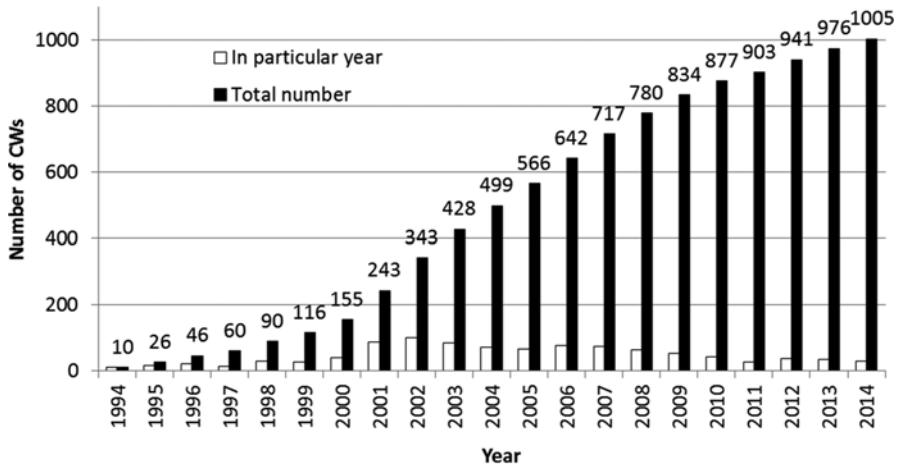


Fig. 2.1 Number of constructed wetlands established to reduce runoff from agricultural lands in Norway

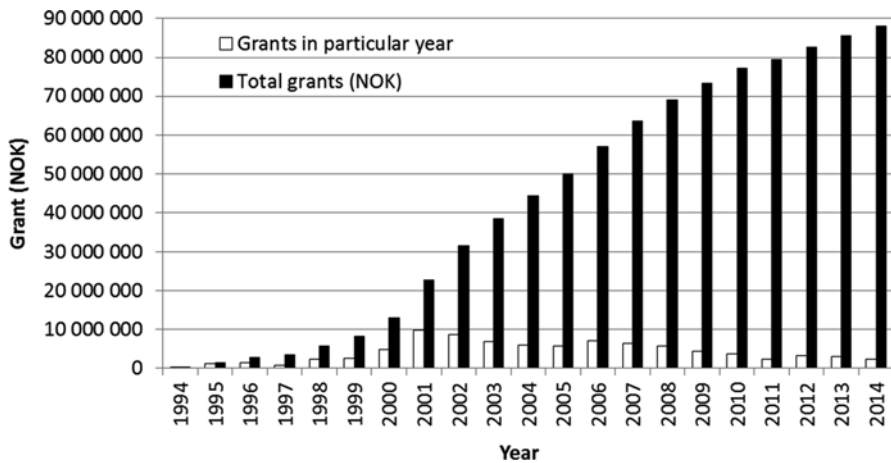


Fig. 2.2 Total governmental subsidies in Norwegian crowns (NOK) per year and in total granted to constructed wetlands treating agricultural runoff

CWs treating agricultural runoff has been in operation in Norway since early 1990s and at present, there are more than 1000 CWs established to reduce sediment, nutrients, pesticides and other pollutants from agricultural runoff (Fig. 2.1). During a period of 20 years the government have spent in total about 88 million Norwegian crowns (NOK) to subsidize the CWs for agricultural runoff (Fig. 2.2), and total costs are assumed to be about 150 mill NOK. Costs per CW varies from about 26.000 NOK to 124.000 NOK, and average cost per CW is approximately 87.500 NOK (Fig. 2.3).

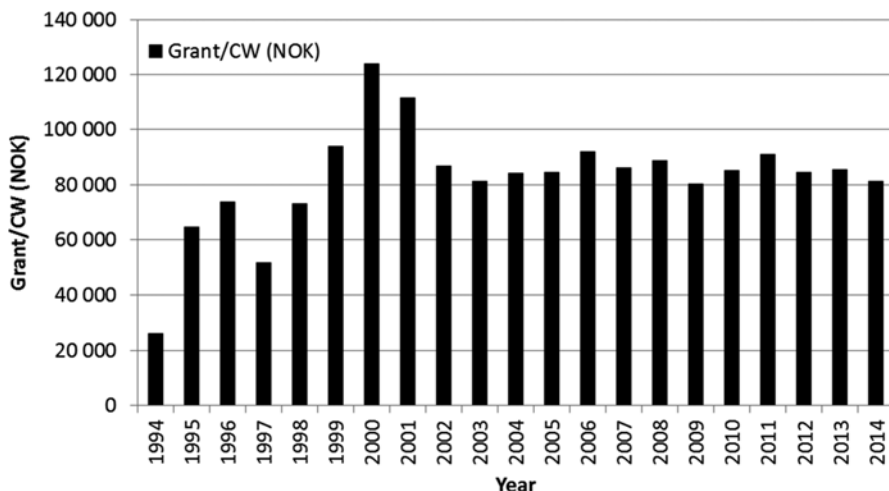


Fig. 2.3 Governmental subsidies in Norwegian crowns (NOK) per year during the last 20 years granted to constructed wetlands treating agricultural runoff

The CWs treating agricultural runoff are usually constructed by expanding the width of natural streams. At the inlet of the CW, the stream water flows into a sedimentation pond. From the sedimentation pond, water passes through a sprinkling zone, and then through one or more vegetated wetland filters, divided with sprinkling zones. Due to the typical small-scale Norwegian agriculture and the landscape with rough topography, CWs are often quite small (<0.1 % of the catchment area). The size of CWs is one of the crucial factors limiting overall treatment efficiency. Norwegian studies show that retention of total phosphorus (TP), both particulate P and dissolved P, increase with increasing area of the CW (Braskerud et al. 2005). The retention of sediments, nutrients and pesticides in different CWs also varies due to other factors like design principles, soil types in the catchment, hydraulic loads, and locations along the streams (Braskerud and Blankenberg 2005; Blankenberg et al. 2007, 2008; Elsaesser et al. 2011). Over the years, the CW will fill up with sediments, and to maintain good treatment efficiency, it is required to empty the CWs periodically (Blankenberg et al. 2013). Braskerud (2001) showed that average retention in six CWs in Norway varied from 45 to 74 % for soil particles and 21–44 % for TP.

The main objectives of this study is to present some overall data on the present status of CWs treating catchment runoff in Norway, and in particular recent results of source tracking and retention of sediments and TP in a model, full-scale, long-term operated CW, which in practice treats runoff from a typical rural catchment with pollution from both point and diffuse sources along the Gryteland stream at the Skuterud study site located in southeastern Norway.

2.2 Methods

2.2.1 Phosphorus Losses from Rural Catchments

With the purpose of quantifying the nutrient contribution from agriculture runoff and OWTs to the water contamination, this article evaluates the effect of rural wastewater on water quality in eight selected agricultural catchments, all included in the Norwegian Agricultural Environmental Monitoring Programme (JOVA). JOVA is a national programme for soil and water monitoring in agriculture dominated catchments in Norway, and was initiated in 1992 with the aim to document the effects of agricultural practices and measures on runoff and water quality. The geographical distribution of the eight catchments and an overview of the total area, cultivated area, temperature and precipitation conditions, soil type and dominant crops in the fields are shown in Fig. 2.4. P losses from the fields are measured from May to May each year. The quantification is carried out using discharge measurements and water sampling. The water sampling system is based on a volume proportional, composite sampling routine. The advantage of such a system is that accurate information is obtained about the concentration in the runoff water during the composite sampling period, representing an average concentration for the sampling period. No information about the variation in concentration during episodes with high runoff and large diurnal variations in discharge is obtained using this method. The method is described in great detail elsewhere (Deelstra et al. 2013).

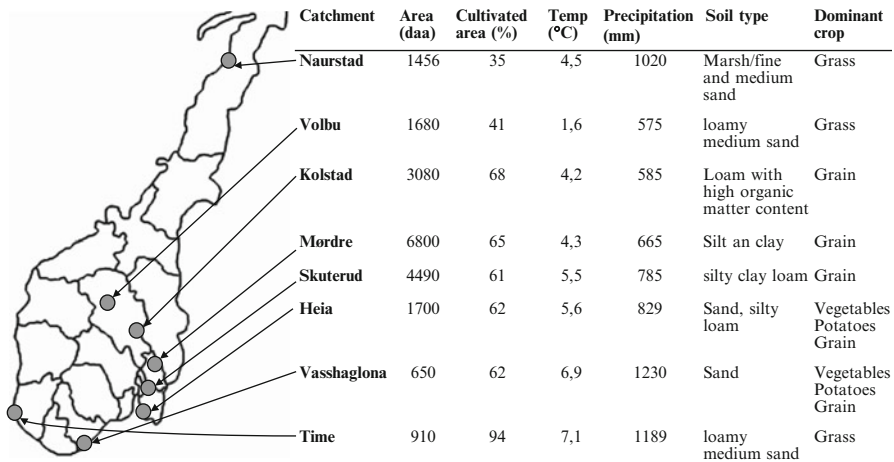


Fig. 2.4 The geographical distribution of the JOVA catchments and an overview of the total area, cultivated area, temperature and precipitation conditions, soil type and dominant crops in the catchments

2.2.2 Phosphorus Losses from On-site Wastewater Treatment Systems

In the eight catchments, there are scattered settlements with OWTs that contribute to nutrient and intestinal bacteria discharges into water bodies. In 2012/2013 information about the OWTs in the catchments were collected. Nutrient emissions from the OWTs were calculated using a web-based model with Geographical Information System (GIS) “WebGIS avløp” (Kraft and Turtumøygaard 1997), in which the emissions from OWTs are assumed equally distributed throughout the year. The WebGIS avløp (English: WebGIS wastewater) is a practical tool applied by Norwegian municipalities for registration and administration of OWTs.

2.2.3 Case Study

In order to carry out further investigations on source tracking, Skuterud catchment was selected purposely to be the case-study area, as it represents perfectly the typical Norwegian catchment with pollution runoff from both point and diffuse sources. This pollution is treated by a CW established in the outlet of the Gryteland stream, which receives runoff from both scattered settlements and agricultural areas including livestock and grain production.

2.2.3.1 Site Description

The catchment is located in southeastern Norway, approximately 30 km southeast of Oslo (Fig. 2.5). Marine deposits in addition to glacial moraine deposits dominate in the soils of the catchment. The soil texture of the arable land in the upstream

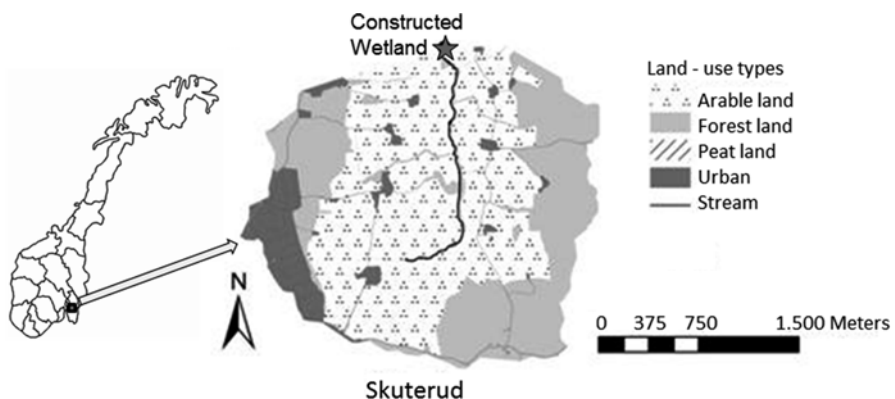


Fig. 2.5 The catchment of the Gryteland stream at the Skuterud study site

catchment is mainly silty clay loam, silt loam, and some loamy sand. A detailed soil classification of the catchment is described in great detail elsewhere (Hauken and Kværnø 2013). The climate is characterized by unstable winters and an average yearly precipitation of 775 mm. The catchment of the Gryteland stream has an area of about 450 ha, of which 270 ha is agricultural land, 140 ha forest/marshland and 40 ha occupied by settlements (Fig. 2.5).

2.2.3.2 Wetland Design

The CW in the Gryteland stream was constructed in 2000 with the objective of reducing the transport of sediments, nutrients, and pesticides to the lake Østensjøvannet, which is located at the outlet of the Skuterud catchment (Fig. 2.6). At the inlet of the CW, the stream water flows into a sedimentation pond, passes through a sprinkling zone, and then runs through two vegetated wetland filters (Fig. 2.6). Long and narrow shapes of the CW help to increase the hydraulic efficiency and makes maintenance easier. Retention-time in the CW varies with the water flow, but is on average approximately 4.5–5 h. The water surface of the entire CW is approximately 2300 m² covering about 0.09% of the cultivated area and 0.05% of the total catchment area.

The sedimentation pond is 1.5–2 m deep, 8–10 m wide and 50 m long. The first vegetated wetland filter is about 100 m long, 0.5 m deep, and 8 m wide and the second vegetated wetland filter is about 75 m long, 0.5 m deep, and 8 m wide (Fig. 2.6). The filters are planted with local wetland vegetation such as: *Phalaris arundinacea* L., *Typha latifolia* L., *Iris pseudacorus* L., *Glyceria fluitans* L., and *Sparganium erectum* L.

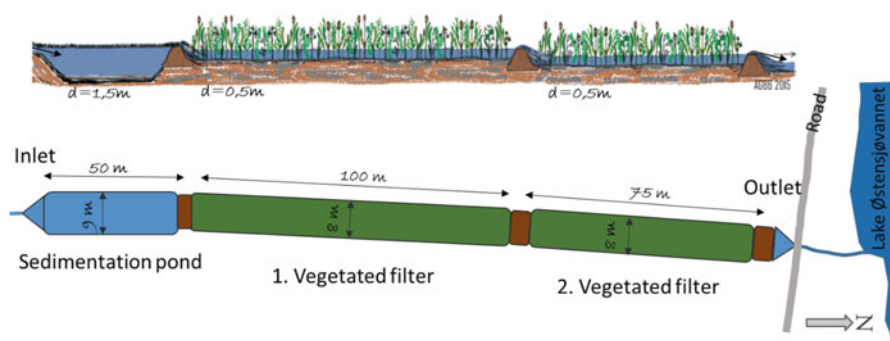


Fig. 2.6 Cross section- and bird's eye sketch of the constructed wetland in the Gryteland stream in the Skuterud catchment

2.2.3.3 Source Tracking and Distribution of Pollutants Through the Seasons

It is assumed that emission from OWTs is approximately equal throughout the year, while runoff from agricultural area varies due to factors like temperature, precipitation, tillage and growing season. The P-input from agriculture runoff and OWTs on monthly basis into the Gryteland stream has been calculated based on the same model described in Sect. 2.2.1. Additionally eight water (grab) samples were collected at the inlet and outlet of the CW in the Gryteland stream throughout the investigation period November 2014–September 2015. The water samples were analyzed for various physico-chemical parameters by ALS Laboratory Group Norway AS applying ISO and national standards. At the same time, water samples were collected and analyzed for faecal contamination, reported in terms of faecal indicator bacteria, i.e. *Escherichia coli* (*E. coli*) and coliform bacteria concentrations in 100 ml of tested water and expressed as the most probable number (MPN)/100 ml with a detection limit of <1 MPN/100 ml. The samples were analyzed by using the Colilert 18/Quanti-Tray®2000 Method (IDEXX Laboratories Incorporated, Westbrook, Maine, USA). The procedure of this method has been described in greater detail elsewhere (Paruch et al. 2015a). Contamination from domestic wastewater can be tracked by faecal pollution providing it is clearly distinguished between human and non-human origin. To confirm/decline contribution of human, thus contamination by domestic wastewater, molecular analyses of the eight water samples at the inlet and outlet of the CW in the Gryteland stream were performed. These analyses employed real-time quantitative polymerase chain reaction (qPCR) on host-specific 16S rRNA gene markers *Bacteroidales* for microbial source tracking (MST) to determine the origin(s) of water contamination. The scientific background and procedures of the MST technique have been described in greater detail elsewhere (Paruch et al. 2015b).

2.3 Results and Discussion

2.3.1 Phosphorus Losses from the Catchments

The TP runoff from rural catchments is largely affected by precipitation and runoff situation, and therefore varies both throughout the year (every single year), and from one year to another. The large variations in the studied catchments with respect to the size, the share of cultivated lands, precipitation and dominant agricultural practices contribute to wide variation in TP runoff from these catchments. During the years 2010–2013, the average TP-losses from all catchments varied from 46 to 2246 kg/year, and the annual variation in TP-losses from each catchment varies largely (Table 2.1). For example, TP-runoff from the Skuterud fields was in

Table 2.1 Average TP-losses 2010–2013 and annually TP-losses in 2010/2011, 2011/2012, 2012/2013 in the eight catchments

Catchment	Average TP-loss 2010–2013: kg/year ^a	Annually TP-loss (kg/year) ^a		
		2010/2011	2011/2012	2012/2013
Skuterud	827	718	566	1199
Mørdre	2246	1564	1578	3597
Kolstad	207	237	145	240
Heia	299	250	216	452
Vasshaglona	195	100	269	216
Time	114	73	94	107
Volbu	46	55	59	25
Naurstad	172	157	181	179

^aCalculations are based on the total catchment area (Fig. 2.4)

2012/2013 about twice as large as in 2011/2012. This can partly be explained by some episodes of high runoff, which has given high nutrient concentrations in streams.

2.3.1.1 Phosphorus Losses from On-site Wastewater Treatment Systems

Seven different types of OWTSSs were registered in the eight catchments, and the total number of OWTSSs in these catchments varied from 4 to 48 (Fig. 2.7). There are many OWTSSs of poor treatment standard (24–100 %) and these are mainly septic tanks without any following treatment steps, such as e.g. infiltration systems, CWs or chemical and/or biological treatment plants. In the Skuterud catchment there are 17 OWTSSs, whereof 4 is not approved after Norwegian standards. The catchments Volbu and Naurstad have the highest percentage of OWTSSs with poor treatment standards.

P-losses from OWTSSs was based on input data registered in 2012–2013, while TP runoff from catchments was based on the average runoff from the years 2010/2011, 2011/2012 and 2012/2013. P-losses from agriculture areas were calculated by subtracting the modelled contribution of OWTSSs from the TP-runoff measured in JOVA. Discharges from OWTSSs were 3–51 kg P / year, equivalent to 1–5 % in seven of eight of catchments, while as much as 19 % in Volbu catchment (Table 2.2). The large contribution of TP from OWTSSs in Volbu was probably due to many treatment systems with poor standard, combined with small agriculture area in the entire catchment (Fig. 2.7). In addition, contribution from the OWTSSs were probably overestimated, since the model calculate “worst case scenario” of runoff to the streams. For example, some factors (including soil type, structure, thickness, density) that influence purifying processes in the soil before the runoff enters the streams were not included in the model.

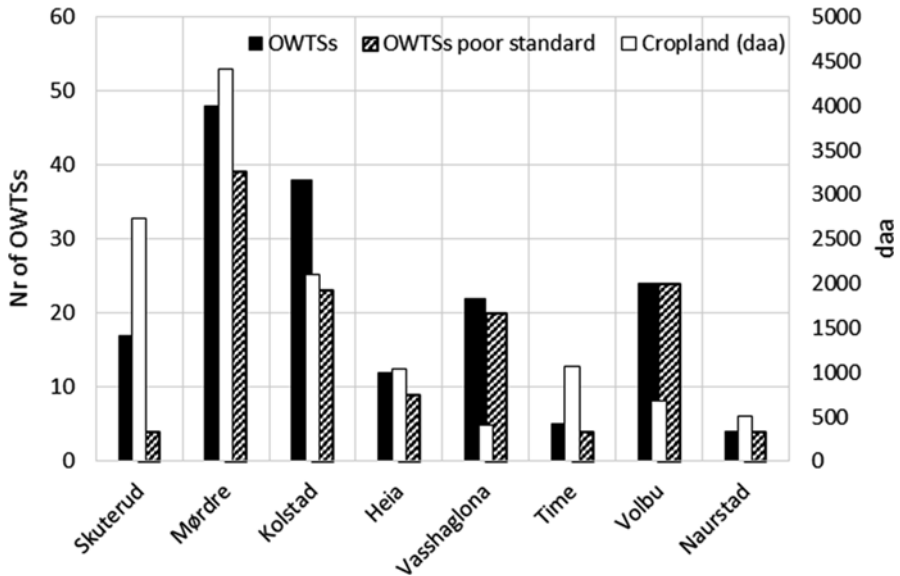


Fig. 2.7 Total number of OWTSSs, numbers of OWTSSs with poor treatment standard and agricultural area in the catchments (daa=decare=0.1 ha)

Table 2.2 Estimated TP (kg/year) from (OWTSSs) in the eight catchments and the percentage of TP from OWTSSs of the TP inputs in catchment areas

Catchment	TP from OWTSS (kg/year) ^a	TP from OWTSS (%) ^b
Skuterud	7	1
Mørdre	51	2
Kolstad	10	5
Heia	12	4
Vasshaglona	8	4
Time	5	5
Volbu	11	19
Naurstad	3	1

^aTheoretical calculations (“WebGIS avløp”)

^bBased on average runoff in 2010/2011, 2011/2012 and 2012/2013

2.3.2 Case Study

In the Skuterud catchment, only 1% of the annual TP contribution originated from OWTSSs, but as shown in Fig. 2.8 the monthly contribution was relatively high (even >30%) during warm/dry season; in spring and summer (May, June and July), and cold months with frost in wintertime (January, February, March). Former studies have also shown that average monthly TP contribution from OWTSSs could be as

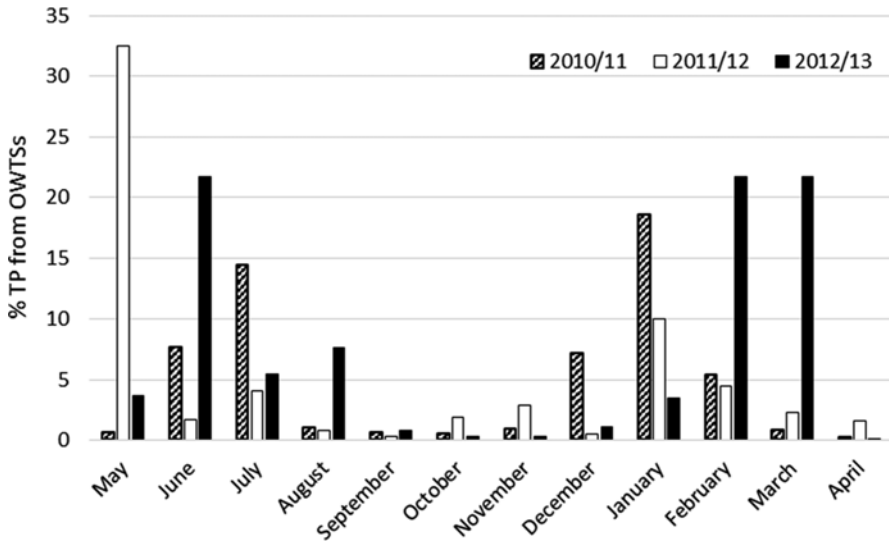


Fig. 2.8 Average monthly percentages of TP runoff from OWTs into the Gryteland stream, in the years 2010/2011, 2011/2012 and 2012/2013

much as 100% in the dry and/or cold periods (Blankenberg et al. 2015). This is of great importance, since P inputs into water bodies during spring and early summer contribute to higher P concentrations in algae growing season. In addition, P from OWTs are more accessible to algae than P from surface runoff from agricultural areas, because P from surface runoff is largely bound to soil particles and thus less available for algae.

2.3.2.1 Effect of the Constructed Wetland in the Gryteland Stream

The retention of P and sediments has been monitored in the CW in the Gryteland stream since 2003. Sediments were removed from the CW twice, in 2005 and 2011. There are considerable variations in retention of sediments and TP in the CW between the years (Fig. 2.9). The annual retention of TP in the years 2003–2015 varied from 5 to 43%, except in 2011/2012, where there was an annual leakage from the CW. This was probably caused by the fact that sediments were removed from the CW in February 2011, and this led to many particles in the stream. The average retention of TP for the entire period was 16%. There were also substantial variations in retention of sediments and TP in the CW between the years, and furthermore, there was a relationship between yearly runoff and the retention. There was a trend of increasing the retention of both sediments and TP with decreasing runoff. This was probably due to longer retention time in the CW. Other study (Braskerud 2001) revealed that as runoff increases, erosion processes in the

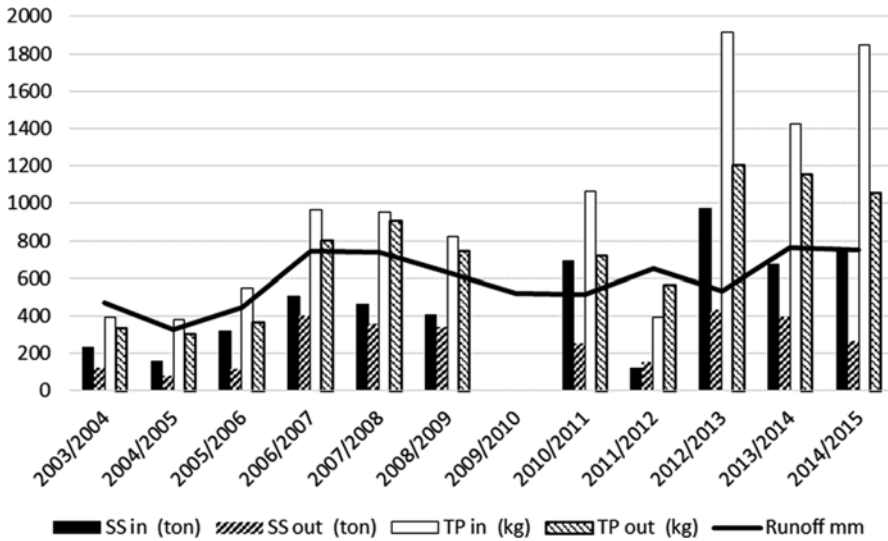


Fig. 2.9 Annual particle- and TP-transport in the inlet and the outlet of the constructed wetland, and runoff in the Gryteland stream in the years 2003–2015. Results from SS and TP transport in the inlet and the outlet of the constructed wetland in 2009/2010 are not included in the figure, since there were problems with the sampling system, due to frost from December 2009 to April 2010

watershed deliver more and larger particles and aggregates to the CWs, and therefore the retention also increases. The same effect was observed for particulate P (Braskerud 2001).

In the investigation period (November 2014–September 2015), the average monthly retention of sediments and TP in the CW varied from 13 to 77 % and –23 to 62 % respectively (Figs. 2.10 and 2.11). The average monthly retention of TP and sediments was 32 % and 54 %, respectively. For both sediments and TP the average monthly concentrations were higher in the inlet than in the outlet, except for TP leakage in September 2015 (Fig. 2.11). This was probably due to heavy rainfall in the autumn, followed by an increased runoff.

The concentrations of sediments and TP in the grab samples were lower than in the volume proportional, composite samples. Content of sediments in the inlet and outlet of the CW was about the same in five of the eight grab samples. In December 2014 and January 2015, the content was higher in the outlet than in the inlet, while in February 2015 the content was highest in the inlet of the CW. TP concentrations in the grab samples in the inlet and outlet of the CW were also about the same in five out of the eight samples. Despite low runoff in wintertime, TP concentrations were quite high in grab samples collected in December 2014 and February 2015. This could clearly indicate on nutrients origin from other sources than runoff from agricultural areas.

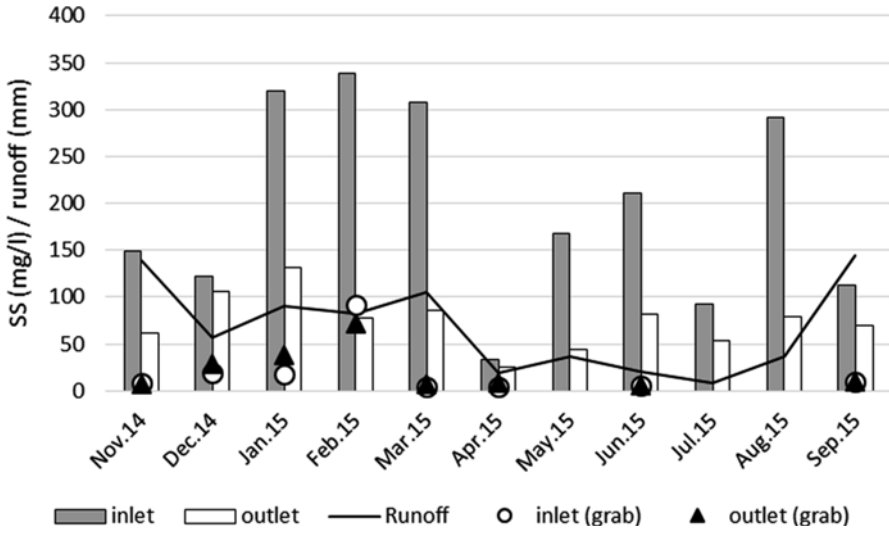


Fig. 2.10 Concentrations of sediments (mg/l) in the inlet and outlet of the constructed wetland in the Gryteland stream, and runoff (mm) throughout the investigation period (November 2014–September 2015)

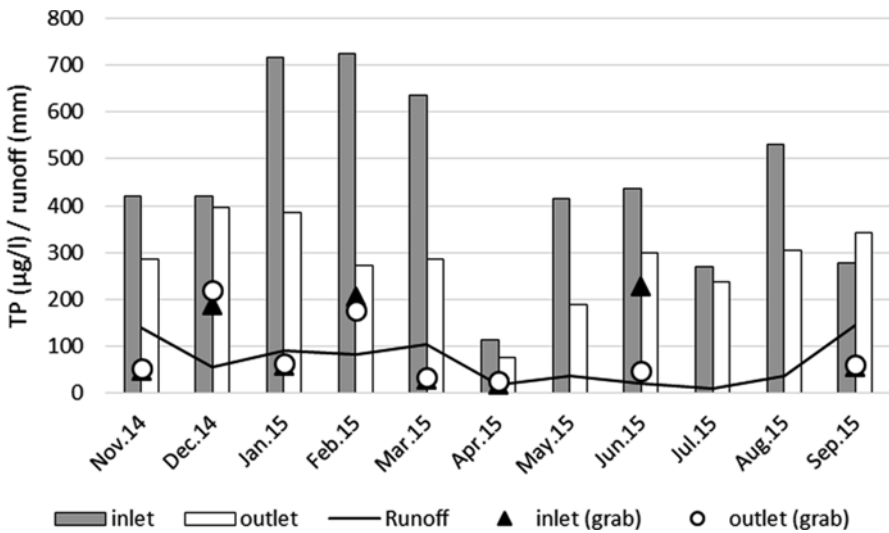


Fig. 2.11 TP concentrations (µg/l) in the inlet and outlet of the constructed wetland in the Gryteland stream and runoff (mm) throughout the investigation period (November 2014–September 2015)

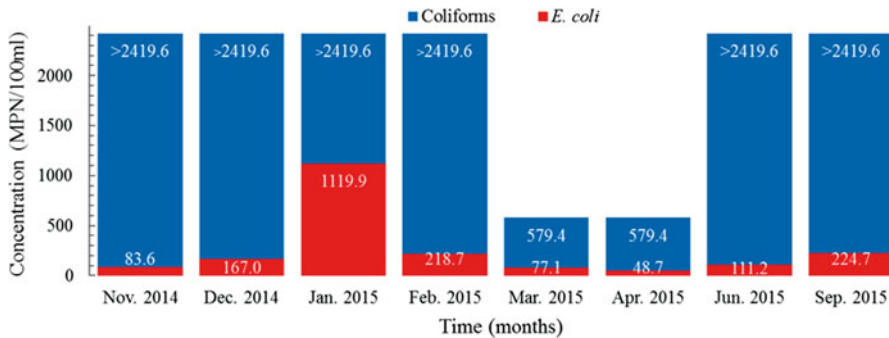


Fig. 2.12 Faecal contamination reported in terms of *E. coli* and coliform bacteria concentrations expressed as the most probable number (MPN)/100 ml of water samples collected throughout the investigation period (November 2014–September 2015) in the inlet of the constructed wetland in the Gryteland stream

2.3.2.2 Faecal Contamination in the Catchment

It has been indicated that P in the Gryteland stream might originate from other sources than agricultural runoff. To determine the real sources, and particularly distinguished between human and non-human origins of water pollution, the MST technique has been implemented. Water contamination from humans occurs through discharge of wastewater, which can be tracked by faecal pollution measures. Microbial analyses of water samples collected during the study period November 2014–September 2015 in the inlet and outlet of the CW in the Gryteland stream detected both coliforms and *E. coli* in all the samples. The highest concentration of *E. coli* among all the inlet samples was in January 2015 (1119.9 MPN/100 ml) while the lowest concentration (48.7 MPN/100 ml) was in April 2015 (Fig. 2.12). This corresponded respectively with both high and low average monthly TP concentrations in the inlet of the CW in the Gryteland stream (Fig. 2.11). Molecular tests revealed human dominance in water faecal pollution, 99% and 89% in December 2014 and April 2015, respectively (Fig. 2.13). This was well associated with the earlier estimated TP inputs from wastewater (discharge from OWTSSs) to the stream during the cold season and relatively dry period (e.g. little agriculture runoff in April 2015, Fig. 2.11).

Although the CW in the Gryteland stream was established mainly for reducing the transport of sediments, nutrients and pesticides, it also reduces concentrations of faecal indicator bacteria. As shown in Fig. 2.14, concentrations of *E. coli* in water samples collected during the study period November 2014–September 2015 in the outlet of the CW were generally lower (except substantially higher concentration in April 2015) than the respective concentrations in the inlet water samples (Fig. 2.12). High *E. coli* concentration measured in April 2015 might have been of a multiple reason, as for instance: (i) direct wastewater pollution, as revealed by human dominance in the inlet and outlet water samples (89% and 94% respectively, Figs. 2.13 and 2.15) and at the same time a little agriculture runoff (Fig. 2.11), which was also

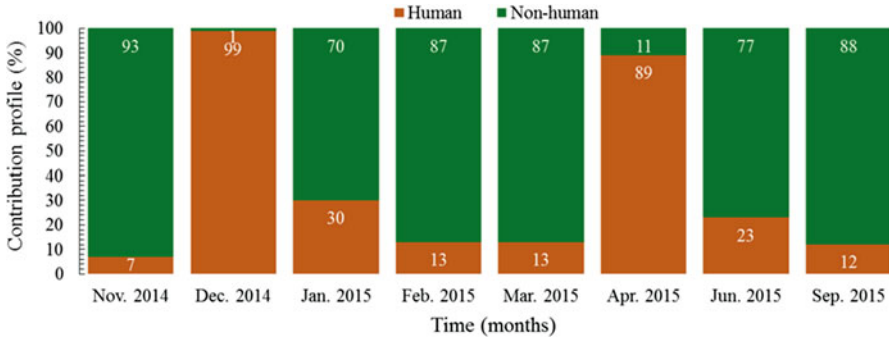


Fig. 2.13 Relative abundance among microbial source tracking (MST) tested markers in terms of their contribution in water samples collected throughout the investigation period (November 2014–September 2015) in the inlet of the constructed wetland in the Gryteland stream

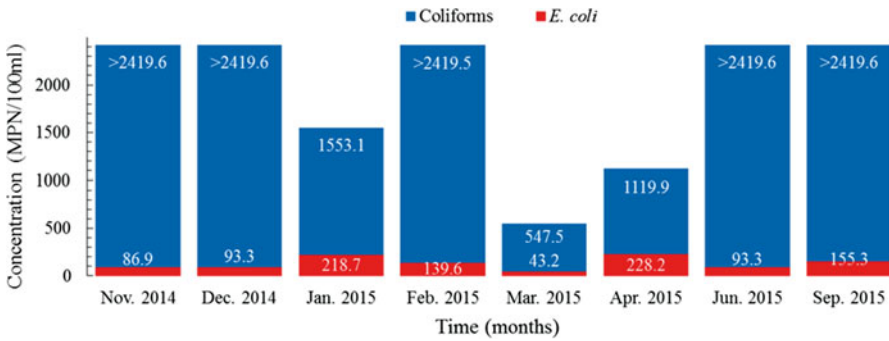


Fig. 2.14 Faecal contamination reported in terms of *E. coli* and coliform bacteria concentrations expressed as the most probable number (MPN)/100 ml of water samples collected throughout the investigation period (November 2014 - September 2015) in the outlet of the constructed wetland in the Gryteland stream

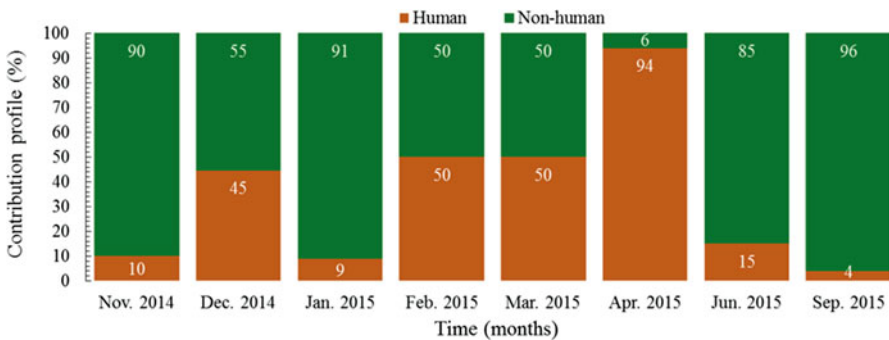


Fig. 2.15 Relative abundance among microbial source tracking (MST) tested markers in terms of their contribution in water samples collected throughout the investigation period (November 2014 - September 2015) in the inlet of the constructed wetland in the Gryteland stream

documented elsewhere (Blankenberg et al. 2015); (ii) resuspension of faecally contaminated wetland sediments during annual changes of wetland vegetation (e.g. roots and shoots development in growing season), which causes higher concentrations of *E. coli* in water (Brinkmeyer et al. 2015; Sanders et al. 2005); (iii) ambient turbulences that hinder settling of solids and flocculation in wetlands and further distribute solids over the entire water depth (Tchobanoglous 1993); (iv) long survival and possible multiplication of faecal indicator bacteria in wetland sediments (Sanders et al. 2005) as particles can harbor bacteria and some wetland filters possess suitable conditions for long-term growth and persistence of *E. coli* (Paruch 2011).

2.4 Conclusions

This study presents data on the actual status of CWs treating catchment runoff in Norway, and in particular most recent results of source tracking and retention of sediments and TP in a model, full-scale, long-term operated CW treating runoff from a typical rural catchment with pollution from both point and diffuse sources along the Gryteland stream in the Skuterud catchment located in southeastern Norway. The study reveals that runoff from agricultural areas was the main P-source annually in the eight studied rural catchments, however the sediment and nutrient runoff was largely affected by the precipitation and runoff situation, and therefore varied both throughout the year (every single year), and from one year to another.

Annual TP contribution that originates from OWTs was in general limited, and in the case study of the Skuterud catchment it was only 1%. Monthly contribution, however, was higher than 30% during warm/dry season, and cold months with frost during the winter. This is of great importance, since P inputs into water bodies during spring and early summer contribute to higher P concentrations in algae growing season. In addition, P from OWTs are more accessible to algae than P from surface runoff from agriculture areas, due to that P from surface runoff is largely bound to soil particles. Water contamination from humans occurs through discharge of wastewater, which was tracked by faecal pollution measures and confirmed in MST tests. High *E. coli* concentration was well associated with high TP inputs from wastewater to waterbodies during dry or/and cold season with little or no agriculture runoff, and further MST tests proved human contribution.

Retention of sediments and P has been monitored in the CW in the Gryteland stream since 2003 and there are considerable variations in retention of sediments and TP in the CW between the years. The annual retention of TP in the years 2003–2015 varied from 5 to 43%, except in 2011/2012, where there was an annual leakage from the CW. This was probably caused by the fact that sediments were removed from the CW in February 2011, and this led to many particles in the stream. The average retention of TP for the entire period was 16%. There was a trend of increasing the retention of both sediments and TP with decreasing runoff.

During the study period (November 2014–September 2015), the average monthly retention of sediments and TP was 54 % and 32 %, respectively. In September 2015 it was a leakage of TP, probably due to heavy rainfall in the autumn, followed by an increased runoff.

Although the CW in the Gryteland stream was established mainly for reducing the transport of sediments, nutrients and pesticides from agriculture runoff, it also reduced concentrations of faecal indicator bacteria. Concentrations of *E. coli* in water samples collected during the study period in the outlet of the CW were in most cases lower than the respective concentrations in the inlet water samples.

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Chapter 3

Performance of Constructed Wetlands Treating Domestic Wastewater in Norway Over a Quarter of a Century – Options for Nutrient Removal and Recycling

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Anne-Grete Buseth Blankenberg, and Guro Hensel

Abstract Norwegian constructed wetlands (CWs) that treat domestic wastewater are classified as horizontal subsurface flow constructed wetlands (HSFCWs). Over the years of continuous performance, the HSFCWs operating under cold climate conditions have shown a high and stable treatment efficiency with regard to the removal of organic matter (>90 % BOD), nutrients (>50 % N and >90 % P) and microbes (>99 % bacteria). The majority of Norwegian HSFCWs are categorised as small (<50 pe) on-site, decentralised wastewater treatment systems. The Norwegian systems consist of three fundamental elements: a septic tank, a pre-filter (i.e. an aerobic vertical flow biofilter) and a horizontal flow saturated filter/wetland bed. The first, primary treatment step begins in the septic tank from which effluents are pre-treated in the second step occurring in the pre-filter/biofilter section and further in the third, final step taking place in the filter bed/HSFCW. The first and third treatment steps are quite common in systems with CWs, but the pre-treatment in biofilter(s) is mainly known from Norway. The main purpose of using the pre-treatment phase is to supply air during the cold season, to enhance nitrification processes, and to reduce the load of organic matter before entering the filter/wetland bed. If constructed and maintained correctly, the biofilters alone can remove 90 % BOD and 40 % N. Various filter/CW beds have been introduced for treatment of domestic wastewater (as complete or source-separated streams) in Norway, but the most common feature is the use of specific filter media for high phosphorus (P) removal. A few Norwegian municipalities also have limits with respect to nitrogen (N) discharge, but the majority of municipalities use 1.0 mg P/l as the discharge limit for small wastewater treatment systems. This particular limit affects the P retention lifetime of the filter media, which varies from system to system depending on the filter media applied, the type of wastewater treated, and the system design and loading rates. An estimated lifetime of filter media with regard to P removal is

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approximately 15–18 years for a filter/CW bed of a single household. After completing the lifetime, the filter media is excavated and replaced with new/fresh materials, allowing the system to operate effectively for another lifespan. Since the exploited media are P-rich materials, the main intention is their reuse in a safe and hygienic way, in which P could be further utilised. Therefore, the Norwegian systems can represent a complex technology combining a sustainable technique of domestic wastewater treatment and a bio-economical option for filter media reuse. This is a quite challenging goal for reclamation and recycling of P from wastewater. Thus, there are some scenarios of reusing the P-rich filter media as a complementary P fertiliser, a soil amendment or a conditioner, provided the quality is acceptable for utilisation in agriculture. Alternatively, the filter media could be reused in some engineering projects, e.g. green roof technology, road screening or construction of embankments, if the quality allows application in the environment. The core aspect of the reuse options is the appropriate quality of the filter media. As for the theoretical assumption, it should not be risky to reuse the P-rich media in agriculture. In practice, however, the media must be proven safe for human and environmental health prior to introducing into the environment.

Keywords Horizontal subsurface flow constructed wetlands (HSFCWs) • Nutrients • Pre-filter/biofilter • Filter media reuse

3.1 The State of the Art in a Nutshell

It has been a quarter of a century since the first wastewater treatment system employing constructed wetland (CW) was implemented at Haugstein farm, approximately 20 km southeast of Oslo, southeast Norway. The main purpose behind this implementation was to assess the ability of CW in treating domestic wastewater under cold climate conditions. The pioneering system was constructed with two wetland beds: one filled in with sand and the other with light expanded clay aggregates (LECA). As fast as the system revealed that it was well adopted with regard to climatic variations, plant assimilation and treatment efficiency, the popularity of such a nature-based treatment technology gained more interest. It turned out that filter/CW beds were widely implemented for treatment of different types of wastewater with respect to the origin, diversity of pollutants and specific treatment goals.

At present, there are generally two types of CWs adapted to wastewater treatment under Norwegian conditions, i.e. systems with surface and subsurface flow. Systems of the first type with free flow of water at the surface of the CW bed are commonly used for treatment of agricultural and/or urban (including road and tunnel) runoff, storm waters and landfill leachates. For treatment of domestic wastewater, in particular at decentralised locations, CWs with subsurface flow of water (underneath the surface of the filter bed) are the most popular in Norway. Furthermore, systems with horizontal flow have been commonly used; hence, the

Norwegian CWs are classified as horizontal subsurface flow constructed wetlands (HSFCWs).

Although various designs of CWs have been introduced for treatment of domestic wastewater in Norway, the most common feature is the use of specific filter media for high P removal. This is because phosphorus (P) is considered the main element triggering eutrophication in Norwegian waterways. Thus, its efficient removal is required for small, decentralised treatment systems. A few Norwegian municipalities also have limits with respect to nitrogen (N) discharge, in order to protect local groundwater or sensitive fjord areas; however, the majority of municipalities normally use 1.0 mg P/l as the discharge limit for small wastewater treatment systems. This particular limit affects the P retention lifetime of the filter media, which varies from system to system depending on filter media applied, the type of wastewater treated, and the system design and loading rate. After completing the lifespan, the filter media have to be excavated and replaced with new materials, allowing the system to operate effectively for another lifespan. Since the exploited media are P-rich materials, the main intention is not to waste them, but rather to reuse them in a safe and hygienic way, in which P could be further utilised. Alternatively, the media could also be reused as either soil conditioners or road constructing amendments.

The majority of Norwegian HSFCWs are categorised as small (<50 pe) decentralised, on-site wastewater treatment systems, but there is also a range of systems operating for a higher number of individuals. It is therefore quite difficult to provide an exact figure of total CWs treating domestic wastewater over the entire country. However, according to the latest statistical data (Statistics Norway 2014), there were 203 filter/CW beds in Norway (Fig. 3.1).

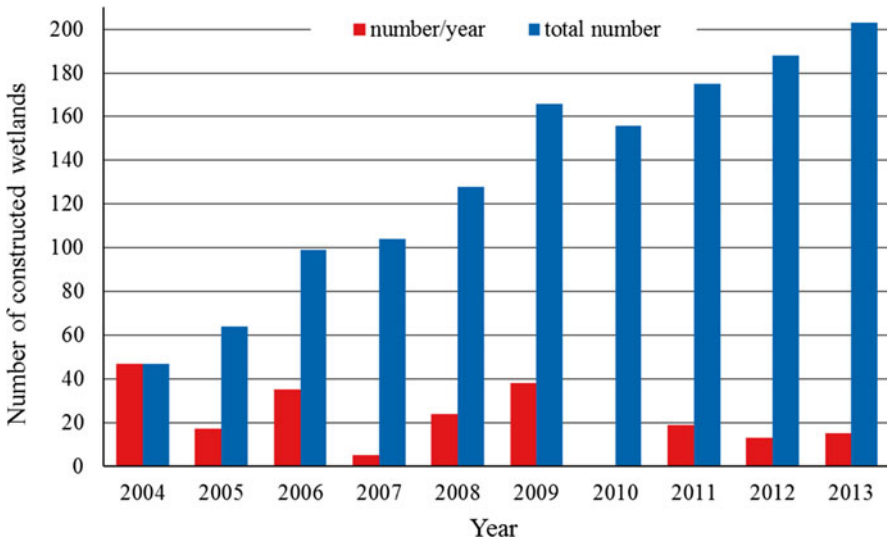


Fig. 3.1 Number of constructed wetlands and filter beds employed in small (<50 pe) systems treating domestic wastewater in Norway (Statistics Norway 2014)

This number, however, considers only the small systems, thus CWs serving for more than 50 pe are not included in these statistics. This is strictly related to the fact that one of the categories of Norwegian municipal wastewater treatment plants includes nature-based systems (212 in total number), thus also all the CWs operating for >50 pe (Statistics Norway 2014). In addition, it happens that not all treatment systems (especially the decentralised, on-site ones) are reported by municipalities, hence are not included in published documents (Statistics Norway 2014). It can therefore be estimated that the total number of CWs treating domestic wastewater in Norway is actually much higher than the reported figures.

Scientists from NIBIO have been actively involved in developing, designing, constructing and studying the range of small on-site, decentralised systems treating domestic wastewater in Norway, including long-term investigation of the first, pioneering CW in Haugstein. The results presented throughout this chapter in tables and figures that have no external references have been derived from the research database established by engineers and scientists from NIBIO.

3.2 General Characteristics and Design Principles

The main goal for all HSFCWs treating domestic wastewater in Norway is high nutrient and organic matter removal, with particular attention paid to phosphorus (P) and BOD reduction (>90 % in sensitive areas). A few municipalities have established limits with respect to nitrogen (N) removal, but the majority of municipalities use 1.0 mg P/l as the discharge limit. To achieve these limits, Norwegian systems have been designed with three fundamental treatment steps. The first (primary treatment) represents an anaerobic method, the oldest and most common in on-site treatment systems (Ntengwe 2005; Seghezze et al. 1998), and occurs in a septic tank. The second (pre-treatment) is characterised by an aerobic phase carried out in a biofilter section. The third (final) treatment step takes place in a filter bed of HSFCW. Consequently, the typical CW system treating domestic wastewater in Norway consists of three principal elements: (1) the septic tank followed by (2) the pre-filter/biofilter with vertical unsaturated flow and (3) the subsequent saturated horizontal flow filter/wetland bed. The first and third treatment steps are quite common in systems with CWs, but the pre-treatment in biofilter(s) is mainly known in Norway, where it has been introduced for supporting the entire operation of the treatment system under cold climate conditions (Jenssen et al. 1991).

3.2.1 *Septic Tank*

A three-chamber septic tank with minimum volume of 4 m³ was required for single households (5 pe producing 1 m³ wastewater daily) in Norway (Miljø blad 2001b). This is no longer necessary as the Norwegian norm has been revised, and from 2013 prefabricated one-chamber septic tanks can be used. Yet, the highest number of

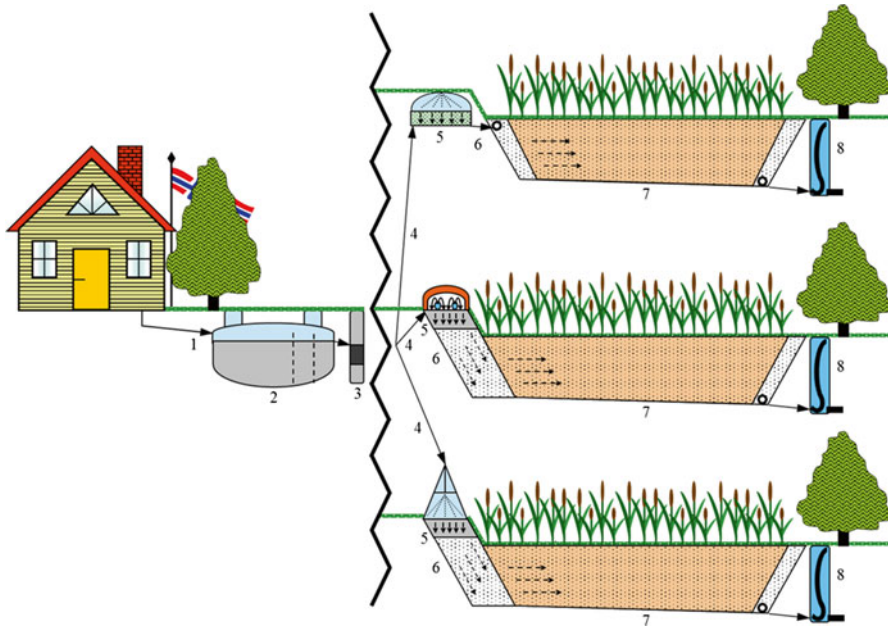


Fig. 3.2 General layout of the Norwegian HSFCW in three configurations where: 1 – inlet (domestic wastewater), 2 – three-chamber septic tank (the first treatment step), 3 – pump well, 4 – effluent from the septic tank, 5 – typical configuration of an aerobic pre-filter/biofilter (the second treatment step) constructed in dome (*top configuration*), infiltration bed (*middle configuration*) and shelter (*bottom configuration*), 6 – effluent from the biofilter, 7 – wetland bed with vegetation (the third treatment step), 8 – outlet (effluent from the entire system)

filter/CW beds was constructed with three-chamber septic tanks; hence this type of tank has been described here. The first chamber normally has a larger volume than the others, as is designed for the collection of sludge. It is estimated that 0.25 m³ of sludge per pe during 1 year can be collected from households equipped with WC. Normally, there is an 18-h retention time for domestic wastewater in the septic tank before entering the next treatment steps. Effluent from the septic tank is first pre-treated by the pre-filter/biofilter units before entering the final treatment step in the filter/wetland bed. The effluent is normally pumped up from the tank and then pumped into the pre-filter/biofilter section. A high-pressure borehole peristaltic pump is used for dosing the septic tank effluent into the pre-treatment section. The pump is commonly installed in a well between the septic tank and the pre-filter section (Fig. 3.2).

3.2.2 Pre-filter/Biofilter

The main purpose of implementing the pre-filter/biofilter section was to supply air during the cold season and to enhance the nitrification processes improving N removal (Mæhlum et al. 1995). An additional reason was to pre-treat effluents from

septic tanks through reduction of organic matter, thus avoiding possible clogging occurring in CWs (Mæhlum and Jenssen 1998). Furthermore, pre-filters decrease the load of these effluents into wetland/filter beds and thus a stable and high effect of treatment can be achieved (Miljø blad 2001a).

For the best performance of this treatment step, the septic tank effluent has to be distributed evenly onto the surface of the biofilter media. The effluent distribution can be performed either by infiltration pipes (placed in splashing capsules or covered by coarse materials, e.g. gravel and/or crashed stones) or by spray nozzles. The latter have been found to be more effective in providing an even distribution of the effluent over the whole biofilter media. Furthermore, use of the pre-filters with spray nozzles has revealed a higher treatment efficiency (Jenssen et al. 2005); thus, these so-called trickling biofilters became standard pre-filters used in Norwegian design of CW systems treating domestic wastewater.

The pre-filter constitutes a down (vertical) flow aerobic filter (biofilter) filled in with special media. It can be built in a dome, a tank or a sheltered bed (Fig. 3.2) depending on local conditions. The minimum depth of the pre-filter is 0.5 m. The depth is equal to the minimum height of the biofilter bed, if it is integrated with the front edge of CW (Fig. 3.2). For biofilters built in a separate units (domes or tanks), the minimum height is 0.6 m, due to the 0.1 m drainage zone for the effluent collected (Miljø blad 2001a).

To obtain a high treatment efficiency, the biofilter has to be built with an adequate surface area for receiving wastewater. The surface area is a dimensioning parameter and depends mainly on two factors: daily water consumption (l/pe/d) and hydraulic load (cm/d). According to the Norwegian guidelines (Miljø blad 2001a), the first factor is defined as 200 l/pe/d, while the second is 10 cm/d (100 l/m²/d) or 20 cm/d (200 l/m²/d) for infiltration or trickling biofilters, respectively. Based on these factors, the surface area in the standard pre-filters (trickling biofilters) can be estimated as 1 m²/pe. However, the minimum surface area of one trickling biofilter should be at least 3 m² (Føllesdal 2005).

In general, wastewater discharged from one household (5 pe) should be treated by two pre-filters (minimum total surface area of 6 m²). The number of pre-filters increases by one with the number of households; thus two households (10 pe) have three pre-filters, three households (15 pe) have four pre-filters, and so on (Føllesdal 2005). In some situations, e.g. when the local conditions limit available space, the surface area could be scaled-down. This, however, require recirculation of the effluent through the pre-filter section.

Hydraulic loading rates depend on the type of the pre-filter (i.e. infiltration or trickling biofilter, Fig. 3.2) and the type of treated wastewater. For the best performance of the standard pre-filters (trickling biofilters), the suggested maximum hydraulic loading rate is 20 cm/d (Miljø blad 2001a). Some studies, however, have revealed that high performance could also be achieved with higher loading rates of up to 30 cm/d (Heistad et al. 2006; Jenssen et al. 2005). The best treatment efficiency can be obtained if the loading is dispensed with fixed small doses, optimally 18–48 doses/d (Miljø blad 2001a). The spray nozzles (1–2 per biofilter), which are

suspended over filter media, enable an even distribution of the septic tank effluent over the entire biofilter. The effluent flows further vertically down to the drainage zone at the bottom of the biofilter and, finally, the drained effluent runs off gravitationally to the inlet side of the third/final treatment step occurring in the filter/wetland bed (Fig. 3.2). In the case of separately built pre-treatment units, the drained effluent has to be transmitted gravitationally through a sealed pipe into the perforated distribution pipe (Fig. 3.2). This pipe can be placed in the upper layer of the inlet side in a mass of crushed stones or coarse gravel (10 mm in diameter, clean and free of ash, dust and fine particles) constituting the first part (approx. 0.6 m) of the filter/wetland bed (Miljø blad 2001a).

3.2.3 Constructed Filter/Wetland Bed

Depending on the design of the pre-filter/biofilter section, effluents can run directly into the underlying submerged basin of horizontal flow filter/wetland bed, or indirectly through the distribution pipe in the inlet side integrated with the saturated filter/wetland bed (Fig. 3.2). This is the largest element of the entire wastewater treatment system and has a retention time of 10 days (Miljø blad 2001a). The saturated bed is mainly constructed for P removal to achieve the discharge limit of 1.0 mg P/l (Heistad et al. 2006; Jenssen et al. 2005). For this purpose, the use of filter media with a high P-binding capacity is recommended (Miljø blad 2001a).

The filter/CW bed is an excavated basin of 0.9–1.0 m depth, screened/sealed/insulated at the bottom and edges with a watertight material (e.g. geomembrane made of polyethylene, PVC or bentonite) and filled in with the appropriate filter media. The entire screening of the basin can be avoided if the system is constructed on sites with clay soil (hydraulic conductivity $< 10^{-8}$ m/s) and where the groundwater table is permanently lower than 1.5 m, assuring adequate protection of groundwater from wastewater contamination (Miljø blad 2001a). In this case, however, the use of geotextiles is recommended to separate the filter media from the native soil. The bottom of the filter/CW bed needs to be built with a slight slope of 0.5–1 % between the inlet and outlet side of the bed, i.e. at 1:200–1:100 hydraulic gradient respectively (Miljø blad 2001a). The outlet side of the bed can be constructed in a similar manner as the inlet side. Thus, the perforated pipe collecting the effluent and transmitting it gravitationally into a well/manhole chamber can be installed (Fig. 3.2). The collecting pipe is placed at the bottom of the outlet side in a mass of crushed stones or coarse gravel (10–30 mm in diameter, clean and free of ash, dust and fine particles) constituting the last part (approx. 0.6 m) of the filter/wetland bed (Miljø blad 2001a). The entire bed is normally planted with macrophyte vegetation, in Norway typically with *Phragmites* spp. (common reeds) and *Typha* spp. (cattails). There are also some systems, in particular filter bed treatment systems, established without wetland vegetation at all.

3.2.4 Filter Media

Different media have been applied for construction of pre-filters and filter beds in on-site domestic wastewater treatment systems in Norway. The applied media replace native soils with gravel, sand, shell/coral sand, crushed limestone and light-weight aggregates (LWA). LWA are typical media used in Norwegian CW systems, and among them, the products with the brand name LECA® (light expanded clay aggregates) and Filtralite® are the most frequently used (Søvik and Kløve 2005). In addition, shell sand products under the brand name Filtramar™ have been implemented on the commercial market.

The LWA filter media are normally applied for the second and third treatment steps, pre-filters/biofilters and filter/CW beds, respectively. These porous materials (with a large surface area of over 5000 m²/m³) are ideal media for biofilm growth. Specially developed, Filtralite® NR (N=normal density, R=round material) with a grain size of 2–4 mm is the most frequently used media in Norwegian biofilters. To avoid eventual clogging and formation of puddle over the surface of the biofilter, more coarse materials, e.g. Filtralite® NR with a grain size of 4–10 mm, could also be implemented in the upper 10–20 cm layer (Føllesdal 2005). The main function of filter media applied in filter/CW beds is an efficient P removal; hence, specially developed Filtralite®P (P=high phosphorus sorption capacity) has been widely applied in Norway. In principle, 8–10 m³ of the filter materials per pe is projected; thus, 40 m³ of filter media is normally applied in the system treating wastewater discharged from a single household (5 pe = 1 m³ wastewater daily) equipped with a septic tank, pair of trickling biofilters and filter/CW bed (Miljø blad 2001a).

3.3 Overall Treatment Performance

The range of major pollutants in Norwegian domestic wastewater treated by small on-site, decentralised systems studied by scientists from NIBIO over the years is presented in Table 3.1. Although there is a quite a large variety in concentration rates of the contaminants and in the overall treatment performance, the removal of nutrients is relatively sufficient, especially the average P reduction below the Norwegian limit of 1.0 mg P/l.

An efficient treatment of domestic wastewater in on-site, decentralised systems relies on an effective performance of each and every treatment step in the entire system. Thus, sufficient removal of particles from wastewater hangs on the optimal operation of the septic tank; the necessary reduction in organic matter and N depends on the appropriate function of the pre-filter/biofilter section; while high removal of P and pathogenic microbes rests on vital activities occurring in the filter/CW bed. If all the treatment steps work effectively, then the treatment efficiency of the entire system can be expected to be relatively high.

Table 3.1 The range (min=minimum, max=maximum and st.d. = standard deviation) of contaminants (mg/l) in domestic wastewater entering selected small on-site treatment systems (inlet content) and in the effluents from these systems (outlet content)

Parameter	Inlet content				Outlet content			
	Min	Mean	St.d.	Max	Min	Mean	St.d.	Max
BOD	3	155	123	400	<2	52	43	130
COD	205	648	399	1980	<5	53	51	206
TSS	12	134	143	860	<2	41	39	120
TOC	37	121	63	250	3	24	20	57
P tot	0.22	9.3	5.4	28.6	0.01	0.16	0.28	1.4
N tot	6	80	39	153	7	39	28	103
N-NO ₂ +NO ₃	0.04	0.12	0.11	0.20	0.11	3.2	4.1	12
N-NH ₄	60	123	24	149	0.02	40	37	101
Cl	7	37	24	79	8	29	21	73
pH	6.9–7.9				10.1–>12.0			

Primary treatment occurring in the septic tank (through retention, sedimentation and digestion of organic matter settled as sludge) provides the most effective and undisturbed sedimentation of solids, eggs and (oo)cysts of some pathogenic intestinal parasites. The digestion process conducted by anaerobic bacteria defines the septic tank as a simple/preliminary biological treatment step (Paruch 2010).

The second treatment phase carried out in the pre-filter/biofilter section provides a high removal of organic matter, which can reach >99 % in terms of BOD reduction (Jenssen et al. 2010). In addition, the pre-filters/biofilters can efficiently nitrify and reduce ammonia concentrations (Heistad et al. 2006). It has been revealed that the pre-treatment section can remove up to 40 % of total N (Jenssen et al. 2005). As there is no sludge accumulation at the bottom of the pre-filter/biofilter units, denitrification can happen in the deeper, anoxic layers of a biofilm formed in the biofilter or in anaerobic sites of filter particles. The biofilter can also remove substantial P concentrations, but it becomes quickly saturated (depending on loading rates), then the major removal processes occur in the filter/wetland bed.

The final treatment occurring in the HSFCW can provide a high reduction of P of more than 90 % within the first 10 years of operation. In general, high P removal can be associated with high removal of bacteria and viruses (Schijven and Hassanizadeh 2000). It has been revealed that patches of positively charged aluminium (Al) and iron (Fe) oxides in Filtralite®P media can attract negatively charged viruses (Heistad et al. 2006). Furthermore, a high concentration of magnesium (Mg) and calcium (Ca) ions in these media may facilitate salt bridge effects between negatively charged surfaces (Heistad et al. 2009).

In addition to the main function of the specific filter media, the macrophyte vegetation also plays an essential role in the removal of different contaminants, in particular P and N (eutrophication-limiting nutrients in aquatic environments), through several mechanical, chemical and biological routes. These employ very complex processes accelerated by the natural synergies between the biologically active (with

micro- and macro-organism interactions) ecosystems of soil, water, vegetation and atmosphere. The removal processes include retention, sorption, accumulation, plant uptake, photo degradation and microbial activities enhanced during the passage of wastewater through the rhizosphere. The developed root system expanded throughout the vegetated bed plays a substantial role in transporting contaminants, serving as pathways for gases, and moving particles (Scholz et al. 2002). Vegetated systems act as natural filters that hold particles and inhibit sediments against re-suspension by stabilising them within roots. Rhizomes of reeds create a natural barrier for parasite eggs, so they can be easily destroyed by antagonistic organisms (e.g. earthworms) settled in the filter beds (El-Khateeb et al. 2009; Mandi et al. 1996; Reinoso et al. 2008). Also, the numbers of the bacterial distribution on macrophyte roots sharply decrease within the first few metres along the horizontal flow in vegetated filters (Vymazal et al. 2001).

3.4 Recycling Options for Filter Media

To maintain a high treatment efficiency, the filter media have to be exchanged when the P concentration in the effluent exceeds the discharge limit (i.e. 1 mg P/l). An estimated lifetime of filter media is approx. 15 years for a bed of a single household when the inlet values are about 10 mg P/l (Jenssen and Krogstad 2002). In practice, however, many Norwegian systems have already exceeded the theoretical lifespan and the filter media have reached their P-sorption capacity, but they are still in continuous operation. There are two options for such systems: either to completely shut down the entire system or to re-establish it with fresh/new media replacement. Usually, the second option is considered; however, there are many uncertainties regarding dealing with the excavated/exchanged filter media. On the one hand, the used media are P-rich structural materials that may be utilised as soil amendments and conditioners. Results from laboratory-scale and pot experiments carried out in the Nordic countries demonstrate that P-rich filter media have a positive fertiliser and liming effect, thus could be considered for reuse as a plant fertiliser (Jenssen et al. 2010). On the other hand, reuse of these media has to be done in a safe and hygienic way, thus the media need to be considered as harmless and non-toxic for human and environmental health.

So far, to our knowledge, there have been limited data on the range of contaminants in filter media of CWs in Norway. This is perhaps due to the main focus being given to the effectiveness of these systems in wastewater purification; therefore their year-by-year performance was the main interest, not the fate of the exploited filter media. The initial survey on contaminants accumulated in filter media of the representative CWs in Norway was described by Paruch et al. (2007). The study was further extended and carried out in selected full-scale operating CWs; including the oldest CW in Norway, Haugstein (Table 3.2). The contaminants of concern were heavy metals (Cd – cadmium, Pb – lead, Cu – copper, Zn – zinc, Ni – nickel and Cr – chromium), thermotolerant coliform bacteria (TCB), *Salmonella* bacteria and parasite eggs, as defined in the Norwegian regulation on materials applied in cultivated areas (FOR 2003).

Table 3.2 General characteristics of four selected full-scale operating CWs in Norway

CWs	Construction year	Number of persons/facility connected	Type of filter media
Haugstein	1991	7/household	Sand and LECA®
Tveter	1993	7/household	LECA®
Holt	1999	12/household	Filtralite®P
Dal	2000	39/primary school	Filtralite®P

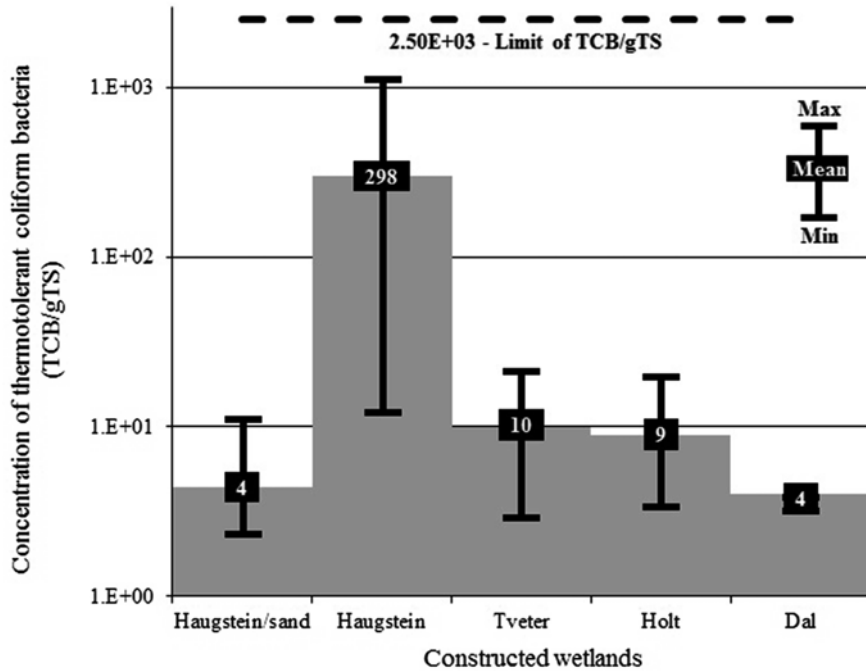


Fig. 3.3 The range of concentrations (min = minimum, mean and max = maximum) of thermotolerant coliform bacteria (TCB/g TS) in filter media of selected constructed wetlands and the limited number for these bacteria in materials applied in cultivated areas in Norway, i.e. 2500 TCB/g TS

The commonly applied filter media are porous materials with a large surface area, which makes them ideal media for biofilm growth. These porous media can harbour microorganisms, yet the mean TCB concentrations in materials tested from four different treatment systems did not exceed the upper limit of 2500 TCB/g TS (Fig. 3.3) set for materials applied in cultivated areas in Norway (FOR 2003). In addition, the survey on the presence of pathogenic microbes in the specific LWA filter media of full-scale operating CWs did not reveal any contamination with virus indicators (FRNA phage MS2 and phi X174), *Salmonella* and parasite eggs (Paruch et al. 2007).

The tested filter media demonstrated a relatively low sorption capacity for most of the heavy metals tested (Table 3.3), showing that their concentrations did not exceed the maximum permissible contents of heavy metals in the materials applied in cultivated areas in Norway (FOR 2003). Theoretically, domestic wastewater from

Table 3.3 Average concentrations of heavy metals in filter media (mg/kg TS) at the inlet and outlet sides of selected constructed wetlands (CWs) and the acceptable maximum contents of these metals in materials applied in cultivated areas (class 0 – no restrictions, class I – 40 t/ha/10 years, class II – 20 t/ha/10 years) and nonproductive green areas (class III – application every 10 years with restriction to cultivation of neither food crops nor fodder crops) in Norway (FOR 2003)

CWs	Quality classes	Heavy metals (mg/kg TS)					
		Cd	Pb	Cu	Zn	Ni	Cr
	From 0-highest to III-lowest						
	0	0.4	40	50	150	20	50
	I	0.8	60	150	400	30	60
	II	2	80	650	800	50	100
	III	5	200	1000	1500	80	150
Haugstein/sand							
Inlet	0	0.4	9	18	40	14	21
Outlet	0	0.4	7	15	40	16	22
Haugstein							
Inlet	0/II(Cr)/III(Ni)	0.4	4	26	54	52	91
Outlet	0/I(Ni)	0.4	5	22	56	30	39
Tveter							
Inlet	0/I(Ni)	0.4	8	18	74	24	35
Outlet	0	0.4	7	36	38	19	26
Holt							
Inlet	0/III(Ni and Cr)	0.4	4	22	44	56	109
Outlet	0/II(Ni and Cr)	0.4	4	21	43	37	69
Dal							
Inlet	0/I(Ni)	0.4	5	22	24	28	25
Outlet	0/II(Ni and Cr)	0.4	4	24	24	49	66

common households should not carry high concentrations of heavy metals, yet continuous operation of CWs treating wastewater may cause an accumulation of some metals over time.

As shown in Table 3.3, the contents of Cd, Pb, Cu and Zn in filter media did respond to the concentrations of these metals defined for the highest quality class of materials applied in cultivated areas in Norway. The contents of Ni and Cr at the inlet sides of two CWs (Haugstein and Holt) slightly exceeded their maximum acceptable concentrations defined for quality class II in materials applied in cultivated areas (Table 3.3). However, these contents were still within the maximum permissible concentrations of Ni and Cr in materials applied exclusively in nonproductive green areas (i.e. not cultivated for food- and/or fodder crops). On the other hand, materials collected from the outlet sides of CWs did not reveal concentrations of all the metals tested higher than the maximum permissible contents for materials applied in cultivated areas (Table 3.3). It could therefore be assumed that these filter media were suitable for reuse in terms of contaminant contents defined by Norwegian regulations. However, other pollutants should also be considered prior to the reuse options, in particular emerging contaminants present in domestic wastewater, e.g. pharmaceutical-derived compounds. Pharmaceutical and personal care products

(PPCPs) are commonly used nowadays; thus their presence in water and wastewater cannot be neglected. For instance, Bergersen et al. (2012) traced citalopram and sertraline (the most commonly used nervous system drugs in Norway) in sewage sludge, which indicates that these drugs must also be present in wastewater. Filtralite® media have proved effective in PPCPs removal, achieving over 80 % efficiency (Matamoros et al. 2009). However, the fate of PPCPs in these well-performed filter media was never considered, thus the contamination effect of these media cannot entirely be stated.

Polycyclic aromatic hydrocarbons (PAHs) are the other water pollutants of environmental importance, because of their carcinogenicity and mutagenicity (Nkansah et al. 2012). There is a recent global concern about the increase in PAHs' contamination of water. The PAHs detected in highest concentrations in drinking water are fluoranthene, phenanthrene, pyrene and anthracene. Since they are in drinking water, they are also expected in wastewater; therefore, PAHs removal by on-site treatment systems should be highly considered. So far, this has been tested in laboratory studies, which revealed that phenanthrene, fluoranthene and pyrene can effectively be removed from water during batch sorption experiments using LECA as a sorbent (Nkansah et al. 2012). Physical sorption was the main mechanism that governed the removal process, which on the other hand caused the contamination of LECA filters.

All these examples demonstrate a high performance of LWA filter media with regard to the removal of different micropollutants, but the fate of these contaminants and their effect on the filter media must also be extensively investigated. This is of high importance for the reuse options of the exploited media from CWs treating domestic wastewater. It has been revealed that, even if the media are not highly contaminated, e.g. with faecal indicator bacteria – *Escherichia coli* (*E. coli*), this can drastically change during the storage period of these media (Paruch 2011). *E. coli* could survive and re-grow from relatively low initial concentrations and further persist in the specific LWA–Filtralite®P media for an extended period of time, over 14 months (Paruch 2011).

3.5 Conclusions

The concept of implementation of CWs in Norway was to imitate the natural ecosystems in soil media and wetlands with respect to their role in environmental pollution control through several mechanical, chemical and biological routes. Over the years of continuous performance, the treatment systems with filter/CW beds operating under cold climate conditions have shown a high and stable treatment efficiency regarding organic matter (high reduction of BOD and COD), nutrients (high P-binding capacity) and pathogens (high removal rates of viruses and bacteria). The removal efficiency of the main pollutants in treatment systems employing HSFCWs is relatively high, >90 % BOD, >50 % N, >90 % P and >99 % bacteria, providing that the systems are based on an appropriate design according to Norwegian

recommendations. Therefore, HSFCWs are one of the most efficient nature-based treatment systems in Norway with respect to removal of contaminants from domestic wastewater.

The Norwegian systems represent a sustainable technique of domestic wastewater treatment and a bio-economical option for filter media reuse. This is a challenging goal to reclaim and recycle P from wastewater. Thus, there are some options of reusing the P-rich filter media as a complementary P fertiliser, a soil amendment or a conditioner, provided the quality is acceptable for utilisation in agriculture. Alternatively, the filter media could be reused in some engineering works (e.g. growth media in urban greening projects, green roof technology, road screening technics or construction of embankments), if the quality allows application in the environment. Therefore, the core aspect of the reuse options is the appropriate quality of the filter media. As for the theoretical assumption, it should not be risky to reuse the P-rich media. In practice, however, they must be proven safe for human and environmental health prior to introducing into the environment.

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Chapter 4

Decomposition of *Phragmites australis* in Relation to Depth of Flooding

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Abstract Decomposition of *Phragmites australis* in relation to water depth was studied in the littoral reed stands of two fishponds near Prague during 2014. In both fishponds, three sites were selected along the water depth gradient – deep and shallow water in the littoral zone and above the water. The sites above the water were located in the area which never got inundated. *P. australis* aboveground biomass was collected at all sites in both fishponds in October 2013 and the collected biomass was divided into leaves, lower stems and upper stems (stem was divided in two parts of equal length). The litter bags with mesh size of 1 mm were filled with about 3 g of each plant part and exposed at all locations in six replications in December 2013. The litterbags were taken out in regular 3-month intervals, the biomass was divided into particular plant parts, carefully cleaned from the sediment, dried at 60 °C to a constant weight and weighed. It has been found that stems and leaves of *Phragmites australis* decompose at various rates in the order of leaves > upper stems > lower stems. The difference in decomposition rates for upper and lower stems and leaves was significant. Average time necessary for 50 % dry mass loss was 708 days for lower stems, 411 days for upper stems and 233 days for leaves. The decomposition of *P. australis* proceeded slower above the water than in the littoral zone under submerged conditions. There was no significant difference between decomposition rates in deep and shallow water. The decomposition rate above the water was slower by 20 %, 39 % and 45 % for lower stems, upper stems and leaves, respectively.

Keywords Decomposition • Macrophytes • *Phragmites australis* • Fishponds

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4.1 Introduction

In wetlands, the term decomposition is mostly confined to the breakdown and subsequent decay of dominant macrophytes which leads to the production of detritus (Boyd 1970). Most annual production in wetlands is not consumed by herbivores but decomposes on the wetland surface and becomes a part of a particulate carbon pool (Gallagher 1978; Chamie and Richardson 1978) because much of the carbon within macrophyte tissues (lignocellulose) is not readily digested and assimilated (Teal 1962). The macrophyte biomass eventually enters the detrital pool, where the microorganisms (bacteria and fungi) are involved in the breakdown and mineralization (Kuehn and Suberkropp 1998). Also, considerable part of organic material production plays an important role in the detritus food chain (e.g., Cummins et al. 1973; Menéndez et al. 2003).

The processes of macrophyte decomposition includes several steps, namely (a) mechanical fragmentation by animals, weather or other mechanisms, (b) leaching and/or autolytic production of dissolved organic matter, and (c) digestion of labile and recalcitrant materials by bacteria and fungi (Carpenter et al. 1983). The decomposition of *Phragmites australis* litter has been estimated by various methods such as dry mass loss (van der Valk et al. 1991; Gessner 2000; Dinka et al. 2004; Eid et al. 2014), ash-free dry mass loss (Hietz 1992; Longhi et al. 2008), microbial respiration (Mason and Bryant 1975; Komínková et al. 2000), nutrient loss from the biomass (van der Valk et al. 1991; Eid et al. 2014), bacteria and fungi dynamics on decomposing biomass (Tanaka 1991; Komínková et al. 2000; Van Ryckegeem et al. 2007) or changes in fibre materials (cellulose, hemicellulose, lignin) content of decomposing reed litter (Tanaka 1991; Dinka et al. 2004).

Decomposition processes are regulated by three interacting sets of factors: physico-chemical parameters in the environment (Carpenter and Adams 1979; Hietz 1992; Sangiorgio et al. 2008; Lopes et al. 2011; Qu et al. 2014), quality of decomposing material (Boyd 1970; Brinson et al. 1981; Wieder and Lang 1982; Aerts 1997; Liski et al. 2003; Smith and Bradford 2003; Rejmánková and Houdková 2006) and the decomposing organisms (Komínková et al. 2000; Findlay et al. 2002; Van Ryckegeem et al. 2006). Also, flooding conditions, i.e., drying and wetting of the decomposing material may affect substantially the decomposition rate (Harner and Stanford 2003; Langhans and Tockner 2006).

Decomposition rate proceeds faster under aerobic conditions (e.g., Balogh et al. 2006), however, it has been found that the macrophyte biomass decomposes faster under water than in dry conditions (Boyd 1970; Langhans and Tockner 2006; Wallis and Raulings 2011). Several studies have indicated that the fastest decomposition takes place under conditions of changing wet and dry conditions (Brinson et al. 1981; Battle and Golladay 2001; Wallis and Raulings 2011).

The objective of this study was to compare decomposition rate of various parts of *Phragmites australis* aboveground biomass in relation to various water depths.

4.2 Materials and Methods

Decomposition of *P. australis* (Common reed) was studied in the littoral zone of two fishponds in the outskirts of Prague, Czech Republic which are used to grow stock of common carp. The reed stands grew about 30 m into the littoral zone and the stands extend about 20 m above the water level. In both fishponds, three sites were selected along the water depth gradient – deep and shallow water in the littoral zone and above the water. The sites above the water were located in the area which never gets inundated. The water quality and water depth were measured during 2014 in 3 weeks interval. Dissolved oxygen, oxygen saturation, pH, temperature and electric conductivity were measured in the field using HQ30d Portable Multi-Parameter Meter (Hach Lange, Loveland, USA). Nitrate-N, nitrite-N, ammonia-N, orthophosphate and total phosphorus were measured within 24 h according to the Standard methods (APHA 1998) using UV-VIS spectrophotometer Cary 60 (Agilent Technologies, Walbronn, Germany). Total nitrogen and carbon were measured using the Formacs TOC/TN analyzer (Skalar, Breda, The Netherlands).

P. australis aboveground biomass was collected at all sites in both fishponds in October 2013 and the collected biomass was divided into leaves, lower stems and upper stems (stem was divided in two parts of equal length). Total nitrogen and carbon in the biomass were measured using the Primacs SNC analyzer (Skalar, Breda, The Netherlands). Phosphorus was analyzed following the $\text{HNO}_3/\text{HClO}_4$ method of Sommer and Nelson (1972). NIST 1547 Peach Leaves were used as the standard (National Institute of Standards and Technology, Gaithersburg, MD, USA).

The litter bags (15×15 cm) with mesh size of 1 mm were filled with about 3 g of each plant part and exposed at all locations in six replications in December 2013 (Fig. 4.1). The submergence of litterbags on the bottom was ensured by placing a small stone in each bag. The litterbags were taken out in regular 3-month intervals, the biomass was divided into particular plant parts, carefully cleaned from the sediment, dried at 60 °C to a constant weight and weighed. For each location decomposition constant was calculated according to the first-order exponential decay model (Jenny et al. 1949; Olson 1963; Wieder and Lang 1982):

$$\ln(W_t / W_0) = -kt,$$

where W_t and W_0 is the dry biomass at the end and in the beginning of the experiments, respectively and t is time in days (in this case 365 days). The time necessary for 50% loss of dry mass (t_{50}) was calculated according to the formula $T_{50} = 0.693 k^{-1}$, time necessary for 95% loss was calculated according to the formula $T_{95} = 3 k^{-1}$ (Olson 1963).

Statistical analysis of variance (ANOVA) with Turkey HSD test was used to determine difference between decomposition at various water depths and between various plant parts. Significance level was set at $p < 0.05$.

Fig. 4.1 Decomposition sites – above the water (*top*), shallow water (*middle*) and deep water (*bottom*)



4.3 Results and Discussion

4.3.1 Water Chemistry and Water Depth

The water depth at the deep littoral sites in Ponds I and II averaged 31.5 cm and 44.8 cm, respectively (Table 4.1) while respective average values for shallow littoral sites were 1.3 cm and 4.5 cm. The water level in shallow site in Pond I was below the sediment surface during 6 out of the 18 visits and the lowest water level was 7 cm

Table 4.1 Water chemistry of Ponds I and II littorals in the period of January 21 – December 29, 2014, n=18, standard deviations in parentheses

	Pond I	Pond II
Water depth, deep (cm)	31.5 (7.5)	44.8 (5.7)
Water depth, shallow (cm)	1.3 (5.3)	4.5 (3.4)
Bank, water below surface (cm)	-28.1 (6.4)	-12.2 (4.7)
Dissolved oxygen (mg/l)	4.3 (2.4)	10.6 (3.5)
Oxygen saturation (%)	39.1 (20.1)	100.9 (31.2)
pH	7.00 (0.27)	7.85 (0.84)
Electric conductivity (mS/m)	53.8 (11.9)	48.7 (67.1)
N-NH ₄ (mg/l)	0.57 (0.48)	0.34 (0.23)
N-NO ₂ (mg/l)	0.03 (0.02)	0.10 (0.08)
N-NO ₃ (mg/l)	3.5 (3.8)	2.6 (2.2)
TN (mg/l)	5.4 (2.6)	3.8 (1.5)
P-PO ₄ (mg/l)	0.15 (0.15)	0.21 (0.16)
TP (mg/l)	0.67 (0.36)	0.73 (0.29)
TN/TP	12.9 (14.9)	7.7 (8.5)
TC (mg/l)	33.8 (8.7)	28.1 (8.7)
TC/TN	7.8 (3.7)	8.9 (5.0)
TC/TP	82.2 (93.3)	52.7 (53.3)

below the surface. At Pond II, the water level was observed below the surface only once (2 cm below the surface). Water level at the sites above the water was well below the surface all the time (Table 4.1) and the highest water levels were 17 cm and 7 cm below the surface at Ponds I and II, respectively. The *P. australis* stand was much denser in Pond I with much less light penetrating the canopy. As a consequence the average dissolved oxygen concentration and pH were much higher in Pond II. In Pond I the total nitrogen was slightly higher than in Pond II (Table 4.1) while total phosphorus concentration was slightly higher in Pond II (Table 4.1). The nutrient content in water of both Ponds were quite high due to fertilization of the ponds and also most probably due to drainage of nutrients from surrounding agricultural fields. The TN/TP ratios in both Ponds (12.9 in Pond I and 7.7 Pond II) indicated nitrogen limitation (Verhoeven 1986). However, the concentrations of nutrients did not differ significantly in both Ponds ($p > 0.05$).

4.3.2 Decomposition of Various Plant Parts

The decomposition constants varied among various parts of *P. australis* and increased in the order lower stems, upper stems and leaves (Fig. 4.2) indicating fastest decomposition of leaves. The difference between decomposition of various plant parts at all sites were statistically significant ($p < 0.05$). Our results fully support the suggestion reported by Gessner (2000) who pointed out that in order to evaluate

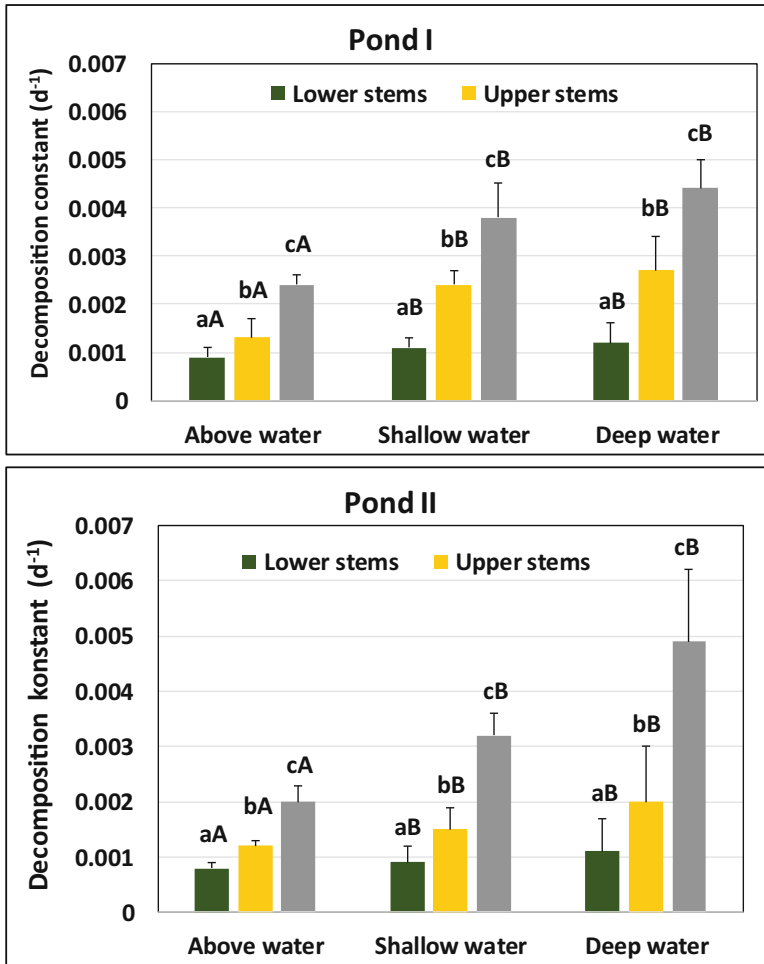


Fig. 4.2 The decomposition constants of various *P. australis* parts at the study sites. Bars represent standard deviations. Different letters indicate significant difference at $\alpha=0.05$ between the means. The *lower case letters* compare decomposition constants for the leaves, upper and lower stems at each site. *Upper case letters* compare the same plant part at various sites in relation to water depth

breakdown of *Phragmites* shoots, it is necessary to separate leaves and stems and also to consider leaves and stems from various heights.

Gessner (2000) speculated that the slow decomposition of the stems may be caused by the fact that the microbial and detritivore attack may be restricted initially to the most refractory outer tissue layers of the culms similar to the peripheral decay of woody debris in freshwaters (Anderson et al. 1984). Ágoston-Szabó and Dinka (2008) reported that decomposition differences of plants are often related to lignin concentrations found in plant cell walls. Lignin is especially resistant to microbial

degradation because of its complex chemistry of cross-linked polymers and may also reduce bioavailability of other cell wall constituents to decomposition. Dinka et al. (2004) explained the faster decomposition of leaves than stems by higher content of slow decomposing lignin in stems.

The decomposition rates of the leaves, upper and lower stems differed significantly ($p < 0.05$) at all sites regardless of inundation (Fig. 4.2). On average, decomposition of the lower stems was $3.38 \times$ slower than leaves and $1.81 \times$ slower than upper stems. Zhang et al. (2014) studied decomposition of *P. australis* stem segments of 0–30 cm, 30–80 cm, 80–130 cm and 130–180 cm above the ground. These results of this study revealed that the decomposition of stems from the 0–30 cm segments was by about 70 % slower than decomposition of stems from the 80–130 cm segments above the ground.

Important factors affecting the decomposition rate are the C/N and C/P concentration ratios in the biomass. Reddy and DeLaune (2008) reported that decomposition rate decreased very sharply with decreasing C/N and C/P concentration ratios and decomposition was negligible at $C/N > 40$ and $C/P > 200$. Enriquez et al. (1993) suggested that C/N and C/P ratios were excellent predictors of decomposition rates with strong negative correlation between decomposition rates and C/N and C/P ratios. A similar trend was reported also by Li et al. (2013) or Rejmánková and Houdková (2006). In our study, C/N and C/P ratios increased in the order leaves (27.4 and 449, respectively), upper stems (67.1 and 1897) and lower stems (81 and 3155) while the decomposition rates decreased in the same order (Vymazal and Březinová 2016).

4.3.3 Decomposition in Relation to Water Depth

The results shown in Fig. 4.2 indicate that the decomposition above the water was slower ($p < 0.05$) than the decomposition at shallow and deep water sites. The differences between the submerged sites were not significant (Fig. 4.2). In Tables 4.2 and 4.3, comparison with literature data for *P. australis* leaves and stems are presented. The comparison reveals that the decomposition rates measured above the water belong to the slowest rates reported for *P. australis*.

Slow decomposition under dry conditions has been reported by several authors (Van der Valk et al. 1991; Hietz 1992; Bedford 2005; Liu et al. 2010; Wallis and Raulings 2011). Van der Valk et al. (1991) observed that the time necessary for 50 % dry mass loss of *P. australis* on the wetland surface was about 2.5 times longer than 100 cm below the water surface of the Delta Marsh, Canada. Hietz (1992) reported that *Phragmites* leaves ash-free dry weight (AFDW) remaining after 1 year was by 23.9 % higher above the water as compared to leaves decomposing below the water in the Lake Neusiedl, Austria. Bedford (2005) found that decomposition of *P. australis* leaves was three times slower at a dry site as compared to a seasonally flooded site in Leighton Moss, N.W. England (Table 4.2). The author suggested that drying of the litter led to slowing or complete cessation of breakdown.

Table 4.2 The exponential decomposition constants ($-k$) and time necessary for 50% (T50) and 95% (T95) dry mass decrease of *P. australis* leaves reported in the literature and comparison with values observed in this study

Location	$-k$ (d^{-1})	t50 (d)	T95 (d)	Reference
River Tirso basin, Italy	0.0193	36	155	Pinna and Basset (2004)
Lake Burullus, Egypt	0.0117	59	256	Eid et al. (2014)
Sunnyside Swamp, Australia	0.009	77	333	Janssen and Walker (1999)
Lake Aiguebelette, France	0.0087	80	344	Blake (1982)
Lake Baiyangdian, China	0.0081	86	370	Lan et al. (2012)
Lake Geestmerambacht, The Netherlands	0.0078	89	384	Van Dokkum et al. (2002)
Ria de Aveiro, Portugal	0.0076	91	395	Lopes et al. (2011)
Shanyutan wetland, China	0.0053	131	566	Zhang et al. (2014)
Hackensack Meadowlands, NJ, USA	0.0052	133	576	Windham et al. (2004)
Pond II – deep water	0.0049	141	612	This study
Lake Belau, Germany	0.0045	154	667	Gessner (2000)
Pond I – deep water	0.0044	160	681	This study
Pond I – shallow water	0.0038	182	789	This study
Leighton Moss wetland, U.K. ^a	0.0036	193	833	Bedford (2005)
Scheldt Estuary, The Netherlands	0.0035	198	857	Van Ryckegem et al. (2007)
Pond II – shallow water	0.0032	217	938	This study
Lake Sortesø, Denmark	0.00286	242	1049	Larsen and Schierup (1981)
Kis-Balaton, Hungary	0.0028	248	1071	Balogh et al. (2006)
Lake Fertő, Hungary	0.0026	267	1154	Dinka et al. (2004)
Pond I – above water	0.0024	289	1250	This study
Lake Hampen, Denmark	0.00237	293	1266	Larsen and Schierup (1981)
Pond II – above water	0.0020	347	1500	This study
Leighton Moss wetland, U.K. ^b	0.0013	533	2307	Bedford (2005)

^aWet side

^bDry site

Wallis and Raulings (2011) found out that litter exposed at a dry site decomposed more slowly due to desiccation of microbial and fungal communities. The authors found that the time necessary for 50% dry mass loss was 289 and 630 days in samples located 10 cm below the water and above the water, respectively. Another important factor favoring decomposition in wet conditions is the number of bacteria on decomposing material. Úlehlová (1976) found substantially higher bacteria counts on *P. australis* decomposing in aquatic conditions than in standing litter above the water.

Table 4.3 The exponential decomposition constants ($-k$) and time necessary for 50% (T50) and 95% (T95) dry mass decrease of *P. australis* stems reported in the literature and comparison with values observed in this study

Location	$-k$ (d ⁻¹)	T50 (d)	T95 (d)	Reference
Lake Baiyangdian, China	0.0042	165	714	Lan et al. (2012)
Lake Burullus, Egypt	0.0036	193	833	Eid et al. (2014)
Lake Aiguebelette, France	0.0031	224	968	Blake (1982)
Pond I – Upper stems, DW	0.0027	257	1111	This study
Pond I – Upper stems, SW	0.0024	288	1250	This study
Shanyutan wetland, China	0.0023	301	1304	Zhang et al. (2014)
Pond II – Upper stems, DW	0.002	347	1500	This study
Seaftinghe Marsh, The Netherlands	0.0019	364	1579	Van Ryckegem et al. (2006) ^c
Hackensack Meadowlands, NJ, USA	0.0018	386	1667	Windham et al. (2004)
Lake Belau, Germany	0.0015	462	2000	Gessner (2000)
Pond II – Upper stems, SW	0.0015	462	2000	This study
Pond I – Upper stems, AW	0.0013	533	2308	This study
Pond I – Lower stems, DW	0.0012	578	2500	This study
Pond II – Upper stems, AW	0.0012	578	2500	This study
Pond I – Lower stems, SW	0.0011	630	2727	This study
Pond II – Lower stems, DW	0.001	635	3000	This study
Leighton Moss wetland, U.K. ^a	0.001	693	3000	Bedford (2005)
Pond I – Lower stems, AW	0.0009	770	3333	This study
Pond II – Lower stems, SW	0.0009	770	3333	This study
Kis-Balaton, Hungary	0.00082	845	3659	Balogh et al. (2006)
Leighton Moss wetland, U.K. ^b	0.0008	866	3750	Bedford (2005)
Pond II – Lower stems, AW	0.0008	866	3750	This study
Lake Sortesø, Denmark	0.00071	976	4225	Larsen and Schierup (1981)
Lake Fertő, Hungary	0.0005	1386	6000	Dinka et al. (2004)
Lake Hampen, Denmark	0.00042	1665	7143	Larsen and Schierup (1981)

^aWet side^bDry site^cAFDW

4.4 Conclusions

The results revealed that stems and leaves of *Phragmites australis* decompose at various rates in the order of leaves > upper stems > lower stems. The difference in decomposition rates for upper and lower stems and leaves was significant. Average time necessary for 50% dry mass loss was 708 days for lower stems, 411 days for upper stems and 233 days for leaves. The higher decomposition rate of leaves is supported by higher C/N and C/P ratios in the leaves.

The decomposition of *P. australis* proceeded slower above the water than in the littoral zone under submerged conditions. There was no significant difference

between decomposition rates in deep and shallow water. The decomposition rate above the water was slower by 20 %, 39 % and 45 % for lower stems, upper stems and leaves, respectively.

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Chapter 5

Distribution of Phosphorus and Nitrogen in *Phragmites australis* Aboveground Biomass

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Abstract The objective of this study was to evaluate nitrogen and phosphorus compartmentalization in the aboveground biomass of *Phragmites australis* and its seasonality. The study was carried out at four sites in the littoral zones of two fishponds near Prague, Czech Republic. The aboveground biomass was harvested in early July and late August and the biomass was divided into thirds and stems and leaves. Besides dry weight of the biomass, concentrations of nitrogen and phosphorus were determined. The results revealed that the proportion of biomass formed by stems increases during the season. The concentrations of both nitrogen and phosphorus were higher in July and higher in leaves. Phosphorus standing stock is higher in stems than leaves while nitrogen standing stock is higher in leaves than in stems. The highest N and P standing stocks were found in upper leaves both in July and August.

Keywords Standing stock • Nitrogen • Phosphorus • *Phragmites australis*

5.1 Introduction

Nitrogen and phosphorus are taken up and assimilated by growing plants throughout the growing season. However, the highest concentrations in live plant tissue occur early in the growing season and decrease as the plants mature and senesce (Tyler 1971; Johnston 1991). The length of growing season varied widely among macrophytes with senescence occurring between July and September in temperate regions (Dykyjová 1973).

Total storage of a substance in a particular compartment is called standing stock. Nutrient standing stocks in the vegetation are commonly computed by multiplying nutrient concentrations in the plant tissue by dry mass per unit area and are expressed

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as mass per unit area (usually g m^{-2} or kg ha^{-1}) (Johnston 1991; Richardson and Vymazal 2001). The standing stock is affected both by dry mass and concentration of a particular element of interest. However, it has been shown that the biomass is the more important factor as it changes during the growing season more than nitrogen and phosphorus concentrations in stems and leaves (e.g. Boyd 1970; Eid et al. 2012). The nutrient standing stock seasonality has been documented many times in the literature (e.g. Dykyjová 1973; Vymazal 2005), however, the information how is the nutrient standing stock distributed among leaves and stems during the season is very limited. The objective of this study was to evaluate nitrogen and phosphorus compartmentalization in the aboveground biomass of *Phragmites australis* and its seasonality.

5.2 Materials and Methods

The experiments were carried out at four sites in the littoral zones of two fishponds near Prague, Czech Republic. The aboveground biomass was sampled on July 5 and August 29, 2014. Aboveground biomass of *P. australis* was harvested from the 0.25 m^2 ($50 \times 50 \text{ cm}$) quadrants in four replicates at each site. The shoots were clipped at the ground or sediment level, and all the shoots were divided into thirds and divided into leaves (L, including leaf sheaths), stems (S) and flowers (if present). The lower parts were labelled S1/3 and L1/3, middle parts S2/3 and L2/3 and upper parts of the shoots were labelled S3/3 and L3/3.

The biomass was dried at $60 \text{ }^\circ\text{C}$ to a constant weight and weighed. For the further analyses the biomass was ground using the cutting mill Pulverisette 15 (Fritsch, Idar-Oberstein, Germany) and analyzed for nitrogen and phosphorus. Total nitrogen was measured directly by using the Primacs SNC analyzer (Skalar, Breda, The Netherlands). Phosphorus was analyzed following the $\text{HNO}_3/\text{HClO}_4$ method of Sommer and Nelson (1972). NIST 1547 Peach Leaves were used as the standard (National Institute of Standards and Technology, Gaithersburg, MD, USA).

Statistical analysis of variance (ANOVA) followed by post-hoc Tukey HSD test was used to test differences between biomass and nutrients standing stocks of various plant parts. The significance level was set at $p < 0.05$.

5.3 Results and Discussion

5.3.1 *Distribution of Aboveground Biomass of P. australis Between Stems and Leaves*

The distribution of biomass among stems, leaves and flowers is shown in Fig. 5.1.

In July, the stems constituted 55.8% (± 2.3) of the aboveground biomass while the contribution of leaves was only 44.2% (± 2.5) ($p < 0.05$). In August, stems

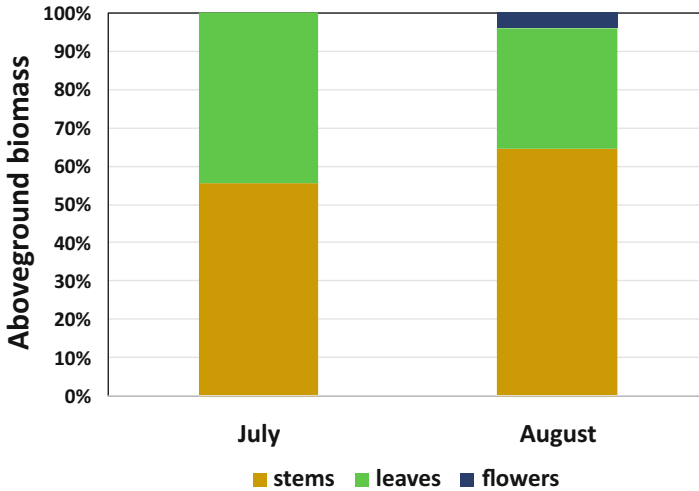


Fig. 5.1 Distribution of *P. australis* aboveground biomass among stems, leaves and flowers in July 4 and August 29, 2014

increased their contribution to 64.4% (± 1.5) while leaves formed only 31.7% (± 0.66) of the biomass ($p < 0.05$). In addition, flowers comprised 3.9% (± 1.1) of the biomass. Both the increase of stem biomass and decrease of leaf biomass were statistically significant ($p < 0.05$).

5.3.2 Distribution of Phosphorus in Aboveground Biomass

Concentrations of phosphorus in various *P. australis* parts are shown in Fig. 5.2. The data presented in Fig. 5.2 indicate that (a) P concentrations are higher in July than in August for both leaves ($p > 0.05$) and stems ($p < 0.05$), (b) phosphorus concentrations differ significantly ($p < 0.05$) in the lower, middle and upper leaves while there is no significant difference for stems. The highest concentrations of phosphorus were found in upper leaves both in July and August. In July, the difference from most other compartments was not significant ($p > 0.05$) while in August the upper leaves concentration was significantly higher than all other compartments ($p < 0.05$).

The major compartments for phosphorus storage are upper leaves and lower parts of the stems (Fig. 5.3). There is significant difference ($p < 0.05$) in standing stocks among lower, middle and upper parts of leaves and stems but there is no significant difference among values found in early July and late August. The highest P standing stock was found in upper leaves in July (33.1%) and in lower stems in August (28.7%).

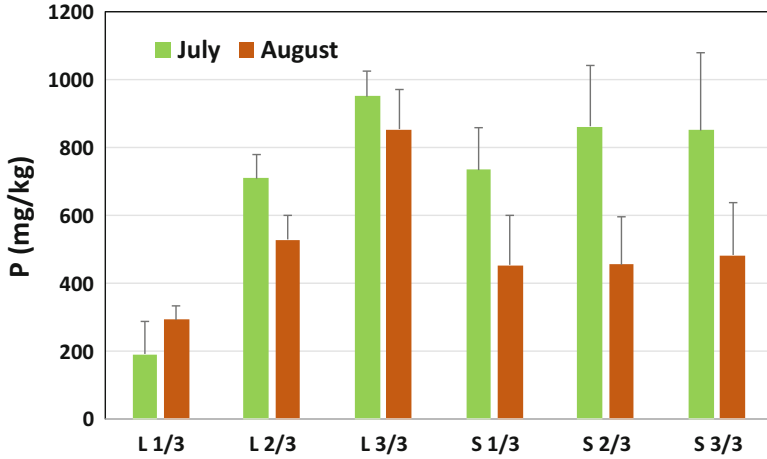


Fig. 5.2 Concentration of phosphorus in various parts of *P. australis* biomass. Bars represent standard deviation

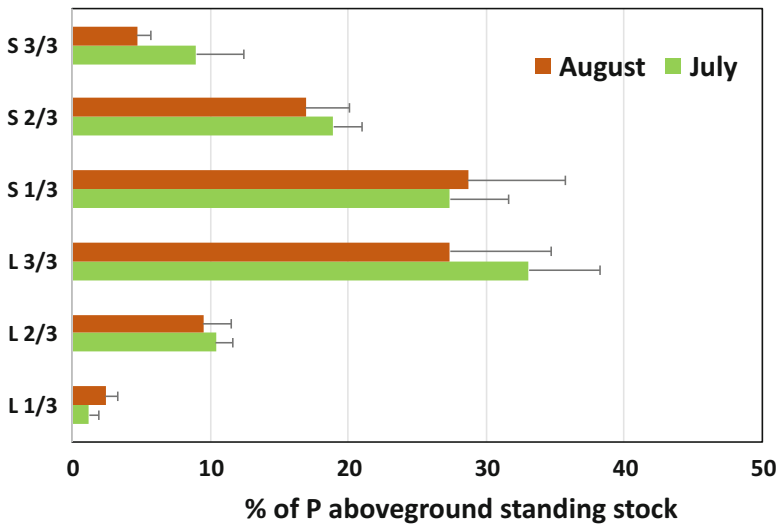


Fig. 5.3 Distribution of P standing stock in various compartments of *P. australis* shoots. Bars represent standard deviation

In July, the total P standing stock amounted to 55.3% (± 5.2) in stems and 44.7% (± 5.2) in leaves ($p < 0.05$). In August, the average stem standing stock dropped to 50.3% (± 10.4) and leaf standing stock decreased to 39.3% (± 7.3). At this time, the difference was not significant ($p > 0.05$). Flowers contributed with 11.4% (± 2.4) (Fig. 5.4). In the only comparable study, Dykyjová (1973) describing distribution of

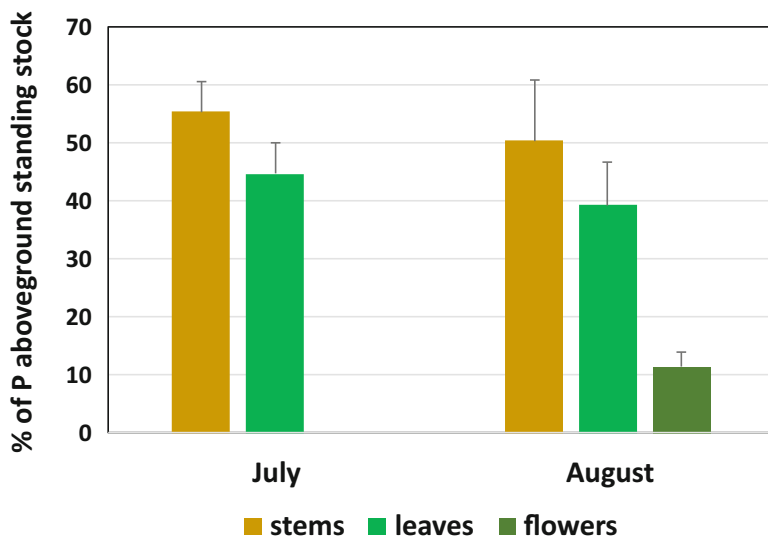


Fig. 5.4 Distribution of P standing stock among stems, leaves and flowers of *P. australis* shoots. Bars represent standard deviation

P standing stock in *P. australis* as follows: 67 % stems and 33 % leaves in July. In August, the respective compartments decreased to 42 % and 26 % with as much as 32 % of phosphorus being accumulated in flowers.

5.3.3 Distribution of Nitrogen in Aboveground Biomass

Concentrations of nitrogen (Fig. 5.5) followed exactly the same pattern as concentrations of phosphorus. The highest concentrations were found in the middle and upper leaves and concentration of nitrogen was significantly higher in these compartments than in other compartments both in July and August ($p < 0.05$). The highest nitrogen concentrations in the middle and upper leaves were also reported by Dykyjová (1973). Figure 5.6 indicates that the highest nitrogen standing stock is in upper leaves. In July it formed as much as 48.5 % of the total nitrogen standing stock, while in August it was 36.8 %. In all other compartments, the nitrogen standing stock was significantly lower ($p < 0.05$).

Total nitrogen standing stock (Fig. 5.7) was distributed in opposite way to phosphorus. The higher standing stock was found in leaves (65 ± 4.0 % in July, 55 ± 3.0 % in August) than in stems ($p < 0.05$). The contribution of stems to the total N standing stock was very similar in July (35 ± 4.0 %) and in August (34.4 ± 5.3 %). In August, 10.6 ± 2.7 % of N standing stock was allocated to flowers. Dykyjová (1973) observed 47 % and 53 % of N standing stock in stems and leaves, respectively in July. In August, the respective compartments decreased to 29 % and 54 % with as much as 17 % of phosphorus being accumulated in flowers.

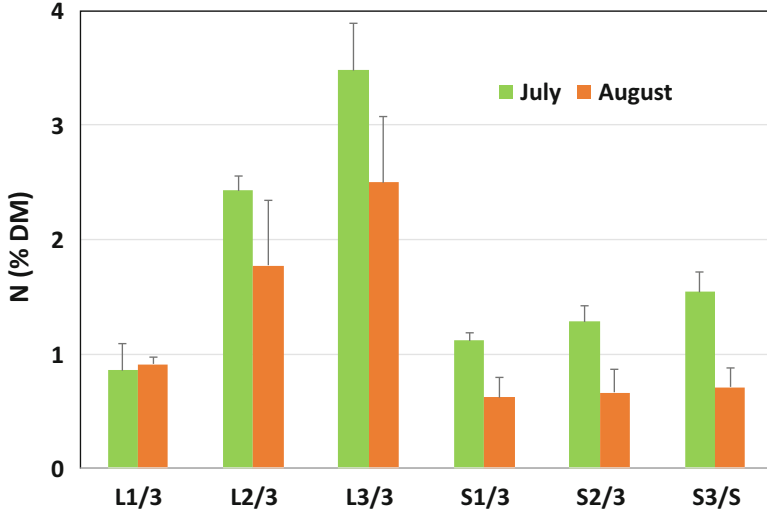


Fig. 5.5 Concentration of nitrogen in various parts of *P. australis* biomass. Bars represent standard deviation

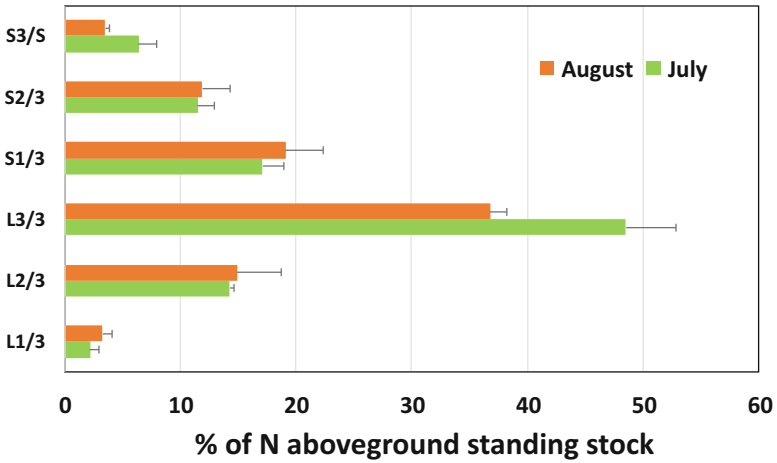


Fig. 5.6 Distribution of N standing stock in various compartments of *P. australis* shoots. Bars represent standard deviation

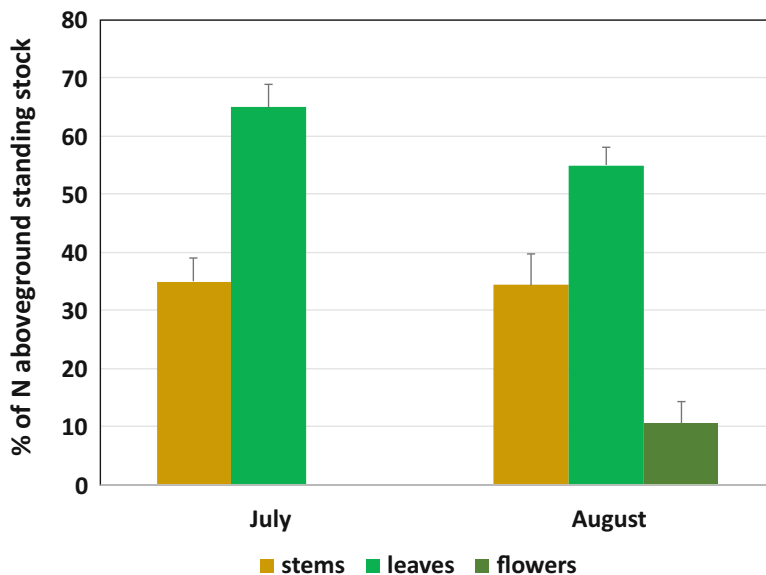


Fig. 5.7 Distribution of N standing stock among stems, leaves and flowers of *P. australis* shoots. Bars represent standard deviation

5.4 Conclusions

The results of our experiments revealed that the stem:leaf biomass ratio increased from 1.26 in early July to 2.03 in late August.

Concentrations of both nitrogen and phosphorus were higher in July than in August with the highest concentrations of both elements being found in upper leaves.

Phosphorus standing stock is higher in stems than leaves while nitrogen standing stock is higher in leaves than in stems. The highest N and P standing stocks were found in upper leaves.

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Chapter 6

How Many Samples?! Assessing the Mean of Parameters Important for Denitrification in High and Low Disturbance Headwater Wetlands of Central Pennsylvania

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Abstract Due to large time and monetary costs involved with sampling for denitrification and other biogeochemical processes, many studies are limited to only a few samples per site. However, denitrification is a highly spatially and temporally variable process due to high variability of the biogeochemical parameters underpinning denitrification such as nitrate (NO_3^-), dissolved organic carbon (DOC), dissolved oxygen (DO), temperature, and pH. Accurately assessing wetlands for the means of parameters underpinning denitrification is vital, as inaccurate means can result in incorrect estimates of wetland-scale denitrification as well as inaccurate parameterization and validation of denitrification models. To examine adequate sample size for these parameters, shallow groundwater samples were collected from twenty wells each at two high disturbance (surrounded by agriculture) and two low disturbance (surrounded by forest) headwater riparian wetlands in central Pennsylvania on May 21st, 2013. A Monte Carlo analysis was used along with a cutoff of 10% coefficient of variation to assess the appropriate number of samples required to accurately assess mean temperature, pH, DO, DOC, and NO_3^- within headwater wetlands. Temperature and pH required only one sample to accurately assess the site mean. DO required seven or more samples, while DOC and NO_3^- required ten or more samples to accurately assess the site mean. No differences were noted in required sample size between high and low disturbance sites. These results highlight the importance of taking numerous samples for wetland denitrification studies.

Keywords Denitrification • Sample size • Headwater wetlands • Dissolved organic carbon • Nitrate

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6.1 Introduction

Because of the great importance of denitrification in maintaining water quality, many studies have been devoted to understanding the underpinnings of this vital ecosystem service. However, measuring denitrification is resource intensive, resulting in studies that often use a handful of sampling points per site or treatment (Groffman et al. 2006). Furthermore, the most common methods for measuring denitrification, such as acetylene block, involve small point-scale measurements (Groffman et al. 2006). Denitrification is known as a highly variable process (Hedin et al. 1998; McClain et al. 2003), and thus a low number of small-scale measurements are unlikely to properly assess denitrification at a site scale. This results in studies that are unable to accurately assess connections between denitrification and the underlying parameters driving this process.

As a microbial process, denitrification is primarily driven by environmental parameters that directly affect microbes. While many factors can influence the ability of microbes to perform denitrification, the most influential factors are temperature, pH, dissolved oxygen (DO) concentration, dissolved organic carbon (DOC) concentration, and nitrate (NO_3^-) concentration (Hochstein et al. 1984; Parkin and Tiedje 1984; Seitzinger 1994; Simek and Cooper 2002; Sirivedhin and Gray 2006). The spatial and temporal variability of these parameters in soil and groundwater have been found to highly influence denitrification (Hedin et al. 1998; McClain et al. 2003), and thus accurate assessments of these parameters at a site or treatment level are important in order to assess drivers of denitrification within a study. These parameters are also often found within many popular models designed to predict denitrification in both soil and water, such as the DeNitrification-DeComposition (DnDC) model and Created Wetlands Model #1 (CWM1) (Li et al. 1992; Langergraber et al. 2009). Thus, it is important to be able to accurately characterize these parameters at a site or treatment level to ensure that these models can provide accurate predictions of denitrification.

Disturbance to wetlands has been found to influence not only the overall concentration, but the variability of these important drivers within a site. A study by Moon (2012) found that spatial variability of multiple edaphic factors such as soil organic matter (SOM) and ammonium (NH_4^+) within a 1000 m² area was greatly reduced in headwater wetlands set in an agricultural landscape compared to headwater wetlands set in forested landscapes. Overall SOM levels were also found to be significantly lower in wetlands in the agricultural landscape (Moon 2012). Because SOM is known to be an important source of DOC (Hagedorn et al. 2004), and NH_4^+ is able to be transformed into NO_3^- via nitrification (Jones and Hood 1980), the differences in concentration and variability of these edaphic parameters may influence the concentration and variability of DOC and NO_3^- within wetland shallow groundwater, which may then affect denitrification.

Accurate denitrification studies cannot be done without an appropriately large sample size; however, resource limitations often stymie the ability of researchers to obtain them. Because disturbance has been shown to affect the spatial variability of

edaphic factors important to denitrification (Moon 2012), different sample sizes may be necessary to properly assess sites of differing disturbance levels. Therefore, we wish to examine the number of samples required to accurately assess the parameters underpinning denitrification in shallow groundwater in both high and low disturbance sites to provide researchers with guidelines as to the minimum number of samples required to provide accurate assessments of these important variables. To do this, we installed twenty shallow groundwater wells within a 400 m² area and analyzed water samples for temperature, pH, DO, DOC, and NO₃⁻ from two headwater wetland sites in forested settings (low disturbance sites) and two headwater wetland sites in agricultural settings (high disturbance sites) in central Pennsylvania. With our findings we hope to provide clearer guidelines for those studying denitrification within wetlands.

6.2 Methods

6.2.1 Sites

Four headwater wetland sites located in the Upper Juniata watershed (HUC 02050302, 255,717 ha) were selected for this case study. The Upper Juniata watershed is located in the Ridge and Valley physiographic province of central Pennsylvania and is approximately 70% forested, 27% agriculture, and 3% urban (Wardrop et al. 2007b). Average wetland size in the Ridge and Valley physiographic province is small, with most sites being about 0.4 ha in size (Brooks et al. 2013). Headwater wetlands are the most common type of wetlands in the Ridge and Valley physiographic province and are associated with small (1^o, 2^o, and 3^o) streams (Penn State Cooperative Wetlands Center 2004). For the sites in this study, all soils are alluvial in deposition and overlie shale primary lithologies. The wetlands are groundwater supported, with lower water levels during the summer months and high water levels fall through spring.

Sites were sorted into two disturbance classes: high disturbance (Mosquito and Cauldron) and low disturbance (Whipple Dam and Shavers), based on a Rapid Assessment score (Wardrop et al. 2007a). The Rapid Assessment score is an indicator of wetland condition based on surrounding land cover, number of stressors within a wetland, and the size and quality of the buffer zone around the wetland (Moon and Wardrop 2013). The score ranges from 0 to 100 with higher numbers indicating greater anthropogenic impacts (Moon and Wardrop 2013). Rapid Assessment scores had been previously calculated for each site (Wardrop et al. 2007b). Sites with a score higher than 50 were classified as high disturbance while sites with a score lower than 50 were classified as low disturbance. Low disturbance sites are primarily surrounded by forest (>75%) within a 1 km radius while high disturbance sites are surrounded by a mixed-use landscape of agriculture (30–60%), urban (0–1%), and forest (<40%) within a 1 km radius.

6.2.2 Water Quality Analysis

To sample shallow groundwater, twenty shallow groundwater wells were installed in a random pattern across a 400 m² plot located within each site. Wells were constructed of 20-slot PVC screen approximately 15 cm in length, with 5.1 cm diameter PVC casing. Wells were installed to collect shallow ground water between the depths of –30 and –45 cm below ground level; slightly below the average depths of water for headwater wetlands in the area (Hychka et al. 2013). Water samples were collected from each well that contained water on May 21st, 2012. Wells were purged 1–2 days prior to sampling. A peristaltic pump with Teflon® tubing was used to collect water samples in 125 ml ashed amber glass bottles (DOC) and 125 ml HDPE plastic bottles (NO₃⁻). DO was recorded in the field using a YSI ProODO optical oxygen sensor. Temperature and pH were measured with an YSI 63 m. All samples were kept on ice in coolers until transport back to the lab where they were stored at 4 °C. Samples were filtered within 6 days and analyzed within 10 days of collection. DOC samples were filtered using Whatman GF/F filters while NO₃⁻ samples were filtered using GH Polypro 47 mm 0.45 µm Hydrophilic Polypropylene membrane filters. NO₃⁻ was analyzed using standard method 4110B (Dionex ICS-3000 Ion Chromatograph). DOC (as NPOC) was analyzed using the high temperature combustion method (Shimadzu TOC-VCPH w/ASI-V).

6.2.3 Monte Carlo Analysis and Statistical Analyses

Monte Carlo simulations were performed in R to randomly resample and calculate the average for 1, 3, 5, 7, and 10 samples 100 times for each variable. The coefficient of variation (CoV) was then determined by dividing the standard deviation of the 100 simulations by the actual measured mean of each variable as determined by the water quality analysis. We used a 10% cutoff as the marker for accurate assessment of the population mean, with CoV's below 10% representing an accurate assessment (Cohen et al. 2008; Moon 2012). Comparisons between high and low disturbance sites for all water quality variables were done using non-parametric Kruskal Wallis tests in JMP®. Version. 9.0.2. SAS Institute Inc., Cary, NC, 1989–2007. All graphs were created using R Version 2.15.1.

6.2.4 Literature Review

We examined the literature for studies on denitrification in riparian wetlands between the years of 2000 and 2015 to see, on average, how many samples were being taken in contemporary denitrification studies within these systems. The literature search was performed using Google Scholar, with the search terms

“denitrification” and “riparian wetlands”, and excluding the terms “constructed”, and “tidal”. We then examined the results and selected papers that directly measured denitrification in both natural and reconstructed riparian wetlands using any type of denitrification method. Papers where sample sizes or methods were unclear were removed from the analysis. We found 34 papers that fit our criteria and recorded the methodology used, the number of samples collected per site or treatment, and size of the site or treatment area (when available). Notes were made if samples sizes differed between sites or sampling events. If sample sizes were the same between sites or sampling events then only the number of samples per site or sampling event was noted. If a study examined more than riverine wetlands, only the number of riverine wetlands was noted for the number of sites. Though we also wished to compare sampling density between the studies, in the vast majority of cases there was not enough detailed information in the methods sections of the papers to accurately calculate this metric. However, we did include information on the areal size of the studies when available.

6.3 Results

A minimum of 11 samples were able to be collected at each site. Mean pH was slightly acidic, between 6.18 and 6.82, and was significantly higher at high disturbance sites compared with low disturbance sites (Table 6.1). Mean temperature was between 15.75 and 18.26 °C and while there were significant differences between sites, they did not differ across the disturbance gradient (Table 6.1). Mean NO_3^- was very low (below 0.06 mg/L) for all sites, and highly variable at high disturbance sites (Table 6.1). While significant differences in mean NO_3^- were found between sites, they were not found across a disturbance gradient. Mean DO was between 1.25 and 2.29 mg/L for all sites, and while significant differences were noted between sites, the differences did not align along the disturbance gradient (Table 6.1). Mean DOC was between 2.33 and 6.64 mg/L, with mean DOC being significantly higher at high disturbance sites (Table 6.1).

For the 400 m² study areas, only one sample was needed to accurately characterize mean pH and temperature at the site level within 10% of the population mean,

Table 6.1 Means and standard deviations of parameters important for denitrification at low and high disturbance sites. Significant differences denoted with different letters.

Dist. level	Site name	N	pH	Temperature (°C)	NO_3^- (mg/L)	DO (mg/L)	DOC (mg/L)
Low	Whipple Dam	19	6.19 ± 0.4 ^a	15.79 ± 0.71 ^a	0.02 ± 0.01 ^{ab}	2.29 ± 0.86 ^a	2.71 ± 1.2 ^a
	Shaver's Creek	17	6.49 ± 0.26 ^b	17.69 ± 1.08 ^b	0.02 ± 0.01 ^a	1.46 ± 0.55 ^b	2.33 ± 0.82 ^a
High	Mosquito	12	6.82 ± 0.12 ^c	18.26 ± 0.99 ^b	0.06 ± 0.08 ^{ab}	1.11 ± 0.5 ^c	6.64 ± 3.92 ^b
	Cauldron	11	6.81 ± 0.16 ^c	15.75 ± 0.46 ^a	0.04 ± 0.02 ^b	1.25 ± 0.37 ^{bc}	3.56 ± 1.48 ^b

though with three samples accuracy improved to within 5 % at all sites (Fig. 6.1). Seven or more samples were needed to accurately characterize mean DO and a minimum of ten samples were needed to characterize mean DOC at all sites within 10 % of the population mean (Fig. 6.1). Greater than ten samples were required to accurately characterize mean NO_3^- within 10 % of the population mean, possibly due to the low concentrations of NO_3^- found in this study (Fig. 6.1, Table 6.1). While there was some variability in coefficients of variation between sites (Fig. 6.1), there were no consistent trends indicating that a greater number of samples would be needed to characterize low or high disturbance sites.

The literature search resulted in 34 papers that examined denitrification in natural or restored riparian wetlands. Of these 34 papers, 30 (88 %) examined denitrification in soil, while 5 (15 %) examined denitrification in water (Table 6.2). Acetylene block was the most popular technique, with 17 studies (50 %) using this method, while DEA assay was the second most popular technique with 14 (41 %) studies using this method (Table 6.2). Of the 5 studies that examined denitrification in water, 4 used a variant of the push-pull method developed by Istok et al. (1997). Only 13 of the studies (38 %) took 10 or more replicate samples per site or treatment during the course of the study, while 17 studies (50 %) took 5 or fewer replicate samples (Table 6.2).

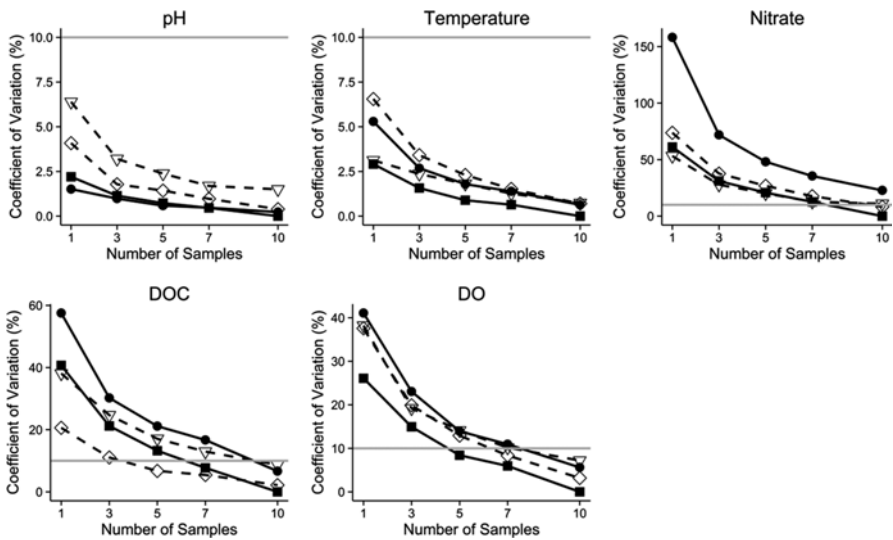


Fig. 6.1 Coefficient of variation by sample size for pH, temperature, NO_3^- , DOC, and DO

Table 6.2 Literature search results

Paper	Method	Sample type	# of wetland sites/treatments	# of replicate samples	Size of study area ^a
Baker and Vervier (2004)	Push pull in situ acetylene block	Water	1	30	200 km ²
Bernal et al. (2007)	Intact core incubation with acetylene inhibition	Soil	1	9	Not specified
Bettez and Groffman (2012)	DEA assay	Soil	6	4	0.5 m transects
Burgin et al. (2010)	Nitrogen-free atmospheric recirculation	Soil	2	2	Samples taken 100 m apart
Casey et al. (2001)	Acetylene block	Soil	1	39	Not specified
Casey et al. (2004)	Acetylene block	Soil	2	8 at CSP site, 20 at DAR site	Study area size not specified. CSP wetland is 6000 m ² , DAR wetland is 60 m wide
Clément et al. (2003)	In situ static core acetylene inhibition	Soil	1 site, 3 zones	3 samples per zone	300 m long study site
Cosandey et al. (2003)	DEA assay and Acetylene inhibition	Soil	1 site, 2 soil types	15 per soil type	2 sampling areas 100 m apart
Davis et al. (2008)	Acetylene block and DEA assay	Soil	1 site, 3 zones	5 acetylene block, 3 DEA	3 zones less than 50 m apart
Dhondt et al. (2004)	DEA assay	Soil	3	10	Each sampling area 10 m wide
Gift et al. (2010)	DEA assay	Soil	6	4	Not specified
Groffman and Crawford (2003)	DEA assay	Soil	8	2–3 per site	Not specified
Hefting et al. (2004)	Intact core incubation with acetylene inhibition	Soil	13	9	Not specified

(continued)

Table 6.2 (continued)

Paper	Method	Sample type	# of wetland sites/treatments	# of replicate samples	Size of study area ^a
Hefting et al. (2003)	Intact core incubation with acetylene inhibition	Soil	2	9	Not specified
Hill and Cardaci (2004)	Acetylene block	Soil	1 site, 3 zones	10 per zone	25 m transects
Hill et al. (2000)	Push pull in situ acetylene block	Water	1	27	2 piezometer nests 400 m apart
Hunter and Faulkner (2001)	DEA assay	Soil	6	12	600 m ² area in each site
Roach and Grimm (2011)	DEA assay	Soil	1 site, 8 transects	5	45 m transects
Jordan et al. (2007)	DEA assay	Soil	9	5	Not specified
Kellogg et al. (2005)	In situ push pull with N ¹⁵	Water	4	3	300 m ² area in each site
Maitre et al. (2005)	DEA assay	Soil	1	18	Not specified
McCarty et al. (2007)	DEA assay, acetylene block	Soil	1	5	Not specified
Orr et al. (2007)	Acetylene block	Soil	1	50	338 ha
Pinay et al. (2007)	Static core acetylene inhibition	Soil	15	9	Not specified
Racchetti et al. (2011)	Isotope pairing	Soil	22	4	Not specified
Rotkin-Ellman et al. (2004)	DEA assay	Soil	4	2–3 per site	Samples collected 1 m apart
Sgouridis et al. (2011)	15NO ₃ additions	Soil	1 site, 4 zones	15 per zone	Not specified
Stander and Ehrenfeld (2009)	Acetylene block	Soil	14	5	37.5 m transects
Sutton-Grier et al. (2009)	DEA assay	Soil	21	3	20 m ² plots in each site

(continued)

Table 6.2 (continued)

Paper	Method	Sample type	# of wetland sites/treatments	# of replicate samples	Size of study area ^a
Sutton-Grier et al. (2010)	DEA assay and denitrification potential via acetylene inhibition	Soil	2	68 at one site, 256 at the other	Not specified
Ullah et al. (2005)	Acetylene block	Soil	2	8 in 2002, 15 in 2003	Not specified
Olde Venterink et al. (2006)	Acetylene inhibition	Soil	2 rivers, 5 wetland communities, 8 replicate plots	3 per replicate plot	Not specified
Watson et al. (2010)	Push-pull method	Water	9	3	Not specified
Woodward et al. (2009)	Acetylene block (soil), gradient tracer tests with acetylene block (water)	Soil and Water	1	1 for water, 4 soil cores	Not specified

^aSize of study area is the areal extent over which sampling occurred, not the size of the wetland studied (unless specified)

6.4 Discussion

Our results highlight the difficulty in characterizing wetland biogeochemistry with a limited number of samples. Denitrification has been known as a highly variable process due to the spatial and temporal heterogeneity of the parameters underpinning denitrification (Hedin et al. 1998; McClain et al. 2003). While our results show that temperature and pH only require small numbers of samples, DO, DOC, and NO_3^- required multiple samples to obtain an accurate site level mean. The number of samples required is directly linked to the variability of the different parameters. Temperature and pH displayed low within-site variability, while DO, DOC, and NO_3^- displayed higher within-site variability (Table 6.1). Unlike the study by Moon (2012), we did not find differences in variability between high and low disturbance sites. Furthermore, no consistent differences were noted in the number of samples required between high and low disturbance sites, indicating that both high and low disturbance sites should be treated similarly with respect to sample size. These findings are reassuring, as differences between sample sizes for high and low disturbance sites would indicate that we may be properly characterizing one type of site, but not the other, which could result in an inappropriate comparisons between sites of high and low disturbance.

Because DOC and NO_3^- are especially important factors that control denitrification (Hedin et al. 1998; Sirivedhin and Gray 2006), studies with fewer samples run

high risks of obtaining inaccurate site means for NO_3^- and DOC especially. This can result in studies that either fail to find or draw inaccurate relationships between these variables and denitrification. Furthermore, low sample sizes also make the creation of accurate denitrification models difficult, as they need to be validated with field and experimental data. Too few samples will likely result in incorrect parameterization of denitrification models, which will lead to inaccurate model predictions. According to our literature review, only 38 % of the 34 studies we found in riparian wetlands took ten or more samples per treatment or study site (Table 6.2). While this is understandable given the resource intensive nature of measuring denitrification, the low number of samples taken in the majority of studies on denitrification limits the ability of these studies to draw accurate conclusions about the nature of this process.

A potential solution to the issue of low sample numbers would be to use spatially integrative techniques, such as “push-pull” methods, to assess denitrification (Istok et al. 1997; Addy et al. 2002). With the push-pull method, $^{15}\text{NO}_3^-$ and a conservative tracer is added to a specified amount of groundwater extracted from the study site, which is then re-pumped back into the site and allowed to incubate before being extracted. Samples are then analyzed for $^{15}\text{N}_2$ and $^{15}\text{N}_2\text{O}$ to determine potential denitrification rates. Because a large amount of groundwater is extracted in order to obtain samples, this method results in an average rate over a larger extent of space compared with extracting small soil cores. While only two of the five studies that examined denitrification within wetland groundwater took more than five samples, the vast majority of these studies used a variant of the push-pull method, meaning that they covered a greater spatial area and likely required fewer samples in general. However, this method does have its limitations. Materials are costly and the method is time consuming to perform. Wetland systems with flashier hydrology may not provide adequate groundwater supplies during certain seasons and riparian systems with quick water flows may result in poor retention of $^{15}\text{NO}_3^-$. Furthermore, the spatial integration of a larger area by the push-pull technique makes it less useful for studies that wish to examine hot spots or hot moments of denitrification. Yet, push-pull may be useful for studies that wish to obtain a site average for denitrification.

6.5 Conclusions

The purpose of our study was to examine the number of samples required to accurately assess the parameters underpinning denitrification in shallow groundwater in both high and low disturbance sites to provide researchers with guidelines as to the minimum number of samples required to provide accurate assessments. We found that only one sample was needed to accurately characterize pH and Temperature, seven or more were needed to characterize DO, ten or more were needed to characterize DOC, and greater than ten samples were required to characterize NO_3^- within a 400 m² area. Due to the greater numbers of samples needed to characterize DOC and NO_3^- , we recommend that researchers either take a minimum of 10 samples per

400 m² area or use a spatially integrative method to appropriately characterize these parameters for denitrification studies.

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Chapter 7

Indirect and Direct Thermodynamic Effects of Wetland Ecosystems on Climate

Jan Pokorný, Petra Hesslerová, Hanna Huryna, and David Harper

Abstract This chapter deals with a largely unrecognised service of wetlands – their role in regulating air temperature through evapotranspiration. We explain quantitatively how solar energy striking the earth’s surface is dissipated by water (expressed in energy units (W m^{-2})) in three processes: dissolution-precipitation of salts, disintegration-recombination of the water molecule in biological processes and evapotranspiration-condensation. The direct effect of wetlands on regional climate, through reduction of temperature gradients and the role of water vapour and clouds in lowering the passage of solar radiation are then described. We quantify the huge upsurge of sensible heat (warm air) that must have occurred after the drainage of wetlands in the northern hemisphere over the past 260 years. The radiative forcing that was caused by the increase in greenhouse gases in the atmosphere over the same period (from 1 to 3 W m^{-2} from 1750 to the present day) is markedly lower than radiative forcing caused by wetland drainage and indeed, is too small to measure. The amounts of carbon dioxide, methane and water vapour in atmosphere and their dynamics are compared. We question the meaning of ‘average temperature’ as the criterion of climate change in terms of thermodynamics. We show temperature differences in the present-day cultural landscape, on a clear sunny day, in thermovision pictures: wetlands and forests are upto 20 °C cooler than drained surfaces. We argue that persisting with the dogma of climate change caused by the greenhouse effect alone results in society ignoring the most important functions of natural vegetation, manifest through their direct effect on climate and water cycling. This facilitates further wetland drainage and deforestation. We believe that it is now essential to support and restore natural vegetation structures, like wetlands and forests, in order to make any serious reduction in climate warming.

Keywords Solar radiation • Evapotranspiration • Sensible heat • Water cycle • Radiative forcing • Surface temperature

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7.1 Introduction

Wetlands are ecosystems driven by different dominant plant species, dependent upon the quantity and duration of water within them. They have high primary production which supplies energy for diverse food webs and organic matter for soil formation. (Cooper 1975; Dykyjová and Květ 1978; Květ et al. 1998) Wetlands are also well-known as important units in the landscape for their role in regulating the hydrological cycle. They retain flood water and moderate floods in lower part of catchments. Wetland plants in floodplains, the littoral zone of lakes, swamps and marshes tolerate flooding and recycle nutrients from flood water. Their role in flood prevention is now beginning to be recognized and taken into account in legislation in many countries.

Wetlands however, have been removed from the natural landscape over most of the earth's surface as a result of drainage to promote urban and agricultural developments. For example, more than 51 % (45.9×10^6 ha) of the total area of wetland in the USA has been replaced by cropland since European settlement (Mitsch and Hernandez 2013). This removal has resulted in considerable losses of biodiversity and several other services not formerly recognised as beneficial to humankind, but now increasingly being quantified (Gopal et al. 2000, 2001). The important ones are the direct provisioning services, such as food (e.g. fish) and building materials (e.g. reeds for roofing); recreational services (e.g. tourism) and supporting services (e.g. flood control, climate through carbon sequestration) which wetland recreation and restoration now seek to provide (Mitsch and Gosselink 2007).

A largely unrecognised service, but potentially the most important of all those known, which is provided by wetlands, is their role in regulating air temperature – hence climate change – through evapotranspiration. This chapter explains and quantifies the role of wetlands as climate regulators.

7.2 Solar Energy Striking the Earth's Surface

The Sun's energy drives our water cycle, plant production and all other living processes in the biosphere; it warms the Earth to an average temperature of around 15 °C or 288 °K. The actual direct solar irradiance at the top of the Earth's atmosphere fluctuates over a year from 1412 W m⁻² in early January to 1321 W m⁻² in early July due to the Earth's varying distance from the Sun. (Geiger et al. 2003; Kopp et al. 2005). The **Solar Constant**, which is the total radiation energy received from the Sun per unit of time per unit of area on a theoretical surface perpendicular to the Sun's rays and at Earth's mean distance from the Sun (Anon 2016) is approximately 1.366 kW m⁻². It is most accurately measured from satellites where atmospheric effects are absent. It is not exactly constant, increasing by 0.2 % at the peak of each 11-year solar cycle. Sunspots block out light and reduce the emission by a few tenths of a percent, but bright spots, called plages, that are associated with solar

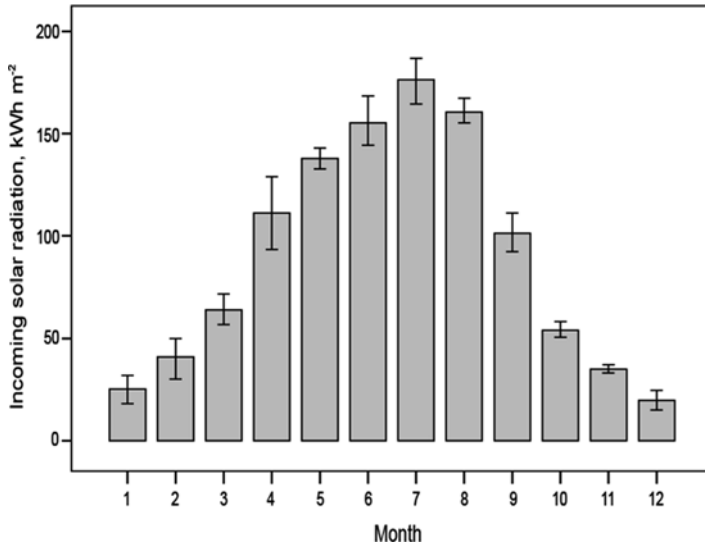


Fig. 7.1 Monthly values of incoming solar radiation data recorded for the whole year of 2009. Mean values measured at the wet meadow and the dry land site in Třeboň region

activity, are more extensive and longer lived, so their brightness compensates for the darkness of the sunspots. Total solar output has varied over the last three 11-year **sunspot** cycles by approximately 0.1 % (Willson and Hudson 1991).

The difference between the amounts of incoming radiation on a clear day ($29.3 \text{ MJ m}^{-2} = 8.14 \text{ kWh m}^{-2}$, maximum flux 1000 W m^{-2}) is about an order of magnitude greater than on an overcast day (0.78 kWh m^{-2} , maximum flux 100 W m^{-2}) (Figs. 7.1 and 7.2). The best instruments to measure solar irradiance (e.g. Kipp Zonen, Eppley or Middleton pyranometers and pyrheliometers) have an accuracy about 1 % – i.e. of 10 W m^{-2} (WMO 2008). *The total irradiance fluctuation caused by changes of the Sun's activity varies by a negligible amount (0.1 %) with the 11-year sunspot cycle* (e.g. Cahalan et al. 2010; Priest 2014); thus the stability of irradiance is considered to be greater than the accuracy of the measurements we can currently make.

7.3 Direct Effect of Wetlands on Climate via Evapotranspiration and Other Life Processes

There is a large difference between the distribution of solar radiation in functioning natural ecosystems of high plant biomass well supplied with water compared to dry, biomass-poor ecosystems with far more non-living physical surfaces. This is because of the impacts of solar energy upon water molecules.

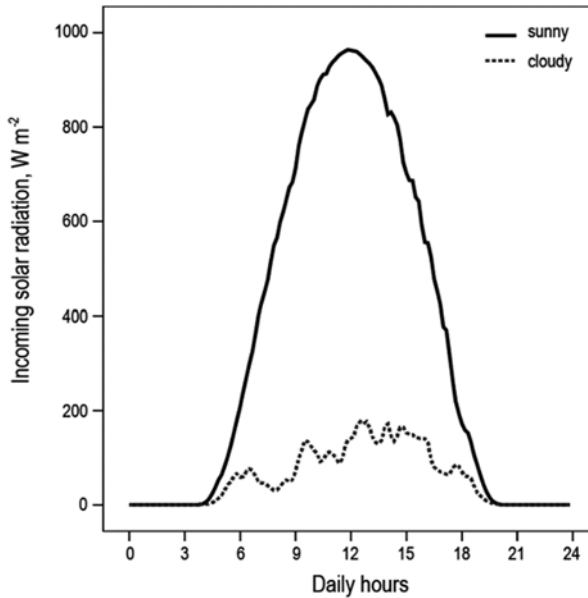


Fig. 7.2 Daily mean series of incoming solar radiation (W m^{-2}) on five sunny (3.06, 13.06, 14.06, 16.07, 27.07) and five cloudy (19.06, 23.06, 24.06, 18.07, 28.07) days in 2009 at the wet meadow and the dry land site in Třeboň region

The incoming solar radiation is dissipated at the surface of the earth by three main processes – dissolution-precipitation of salts, disintegration-recombination of the water molecule in biological processes and evapotranspiration-condensation. Willy Ripl proposed a conceptual model to help understand these reactions, called the ETR (Energy – Transport – Reaction) Model (Ripl 1995, 2003). All three processes are driven by the gradient of solar energy; they slowdown in winter when the supply of solar energy is low, then accelerate in summer (Fig. 7.3). Similarly they fluctuate between day and night.

7.3.1 Dissolution-Precipitation of Salts

Energy transformations associated with processes of dissolution and precipitation of salts are less often considered in studies of energy fluxes in ecosystems than the other two and their role has not been fully evaluated. The free energy of formation of a pure substance – taken to be the free energy change when 1 mol of the substance is formed from its elements at 1 atm pressure – is negative for most compounds at 25 °C and 1 atm pressure. This implies that formation of a compound from its elements under these conditions is ordinarily a spontaneous process. Conversely, most compounds are stable with respect to decomposition into their

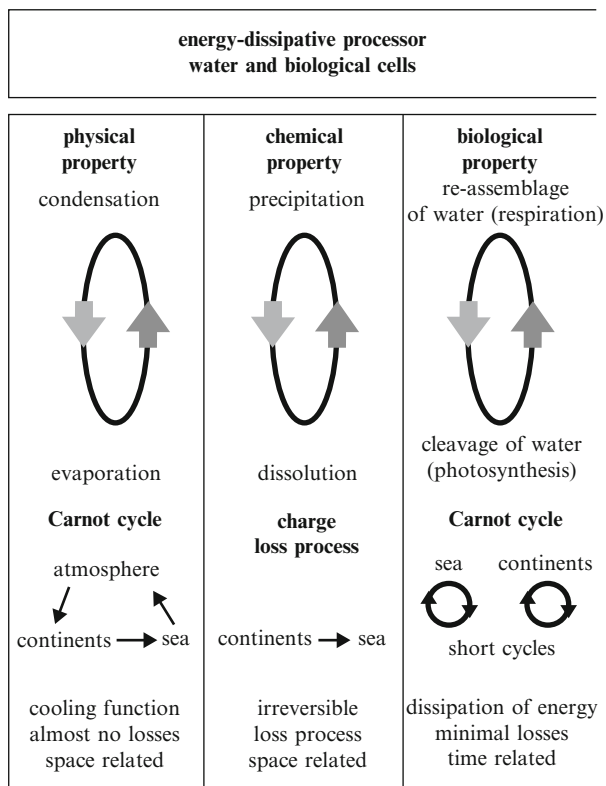


Fig. 7.3 W. Ripl's scheme of three dissipative properties of water (Ripl & Hildmann 2000)

elements. For example formation of solid CaCO_3 and CaSO_4 are linked with release of -1206.9 kJ (-335.25 Wh) mole^{-1} and -1431.1 kJ (-397.52 Wh) mole^{-1} . There are exceptions and two of the more interesting are NO ($+90.25 \text{ kJ mole}^{-1}$) and NO_2 ($+33.18 \text{ kJ mole}^{-1}$). In principle these compounds should decompose to N_2 and O_2 under ordinary conditions. The fact that they remain long enough to be major problems in air pollution implies that the rate of decomposition must be extremely slow. The standard enthalpies ('heat content') of formation of selected inorganic compounds are given in the Table 7.1.

7.3.2 Disintegration-Recombination of Water Molecules

The disintegration-recombination of water molecules are the principal processes of photosynthesis and respiration. When 1 mol of hydrogen reacts with $\frac{1}{2}$ mole of oxygen then 1 mol water is formed and energy of 286 kJ (79 Wh) is released. One mole of water (18 g) thus has enthalpy of -286 kJ (-79 Wh). The maximum net

Table 7.1 Standard enthalpies of formation of inorganic compounds at 298 °K

Compounds	Enthalpy	
	kJ mol ⁻¹	Wh mol ⁻¹
H ₂ O (l)	-285.83	-79.39
H ₂ O (g)	-241.82	-67.17
NH ₃ (g)	-46.11	-12.81
NO ₂ (g)	+33.18	9.22
CaCO ₃ (s)	-1206.9	-335.25
CO ₂ (g)	-393.51	-109.31
NO (g)	+90.25	2.075
CaSO ₄ (s)	-1431.1	-397.2

According to Atkins and de Paula [2010](#)

photosynthesis may take up as much as 20 W m⁻², with an average rate of about 2 W m⁻² in closed-canopy ecosystems (Cooper [1975](#); Pokorný et al. [2010a](#)). One kg of plant biomass, consisting mostly from cellulose, contains roughly 5 kWh energy. Primary production in the temperate zone is 0.1–1 kg dry mass from 1 m² per vegetation season (Patten [1990](#); Květ et al. [1998](#); Pokorný et al. [2010b](#)). Only exceptionally can the long term production of plant biomass be higher (Hejný et al. [1981](#)). The maximum annual primary productivity (about 1 kg m⁻² year⁻¹) compared with incoming annual income of solar energy in the temperate zone (about 1100 W m⁻²) makes it clear that the efficiency of conversion of solar energy into plant biomass is less than 0.5%.

Decomposition of organic matter (respiration), in contrast to biomass production, results in release of the stored energy. Decomposition release is accelerated by drainage of wetlands. Its rate in drained wetland soil can be several times higher than primary production.

7.3.3 *Evapotranspiration-Condensation*

Phase changes between liquid water and water vapour are linked with the consumption of a large amount of energy. The enthalpy of liquid water is -2.5 kJ g⁻¹. Evapotranspiration (ET) or latent heat flux, represents large, invisible fluxes of water and energy in the landscape: the scale of several hundred W m⁻². An ET of 250 W m⁻² produces 100 mg H₂O m⁻² s⁻¹, equivalent to evaporation of 100 L s⁻¹ from 1 km², an order of magnitude more than the surface water outflow from 1 km² of land. This is thus an energy gradient reduction (“air-conditioning”) of 250 MW km⁻².

ET is a powerful cooling process, having a double air-conditioning (gradient reducing) effect upon the landscape – (a) evaporation **cools** places, consuming solar energy for transfer of liquid water into water vapour (b) subsequent condensation of water vapour **warms** air where it occurs, releasing latent heat when the dew point is achieved on cool surfaces. ET can thus be considered as a perfect process of

gradient reduction (equalisation of temperature), linked to the growth of plants (primary production), and to nutrient uptake and water recycling (Ripl 1995). Warm air can absorb and contain high amounts of water vapour and transport it to high levels of the atmosphere where, after mixing with cooler results in air rapid condensation and release of latent heat.

These processes of evapotranspiration-condensation, dissolution-precipitation of salts and disintegration-recombination of the water molecule slow down when there is water shortage; solar energy is consequently converted into sensible heat. The sensible heat flux (H) represents the sum of all the heat exchanges between the surface of a landscape and its surrounding atmosphere by conduction or convection. On dry surfaces, H may reach values of several hundred $W m^{-2}$. The H of a heated surface thus warms air, which rises up in a turbulent motion creating atmospheric instability. Drainage of wetlands and deforestation bring about a large shift from latent heat flux (air-conditioning, temperature gradient equalizing via evapotranspiration) into sensible heat flux (increase of local temperature and turbulent motion of air, strong wind, cyclones).

7.3.4 Ground Heat Flux and Warming of Biomass

The ground heat flux (G) is that transferred from the surface downward via conduction. G slows down in dry soil as well as in dry plant litter, is also low in dense vegetation cover. In summer months, G is typically 10% of net radiation and ranges from $10 W m^{-2}$ to $100 W m^{-2}$ for growing crops (Jones 1991; Kustas et al. 1993; Huryňa et al. 2014). The physical sink of energy depends on the amount of living biomass and its water content (Fig. 7.4). The maximum heat-flux warming of biomass is approximately $20 W m^{-2}$.

7.3.5 Ratio Between the Amount of Energy Bound in Biomass and That Dissipated by Evapotranspiration

The ratio between the amount of energy needed for primary production and cooling (air-conditioning) is very low: 5 kWh stored in 1 kg of dry biomass is equal to the latent heat of evaporation of 7.4 L of water. A stand of wetland plants evaporates about 500 L of water in a year, which is 350 kWh; this is 70 times more than amount of energy bound in the biomass. Plants have evolved to invest a very low portion of incoming solar energy in their biomass, but a high portion into the cooling process of transpiration. Photovoltaic panels have to produce electricity for 1–2 years to cover the amount of energy needed in their production. Wetland plants cover amount of energy needed for their production (bound in biomass) in just 2 days of evapotranspiration. Wetland plants are thus an effective and perfect air-conditioning system (Table 7.2)!

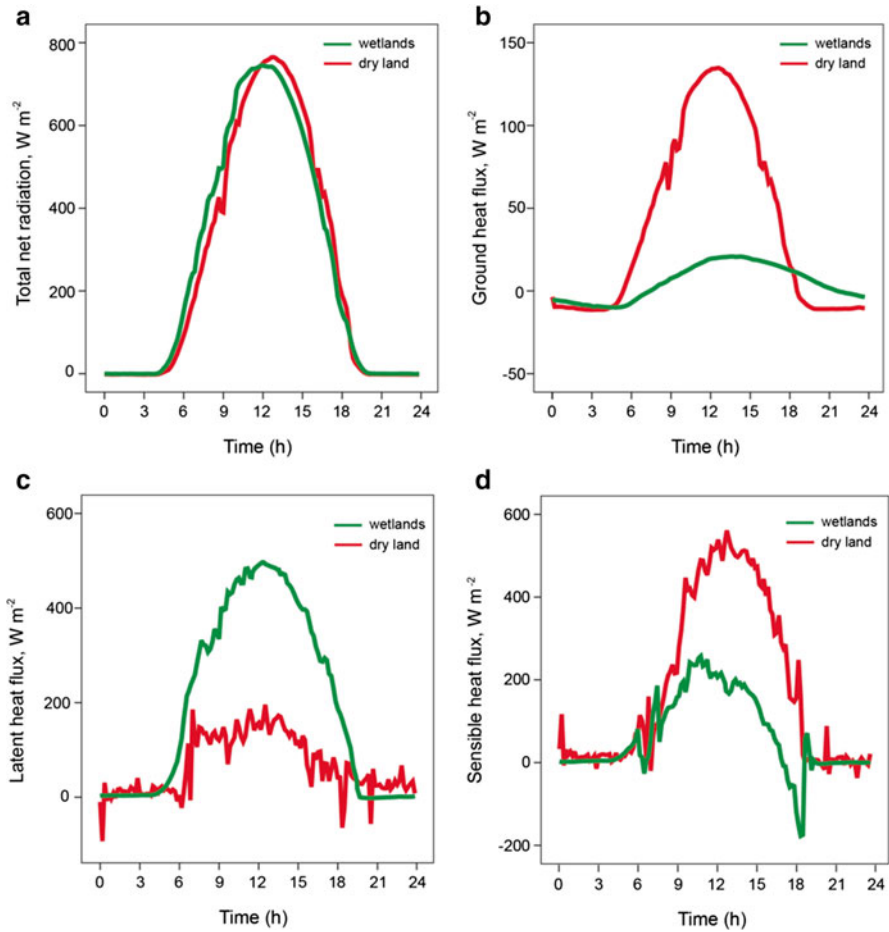


Fig. 7.4 Daily mean series of total net radiation (Wm^{-2}) (a), ground heat flux (Wm^{-2}) (b), latent heat flux (Wm^{-2}) (c) and sensible heat flux (Wm^{-2}) (d) on five clear days (13.06, 14.06, 16.07, 27.07, 01.08) for 2009 period at the wet meadow and the dry land site in Třeboň region

Table 7.2 The distribution of incoming solar radiation in two types of ecosystems

	Wetlands		Dry lands	
	$W m^{-2}$	%	$W m^{-2}$	%
Reflectance	155	18	235	28
Evapotranspiration	452	54	65	8
Sensible heat flux	173	20	400	47
Ground heat flux	50	6	150	17
Biomass	20	2	–	–

According to Pokorný et al. (2010b), Huryňa et al. (2014)

7.4 Wetland Losses and Consequent Impact on Climate

The drainage of wetlands causes a shift from latent heat to sensible heat flux, which results in an increase of temperature and thus water loss from the landscape. The distribution of latent and sensible heat between a wetland and dry (drained) land differ by several hundred Wm^{-2} during a sunny day. In wetland ecosystems, latent heat prevails and uses about 60–80 % of net radiation, sensible heat uses about 20–30 % while ground heat flux 10–20 % (Kedziora 2011) The decrease in ET from 1 km^2 (100 ha) as a result of drainage is approximately 250 Wm^{-2} (equivalent to 100 mg s^{-1} of water vapour) and this 250 MW of solar energy in 1 km^2 is thus released into atmosphere as warm air (sensible heat). In August 2015 in the Czech Republic, the total surface area of harvested wheat and rape seed fields was about 18,000 km^2 (1,800,000 ha) and sensible heat released from this dry surface area was thus at least 4,500,000 MW. Human generation of such heat in electricity production would require 4500 Nuclear Power Stations, each of 1000 MW. The real quantity of sensible heat is even higher because 1000 Wm^{-2} of incoming solar energy is partly reflected (200 W m^{-2}), partly heats the ground (up to 100 W m^{-2}), is partly used for ET (maximum 200 W m^{-2} from and almost dry surface), so 500 W m^{-2} is realised as sensible heat warming the atmosphere. High air pressure is developed as a consequence, which prevents the income of wet air from the Atlantic, in this example of Central Europe.

The IPCC claims and works with the premise that man does not substantially affect emissions of water vapour. Yet there is a huge difference between the amount of water vapour and dynamics of the phase (states) changes of water above a wetland (defined as vegetation well supplied with water) or a forest, and an agricultural field or a sealed surface which were created by draining the wetland; the wetland has lower temperature and air above, higher relative humidity and a tendency to create fog and clouds. Satellite pictures of large wetlands areas demonstrate this. Similarly clouds are common above rain forests (Earth Science Data, NASA Land Processes Distributed Active Archive Center). The high content of water vapour in the air also reduces the passage of solar radiation to the Earth's surface and hence effectively reduces surface temperature. **This is the opposite conclusion than would be reached based on an interpretation of the greenhouse effect: conventional thinking says the higher greenhouse effect results in a higher temperature.** Wetlands reduce temperature however, by the cooling effect of ET and the shading by fog and clouds formed from water vapour. The water vapour does not rise quickly into the atmosphere, because there are no hot surfaces on wetlands. The water vapour condenses by night and prevents infra red (IR) radiation moving from the Earth's surface towards the sky. In this way, wetlands moderate extreme day and night temperature, as do forests. For example substantial clearance of the Mau Forest in Kenya at an altitude 2800 m, resulted in early morning frosts which hindered crop production (Hesslerová & Pokorný 2010).

Water vapour rises faster from crop plants than from wetlands and forest, which have dense vegetation and therefore lower temperatures at the ground. The conversion of natural vegetation to agricultural fields changes the land surface characteristics, which lead to redistribution of surface energy components (Esau and Lyons 2002). More than 51 % (45.9×10^6 ha) of the total area of wetland had been replaced by cropland in USA by European settlement (Mitsch and Hernandez 2013). About 400 W m^{-2} is moved from latent to sensible heat flux for days with the highest solar irradiance (Huryňa et al. 2014), so it can be calculated that more than 175,000 GW of energy was converted into sensible heat over the USA these past 260 years, which has strongly affected dynamic processes in the atmosphere. Warming of Northern Hemisphere has been faster than that of Southern Hemisphere (Climatic Research Unit), because the Northern Hemisphere has a substantially higher portion of continents than the Southern Hemisphere and almost 90 % of the world population lives there. Changes of land cover, particularly deforestation and drainage, have affected the Northern Hemisphere much more. Sensible heat (hot air rising into the atmosphere) would contribute to the melting of mountain and arctic glaciers. The amount of sensible heat released as a consequence of land cover changes is very much higher than heat caused by increased carbon dioxide concentrations creating the greenhouse effect.

7.5 Indirect Effect of Wetlands on Climate *via* Greenhouse Gases (GHG); Sink or Source?

The indirect effect of wetlands on climate as either a source, or a sink of GHGs such as CO_2 and CH_4 , has been studied intensely. GHGs act on global climate through Radiative Forcing (RF), which is the change in net radiative flux expressed in W m^{-2} at the top of the atmosphere (Fig. 7.5). The Intergovernmental Panel on Climate Change (IPCC 2007) documents the RF caused by an increase in GHG in the atmosphere from 1750 to the present day as between 1 and 3 W m^{-2} . In the next 10 years, the RF is expected to increase by 0.2 W m^{-2} . Changes of radiative forcing over time are too small to be measured, so they are calculated. The effect of GHGs on changes of RF cannot thus be tested by the scientific method. The IPCC focuses on global average temperature (GAT) and warns of global warming caused by increasing concentration of GHGs. Water vapour is considered only as a “feedback agent”, rather than an agent directly forcing climate change (IPCC 2013). Myhre et al. (2013) state (p. 666) “*As the largest contributor to the natural greenhouse effect, water vapour plays an essential role in the Earth’s climate. However, the amount of water vapour in the atmosphere is controlled mostly by air temperature, rather than by emissions. For that reason, scientists consider it a feedback agent, rather than a forcing to climate change. Anthropogenic emissions of water vapour through irrigation or power plant cooling have a negligible impact on the global climate*”.

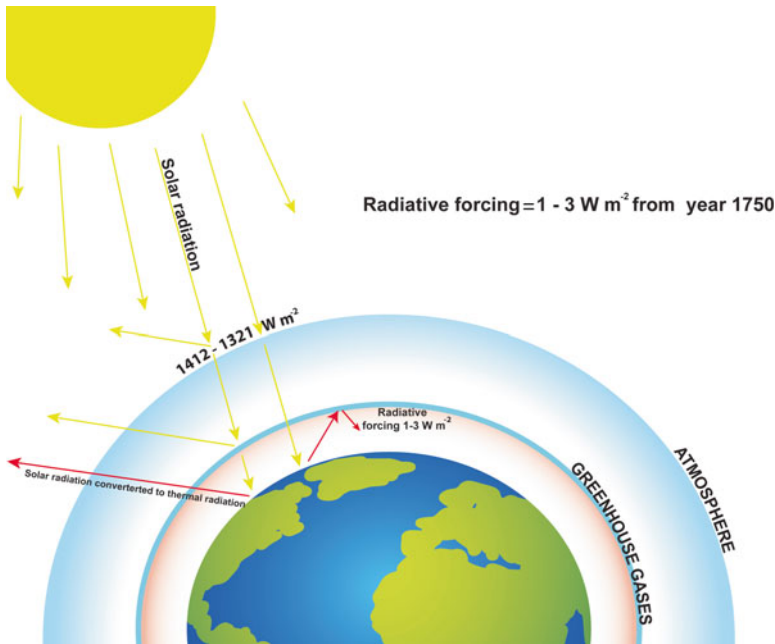


Fig. 7.5 Scheme of greenhouse effect and radiative forcing

This is their principal premise, which thus excludes water and land cover as controlling factors of climate, assigning CO_2 and CH_4 alone as the controlling agents of global climate. Yet the amount of water vapour in air is one to two orders of magnitudes greater than that of CO_2 the concentration of CO_2 and CH_4 in the atmosphere is 390 ppm and 1.8 ppm, respectively. Concentration of CO_2 fluctuates during a year within 20 ppm being lower in period of summer in Northern Hemisphere (IPCC 2013), which indicates important role of terrestrial vegetation. Water vapour in air expressed in mass units (grams m^{-3}) and in volume units (ppm calculated according to Avogadro Law i.e. 1 mol has a volume 22.4 L) are shown in Fig. 7.6.

As an example, air saturated with water at 21 °C contains 22,400 ppm of water vapour; air saturated with water at 40 °C contains 62,200 ppm. The concentration of water vapour in air changes rapidly both in time and space. Water exists in three phases/states – liquid, solid, gaseous – within the normal range of temperatures on our planet and transition between states is linked with release and consumption of energy. Water vapour forms clouds which prevent passage of solar energy to Earth surface and reduce temperature.

How does land cover change – especially drainage of wetlands – affect the amount of water vapour in air? Is the amount of water vapour in the atmosphere really controlled mostly by the concentration of GHGs? Does water vapour affect climate only as a greenhouse gas? These are urgent questions for wetland scientists to answer as a community.

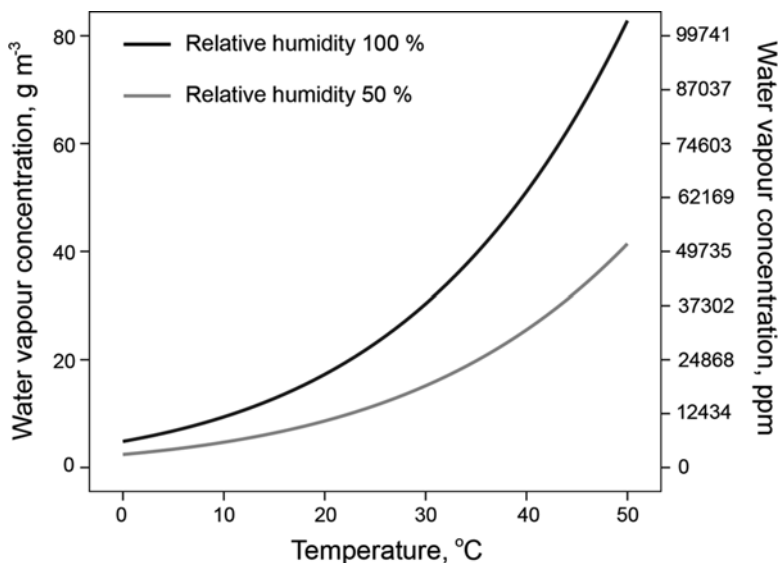


Fig. 7.6 Water vapour concentration in air at 100 % and 50 % saturation at different temperatures expressed in g m^{-3} and ppm volume units

7.6 Meaning of Average Temperature in Thermodynamics and the Role of Gradients

The recommendations of the IPCC are designed for the reduction of GHGs in the atmosphere; climate change is expressed as a change of global average temperature (GAT). The UN Conference on Climate Change 2015, in Paris (COP21/CMP11) set a goal for limiting global warming to less than 2°C GAT compared to pre-industrial levels. The goal should be achieved by reduction of GHGs. The agreement calls for zero net anthropogenic GHG emissions to be reached during the second half of the twenty-first century. According to the IPCC, the quantifiable criterion of climate change is GAT and the reason for global warming is increasing GHGs, particularly CO_2 .

In terms of thermodynamics, average temperature alone does not produce power. The founder of thermodynamics, Carnot, pointed out in the early 1800s that it was not simply the temperature of steam produced in a boiler that made the pistons pump hard and fast in an engine, but rather the difference between the temperature of the hot boiler and cooler radiator. The difference across distance – the gradient – is what makes the conditions for the flow to take place. Similarly, in the atmosphere and the landscape, it is gradients of temperature, heat, air pressure, which drive wind and transport water vapour.

Real tornados, thunderstorms and hurricanes are macroscale dissipative processes driven by gradients. The relationship pressure-volume-temperature is crucial. The gradient that drives a hurricane is between the warm ocean, at least 27°C

and much cooler temperatures higher in the atmosphere. This temperature gradient between a warm ocean and cool air creates an updraft of warm air (Schneider and Sagan 2005).

In 1886, Boltzmann suggested that the energy gradient imposed on the Earth by the sun drives living processes *“Life struggles for entropy which becomes available through transition of energy from the hot sun to the cold Earth. In order to exploit this transition, plants spread their immense surface of leaves and utilise the sun’s energy before it falls to the Earth’s temperature”* (Schneider and Sagan 2005).

Plants are the most advanced instrument yet evolved for degrading incoming solar radiation. Nature is said to “abhor a gradient” – a corollary to this is that when a gradient is imposed on a system it can develop processes and structures that will hold material and energy from going to equilibrium immediately, while degrading the imposed gradient as thoroughly as possible.

A tree is like a giant water fountain ‘spewing’ water in the form of latent heat. It is like a candle burning high-exergy waxes and degrading that high-exergy fuel into low-grade heat. Thus a tree is best understood as a giant degrader of energy. Each new leaf, each new phototrophic rearrangement, is a new opportunity for energy degradation.

Prigogine popularized the term dissipative structures introduced by Lotka (1922). Dissipative systems are non-equilibrium, open, dynamic systems with gradients across them. They degrade energy and exhibit material and energy cycling. Dissipative structures grow more complex by exporting – dissipating – entropy into their surrounding (Schneider and Sagan 2005).

The IPCC, and consequently the conclusions of the United Nations Conference on Climate Change 2015 in Paris (COP21/CMP11), regard increasing concentration of GHGs as the main cause of climate change, which is indicated and quantified by increasing GAT. Can Radiative Forcing, which has a value less than one thousandth of solar energy income ($1\text{--}3\text{ W m}^{-2}$ from 1750, or 0.2 W m^{-2} during a decade) really be transformed into such dramatic events which need a great deal of energy, as winds, torrential rains, floods caused by gradients of temperature and air pressure? No, there are other energy fluxes on the interface of the Earth’s surface and atmosphere. These are the fluxes of solar energy in the landscape, the fluxes of energy in ecosystems mediated by living organisms. Life processes directly transform the incoming solar energy through the well-known processes described above.

7.7 Exchange of Water and CO₂ in Plant Stands

The cooling process of transpiration is often considered a side effect rather than a mechanism to control leaf temperature and that of surroundings. It is often called ‘evapotranspiration losses’ or ‘a tax paid for the opening of stomata’. Yet, the quantity of water molecules exchanged by plants is at least two orders of magnitude higher than the quantity of carbon dioxide fixed in biomass (Table 7.3).

Table 7.3 Comparison of concentrations of CO₂, CH₄, H₂O and their turnover rates in air

Greenhouse gas	CO ₂	CH ₄	H ₂ O
Concentration ppm in atmosphere	380	1.5	1000–40,000 rapidly changing
Phase	Gas	Gas	Solid, liquid, gaseous ex/endo thermic; 18 ml liquid equivalent to 22,400 ml gas
Turnover rate	Years	Years	Days
Climate policy respect	Emission trading, incentives	Emission trading, incentives	Ignored

7.8 Surface Temperature Distribution in a Cultural Landscape with Wetlands – An Example

Spatial resolution of aerial thermal image (2 m) allows a more detailed analysis of the temperature distribution in the landscape. The image (0.95×2.8 km) was acquired on 29 July 2008 at 1:00 p.m. and shows Třeboň town and its surroundings – wet meadows area (Fig. 7.7). The temperature values for ten categories of land cover were calculated along the transects, displayed within the image (Table 7.4, Fig. 7.8).

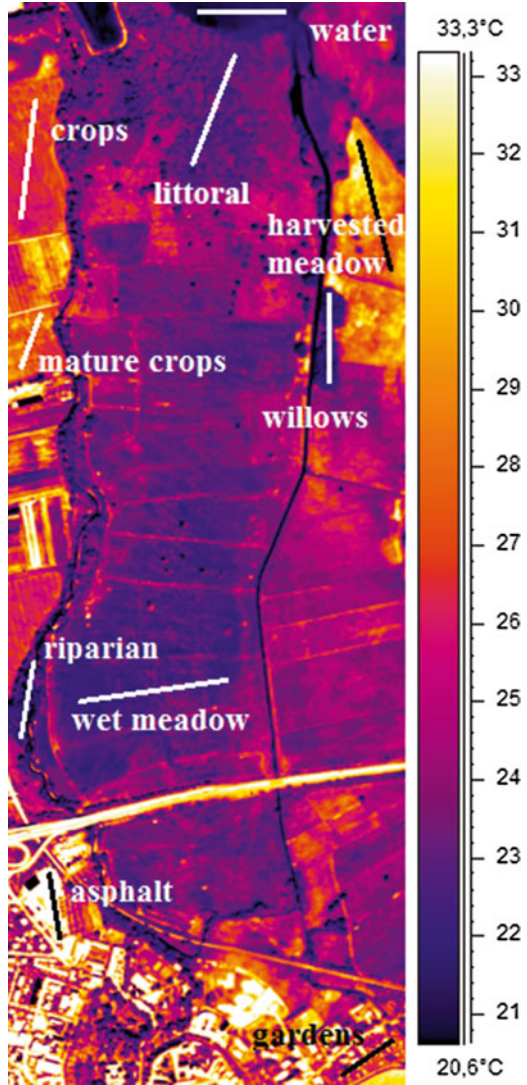
The lowest values of the average surface temperature are found in water, willows and riparian vegetation, slightly less than 22 °C. Wet meadow and littoral, with temperatures around 22.5 °C follow; they are surfaces with wetland vegetation. Categories of agricultural crops and mixed surface–gardens have an average surface temperature of about 26.5 °C. The warmest surfaces are mature crops and harvested meadow (28.3 and 29.3 °C). In this case, the surfaces are covered by dry vegetation. The hottest surface is asphalt with 34.1 °C.

The highest temperature differences (18 °C) are between physical surfaces – water and asphalt. When comparing the coldest and warmest vegetation surface, which is willow and harvested meadow, this difference is close to 10 °C. The difference between the asphalt and willow growth is 17 °C. Temperature differences between structured vegetation well supplied with water and vegetation, with water deficit are therefore evident. Through the vegetation cover one can affect the surface temperature of the landscape, which reflects the way of solar energy distribution and consequently energy flows in the landscape.

7.9 Conclusions

Global average temperature (GAT) as a criterion and an indicator of global change is misleading. Gradients drive climate change. GAT does not pick up gradient increase and is unable to reduce gradients. For example: if a large wetland having

Fig. 7.7 Thermovision picture of culture landscape with wetlands showing direct effect of water and vegetation on temperature and local climate



diurnal temperature fluctuation from minimum 18 °C, to maximum 22 °C is drained and changed into an agro-industrial area having temperature fluctuation minimum 10 °C, maximum 30 °C, the average temperature has not changed, but the distribution of solar energy and water fluxes i.e. the local climate changed substantially.

The Greenhouse Effect acts, according to the IPCC, via Radiative Forcing caused by the rise of CO₂ and CH₄, which have increased since 1750 from 1 to 3 W m⁻² and will increase during the next decade by a further 0.2 W m⁻². Changes of Radiative Forcing are so small in comparison with incoming solar radiation, that they cannot even be monitored. In order to affect global climate, Radiative Forcing and the

Table 7.4 Surface temperature values of different land cover types in wet meadow area near Třeboň, South Bohemia, Czech Republic calculated from the transects in thermal image as showed in Fig. 7.7

Surface temperature (°C)					
	Min	Max	Max – min	Average	Standard deviation
Harvested meadow	26.2	31.9	5.6	29.3	1.7
Litoral	21.5	23.5	2.1	22.8	0.4
Water	20.8	21.5	0.7	21.2	0.2
Willows	20.8	22.2	1.4	21.5	0.3
Wet meadow	21.6	23.7	2.1	22.6	0.4
Crops	25.5	27.9	2.4	26.6	0.5
Riparian vegetation	18.9	23.9	5.0	21.7	0.9
Mature crops	26.6	30.7	4.1	28.3	1.0
Gardens	24.0	33.2	9.2	26.8	1.8
Asphalt	33.1	39.3	6.2	34.1	0.9

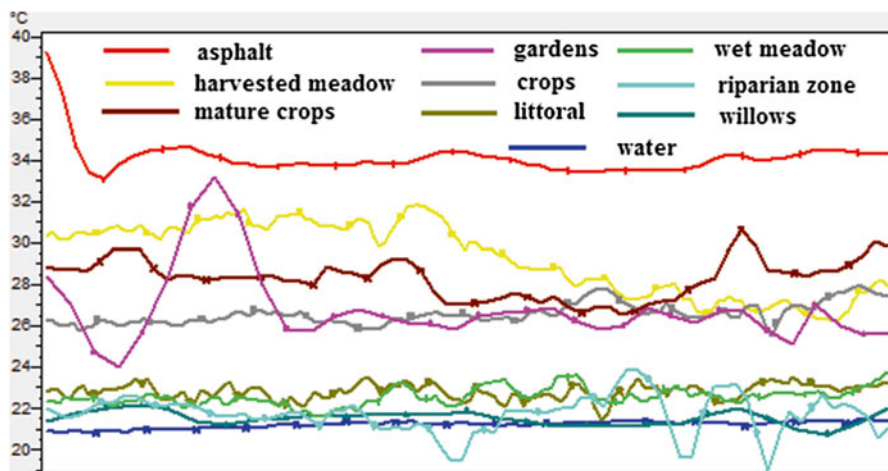


Fig. 7.8 Surface temperature transects of ten different surfaces as displayed in Fig. 7.7

consequent increase in GAT must be in some way be transferred into temperature gradients. Reporting only GAT ignores the main reasons for climate change i.e. the gradients which may drive torrential rains, cyclones etc. GAT does not change significantly and it results in the false conclusion by climate sceptics that there is no climate change and therefore no measures are needed. The present approach to climate change which reduces the role of plants to a sink or a source of GHG and albedo is an approach of astrophysics which largely ignores life processes, which moderate extremes of temperatures. Plants are presented in text books as homoiothermic (do not regulate temperature), the reality is the opposite: plants are well supplied with water, they cool and create climate and conditions for other organisms with an efficiency of several hundred $W m^{-2}$.

Very high gradients are caused by changes of land cover, namely by drainage of wetlands and deforestation. These have been considerable since 1750 in North America and Europe in particular and continue on large scales in Asia and Africa since the middle of the twentieth century. Drainage of 1 km² is associated with a daily shift from latent heat of ET to sensible heat of hundreds of MW. In the Czech Republic 10,000 km² of agriculture fields with small flood plains and wet meadows were drained over the past 100 years, which has caused a decline of ET and increase of sensible heat in the order of millions of MW. The effect of drainage of the world's wetlands should be evaluated in this way. Satellite pictures from the 1980s provide exact information on changes of land cover and surface temperature.

Persisting with the dogma of greenhouse effect alone results in us ignoring the most important functions of wetlands through their direct effect on climate and water cycling and hence enables further drainage and deforestation. We have to support life structures like wetlands to improve and retain our climate.

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Chapter 8

Application of Vivianite Nanoparticle Technology for Management of Heavy Metal Contamination in Wetland and Linked Mining Systems in Mongolia

Herbert John Bavor and Batdelger Shinen

Abstract A demonstration of the feasibility of application of iron phosphate (vivianite) nanoparticle techniques, coupled with phytoremediation, for remediation of heavy metal contaminated soil and water resources is proposed for urban and peri-urban areas of Ulaanbaatar and also developing mining regions of Mongolia. The work will be followed by more wide spread application of the remediation technique and strengthening of the environmental assessment and analytical capacity of government public health and environmental management agencies. Remediation of metal contaminated soils due to anthropogenic inputs from urban development and mining activity in Mongolia is considered an important issue by the World Health Organization and the Mongolian Government. Such remediation is urgently required within Ulaanbaatar and at sites such in the Zaamar district, Tuv Province, downstream in marshes and mires of the Tuul River and also in the Boroo River catchment. Silt, mud and metals have contaminated large areas as a consequence of large multi-national mining operation, widespread artisanal mining and other unregulated mining activities. The current situation and approaches to be taken are considered.

Keywords Bioavailability • Heavy metals • Mining remediation • Nanoparticles • Phytoremediation • Wetlands

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8.1 Introduction

Remediation of metal contaminated soils and water resources, due to urbanisation and mining activity in Mongolia, is considered an important issue by the World Health Organization and the Mongolian Government. The problem of heavy metal contamination has been reported both in the capital city, Ulaanbaatar with a population of approximately 1.4 million, and also in many regional areas of Mongolia (Chung and Chon 2014; Murao et al. 2006; Tumenbayar 2003; World Bank 2006). The body burden and human health impact resulting from the use of mercury in artisanal gold mining is high not only in the miners themselves, an increased mercury hazard was also found for inhabitants of mining areas who were not actively involved in mining (Steckling et al. 2011).

8.2 The Situation

Ulaanbaatar, the capital city of Mongolia in the central Asia (Fig. 8.1), is reliant on a coal-based fuel–energy complex which is one of the major sources of environmental pollution. The city has three coal-fired thermal power plants and about 100,000 dwellings using coal for their heating and cooking. About half of the city inhabitants live in apartments and the other half in traditional tents (*gers*) and small individual houses. About 80% of inhabitants living in apartments use central heating and hot water from the three thermal power plants which are located in the city center, and the rest of the inhabitants of apartments use heating boilers and individual stoves. Their basic fuel is brown coal of the Baga Nur, Nalaikh, and Chulut deposits, which

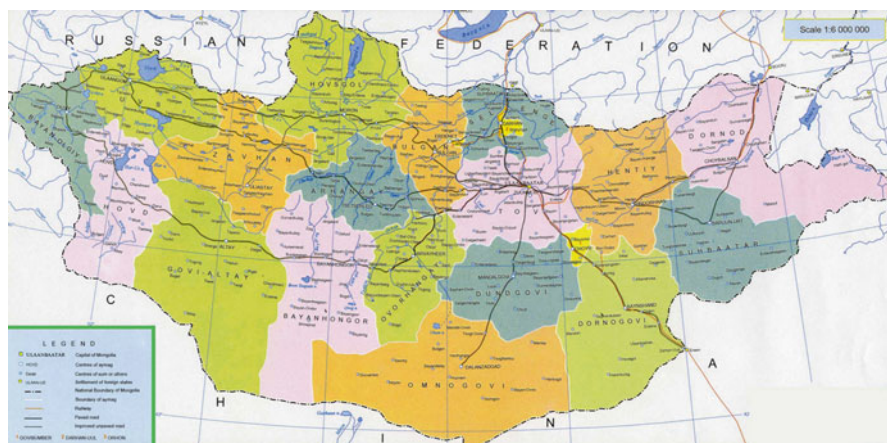


Fig. 8.1 Map of Mongolia showing Provinces, towns and road/rail network

are enriched in Pb, As, and Mo. Control or reduction of coal combustion emissions is uncommon. Gasoline in the country is treated with tetraethyl lead and catalytic converters are rarely present or functional. These conditions have resulted in poor air quality, soil and water pollution with contaminants including Hg, As, Zn, Cu, Cr, Cd, Mo and Pb (Kasimov et al. 2011).

In recent years, artisanal gold mining (digging and smelting by individuals, usually unregulated) is increasingly gaining momentum in developing countries as a subsistence activity. In Mongolia, it is largely practiced in rural areas by unemployed or struggling herder populations who lack the requisite education, training, management skills and equipment required to carry-out such activities in an environmentally sustainable fashion. It is estimated that the total number of artisanal miners in Mongolia is about 50,000, including approximately 25,000–35,000 artisanal miners who are panning placer gold in the North Khentei gold belt. Contamination from their activities includes elemental Hg waste, tailings, silt and mud runoff containing Hg, Pb and other metals. These workers are called “Ninja” miners because their clandestine activity in narrow, irregular, mining excavations reminds people of historical Japanese secret service mercenaries, *Ninja*. (Murao et al. 2006).

Silt, mud and metals have contaminated large riverine, wetland and mire areas as a consequence of widespread artisanal mining and other mining activities (Figs. 8.2 and 8.3). It has been reported that placer gold mining (mining of stream bed or allu-



Fig. 8.2 Artisanal mining diggings site, Tuul River catchment, June 2015 (Photo: Batdelger Shinen)



Fig. 8.3 Water turbidity and silt deposits from mining activity in the Tuul River marshes, October, 2013 (Photo: Batdelger Shinen)

vial deposits) at the Zaamar site has increased the total riverine mass flows of Al, As, Cu, Fe, Mn, Pb and Zn by 44.300, 30.1, 65.7, 47.800, 1.480, 76.0 and 65.0 tonnes per year, respectively (Thorslund et al. 2012). Remediation is urgently required at sites within Ulaanbaatar and in the Zaamar district, Tuv Province, downstream in marshes and mires of the Tuul River and in the Borro River catchment. The importance of wetland nutrient transformation has been noted in Mongolia, however, the potential of wetland systems for heavy metal and sediment management has not been recognized (Itoh et al. 2011). The above contaminated areas are within an approximate 200–300 km radius of Ulaanbaatar (Fig. 8.4), predominantly in the NW and NE sectors.

Unlike many other developing countries, artisanal mining is not part of traditional subsistence economy in Mongolia. It is suggested that the initiation of artisanal mining and its dramatic growth over the past decade, particularly in terms of organizational structure, is an outcome of poverty-driven self-help efforts to reduce some of the more insidious effects of developing a market economy, including job loss, declining real incomes, decline in rural services, and the difficulty all governments face in providing a rural safety net particularly in the face of natural disasters and climate change (World Bank 2006).

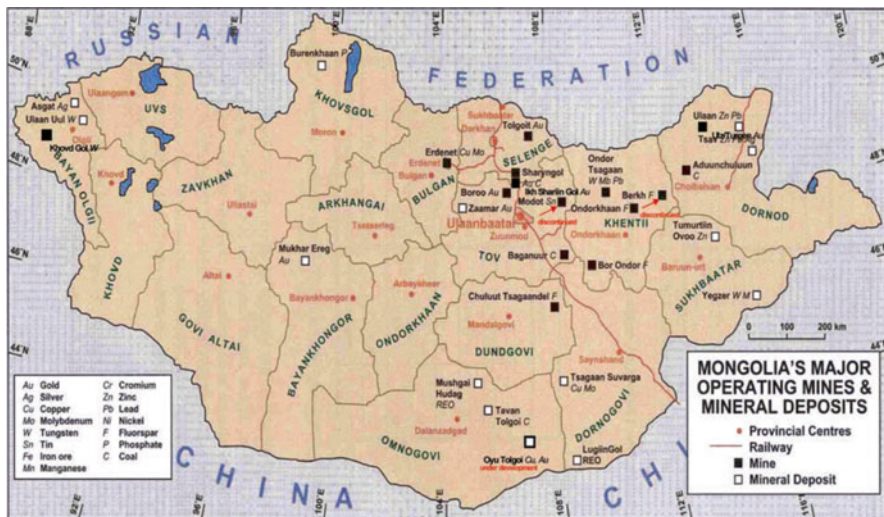


Fig. 8.4 Major operating mines and mineral deposits in Mongolia (Source: Rheinbraun Engineering und Wasser GMBH. 2003. “Review of the environmental and social policies and practices for mining in Mongolia.” Rheinbraun engineering background papers. Cologne)

8.3 Remediation Options and Recommendations

A number of remediation approaches have been used to diminish toxic effects of metals in the environment. Remediation of metal contaminated soil and sediment may involve separation, excavation, thermal extraction, stabilisation or biological methods. Physical separation processes are used to reduce the volume of metal contamination in size or type for further treatment. Excavation of contaminated soils and impoundment in landfill is a conventional method in developing counties (Figs. 8.5 and 8.6). However, this technique is environmentally disruptive and can result in an increased risk of heavy metal leaching (Mulligan et al. 2001).

Prevention of pollution is always preferable to follow-on remediation. It is important to emphasise that mining, industrial and agricultural activities should always use environmentally sustainable and responsible methodologies. As such practices are commonly not rigorously followed in both developed and developing countries, metal contamination is a difficult and wide spread issue in many parts of the world. Heavy metal remediation techniques such as excavation and disposal or physical separation of contaminants from polluted sites are often inappropriate to apply in large and heavily contaminated areas due to economic, site social and ecosystem disruption and energy constraints. It is suggested that immobilization techniques, coupled with phytoremediation options, are more feasible and economical approaches in situations such as are present in Mongolia.

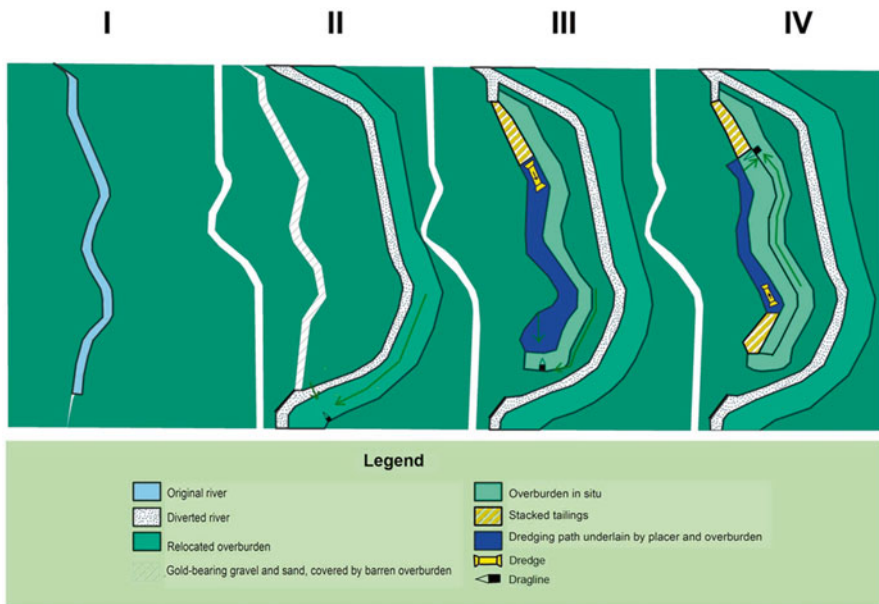


Fig. 8.5 Schematic representation of river diversion, dredging and overburden placement as used in mining operations at Zaamar, Tuv province, Mongolia. The technique results in major ecosystem disruption, sediment discharge, downstream metal contamination and is often left with no rehabilitation, resulting in further erosion and sediment discharge. The sequence could be modified with wetland creation to minimize sediment inputs to downstream waters and erosion of diversion channels



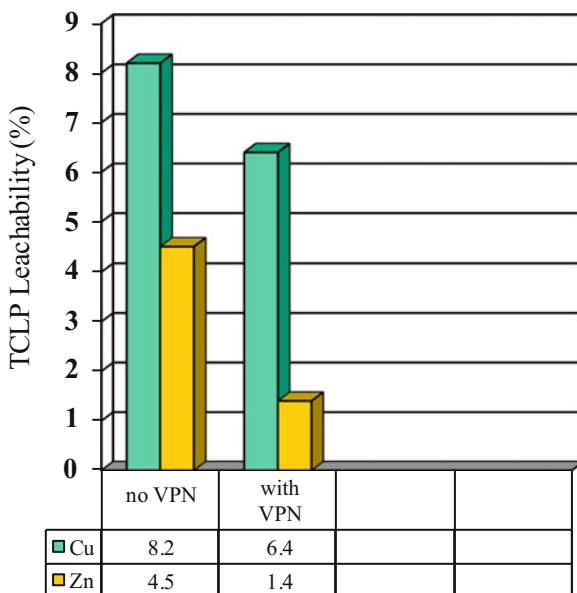
Fig. 8.6 Sediment sampling from tailings settling dam, large-scale mining site at Oyu Tolgoi, July, 2013 (Photo: Batdelger Shinen)

Previous studies have investigated the feasibility of iron phosphate (vivianite) nanoparticles for stabilisation of heavy metals in wetland sediments. Plant species such as *Helianthus annuus* were also used to phytoremediate the stabilised heavy metals. The effectiveness of the treatment was verified by examining nanoparticle sequestered metal leachability (Fig. 8.7), bioavailability and speciation in sediments using a Toxicity Characteristic Leaching Procedure, Physiologically Based Extraction Test and Sequential Extraction Procedures, respectively (Bavor and Shinen 2015; Rauret 1998; Ruby et al. 1999; USEPA 1992). These studies focussed on Cu and Zn, however, a number of researchers have reported significant immobilisation/phytoremediation, using similar approaches for As, Pb and Hg (Almaroia et al. 2014; Cabrejo and Phillips 2010; Ferreyroa et al. 2014; Goldowitz 2006).

A demonstration program for the feasibility of application of iron phosphate (vivianite) nanoparticle techniques, coupled with phytoremediation, for remediation of heavy metal contaminated soil and wetland linked water resources is recommended for establishment in selected contaminated sites within Ulaanbaatar and at sites such in the Zaamar district catchment area, Tuv Province, downstream in marshes and mires of the Tuul River and also in the Boroo River catchment.

Fig. 8.7 Toxicity Characteristic Leaching Procedure (TCLP) showing reduction in Cu and Zn leachability in untreated and nanoparticle (VPN) amended sediment. Leaching solution consisted of glacial acetic acid & sodium hydroxide (USEPA Method 1311, 1992). Total reduction of leachability was 22% for Cu, 69% for Zn

Leachability of Cu, Zn



The program should include the following components in an integrated remediation approach:

- Strengthen laboratory capability and field monitoring skills.
- Staff training.
- Infrastructure development – equipment, reagents and standards for sediment, plant biomass and water analysis.
- Characterize concentration, form and distribution of metals at selected site(s).
- Establish objectives and linkages with local stake holders.
- Establishment of small scale demonstration site(s) to verify remediation technique, linking immobilisation and phytoremediation techniques.
- Up-scale application of selected technique protocols to large field site.
- Monitoring to manage and confirm efficacy of remediation.
- Consideration of more wide-scale application of remediation.

8.4 Conclusions

Pollutant inputs from increased urbanisation and both regulated and unregulated expansion of the mining activities in Mongolia have resulted in an urgent need for heavy metal remediation of soil and water resources in urban and peri-urban areas of Ulaanbaatar and also in contaminated mining-activity regions of Mongolia.

A number of studies have investigated nanoparticle and phytoremediation technologies for the stabilisation of heavy metals. The studies support the establishment of an integrated feasibility program to demonstrate use of wetland-linked nanoparticle and phytoremediation technology for stabilisation/removal of heavy metals in contaminated sediments and water resources. Experimental data indicates that stabilised nanoparticles can effectively decrease the leachability and bioimpact of heavy metals. Further, the approach has been estimated to be economical and have ancillary ecosystem benefits. Such a program would require a government multi-agency approach and, given the developing country status of Mongolia, mining industry and/or international funding assistance would be needed.

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Chapter 9

Sludge Treatment Reed Beds (STRBs) as a Eco-solution of Sludge Utilization for Local Wastewater Treatment Plants

Katarzyna Kolecka, Hanna Obarska-Pempkowiak, and Magdalena Gajewska

Abstract Sewage sludge is a by-product of wastewater treatment processes. In Poland, the amount of sewage sludge has been still growing in recent years, due to enhancement of pollutants removal in wastewater treatment plants (WWTPs). One of the systems used to transform sewage sludge into fertilizer is a technology called Sludge Treatment Reed Beds (STRBs). Reed systems are periodically irrigated with sewage sludge with a low dry matter content (from 0.5 to 1.5 % DM). The accumulation time of sludge necessary to achieve stabilization is approximately 10–15 years. Sludge is transformed into humus substances. After checking the microbiological and physico-chemical properties stabilized sludge can be used for agricultural purposes. In the recent years, Denmark, France and Germany have applied integrated technologies of dewatering and stabilization of sewage sludge in these systems. The aim of the study is to present: (i) general principles for the construction and for the design of reed systems for integrated dewatering and stabilization of sewage sludge, (ii) conditions of proper operation of the systems (iii) the economic and environmental aspects of using reed technology based on the literature and our own research. Conducted research confirmed that sewage sludge dewatered and stabilized in STRBs can be used as fertilizer in agriculture.

Keywords Quality of sewage sludge • Integrated dewatering and stabilization • Reed beds

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9.1 Introduction

Treatment of municipal wastewater includes many processes which remove pollutants from wastewater. In wastewater treatment processes sewage sludge is produced as a by-product. The amount of generated sewage sludge depends on used technology of wastewater treatment and it usually equals from 1 to 5% of the inflowing wastewater. A large amount of produced sewage sludge with a high content of organic matter causes that the sludge management and utilization are the main problems of wastewater treatment plants. In recent years, many municipal wastewater treatment plants have been upgraded to meet the current requirements concerning wastewater effluent quality. At the same time, many local and on-site wastewater treatment plants were built on rural areas. As a result, a significant increase in the production of sewage sludge has been observed. In Poland, the amount of sludge has increased by 33.4% during the last 13 years (from 359 800 of dry matter in 2000 to 540 3000 in 2013). According to the National Waste Management Plan 2014 (NWMP 2014), this trend will be continue in the next years. However, it is assumed that since 2018 the growth of sludge will decrease about by one half.

In Poland, in the 1970s and 1980s in many wastewater treatment plants the management of sewage sludge was limited to the storage within the wastewater treatment plant area. The approach changes have been initiated in the 1990s after gaining the access to new efficient dewatering equipments. The use of these equipments is a subject of economic and environmental issue. So far, in small and medium-sized wastewater treatment plants sludge treatment is often restricted to the mechanical dewatering and disinfection, and its storage. According to the Polish legislation, since 2016 storage of sewage sludge (which exceeds the limit values for total organic carbon: $\text{TOC} > 5\%$, loss on ignition $> 8\%$, and heat of combustion $> 6 \text{ MJ/kg}$) is impossible.

Sewage sludge produced in small and medium-sized wastewater treatment plants, for the legal, practical and aesthetic reasons should be used in agricultural due to the high content of organic matter and the potential fertilizing properties. The sludge produced in large wastewater treatment plants should be thermally utilized (NWMP 2014).

In the recent years, Denmark, France and Germany have applied integrated technologies for dewatering and stabilization of sewage sludge in Sludge Treatment Reed Beds (STRBs) (Brix 2014; Nielsen et al. 2012; Uggetti et al. 2010; Troesch et al. 2009). For economic and environmental reasons it is an attractive solution, especially in rural areas. Reed systems are considered as eco-engineering solutions. Reed systems simulate natural processes occurring in the environment and the systems are specially designed and operated in order to achieve benefits for both people and the environment. Therefore, these systems meet the requirements of eco-engineering (Odum and Odum 2003; Jørgensen and Mitsch 1989).

The aim of this paper is to present the current experience on the integrated dewatering and stabilization of sewage sludge in STRBs technology based on the literature survey and our own research.

9.2 Construction and Design

Reed systems are periodically irrigated with sewage sludge with low dry matter content (from 0.5 to 1.5% DM). The accumulation time of sludge necessary to achieve stabilization is approximately 10–15 years. Sewage sludge can be supplied directly from the activated sludge chamber, secondary settling tank, sludge thickener or the digester chamber. Before supplying into the reed system, sewage sludge is often stored in a retention tank, where it is mixed and aerated (Nielsen 2003).

STRBs are typically built as beds or basins with horizontal layers of substrate. They are composed of several layers and are the most commonly planted with macrophytes. The rate of dewatering is dependent on the particle size of layers and properly designed drainage system connected to the ventilation which prevents the formation of anaerobic conditions. Reed systems are built as a free-standing concrete ground tanks or as sealed ground basins. The bottom of the system must be protected by impermeable geomembrane to eliminate leaks and contamination of groundwater.

Taking into account the duration of the system use, it is important to leave a space for the storage of sludge. Considering the time required to stabilize the sludge, the height of beds/basins should be about 1.5 m. According to Nielsen (2003) the depth of systems operated for 8–12 years should be at least 1.7–1.8 m (above the filter layers). This approach enables the accumulation of sludge to a depth of 1.5–1.6 m. Most often objects with a depth of 2.4 m are designed, where filter layer is 0.6–0.7 m. During a resting period (before removing the stabilized sludge) sludge thickness is reduced to approx. 1.2–1.5 m (Nielsen 2002; Kofecka and Obarska-Pempkowiak 2013). Increase of sludge layer depth is mostly monitored by measuring patches installed at selected points. In general, the height of the system should provide accumulation of 1 m layer of sludge, which corresponds to an average sludge accretion rate about 0.1 m year⁻¹ (Vincent et al. 2012).

Dimensioning of reed systems requires the consideration of many aspects, among them quantity of the sludge (tons of dry matter/year), the quality of sludge (usually based on a dry matter basis), capillary suction time, the type of sludge (depends on sludge treatment methods) and climatic conditions (De Maeseneer 1997).

There are no standard values for design factors and configurations of STRBs. The main design factor is the sludge loading rate (Uggetti et al. 2010). On its basis indispensable area of the system is determined. Selected loading rate should take into account the origin and type of sludge, climate and assumed time of sludge accumulation. Previous research has shown that activated sludge (average moisture content of 98.4%) has better filtration properties than sludge from septic tanks (average moisture content of 87.5%). Consequently, in reed systems where

secondary sludge is supplied, it is possible to use higher loading rate (Vincent et al. 2011). After the commissioning period, which lasts approximately 2 years, loading rate can increase to the maximum design value. This value for secondary sludge is approx. $60 \text{ kg DM m}^{-2} \text{ year}^{-1}$. In case of sludge with a high content of fat or fermented substances the recommended loading rate is $50 \text{ kg DM m}^{-2} \text{ year}^{-1}$. To provide sufficient time for the transformation of accumulated sludge, the STRBs should have at least 8 basins/beds (Nielsen 2003; Kofecka and Obarska-Pempkowiak 2008; Vincent et al. 2012).

An important element of the system is the correct construction of the drainage layer, which drains leachate and ensures the inflow of air. The drainage system should be designed for maximum outflow of leachate, which is $1.4 \text{ dm}^3 \text{ min}^{-1} \text{ m}^{-2}$ (Nielsen 2002; Troesch et al. 2009). Filter layers should be formed with fine gravel to provide filtration of water from supplying sludge. The surface layer, which has contact with the air, should be planted with reed (Fig. 9.1). Above designed height of system ventilation chimneys should be built to provide air inflow.

The reed (*Phragmites australis*) is used in treatment sludge systems due to many advantages, among other the branched root and rhizomes system, the capacity of water absorption from sludge, and then it release into the atmosphere through evapotranspiration, the tolerance to cold and arm weather, high resistance against toxic micropollutants, ability to transport oxygen from the air to the rhizosphere. Stems and roots of reed allow the filtration of water to drain the system. Vigorous growth of reed rhizomes creates free spaces inside the system, which allows for water and air penetration into the system. Reed movements caused by wind form

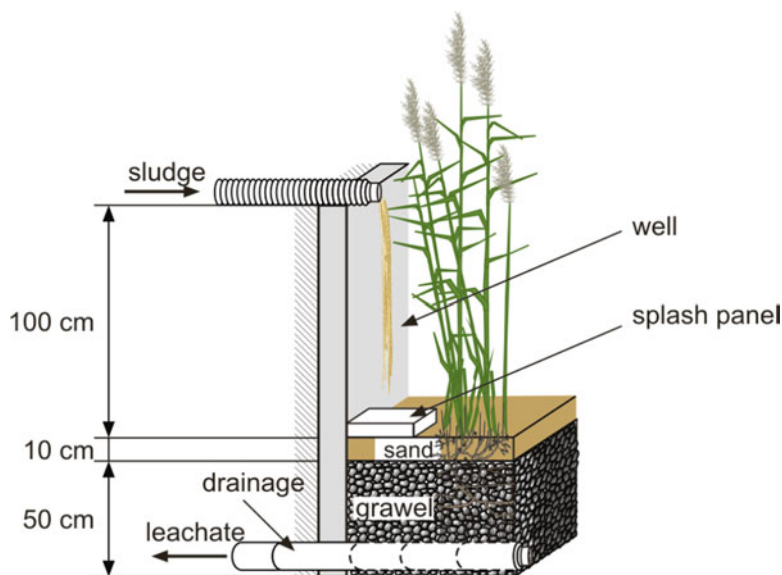


Fig. 9.1 The scheme of STRBs (Modified from Zwara and Obarska-Pempkowiak (2000))

free space on the surface of the system, which positively affects the process of percolation (Peruzzi et al. 2009). The number of reed seedlings should be 4 m^{-2} in the layer of growth (Obarska-Pempkowiak et al. 2003; KołECKA and Obarska-Pempkowiak 2009).

9.3 Operation

During the operation of reed systems there are three periods (commissioning, full operation, emptying), which constitute one cycle. This cycle lasts for 8–12 years, although studies show that it may take even more than 15 years. The total period of the system operation consists of 2–3 cycles and it may last 30 or more years.

During the first period of commissioning (about 2 years), one dose of sludge should be lower than designed values (Nielsen 1994). After this period, the plants are already fully developed and the system can be operated with full loading rate which is typically about $50\text{--}60 \text{ kg DM m}^{-2} \text{ year}^{-1}$. Sludge is supplied until the system is full. The stabilization period is usually from 8 to 12 years. Then in the third period, dewatered and stabilized sludge is removed from the system and the beds/basins are sequentially emptied. Number of beds/basins, which are simultaneously emptied, is usually two from the eight operated. Before emptying beds/basins should not be used for some time (preferably in summer) to increase the content of dry matter (Troesch et al. 2009; Nielsen 2011; KołECKA and Obarska-Pempkowiak 2013).

During the operation, the individual beds/basins are fed periodically. Resting periods (no sludge supply) may take several days, depending on beds/basins efficiency, weather conditions, the age of a system, and the dry matter content in sludge as well as thickness of accumulated sludge (Nielsen 2003). More beds/basins provide longer rotation, which causes longer rest between supplying of sludge. This improves the efficiency of dewatering and stabilization of the sludge.

Evapotranspiration is the main factor in the ability of reed beds to filter and dewater the sludge. Another important factor is the vertical flow of leachate through the beds/basins. Meteorological parameters such as sunlight and temperature have also impact on the process of dewatering. For the proper dewatering and stabilization of the sludge, the supply schedule of the beds/basins is very important.

For the countries from the northern part of the Mediterranean Region, the period of sludge supplying is 1 week during the winter. In this case, the resting period should last about 3 weeks. In the summer, supply and rest periods last 1 and 2 weeks, respectively (Stefanakis and Tsihrintzis 2011).

9.4 Methodology

9.4.1 Research Objects

The characteristic of STRBs from Danish and Polish wastewater treatment plants is presented in Table 9.1.

Analyzed sewage sludge originated from conventional wastewater treatment plant located in Helsingør, Rudkøbing, Nakskov and Vallo in Denmark and Gniewino in Poland. Danish systems consist of 8 or 10 beds and treat sludge from 9,000 to 40,000 pe. The operation time is from 7 to 15 years (Kofecka and Obarska-Pempkowiak 2013). The system in Gniewino is relatively new. Full operation of the facility started in 2013 after 2 years of commissioning phase. This system stabilizes sludge from 24,000 pe and consists of 6 beds. In spring 2016, the next 3 beds will be constructed.

9.4.2 Sampling and Analyzes

Samples of sludge were collected from selected beds by using a special probe at six points and the results were averaged in each STRB.

Determinations of dry matter, organic matter, nitrogen and phosphorus were carried out according to the Polish Standards and APHA (2001). Determinations of selected heavy metals in the sludge were made by using of atomic absorption after their mineralization.

Table 9.1 The characteristic of analyzed STRBs

Location	Polish system	Danish systems			
	Gniewino	Rudkøbing	Nakskov	Vallo	Helsingør
Operation time (years)	3	13	15	7	9
Number of beds	6	8	10	8	10
Total area (m ²)	2 400	5 000	9 000	3 900	10 500
Amount of sludge (t DM year ⁻¹)	100	232	870	300	630
pe ^a	15 000	13 000	33 000	9 000	40 000

^aPerson equivalent

9.5 Results and Discussion

9.5.1 Dry Matter and Organic Matter

The results of dry matter and organic matter content in sludge stabilized in STRBs and in raw sludge are presented in Figs. 9.2 and 9.3.

The research revealed that the reed system in Gniewino had the lowest dry matter content and the highest organic matter content. Significant differences in dry matter and organic matter content between the Polish and Danish STRBs were mostly caused by the length of operation. Danish systems have been in operation from 7 to 15 years, while the Polish facility only for 2 years.

The dry matter content depends not only on the operation time, but also on the duration of resting period before sampling. Rudkobing, where the dry matter content was the highest, the raw sludge has not been supplied for 2 years before

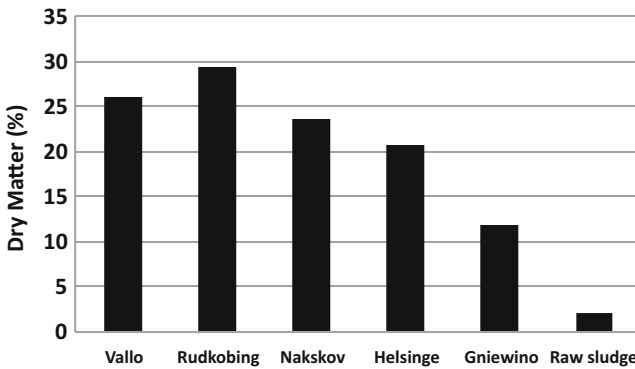


Fig. 9.2 The average content of dry matter in sludge stabilized in STRBs and in raw sludge

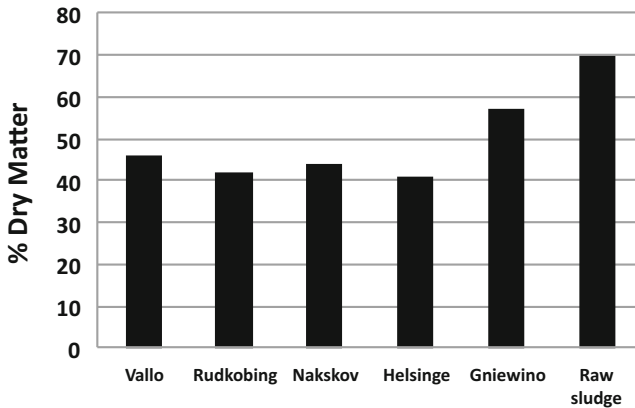


Fig. 9.3 The average content of organic matter in sludge stabilized in STRBs and in raw sludge

sampling. By contrast, the system in Helsingø with the low content of dry matter was supplied with raw sludge just 2 weeks before sampling. On this basis it can be concluded that after long-term dewatering and stabilization period the content of dry matter in sludge can achieve 30 %.

9.5.2 Nutrients

The concentrations of nutrients in sewage sludge dewatering and stabilized in STRBs are shown in Figs. 9.4 and 9.5.

The concentration of total nitrogen in sludge from Gniewino reed system was two times higher than in Danish systems. Relatively high nitrogen concentration was caused probably by inflow of industrial wastewater with high concentrations of this element. Also, short operation time may have negative impact. In STRBs, nitrification takes place in aerobic zones nearby roots and rhizomes while denitrification takes place in anoxic/anaerobic zones in the beds. These processes contribute to the removal of nitrogen compounds out of the system in gaseous forms (Kofecka and

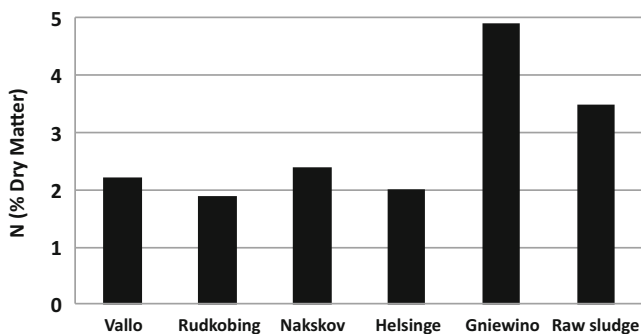


Fig. 9.4 The average concentration of nitrogen in sludge stabilized in STRBs and in raw sludge

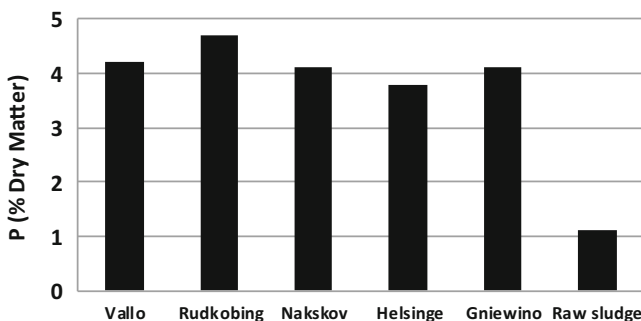


Fig. 9.5 The average concentration of phosphorus in sludge stabilized in STRBs and in raw sludge

Table 9.2 Average heavy metals concentrations in sewage sludge from STRBs and limit values for agricultural use according to The Regulation of the Environment Minister, mg kg⁻¹ DM

Heavy metal	System					Permissible values of metals
	Gniewino	Helsingø	Rudkøbing	Nakskov	Vallo	
Cd	1.54	0.95	0.84	0.74	1.07	10
Pb	16.0	10.7	14.6	15.6	10.0	500
Ni	16.6	20.7	20.3	22.4	26.3	100
Cr	22.4	18.1	32.1	17.5	11.3	500
Cu	105.0	237.0	219.0	81.0	166.0	800
Zn	749.0	416.0	542.0	437.0	520.0	2500

Obarska-Pempkowiak 2013). The values obtained in Gniewino were comparable with the nitrogen concentrations in the manure, which average is 2.0 % DM. (Kotecka and Obarska-Pempkowiak 2013).

The concentration of total phosphorus was approximately 4 % DM and it was at the same level in all surveyed systems. The obtained results were above the average values of phosphorus in natural fertilizers. For example, in manure the average concentration of total phosphorus is approximately 1.2 % DM. (Kotecka and Obarska-Pempkowiak 2013).

9.5.3 Heavy Metals

Due to the high concentrations of nutrients, stabilized sludge can be considered as a potential fertilizer. In Poland, however, to use the sludge for agriculture the concentrations of heavy metals must be below the permissible values specified in The Regulation of the Minister of the Environment from 6 February 2015 for municipal sewage sludge (year 2015, item 257).

Average concentrations of six selected heavy metals in sewage sludge from STRBs and limit values for agricultural use are given in Table 9.2.

Based on obtained results it was indicated that analyzed heavy metals concentrations in sewage sludge from STRBs were substantially below the permissible values for the agricultural use.

9.5.4 Pathogenic Microorganisms

Dewatered and stabilized sludge can be used as fertilizer if it is free of pathogenic microorganisms. Raw sewage sludge contains a large number of pathogens. For example, the counts of *Salmonella* can amount to 10–100 1 cm³, *Enterococci* and *Escherichia coli* 10,000–1,000,000. The research have shown that the

Table 9.3 The comparison of mineralization degree of linear alkylbenzene sulphonates (LAS) and nonylphenoethoxylates (NPE) for different sludge treatment methods (Nielsen 2005)

	Methods			
	STRBs (%)	Sludge storage (%)	Mechanical aeration (%)	Composting (%)
LAS	98	10	75–95	78–95
NPE	93	41	95	100

above-mentioned microorganisms in sludge from STRBs are significantly reduced within 3–4 months. After this period, number of bacteria is 2/100 g of sludge for *Salmonella*, <10/100 g for *Enterococci* and <200/100 g for *E. coli* (Nielsen 2007).

9.5.5 Hazardous Organic Compounds

Highly-effective methods of wastewater treatment mostly produce more sludge containing hazardous organic micropollutants. These micropollutants should be removed because they can bioaccumulate and biomagnificate in living organisms. This can cause carcinogenic and mutagenic diseases because of their toxic properties. Linear alkylbenzene sulphonates (LAS) and nonylphenoethoxylates (NPE) are defined as one of such hazardous organic compounds. However, they are biodegradable in stabilized sludge. According to Nielsen (2005) the degree of mineralization is 98 % for LAS and 93 % for NPE in reed systems. The comparison of changes in the concentrations of organic micropollutants for different sludge treatment methods is given in the Table 9.3. Matamoros et al. (2012) showed that cosmetics and pharmaceuticals can also be removed with high efficiency during stabilization of sewage sludge in reed systems.

9.6 Ecological and Economic Aspects of the Integrated Sludge Treatment in STRBs

Reed systems of integrated dewatering and stabilization of sewage sludge is the natural and environmental friendly technology. These systems do not require the use of additional chemicals (e.g., coagulants or polyelectrolytes) to improve the process. In comparison with other conventional methods of sludge treatment, such as composting, or thermal incineration STRBs do not cause unpleasant odors.

It is a low-emission method. It has been estimated based on a calculation that the reed system sludge causes emissions 46.6 t/annually of CO₂, 5440 kg/annually of NO_x and 363 kg SO₂/annually during stabilization of 100 tonnes of dry matter. For comparison agricultural use causes emission about 37 % higher and thermal combustion about 43 % higher. The operation of reed systems does not require the use

of heavy traffic and other dangerous machines. Such an equipment is needed only during the emptying of beds/basins, which is every 10–15 years.

In addition, this is low-energy-consuming methods, because electricity is needed only to pumps supplying sludge to the system. Operating costs are very low and amount about 5–10 % of the cost of conventional methods (Kofecka and Obarska-Pempkowiak 2013).

9.7 Conclusions

Based on own research and literature reports the following conclusions can be formulated:

1. Sludge Treatment Reed Beds are examples of eco-engineering solutions not only due to natural processes, but also for economic and environmental reasons
2. The quality of sludge as well as climatic conditions should be taking into account to proper design of reed system.
3. Proper operation of the systems requires at least 8 beds.
4. Sewage sludge stabilized in STRBs has high content of dry matter and low content of organic matter in comparison to sludge stabilized with conventional technology.
5. The final residual sewage sludge can be used as potential fertilizer, thanks to the relatively high concentrations of nitrogen and phosphorus in comparison to organic fertilizers, eg. manure.

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Chapter 10

Dairy Wastewater Treatment by a Horizontal Subsurface Flow Constructed Wetland in Southern Italy

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Abstract Evidence on the efficiency of horizontal flow constructed wetlands (HF CWs) in treating dairy wastewater in the Mediterranean region is reported, showing results from the 3 year long monitoring of a HF CW treatment plant situated in southern Italy. The HF CW treats a mixture of different wastewaters produced by a dairy farm (dairy, milking, milk cooling, restaurant, and house). Samples of wastewater quality (pH, COD, and N-NH₄⁺) were collected at the inlet and outlet of HF CWs from February 2012 to May 2015. The effluent COD concentrations from dairy activities alone were also collected during the same period, showing the most relevant contribution of dairy wastewater in terms of organic loads. The start-up phase was influenced by the influent pH being too low, which was fixed by adopting a serum separation. The system showed some stress during a management phase lasting 2 months, requiring then 1 month of recovery period. The overall treatment performance is now very satisfactory, with 94.3 % COD removal efficiency based on average influent and effluent values, while a slight increase in effluent N-NH₄⁺ was registered, probably due to organic matter ammonification.

Keywords Dairy wastewater treatment • Horizontal subsurface flow constructed wetland • Mediterranean climate

10.1 Introduction

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 parlor (Kadlec and Wallace 2009). These activities lead to a dairy

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wastewater characterized by high organic matter concentrations and a wide range of pH as reviewed by Vymazal (2014). The organics present in the wastewater are mainly carbohydrates, proteins and fats originating from the milk. A wide range of pH (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⁻¹; COD 2000–90,000 mg L⁻¹; N-NH₄⁺ 20–150 mg L⁻¹).

Treatment of dairy wastewater through conventional biological treatment technologies (e.g., activated sludge) is problematic for a number of reasons: (i) the high variability in both hydraulic and organic loads of dairy wastewater can be difficult to manage, since more temporally constant influent loads are needed to ensure appropriate performance; (ii) the high organic loads lead to high sludge production, increasing the management costs; and (iii) the need for specialized operation and management staff is a burdensome extra cost, especially for small- and medium-sized dairies. Hence, more flexible constructed wetland (CW) treatment technologies have been widely adopted to treat dairy wastewater (Kadlec and Wallace 2009; Vymazal 2014). Among the different CW configurations, horizontal subsurface flow (HF CW) has been one of the most widely used solutions for treatment of dairy wastewater, showing satisfactory removal efficiencies of 50–98% for BOD_5 and 40–96% for TSS (Vymazal 2009, 2014). However, a recent literature review (Vymazal 2014) found a lack of evidence of HF CW capability to treat dairy wastewater in the Mediterranean region. For this reason, a 3-year case study of HF CWs used to treat wastewater from a dairy farm in southern Italy is reported here as a representative case study of HF CW performance in the Mediterranean climate.

10.2 Material and Methods

The experimental case study is located at Fattoria della Piana, Candidoni (Italy – 38°N, 15°E), a farm situated in the South of Italy. The farm promotes an eco-sustainable approach, recovering the end-products from the dairy (serum) and stable (manure) to produce biogas, and treating wastewaters with CWs.

Fattoria della Piana wastewater comes from a number of sources: houses and a restaurant (maximum 12 residents and 100 restaurant users); the milk cooling plant (average 20 m³d⁻¹); the dairy (20 tons per day of processed milk, which produces 20 m³d⁻¹ of wastewater); and milking (200 livestock, which produce 20 m³d⁻¹ of wastewater). On average, the daily wastewater quantity is 85 m³d⁻¹.

A CW treatment plant has been built to treat this wastewater. According to Italian law, the Fattoria della Piana wastewater can be classified as domestic, and therefore quality limits are only imposed for COD, TSS, and pathogens concentrations. The scheme of the wastewater treatment plant is reported in Fig. 10.1: an equalization

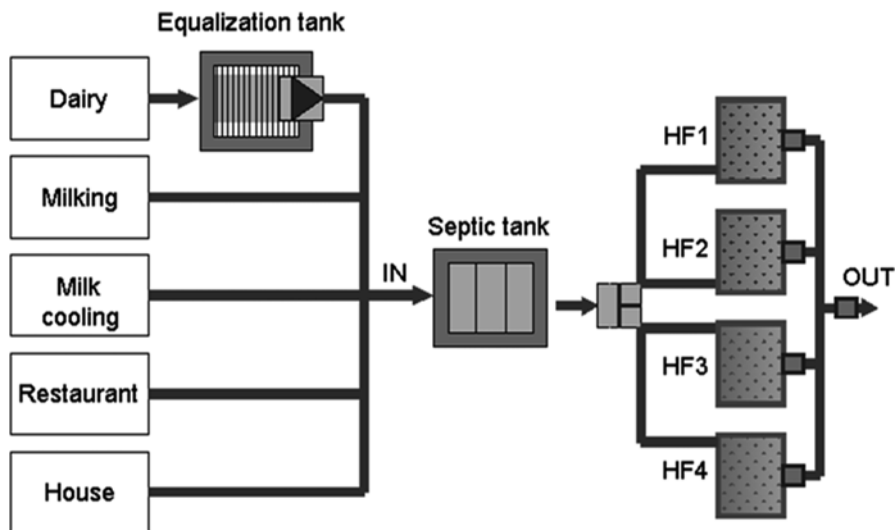


Fig. 10.1 Scheme of CW wastewater treatment plant

tank is installed for the dairy wastewater; primary treatment for wastewater from all the different sources is performed by a three-chamber septic tank; four horizontal flow constructed wetlands (HF CWs) are installed in parallel as secondary treatment, with a total surface area of 2280 m² (theoretical hydraulic retention time – HRT – 5.3 days; gravel Ø 8 mm, depth 0.8 m, 15 m W / 38 m L for each of the four HF beds). Each HF CW is also subdivided in 2 hydraulically separated sectors to facilitate management operations, therefore the CW treatment plant is composed of 8 separated sectors. The treated effluent is discharged into a small stream. The treatment plant started operation in February 2012. Since the pH of influent wastewater was too low in the first operational period, the serum from dairy activities (highly acidic) has been separated and sent to an anaerobic digester for biogas production since the middle of May 2012. Management activities were performed between December 2014 and January 2015, during which only 7 of 8 sectors were operational and the water table within the beds was reduced.

Samples of wastewater quality (pH, COD, and N-NH₄⁺) were collected at the inlet and outlet of the HF CWs from February 29, 2012 to May 11, 2015. Effluent COD concentrations from the dairy equalization tank were also collected during the same period. The wastewater quality dataset is analyzed here to investigate the functioning and efficiency of the HF CW treatment plant for dairy wastewater mixed with other pollutant sources (e.g., milking, domestic). Moreover, we describe in detail the atypical functioning phases of start-up and management, to understand the HF CW response during these critical phases.

10.3 Results and Discussion

10.3.1 Role of Dairy Wastewater on the Mixed Wastewater Composition

The samples collected at the effluent of the equalization tank can be considered representative of the dairy wastewater composition, while those collected at the influent of the HF CW are representative of the mixed wastewater. The dairy almost always has higher COD concentrations than the mixed wastewater (Fig. 10.2). The average COD concentration in dairy wastewater during the study period was 4079 mg L^{-1} (Table 10.1), which is within the range reported by Vymazal (2014; 2000–90,000 mg L^{-1}) and twice the average influent concentration of the mixed wastewater. No relevant changes in pH were observed (Table 10.1). Since only three values of N-NH_4^+ concentration from the equalization tank effluent have been measured, the same analysis has not been done for N-NH_4^+ .

The mixing of dairy wastewater with other, less concentrated pollution sources (wastewater from milking, milk cooling, the restaurant, and houses in this specific case) has beneficial effects and should be always performed when possible in farm wastewater treatment. Mixing dairy wastewater with less concentrated wastewater reduces the organic matter concentration in the HF CW influent, limiting the risk of clogging and thus increasing the lifespan of the treatment plant. Furthermore, the domestic blackwater provides nutrients that are often lacking in agricultural effluents.

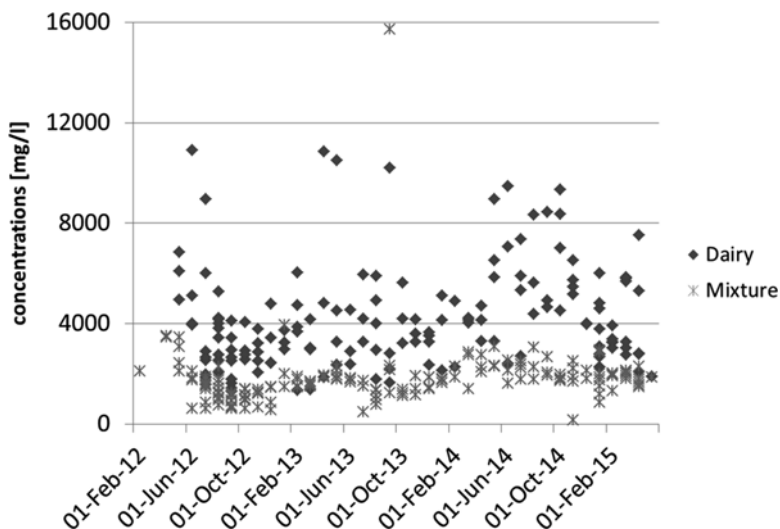


Fig. 10.2 COD concentrations in effluent from the equalization tank (representative of the dairy wastewater – diamonds) and inflow of the HF CW (representative of the mixed wastewater – stars) at Fattoria della Piana (Candidoni, Italy) from February 2012 to May 2015

Table 10.1 Descriptive statistics of pH values and COD concentrations in the effluent from the equalization tank (representative of the dairy wastewater) and in the inflow of the HF CW (representative of the mixed wastewater)

	pH		COD (mg L ⁻¹)	
	Dairy	Mixture	Dairy	Mixture
Mean	5.8	5.9	4079	1766
St. dev.	0.7	0.4	1982	1281
Min	2.5	5.1	1284	140
Max	10.0	6.8	10,843	15,730
# of samples	153	154	147	152

Data collected after the starting period, from July 2012 to May 2015

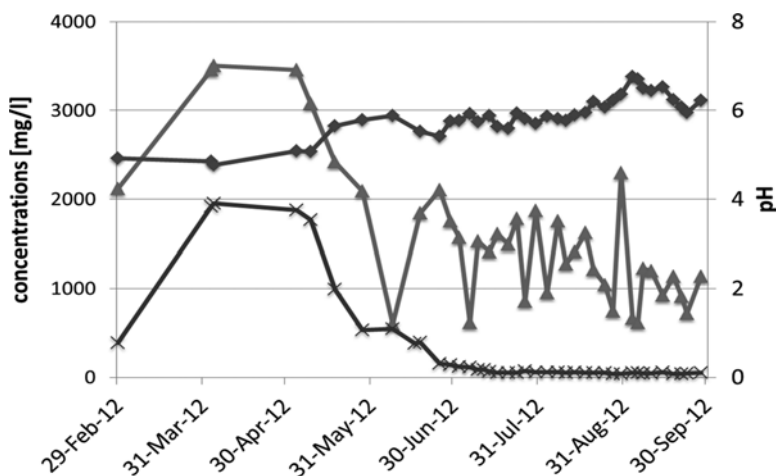


Fig. 10.3 Correlation between influent (triangles) and effluent (crosses) COD concentrations from the HF CW and influent pH (diamonds) in the starting period (from February to September, 2012)

10.3.2 Start-Up Phase

The first operational period failed in COD removal due to low influent pH (average value of 4.9 from March 2012 to the middle of May 2012). In order to deal with this issue the serum (highly acidic) has been separated at the source and sent to an anaerobic digester for biogas production since the middle of May 2012. This approach promoted an increase in influent pH (around 6), which produced a more suitable environment for the bacterial communities and, consequently, a higher HF CW removal efficiency. This is confirmed by Fig. 10.3, where the improvement in COD treatment performance is clearly correlated with the increased influent pH prompted by serum separation.

Data reported in the literature show a wide range of pH values for dairy wastewater (Vymazal 2014), 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 to optimize 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 (in this case, the possibility of separating the serum) and carefully analyze treatment performance during the start-up phase.

10.3.3 Management Phase

After two and a half years of proper functioning, some management activities were performed at the end of 2014, for an early appearance of clogging signals in 1 of the 8 sectors of the CW system, mainly due to some overloads events that took place in the former period. The management, which was done between December 2014 and January 2015, required a limitation on the number of hydraulically separated sectors to treat the wastewater (7 instead of 8, which increased the hydraulic loading rate for each of the remaining 7) and lowering the water table within the beds (reducing the hydraulic retention time). As visible in Fig. 10.4, during the management phase the system showed some stress, maintaining good removal efficiencies but exceeding the limit of 160 mg L^{-1} (Fig. 10.4). After all 8 hydraulically separated sectors and the correct water level were re-established (in February 2015), the HF CW needed a recovery period of almost 1 month to re-establish the removal efficiencies obtained before the management phase.

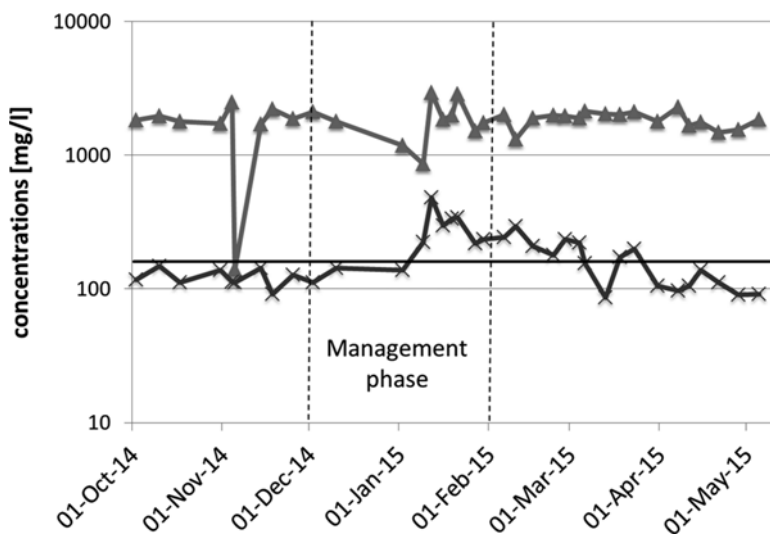


Fig. 10.4 Influent (triangles) and effluent (crosses) COD concentration from the HF CW treatment plant at Fattoria della Piana (Candidoni, Italy) before, during, and after the management phase, which lasted from December 2014 to January 2015. Black continuous line indicates the Italian limit for discharge into surface water. Vertical axis is in logarithmic scale

These results suggest that HF CWs are able to recover the treatment performance after a management phase in a relevantly short time period and the frequency of such maintenance operations is very low.

10.3.4 Overall Treatment Performance

Influent and effluent COD concentrations from the HF CWs for the analyzed period (from February 2012 to May 2015) are shown in Fig. 10.5, highlighting very good performances in COD treatment. Aside from the start-up and management phases, the effluent COD concentration was almost always below the Italian limit of 160 mg L^{-1} for discharge of wastewater in fresh water bodies, except in a few cases.

Table 10.2 summarizes the influent and effluent pH, N-NH_4^+ , and COD from the HF CWs, using only data after the start-up phase, but including the management phase (i.e., from July 2012 to May 2015). pH was quite stable (low standard deviation) both at the influent and the outlet, with values within the range for optimal bacterial activity (approximately 6–9 – Kadlec and Wallace 2009). COD removal efficiency was very high, with a relatively stable, low effluent concentration (Fig. 10.6). Average COD removal efficiency was 94.3% based on all 156 samples (Table 10.2), and $93.1 \pm 4.9\%$ (69–97.8%) when considering only the samples corresponding to a HRT of 4–6 days (52 samples). A slight increase in average N-NH_4^+ effluent concentration was observed (Fig. 10.6); this is probably due to ammonifica-

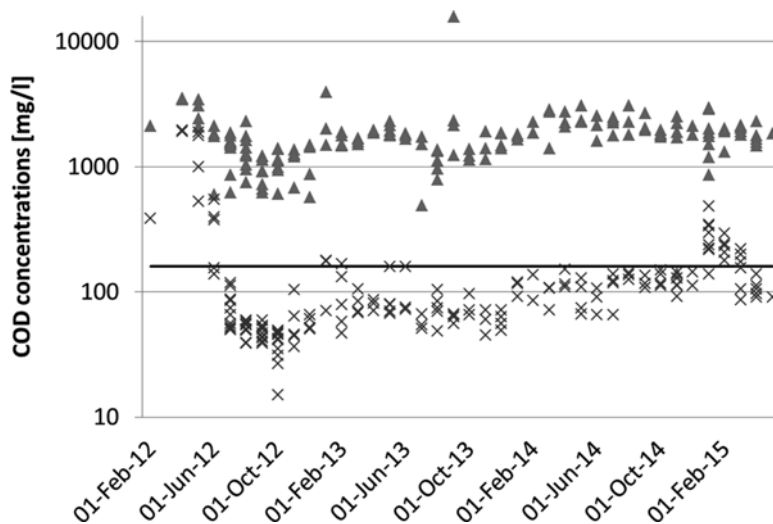


Fig. 10.5 Influent (*triangles*) and effluent (*crosses*) COD concentration from the HF CW treatment plant at Fattoria della Piana (Candidoni, Italy) from February 2012 to May 2015. *Black continuous line* indicates the Italian limit for discharge into surface water. Vertical axis is in logarithmic scale

Table 10.2 Descriptive statistics of pH, N-NH₄⁺, and COD concentrations at the inflow and outflow of the HF CW. Data collected after the starting period, from July 2012 to November 2014

	pH		N-NH ₄ ⁺ (mg L ⁻¹)		COD (mg L ⁻¹)	
	In	Out	Out	In	In	Out
Mean	5.8	6.7	40.0	49.4	1766	100
St. dev.	0.4	0.2	13.2	11.7	1281	67
Min	5.1	6.2	3.0	25.1	140	15
Max	6.8	7.3	77.6	74.9	15,730	484
# of samples	154	155	125	131	152	156

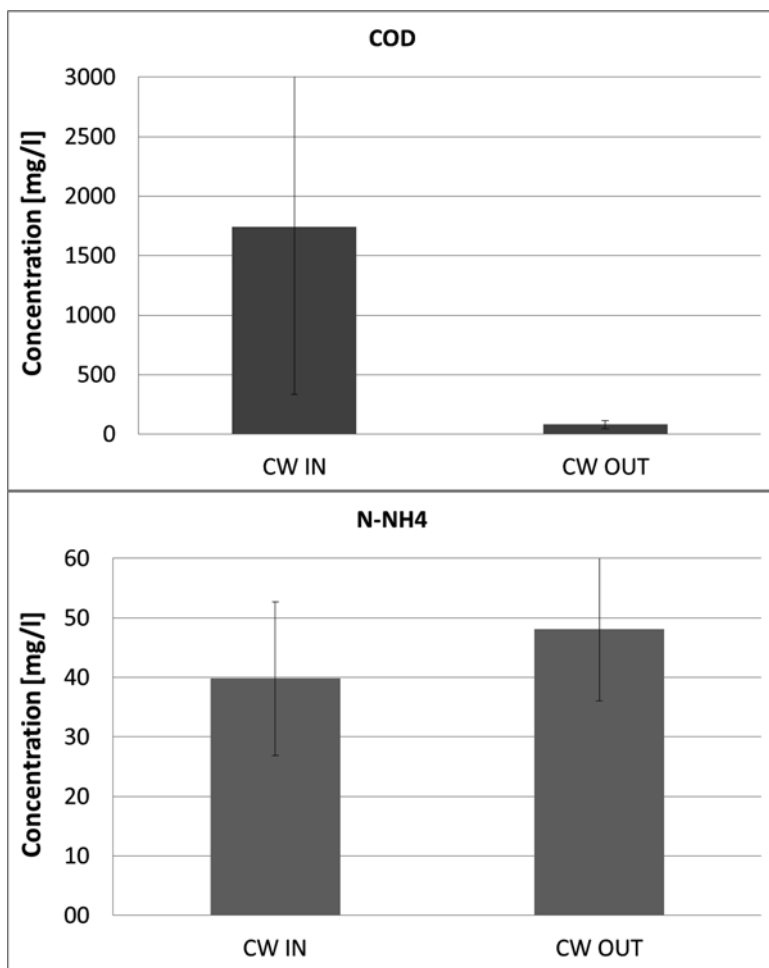


Fig. 10.6 Average and standard deviation of COD and N-NH₄⁺ concentrations at HF CW inflow and outflow

tion of the high organic nitrogen content within influent loads (Kadlec and Wallace 2009), determined by high concentrations of proteins in the dairy wastewater (Vymazal 2014). These results suggest that a second, aerobic stage to promote nitrification (e.g., vertical flow constructed wetland – VF CW – Kadlec and Wallace 2009) should be included in cases where there are limits on effluent nitrogen concentrations. A future analysis of influent and effluent total Kjeldhal nitrogen (TKN) is planned to confirm this hypothesis.

10.4 Conclusions

The organic load of dairy wastewater from Fattoria della Piana, Candidoni (Italy – 38°N, 15°E), has been successfully reduced by the HF CW treatment plant from 2012 to 2015. The wastewater effluent from the HF CW met the current Italian water quality limits except during the start-up phase and a particular management phase. This study demonstrates the validity of CW technology in treating wastewaters produced by dairies in Mediterranean regions and provides useful insights on how to deal with start-up and management phases for this particular application of the HF CW treatment plant.

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Chapter 11

Phosphorus Recycling from Waste, Dams and Wetlands Receiving Landfill Leachate – Long Term Monitoring in Norway

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Abstract Nutrients for food production are traditionally extracted from natural resources, most importantly as nitrogen from the air, and phosphorous from limited mineral resources. They can also be recovered and recycled from human waste products. There is generally a low P status in the world's soils, while Norwegian soils are rich in phosphorous. Most recyclable P is in human and animal waste products as wastewater and manure, but also municipal solid waste and more recently, organic waste contain a considerable amount of P that ideally can be utilized.

Keywords Recycling • Waste • Resources • Phosphorous

11.1 Introduction

Nutrients are necessary for food production. Nitrogen (N) is easily but not freely extracted from air, and phosphorus (P) from minerals. Within a few generations, the P demand might be greater than the production of mineral P making recycling a top priority (Hanserud 2014). The consumed P ends up in different waste products (Table 11.1). It seems during the years of ordinary landfilling, the amount of P from ordinary waste (80 kg per person annually), was a major fraction, ca. 15% of what was lost through wastewater. Also we have to separate between P recovery which include the retrieval of P to prevent downstream pollution, and P recycling, meaning making P available for (useful) plant growth. In this context, some queries arise: Is it practical to retrieve these fractions? Would the P be readily available for agricultural purposes? Could peat materials be “backwashed” or treated to re-mobilize P and allow capture/reuse of the P for agriculture? How tightly was the P bound and

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Table 11.1 P cycles in Norway^a and the World^b

	Norway (t/year)	%	World (Mt/year)	%
Added P total:	22,700		22,5	
Mineral fertilizer	8400	37	14	62
Manure	11,600	51	8	36
Other organic waste	2500	11	0	1
Precipitation	200	1	0,2	1
Removal total, % of added:	12,200	54	22	96
Plant uptake	11,000	90	12	56
Runoff	1200	10	9,5	44
Accumulated in soil:	10,500	46	1	4

^aHamilton et al. (2015), ^bCordell and White 2011

Table 11.2 Nutrients in waste fractions (mg/kg DM), concentration in leachate from mature waste (mg/l), treatment cost (euro/kg), nutrient content (kg pe⁻¹ year⁻¹) in food waste, municipal solid waste (MSW) and wastewater (WW)

	Food	MSW	Comp	Bio	Org	Conc	Cost	Food g	MSW g	WW g
Tot-N	2.8	0.83	2.5	2.2	1.5	205	19	50	332	
Tot-P	0.14	0.2	0.26	0.092	0.08	15	0!	2	80	548
TOC	50	40	40	36	46	546	0.13	885	16,000	
kg pe ⁻¹ year ⁻¹	17.7	400								
C/N	18	48	16	16	31					
N/P	35	42	10	24						

From Haarstad et al. 2005

Comp compost, *bio* biocell, *org* organic fraction of water, *conc* concentration in water

did “washout” or leakage frequently occur - just returning the P to waterways and increasing eutrophication? Bits of these questions have been addressed in various research - but looking at developing a “demonstrated” management option for long-term sequestration or, even better, reuse of the P is interesting. As shown in Table 11.1 the P cycle in Norway is dominated by an almost equal share of P losses and soil accumulation, while the figures for the World shows an almost total loss of P, and no accumulation, due to a gross under fertilization of P on a global scale.

Some data on the P content in waste material deposited in few selected Norwegian landfills is presented in Table 11.2, showing that with the current concentrations of P in landfill leachate it takes “forever” to leach out the P. The P content accumulated in landfills for municipal solid waste (MSW) over the last 5 decades constitutes almost 15% of the P content in wastewater.

Filter material from wetlands receiving wastewater and saturated with P in the laboratory adsorbed about 7500 mg P/kg DM (Jenssen et al. 2010). Most of it was extractable with ammonium lactate (pH 3.75) but only a small proportion of it could be dissolved in water. This outcome is in line with the results of the Chang and Jackson fractionation, where the majority of retained P in the saturated filter mate-

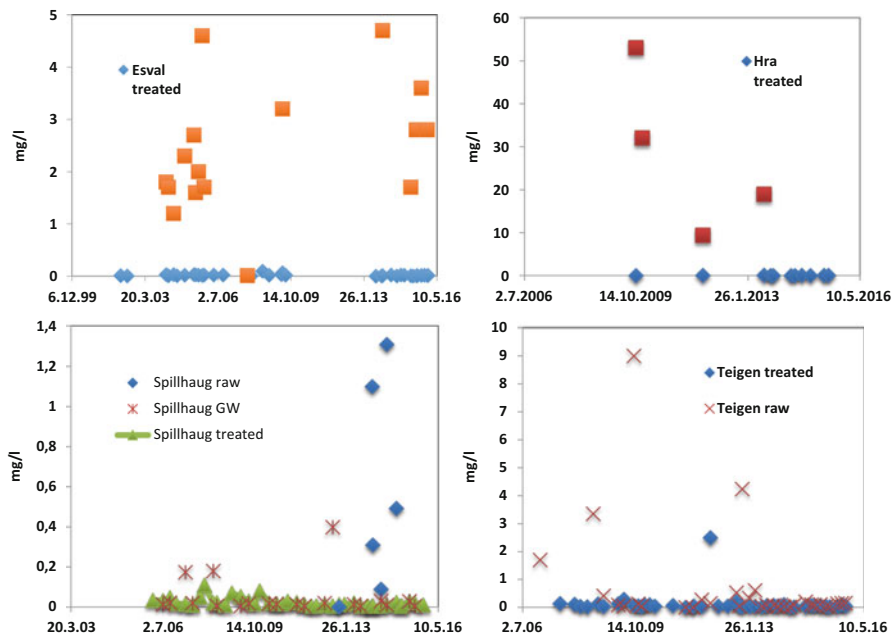


Fig. 11.1 Total-P (mg/l) from four Norwegian landfills

rial was recovered by H_2SO_4 extraction. Typically, P bound at high pH by materials rich in Ca, such as Filtralite-P, is in the form of Ca-phosphates, which are easily soluble in acids. As observed earlier (Zhu et al. 2003), the concentration of NaOH extractable P, representing Fe bound forms, was negligible in the Filtralite-P materials, as compared to soil. It is likely that during manufacturing of the material at high temperature (1200 °C, Zhu et al. 2003) the crystallinity of Fe minerals is increased, decreasing their P-sorption capacity. Full scale experiments (Søvik et al. 2007) with wetlands show P adsorption capacities below 1000 mg/kg, while potential adsorption reaches more than 15,000 mg/kg (shell sand).

Wetlands have also been implemented for treatment of landfill leachate. Figure 11.1 shows that the average P concentration in four Norwegian landfills are 2–4 mg/l, although one is much higher, up to 50 mg/l, indicating either dilution or that P is adsorbed in ordinary waste landfills compared to leachate from organic waste (Table 11.3).

Landfills are at the bottom of the waste hierarchy, where reuse or recycling and source separating are at the top. Nevertheless, in Europe landfilling kept its position until 2010, being replaced by 35% recycling and 60% incineration. The organic matter in landfills is to a larger degree dominated by fulvic, humic and organic acids, the former more prominent in fresh MSW waste. Both the organic fraction and sulphides are able to retain heavy metals and possibly also phosphorus in landfills (Aulin et al. 1997).

Table 11.3 Operation years, size of landfill in tons (Haarstad and Mæhlum 2008), total waste in kg, total P^a content (kg), year of supply as mineral fertilizer, and years required to empty P in landfill

Landfill	Open	Tot. waste	Tot-P	Years	Years 2
Esval	–1962	18,000,000	3,600,000	408	2.E+07
Spillhaug	1973–2009	90,000	18,000	2	4.3+05
Teigen	1941–1995	108,000	21,600	2	2.E+06
HRA	–1970	736,000	147,200	17	7.E+05

^aAssuming 20 g P/DM waste

11.2 Results and Discussion

Phosphorus is a plant available mineral P as (a) phosphate ions and (b) organic P, and not directly available as (c) adsorbed P, (d) secondary (clay, Al and Fe oxides) and (e) primary mineral P (mainly apatite) (Hyland et al. 2005). Estuaries and wetlands adsorb most of the P (Wang and Li 2010). Water flow typically delivers as little as 1–5 % of a plant's P demand, and the amount intercepted by growing roots is only half of that (Lambers et al. 1998, 2006). Fe, Al or Mn metal ions adsorb P favored by a low pH (McBride 1994). Lowering the pH or increasing the Al concentration makes P precipitate. In calcium rich material with high pH levels P precipitates as hardly water soluble Ca-compounds (Wang and Li 2010). Work in Nordic countries has shown relatively high accumulation of P in peat lands that received treated sewage, mostly in peat lands that had received effluent, which have been treated with ferric chloride or Al precipitation. As the effluents still contained residual iron or Al, the high accumulation of P was suggested to be due to the Fe or Al complexation processes and may not have been so much due to the inherent properties of the peat, itself.

Wetlands can treat effluent sewage in rural areas. Their P retention efficiency increase with wetland size (Braskerud et al. 2005). Investigations in Scandinavia showed a content of P in filters of up to 7500 mg/kg dry matter (Jenssen et al. 2010). The uptake of P in ryegrass pot experiments was, however, just 3 % of the P applied. In filters saturated with P as sodium phosphate, the average uptake increased to 24 %, compared to the uptake of 37 % when applied as fertilizer. The Fe content in wetland filters receiving leachate is high, between 40 and 400 g/kg DM. Wetland filters may also have a liming effect, which increase the availability of phosphorus in acidic soils. The adding of simple organic acids to soil reduced P retention in materials with high pH (Øgaard 1996). P precipitates as magnesium, ammonium or potassium phosphates have been shown to be as effective P sources as water-soluble magnesium calcium phosphate, showing that water solubility is not necessarily a good measure for plant P uptake (Johnston and Richards 2003). Wetlands that becomes anaerobic will release some of its retained P (Reddy et al. 2005).

Figure 11.1 shows that the average P concentration in four Norwegian landfills are 2–4 mg/l, although one is much higher, up to 50 mg/l, indicating either dilution or that P is adsorbed in ordinary waste landfills compared to leachate from organic waste (Table 11.3).

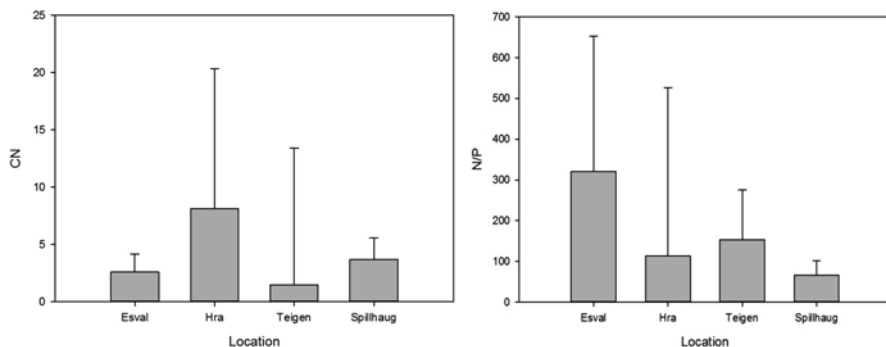


Fig. 11.2 C/N (left) and N/P ratio in leachate from Esväl (left), HRA, Spillhaug and Teigen landfills

For bacterial growth and metabolism influencing the N and C concentrations in water, there are optional C/N/P ratios depending on the organism's cellular demand. The C/N-ratio should not be too low (<5) in order to degrade ammonia to nitrate, according to Choi (2008). For biogas production, a high C/N ratio is preferred (30–300). Figure 11.2 shows that the measured levels in leachate in Norway are high in nitrogen and low in C and P for microbial metabolism. The landfill with fresh waste and organic waste disposal (HRA) shows the highest C/N-ratio, while the biggest landfill (Esväl) shows the lowest P content (Fig. 11.2).

11.3 Conclusions

There are large amounts of P in landfills (almost a fifth of wastewater, but stored!)
 The concentrations in leachate are high shortly after storage or from organic waste,
 later negligible

There is a surplus of N and shortage of C and P for biological treatment of
 leachate

The Fe content in wetlands treating leachate is high making P recycling more
 difficult

The P needs to be extracted during landfilling or shortly afterwards

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Chapter 12

Application of the NaWaTech Safety and O&M Planning Approach Re-Use Oriented Wastewater Treatment Lines at the Ordnance Factory Ambajhari, Nagpur, India

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Abstract Not only due to a lack of infrastructure such as treatment plants, but also because the majority of existing treatment plants are showing poor or very poor operating conditions and fail to meet their performance targets, India faces increasing water shortage and degradation of fresh water resources. The paper gives an overview on the methodology of a safety and O&M (operation and maintenance) planning approach developed and implemented for supporting sustainable long-term operation of wastewater treatment systems. The implementation of the methodology is shown for the pilot installation at Ordnance Factory Ambajhari, Nagpur, India. At this site, two treatment lines have been installed: Line 1 is designed for 100 m³/day and comprises anaerobic pre-treatment, a vertical upflow constructed wetland, followed by a disinfection step, line 2 designed for 8 m³/day is a 2-stage French Reed Bed system. The effluent of the French Reed Bed system is used for

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irrigation of a Short Rotation Plantation. The safety and O&M planning approach was used to identify critical O&M tasks, develop site-specific trainings of operators as well as a basis to develop the O&M manual and materials for operators (such as check-lists, etc.).

Keywords Constructed wetlands • French Reed Bed • Sanitation safety plan

12.1 Background

By 2050, India's population is projected to exceed 1.6 billion people with more than 50 % of them living in urban areas (UN 2015). Rapid urbanization and population growth and the discharge of untreated sewage already now results in water shortage, degradation of river, streams and aquifers and over exploitation of groundwater resources. This is not only due to a lack of infrastructure such as treatment plants, but also because the majority of existing treatment plants are showing poor or very poor operating conditions and fail to meet their performance targets (CPCB 2007). Efficient and effective operation and maintenance (O&M) is necessary to maximize pollution control efforts and to ensure the sustainability of wastewater management facilities (Murray and Drechsel 2011; USEPA 1974). Maximum health and environmental benefits are only achieved, if the facilities operate continuously and at full capacity, while complying with acceptable standards of quality and (Müllegger et al. 2012).

In general, reasons for poor O&M can be tracked back to both the design phase and the actual operation phase. During the system design phase, e.g. inaccurate technology selection neglecting local conditions and requirements for long-term operation can be a reason. However, critical problems are often also the result of institutional or managerial bottlenecks. Starting with lacking ownership and poor delegation of responsibilities, insufficient training of operating staff and laboratory staff also in combination with high staff rotations, deficient supervision of O&M due to missing procedures for reporting and recording of operational data and system problems; poor plant monitoring, respectively, poor communication of monitoring results – often in combination with a missing understanding of the processes involved in the treatment and their operational. Continuous overloading of systems, the lack of spare parts or consumables and the damage and theft of components are also undermining efficient operation and maintenance practices (CPCB 2007; Turon et al. 2007; Murray and Drechsel 2011; Müllegger et al. 2012). The root causes can be summarized as poor management, lack of planning and failure to cover costs to operate and maintain the systems.

In the DST-EU funded project NaWaTech (“Natural Water Systems and Treatment Technologies to cope with Water Shortages in Urbanised Areas in India”), the use of natural and compact wastewater treatment systems in urban India

is demonstrated. One aim of the NaWaTech project was to support long-term sustainability of the project treatment plants through systematic O&M management – taking into account that various technologies are implemented in quite different stakeholder contexts. Some of the sites are managed by well-experienced plant operators, for other sites people with no wastewater management background are involved in site operation. Thus, an O&M planning and management approach was developed and implemented that is flexible enough to be applied in various contexts. Moreover, a main interest of the NaWaTech project is to develop solutions, which are replicable. Therefore, a generic approach was required which is practice-oriented and simple and not require complex data collection and processing. The idea was to start to sensitize system designers for O&M and safety requirements in an early project stage as well as iteratively develop an appropriate planning methodology alongside.

Water and Sanitation Safety Planning promoted by WHO (Davison et al. 2005; Bartram et al. 2009; WHO 2015), is a risk-based management tool that recognizes the importance of institutionalizing good O&M practices (NIPHJ/IWA 2015). However, the miss-perception of the concept being an “additional” activity to existing managerial or operational tasks or the missing integration of planning results into day-to-day routines due to financial or institutional barriers can limit the success of the concept (Williams and Breach 2012). Since Sanitation Safety Planning is still rather new, lessons learnt from Water Safety Planning are taken as a basis for further developing Safety Planning as a means for O&M management in the sanitation sector. Through simplifying and adapting principles of WHO water and sanitation safety planning, preventative, technology-focused O&M schemes are developed and stakeholder involvement is initiated at each project site. Major objective of safety and O&M planning is to overcome shortcomings of conventional O&M for establishing effective and more informed and target-oriented O&M and troubleshooting routines.

12.2 Materials and Methods

12.2.1 Pilot Systems

In this paper, the implementation of the methodology is shown for a pilot site in Nagpur, India at the Ordnance Factory Ambajhari (in the following also referred to as “OFAJ site”), one of 40 ordnance factories in India. In the NaWaTech project, two treatment lines are implemented to treat 108 m³ domestic wastewater per day from the workers housing blocks for irrigation purposes on factory premises, respectively. Moreover, a Short Rotation Plantation pilot is implemented using treated effluent from the French Reed Bed system (treatment line 2, Molle et al. 2005) for irrigation. An overview of the treatment systems is given in Table 12.1.

Table 12.1 Overview treatment systems implemented at Ordnance Factory, Nagpur, India

Treatment line and capacity	Pre-treatment	Treatment steps	Disinfection	Re-use purpose
<i>Treatment Line 1</i> 100 m ³ /day	Screen	Primary clarifier, Up-flow Anaerobic Attached Growth Filter, Vertical upflow CW, Cascade Aeration, Dual Media Filter, Activated Carbon Filter + Sludge Drying Reed Bed	UV	Irrigation/gardening
<i>Treatment Line 2</i> 8 m ³ /day	Screen	2-stage French Reed Bed	–	Short rotation plantation

12.2.2 Safety and O&M Planning

The following steps were implemented as part of the safety and O&M planning process:

1. Establishment of the safety and O&M planning team

A core-team coordinating and initiating the safety and O&M planning was assembled with the NaWaTech project (including European and Indian project partners). After project hand-over, on-going tasks (e.g. revision of O&M calendars and routines, system monitoring etc.) will be taken over from representatives of Ordnance Factory management and the local contractor who will be operating the system.

2. Definition of system boundaries, stakeholder identification, and system description

System description as well as stakeholder identification was based on literature review and refined during field-visits and expert consultation with the engineers designing and supervising execution of the construction works. System flow diagrams and descriptions were elaborated following a sanitation system approach (Tilley et al. 2014).

One activity associated to this planning step, which was started but not formally completed during the project, is the assignment of O&M roles and responsibilities for each stakeholder in structured discussions (e.g.: in a workshop setting). The assignment of roles and responsibilities is relevant to seek consensus and transparency of potential future tasks of staff at different levels (operator level, supervisor level and management level), the system designers and system owners directly/indirectly relevant for system operation. This joint discussion is also necessary to foster a common understanding of how many different tasks are actually involved in

extensive, preventive system operation and maintenance. The underlying idea is to distinguish between

- (1) Preparatory tasks such as the development of SOPs or initial training of operators and supervisors, etc.;
- (2) the regular (“day-to-day”) O&M tasks such as the plant operation and inspection routines, regular maintenance and spare part management or the supervision of regular O&M;
- (3) trouble-shooting and major maintenance;
- (4) safety and performance monitoring including e.g.: taking samples, sample analysis, external audits, etc.;
- (5) resource and staff management involving legal responsibilities and resource allocation etc.

At Ordnance Factory, the existing hierarchical structures for construction and maintenance activities at the factory premises will build the formal framework for system operation. The actual O&M works will be carried out by an external company contracted for system construction and 1 year system operation. After 1 year, this contract will either be extended or opened to other specified contractors.

3. *Identification of hazards, risk assessment and prioritization*

Preliminary hazard identification was based on an extensive literature review on recorded operational experiences with the different system components as proposed by Hokstad et al. (2009). This first hazard list was carried out before system implementation and site visits were limited to the visit of future construction sites. However, the list was then iteratively discussed with experienced engineers and system designers as the project implementation proceeded. For preparation of hazard prevention and reduction, it was distinguished between hazards and hazardous events that are linked to the (1) design phase, (2) construction phase and (3) operation phase.

Each identified hazard was assessed using the semi-quantitative risk-scoring matrix shown in Table 12.2. Once the risks are assessed, risk ranking helps prioritizing risks and evaluates the “tolerability of a risk”. The ranking is based on calculating risk scores (R) which is calculated by

$$R = P * D * S, \quad (12.1)$$

where P=probability of the hazardous event, D=detectability of the hazard, and S=severity of the consequence of the hazard.

Results of the risk scoring were then grouped in two ranking classes indicating the criticality of the estimated risk. A risk is considered critical if $R > 7$, $P = 3$ or $S = 3$. This evaluation rules were adapted from Mayr et al. (2012) – with respect to the classes defined in the NaWaTech project.

Risk assessment was carried out iteratively. Starting with a draft assessment based on a literature study and followed by several rounds of expert discussions with system engineers during both project consortium workshops and individual sessions.

Table 12.2 Semi-quantitative risk score matrix for NaWaTech safety and O&M planning

Score	Probability (P) (of the hazardous event)	Detectability (D) (of the hazard)	Severity (S) (of the consequence of the hazard)
1	Occurs less than once in 5 year	Hazard is detected based on visual inspections	Will not result in major system degradation and will not produce system functional damage
2	Occurs once a month to once in 1 year	Hazard detection requires stepwise analysis (e.g. sampling required)	Will degrade system performance but can be counteracted or controlled without major damage
3	Occurs more often than once a month	No detection in normal operation; problem analysis is stepwise and complex	Will (severely) degrade system performance by substantial damage (component failure), interrupt system feeding, requiring immediate corrective action for system survival

In order to limit the subjectivity of the estimations – which is especially relevant for the probability-of-hazardous event – participants were asked to estimate on the assumption of having “unreliable or insufficient O&M practices/staff” in place. At the time of paper submission, a joint risk assessment with representatives of operating staff at the Ordnance Factory was not carried out yet.

4. Identification of risk prevention and risk reduction measures (control measures)

In the next step, for each identified hazard (hazardous event), control measures were defined – no matter if the risk assessment resulted in a “critical” or “non-critical risk”. A control measure is a preventive activity aiming at eliminating potential component failures, respectively, the probability of their occurrence.

5. Monitoring measures for surveying the efficacy of the control measures

Monitoring measures were defined to check whether the control measures, respectively, the treatment system, are operating as intended and to describe what to do if they are not (WHO 2015). Regular monitoring activities are defined to give simple and rapid feedback on how effectively the control measures are working so that corrective measures can be quickly initiated if needed. Similar as in case of the control measures, each activity was formulated in short descriptions, indicating the aim and content of the activity, responsibilities and implementation frequencies. Critical limits were defined which indicate if corrective measures (“trouble-shooting”) have to be initiated.

6. Formulation of a generic trouble shooting and communication routine

For each hazard, also some general actions to be taken when the “critical limit” is exceeded and/or a deviation from regular operation conditions is observed were defined. The description encompasses a general procedure only and aims at lowering

the time span between problem identification and remediation and to fast return to regular operation. This means to timely report failures/anomalies to the respective people in decision-making position (or also to inform affected users), but also to initiate measures like the bypassing of mal-functioning system units, repairs or substitution with spare parts. Besides that, these routines also involve the establishment of learning loops for involved staff, e.g. the revision of operator routines or the initiation of a refresher training. More specific and detailed emergency operation procedures are not described in the safety and O&M planning documents, but included in the O&M manuals.

12.3 Results and Discussion

In the following section, selected safety and O&M planning results are presented for two treatment lines implemented at Ordnance Factory. Table 12.3 shows selected examples of hazards and hazardous events and their risk assessment for different components (from different treatment lines), respectively, “general” hazards. The column “project phase” indicates whether the respective hazard has is controllable during the design, construction or operation phase. For hazards controlled during design or construction phase, no likelihoods are assessed.

A summary of the results for hazard identification and risk assessment for the two systems implemented at OFAJ site is provided in Table 12.4 illustrating the total number of hazards as well as the proportion of hazards ranked as critical distinguishing those with a high total risk rank (R), those with high probability (P) and those with high severities (S).

For the main line, out of this 106 total number of hazards, 39 were identified which can be controlled in the design, pre-construction or construction phase. For the French Reed Bed System, 29 hazards can be controlled before actually starting system operation. To provide an example for such control measures, Table 12.5 lists those identified for the French Reed Bed System.

When comparing the total number of hazards for the main treatment line and the French Reed Bed system, there is quite a considerable difference in the total number of hazards as well as critical hazards. This difference is not there, when taking into account the number of components involved. However, what is important to note is that for the French Reed Bed 17 out 24 identified critical hazards are associated with the two pumps used for system feeding and the feeding of the second stage French Reed Bed. For the main treatment line, 47 of the identified hazards associated with the operation phase that were ranked as critical are not only related to pump operation. Besides pump operation, main hazards were related to: sludge removal from the sump and the primary clarifier, sludge scraper maintenance, loading of the activated carbon column, UV disinfection unit maintenance (e.g. lamp cleaning, replacement of lamps etc.), power fluctuations and power cuts and maintenance of valves, staff dropout and rotations as well as spare-parts.

Table 12.3 Examples for hazard identification and risk assessments for selected components

Hazard identification and risk assessment							
Hazard ID	Project phase	Hazards and their effects		Risk assessment			
		<i>Hazardous event</i>	<i>Hazard</i>	<i>P</i>	<i>S</i>	<i>D</i>	<i>Risk</i>
Screen 1	3	Insufficient cleaning of the screen, build-up of solid material (e.g. plastic bags, hair, hygiene articles, paper, plastics etc.)	Reduction of flow; choking (clogging) and eventually surges in flow when piling screenings are removed	3	2	1	6
Manually operated valve 2	1,3	Valves aren't kept open long/frequent enough	Plant sickness/death when loading is insufficient; or resting intervals too long (not enough wastewater)	2	2	1	4
Primary clarifier 2	3	Infrequent removal of sludge (dropout of suspended solids/ high levels of suspended solids)	Floating sludge, Surface scum; creation of anaerobic conditions; Reduction of the effective volume available (hydraulic detention time affected). Non-adequate removal of solids, overloading (clogging) of subsequent treatment steps	2	2	2	8
Manually operated valve 3	3	Infrequent or no maintenance of valves	Valve breaks; not easy movable anymore	2	3	1	6
General 6	3	Temporary or long-term drop-out of operational staff	Knowledge management impeded; Loss of experience; temporary/ long-term understaffing	2	3	1	6

Project phases: (1) design (2) construction (3) operation;

Risk assessment: *P* Probability, *S* Severity, *D* Detectability

Table 12.4 Overview of main results for hazard identification and risk assessment for two technical and two natural treatment lines

Treatment line	Capacity	# components	# hazards	# critical hazards	Hazards with		
					$R \geq 7$	$P=3$	$S=3$
Main line	100 m ³ /day	16	106	47	25	0	22
French reed bed system	8 m ³ /day	8	66	24	12	0	12

Table 12.5 Overview of identified control measures to be implemented during design and construction phase of a French Reed Bed system

The selection of appropriate material for the screen and manhole covers in order to prevent corrosion
Selection of quality material for the filter material in order to lower proneness of clogging and assure targeted hydraulic characteristics within the bed
Cautious handling of the liner during installation to prevent leakage ^a
Protection and watering of the sprouts after plantations
Prevention of system components flooding through appropriate positioning of the facilities ^a
Protection of valuable system components (e.g. pumps, flow meters) against theft, e.g. through fencing
Protection of the filter beds against entering of residents e.g. through fencing ^a
Provision of substitute or alternate pumps for system feeding pumps
Provision of emergency power supply to continue system feeding during longer power cuts
Provision of circuit breakers and buffers to protect pumps against power fluctuations (high voltage specifically)
Selection of tight manhole covers

^aIn the case of this site, the reed beds were not excavated in the ground but installed in concrete basins, therefore, these control measure specifically were not valid

Considering the results in terms of critical control and monitoring measures to be integrated and prioritized in the day-to-day system operation, the following could be observed:

- For the French Reed Bed system, the maintenance of the screen and the pumps, protection against theft, power cuts and power fluctuations and the handling of staff rotations are the most critical O&M activities.
- For the main treatment line, the list of critical O&M concerns more components. In Table 12.6 all critical O&M measures identified for the main treatment line are listed. One significant outcome of the safety planning approach was to specifically distinguish and point out O&M activities carried out by the operator and hired personal such as electricians (=“control measures”) and those activities carried out by plant supervisors (“monitoring measures”) to assure the effectiveness of the control measures. This outcome is especially relevant for preparing “role”-specific O&M materials and training contents.

Trouble shooting routines identified for the two treatment lines underline the importance of reviewing O&M routines after major incidents – besides the actual technical measures for remediation. Thus, communication between the different levels of staff involved in system operation and maintenance is foreseen in all defined trouble-shooting routines. Only through systematic re-fresher trainings and joint discussions with operational staff, a learning process for system improvement can be established. Similar as in case of the control measures, trouble-shooting routines defined in the safety planning should be linked to more detailed step-by-step work descriptions contained in the O&M manuals provided by system designers and component manufacturers.

Table 12.6 List of critical O&M measures identified for the OFAJ main treatment line

Component+Hazard ID	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
Sump 1	1) Regular inspection of sump for any accumulation of suspended solids/sludge.	Unannounced inspection of sump, check for	Every 6 months
	2) Removal of accumulations and disposal on SDRB		
Primary Clarifier 1	1) Regular inspection of outlets	(a) accumulated sludge levels	Once in 6 months
	2) Regular clearing from build-ups or blockages	(b) operator routines/documentation (e.g. maintenance/inspection list, daily log)	
Primary Clarifier 2	Regular check of	Unannounced check of	Once in 6 months
	1) floating material in primary clarifier, skimming and removal of floating sludge.	(a) outlets for presence of build-ups/blockages	
	2) sludge scraper and associated working parts (including regular maintenance according to manufacture specifications) ¹	(b) check of operator routines/documentation (e.g. maintenance/inspection list, daily log)	
Primary Clarifier 1	1) Regular inspection of outlets	Unannounced check of	Once in 6 months
	2) Regular clearing from build-ups or blockages	(a) wastewater flow rate (checked with flow meter; running according to design specifications)	
		(b) sludge scraper (maintained according to manufactures details, time and duration of sludge scraping)	
		(c) check of operator routines/documentation (e.g. maintenance/inspection list, daily log)	

Primary Clarifier 4	Regular inspection and maintenance of sludge scraper and associated parts according to manufacture specifications	<p>Unannounced check of</p> <p>(a) water level</p> <p>(b) visible changes in wastewater colour, odour, floating debris etc.</p> <p>(c) maintenance schedule of scraper has been undertaken and documented</p> <p>(d) inspection of scraper – in working order (if possible without having to drain tank)</p> <p>(e) sludge feeding to SDRB shows normal flow conditions</p> <p>(f) check of operator routines/documentation (e.g. maintenance/inspection list, daily log)</p>	Once in 2 years
Upflow Anaerobic Filter 3	Regular monitoring of sludge volume in inflow	<p>Unannounced checks of</p> <p>1) sampling results</p> <p>2) operator routines/documentation (e.g. maintenance/inspection list, daily log)</p>	Once in 6 months
VF Upflow CW 15	<p>1) During construction: Hydraulic Testing after construction before commissioning</p> <p>2) During operation: Check level of bed free surface under the outlet pipeline bottom (indication of leakage) Visible signs of damage to civil works (cracks)</p>	<p>Unannounced check for</p> <p>1) Structural integrity</p> <p>2) Visual inspection for leakage, cracks, damage</p>	Once in 3 years

(continued)

Table 12.6 (continued)

	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
Component+Hazard ID Activated Carbon Columns 1	Change of activated carbon according to manufacturer/vendor specification	Unannounced check of 1) AC column 2) Influent and effluent concentration (influent: turbidity; effluent: turbidity, pH, BOD ₅ , TOC, COD) 3) Filtration rate (flow rate controlled by manual multiport valve) 4) Compliance with media replacement intervals 5) Operator routines/documentation (e.g. maintenance/inspection list, daily log)	Yearly
Component+Hazard ID	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
UV Disinfection 2	Regular cleaning and O&M of UV lamps, quartz sleeves or Teflon tubes	Unannounced check of: (a) Cleaning intervals; within manufacturer recommendations (b) Lamp condition (in good working order) (c) Effluent sampling: turbidity; disinfection effectiveness (d) check of operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 6 months
UV Disinfection 1	1) Regular check of lamp: in good working order; lamp intensity; hours of use 2) Replacement of lamp	Unannounced check of: (a) Hours of use; within manufacturer recommendations (b) Lamp condition (in good working order) (c) Effluent sampling: turbidity; disinfection effectiveness (d) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 1 year

UV Disinfection 7	1) Mini circuit breaker	Unannounced check of electricians	Once in 6 months
	2) Supervision of installations during construction		
	3) Regular inspection and maintenance of connections, cables, contacts		
Pumping Stations 5	Regular check of valves	Unannounced check of 1) valves, pumping system 2) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 1 year
Pumping Stations 10	Instruction for operators to close the manhole after all inspections	Unannounced check of 1) missing caps on manholes 2) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 2 weeks
Pumping Stations 2	Regular inspection of sludge level in pump shaft and removal of sludge through third party if necessary	Unannounced check of 1) sludge level in pump shaft 2) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 1 year
Pumping Stations 4 ^a	1) Selection of non-clogging pump 2) Regular inspection of feeding pump and check for accumulating material; clean/remove coarse accumulating material from pump	Unannounced check of pump shaft for (a) accumulation of coarse material (pump is clean and in good working order, maintained according to manufacturer details) (b) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 6 months
Pumping Stations 7 ^a	Regular check of time switch in pump shaft	Unannounced check of 1) functionality of timer 2) operator routines/documentation (e.g. maintenance/inspection list, daily log)	Once in 6 months

(continued)

Table 12.6 (continued)

	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
Component+Hazard ID	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
Component+Hazard ID	Control measure to be carried out by operator	Monitoring measures to be carried out by supervisor	Minimum monitoring interval
General Potential Hazards 1	Preparatory training of staff; system for knowledge transfer during staff rotations; regular refresher courses	Regular feedback round with all involved personnel with general remarks on plant operation	Once per year
General Potential Hazards 2	Plan for replacement staff/substitutes; protocols/routines for short-term substitutes/long-term-hand-over	Regular feedback round with all involved personnel with general remarks on plant operation	Once per year
General Potential Hazards 3	Spare part and consumables management: list of all manufactures or dealers of various equipment used in the plant storage of consumables (e.g. chlorine) + tools for day-to-day operations (brooms, protection gear, etc.) line out responsibilities for repairs etc. plan for troubleshooting solutions (some spare parts may have long delivery times)	Unannounced check of spare part management; cross check spare part and supply list with storage	Yearly
General Potential Hazards 4	Protection of vulnerable parts against theft (e.g. pumps, flow-meters, UV lamps, other valuable material)	Unannounced inventory and check of integrity of protection measures	Monthly
General Potential Hazards 5	Regular maintenance of alternate power supply	Unannounced check of generators	Yearly

^aonly for system feeding pump

Table 12.7 List of outcomes from safety and O&M planning

Safety and O&M planning outcomes
Design adaptations based on hazard identification in discussions with designers
Fostering O&M roles and responsibilities in stakeholder workshops
Prioritization of critical control and monitoring measures to be integrated in O&M work plans for operators and supervisors/skilled and unskilled workers
Input for technical trainings: focus topics, cause-effect chains
Development of or adaptation of O&M manuals and supporting materials
Identification of operator-friendly O&M materials and tools
Introduction of failure-reporting documentation

Besides at OFAJ site, safety and O&M planning was also carried out at three other NaWaTech project sites where further ten treatment lines have been or are about to be implemented. Table 12.7 provides a general list of outcomes that were obtained from safety and O&M planning at these different sites. One of the first visible outcomes of safety and O&M planning, is to take into account O&M requirements already in the early design phase and discuss modifications of the original design of the plants where needed. The second kind of outcomes are the definition and assignment of an agreement on roles and responsibilities of operators and supervisors for O&M during stakeholder workshops. Additionally, prioritizing critical control and monitoring measures to be integrated in O&M work plans, respectively, can be achieved and thus prioritizing them during technical trainings. For operator and supervisor technical trainings before plant hand-over, also pointing out of cause-chain can be derived. The differentiation of activities relevant for operators and supervisors and for skilled and unskilled workers, respectively, can help designing operator-friendly O&M materials including checklists and workplans for operators and supervisors, respectively. Finally, creating awareness for O&M from the beginning and clear responsibilities can support budgeting for O&M.

12.4 Conclusions

Since both systems are considered “natural” treatment systems there are some aspects, which are in common for both systems (in comparison to more technical options), such as:

- The need for regular and reliable O&M of the screen to prevent coarse materials from entering the systems, as well as the considerable number of “bottle-necks” related to pump operation.
- Thus, a reasonable number of critical hazards can be eliminated when designing/ positioning the treatment lines in a setting that allows substituting the system feeding pump or other pumps through gravity flow.

- Also it has to be noted that even though natural treatment systems require less skilled O&M workers, the training of the involved staff (operators and supervisors), the preparation of knowledge transfer and the organization of re-fresher trainings, but also the planning for staff rotations are also critical control measures as well as the preparation of spare part management.

However, probably the most obvious benefit of the safety and O&M planning activities can be seen in the initiation of a discussion process on O&M requirements throughout the implementation of a project – ideally, starting already during the design phase. Especially, when starting safety planning discussions already before system implementation, the process not only requires time but also the willingness to identify system weaknesses. To facilitate this process, hazard lists, proposed control and monitoring measures and trouble-shooting routines elaborated during the NaWaTech project can be used as a baseline and adapted to a particular project application.

Safety and O&M planning outcomes can support illustrating problem causes, operational effects, and hereby helping putting the focus on bottlenecks for system integrity. This can be of particular interest for a sound preparation of operator and supervisor trainings to foster their system understanding. Moreover, work scheduling and resource allocation is facilitated through the identification of high-priority O&M tasks. The differentiation of regular operator tasks from the ones of the supervisor, and indicating minimum intervals for critical monitoring activities helps to better line out roles and responsibilities for regular O&M, spare part management and system supervision.

An issue raised several times during stakeholder interventions, was the need for the provision of user-friendly O&M materials. Also OFAJ site staff, which has already experience in operating the wastewater treatment plant of the production facility, underlined the importance of materials which are easy to understand and practical for use. This also refers to aspects such as language or comprehensiveness for staff (e.g. using visual illustrations), which are not trained engineers or have some other professional background in wastewater management. It was emphasized that there is a need for checklists and workplans for operators and supervisors, which give them an overview of all regular O&M tasks as well as help in using the extensive O&M manual provided by the system designers. Safety and O&M planning can help in developing check-list for operators and supervisors, or review and extend existing materials included in O&M manuals. Moreover, in the stakeholder discussions during the planning process, formats (e.g. language, illustrations) appropriate for the staff involved in operation can be jointly elaborated to address their particular needs.

In November 2015, the treatment systems were still under construction. Outcomes of the safety planning presented in this outline paper are considered as preliminary results based on expert ranking and are yet to be validated with local O&M personnel and supervisors once the systems are taken into operation by the end of 2015.

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Chapter 13

Clogging Measurement, Dissolved Oxygen and Temperature Control in a Wetland Through the Development of an Autonomous Reed Bed Installation (ARBI)

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Abstract The suitability of different magnetic resonance sensor types, based on magnet arrangement, for the detection of different solids loadings in wetland sludges were explored. Unilateral and borehole design sensors were investigated. A development of the Helmholtz-style borehole design was selected due to its simplicity of design, low cost and large field of detection. The effect of aerating and heating a pilot wetland on organic matter and $\text{NH}_4\text{-N}$ reduction was investigated. Aeration improved performance across a range of oxygen concentrations down to 0.5 ppm below which little benefit was seen. No significant improvement was seen with heating between control and experimental bed. This is thought to be due to seasonal effects raising the temperature of the control to within 10 °C of the experimental bed, a difference thought not to be significant at the measured temperature range, (15–25 °C). Temperature driven hydrolysis of embedded organics may also be causing increased soluble COD outlet loads. We have demonstrated a Helmholtz like NMR sensor can detect differences in solids loading when buried in a gravel mix. Aeration can significantly increase treatment performances by 56% and 69% for COD and $\text{NH}_4\text{-N}$ reduction respectively at an oxygen tension of 5 ppm.

Keywords Aerated wetland • Clogging • Feedback • Sensor • Magnetic resonance • Temperature

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13.1 Introduction

13.1.1 General

The use of wetland waste water treatment systems is increasing as the demand for robust, low maintenance, sustainable technologies rises across both the municipal and industrial sectors. Concomitant end-user driven developments in the design and operation of the technology have been directed at key factors associated with performance and longevity including oxygen transfer rates and clogging.

Significant performance improvement has been seen with the introduction of direct aeration such as Forced Bed Aeration™, (Oullet-Plamondon et al. 2006; Nivala et al. 2007, 2013) though the addition of feedback control systems also linked to similar temperature control systems would help to optimise performance and minimise power consumption.

The relationship between areal organic load and clogging rates is becoming better understood (Nivala et al. 2013) and the use of different effluent delivery point aspects, horizontal/vertical, width/length and varying these with time is being explored generally. The ability to monitor clogging *in situ* would improve our understanding of the relationship between clogging and performance and enable improved prediction and control of the clogging process. This would allow operators to maximise the efficiency of their systems as well as budget for and control refurbishment cycles.

A consortium of research institutes and SMEs are involved in a 2 year study to develop and trial a modular wetland system ARBI (Autonomous Reed Bed Installation) capable of measuring and monitoring clogging factors, and controlling environmental conditions including, aeration, temperature and effluent delivery point.

The objective of the project is to construct and design a working wetland treatment system which offers treatment performance based feedback control of aeration and heating systems and a prototype magnetic resonance (MR) sensor which can detect differences in levels of clogging and generate a control signal to be interpreted by the heating and aeration systems.

Water quality monitoring sensors and MR clogging sensors will monitor the reed bed and adjust parameters including temperature, aeration and effluent delivery point to optimise treatment and minimise clogging. The development of algorithms to ‘program’ the operation of the clogging control system will be an important area of future work but is outside of this project scope. The modular nature of ARBI will allow easy extension of treatment capability and replacement of clogged sections as well as providing a vehicle to take the system to market.

13.1.2 *MR Probes for Clogging*

Magnetic resonance (MR) is a non-invasive technique best known for its application in medical imaging (Johnston et al. 2006). MR can be used to measure the properties of non-medical samples using a fixed magnetic field and radio frequency (RF) coil which transmits a sequence of high power pulses and collects a signal from the sample. This technique can be applied in a variety of ways, however this work focusses on MR relaxation measurements. The relaxation measurements probe the decay of the energy transferred to the sample which can provide information about the sample and its environment. One of the most basic sequences for obtaining a relaxation value (in our case T_2^{eff}) is a CPMG sequence (Meiboom and Gill 1958), which was employed in this work. Another relaxation parameter, T_1 , can be recorded using a series of CPMG measurements as described elsewhere (Morris et al. 2011).

It has been demonstrated in a laboratory environment that the MR relaxation times, T_1 and T_2^{eff} , are sensitive to the levels of clogging in a water sample (Morris et al. 2011; Shamim et al. 2013). There is potential that, using MR probes, the relative proportions of free water, suspended solids, biofilms and plant matter can be detected in wetlands.

MR sensors can be produced in a near infinite number of designs depending on the arrangement of the magnets and therefore can be used to create magnetic fields of different strengths and orientations (between about 0.2 and 0.5 Tesla in this work). The magnets also required a tuned RF circuit that resonates at a frequency corresponding to the field strength of the magnet. The initial MR sensor work undertaken for ARBI, as outlined below, explored the suitability of different MR sensors for the application of being embedded into wetlands to monitor clog levels.

Multiple magnet configurations have been investigated for the MR probe. These have included an Earth's field NMR (EFNMR) system (Packard and Varian 1954) which uses the Earth's magnetic field instead of a permanent magnet, a four-magnet surface sensor (similar to a sensor using the stray field from a Halbach array (Chang et al. 2006), a Helmholtz-style permanent magnet arrangement (Morris et al. 2011), and a simple bar magnet design (Blümich et al. 2002). The use of the designs discussed above have either previously been disseminated in the literature or are about to be published in regards to their applications in reed beds (Morris et al. 2011; Hill-Casey et al. 2014; Hughes-Riley et al. 2014a, b, c, d, 2016). This work will briefly overview this enterprise.

13.1.3 *Heating, Aeration and Step Feeding*

The aeration of reed bed treatment systems to enhance performance has been studied widely as have the effects of temperature. Active heating to enhance performance has not received so much attention but the effects of both on clogging may be multifaceted and complex.

Aeration fuels aerobic microbial activity and biomass production potentially enhancing clogging but the physical action of aeration bubbles may scour solids/biofilms from media thus reducing clogging. Similarly, rising temperatures may enhance temperature sensitive microbial activity, creating biomass and increasing clogging, but may also enhance the hydrolysis of organic matter converting it from solid matter to soluble matter (García et al. 2010) effectively de-clogging systems.

Changing the point at which effluent is applied to a wetland treatment system should help in spreading the development of biomass across the whole of the system rather than just at the dosing point. We would expect this therefore to increase heterogeneity in terms of hydraulic conductivity through a wetland system reducing the tendency for effluent short circuiting. This could maximise the use of the whole bed for longer periods effectively extending its performance life.

The work outlined below was undertaken by the GEMMA-Group of Environmental Engineering and Microbiology, Universitat Politècnica de Catalunya-BarcelonaTech (UPC) and provides an initial investigation on the impact of aeration, heating and, to a limited extent, step feeding on treatment wetland performance.

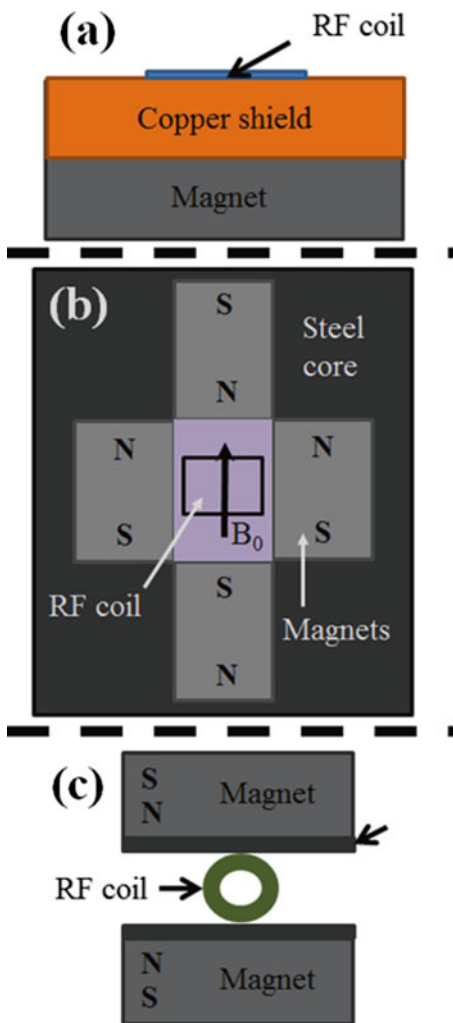
13.2 Methods

13.2.1 *Development of MR Sensor for Clog State Measurements*

Details of the construction and design of the sensors have been discussed in full detail elsewhere (Hill-Casey et al. 2014; Hughes-Riley et al. 2014a, b, c, d, 2016), but for completeness a diagram showing the basic design of each sensor has been included below (Fig. 13.1). It can be seen that there are two principle configurations. A bore-hole design where reed bed material has to fill the inside of the sensor (the Helmholtz probe) whilst the other two are unilateral, or single-sided, devices which probe a volume some distance above the sensor surface (the four-magnet surface sensor, the bar-magnet sensor).

The data presented in this work was performed using three of the Helmholtz-style sensors (Fig. 13.1c). Each was built using two N42 neodymium magnets (diameter=35 mm, height=20 mm; HKCM GmbH, Eckernförde, Germany), capped with steel disks, and separated by 20 mm to give central magnetic field strengths with corresponding proton resonant frequencies at 11.6 MHz, 11.4 MHz and 11.9 MHz. A parallel-series tuning board (using two 5.5–50 pF variable capacitors; Johanson Manufacturing, Boonton, USA) was attached to an enamelled wire solenoid (Rowan Cable Products Ltd., Potters Bar, UK) via an inductive tuning element to complete a resonant circuit for the transmission and receiving of RF signals.

Fig. 13.1 Schematics, (not to scale) of the sensors explored in this work. **(a)** A simple bar magnet sensor: A pair of 75 mm diameter neodymium magnets was combined to provide the magnetic field. The coil is placed atop the magnet. A copper shield is used to minimize external RF interference. **(b)** Four-magnet surface sensor: Neodymium magnets are arranged as shown to produce a uniform field across the surface of the magnet gap. The RF coil was placed in this gap. Steel was used as a base to the magnet arrangement to improve the magnetic field homogeneity. **(c)** Helmholtz sensor: Two magnets are separated by a gap generating a uniform magnetic field where an RF coil is placed



Magnetic resonance experiments were conducted using a CPMG sequence where 128 echoes (echo time = 150 μ s) were summed to provide a signal intensity for each repetition time explored. Different repetition times between 25 ms and 15 s were used and were plotted against the signal intensity. This was monoexponentially fit using IGOR Pro (Wavemetrics, OR, USA) allowing for the relaxation time T_1 to be extracted. All experiments were carried out using a Kea² spectrometer (Magritek, Wellington, New Zealand).

The three MR probes were embedded into wetlands maintained by ARM Ltd (Rugeley, UK). Probes were embedded at locations that were believed to have different levels of clogging.

Table 13.1 Technical specifications of the experimental pilot plant at UPC

Parameter	Value
Dimensions (W×L×H)	55.2 cm×70.5 cm×35 cm
Water level	30 cm
Surface	0.39 m ²
Loading rate (design)	63 l/d
Hydraulic retention time	0.8 d
Organic loading rate	60 g COD/m ² •d
Ammonium loading rate	5 g NH ₄ ⁺ -N/m ² •d

13.2.2 *Effects of Aeration and Heating on Treatment Wetland Performance*

Aeration and heating experiments were carried out in a pilot scale reed beds consisting of two horizontal subsurface flows constructed wetlands with the specifications outlined in Table 13.1.

One of the beds (the experimental wetland) was used to investigate aeration, heating and step feeding, the other wetland (control wetland) operated under normal saturated conditions.

13.2.3 *Aeration*

To test aeration effects the experimental wetland was equipped with an aeration system consisting of a pierced resin pipe of approximately 50 cm rolled at the bottom of the wetland at its central zone. The aeration roll occupied a surface of 0.07 m², which was about 20% of the total bed surface. Air was injected by means of an air pump at a flow rate of 720 l/h. Dissolved oxygen within the aerated and non-aerated bed were continuously monitored by means of a dissolved oxygen probe (CS512 Oxyguard Type III, Campbell Scientific Inc., USA) connected to a data logger (CR1000, Campbell Scientific Inc., USA).

At the beginning of the experiment air was continuously injected (24 h/d), reaching oxygen concentrations within the bulk liquid of approximately 5 mg/L. After 9 days the oxygen concentration within the bed was set at 3 mg/L by means of a control program in the data logger (control Deadbond version 2.5).

The oxygen concentration within the bed was successively reduced to 1.5, 0.5 and 0.2 mg O₂/L in order to test the treatment performances at different aeration rates. Each configuration was tested for 2 weeks and power consumption for each set point calculated. Water samples were collected twice a week from the inlet and from the outlet of the two beds and immediately analysed for Chemical Oxygen Demand (COD) and ammonium according to the *Standard Methods for Wastewater*

Treatment, (APHA-AWWA-WPCF 2001 APHA-AWWA-WPCF Standard Methods for the Examination of Water and Wastewater (twentieth ed.) American Public Health Association, Washington DC (2001)).

13.2.4 Heating

The experimental wetland was heated by means of a heat pipe embedded in the aeration system, set at 21 °C. Temperature was monitored by means of a temperature probe (Model 107, Campbell Scientific Inc., USA) located within both reed beds. Water samples were collected twice a week from the inlet and outlet of the two beds and analysed for Chemical Oxygen Demand (COD) and ammonium according to the *Standard Methods for Wastewater Treatment*.

13.2.5 Step Feeding

The feeding position was moved from the inlet front (feeding position 1) to the middle part of the bed (feeding position 2). Water samples were collected twice a week from the inlet and outlet of the two beds. During part of this period heating and aeration were run simultaneously in the experimental bed. Aeration was controlled to hold an oxygen tension of 0.5 mg/l determined from optimisation in the earlier aeration studies. The temperature in the experimental and control beds were 25 °C and 15–20 °C respectively.

13.3 Results

13.3.1 Magnetic Resonance Sensors

As outlined earlier, various MR sensors have been developed for use with the ARBI unit. An intention of the project was to select the most successful design after a trial period and bring this forward for implementation into an active reed bed.

13.3.2 Unilateral Surface Sensors

Unilateral surface sensors have one main advantage over a bore-hole design sensor which is that they cannot become completely clogged with material (such as a reed rhizomes) making the sensor inoperable as might happen with an enclosed design. Unfortunately, these sensors are only capable of interrogating a small region above

the coil surface. This becomes increasingly problematic when the sensor is water-proofed as this process will remove part of the already small sensitive region.

A simple bar-magnet design (Blümich et al. 2002) was investigated as this was the most simple magnet arrangement available for a sensor design. Unfortunately, while providing a strong signal, an extremely short T_2^{eff} ($309 \pm 60 \mu\text{s}$ on a thin wetland sample) led to a T_1 insensitivity to the clog state. This study was covered in more detail elsewhere (Hughes-Riley et al. 2014c).

The four-magnet surface sensor proved more promising. A sensor of this configuration showed differences in the T_2^{eff} as well as T_1 sensitivity (Hughes-Riley et al. 2014b, c).

13.3.3 *Helmholtz-Style Permanent Magnet Arrangement*

The Helmholtz-style permanent magnet arrangement was the first design used for this application (Morris et al. 2011). For ARBI two variations on this design have been explored. A small unit (using 35 mm diameter magnets) similar to Morris et al.'s earlier design and a larger unit (using 75 mm diameter magnets), allowing for a larger sample volume to be explored. The later design was able to collect signal however the signal intensity was inferior to that of the smaller unit. Additionally, under normal laboratory conditions ($\sim 20^\circ\text{C}$) the entire region within the solenoid of the larger sensor was not able to collect MR data. Given the favourable signal intensity of the smaller Helmholtz-style sensor further effort has been expended to ascertain the viability of the design for long term deployment into a wetland. One important area that has been explored was how temperature variations effected the probes operation. This was an important factor as magnetic field strength is known to change with temperature significantly (Clegg et al. 1991) and large temperature changes may have significantly affected the signal strength acquired from the sensor. Preliminary work showed a major reduction in the signal intensity collected by the probe as it became warmer or cooler than standard laboratory temperature (Hughes-Riley et al. 2014a). Later work, with a slightly improved sensor, has shown a far less extreme effect, with the signal intensity only reducing by a factor of two at the extremes of the temperature range that the probe would likely encounter while in a wetland in Europe Hughes-Riley et al. (2016).

A water-tightened sensor of this design has been used to collect T_1 measurements in a test wetland (Hughes-Riley et al. 2014d). Further to this, probes have been embedded into operational wetlands as part of a study into how the probes operate with seasonal variation and how the deviation in environmental factors over many months affect the T_1 values recorded from the probes, as well as the probes general operability. Figure 13.2 shows T_1 values collected from three operational probes after they had been embedded for 62 days.

Figure 13.2 shows a very clear distinction between clog levels where Probe A recorded a long T_1 time, Probe B a shorter T_1 time, and Probe C collected a short T_1

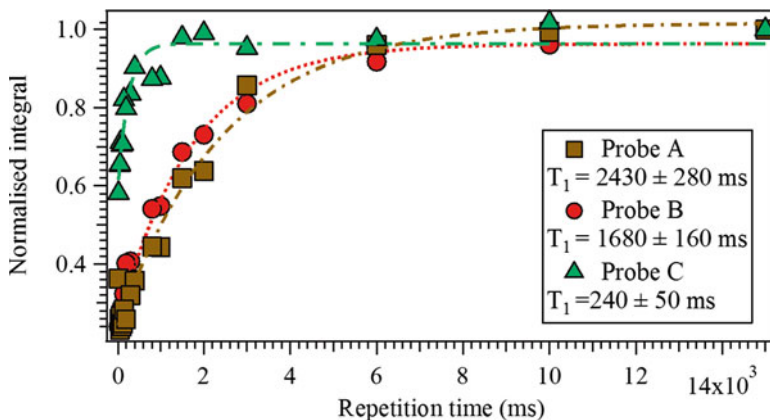


Fig. 13.2 T_1 measurements taken using Helmholtz-style MR sensors embedded into operational constructed wetlands. Each shows a very different clog level, with the short T_1 time recorded by Probe C representing a greater clogging than the long T_1 time recorded using Probe A

time; where a short T_1 corresponds to a greater level of clogging (Morris et al. 2011). Ultimately, it would be desirable to correlate T_1 times to the hydraulic conductivity within wetland samples, and this will be the focus of future work.

Long term monitoring of the embedded probes is on-going with preliminary results showing little change in the T_1 values over time. This would be expected for these already well-established beds where a change in clog level should only occur over a very long time scale.

The Helmholtz design was the simplest of all the MR sensor designs explored, it was the least expensive and had the largest sensitive volume of all the permanent magnet geometries considered. As a result, this made it the primary contender for use in the final ARBI unit.

13.3.4 Aerated System

The organic matter removal significantly increased in the aerated bed ($p < 0.05$). The COD inlet concentrations of about 400–300 mgO_2/L decreased to 160–140 mgO_2/L in the control bed, whereas lower values (about 70 mgO_2/L) were reached in the aerated bed, however the COD concentration in the outlet of the two beds were similar when the aeration set point was 0.2 ppm, shown in Fig. 13.3. The percentage increase in COD removal seen across the set point oxygen concentrations of 5, 3, 1.5, 0.5 and 0.2 mgO_2/L was 56%, 46%, 40%, 35%, and 3% respectively.

Differences between systems with and without aeration were more evident when looking at ammonium nitrogen ($\text{NH}_4 + \text{N}$) (Fig. 13.4). Ammonium nitrogen concentrations in the inlet ranged between 14 and 32 $\text{mgNH}_4 + \text{N}/\text{L}$ due to variation in the wastewater quality. The concentrations in the outlet were generally higher in the

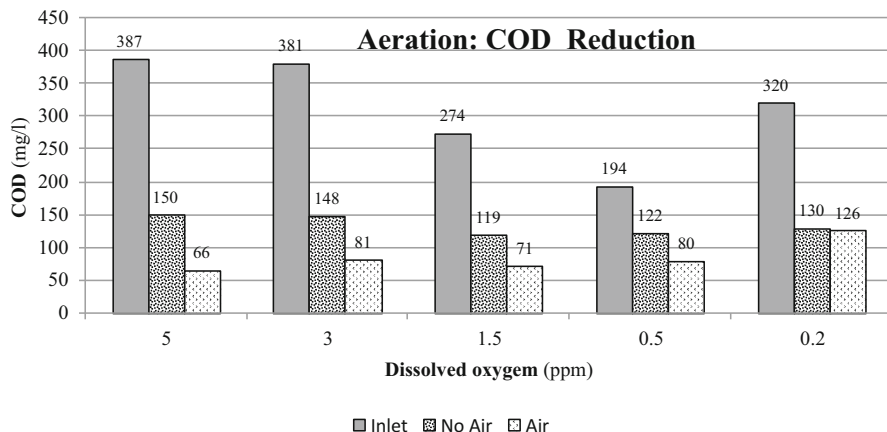


Fig. 13.3 COD concentration in the aerated and control beds at five different set point oxygen concentration levels

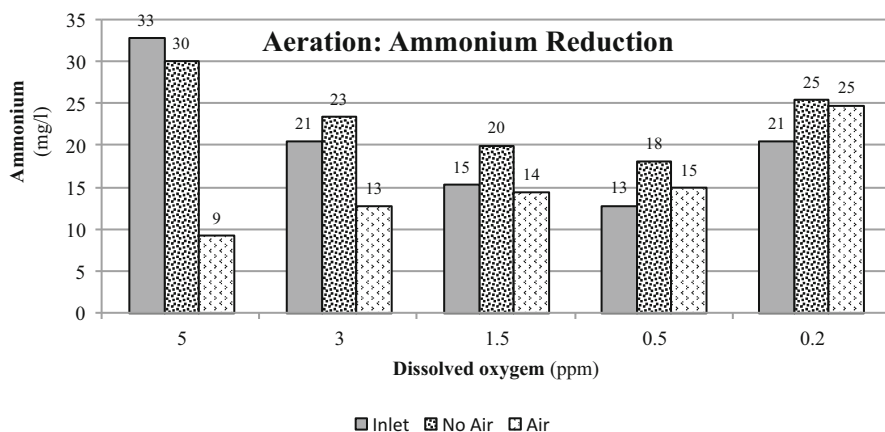


Fig. 13.4 Ammonium concentration in the aerated and control beds at five different set point oxygen concentration levels

control (18–29 mg $\text{NH}_4^+ - \text{N}/\text{L}$) than in the experimental beds (9–24 mg $\text{NH}_4^+ - \text{N}/\text{L}$) and on a percentage basis, aeration increased ammonium removal by 69%, 45%, 28%, 18% and 2% for the 5, 3, 1.5, 0.5 and 0.2 mg O_2/L set points respectively. As for COD, the difference between the aerated and the non-aerated bed was visible down to 0.5 ppm, whilst no differences (2%) were observed at 0.2 ppm mg O_2/L .

The optimal aeration rate, balancing performance effect and power consumption is 0.5 mg O_2/L , which provides a 35% and 18% increase in COD and $\text{NH}_4^+ - \text{N}$ removal respectively at a power consumption rate of 0.2 kWh/m³ water treated.

13.3.5 *Heated System*

Chemical Oxygen Demand (COD) concentrations ranging between 350 and 520 mgO₂/l in the inlet decreased to 150–180 mgO₂/l in both beds with the application of heating to the bed. The similar performance indicates that the difference in temperature between the two beds is not great enough to affect microbial degradation significantly at this particular temperature range.

Even if differences were not significant it was observed that initially (during the first week) that the COD removal was slightly higher in the control. This may be attributed to the hydrolysis of the embedded organic matter present within the granular media (García et al. 2010) which occurred to a greater extent in the higher temperature bed, effectively increasing the soluble COD load in the effluent indicating a decrease in the performance of the wetland. During the following week the tendency was changing and slightly better COD removal was observed within the experimental bed. This temperature based hydrolysis could have a positive effect through reduction of solids and hence de-clogging: as the clogging phenomenon is mostly attributed to the organic matter retention within the granular media.

13.3.6 *Step Feeding*

The impact of different feeding positions or changing the position of the influent flow can only realistically be seen with long term comparisons; thus no significant differences in COD removal were detected between the outlet concentrations of the two beds when the feeding position was changed. Moreover, no significant differences ($p > 0.05$) of COD removal were detected between the two feeding positions. Similarly with respect to ammonium nitrogen (NH₄ + -N), no significant differences ($p > 0.05$) were detected between beds nor between feeding positions.

The COD reduction results obtained whilst both heating and aeration were operated simultaneously indicated a good overall reduction but little difference in performance against the control this is likely to be due to the relatively low aeration level (0.5 mgO₂/L) and also suggests, when compared to the previous aeration results, that the impact of aeration on aerobic microbial degradation is greater at lower temperatures.

13.4 **Conclusions**

We have demonstrated in this work that a Helmholtz-like sensor with a comparably large sensitive volume can be used to monitor T₁ when buried in operational, established wetlands. Furthermore we have again demonstrated the sensitivity of this parameter to clogging. The large sensitive volume makes it most representative of the bed health on the whole which is highly desirable.

Aeration can significantly increase treatment performance, by 69 % for ammonia reduction and 56 % for COD reduction based on a dissolved oxygen level of 5 ppm. A balance of performance and electric consumption indicated the oxygen set point at 0.5 ppm as the best configuration. This configuration increased COD and ammonium removal by 35 % and 18 % respectively. The positive effects on treatment performance seen with aeration may be better in colder climates through offsetting of low temperature effects. Considering that the control already achieves the regulation thresholds, results suggest that wetland surface could be reduced by 18 % without affecting performances. Heating did not show a significant effect on COD and ammonium removal but this could be due to the already high temperature in the control bed (10–15 °C) which will be confirmed with further work in a cooler climate. However, the hydrolysis of organic matter caused by heating suggests that it could be applied to delay or reduce the level of particulate organic matter in the medium and, consequently, reduce the clogging phenomenon.

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Chapter 14

Constructed Wetlands Treating Municipal and Agricultural Wastewater – An Overview for Flanders, Belgium

Hannele Auvinen, Gijs Du Laing, Erik Meers, and Diederik P.L. Rousseau

Abstract Flanders is one of the most populated regions in Europe where agriculture is large-scale and intensive. It is therefore challenged by high nutrient loads which deteriorate the surface water quality through direct discharge from wastewater treatment plants and through runoff from agricultural fields. According to our survey, constructed wetlands in Flanders mainly used to treat domestic and agricultural wastewater in rural areas. The use of constructed wetlands as individual treatment systems has increased in the last years due to the obligation of wastewater treatment at remotely located households. In this overview, the performance of constructed wetlands treating municipal wastewater and manure treatment wastewater is analyzed. In general, the constructed wetlands studied were found to meet the limits set in the Flemish legislation for wastewater effluent quality. Many of the wetlands treating municipal wastewater are, however, dealing with high nitrogen loading which results in low total nitrogen removal efficiency. Also the phosphorus removal capacity in these wetlands was found to be limited. Therefore, these wetlands do not majorly decrease the nutrient discharge to the environment. The constructed wetlands treating manure treatment wastewater are, on the contrary, over-dimensioned and thus able to achieve excellent nutrient removal efficiency.

Keywords Nitrogen • Phosphorus • Efficiency • Domestic wastewater • Manure

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14.1 Introduction

14.1.1 Treatment of Municipal Wastewater

Flanders, the northern region of Belgium, has a total surface area of 13,522 km², and is inhabited by approximately 6.2 million people (Belgian Federal Government 2010). By the end of 2014, 86 % of the households in Flanders were connected to a sewer network, the connection rate being higher in urban areas than rural areas, and over 94 % of the collected wastewater was effectively treated (VMM 2015). For the other households, there has been uncertainty for many years whether or not they will still be connected or will have to take own responsibility in treating wastewater. This issue has been solved in the period 2006–2008 by the elaboration of zonation plans. For this, the territory has been divided into three zones: where a sewer network already exists, where it is planned to be built and where treatment of wastewater is the responsibility of the individual households.

The drinking water companies are responsible for treatment of wastewater collected in the sewer systems. They co-operate with Aquafin N.V. and municipalities which organize the collection and treatment in practice. In case a household is remotely located and no sewer system is available and planned to be built, an individual small-scale treatment system must be installed. These systems must be selected from a list of certified installations according to a Belgian standard BENOR which is based on European standard EN 12566/3 (Certipro 2015). This list contains constructed wetlands among others.

Small-scale treatment systems for the treatment of municipal wastewater only have to meet a COD standard of 125 mg/L and an influent load reduction of at least 90 %; a BOD₅ standard of 25 mg/L in combination with an influent load reduction of 75 %; and a TSS standard of 35 mg/L (500–2000 IE) or 60 mg/L (<500 IE) in combination with an influent load reduction of 70 %. For plant-based systems, there is even an exception in the sense that they are exempted from these standards in case the temperature drops below 5 °C, indicating an (unwarranted) lack of confidence in constructed wetlands (VLAREM 2016). There are no nutrient limitations for small-scale treatment systems treating municipal wastewater.

14.1.2 Treatment of Agricultural Wastewater

Farming in Flanders is large-scale and intensive, which is linked to high consumption of fertilizers (both inorganic and manure) and deterioration of soil and water quality (EU 2014). High concentrations of nitrates have been (and are still being) measured in groundwater and surface waters (MIRA 2014), and therefore, the whole of Flanders is classified as Nitrate Vulnerable Zone as described in the Nitrates Directive (EU, 1991^a). Tertiary treatment of agricultural wastewater originating from pig manure processing activities has developed in Flanders over the last decade to decrease the nutrient loading to the surface waters.

Meers et al. (2008) described for the first time the successful application of a pilot scale pilot installation of constructed wetlands, designed for the downstream processing of the liquid fraction coming from physical separation of manure (centrifuge, sieve band press or fan press) and pre-treated by biological processing (nitrification / denitrification). The pilot-scale installation was initially designed for 1000 t/year of pig manure processing down to dischargeable concentrations. After proof-of-principle at this pilot installation, the technique was commercially rolled out as the “Innova Manure” system. Currently multiple treatment wetlands have been constructed in Flanders, with a capacity treating of 70,000 t/year of animal manure. The wetland systems in actuality form a green buffer around industrial pig farming and manure processing facilities (nitrification/denitrification tanks) and perform additional ecoservices, such as establishing an enhanced biodiversity (Boets et al. 2011). The effluent standards which all manure treatment CWs need to meet are the same as those for municipal wastewater treatment systems of 2000–10,000 I.E.: 125 mg COD/L, 25 mg BOD₅/L, 15 mg TN/L, 2 TP mg/L and 35 mg TSS/L.

14.2 A Database on Constructed Wetlands in Flanders

14.2.1 Data Collection

An elaborate review on constructed wetlands in Flanders has been published about a decade ago (Rousseau et al. 2004) with a state-of-the art *anno* 2004. To create an updated database of constructed wetlands in Flanders, organizations active in wastewater treatment were contacted in the course of 2014. These included both government-related organizations (Aquafin N.V. and the Flemish Land Agency VLM) and private wetland constructors. The latter were found via online queries.

As a first step, data on the location, size and type of wetland and treated wastewater were collected. Secondly, data on treatment performance were collected. Because there is no monitoring obligation for such small treatment plants, data are quite scarce.

For this chapter we processed the data from the Aquafin operated wetlands (n=35) and from six manure treatment wetlands (“Innova Manure”). The Aquafin operated wetlands are monitored at least 12 times per year for common water parameters, i.e. chemical oxygen demand (COD), biochemical oxygen demand (BOD₅), ammonium (NH₄⁺), nitrate (NO₃⁻), nitrite (NO₂⁻), Kjeldahl nitrogen (KjN), total nitrogen (TN), total phosphorus (TP). These data with the flow data are publicly available at <http://geoloket.vmm.be/Geoviews/map.phtml>, which is regularly updated by the Flemish Environment Agency VMM. The “Innova Manure” wetlands are also monitored on a monthly basis. These systems are monitored by an external company (i.e. not the manure processing installation or pig farmer themselves) at several locations within the wetland system, from inlet to discharge. The data from the “Innova Manure” wetland systems were provided by the constructors.

These systems are monitored for ingoing and outgoing concentrations of COD, TN and TP.

14.2.2 Data Processing and Analysis

The raw water quality data from the year 2005 (or any later start-up of the wetland) until the end of 2014 (or prior shutdown of the wetland) was processed for the data analysis. The data in the Geoloket are based on time- or flow-dependent sampling and in some cases the samples are taken as grab samples. The samples at the manure treatment installations are grab samples.

Based on these data, the following characteristics were calculated as averages from all data points during the surveillance period:

- Removal efficiencies (%), based on influent and effluent concentrations
- Histogram represents the distribution of the probability for a sample to be below a certain value (e.g. the 90-percentile means that 90 % of all samples in the data-base have a concentration below the indicated concentration, 10 % have a concentration above the indicated concentration)
- Areal loading rates ($\text{g}/\text{m}^2/\text{year}$)
- Actual pollution load in population equivalents (pe), assuming 44 g BOD/pe/day and 10 g TN/pe/day (Aquafin 2013). For the manure treatment wetlands the pe was calculated based on nitrogen content and assuming 10 g TN/pe/day.
- Hydraulic retention time (HRT: d)
- Hydraulic loading rate (HLR: $\text{m}^3/\text{m}^2/\text{d}$)
- Denitrification potential, i.e. the amount of nitrogen that can be denitrified, was defined as the amount of kjeldahl nitrogen removed in the system plus nitrite and nitrate present in the influent (mg N/L). The denitrification efficiency was calculated based on the remaining effluent nitrite and nitrate concentrations relative to the denitrification potential (%).
 - E.g. Denitrification potential at the Deurle installation:
The average influent KjN concentration was 26.0 mg N/L and the sum of NO_3^- and NO_2^- was 1.2 mg N/L. In effluent there was still 21.8 mg KjN/L present. The denitrification potential is thus $(26.0-21.8) \text{ mg N/L} + 1.2 \text{ mg N/L} = 5.4 \text{ mg/L}$
 - E.g. Denitrification efficiency at the Deurle installation:
The average sum of nitrite and nitrate concentrations in the effluent was 0.43 mg N/L. The denitrification efficiency is thus $(5.4-0.43) \text{ mg N/L} / 5.4 \text{ mg N/L} = 92 \%$

The data obtained this way were further analyzed by using Excel (Microsoft 2010). To analyze the treatment efficiency of each type of wetland, a histogram was produced using the influent and effluent concentrations for each type of wetland per

component of interest (TN, TP). The dependence between the phosphorus removal efficiency and the age of the CW was studied by calculating the Pearson correlation factor (R^2) for the Aquafin operated wetlands, as an indicator of adsorption saturation.

14.3 Location, Number and Types of Constructed Wetlands

The database contains currently information on 406 constructed wetlands for wastewater treatment. More than half of these wetlands (56 %) are individual wastewater treatment systems for single families (Fig. 14.1, <10 %). Almost a fifth (18 %) of the wetlands is constructed to treat agricultural wastewater, and 14 % of the wetlands were built to treats wastewater from e.g. companies, schools and restaurants. Only 10 % of the wetlands are operated by Aquafin N.V, but they correspond to approximately 40 % of the total I.E. that is treated in constructed wetlands according to our database. The “Innova Manure” wetlands also discussed in this chapter constitute approximately 12 % of the total I.E. that is treated in constructed wetlands. The location of the Aquafin operated wetlands and the “Innova Manure” wetlands discussed in this chapter are illustrated on the maps in Figs. 14.2 and 14.3.

Currently, the most popular type of wetland is the vertical flow type (VF; Fig. 14.4). Their numbers have increased tremendously since 2004 when the previous overview on Flemish wetlands was published (Rousseau et al. 2004). In 2004, there were only 34 VF CWs in the database, now we found references to 287 VF CWs. The change is mainly caused by the popularity of VF CWs among the individual treatment systems which have been built in recent years to meet the discharge regulations set up by EU (1991^b). Free water surface flow (FWS) CWs are more popular than the horizontal sub-surface flow (HSSF) CWs. The HSSF CWs are not used

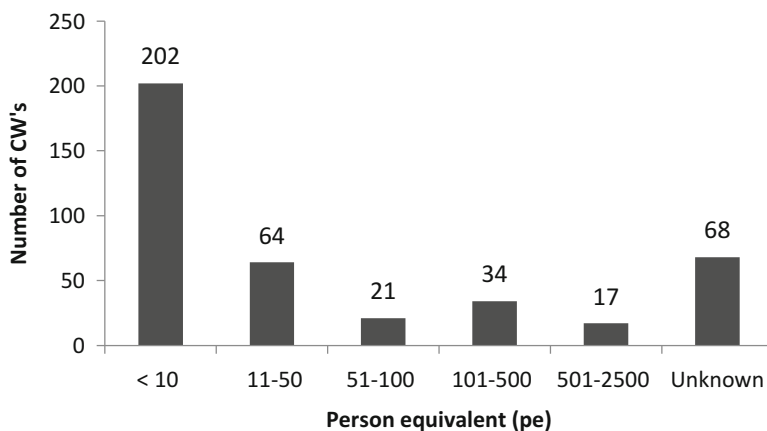


Fig. 14.1 Number of constructed wetlands in Flanders based on design person equivalent (pe)



Fig. 14.2 Location of constructed wetlands owned by Aquafin N.V. (■ = FWS; ● = VF; ● = HSSF; ● = VF-HSSF (combined); ● = SAF-SSF (tertiary); ● = RBC-SSF (tertiary)) (The interactive version of this map is available at: www.enbichem.ugent.be/FlandersWetlands ©GoogleMaps)



Fig. 14.3 Location of the “Innova Manure” wetlands (The interactive version of this map is available at: www.enbichem.ugent.be/FlandersWetlands ©GoogleMaps)

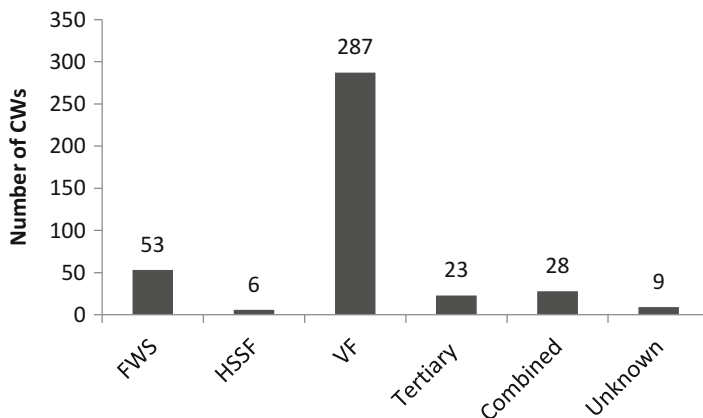


Fig. 14.4 Number of constructed wetlands in Flanders based on the wetland type

often as single stage wetlands but more often built as a tertiary treatment system after a rotating biological contactor or submerged aerated filter (Tertiary) or used in combination with vertical flow CWs (Combined). The number of unknown wetlands is caused by the incompleteness of some of the data acquired from newspaper articles and internet sites of the wetland contractors.

14.4 Removal of Nutrients from Municipal Wastewater

14.4.1 Free Water Surface Wetlands (FWS)

Only two of the free water surface wetlands are regularly and extensively monitored. Both are treating pre-settled municipal wastewater. In Deurle, the wastewater flows through 9 FWS CWs built in series, and in Latem through 13 FWS CWs. The design and operational characteristics of these installations can be found in Table 14.1.

It can be noticed that the actual loading in terms of inhabitant equivalent values based on BOD_5 and TN vary significantly. This can be due to the degradation of BOD_5 in the sewers and septic tanks. Septic tanks are still widely used at especially older households and it is thus possible that the organic matter degradation occurring in these tanks affects the influent composition at the WWTP. Nitrogen concentrations are not affected by presence of the septic tanks because no nitrification (and subsequent denitrification) occurs in these tanks. Some of the preliminary BOD_5 degradation is already taken into account in defining the inhabitant equivalents as 44 g BOD_5 /pe/d instead of 54 g BOD_5 /pe/d (Aquafin 2013), but it appears to be inadequate in the context of Flemish wastewaters. Furthermore, calculation of *pe* based on 12 g TN/pe/d has been shown to be more suitable for municipal wastewater in Flanders (Coppens et al. 2013).

Table 14.1 Design and operational characteristics of the FWS CWs

CW	Year of construction	Design capacity (pe)	BOD ₅ loading (pe) ^a	TN loading (pe) ^b	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
Deurle	1989	900	314	938	3060	3.4	0.11
Latem	1989	720	260	815	3000	4.2	0.13

^aBased on 44 g BOD₅/pe/d (Aquafin 2013)

^bBased on 10 g TN/pe/d (Aquafin 2013)

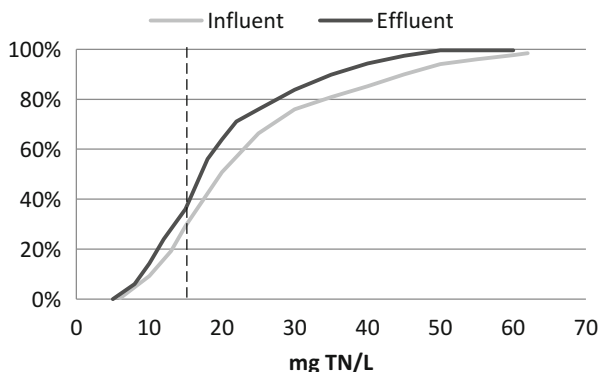


Fig. 14.5 The histogram for total nitrogen in the FWSs ($n_{\text{infl}}=250$; $n_{\text{eff}}=266$). The dashed line indicates the limit values set for effluent in the Flemish legislation VLAREM for installations >2000 pe

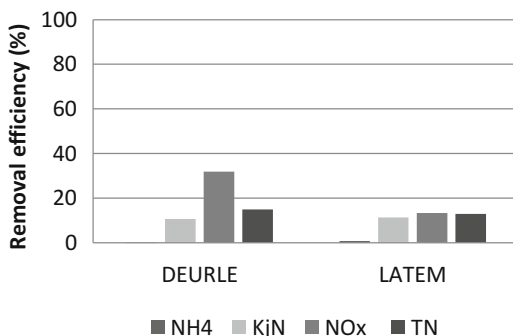
14.4.1.1 Nitrogen

The average TN concentrations in the influent are 21 ± 12 and 27 ± 14 mg TN/L for Latem and Deurle installations, respectively. The influent is thus quite diluted which is likely to be caused by the wide use of combined sewer systems. The influent total nitrogen content consists mainly of ammonium nitrogen (73%), the remainder being organic nitrogen.

From Fig. 14.5 it can be seen that TN removal in the FWS CWs is very poor. As mentioned in the introduction, there are no nutrient limitations for wastewater treatment plants smaller than 2000 pe in Flanders. When the data on total nitrogen are compared to the Flemish standard for larger installations (15 mg TN/L) it can be seen that only 36% of the effluent samples would comply. Actually, the poor nitrogen removal in the two FWS wetlands is also emphasized, as 29% of the influent samples already achieve the standard limit value.

It is clear that the nitrification efficiency is very low in these systems (Fig. 14.6). A reduction in NH_4^+ of only 1 ± 2 mg N/L is achieved in these systems. The low nitrification efficiency is likely due to high nitrogen loading. Indeed, the observed TN influent loading rates 600 ± 335 g TN/m²/year and 837 ± 381 g TN/m²/year for the Latem and the Deurle system, respectively, are high in comparison to values

Fig. 14.6 Removal efficiency of nitrogen species in the FWSs



summarized in e.g. Vymazal (2007) of 466 g TN/m²/year. In addition, the design flow rate (based on 150 L/pe/d), was met only in 35 % of the sampling events at the FWS installations. These high flow rates decrease the HRT, and this has been shown to affect nitrification efficiency (Toet et al. 2005).

It is logical to assume that the low nitrification efficiency also limited denitrification through low production of nitrite and nitrate. Denitrification potential was estimated by taking into account the ammonia oxidation that occurred in the system, producing nitrite and nitrate. The denitrification potential was very low, only 4.55 ± 1.13 mg N/L, and 92 % of the nitrate and nitrite present was removed in the FWS CWs.

14.4.1.2 Phosphorus

The removal of total phosphorus is very low in the concerned FWS CWs: average removal efficiencies were 5 % and 7 % for the Latem and Deurle installations, respectively. According to Fig. 14.7, only 40 % of the samples were under the Flemish standard (2 mg TP/L) for larger installations. However, this is a typical phenomenon in FWS wetlands where adsorption on the matrix is not possible. The phosphorus taken up by the plant will be released upon biodegradation of the plant biomass. Wallace and Knight (2006) stated that phosphorus removal in a FWS wetland is a result of accretion to sediments on the bottom of the wetland.

14.4.2 Vertical Flow Systems (VF)

Three of the VF systems in our database are regularly sampled. These VF CWs are used to treat pre-settled municipal wastewater. The Zemst-Larebeek and Sint-Niklaas Heimolen installations are composed of two parallel VF CWs. The Rillaar installation is the largest CW in Flanders, and it consists of four VF CWs operating in parallel. The design and operational characteristics of these installations can be found in Table 14.2. From this table, the high surface loading rate at the

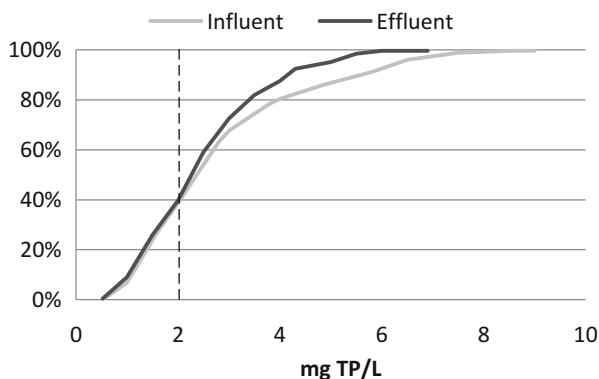


Fig. 14.7 The histogram for total phosphorus in the FWS wetlands ($n_{\text{infl}}=250$; $n_{\text{eff}}=266$). The dashed line indicates the limit values set for effluent in the Flemish legislation VLAREM for installations >2000 pe

Table 14.2 Design and operational characteristics of the VF CWs

CW	Year of construction	Design capacity (pe)	Actual BOD ₅ capacity (pe) ^a	Actual TN capacity (pe) ^b	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
Zemst-Larebeek	2000	423	463	1880	940	2.2	0.57
Rillaar	2009	1800	1170	1880	12,000	6.7	0.02
Sint-Niklaas Heimolen	2000	243	209	408	540	2.2	0.13

^aBased on 44 g BOD₅/pe/d (Aquafin 2013)

^bBased on 10 g TN/pe/d (Aquafin 2013)

Zemst-Larebeek installation jumps out. This is probably due to the combined sewers and the exceptionally high proportion of surface and/or infiltrated groundwater in the influent stream.

14.4.2.1 Nitrogen

The three VF CWs in the database, Zemst-Larebeek, Rillaar and Sint-Niklaas Heimolen CWs, receive influent which contains on average 35 ± 34 mg TN/L, 66 ± 25 mg TN/L and 58 ± 25 mg TN/L for, respectively. The TN at the Rillaar and Sint-Niklaas Heimolen installations consists mainly of NH₄⁺ nitrogen (78%). At the Zemst-Larebeek installation the proportion of NH₄⁺-N is slightly lower (58%) due to the presence of nitrite and especially nitrate in the influent (5.2 ± 4.4 mg N/L; 13%), which are most likely designated to agricultural runoff.

Compared to the FWS CWs, the VF CWs achieve a better total nitrogen removal. However, only 23% of the effluent samples would meet the Flemish standard 15 mg TN/L set for larger installations, while almost 8% of influent samples are already under this limit (Fig. 14.8).

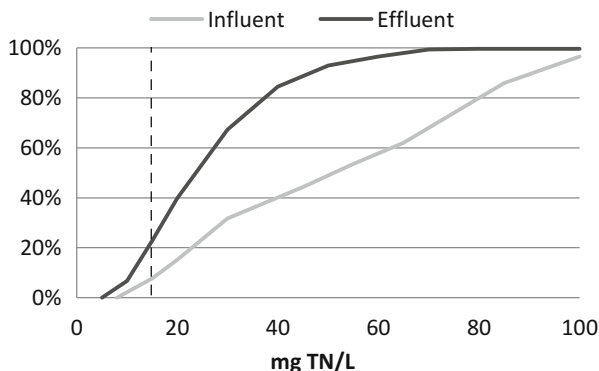


Fig. 14.8 The histogram for total nitrogen in the VFs (n=284). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

Fig. 14.9 Removal efficiency of nitrogen species in the VFs

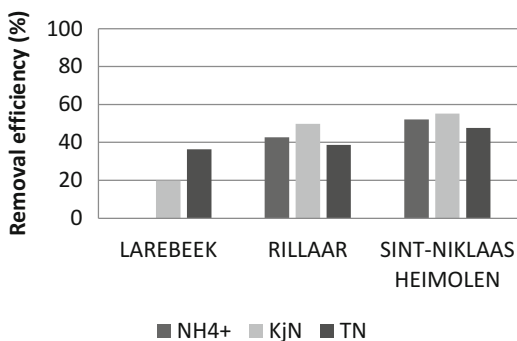


Figure 14.9 gives an overview of the average removal efficiency of nitrogen species in the three VF CWs. In general, kjeldahl nitrogen is rather efficiently removed by ammonification and nitrification. It seems though, that nitrification is poor at the Zemst-Larebeek installation because the average concentration of ammonium nitrogen increases by approximately 20% during the treatment. It is likely that the hydraulic conditions at the Zemst-Larebeek installation could explain the poor ammonium removal. The biological processes could be hindered because of the high hydraulic loading rate (Table 14.2). The actual HLR (0.57 m/d) is approximately 5 times higher than the design HLR (0.12 m/d) and the design HLR was met at 22% of the sampling events. The influent TN loading rate at the Zemst-Larebeek installation is thus much higher than at the other two VF CWs (Rillaar: 475 ± 183 g TN/m²/year; Sint-Niklaas Heimolen: 1970 ± 1230 g TN/m²/year). The influent loading rate at the Zemst-Larebeek installation was on average 5060 ± 8630 g TN/m²/year, which is also remarkably higher than usually reported for VF CWs, 1222 g/m²/year Vymazal (2007).

Denitrification was rather efficient in the VF CWs. Usually, denitrification is limited for single stage VF constructed wetlands because of the lack of anoxic zones (Vymazal 2007). The denitrification potential was calculated to be 35 mg N/L in the

Rillaar and Sint-Niklaas Heimolen CWs and slightly lower, 21.8 mg N/L, in the Zemst-Larebeek CW. On average, an 85 % denitrification efficiency was obtained in the VF CWs. The concentration of nitrification products, nitrate and nitrite, was found to increase in the effluent of Rillaar (5.9 ± 3.1 mg N/L) and Sint-Niklaas Heimolen (4.3 ± 4.3 mg N/L), but these values are low in comparison with literature (24.4 mg N/L; Vymazal 2007).

14.4.2.2 Phosphorus

Phosphorus removal in the VF wetlands is in general more efficient than in the FWS wetlands, due to the presence of a matrix. However, the matrix, e.g. gravel, has usually a low sorption capacity which also decreases over time due to saturation (Xu et al. 2006). Furthermore, the phosphorus removal in VF CWs can also be restricted due to changing redox conditions (Vymazal 2007).

The average removal efficiency of TP varies greatly between the VF CWs: $50 \pm 30\%$, $29 \pm 49\%$ and $2.4 \pm 108\%$ for the Rillaar, Sint-Niklaas Heimolen and Zemst-Larebeek installations, respectively. Only 25.3 % of the effluent samples reach the Flemish standard for large installations, 2 mg TP/L (Fig. 14.10). Pearson correlation analysis was further used to investigate whether the phosphorus removal efficiency is dependent on the age of the CW, i.e. the saturation state of the matrix. It appeared that there was no correlation between these two parameters ($R^2 \approx 0$; data not shown).

14.4.3 Horizontal Sub-Surface Flow Systems (HSSF)

Three HSSF CWs were studied based on the data on removal efficiencies and flow rates collected from the online database. Two of these systems are currently in operation. The Kiewit CW, consisting of 9 parallel wetlands, was shut down in spring 2014 after 15 years of operation. The Dikkelvenne CW is operated with 4 parallel

Fig. 14.10 The histogram for total phosphorus in the VF wetlands ($n=395$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

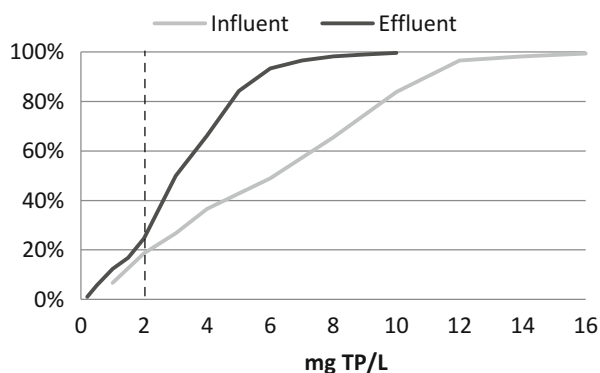


Table 14.3 Design and operational characteristics of the HSSF CWs

CW	Year of construction	Design capacity (pe)	BOD ₅ loading (pe) ^a	TN loading (pe) ^b	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
Dikkelvenne	2007	900	787	1320	1800 ^c	2.0	0.21
Hasselt-Kiewit	1999	137	84	131	640	4.7	0.04
Zemst-Kesterbeek	2001	315	279	586	1300	4.1	0.07

^aBased on 44 g BOD₅/pe/d (Aquafin 2013)

^bBased on 10 g TN/pe/d (Aquafin 2013)

^cBased on an aerial photo

HSSF CWs and the Zemst-Kesterbeek CW has 2 parallel HSSF CWs. The design and operational characteristics of these installations can be found in Table 14.3.

Similarly as in the case of the FWS and VF CWs, the nitrogen loading can be seen as the dominant parameter in determining the actual loading in the HSSF CWs. What also can be noticed is the smaller design footprint of the Dikkelvenne installation in comparison to the two older systems.

14.4.3.1 Nitrogen

The total nitrogen concentrations in the influent were 33.8 ± 14.9 mg N/L, 54.4 ± 20.6 mg N/L and 61.7 ± 28.5 mg N/L for the Dikkelvenne, Hasselt-Kiewit and Zemst-Kesterbeek installations, respectively. The TN at the Hasselt-Kiewit and Zemst-Kesterbeek installations consisted mainly of NH₄⁺-N (80 %) and organic nitrogen, but the Dikkelvenne installation received also some nitrite and nitrate (NH₄⁺-N: 61 % NO_x-N:8 %).

Based on the histogram for total nitrogen, the HSSF CWs achieve similar efficiency as the VF CWs. 28 % of the effluent samples of the HSSF CWs were under 15 mg N/L, as required for large wastewater treatment installations in Flanders (Fig. 14.11).

When the removal efficiency of the different nitrogen species is observed in more detail, one can notice that, overall, the performance of the HSSF CWs is similar to that of the VF CWs (Fig. 14.12) the nitrification efficiency seems limited. In case of single stage HSSF CWs, the low nitrification efficiency can, at least partly, be explained by limited oxygen transfer which is typical in these wetlands (Tyroller et al. 2010; Vymazal 2007). Furthermore, it is likely that the high influent loading rate of the two systems (Dikkelvenne: 2430 ± 1390 g TN/m²/year; Zemst-Kesterbeek: 1160 ± 1080 g TN/m²/year) and the lowered HRT, which is linked to the application of combined sewers, decrease their overall nitrogen removal efficiency. The total nitrogen loading rate reported in literature, 945 g/m²/year (Vymazal 2009) and 644 g/m²/year (Vymazal 2007), is lower than the values found here. The treatment performance at the Hasselt-Kiewit installation was even lower in comparison to the two

Fig. 14.11 The histogram for total nitrogen in the HSSFs ($n_{infl}=283$; $n_{eff}=284$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

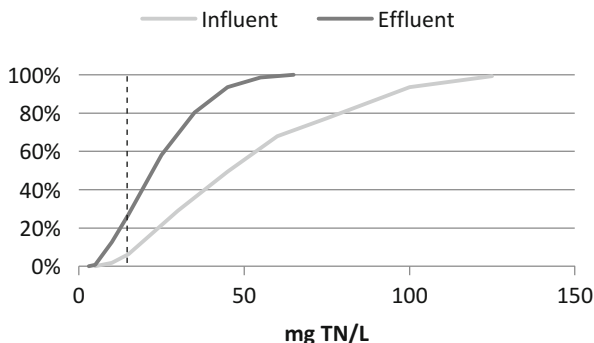
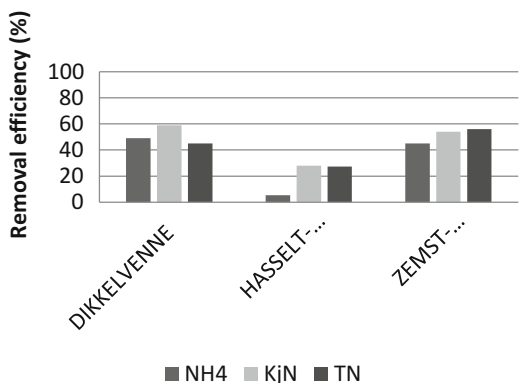


Fig. 14.12 Average removal efficiencies of nitrogen species in the HSSF wetlands



other installations. This can have been caused by age related clogging of the wetland matrix but it could not be verified.

Denitrification seems to be more efficient in the HSSF CWs than in the VF CWs as expected. Firstly, a lower concentration of nitrite and nitrate could be found in the effluent of the HSSF CWs (2.2 ± 1.7 mg N/L) in comparison to the VF CWs (4.5 ± 3.7 mg N/L). Secondly, the HSSF CWs were able to achieve denitrification efficiency of 97%, the average denitrification potential being 32 ± 8.7 mg N/L, in comparison to 85% in the VF CWs.

14.4.3.2 Phosphorus

As is usual for constructed wetlands, the phosphorus removal in the studied HSSF CWs was also rather low. The Zemst-Kesterbeek installation achieved the best average removal (44%) in comparison to the other two HSSF CWs (Dikkelvenne: 29% and Hasselt-Kiewit: 7%). Thus, only 23.2% of the effluent samples reached the Flemish discharge standard of 2 mg TP/L for large installations (Fig. 14.13). No correlation between the TP removal efficiency and age of the system was found ($R^2 \approx 0$).

Fig. 14.13 The histogram for total phosphorus in the HSSFs ($n_{infl}=283$; $n_{effl}=285$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

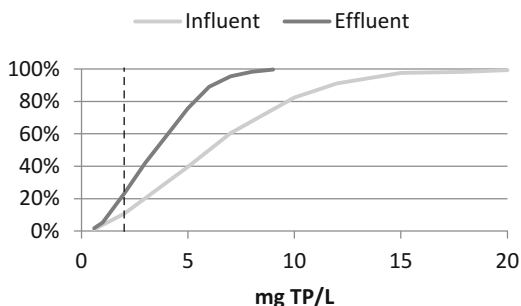


Table 14.4 Design and operational characteristics of the VF-HSSF CWs

CW	Year of construction	Design capacity (pe)	BOD ₅ loading (pe) ^a	TN loading (pe) ^b	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
Bierbeek-Kleinbeek	2000	189	162	276	660	3.5	0.21
De Pinte	2000	675	390	1140	2250	3.3	0.10
Ieper-Hollebeek	2000	360	690	1060	1080	3.0	0.27
Mol-Postel	1998	270	826	327	1440 ^c	5.3	0.03
Pervijze	2000	630	406	1176	2212	3.5	0.14

^aBased on 44 g BOD₅/pe/d (Aquafin 2013)

^bBased on 10 g TN/pe/d (Aquafin 2013)

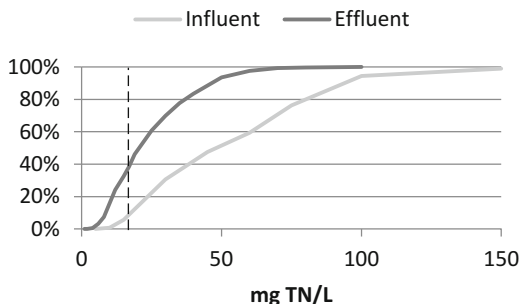
^cBased on an aerial photo

14.4.4 Combined Wetlands: VF-HSSF

Combined systems, which combine two or more wetlands in series, have the advantage over single stage CWs that they are usually better at total nitrogen removal (Vymazal 2007). This is due to the existence of both oxic and anoxic zones. In the five combined wetlands discussed here pre-settled wastewater is first treated in a vertical subsurface flow CW after which the secondary effluent is sent to a horizontal subsurface flow CW. The design and operational characteristics of these installations can be found in Table 14.4.

The same trend, with a few exceptions, is noticeable as with previous CW configurations, i.e. that the TN load is high in comparison to the BOD₅ one in terms of I.E. In case of the Mol-Postel treatment plant, the BOD₅ loading exceeds that of the TN when measured in I.E. This is due to the special influent composition caused by a nearby cheese factory. At the Ieper-Hollebeek installation, the influent is highly loaded in both organic matter and nitrogen. The specific reason for this is unknown.

Fig. 14.14 The histogram for TN in the VF-HSSF wetlands ($n_{\text{infl}}=562$; $n_{\text{eff}}=602$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe



14.4.4.1 Nitrogen

The total nitrogen content of the influent wastewater differed quite remarkably between the different VF-HSSF installations. The average total nitrogen concentration was 53.9 ± 69.9 mg N/L of which on average 66% consisted of ammonium nitrogen. The lowest TN concentration was observed in the influent of Ieper-Hollebeek (35.6 ± 23.1 mg N/L) which was the only one to receive some nitrite and nitrate in the influent (11%). The highest TN concentration was recorded at Mol-Postel (84.0 ± 26.9 mg N/L) where a large proportion of organic nitrogen (52%) in addition to ammonium nitrogen (48%) was registered. The reason for the high organic nitrogen at Mol-Postel load is a nearby located cheese factory.

As is theoretically expected, the removal of total nitrogen is better in combined systems than single stage wetlands (VF and HSSF). The combined systems achieved a cumulative removal efficiency of 33% (Fig. 14.14), when considering effluent samples containing maximum 15 mg TN/L. In comparison, the cumulative removal efficiencies of TN were 23% and 28% for the VF and HSSF CWs, respectively.

When the nitrogen removal is observed in more detail, clear differences between the installations can be noticed (Fig. 14.15). Especially the NH_4^+ removal efficiency in the Ieper-Hollebeek and Mol-Postel installations appears to be low in comparison to the others. The characteristics of the influent at Ieper-Hollebeek and Mol-Postel are likely to cause these differences. Firstly, the influent total nitrogen loading rate at the Ieper-Hollebeek installation (2410 ± 2024 g TN/m²/year) is higher than the loading rates of the other installations (1260 ± 838 g TN/m²/year) and also higher than the values reported in literature for VF CWs (1220 g N/m²/year; Vymazal 2007). Secondly, it appears that the NH_4^+ -nitrogen removal efficiency is comparable to the organic loading of the combined CWs. In short, when the BOD_5 loading is high, the NH_4^+ removal decreases. It is thus probable that the oxygen consumption of the organic matter degradation restricts the growth and activity of nitrifying organisms.

The concentration of nitrification products, nitrite and nitrate increases in the effluent of most of the combined wetlands (not in Ieper-Hollebeek). The lowest concentrations of nitrite and nitrate are found in the effluent at Mol-Postel, which is the CW receiving the highest organic load. It appeared also that the Mol-Postel CW is the most efficient one at denitrification. It achieved 96% denitrification efficiency

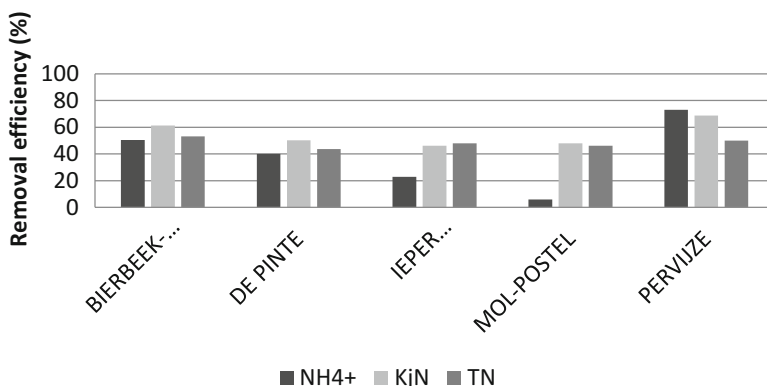


Fig. 14.15 The average removal efficiency of nitrogen species in VF-HSSF CWs

(denitrification potential 43.2 mg N/L) whereas the performance of the other combined wetlands was comparable to the single stage CWs (85%; 30.5±8.7 mg N/L).

14.4.4.2 Phosphorus

The phosphorus removal efficiency in the combined CWs varied between 23% and 30%. Because of the poor P removal, only 22% of the effluent samples achieved the discharge limit of large installations, 2 mg TP/L (Fig. 14.16). The P removal efficiency did not decrease with the age of the combined systems (data not shown).

14.4.5 Tertiary Treatment Wetlands: RBC-HSSF and SAF-HSSF

There are 22 CWs in Flanders which are used as tertiary treatment for domestic wastewater, and of which sufficient data are available. 17 of these are treating effluent from a rotating biological contactor (RBC) and 5 of these are placed after a submerged aerated filter (SAF). The purpose of the CWs is to function as a post-treatment for the secondary effluent but also to act as a buffer system in case of high flow, e.g. due to a heavy rain fall. More specifically, the CW receives untreated influent if the flow rate exceeds 6 times the design flow rate Q_{14} ($Q_{14}=1,7$ *dry weather flow) and pre-settled influent if the flow rate is lower than this but higher than 3 times the Q_{14} . At most treatment sites there are one or two CWs after the secondary treatment. The largest installations are in some cases divided in three or four CWs. Further design and operational characteristics of the tertiary CWs are compiled in Table 14.5.

Fig. 14.16 The histogram for TP in the VF-HSSF wetlands ($n_{infl} = 562$; $n_{eff} = 600$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

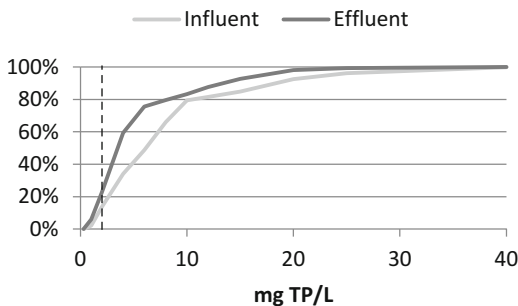


Table 14.5 Design and operational characteristics of the tertiary CWs

	CW	Year of construction	Design capacity (pe)	BOD ₅ loading (pe) ^a	TN loading (pe) ^b	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
RBC-HSSF-CW	Aalbeke	1996	450	326	596	500	1.1	0.30
	Asse-Bekkerzeel	2010	360	237	483	820 ^c	2.3	0.10
	Avekapelle	2012	330	266	323	N/A	N/A	N/A
	Beauvoorde	2012	360	929	1010	840 ^c	2.3	0.37
	Duffel	2011	800	318	654	1130 ^c	1.4	0.10
	Gistel	2012	500	606	1010	N/A	N/A	N/A
	Lebbeke-Rooien	2012	113	62.8	136	240 ^c	2.1	0.09
	Lokeren-Daknam	2013	500	619	1280	1170 ^c	2.3	0.39
	Lovie	2012	540	276	600	900 ^c	1.7	0.16
	Moerbeke	2006	405	326	580	760 ^c	1.9	0.12
	Ninove-Rendestede	2012	153	51.6	86.0	180 ^c	1.2	0.06
	Schorisse	2010	360	397	658	N/A	N/A	N/A
	St.Margriete	2014	450	208	530	N/A	N/A	N/A
	St.Maria Lierde	2000	850	1860	2960	425	0.56	1.7
	Stavele	2012	378	327	457	360	0.95	0.21
Steenkerke	2013	270	48.8	119	N/A	N/A	N/A	
Vlissegem	2013	500	530	646	N/A	N/A	N/A	
SAF-HSSF-CW	Loker	2007	675	480	657	1360 ^c	2.0	0.07
	Wachtebeke-Overslag	2012	405	245	523	850 ^c	2.1	0.09
	Watervliet	2009	810	456	1050	1570 ^c	1.9	0.24
	Wervik Kruiseke	2011	729	371	575	N/A	N/A	N/A
	Wingene St.Pietersveld	2007	540	712	807	N/A	N/A	N/A

^aBased on 44 g BOD₅/pe/d (Aquafin 2013), ^bBased on 10 g TN/pe/d (Aquafin 2013), ^cbased on an aerial photo
N/A not available

It can be concluded from Table 14.5 that the influent total nitrogen loading is often higher than the design capacity, similarly to other installations. At RBC-HSSF CW sites Gistel, Lokeren-Daknam and Schorisse the design capacity is exceeded by at least 100 %. The organic loading at the Beauvoorde and St. Maria Lierde installations is additionally very high. The St. Maria Lierde installation receives in addition to municipal wastewater also runoff from agricultural land (Lesage 2006), which is likely to increase the loading. The total nitrogen loading at the SAF-HSSF CW Watervliet is also higher than expected based on its design capacity. The results reported here are based on the performance of the complete installation, i.e. secondary and tertiary treatment together.

14.4.5.1 Nitrogen

The influent nitrogen content at the tertiary treatment plants is comparable to the other installations. The TN concentrations in the influent were 51.3 ± 23.9 mg N/L and 52.9 ± 18.5 mg N/L for the RBC and SAF-HSSF CWs, respectively. The fraction of ammonium nitrogen was on average 74 %. Only the $\text{NH}_4^+\text{-N}$ proportion at the Beauvoorde and St. Maria Lierde installations differed from the others being 42 % and 52 %, respectively. It is possible that the runoff from fields, that was stated to occur at the St. Maria Lierde installation, contains soil, plant debris and manure which increase the organic nitrogen content of the influent.

Based on the cumulative removal efficiency of total nitrogen, the SAF-HSSF CWs are more efficient in TN removal than the RBC-HSSF CWs. 59 % of the effluent samples discharged from a SAF-HSSF CW comply with the Flemish standard of 15 mg TN/L, in comparison to 44 % of the effluent samples originating from a RBC-HSSF CWs (Fig. 14.17).

Nitrification is mainly the task of the secondary treatment (RBC or SAF) at these installations. The ammonium and kjeldahl nitrogen removal efficiency in these treatment systems was remarkably better and less variable than in the other installations. The RBC and SAF-HSSF CWs could achieve a removal of 90 % and higher for ammonium and kjeldahl nitrogen (Figs. 14.18, and 14.19) whereas other installations reached barely 40 %.

Denitrification occurs mainly in the tertiary treatment HSSF wetland. The denitrification potential was calculated to be 48.3 ± 15.0 mg N/L for the RBC-HSSF CWs and 52.2 ± 17.7 mg N/L for the SAF-HSSF CWs. Hence, the denitrification potential is higher in the tertiary treatment systems than in the other installations. The denitrification efficiency is better in the SAF-HSSF CWs (78 %) than in the RBC-HSSF CWs (67 %). In comparison to other installations, the tertiary treatment systems achieve lower denitrification efficiency. Because of this, the nitrite and nitrate concentrations are high in the effluent of the tertiary treatment systems (14.5 ± 6.0 mg N/L).

The problem with denitrification in tertiary treatment systems is that a large portion of the bioavailable organic matter is already degraded in the secondary treatment, limiting the denitrification in the tertiary treatment. To analyze the organic

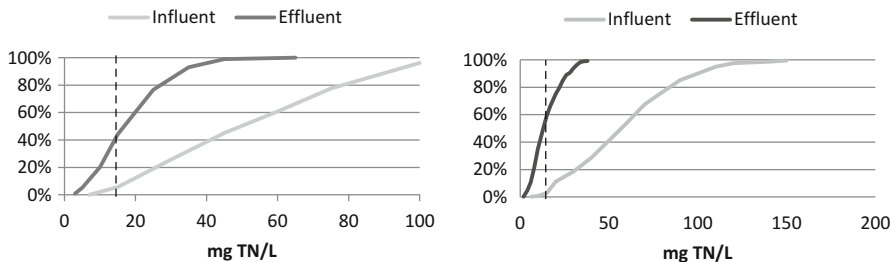


Fig. 14.17 The histogram for TN in the RBC-HSSF wetlands ($n_{infl}=970$; $n_{eff}=977$, *left*) and the SAF-HSSF wetlands ($n_{infl}=383$; $n_{eff}=384$, *right*). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

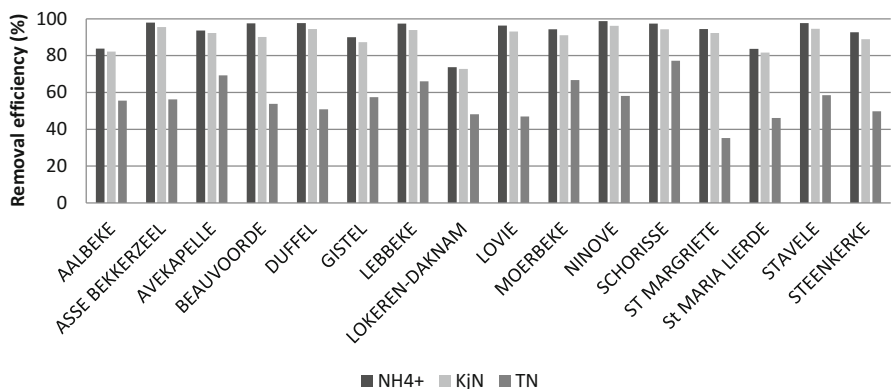


Fig. 14.18 Average removal efficiencies of nitrogen species in the RBC-HSSF CWs

loading received by the CWs, it was calculated based on the available flow rates how often the HSSF CWs receive (pre-settled) influent. Influent was sent directly to the RBC-HSSF CWs on 3% of the operation days and additionally, pre-settled influent on 5% of the operation days. In case of SAF-HSSF CWs the values were somewhat lower, 1% and 3%, respectively. However, as denitrification was more efficient in the SAF-HSSF CWs, the organic loading in the HSSF CWs is presumably not the main limiting factor.

14.4.5.2 Phosphorus

The phosphorus removal efficiency is similar in both types of tertiary systems. 30% of the effluent samples from the RBC-HSSF CWs and 11% of the effluent samples from the SAF-HSSF CWs were under the discharge limit for large installations, 2 mg TP/L (Figs. 14.19, and 14.20). In RBC-HSSF CWs the TP removal varies between 31% and 52%, with two exceptions of 18% and 19%. In SAF-HSSF CWs the TP removal efficiency is generally between 29% and 49%, but in one system

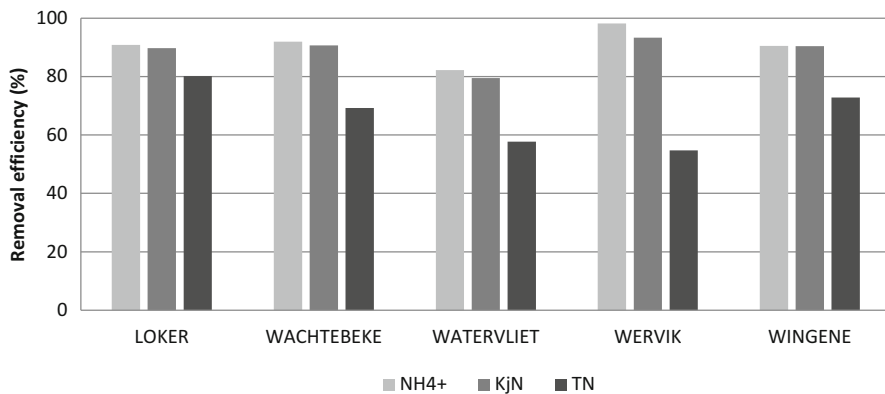


Fig. 14.19 Average removal efficiencies of nitrogen species in the SAF-HSSF CWs

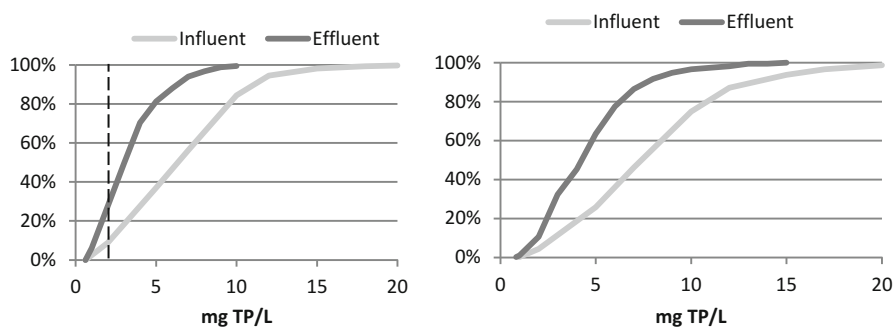


Fig. 14.20 The histogram for TP in the RBC-HSSF wetlands ($n_{infl}=968$; $n_{eff}=976$, left) and in the SAF-HSSF wetlands ($n_{infl}=383$; $n_{eff}=384$, right). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations >2000 pe

slightly lower, 21 %. These values are similar to other installation types. The removal efficiency was not noticed to decrease with the age of the system (data not shown).

14.5 Agricultural Wastewater N and P Removal

The installations for treating pig manure consist of a multi-bed system, with VF and HSSF CWs as well as open ponds. Most of the systems are situated in the province West-Flanders, which is characterized by a very high density of animal husbandry and in particular a very intensive pig farming industry. The systems vary in size between 0.35 and 1.5 ha and having a capacity of 5000–20,000 t/year of manure post-treatment. Based on COD and TN concentrations of ingoing influent, the loading is equivalent to 300–11,00 pe.

Table 14.6 Design and operational characteristics of the “Innova Manure” treatment wetlands

CW	Year of construction	Design capacity (pe)	TN loading (pe) ^a	Surface area (m ²)	Design footprint (m ² /pe)	Hydraulic loading rate (m ³ /m ² /d)
Ichtegem	2006	288	288	3510	12	1.0
Gistel	2007	1159	1159	14,100	12	1.0
Langemark I	2008	356	392	4332	12	1.1
West-Vleteren	2009	1233	863	15,000	12	0.7
Pittem	2009	719	863	8750	12	1.2
Hoogstraten	2013	986	1233	12,000	12	1.3
Langemark II	2015	360	N/A	4375	10	N/A

^aBased on 10 g TN/pe/d

N/A not available

Based on the monthly monitoring, flows to and within the system are adapted to restrict the nutrient losses to the receiving surface water. When concentrations in the final buffer lagoon of the wetland exceed the discharge limits, the pumping schedule within the system is altered to provide for longer HRT within the system and to block all possible discharge until values return to below allowable levels. The loading of the manure treatment wetland systems is dimensioned in function of COD and TN. Table 14.6 presents the design and operational characteristics of the six treatment wetlands studied. The Langemark II CW system is only recently taken into operation and no sufficient data from it is yet available.

14.5.1 Nitrogen

The influent TN concentrations can vary greatly between 50 and 800 mg/L with an average value at around 300 mg/L (Fig. 14.21). The systems are well equipped to deal with fluctuating inlet concentrations and rarely any values over 15 mg/L are ever observed. In those cases when values above the discharge limit are observed, discharge is blocked and the effluent is recirculated back into the wetland.

14.5.2 Phosphorus

The manure treatment wetlands are designed in such a manner that phosphorous build-up within the system is avoided. The loading rate of the pre-treated manure is limited to removal capacity via plant uptake. The reed plants are annually harvested and exported off-site to assure any potential accumulation. TP concentrations of the pre-treated manure influent are <15 mg/L and the wetlands have no problem to achieve the <2 mg/L threshold as can be observed in Fig. 14.21.

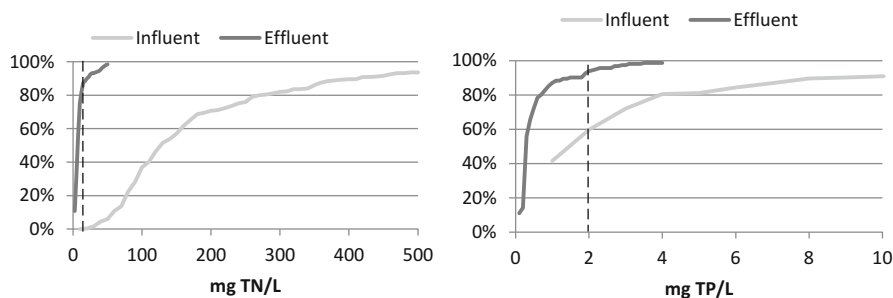


Fig. 14.21 The histogram for TN (*left*) and TP (*right*) in the manure treatment wetlands ($n=249$). The dashed line indicates the limit value set for effluent in the Flemish legislation VLAREM for installations discharging processed manure into surface water (15 mg/l TN, 2 mg/l TP)

14.6 Comparison of the Performance of Constructed Wetlands

In this study, the treatment performance of 35 CWs treating municipal wastewater and of 6 manure treatment wetlands was studied in more detail. The wetlands treating municipal wastewater were located all around Flanders. Most of these wetlands are built after the year 2000, and their design capacities vary between 113 and 1800 pe. Free-water-surface CWs (2 wetlands), vertical flow (3 wetlands) and horizontal flow CWs (3 wetlands) are used as single stage CWs, and five cases are combined systems with VF CW followed by HSSF CW (VF-HSSF CWs). Twenty two wetlands are used for tertiary treatment after RBC (17) or SAF (5). The manure treatment wetlands studied are mainly located in the region of West-Flanders which is known for its intensive agriculture. Their design capacities vary between 300 and 1100 pe. All systems consist of a combination of multiple beds.

The influent composition at the different treatment plants was overall similar. The organic loading was in general lower than expected based on the design capacity (pe), but the total nitrogen load was often higher (Table 14.7). This was expected to be due to the biodegradation of BOD_5 in the sewer system and septic tanks. The TN concentrations were not observed to be higher than reported in literature earlier, but the high flow rate caused the increased TN loading. The high flow rate is caused by the combined sewers in which wastewater is diluted by surface water and infiltrating groundwater. These sewer systems are still widely used in Flanders. When the situation is compared to a large activated sludge treatment plant in Ghent (207,000 pe), a similar trend is observed. Accordingly, a study on Flemish nutrient streams states that the often applied 10 g TN/pe/d possibly underestimates the TN content in the domestic wastewater in Flanders and that 12 g TN/pe/d should be used instead (Coppens et al. 2013).

The nutrient emissions from the CWs were compared to the discharge standards of larger wastewater treatment plants (>2000 pe) because the Flemish legislation on the treatment performance of small-scale treatment systems, which includes all

Table 14.7 Average influent and effluent loading at the CWs

Wastewater	Type CW	Influent loading (g TN/m ² /year)	Effluent loading (g TN/m ² /year)	Reduction (%)	Influent loading (g TP/m ² /year)	Effluent loading (g TP/m ² /year)	Reduction (%)	Influent loading (g TP/m ² /year)	Effluent loading (g TP/m ² /year)	Reduction (%)	References
Domestic	FWS	719	622	13	86	78	9.3	86	78	9.3	This study
		466	219	53	138	68	51	138	68	51	Vymazal (2007)
Domestic	VF	2500	1330	47	256	156	39	256	156	39	This study
		1220	592	52	126	54	57	126	54	57	Vymazal (2007)
Domestic	HSSF	1340	575	57	180	116	36	180	116	36	This study
		945	580	39	248	179	28	248	179	28	Vymazal (2009)
		644	394	39	141	96	32	141	96	32	Vymazal (2007)
		590 ^a	360 ^a	39	133 ^a	84 ^a	37	133 ^a	84 ^a	37	Vymazal (2002)
Domestic	VF-HSSF	1490	809	46	197	146	26	197	146	26	This study
Domestic	Tertiary	N/A	N/A		N/A	N/A		N/A	N/A		
Pre-treated pig manure	Combined	70,300	3790	95	1470	236	76	1470	236	76	This study

^aAverage of reported values

N/A not applicable

CWs in Flanders, is not strict. Thus, there are no nutrient limitations for these small-scale treatment plants. This is rather contradictory taking into account the fact that the whole of Flanders is categorized as a nitrate sensitive area by the EU (EU 2014). Nutrient pollution from households and agriculture is a significant problem in Flanders causing elevated nutrient concentrations in surface and ground waters (EEB 2010; EU 2013). Brooks and small rivers, which are often the discharge waters of small-scale treatment plants, are thus at risk of eutrophication. Based on our calculations, approximately 70–80% of the effluent samples do not meet the requirements set for TN emissions for large installations.

The FWS CWs appeared to show very low overall nitrogen transformation efficiency (Table 14.8). The influent consisted mainly of ammonium nitrogen but only a small fraction was nitrified. Therefore, the production of nitrite and nitrate was essentially not occurring, and was limiting denitrification. It was concluded that the problems related to the low nitrification efficiency are related to the low HRT and poor oxygen transfer in these systems. As a consequence, the TN removal efficiency was lower and the TN concentration higher than in reported in literature for other FWS CWs (Vymazal 2007).

The VF and HSSF CWs showed similar nitrogen removal efficiencies. Nitrification seemed to be the limiting process in both systems. It was surprising that the VF CWs also experienced problems related to oxygen transfer. This was again expected to be caused by the high nitrogen loading rate, which was observed in case of both types of wetlands. As expected, denitrification was slightly more efficient in case of HSSF CWs than VF CWs, which resulted in lower nitrate concentration in the HSSF CW effluent. The remaining TN and ammonium nitrogen concentrations or the TN removal efficiency were in the range reported earlier in literature reviews (Vymazal 2007, 2009).

The combined systems, VF-HSSF CWs, could not achieve remarkably improved performance when compared to the single stage systems, VF and HSSF CWs. It was concluded that the nitrification step was limiting because of high loading rate.

The removal efficiency of nitrogen species was observed to be best in the tertiary treatment systems. Also, the tertiary treatment systems achieved the best nitrification efficiency. The nitrogen in the effluent was mainly in the form of nitrate, unlike in the other systems where it was ammonium. The oxygen transfer in the tertiary treatment systems was thus improved in comparison to basically all CWs as secondary treatment which had poor nitrification efficiency due to nitrification efficiency. Aeration of the secondary treatment CWs could offer a solution to the poor oxygen transfer, and should be considered to obtain better nutrient removal efficiency in these systems.

The denitrification efficiency was found to be slightly poorer in the tertiary systems than in the other systems (not FWS). This was possibly partly due to the consumption of organic matter in the secondary treatment.

The removal of total phosphorus was also studied. The FWS CWs showed the lowest TP removal efficiency, which was attributed to the lack of matrix material in these wetlands and thus a lack of adsorption sites. The other single stage and combined systems achieved a removal of approximately 20%. The best removal effi-

Table 14.8 Average treatment performance of the different types of CW systems

Waste-water	Type CW	Influent TN concentration (mg N/L)	Effluent TN concentration (mg N/L)	Effluent NH_4^+ concentration (mg N/L)	Effluent NO_3^- concentration (mg N/L)	TN removal efficiency (%)	$\text{NH}_4^+\text{-N}$ removal efficiency (%)	Effluent TP concentration (mg N/L)	TP removal efficiency (%)	N
Domestic	FWS	23.8	19.7	16.1	0.275	14	0.35	2.50	6.3	2
Domestic	VF	49.8	26.7	19.2	4.36	41	24	4.19	19	3
Domestic	HSSF	50.0	24.2	20.5	2.12	43	33	3.65	27	3
Domestic	VF-HSSF	53.9	24.3	17.6	3.85	48	38	5.42	22	5
Domestic	Tertiary	51.7	17.8	1.68	14.2	58	93	3.75	38	22
Pre-treated pig manure	Combined	192	10.1	N/A	N/A	95	N/A	0.63	85	6

N/A not available

ciency was achieved in the tertiary systems. The concentration of TP in effluent is comparable to reported values (Vymazal 2007), but these values are, however, above the effluent standards applied for larger treatment plants (>2000 pe). When the percentage of removed TP load is observed, it appears that the CWs in this study perform comparably to those reported in the literature. At the moment the matrix material is not chosen material with the purpose of maximizing the P removal and therefore, it could be possible to improve the TP removal efficiency by opting for an improved matrix, e.g. through iron amendments (Van Oirschot 2010). The aging of the installations during the period 2005–2014 was not found to affect the TP removal efficiency.

In comparison to the CWs treating municipal wastewater, the manure treatment wetlands are very efficient at the removal of both total nitrogen and phosphorus. On average 95 % of the TN load and 76 % of the TP load was removed (Table 14.7). The effluent standards set for the manure treatment wetlands were thus also fulfilled (Table 14.8). The difference is caused by the different operation and design of the systems. The manure treatment CWs are dimensioned larger than the CWs treating municipal wastewater, and mostly they are even over-dimensioned. Furthermore, in these CWs recycling of effluent is applied if needed and P removal is improved by frequent plant harvesting. It must be noted however, that intensive sampling is necessary to define the need for recycling, and this is laborious and costly.

14.7 Conclusions

The number of constructed wetlands for wastewater treatment has increased tremendously since 2004 when the last overview on the Flemish CWs was published. Especially the vertical flow constructed wetlands are popular, and most of the recently built VF CWs are individual wastewater treatment systems of single families. CW technology has thus shown to be an important sustainable solution for wastewater treatment in rural areas. Correct design can also contribute to decreasing nutrient load to the environment. Due to the current high nitrate concentrations in Flemish groundwater and surface water, it is advisable to revise the effluent standards for wastewater effluent from small-scale treatment plants.

The CWs for municipal wastewater treatment studied in this chapter achieve the treatment performance set in legislation. Currently, this only involves the reduction of organic loading and nutrient discharge is not controlled. When we analyzed the data on N and TP discharge, it was clear that most of the systems could not contribute significantly to reducing the nutrient levels in wastewater. The reason for this was found to be the TN loading caused by high flow rates. Due to this it is likely that the oxygen transfer, and thus nitrification, is hampered in most systems, especially the single stage and combined systems. Active aeration should thus be considered as an option in future installations.

Treatment wetlands for processing pig manure down to dischargeable levels are particularly designed and operated in such a manner that effluent above effluent

standards is recirculated back into the wetland system. For TP, accumulation within the system is avoided by dimensioning the loading rate to levels well below annual removal capacity by plant uptake (and harvest export off-site). As regarding TN, the systems are currently over-dimensioned to 12 m²/pe. which explains why reducing effluent levels to below 15 mg/L rarely poses any problems. Even though over-dimensioned, overall costs of building, operations and follow-up amount to approximately to 3–4 €/m³ of pre-treated pig manure (i.e. following separation and nitrification/denitrification). In comparison, spreading pre-treated pig manure as a fertilizer costs about 5 €/m³ in the nutrient intensive region of Flanders, making wetland systems a valuable alternative. Particular interest for further investigation regarding treatment wetlands for this type of application, goes to the role of recalcitrant dissolved organic matter and its impact on elevated COD concentrations which can occur even if BOD₅ concentrations remain well below discharge limits (<25 mg/L) and in the majority of observations even below detection limits (<4 mg/L).

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Chapter 15

Performance Intensifications in a Hybrid Constructed Wetland Mesocosm

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Abstract This chapter presents the study on the performance of a hybrid mesocosm constructed wetland system composed of two treatment step: vertical flow bed and floating emergent macrophyte unit (a tank with a vegetated floating mat). The objectives of this study were to assess the performance intensifications in this system by assessing the effect of: (i) combing two types of constructed wetlands, (ii) increasing saturation level in the vertical flow unit, (iii) applying artificial intermittent aeration in the vertical flow unit, on the removal of carbonaceous compounds, nitrogen species and (phosphate phosphorus) P-PO₄. The duration of the experiment was 378 days and it was divided into 4 periods: two periods with unsaturated vertical flow bed (but with different hydraulic loading rate values) and two periods with saturated bed of the vertical flow unit (in one period the bed was additionally intermittently aerated). The experimental system was fed with synthetic municipal waste water in a batch mode. It was observed that the intensification of the system's performance by integrating a vertical flow bed with partly saturated conditions and a floating emergent macrophyte unit ensured high removal efficiencies for total nitrogen (TN) and P-PO₄. Additionally, the application of artificial intermittent aeration in a vertical flow bed allowed achieving high removal efficiency of carbonaceous compounds (determined as dissolved organic carbon) without compromising the removal of TN and P-PO₄. The scope of further research will include optimization of the system for hydraulic retention time and batch duration and monitoring of the fate of selected micropollutants in the system.

Keywords Hybrid • Vertical flow • Constructed wetland • Floating emergent macrophyte • Intermittent aeration

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15.1 Introduction

The technology of treating municipal and domestic waste water in constructed wetlands (CWs) have rapidly developed over the last three decades (Wu et al. 2014) to become a viable alternative to energy intensive technologies for the sanitation of small communities (Garcia et al. 2010). Increasingly stringent water quality requirements in Europe necessitate further development of CWs and to intensify the treatment efficiency of these systems (Miksch et al. 2015) especially in terms of nitrogen and phosphorus removal (Wu et al. 2014). The intensification of the treatment processes in CWs can be achieved by various methods e.g. artificial aeration or integration of different types of CWs (Wu et al. 2014), the use of various filter media with high sorption capacity for the removal of phosphorus (Kim et al. 2015), or combination with other treatment technologies such as membrane bioreactors or advanced oxidation processes for the treatment of high-strength waste water and elimination of recalcitrant organics, respectively (Liu et al. 2015). The poor oxygen supply in conventional horizontal subsurface flow CWs or in the vertical flow CWs (VF-CWs) with saturated bed is a limiting factor for removal of organic matter and nitrogen (Boog et al. 2014; Wu et al. 2014). Aerated CWs are characterized by improved treatment efficiency, and therefore these intensified systems can have smaller footprint, and offer stable performance even in cold-climate countries and are less susceptible to clogging (Boog et al. 2014). Thus far, artificial aeration has been mostly introduced in the VF-CWs, however, other types of CWs have been also fitted with an aeration system. The artificial aeration in subsurface flow CWs can be operated continuously or in an intermittent pattern. The continuous aeration may lead to complete nitrification because anoxic conditions favourable for denitrification would not be provided. Thus, intermittent operation seems to be a viable solution for total nitrogen (TN) removal by providing alternate aerobic/anoxic conditions advantageous for simultaneous nitrification and denitrification and as a result enhancing overall nitrogen removal (Boog et al. 2014; Wu et al. 2014). Boog et al. (2014) observed improved removal of TN and nitrates and comparable removal of carbonaceous compounds in saturated VF-CWs with intermittent artificial aeration as compared with continuously aerated systems. Apart from the enhanced treatment efficiency, intermittently aerated CWs are more energy-economic than continuously aerated beds (Wu et al. 2014). There were various aeration patterns applied in the CW e.g. continuous and intermittent: 8 h on/4 h off (Boog et al. 2014), 1 h on/5 h off (Fan et al. 2013), or 5 min on/25 min off (Foladori et al. 2013). The removal of phosphorus in CWs is closely associated with the properties the filter material, because this element is mainly sorbed by or precipitated in filter media (Vohla et al. 2011). The conventionally applied filtered media as gravel or sand do not usually provide long-lasting elimination of phosphorus (Vohla et al. 2011). The poor elimination of nitrates and phosphorus in VF-CW (Kim et al. 2015) can be overcome also by combining this type of CWs with wetland systems promoting intensive nutrient removal such as, for example, free water surface CWs (FWS-CWs) (Vymazal 2005). These hybrid (or integrated) constructed wetlands combine different CW

units to optimize the treatment efficiency of the overall system (Fonder and Headley 2013) by complementing the advantages of each treatment unit (Vymazal 2013). It was reported that the application of FWS-CWs as the final treatment step not only improves nutrient removal but also the elimination of bacteria (Vymazal 2005), selected micropollutants (e.g. diclofenac) and microbial activity (Ávila et al. 2014). One of the variants of the FWS-CWs is a floating emergent macrophyte CW (FEM-CW), frequently also termed as floating treatment wetland, in which emergent macrophytes grow on a buoyant artificial mat or raft floating on a surface of a pond (Fonder and Headley 2013). Plant roots grow through the porous mat and hang into the water column (Borne et al. 2014), which provides extensive area for the growth of biofilm and entrapment of fine suspended particulates (Tanner and Headley 2011). As the plants are not rooted, they are forced to acquire the nutrients directly from the water, which may lead to significant improvement in the rate of nutrient uptake rate by plants (Tanner and Headley 2011). The important feature of FEM-CWs is that their buoyancy enables to tolerate wide fluctuations in water depth (Tanner and Headley 2011).

The objectives of this study were to assess the performance intensifications in a mesocosm CW system by assessing the effect of: (i) combining two types of CWs, namely VF-CW and FEM-CW, (ii) increasing saturation level in the VF-CW, (iii) applying artificial intermittent aeration in the VF-CW, on the removal of carbonaceous compounds, nitrogen species and (phosphate phosphorus) P-PO₄.

15.2 Methods and Materials

15.2.1 Experimental System

The experimental system was a mesocosm hybrid CW comprising two treatment step: vertical flow (VF-CW) bed and a tank with a floating island, which will hereafter referred to as a floating emergent macrophyte constructed wetland (FEM-CW) according to the nomenclature proposed by Fonder and Headley (2013).

The VF-CW was constructed using plastic cylindrical tank (diameter 60 cm, height 90 cm), which was filled up to 80 cm with mineral media (15 cm of 4–8 mm gravel at the bottom, and 65 cm of 0.5–1.0 mm quartz sand; both fraction were separated using polyester mesh). The filter media were approved for potable water sand filters and were purchased from Ecopol Sp. z o.o. (Police, Poland). The VF-CW tank had four orifices (diametrically opposite) on the side wall at the bottom. Three orifices were interconnected with a plastic pipe that extended up to the upper brim of the barrel, which was used a passive aeration system until day 350 of the experiment and further on it was connected with an air pump through plastic tubing. The fourth orifice was used as an outflow of the treated waste water. Additionally there were four more outlets situated every 5 cm above the bottom outlet, but they were not used in this experiment. No specific air distribution system, as e.g. perforated

pipes, was used inside the VF-CW assuming that the bottom layer of gravel would distribute the air sucked or pumped into the system evenly. The bed of VF-CW was inoculated using return sludge at the beginning of the experiment. This sludge originated from municipal waste water treatment plants using conventional activated sludge technology. The mixed liquor suspended solids concentration in the sludge was 5 g/L and the volume of the sludge used to inoculate the system was 20 L.

The tank of the FEM-CW was constructed using steel frame and glass panes (length 150 cm, width 100 cm, depth 50 cm). The artificial floating island (120 cm length, 80 cm width and approx. 10 cm thickness) was a buoyant structure made of lime-wood frame with a willow wicker weaving supporting 10-cm layer of light expanded lay aggregates covered with a plastic net. The artificial floating island was purchased from Ogrody Wodne (Gorzycko Stare, Poland). In order to maintain initial buoyancy of the floating island a rectangular frame (120 cm length, 80 cm width, with an additional transversal pipe) made of drainage pipes filled with empty PET bottles (11 bottles in total) was placed beneath the island after a year of the experiment. The water depth in the FEM-CW was 45 cm and was increased to 50 cm on day 329 for aesthetic reasons, which corresponded to active volume 675 L and 750 L, respectively. The VF-CW and FEM-CW were connected by a stainless steel braided rubber hose. The inlet of the waste water in the FEM-CW was situated in the corner of the tank approx. 5 cm above the bottom, and the outlet was situated in the diagonally opposite corner of the tank also approx. 5 cm above the bottom. Prior to the start-up the FEM-CW tank was filled with the effluent of microcosm VF-CWs treating the same synthetic waste water.

The experimental system was operated under indoor conditions with artificial lighting system comprising one 600 W high-pressure sodium lamp (HPS) and one metal-halide lamp (600 W), of which the former was hung above the FEM-CW, and the latter was put on a post in an inclined position adjacent to the VF-CW (Fig. 15.1). Before day 262 only one lamp was used (600 W HPS), however due to increasing shading of the FEM-CW by the vegetation in the VF-CW another lamp was added to the lighting system. The lighting system was purchased from Flora and Fauna Sp. z o.o. (Poland). The light/dark conditions were 16/8 h throughout the experiment.

The FEM-CW tank was aerated continuously using an air pump Tetratrac APS-150 (3.1 W, with a maximum airflow of 2.5 L/min) connected with a ceramic stone diffuser placed on the bottom of the tank. The aeration in the FEM-CW was used to prevent oxygen depletion and to induce mixing. The floating island occupied significant area of the tank (64%) and it would obstruct oxygen diffusion into the system leading to aesthetically displeasing appearance of the system, which was noticed in the week preceding the start-up of the system. On day 351 the use of the Tetratrac air pump was discontinued and it was replaced with an air pump HIBLOW-HP-60 manufactured by Techno Takatsuki (Japan) with a maximum airflow 60 L/min. The HIBLOW-HP-60 air pump was used to aerate both the VF-CW and FEM-CW units in the aeration pattern 4 h on/2 h off. Also the ceramic stone diffuser was replaced by an EPDM-HD200 disc diffuser (effective diameter 184 mm) manufactured by Jäger Gummi und Kunststoff GmbH (Germany) mounted on a concrete base and placed on the bottom of the FEM-CW tank.

Fig. 15.1 Experimental mesocosm system: VF-CW (left) and FEM-CW (right), photo taken on day 342 of the experiment



The VF-CW was planted with 10 rhizomes of common reed (*Phragmites australis*) but it was observed that this plant desiccated completely, probably due to insufficient substrate moisture, which was related to the mode of operation (Table 15.1). For this reason, common reed was completely removed from the system and replaced with 5 seedlings (in 2-L pots) of giant miscanthus (*Miscanthus × giganteus*) on day 65 of the experiment. It should be noted that the bed of VF-CW was not saturated with water prior to the start-up of the system, which probably prevented establishment of common reed. The floating island used in the FEM-CW was initially planted with 24 seedlings of sweet flag (*Acorus calamus*), 3 seedling of yellow flag (*Iris pseudacorus*), 2 seedlings of soft rush (*Juncus effusus*) and 2 seedlings of dwarf scouring rush (*Equisetum scirpoides*). American frogbit (*Limnobium spongia*) was added as a free-floating plant and common duckweed (*Lemna minor*) appeared spontaneously in the FEM-CW. Three seedlings of parrot feather (*Myriophyllum aquaticum*) were planted on the bottom of the tank in 100-mL plastic containers filled with sand. Other plants that were successfully planted on the floating island in the further part of the experiment were: purple loosestrife (*Lythrum salicaria*), yellow monkey-flower (*Mimulus luteus*), water mint (*Mentha aquatica*), bulrush *Scirpus* (*Schoenoplectus*) *lacustris*, bog bulrush (*Schoenoplectus mucronatus*), brooklime (*Veronica beccabunga*), watercress (*Nasturtium officinale*), creeping jenny (*Lysimachia nummularia*), great willowherb (*Epilobium hirsutum*).

Table 15.1 Operation chart for the experiment

Period	Days	Number of batches	Feeding/resting, d	Bed saturation	Operation mode	HLR, mm d ⁻¹	COD load, g m ⁻² d ⁻¹	VF-CW aeration, pattern	HRT (FEM-CW), d
A	1-50	34	5/2	Unsaturated	Batch	32	26	No	81
B	51-251	144	5/2			64	24	No	88
Transitory	252-293	Various settings							
C	294-350	18	2/5	Partially saturated	Batch	80	63	No	157
D	351-378	4	1/6		Batch	213	198	Yes, 4 h on/2 h off	175

The plants that were planted on the island but failed to adapt to its specific conditions were: graceful cattail (*Typha laxmanii*), duck potato (*Sagittaria latifolia*), Japanese sweet flag (*Acorus gramineus*). It was observed that yellow flag had the highest growth rate and greatest biomass production.

The experimental system was fed with simulated municipal waste water prepared according to the protocol by Nopens et al. (2001) with several adjustments (doubled dose of the compounds with the exception of Cr, Cu, Mn, Ni, Pb, Zn, Mg, Fe salts; also soy oil and starch were not used. The influent was prepared in tap water by dissolving the following components (Nowrotek et al. (2016)): urea (208.76 mg/L), NH_4Cl (62.4 mg/L), yeast extract (264 mg/L), skim milk powder (118 mg/L), sodium acetate (510.4 mg/L), peptone (40 mg/L), KH_2PO_4 (41.37 mg/L), $\text{KCr}(\text{SO}_4)_2 \cdot 12\text{H}_2\text{O}$ (0.96 mg/L), $\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$ (0.781 mg/L), $\text{MnSO}_4 \cdot \text{H}_2\text{O}$ (0.108 mg/L), $\text{NiSO}_4 \cdot 7\text{H}_2\text{O}$ (0.359 mg/L), PbCl_2 (0.1 mg/L), ZnCl_2 (0.208 mg/L), $\text{MgSO}_4 \cdot 7\text{H}_2\text{O}$ (4.408 mg/L), $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$ (11.6 mg/L). In period B of the experiment (Table 15.1) the influent was diluted 1:2 using tap water to increase the hydraulic loading rate (HLR) in this period.

The experiment presented in this chapter was started on the 15 October 2014 and the presented data refer to 378-day operation, this is until the 26 October 2015. The operation of the experimental system can be divided into four distinctive periods A-D (Table 15.1) based on the operation parameters. In periods A and B the VF-CW unit was operated as an unsaturated bed and two HLR values were applied. In periods C and D the VF-CW was partially saturated and it was additionally intermittently aerated in period D. In the transitory the bed saturation level was gradually increased in every batch up to the level of 48 cm (maintained in period C), but the waste water samples were not taken. It should be noted that the operation of the system was divided based on the values of the parameters that can be controlled or monitored, therefore the shift from common reed to giant miscanthus in the VF-CW was not considered as an onset of a new period, however the conditions created by the development of the rhizosphere of this plant should not be neglected. The presented values of the hydraulic retention time (HRT) in the FEM-CW (Table 15.1) were calculated based on the average volume of VF-CW effluent (Table 15.1).

In all the periods the VF-CW step was fed in batch mode with various feeding/resting patterns and with the entire cycle being invariably 7 days long. In periods A and B the VF-CW was fed for 5 days (Monday-Friday) and then rested for 2 days (Saturday and Sunday) and was operated in a free draining mode. In periods D and C the bed was fed either for 1 or 2 days (Monday or Monday and Tuesday, respectively) and was rested for 6 or 5 days, respectively. In contrast to period A and B, in periods C and D the bed was partially saturated during the resting period. Based on the bed porosity of 33% it was calculated that the water saturation level in the VF-CW was 48 cm (60% of the bed volume) and 64 cm (80% of the bed volume) in periods C and D, respectively.

The bed was fed manually using a pitcher and the entire daily batch was introduced over a short time of approx. 10 min. The HLR was increased in period B by diluting the waste water in order to reduce HRT in the FEM-CW. However, it can be seen in Table 15.1 that HRT in fact slightly increased due to significant water loss due to rapid growth of giant miscanthus in the VF-CW.

15.2.2 *Sampling Procedure*

The effluent from the VF-CW was collected by draining the entire batch into a container from which the sample was taken. Then the remaining volume was fed into the FEM-CW tank. The samples of the waste water treated in the FEM-CW unit were taken by withdrawing 1 L of the waste water by opening the outlet valve. All the samples were filtered using qualitative filter paper (MN 615 filter for medium fast filtration from Macherey-Nagel GmbH & Co. KG, Germany) prior to analysis.

15.2.3 *Wastewater Analysis*

N-NH₄ was analysed photometrically using 1.00683.0001 Spectroquant Ammonium Test from Merck KGaA (Germany). COD was determined photometrically using 1.14540.0001 and 1.14691.0001 Spectroquant COD Test from Merck KGaA (Germany). Two measuring ranges were applied - 100–1500 mg/L and 10–150 mg/L. DOC concentration was determined by TOC analyser Shimadzu TOC–L. DOC was analysed using the non-purgeable organic carbon method. DOC was not analysed in period A of the experiment.

Manganese (II) ions were determined using 1.4770.0001 Spectroquant manganese test from Merck KGaA (Germany). The Mn²⁺ concentration was measured only in periods C and D. Oxidized nitrogen forms concentration was presented as a sum of N-NO₃ and N-NO₂, because N-NO₂ was found at very low concentrations almost throughout the entire experiment with the exception of period A of the experiment. N-NO₃ was analysed photometrically using 1.09713.0001 Spectroquant nitrate test from Merck KGaA (Germany). N-NO₂ was analysed using 1.4776.0001 Spectroquant nitrite test from Merck KGaA (Germany). P-PO₄ was analysed photometrically using 1.00798.0001 Spectroquant.

The oxidation-reduction potential (ORP) value was measured using an ORP HI98201 m from the Hanna Instruments (USA). This parameter was measured in the effluents from the VF-CW and FEM-CW. The ORP was measured only in periods C and D. pH was measured using an electrode connected to a pH-meter WTW 330.

Sulphate concentration was determined using 1.14548.0001 Spectroquant sulphate cell test purchased from Merck KGaA (Germany). From day 358 of the experiment sulphate was determined using 1.14791.0001 Spectroquant sulphate test from Merck KGaA (Germany). The sulphate concentration was measured only in period C and D. Total nitrogen (TN) concentration was calculated as a sum of TKN (total Kjeldahl nitrogen, results are not presented in this paper), N-NO₃ and N-NO₂ (nitrite nitrogen) in both the influents and the effluents. TKN was determined using the Kjeltex System from Foss Tecator according to the method reported in Nowrotek et al. (2016). Due to servicing, the Kjeltex apparatus was not used in days 228–350

and the cuvette TN test 1.14763.0001 (Spectroquant nitrogen (total) cell test purchased from Merck KGaA, Germany) was used instead.

The inflow and outflow water volumes for the VF-CW were measured by graduated pitcher with an accuracy of ± 0.1 L. The outflow water volume for the FEM-CW corresponded to the volume that was had to be withdrawn to maintain constant water depth in the tank.

15.2.4 Statistical Analysis

The Shapiro-Wilk W test was employed to test for normality of the data. Most of the subsets of the performance data had non-normal distribution, therefore median and the median absolute deviation (MAD) were used as descriptors of the central tendency and dispersion of data distributions. Normally distributed data sets (as independent groups) were compared by the use of Student's t -test and non-normally distributed data sets (as independent groups) were compared using the Mann-Whitney U test. Differences were considered statistically significant if $p < 0.05$. Statistical testing was performed using the STATISTICA 10 software (StatSoft, Inc. 2011).

15.3 Results and Discussion

The efficiency of the treatment in terms of standard waste water quality parameters is listed in Table 15.2. Additionally, water loss as a measure of evapotranspiration is given. All the removal efficiency values (except for pH and water loss) were calculated for the VF-CW (influent against VF-CW effluent), for the FEM-CW (VF-CW effluent against the FEM-CW effluent) and for the entire system (influent against FEM-CW effluent).

It can be seen that the removal of organic matter determined as COD in the VF-CW was affected by the shift from the unsaturated to saturated conditions, which is due to the fact that the rate of biodegradation processes is lower under anoxic conditions (Kadlec and Wallace 2009). The high COD concentration in the FEM-CW effluent in period D could have been affected by high COD values in period C, which were reflected in the quality of the FEM-CW effluent with a time shift due to very low dilution rate. The other reason might be that the installation of a new air pump and a new diffuser on day 350 with much higher air flow resuspended the bottom sediments, which as a result increased the concentration of organic matter in the waste water. It should be also noted that this high value of COD might be an artefact as two different values (80 and 246 mg/L on average) were obtained during the measurements depending on the measuring range. Based on the value of the absorbance it was decided that the higher value was more trustworthy (for the lower COD value the absorbance was > 1). In the other periods, this was not a case,

Table 15.2 Influent and effluent concentrations (mg/L; median±MAD) and removal efficiencies (RE; %; median±MAD) of the standard waste water parameters for all the treatment steps, and influent and effluent water volumes, and water loss (L, median±MAD) for the VF-CW

Parameter	Period	Influent	VF-CW effluent		FEM-CW effluent		Total RE [%]
			Conc. [mg/L]	RE [%]	Conc. [mg/L]	RE [%]	
COD	A, n=4	815±3	138±30	83±4	25±4	77±4	97±0
	B, n=5	380±174	63±34	81±5	43±19	13±19	86±2
	C, n=3	783±35	526±126	30±4	57±20	91±2	94±2
	D, n=4	930±49	320±29	63±4	238±23	31±24	75±10
DOC	A, n=0	–	–	–	–	–	–
	B, n=10	164.2±18.6	13.2±5.0	92±3	12.5±1.7	11±38	92±1
	C, n=3	355.3±60.1	159.4±91.4	46±35	16.5±0.3	90±8	95±1
	D, n=4	372.1±3.6	25.8±9.8	93±3	17.3±0.9	21±29	95±0
N-NH ₄	A, n=4	123.5±18.5	39.0±12.5	69±10	7.5±2.5	80±4	91±16
	B, n=4	25.0±7.0	6.0±1.0	76±5	3.0±1.0	50±17	92±4
	C, n=4	60.3±24.3	43.0±4.8	37±29	6.3±1.8	83±7	90±7
	D, n=4	23.0±2.0	17.2±5.9	30±21	7.1±1.6	57±6	74±2
N-NO ₂ +N-NO ₃	A, n=4	2.7±0.7	75.0±19.0	-3440±1347	37.1±3.9	45±16	-1255±156
	B, n=7	2.8±0.6	58.5±12.3	-2286±505	38.1±5.0	40±12	-1168±290
	C, n=3	1.7±0.7	4.2±0.7	-54±95	7.0±3.5	-45±61	-577±370
	D, n=4	2.3±0.3	11.3±3.8	-327±62	1.7±0.6	83±5	40±8
TN	A, n=4	158±5	113±9	16±5	38±5	71±4	76±6
	B, n=5	79±14	72±6	2±6	37±3	49±3	53±12
	C, n=3	220±20	44±1	76±3	12±2	77±1	94±1
	D, n=4	155±7	32±4	81±2	14±4	57±4	93±1
P-PO ₄	A, n=4	34.8±0.8	28.1±4.3	29±5	13.2±1.1	51±4	64±2
	B, n=8	13.1±4.2	12.8±3.8	15±10	8.2±1.8	29±8	38±13
	C, n=3	33.0±12.9	7.6±0.4	64±8	8.2±1.9	-14±6	59±12
	D, n=4	24.1±1.8	8.2±2.2	66±7	5.6±1.1	30±17	74±7
pH	A, n=4	7.29±0.12	7.16±0.46		7.44±0.08		
	B, n=9	7.93±0.20	8.10±0.18		8.45±0.16		
	C, n=3	7.99±0.26	7.84±0.22		8.29±0.19		
	D, n=4	7.93±0.12	8.41±0.12		8.81±0.05		
Water loss	A, n=3	11.0±0.0 L	12.2±2.0 L	-9±20			
	B, n=2	18.0±0.0 L	10.6±2.9 L	41±16			
	C, n=7	45.0±0.0 L	13.4±5.3 L	70±12			
	D, n=4	60.0±0.0 L	29.5±1.5 L	51±3			

because only one measuring range was valid. The COD analysis could have been interfered by the presence of ferrous iron, heterocyclic compounds or readily volatile hydrocarbons, which could have been present in the effluent from the VF-CW and FEM-CW because of the reducing conditions, presence of plant exudates or specific by-products of microbial degradation (Merck 2013). It is therefore more meaningful to characterize the removal of organic matter using DOC. It can be seen

that the VF-CW displayed very high removal of DOC (>90%) when the bed was operated in an unsaturated mode (period B) and when the bed was artificially aerated (period D). However, in the period C when the VF-CW was operated in a saturated mode without aeration the removal was markedly lower than in period B (at $p < 0.05$), which was due to the fact that organic matter degradation rate is lower under anoxic conditions. The effluent concentration of DOC in the FEM-CW was invariably below 20 mg/L, which corresponded to very high overall removal efficiency (>92%).

The removal efficiency of N-NH_4 appears to be affected by the bed saturation as the removal in periods A and B when the bed was operated in an unsaturated mode is approx. 70% (no statistically significant differences) and in periods C and D when the bed was partially saturated the removal was approx. two-fold lower. The shift from unsaturated to saturated conditions in the VF-CW decreased the nitrification efficiency, but the increased amount of N-NH_4 could have been also a result of dissimilatory nitrate reduction to ammonium, which was determined as an important pathway contributing to nitrate removal in CWs (Zhi et al. 2015). The combined concentration of the oxidized N species (N-NO_2 and N-NO_3) in the effluent from the VF-CW was clearly dependent on the bed saturation. In periods A and B when denitrification was not favoured by bed saturation the concentration of N-NO_2 and N-NO_3 was above 50 mg/L (removal efficiency -3440% and -2286%, respectively) and in periods C and D when the bed was partially saturated the concentration of oxidized N species was approx. 10 mg/L or lower (removal efficiency -54% and -327%, respectively). The concentration of N-NO_2 and N-NO_3 in the FEM-CW effluent was decreasing in the course of the experiment, which corresponded to the concentration in the VF-CW effluent. In period D the concentration of the oxidized N species in the influent and effluent of the whole system was almost equal and at very low level of approx. 2 mg/L.

Similarly, the saturated conditions favoured TN removal by the putative pathways of denitrification and dissimilatory reduction of nitrate to ammonium, which was noticed in period C and D, during which the TN removal efficiency was 76% and 81%, respectively. Noteworthy, these differences were not statistically significant. In contrast, the removal of TN was very low or negligible in periods A and B, when the VF-CW was operated in an unsaturated mode. For the TN removal, statistically significant differences were observed between saturated and unsaturated periods in any combination (period A and C, A and D, B and C, B and D). In any period the FEM-CW provided additional removal of TN in the range of 49–77%. The overall maximum removal efficiency for TN was achieved in periods C and D and was $\geq 93\%$.

The performance of the presented hybrid system for P-PO_4 was in the range from 38 to 74%, which is in the range reported in the review by Vymazal (2013) for various hybrid systems combining subsurface flow CWs with free water surface systems. It can be clearly seen that under saturated conditions the removal of P-PO_4 was increased as compared to unsaturated conditions from 15–29% to 64–66%. Interestingly, partially flooded bed and the corresponding reducing conditions in period C (Table 15.3) did not trigger phosphorus release due to solubilisation of iron

Table 15.3 Influent and effluent concentrations (mg/L; median±MAD) and removal efficiencies (RE; %; median±MAD) of sulphates, manganese (II) ions, and ORP (mV)

Parameter	Period	Influent	VF-CW		FEM-CW		Total RE [%]
			Conc. [mg/L]	RE [%]	Conc. [mg/L]	RE [%]	
SO ₄ ²⁻	C, n=2	102±6	213±102	-104±88	182±24	-4±39	-77±13
	D, n=4	99±13	241±63	-75±75	210±35	9±10	-76±19
Mn ²⁺	C, n=2	<0.50±0.00	10.35±8.25	-1970±1650	<0.50±0.00	87±11	0±0
	D, n=4	<0.50±0.00	0.64±0.14	-27±64	<0.50±0.00	18±18	0±10
ORP, mV	C, n=4		-162±10		50±28		
	D, n=4		75±14		57±5		

minerals and associated phosphorus co-precipitates (Kadlec and Wallace 2009). On the contrary, despite low ORP values (Table 15.3) the P-PO₄ release or decrease of its removal efficiency was not observed as e.g. in the study of Kim et al. (2015), in which the removal P-PO₄ was decreased with increasing saturation level and was released in completely flooded VF-CW system. The duration of the partly and fully saturated phase in the study of Kim et al. (2015) was comparable and also the saturation level was similar: 30 cm – 70 cm (fully flooded) in Kim et al. (2015) and 40 cm (period C) and 64 cm (period D) in the present study. The results obtained in this study may suggest that the enhanced removal efficiency of P-PO₄ was due to increased saturation and the extended retention time, which was 1 week, which in turn probably stimulated plant uptake of P-PO₄. The removal of P-PO₄ in the FEM-CW was less pronounced than in the VF-CW during periods C and D. This is probably due slower development of the plant biomass in the FEM-CW as compared to rapid and riotous growth of miscanthus in the VF-CW. It should be mentioned that, however, the FEM-CW provided additional removal of P-PO₄ in the range of 29–51 %, with the exception of period C. In order to better illustrate the contribution of the VF-CW and FEM-CW to the removal of TN and P-PO₄ relative removal efficiency was calculated as the proportion of the concentration removed in a single step to the overall concentration removed (Fig. 15.2). The onset of the saturated conditions in the VF-CW significantly shifted the balance between the contribution of the treatment units as the sinks for TN and P-PO₄. For both contaminants, the contribution of the VF-CW to their removal was low in periods A and B, however, after the onset of the saturated conditions in the VF-CW this subsystem played the predominant role as the sink of TN and P-PO₄.

The differences between the pH value in the influent and VF-CW effluent were not statistically significant with the exception of period D when the pH value increased by 0.5 during the treatment, which can be attributed to the alkalinity production in the denitrification process (alkalinity was not measured). The increase of the pH value observed in the FEM-CW in the course of the experiment can be attributed to CO₂ consumption by the phytoplankton. The water loss value due to evapotranspiration increased from slightly negative value in period A to 41 % in period B, which should be attributed to the rapid growth of miscanthus that was planted during

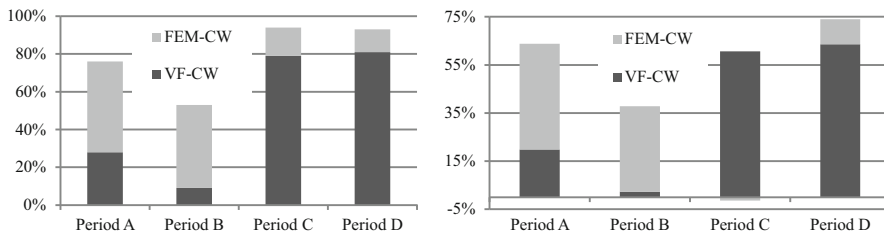


Fig. 15.2 Relative contributions of the treatment units to the overall removal efficiency for TN (*left*) and P-PO₄ (*right*)

this period. In periods C and D it further increased to 70 and 51 %, respectively, due to continuous development of vegetation and also increased contact between the rhizosphere and the treated waste water due to the operation mode. The observed higher water loss in period C (August and September) than in period D (October) was probably due to higher ambient temperature associated with the seasonal changes. Comparing the results presented in Table 15.2 with the performance data of hybrid constructed wetlands with FWS step collected and elaborated by Vymazal (2013) it was observed that the performance of the presented hybrid system was especially high in terms of TN and P-PO₄ (TP was reported by Vymazal 2013) removal, which was higher than the results for 86 % of the systems (6 out of 7) presented in Vymazal (2013). The performance of the system discussed in this chapter for other parameters was in the middle of the range. This comparison was made using the data for period D.

Apart from the parameters routinely measured to characterize the quality of treated waste water additional parameters that reflect oxidation-reduction conditions were determined. These parameters were the concentration of SO₄²⁻ and Mn²⁺ and also the value of ORP and were determined only in periods C and D (Table 15.3), this is before and after the VF-CW aeration was introduced.

As can be seen from Table 15.3, partial flooding of the bed in the VF-CW did not stimulate sulphate reduction, despite ORP values typically occurring in sulphate reduction zone (Reddy and DeLaune 2008), but triggered Mn²⁺ release, because Mn(IV) is an electron acceptor with higher reduction potential than SO₄²⁻. The observed release of SO₄²⁻ in period C and D cannot be elucidated without additional analysis, but it can be hypothesized that it was related to desorption of this anion or rewetting of the filtration material that was not subjected to prolonged contact with waste water percolating through the bed in periods A and B (Whitfield et al. 2010). It seems less likely that the sulphate release was caused by reoxidation of sulphides present in the VF-CW, because this system was operated in a mode preventing sulphate reduction to sulphides. The aeration of the VF-CW increased the ORP value of the effluent by more than 200 mV, which also translated to much less pronounced Mn²⁺ release.

The removal rates and percentage mass removal values are given in Table 15.4 for the VF-CW and the entire system. The removal rates for the entire system are generally lower than for the other hybrid CWs including FWS stage discussed in the

Table 15.4 Treatment performance data in periods B and D of the system operation

Parameter	Period	VF-CW			VF-CW and FEM-CW	
		Loading rate VF-CW, g m ⁻² d ⁻¹	Removal rate VF-CW, g m ⁻² d ⁻¹	Percentage mass removal, %	Removal rate VF-CW, g m ⁻² d ⁻¹	Percentage mass removal, %
COD	B	48.4	46.0	95.1	53.0	96.7
	D	27.5	22.5	81.8	3.77	84.5
DOC	B	20.9	20.4	97.6	3.2	97.7
	D	11.3	10.9	96.4	1.7	97.6
TN	B	10.0	7.4	73.2	1.4	86.2
	D	4.7	4.2	89.2	0.7	95.5
N-NH ₄	B	3.2	3.0	92.9	0.49	96.5
	D	0.7	0.4	61.5	0.1	84.1
P-PO ₄	B	1.7	1.2	71.2	0.2	81.7
	D	0.7	0.6	82.3	0.1	88.1

review of Vymazal (2013). This is due to the fact that the presented system was not optimized in terms of the HRT value in the FEM-CW and the batch duration in the VF-CW, which were both relatively long compared to commonly applied values. The percentage mass removal values in the VF-CW were compared with the results obtained by Boog et al. (2014) for intermittently (8 h on/4 h off) aerated unplanted VF-CWs. It was observed that the values percentage mass removal were higher in both periods (Table 15.4) for DOC (TOC was reported in Boog et al. 2014), in period D for TN, and in period B for N-NH₄. The percentage mass removal reported by Boog et al. (2014) for these parameters was 92.2, 78.0 and 89.2%, respectively.

Based on the population equivalent (pe) 80 g COD/pe (Langergraber et al. 2010) and the data presented in Tables 15.2 and 15.4, it was calculated that the sizing criterion would be 10.4 and 18 m²/pe for the whole system, and 1.7 and 2.9 m²/pe only for the VF-CW (periods B and D, respectively), which is within the range presented in the literature 1.5–20 m²/pe as reported by Kadlec and Wallace (2009), Kim et al. (2015) and Boog et al. (2014).

15.4 Conclusions and Outlook

The intensification of the CW performance by integrating a VF-CW with partly saturated bed and a FEM-CW can ensure high removal efficiencies for TN and P-PO₄. The application of artificial intermittent aeration in a VF-CW can additionally allow achieving high removal efficiency of carbonaceous compounds (determined as DOC) without compromising the removal of TN and P-PO₄. The underlying removal mechanisms were not studied but the putative processes were predominantly microbial nitrification-denitrification and plant uptake (especially in the

FEM-CW) for the TN removal, and plant uptake for the P-PO₄ removal. The observed low removal rates (expressed as g/m²d) are attributed to long overall retention time in the system, which was not determined experimentally and will be further optimized. The development of vegetation on the floating mat in the FEM-CW appears to be lengthy process requiring expertise-based selection of plant adapted to specific conditions of the floating buoyant structure. In the presented experiment it was observed that yellow flag (*Iris pseudoacorus*) adapted best to growing on a floating structure in the FEM-CW, which was based on its growth rate and amount of biomass. The establishment of vegetation in the VF-CW was less challenging as giant miscanthus was successfully applied to replace common reed, which was desiccated after a few weeks of operation. The future tasks will include further optimization of the system, but also monitoring of the removal and fate of selected organic micropollutants. It is assumed that the broad set of abiotic and biotic factors influencing the removal of pollutants in the VF-FEM-CW might be particularly attractive for the removal of these compounds.

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Chapter 16

Treatment of Chlorinated Benzenes in Different Pilot Scale Constructed Wetlands

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Abstract Chlorinated benzenes (CBs) are common pollutants in groundwater due to their broad usage in industry and agriculture. Remediation of CBs from contaminated groundwater is of great importance. Biodegradation has proved to be a suitable approach in eliminating CBs from polluted water, and constructed wetland (CW) is an alternative as cost efficient technology to remove CBs from wastewater. In the present study, a comparison covering five growing seasons (from May 2006 to November 2010) was carried out among four pilot-scale CWs: (1) unplanted horizontal subsurface flow (HSSF) CW; (2) planted HSSF CW; (3) planted HSSF CW with tidal flow; (4) hydroponic root mat (HRM). The unplanted HSSF CW was not efficient for CBs removal, with removal efficiency less than 23 % for the four CBs, and no capability to remove 1,2-DCB. Planted HSSF CW exhibited significantly better treatment performance than the unplanted HSSF CW, and the CBs removal efficiency can be enhanced to some extent (especially after 3 m from the flow path) when running under tidal flow operation. Highest CBs removal efficiency was reached in the HRM system, with mean removal rates for monochlorobenzene, 2-chlorotoluene, 1,4-dichlorobenzene (DCB) and 1,2-DCB were 219, 0.92, 7.48 and 0.86 mg/m²/d, respectively. In conclusion, the HRM is the best variant CW to treat chlorinated benzenes, and it can be an option for the treatment of pollutants which prefer aerobic degradation.

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Keywords Chlorobenzenes • Constructed wetland • Groundwater • Hydroponic root mat • Tidal flow

16.1 Introduction

Monochlorobenzene (MCB) is mainly used as a chemical intermediate in the manufacture of organic chemicals, manufacture of pesticides and degreasing solvent in automobile parts. Dichlorobenzene (DCB) isomers are mainly used in the production of medicine, dyes, herbicides and insecticides, normally, they are not found naturally. The widely used chlorinated solvents such as MCB and DCBs are highly persistent in aerobic and anaerobic circumstances, due to improper handling and storage, they are commonly detected pollutants in sediment and groundwater. Most of the chlorinated benzenes (CBs) are toxic to human which have the potential to cause liver, kidney and central nervous system damage. Some of them are listed on US EPA priority pollutants for drinking water regulation, e.g., maximum contaminant level for MCB, 1,2-DCB, and 1,4-DCB are 0.1, 0.6, and 0.075 mg/L, respectively (USEPA 2002).

Constructed wetlands (CWs) have been widely used as an alternative to conventional intensive systems for different types of wastewater treatment because of their lower energy requirements, easy operation and maintenance (Vymazal 2011). In the past two decades, CWs were used for the clean-up of sediments and groundwater contaminated by CBs (MacLeod et al. 1999; Keefe et al. 2004; Braeckevelt et al. 2007a, b, 2008, 2011; Lee et al. 2009; Cottin and Merlin 2010; Schmidt et al. 2014; Chen et al. 2015). In CWs, the main pathways for chlorinated benzenes removal are microbial degradation, volatilization, plant uptake, and sorption (Kadlec and Wallace 2009; Pardue et al. 1999). Both aerobic and anaerobic MCB degradation pathway was convinced by the observation of carbon isotope shift in both the planted and unplanted horizontal sub-surface flow (HSSF) CWs (Braeckevelt et al. 2007b). It was shown that plants can enhance the removal of MCB, which higher MCB removal efficiency was achieved in the planted HSSF CW than in the unplanted HSSF CW (Braeckevelt et al. 2008). Moreover, the planted HSSF CWs obtained higher MCB removal efficiency in upper layer (Braeckevelt et al. 2007a). Volatilization of MCB amounted to only 2–4% of the total removed MCB (Braeckevelt et al. 2011). Thus, to create more aerobic condition in the CWs will be of great importance for the removal of MCB and DCBs. Tidal flow operation strategy was developed to increase oxygen input in CWs since the later 1990s. During the tidal flow operation, the wastewater acts as a passive pump to repel and draw oxygen into the CWs (Tanner et al. 1999; Green et al. 1997; Sun et al. 1999), therefore, it can improve the aerobic condition in CWs. Due to the lack of substrate, and easy cope with water level, floating hydroponic root mat system has been used for the treatment of different pollutants (Van de Moortel et al. 2010; Headley and Tanner 2012; Chen et al. 2015), including pollutants prefer aerobic degradation.

In the present study, two pilot scale HSSF CWs (one planted and one unplanted) was established to treat groundwater contaminated with CBs at the beginning. In order to enhance the removal performance of CBs, the unplanted system was changed into a hydroponic root mat, and the planted system was running under tidal flow regime to improve the aerobic condition in the HSSF CW. Therefore, the aim of this study was to compare the treatment performance of the four different CWs regarding the elimination of CBs.

16.2 Materials and Methods

16.2.1 Description of Pilot Scale CWs

The unplanted and planted HSSF CWs were established in March 2003 in Bitterfeld Germany. The dimension of the system is 6×1 m, filled with local aquifer material up to 0.5 m. The filling material consisted of gravel (25 %), sand (51 %) and silt/clay (3 %), and the hydraulic conductivity is $2.1 \times 10^{-3} \text{ m s}^{-1}$, porosity of 0.28. One of the CWs was planted with common reed (*Phragmites australis*), the other one was left unplanted as a control. More details of the HSSF CWs are described elsewhere (Braeckevelt et al. 2011; Chen et al. 2012). In March 2010, the unplanted HSSF CW was replaced by a hydroponic root mat (HRM), 3-years old well developed plant root mats from *P. australis*, with a porosity of 0.70. In 2010, the planted HSSF CW was run under 1 week tidal regime, which is a fast outflow flushing of 2 h, causing a rapid decrease in the water level from 0.4 to 0.15 m, a refilling (5.0 L h^{-1}) to water level of 0.4 m took about 34 h, and a further interim period of 132 h. Detail description of the HRM and the tidal flow regime were given previously (Chen et al. 2015). Mean inflow concentrations of CBs were given in Table 16.1.

16.2.2 Sample Collection and Analysis

In total, 43, 52, 6, 14 sampling actions were carried out for the unplanted, planted, planted with tidal flow HSSF CW and HRM, respectively. Pore water samples were taken at different distances (0.5, 1, 2, 3 and 4 m) from the inlet and at different depths (0.3, 0.4 and 0.5 m) in the HSSF CW, same distance but only one depth (20 cm) in the HRM. For the analysis of CBs, 10-mL water samples were collected in 20-mL glass flasks (Supelco, Bellefonte, USA), sodium azide solution was added to the samples to inhibit the microbial activity, and the flasks were sealed with Teflon-lined septa (Pharma-Fix-Septum Buty/PFTE 3.0 MM). The concentrations of the CBs were measured by headspace gas chromatography using a HP 6890 gas chromatograph with flame ionization detector (Agilent Technologies, Palo Alto, USA). For headspace analysis a gas volume of 1 ml was injected at an injection

Table 16.1 Mean concentration and load removal rate after 4 m of the inlet, standard deviation was shown in brackets

	Unplanted (n = 43)			Planted (n = 52)			Tidal flow (n = 6)			HRM (n = 14)		
	In mg/L	Out mg/L	Removal mg/m ² /d	In mg/L	Out mg/L	Removal mg/m ² /d	In mg/L	Out mg/L	Removal mg/m ² /d	In mg/L	Out mg/L	Removal mg/m ² /d
MCB	10.6 (1.8)	9.3 (2.2)	46 (58)	10.6 (1.7)	4.6 (2.4)	199 (49)	7.3 (0.7)	2.5 (2.2)	156 (46)	8.1 (1.2)	1.0 (1.2)	219 (40)
1,4-DCB	0.232 (0.067)	0.181 (0.053)	1.61 (1.31)	0.253 (0.074)	0.085 (0.037)	5.32 (2.07)	0.270 (0.023)	0.061 (0.036)	6.58 (0.94)	0.286 (0.037)	0.045 (0.048)	7.48 (1.21)
1,2-DCB	0.035 (0.010)	0.039 (0.013)	-0.08 (0.20)	0.036 (0.010)	0.023 (0.012)	0.53 (0.25)	0.032 (0.003)	0.014 (0.006)	0.61 (0.14)	0.035 (0.005)	0.007 (0.007)	0.86 (0.18)
2-Chlorotoluene	0.025 (0.009)	0.021 (0.007)	0.14 (0.16)	0.029 (0.011)	0.009 (0.005)	0.63 (0.27)	0.037 (0.003)	0.008 (0.004)	0.91 (0.12)	0.037 (0.004)	0.008 (0.007)	0.91 (0.17)

temperature of 250 °C with split 1:5 (measurements in duplicates). The chromatographic separation was achieved with an HP-1 capillary column (Agilent Technologies, Palo Alto, USA) (30 m length×0.32 mm inner diameter×5 µm film thickness) with the following oven temperature program: equilibration at 60 °C (60 min), GC oven program 45 °C (1 min), 20 °C/min to 200 °C (2.5 min), 65 °C/min to 250 °C (1 min), and the detector temperature was 280 °C. Helium was used as carrier gas with a flow rate of 1.7 ml/min. The detection limits were 1.3, 2.4, 1.2 µg/L, for MCB, 1,4-DCB, and 1,2-DCB, respectively.

16.2.3 Data Analysis

Removal efficiency based on load was calculated with the consideration of water loss by evaporation and plant transpiration. The decrease of the organic load along the flow path was calculated assuming the water loss followed a linear change. The flow distance and the depth related loads were calculated based on the assumption that the concentrations at each depth represent the concentration of the cross section through which the water of this presumed depth layer flows. Detail calculation equation was described previously (Chen et al. 2015). One Way ANOVA was run in order to test the significant difference on chlorinated benzenes load removal efficiencies at 4 m between the four systems, significant difference was set at $p < 0.05$.

16.3 Results and Discussion

16.3.1 MCB Removal

Significant difference in MCB load removal efficiency (at 4 m) was found between the unplanted HSSF CW and other three systems, and the HRM with other three systems, but not between the planted HSSF CW and the tidal flow HSSF CW (Fig. 16.1). The highest residual MCB mass along the flow path, which is significantly differ from the other three systems (Fig. 16.1), as well as lowest mean MCB load removal rate (46 mg/m²/d) was observed in the unplanted HSSF CW (Table 16.1). MCB mean removal efficiency increased from 15 % in the unplanted system to 66 % in the planted HSSF CW, and it increased further to 72 % when the planted HSSF CW operated in tidal flow regime. The highest MCB mean removal efficiency of 90 % was obtained in the HRM, with highest mean load removal rate of 219 mg/m²/d. Although there is no significant difference on load removal efficiency at 4 m between continuous flow and tidal flow in the planted HSSF CWs, the tidal flow operation can enhance MCB removal efficiency at the last half part of the wetland (after 3 m). No significant difference of MCB load was found among the three depths in the unplanted HSSF CW, but significant difference was found

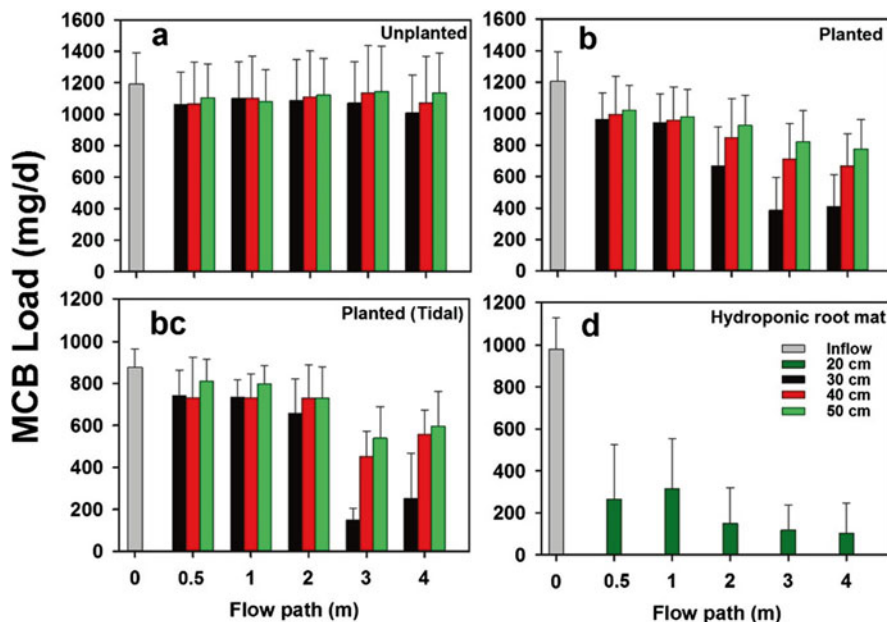


Fig. 16.1 Mean MCB mass load along the flow path in the four systems ($n=43, 52, 6, 14$ for unplanted, planted, tidal flow and HRM). “*abcd*” was shown as significant difference between the four system

regarding depth in the continuous flow and tidal flow planted HSSF CW, with lower MCB load was observed in the upper layer. This confirms that the vegetation has positive effect on MCB removal, which is due to the release of oxygen by the plant roots, and it thus influence the microbial community. However, anaerobic microbial degradation of MCB may be occur in all the four systems as it was proven that ferric iron and nitrate can stimulate the mineralisation of MCB (Schmidt et al. 2014). But, the MCB dechlorination process can be neglect because no benzene accumulation was detected in all the four systems.

Volatilization is a subordinate removal pathway for MCB removal in the HSSF CW, as it's calculated that maxima MCB emission rate is $13.5 \text{ mg/m}^2/\text{d}$, which is less than 4% of the total removed MCB (Braeckevelt et al. 2011). In a large scale surface flow CW, MCB emission rate of $4 \mu\text{g/m}^2/\text{d}$ was reported, which accounts for 89% of the total removal (Keefe et al. 2004). However, it should be note that the MCB inflow concentration is only $0.03 \mu\text{g/L}$ in that study, which is much lower than in previous study (17.6 mg/L). Higher MCB volatilization might be taken place in the HRM, as slightly higher benzene volatilization was observed in HRM (3%) than in the HSSF CW (1.1%) (Chen et al. 2012). This is due to the lack of soil coverage in the HRM compare to HSSF CW, where the soil can prevent the direct emission of pollutants from the pore water. Moreover, slightly higher MCB emission may be expected in the tidal flow HSSF CW due to the variation of water levels. Adsorption may take place at the beginning of the addition of CBs, sorption and

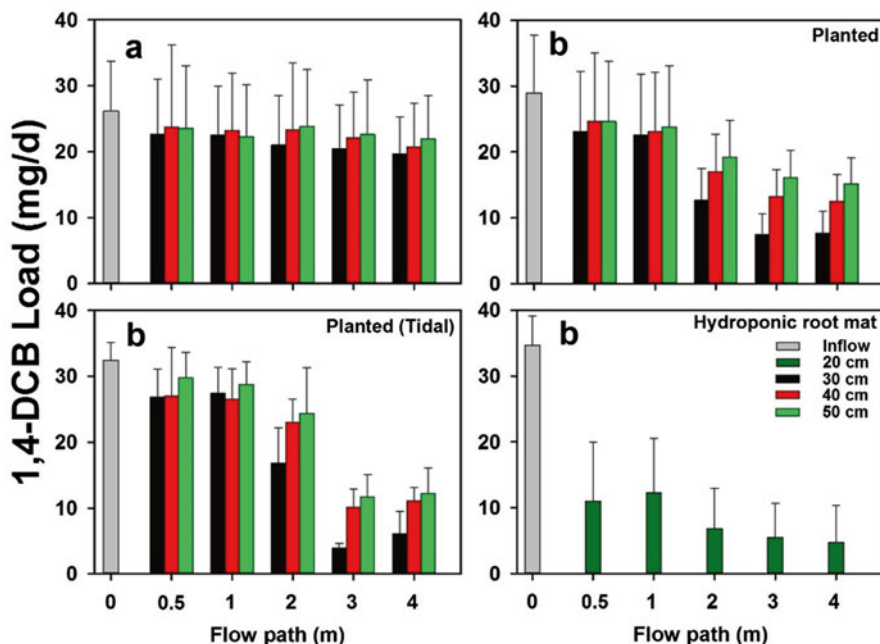


Fig. 16.2 Mean 1,4-DCB mass load along the flow path in the four systems ($n=43, 52, 6, 14$ for unplanted, planted, tidal flow and HRM). “abcd” was shown as significant difference between the four system

desorption-resistance on biodegradation will coexist afterwards. MCB initial mineralization rates of $0.14 \mu\text{g/L/h}$ and $1.92 \mu\text{g/L/h}$ was found in marsh soil and wetland soil respectively, which indicate that microbes associated with MCB degradation can access to the adsorbed MCB, even the desorption resistant part of the sorbed MCB also can be degraded (Lee et al. 2009).

16.3.2 DCBs Removal

There is significant difference in 1,4-DCB load removal efficiency (at 4 m) between the unplanted HSSF CW and other three systems, but no significant difference among the planted HSSF CW, the tidal flow HSSF CW and the HRM (Fig. 16.2). The mean load removal efficiency for 1,4-DCB are 23, 72, 81, and 87% in the unplanted HSSF CW, the planted HSSF CW, the tidal flow HSSF CW and the HRM, respectively. The residual 1,4-DCB mass decreased significantly along the flow path in the three planted systems, but not in the unplanted system. This testified that the plants can enhance 1,4-DCB removal. Moreover, our results give a higher concentration level for *Phragmites australis* to treat CBs, as other research only gives a concentration of 0.2 mg/L MCB and 0.2 mg/L DCB (Faure et al. 2012). The

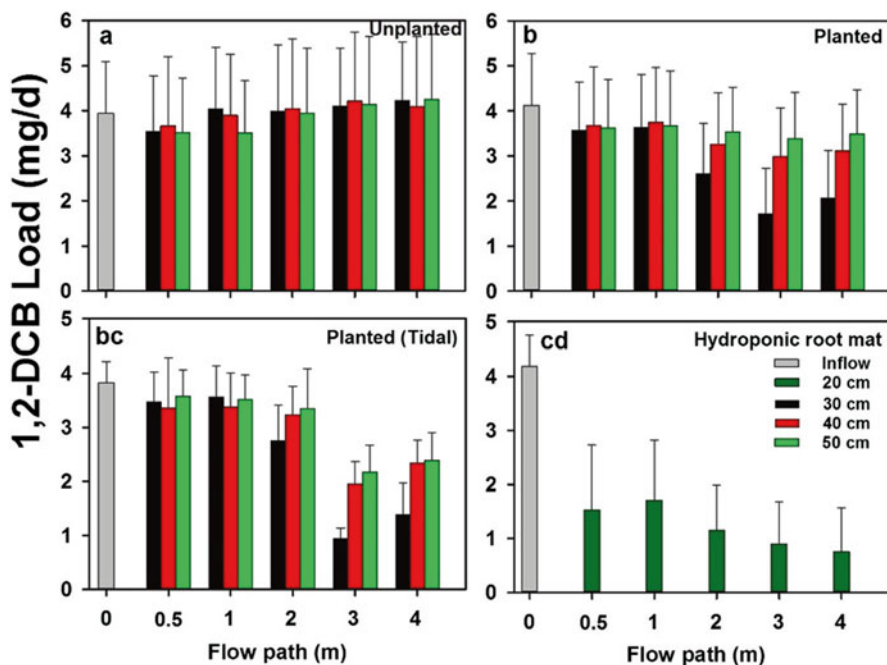


Fig. 16.3 Mean 1,2-DCB mass load along the flow path in the four systems ($n=43, 52, 6, 14$ for unplanted, planted, tidal flow and HRM). “*abcd*” was shown as significant difference between the four system

highest mean removal rate for 1,4-DCB is 7.48 mg/m²/d in the HRM, which is much higher than in a surface flow CWs reached 0.11 mg/m²/d with inflow concentration of 0.74 µg/L (Keefe et al. 2004). Moreover, the volatilization of 1,4-DCB is extremely high which can contribute to 95 % of the total 1,4-DCB removal in the surface flow CW (Keefe et al. 2004). This is due to the direct volatilization from water surface, while such extensive volatilization is not expected to take place in our systems because they are subsurface flow with the coverage of soil and density plant root mat.

Load removal efficiency of 1,2-DCB is significantly different between the unplanted HSSF CW and other three systems, between the HRM with the unplanted and planted HSSF CW, but no significant difference between the tidal flow HSSF CW and the HRM, and no significant difference between the planted HSSF CW and the tidal flow HSSF CW (Fig. 16.3). The results showed that 1,2-DCB is difficult to be removed than 1,4-DCB in CWs. It seems that 1,2-DCB almost can't be removed in the unplanted HSSF CW, and mean removal efficiency are 50, 64 and 83 % for the planted HSSF CW, the tidal flow HSSF CW and the HRM, respectively. In principal, DCBs can be dechlorinated to MCB under anaerobic condition. However, this process is negligible in our study due to the relatively lower DCBs concentrations than MCB concentration in the influent.

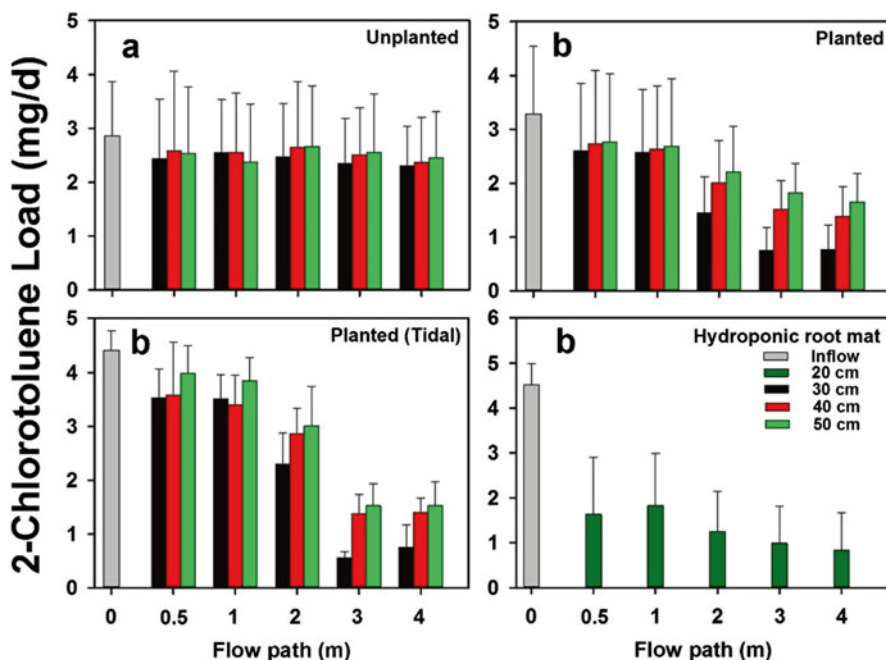


Fig. 16.4 Mean 2-Chlorotoluene mass load along the flow path in the four systems (n=43, 52, 6, 14 for unplanted, planted, tidal flow and HRM). “abcd” was shown as significant difference between the four system

DCBs may be less volatile than MCB due to the lower Henry’s coefficients (20 °C) for 1,2-DCB, 1,4-DCB, and MCB are 0.0012, 0.0015 and, 0.00356 atm m³/mol, respectively (USEPA 2002). Therefore, volatilization for 1,2-DCB and 1,4-DCB in the all the four systems may only account for small portion of the total removal. It was reported that less than 0.15% of DCB was removed through volatilization in lab scale CWs (Cottin and Merlin 2010). Plant uptake is a potential pathway for CBs removal. A bioconcentration factor of 14 was reached for plant uptake of 1,4-DCB after 7 days, and uptake/translocation of CBs are depend on the CBs hydrophobicity, solubility and volatility (San Miguel et al. 2013). Therefore, a higher plant uptake of CBs in the HRM than in the planted HSSF CWs can be expected, especially in the roots which have direct contact with the wastewater.

16.3.3 2-Chlorotoluene Removal

The behaviour of 2-chlorotoluene in the four systems are similar as 1,4-DCB, which the load removal efficiency (at 4 m) is significant different between the unplanted HSSF CW and other three planted systems, but no significant difference between the three planted systems (Fig. 16.4). The mean removal efficiencies of

2-chlorotoluene in the four systems are comparable to MCB's removal efficiencies, with 16, 76, 83, and 82% in the unplanted HSSF CW, the planted HSSF CW, the tidal flow HSSF CW and the HRM, respectively. In general, the biodegradation of the chlorinated benzenes is slower under anaerobic than under aerobic conditions. Therefore, the three plant systems exhibited significantly higher removal efficiency than the unplanted system, and lower concentration was detected in the upper layer than in the deeper layer due to the oxygen release by plants.

16.4 Conclusions

It can be concluded that plants can enhance the removal of chlorinated benzenes in HSSF CW, and tidal flow operation is a useful approach to enhance the treatment efficiency of CBs. The HRM system obtained the highest removal efficiencies for most of the CBs, and due to the lack of substrate, it can be a cost-efficient alternative technology for wastewater treatment, especially for pollutants prefer aerobic degradation.

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Chapter 17

Transformation of Chloroform in Constructed Wetlands

Yi Chen, Yue Wen, Qi Zhou, and Jan Vymazal

Abstract Chloroform is a volatile organic contaminant widely detected in groundwater, surface water and wastewater effluent, thus its fate in the natural treatment systems is of great importance to the environment and human's health. In this study, the transformation processes were studied for six model constructed wetlands (CWs), for treating chloroform in the secondary effluent. Contaminant fate was investigated in the respective water, plant, litter, gravel and atmosphere. Results showed that sorption and biodegradation were the main chloroform removal processes in litter-added CWs, while sorption and plant uptake were the primary contributors to chloroform removal in planted CWs. Volatilization flux of chloroform was always low (2.0–2.5 %) in CWs likely due to the limitation of water–air transfer via diffusion in the SSF CWs. Overall, this study makes the quantitative evaluation of chloroform distribution (i.e. aqueous phase, gaseous phase, vegetation, and bed substrate) and multiple removal pathways (i.e. destructive versus nondestructive processes) in CWs. This will benefit for opening the “black box” of the treatment processes and make people better understand the transformation of trace volatile organic pollutants in natural treatment systems.

Keywords Constructed wetlands • Chloroform • Transformation • Plant litter • *Typha latifolia*

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17.1 Introduction

Chloroform is a volatile organic contaminate widely detected in the environment. Previous studies showed that it is frequently detected in groundwater, wastewater effluent and surface water, thus its fate in the natural treatment systems is of great importance to the environment and human's health. Considering the slow degradation and great health concern of chloroform, its fate and remediation approach is a hot topic (Carter et al. 2012; Chan et al. 2012; Justicia-Leon et al. 2014). Gupta et al. (1996b) found that the removal efficiency of chloroform could achieve 99% in the methanogenic enriched culture under anaerobic condition. In aerobic environment, some microbes can also transform chloroform to CO₂ using different primary substrates (Kim et al. 2000; Wahman et al. 2007; Frascari et al. 2012). However, the high cost of the biotechnology limited the wide application in the field.

Constructed wetlands (CWs) are ecosystems connecting between terrestrial and aquatic systems, and it acts as a "lung of earth" to store and purify the contaminants in the wastewater (Vymazal 2011a). Various types of wastewater and contaminants have been successfully treated in different types of CWs (Vymazal 2007). However, the fate of chloroform has not been evaluated in the wetland systems due to the complexity of wetlands. The wetland systems are ecosystems where four spheres (hydrosphere, biosphere, lithosphere and atmosphere) overlap and thus more heterogeneous and complicated than other natural and engineered systems. The presence of plant which can interact with these spheres makes the wetland systems even more complicated and unique (Vymazal 2011b). Thus, the transformation of contaminants and the related treatment process in wetland systems is always considered as a "black box" for decades (Garcia et al. 2010). This is especially the case for the chloroform investigated in this study, because its trace concentration (ng/L- μ g/L) and highly volatile nature make the quantitative evaluation of pollutants distribution (i.e. aqueous phase, gaseous phase, vegetation, and bed substrate) and multiple removal pathways (i.e. destructive versus nondestructive processes) analytically difficult. As mentioned above, the investigation of chloroform removal in CWs will benefit for opening the "black box" of the treatment processes and make people better understand the transformation of trace volatile organic pollutants in natural treatment systems.

The objective of this study was to assess the fate of chloroform in different lab-scale CWs, including unplanted and planted CWs with or without plant litter. Furthermore, a mass balance of chloroform in CWs was established.

17.2 Materials and Methods

17.2.1 Design and Operation of the SSF CW

The subsurface flow (SSF) CW microcosms (length: 0.3 m, width: 0.3 m, height: 0.5 m) were located in a temperature controlled greenhouse. These CWs were operated as batch systems for 5 years to treat the nitrate and sulfate in the secondary effluent. Details of the microcosm design were illustrated in our previous study (Lin et al. 2008; Wen et al. 2010). All the microcosms were filled with gravel (ϕ 8–13 mm, porosity=0.4) and planted with cattail (*Typha latifolia*). Cattail litter was cut into pieces (1–2 cm) and was used as a potential bio-adsorption material and electron donor for the reductive de-chlorination of chloroform in this study. The collection and preparation of the cattail litter has been described in our previous literatures (Chen et al. 2011).

Six sequencing batch SSF CW microcosms were applied in the study. i.e. unplanted and non-litter added unit (W0), unplanted and litter-added unit (W1, 100 g cattail litter), unplanted and double litter-added unit (W2, 200 g cattail litter), planted and non-litter added unit (W3, 22 plants/m²), densely planted and non-litter added unit (W4, 40 plants/m²), planted and litter-added unit (W5, 22 plants/m², 100 g cattail litter). The feed water of the CWs was the secondary effluent from a neighboring wastewater treatment plant with chlorine disinfection. The concentration of the chloroform was $52 \pm 18 \mu\text{g L}^{-1}$. The batch wetland microcosm was fed with the secondary effluent every 5 days (nominal HRT 5d), and the water level was set to 45 cm, resulting in a vadose zone of 5 cm. All treatments (W0~W5) were triplicated and the duration of each experiment was 100 days which included 20 periods (each period lasted for 5d).

17.2.2 Sampling Procedure and Analysis of Aqueous, Solid and Gaseous Samples

Water samples were collected between 0 and 120 h every 5 days. Water samples were collected from each microcosm at a depth of 20 cm using a 100 ml syringe, and were analyzed in duplicate for chloroform concentration by a gas chromatograph (GC) (Agilent 7890A) with an electron capture detector (ECD). The column used was an HP-5 phenyl Methyl Siloxan column (30 m \times 0.32 mm I.D. with 0.25 μm film thickness). The injector, ECD and GC oven temperature programs for chloroform were: injector of 200 °C; ECD of 300 °C; oven of an initial temperature of 45 °C for 2 min, ramping to 120 °C at 8 °C/min and holding for 2 min. Passive samplers (RADIELLO®, RAD130, Supelco) were used to measure the emission of chloroform in each wetland microcosm. According to the preliminary experiments, the measurements were performed at 10 cm in the air above the centre of each wetland surface for 5 days of exposure time. Background concentrations of chloroform

outside the greenhouse were also measured during the 5 days of exposure time. The RADIELLO® passive sampler was composed of a diffusive body (a semi-permeable membrane) and an adsorbing cartridge packed with activated charcoal with particle size 35–50 mesh. After sampling, chloroform trapped in the adsorbing cartridge was recovered by carbon disulfide, analysis was performed by GC-ECD system. For solid samples, the adsorbed chloroform was extracted using hexane and acetone mixture (50:50) for 5 min, and 1.5 mL of the supernatant was used for analysis.

17.3 Results and Discussion

17.3.1 Sorption

In order to reveal the effect of sorption on chloroform removal in CWs, the sorption rates of gravel, litter and plant were investigated in this study. As shown in Fig. 17.1, all the components of CWs (gravel, litter and plant) could adsorb chloroform to some extent. The litter-added CWs showed the highest sorption rates, followed by the planted and control CWs. Plant litter and gravel was found to be the main sorption sites of chloroform in CWs. This was likely due to the hydrophobic interactions between chloroform and organic carbon attached to litter and gravel (biofilm). Previous studies also found that organic-rich biomass had a good performance for chloroform sorption (Adachi et al. 2002; Cunha et al. 2010). Compared with the blank system (W1), the sorption driven by gravel was lower in litter added CWs. This was probably due to the competitive sorption of chloroform between litter and gravel, which decreased the sorption by gravel.

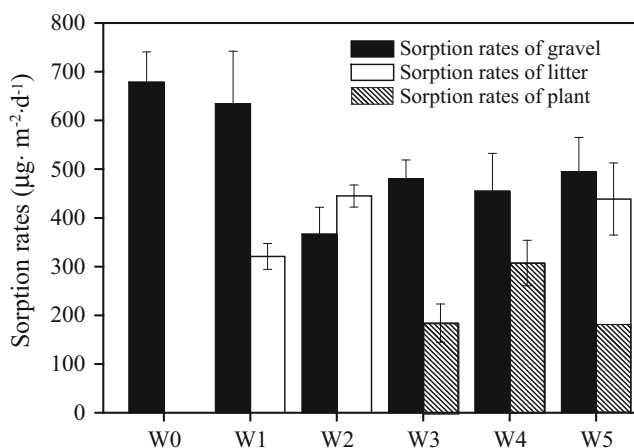


Fig. 17.1 The sorption rates of chloroform on gravel, litter and plant in W0~W5 microcosms

Table 17.1 Total mass and concentration of chloroform on plant tissue in W3 and W4 microcosms

		Leaves	Stems	Roots
Mass (μg)	W3	284.5	291.6	1079.9
	W4	485.9	529.9	1749.8
Concentration ($\mu\text{g/g}$)	W3	27.6	18.7	65.8
	W4	38.9	29.0	88.4

17.3.2 Plant Uptake

As shown in Fig. 17.1, the plant uptake contributes to the chloroform removal in CWs. Previous studies indicated that the organic contaminants were removed by plant through root uptake and transpiration-driven to aboveground biomass (Imfeld et al. 2009). In general, organic contaminants with high $\log K_{ow}$ benefit for their sorption on roots but hinder the transportation inside the plants. Previous studies have suggested that direct uptake of organics by plants via transpiration is generally indicated by low to intermediate $\log K_{ow}$ values ranging from 0.5 to 3 (Seeger et al. 2011). In this study, chloroform is a compound of moderate hydrophobicity with $\log K_{ow}$ value of 1.97, indicating that it could be entered and easily translocated within the wetland plant. In order to understand the distribution and transportation of chloroform inside the wetland plants, the plant was harvested at the end of the experiment, and the content and concentrations of chloroform were detected in plant tissues. As shown in Table 17.1, chloroform was detected in both belowground and aboveground of plant, and chloroform content in belowground was significantly higher than those in aboveground biomass. The results also showed that chloroform was mainly stored in the roots, and less than 40% was transported from belowground to aboveground biomass. The low chloroform content in the aboveground biomass was likely due to the following reasons: (1) the transpiration is relatively low, resulting in inefficient driving force to transport chloroform from belowground to aboveground biomass, (2) the strong sorption between chloroform and roots hindered the transportation of chloroform inside the plant tissues, (3) significant phyto-degradation in the aboveground biomass, (4) significant phytovolatilization from stems and leaves. Firstly, transpiration rates in this study was expected to be high according to the massive water loss in CWs. Secondly, phyto-degradation of chloroform was expected to play a minor role in CWs, because very limited metabolite (i.e. dichloromethane) was detected in the plant tissues. Thirdly, volatilization was very limited (see the next section) in planted CWs, so phytovolatilization of chloroform from stems and leaves was expected to be low compared with its content. As mentioned above, the strong sorption of chloroform on the plant roots might be the obstacle preventing its transportation from belowground to aboveground biomass (Table 17.1).

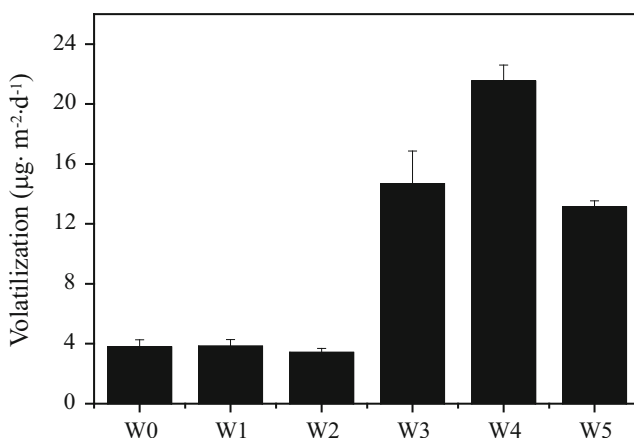


Fig. 17.2 The chloroform volatilization rates in W0~W5 microcosms

17.3.3 Volatilization

As an important component of CWs, plant can remove the volatile organic contaminants from volatilization (Imfeld et al. 2009). In this study, the chloroform emission was detected using the passive samplers. As shown in Fig. 17.2, the calculated volatilization rates were much higher in planted CWs than those in unplanted CWs, and the emission increased as the plant density. This indicated that plant could promote the chloroform volatilization in CWs. According to the mass balance calculation, chloroform removed via volatilization only accounted for 2.0–2.5% of the total chloroform in the influent. This suggested that volatilization plays a minor role in the chloroform removal process in planted CWs. By contrast, volatilization was considered to be a main pathway for chloroform removal in SF-CWs likely due to the direct and efficient contact between water and air (Rostad et al. 2000). The low volatilization observed in this study was likely the consequence that the water level is below the medium surface in SSF-CWs, resulting in hindered water–air transfer via diffusion in the SSF CWs. Seeger et al. (2011) and van Afferden et al. (2011) also found that the contribution of volatilization on volatile organic compounds (i.e. benzene and MTBE) removal was marginal in planted CWs. Previous study revealed that transpiration-driven uptake of volatile compounds can be well estimated based on Dettenmaier model (Dettenmaier et al. 2009). In this study, the model based uptake rates of chloroform was lower than the detected uptake rates. This indicated that Dettenmaier model could estimate the chloroform uptake by wetland plants to some extent, though it could not include the complete uptake pathways. Dettenmaier model calculated uptake rate is based on transpiration rates and the $\log K_{ow}$ values of target organic compounds, and some other uptake pathways (i.e. gas-phase transport) are not included in this model.

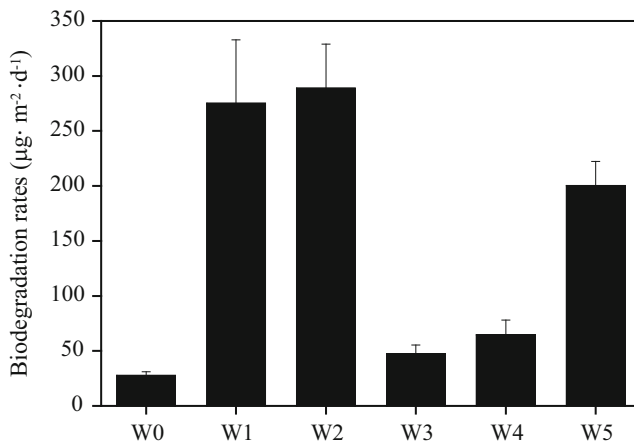


Fig. 17.3 The chloroform biodegradation rates in W0~W5 microcosms

17.3.4 Biodegradation

The biodegradation rates were calculated from the mass balance equation, and the results are shown in Fig. 17.3. The biodegradation rates in litter-added CWs were much higher than those in the other CWs (Fig. 17.3). This suggested that the plant litter could significantly improve the biodegradation of chloroform. Previous studies reveal that redox potential, microbes community and carbon sources are the most important factors for chloroform biodegradation (Gupta et al. 1996b; Pavelic et al. 2006; Wahman et al. 2011). In this study, the redox potential in litter-added CWs were between $-200\sim-100$ mV, suggesting that the anaerobic and sulfate reducing environment were achieved in the litter-added CWs. Previous study showed that the anaerobic reductive dechlorination of chloroform was promoted under the sulfate reducing environment (Gupta et al. 1996a), supporting the efficient biodegradation of chloroform observed in this study. Sulfate-reducing bacteria and methanogens were identified to be the key microbes responsible for chloroform degradation (Gupta et al. 1996a, b). In the litter-added CWs, both microbes were detected and their abundance was correlated with the chloroform degradation rates (Chen et al. 2015). In addition to anaerobic environment and the related degrading microbes, carbon source is also a critical factor for chloroform degradation. In the litter-added CWs, the carbon source is mainly produced during litter decomposition. Firstly, the lignocellulose is hydrolyzed to reducing sugars, and then the monosaccharides are converted to volatile fatty acids (i.e. acetic acid) through the anaerobic microbial metabolism (Chen et al. 2012). Acetic acid has been demonstrated to be primary substrate driving chloroform degradation through cometabolism (Gupta et al. 1996a). Thus, the accumulation of acetic acid in litter-added CWs provided the carbon source for chloroform degradation.

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Chapter 18

Hybrid Constructed Wetlands for the National Parks in Poland – The Case Study, Requirements, Dimensioning and Preliminary Results

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Abstract Poland is proud of 23 National Parks (NPs) which were established to protect unique European areas. Through cooperation between Life Science University in Lublin and the NPs, highly effective methods for environmental protection were introduced in Roztoczański and Poleski National Parks. One method is a hybrid treatment wetland (HTW) for wastewater treatment. HTWs, if well designed and maintained, can effectively treat wastewater generated in houses and museum buildings in the area of the NPs. In 2014, three HTWs were constructed in Roztoczański National Park (RNP) (A – Kosobudy, B – Zwierzyniec, C – Florianka). One of these installations was constructed in the shape of a fir tree (A – Kosobudy). In 2015 another one, in a turtle shape, was built in Poleski National Park (PNP) (D – Stare Załucze). Similar wastewater treatment technology was applied in all facilities. The technology consisted of three steps: (i) mechanical treatment in three

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septic tank chambers, (ii) biological treatment in two or three stages of vertical and horizontal beds, and (iii) inactivation of phosphorus in p-filter with calcium-silica rock. Since the very beginning all facilities have been monitored and the results indicate very low concentrations of pollutants in the final effluents. It can be concluded that HTWs working according to the assumed technology fulfill the strict requirements for wastewater treatment in NP areas in Poland.

Keywords Water protection • Protected areas • Biogenic compounds removal • Phosphorus removal • Wastewater treatment

18.1 Introduction

The majority of institutions dealing with natural resources protection in Poland are facing the problem of unsolved wastewater management in scattered development. Many of the buildings located in the areas of NPs have no access to local sewer systems and, therefore, wastewater is collected in cesspools (Piszczek and Biczkowski 2010). This approach is not adequate since the main function of NPs is to protect natural resources through the most environmentally friendly technologies. As a consequence, not all properties and functions of these areas are fully protected, which could have negative social and economic impacts on the development of the region (Świeca et al. 2012). Lack of adequate infrastructure for wastewater treatment is one fundamental limiting factor for tourism in these protected regions (Batyk 2012). Thus, PNP (Poleski National Park), RNP (Roztoczański National Park) and LSU (Life Science University) in Lublin joined together to solve this problem. Through this partnership, four hybrid treatment wetlands have been constructed to treat wastewater and further constructions are planned. The main aim is to protect the surface and ground waters in accordance with the 2nd Ecological Strategy of Poland [Polish Sejm, 23 August 2001], which stresses both fresh (surface and ground waters) and Baltic Sea water protection by prevention at the source and restoration of good ecological quality.

Kadlec and Wallace (2009), Fonder and Headley (2010) and Vymazal (2005) have shown that it is possible to achieve good quality effluent and protect water bodies in accordance with the Polish Environmental Protection Law (Regulation of the Minister of Environment 2014) through the application of Hybrid Treatment Wetlands (HTWs) for domestic wastewater treatment. Over 20 years of monitoring of HTWs in Poland indicates that this type of facility is appropriate for wastewater treatment in scattered development and ensures consistently good water quality (Józwiakowski et al. 2015). The removal efficiency of suspended solids (TSS) and organic matter (BOD₅ and COD) is over 90 % and removal of nitrogen compounds (TN) varies from 60 to 90 % in HTWs (Börner et al. 1998; Brix et al. 2003; Gajewska 2009; Gizińska et al. 2013; Gajewska and Obarska-Pempkowiak 2008, 2011; Gajewska et al. 2015; Józwiakowski 2012; Obarska-Pempkowiak et al. 2011). As

with all treatment wetlands, HTWs are not dedicated to effective phosphorus removal, and the removal efficiency for phosphorus usually does not exceed 30 % (Kadlec and Wallace 2009).

The aim of this study is to present the concept of wastewater management using hybrid wastewater treatment wetlands in four areas: three settlements located in the forest in Roztoczański National Park and one in the Centre for Teaching and Museum in Poleski National Park.

18.2 The Characteristics of Poleski and Roztoczański National Parks

Roztoczański and Poleski National Parks (50°37'00"N, 22°58'00"E; 51°27'19"N, 23°10'24"E) are located in Poland in the Lublin province (Fig. 18.1). They have cultural and environmental characteristics unique to both the country and Europe.

18.2.1 Roztoczański National Park (RPN)

The Park was established in 1974 and lies entirely in Central Roztocze. It consists of mainly woodland – 10 forest reserves make up 93.81% of the whole area. Protection of the diverse forest ecosystems of Central Roztocze was the main reason for the creation of Roztocze National Park. Currently, RPN covers an area of almost 8500 ha with strict protection of 1029 ha (12.1 %) of its area.

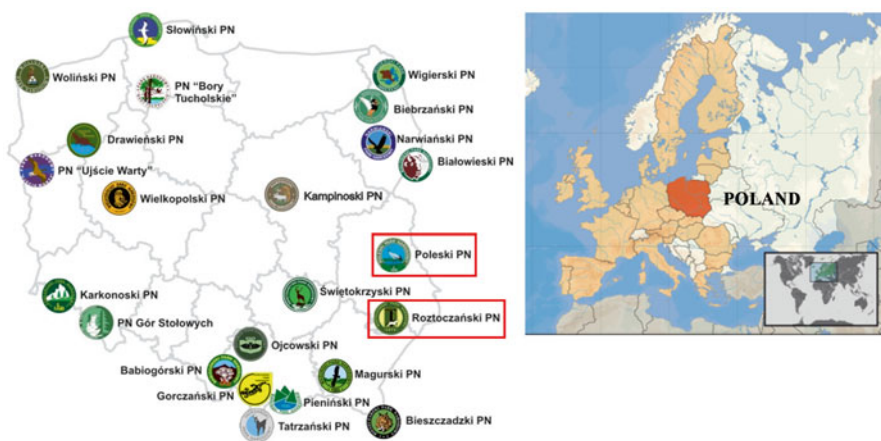


Fig. 18.1 Locations of National Parks in Poland (http://archiwum.ekoportal.gov.pl/warto_wiedziec_i_odwiedzic/Ekoturystyka/Parki_Narodowe_Polska/)

Water resources in RPN consist of fragments of flowing waters: the Wieprz river, three main streams (Świerszcz, Szum, and Krupiec), two ponds (Florianiecki and Black Pond), and a complex of ponds called Echo. These areas of surface water cover 52.6 hectares, only 0.6% of the total area of the Park. RPN has over 3630 species of animals and 900 species of vascular plants, representing nearly half of the total groups forming the Polish flora. Among them are 70 species under strict or partial protection and a similar number of taxa considered rare in the region, and about 20 are threatened with extinction in the country (<http://www.roztoczanski.pn.pl/>).

18.2.2 Poleski National Park (PNP)

The Park was created in 1990, within the Łęczyńsko – Włodawskie Lakeland, which is the southern part of Lublin Polesie. PNP is 9764.31 ha and was created to protect peat- wetland ecosystems. The Park protects unique biotopes in Europe, namely tundra. In the PNP, there are 140 species of northern plants. Moreover, the presence of many plant species of the Atlantic (25) and South (30) zones creates a unique flora (<http://www.poleskipn.pl>). There are four lakes within Poleski National Park; three of them (Moszne, Długie and Karaśne) are small, shallow, dystrophic ones, while the fourth lake is a eutrophic reservoir with an area of 130 ha (<http://www.poleskipn.pl>).

18.3 Water and Wastewater Management in the Area of PNP and RNP

Corporate housing and office space for villages are spread around in the area of RNP. Five buildings are located away from human settlements in forest areas, and two more are located within the village. The water is supplied by the local water supply network or from groundwater intakes. The domestic wastewater is treated in household wastewater treatment facilities or collected in cesspools and then exported to the local wastewater treatment plant in Zwierzyniec. Transport of wastewater from cesspools is very troublesome because of the costs involved, but primarily because of emissions (460 kg CO₂/year) by vacuum trucks moving along the area of the RNP.

A comprehensive program to organize the wastewater treatment in the forest settlements in the RNP was developed in 2013. The program aims to protect the groundwater aquifer (mainly water Cretaceous). In 2014, construction of three hybrid wastewater treatment wetlands and modernization of two other, already existing wastewater treatment plants were completed. For this purpose, the RNP received funding from the National Fund for Environmental Protection and Water Management in the amount of 150,355 PLN (ca. 35,000 EUR), which represents 94% of total investments (159,273 PLN).

The Statistical Vademecum of local government (2014) indicated that in the area of the PNP (in the municipalities of Urszulin, Stary Brus, Hańsk, Sosnowica, Ludwin, and Wierzbica), 64.2–82.9% of households obtain water from a central system, but the central sewage system only covers 18–31.7% of households. This means that the majority of households treat their wastewater through single family WWTPs or, more likely, by cesspools. In the past, single family WWTP drainage system technology was promoted. Nowadays these installations are out-of-date and clogged and, combined with plenty of leaking cesspools, pose a risk to surface and ground water quality. All six villages located in the forest of the PNP receive water from the central water system but for economic reasons they have never been connected to a central sewer system. Thus in 2014, the idea of using wetlands to treat wastewater generated in museum and didactic buildings of the PNP was developed and the construction was completed in September 2015. The concept is based on the newest achievements of wetland technology and included all aspects and advantages of the technology to protect the environment in this unique area.

18.4 The Concept of Hybrid Treatment Wetland Construction for RNP and PNP

Hybrid Treatment Wetlands (HTWs) were constructed in the area of the RNP in the distant forest villages of Kosobody (A), Zwierzyniec (B) and Florianka (C), and in the PNP in Stare Zaucze (D).

The general concept is based on four stages of treatment to ensure good quality of discharged wastewater to protect water bodies in highly protected areas of the National Parks. In the first stage, a three-chamber settling tank was used for mechanical treatment.

Previous studies indicated that the composition of wastewater discharged to treatment facilities in Poland is different (more concentrated) than in other countries in Europe. There are two main reasons for this phenomenon: (i) the smaller amount of water consumed in households (approximately 70 L/person/day) and (ii) poor quality of primary treatment (Gajewska and Obarska-Pempkowiak 2011; Gajewska et al. 2011, Obarska-Pempkowiak et al. 2015). For these reasons the type and construction of the settlement tank is crucial to support proper performance of the second stage – the HTW.

Because of the importance of primary treatment, the facility type was selected from already existing solutions which ensure the following requirements of removal: 40–50% for TSS, 25–40% for BOD₅, 20–40% for COD, and >10% for biogenic compounds. Fats and mineral oils should be removed with 70–80% effectiveness. The construction of a three-chamber tank supports mechanical treatment by processes such as sedimentation and flotation as well as primary biological treatment through fermentation. Gases generated during digestion of organic matter enhance the flotation of pollutants with lower density than water (fats and oils) which are collected in the upper part of the tank. Some pathogenic microorganisms are

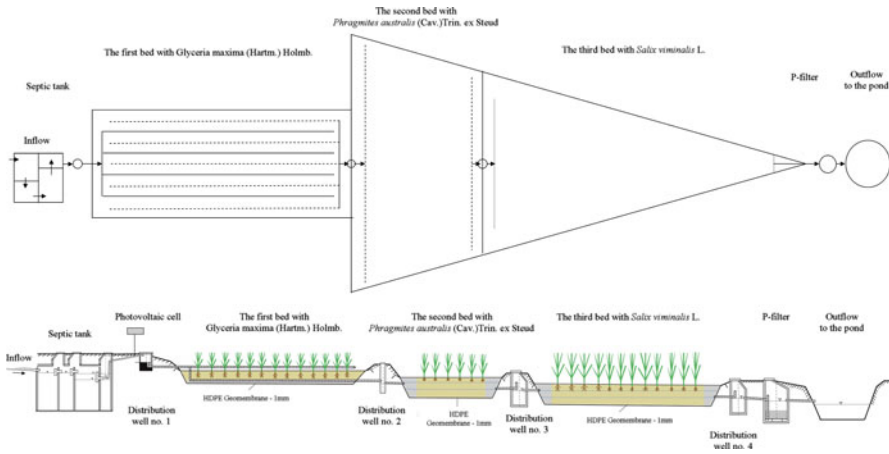


Fig. 18.2 Technological scheme and a longitudinal cross-section of a hybrid constructed wetland in Roztocze National Park – facility A (Jóźwiakowski et al. 2014)

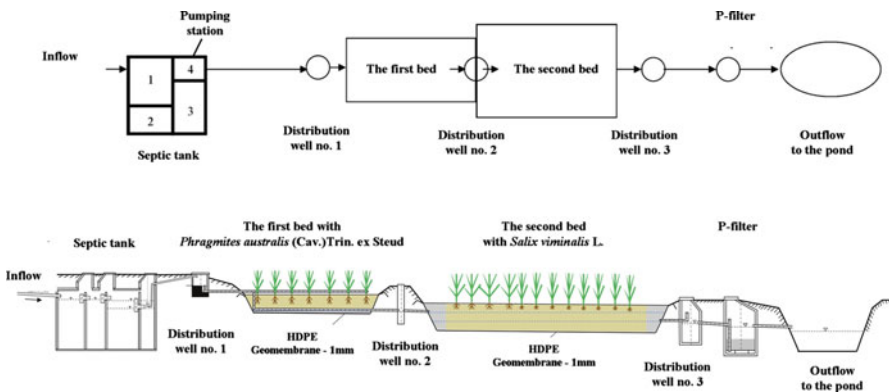


Fig. 18.3 Technological scheme and a longitudinal cross-section of hybrid constructed wetlands in National Parks – Facilities B, C, D (Jóźwiakowski et al. 2014)

eliminated during the processes that occur in the tank (Ødegaard 1998; Pawęska et al. 2011). The volume of the three-chamber tank depends on pe (person equivalent). In the case of the RNP and PNP it was predicted that the tanks should be between 3 and 8 m³. The estimated amount of primer (digested) sludge is 3.35 m³ in RNP and 0.985 m³ in PNP with similar average water content of 93 %. The produced sludge should be utilized safely at a municipal wastewater treatment plant.

In the next stage wastewater is pumped to two or three subsurface flow beds for biological treatment. The general layouts and cross sections of the applied configurations of HTWs are presented in Figs. 18.2 and 18.3. To enhance treatment processes and support biodiversity, different types of macrophytes were planted in the beds. Only local species were selected. In the first bed, usually with vertical flow (VF), *Phragmites australis* or *Glyceria maxima* was planted. *Phragmites australis*

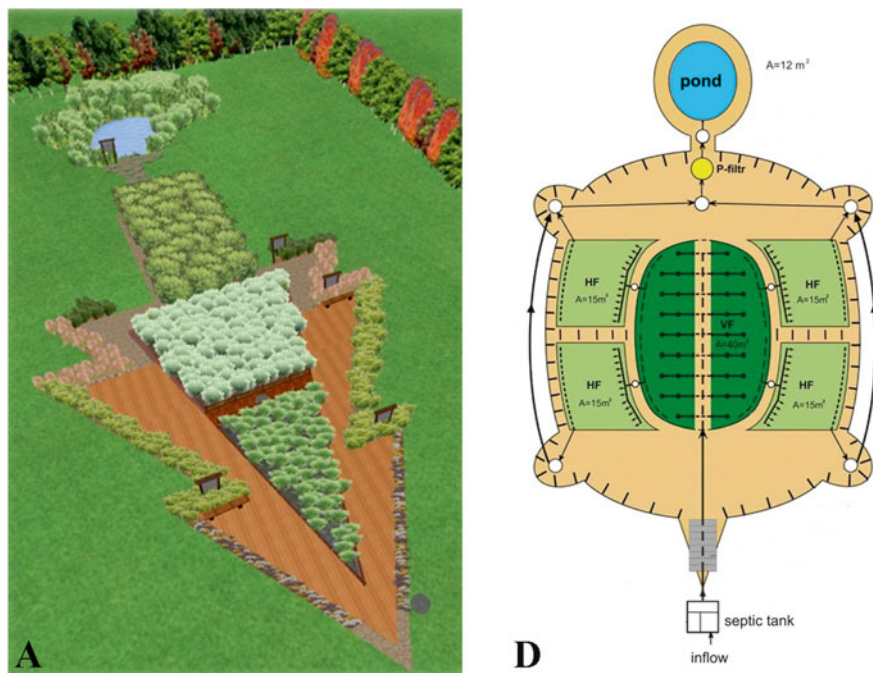


Fig. 18.4 The concept of construction of hybrid constructed wetland in two national parks in Poland: A – fir tree (RNP), D – turtle (PNP)

was planted in the second stage, which had horizontal flow (HF). *Salix viminalis* was used in the last HF stage. If the second stage was the last stage, it was planted with *Salix viminalis* to enhance evapotranspiration and minimize discharge (Jóźwiakowski 2008; Gajewska and Obarska-Pempkowiak 2009; Soroko 2001). According to Jucherski and Walczowski (2012), *Glyceria maxima* is able to remove large amounts of pollutants, especially nitrogen compounds, from the wastewater.

The third stage is a filter for phosphorus removal, and the treated wastewater is collected in a pond at the end of the system. To meet the strict requirements imposed by Polish Regulation on wastewater discharge to water bodies it was necessary to apply a special stage of treatment for phosphorus removal. The chosen technology is based on phosphorus binding with material rich in Ca^{2+} cations. The natural carbon silica rock is treated in high temperature to improve the binding potential. The P removal efficiency of this type of material can be as high as 90% (Nilsson et al. 2013; Bus and Karczmarczyk 2014; Cucarella et al. 2007; Nastawny et al. 2015). This removal efficiency should ensure that the concentration of P in the final effluent is below the required limit of 5 mg P/L. The recipient of the final effluent is an infiltration pond where the water is collected. Depending on the conditions (hydraulic load, weather conditions, etc.) final effluent could be infiltrated into the surroundings or evaporated to the atmosphere. In Fig. 18.4 the visualization of HTWs in the shape of fir tree and turtle is presented.

Requirements to be met by final effluent discharged into surface waters and soil are defined in the Regulation of the Minister of Environment (2014). Wastewater discharged into the waters should not cause any changes which would hamper the proper functioning of aquatic ecosystems and should meet the specific water quality requirements for waters associated with their use.

To meet these requirements, the concentration of pollutants in wastewater discharged from a WWTP serving 2000–9999 pe should be as follows: 50 mg L^{-1} for TSS, a reduction of 87.5 %; 40 mg L^{-1} for BOD₅, a 92.0 % reduction; and 150 mg L^{-1} for COD, an 81.2 % reduction. The design parameters of surveyed facilities are presented in Table 18.1 and the pollutant loads are presented in Table 18.2.

Table 18.1 Characteristics of hybrid constructed wetlands in two national parks in Poland

Facility/Parameter	Roztoczański National Park (RNP)			Polesie National Park (PNP)
	A	B	C	D
	Kosobudy	Zwierzyniec	Florianka	Stare Załucze
Number of person equivalent	20	4	10	10
Flow, Q (m ³ d ⁻¹)	2.0	0.4	1.0	1.0
Active capacity of the septic tank. V (m ³)	8.0	4.0	5.0	3.0
Type and Area of beds				
<i>Glyceria</i> bed (m ²)	I – 60 (VF)	–	–	I – 40 (VF)
<i>Phragmites</i> bed (m ²)	II – 60 (HF)	I – 18 (VF)	I – 40 (VF)	II – 4 × 15 (HF)
<i>Salix</i> bed (m ²)	III – 60 (HF)	II – 30 (HF)	II – 56 (HF)	–
Total area (m ²)	180	48	96	100
P-filter for phosphorus removal (m ³)	1.0	0.5	0.75	0.5
Start up	2014	2014	2015	2015

Table 18.2 The planned loads of pollutants in wastewater in inflow and outflow

Parameter	Sites			
	A	B	C	D
	Kosobudy	Zwierzyniec	Florianka	Stare Załucze
Inflowing load (kg d ⁻¹)				
TSS	0.6	0.12	0.3	0.3
BOD ₅	0.8	0.16	0.40	0.40
COD	1.6	0.32	0.80	0.80
TP	0.04	0.08	0.20	0.20
Outflowing load (kg d ⁻¹)				
TSS	0.07	0.014	0.035	0.035
BOD ₅	0.05	0.020	0.025	0.025
COD	0.25	0.050	0.125	0.125
TP	0.01	0.02	0.05	0.05

18.5 Removal Efficiency of HTWs in RNP – Preliminary Results

18.5.1 Methods

The construction of HTWs in the RNP was completed in October 2014. After a few months of startup period, continuous monitoring of plants was implemented. From February to September 2015, 32 sampling events were carried out. The following parameters were measured in the collected samples of wastewater inflow and outflow:

- total suspended solids (TSS) – direct gravimetric method with paper points,
- biological oxygen demand (BOD₅) – determined using oximeter WTW Oxi 538,
- chemical oxygen demand (COD) – by dichromate method (using WTW MPM 2010 photometer, after oxidation of the test sample in a thermoreactor at 148 °C),
- total nitrogen – with AQUALYTIC PCspectro spectrophotometer, after oxidation of the test sample in a thermoreactor at 100 °C,
- total phosphorus – with WTW MPM 2010 photometer after oxidation of the test sample in a thermoreactor at 120 °C.

In addition, the quantity of coliforms and fecal coliforms was determined according to the [Polish Standards PN-75/C-04615-05](#) and [PN-77/C-04615-07](#).

The efficiency of pollutant removal in the monitored beds was determined on the basis of the average values of the analyzed indicators of contamination in wastewater flowing in (Cd) and out (Co) of the beds according to Eq. 18.1:

$$\eta = (1 - Co / Cd) \times 100 [\%] \quad (18.1)$$

The results were compared with the requirements of the Regulation of the Minister of Environment (2014).

18.6 Results and Discussion

Maximum, minimum, and average values and standard deviations are given in Table 18.3 and treatment efficiency is shown in Fig. 18.5.

18.6.1 Inflow

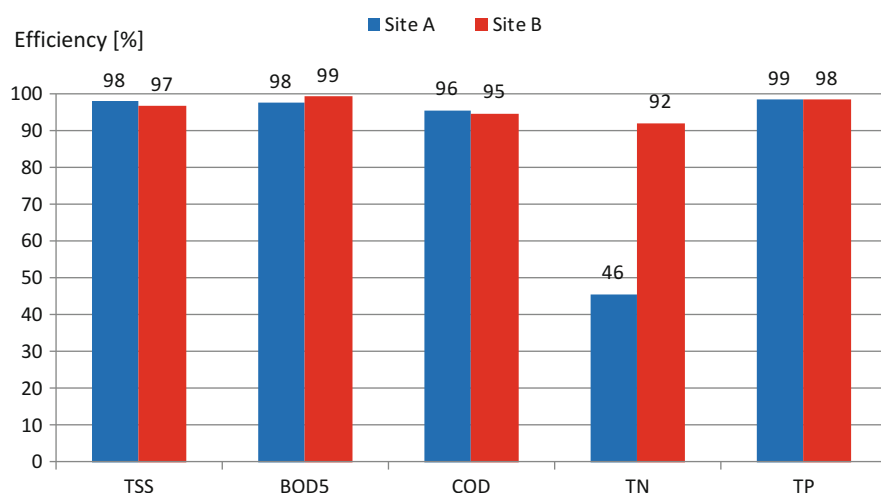
18.6.1.1 TSS

According to Miernik (2007), concentration of TSS in raw sewage in different types of WWTPs in rural areas of Poland varied between 118 and 187 mg l⁻¹. For systems studied by Józwiakowski (2012), the average concentration of total suspended

Table 18.3 The characteristics of inflow and outflow wastewater in Sites A and B

Parameter	Inflow				Outflow			
	Max	Min	σ	\bar{x}	Max	Min	σ	\bar{x}
Site A – Kosobudy								
TSS (mg l ⁻¹)	588	320	141	456	10.2	8.7	0.9	9.7
BOD ₅ (mg l ⁻¹)	791	389	183	521	25.7	4.3	11.7	12.2
COD (mg l ⁻¹)	1370	680	298	965	53.0	31.0	11.0	41.7
TN (mg l ⁻¹)	129	80.0	21.6	104	83.0	40.0	23.3	56.3
TP (mg l ⁻¹)	17.6	11.2	2.7	13.6	0.5	0.07	0.21	0.21
Site B – Zwierzyniec								
TSS (mg l ⁻¹)	491	233	107	360	16.0	2.0	6.5	11.6
BOD ₅ (mg l ⁻¹)	863	338	229	534	11.8	0.7	5.2	4.3
COD (mg l ⁻¹)	1790	930	401	1195	134	31.0	48.3	62.0
TN (mg l ⁻¹)	141	104	15.4	121	14.4	1.3	5.7	9.6
TP (mg l ⁻¹)	29.8	16.8	5.5	22.5	1.2	0.08	0.6	0.4

σ = standard deviation

**Fig. 18.5** Average removal efficiency at Sites A and B (in %)

solids in raw sewage was 120–322 mg l⁻¹. The current study shows that the average concentration of suspended solids in raw wastewater of surveyed facilities amounted to 456 mg l⁻¹ and 360 mg l⁻¹ for sites A and B, respectively (Table 18.3). These results were higher than those reported by Józwiakowski (2012) and Miernik (2007) and slightly lower than those reported by Pawęska and Kuczewski (2008). According to these authors the concentration of total suspended solids in raw wastewater at the treatment plant Brzeźno and Mroczeniu varied between 518 and 550 mg l⁻¹.

18.6.1.2 BOD₅ and COD

The average BOD₅ and COD concentrations in raw wastewater were 521 mg l⁻¹ and 534 mg l⁻¹, respectively (Table 18.3). Miernik (2007) reported a similar value (578 mg l⁻¹) for a wastewater treatment plant in Michniów. Józwiakowski (2012) confirmed that COD in raw wastewater in households ranged from 509 to 562 mg l⁻¹. The average COD concentration in raw wastewater at Site A was 965 mg l⁻¹, with minimum and maximum values of 680 mg l⁻¹ and 1370 mg l⁻¹, respectively (Table 18.3). Miernik (2007) also found high values of COD in the inflow (1125 mg l⁻¹) to this treatment plant. At Site B, an even higher average COD concentration of 1195 mg l⁻¹, with a maximum value of 1790 mg l⁻¹, was recorded. Obarska-Pempkowiak et al. (2013) investigated single family TWs in the Kaszuby Lake district in Poland and found that the average concentration of BOD₅ varied from 500 to 700 mg l⁻¹ whereas for COD the inflow concentrations ranged between 900 and 1300 mg l⁻¹. Extremely high concentrations of COD (in excess of 1000 mg l⁻¹) and BOD₅ were discharged to TWs at four out of nine surveyed farms.

18.6.1.3 Total Nitrogen

This study found that the average concentration of total nitrogen in raw wastewater in facility A was 104 mg·l⁻¹, while the average concentration was 121 mg·l⁻¹ in system B (Table 18.3). High concentrations of total nitrogen together with the large amount of organic matter in the raw wastewater could indicate a small amount of water consumed in surveyed households. Pawęska and Kuczewski (2008) obtained similar values of total nitrogen in raw sewage flowing into the Treatment Wetlands in Brzeźno and Mroczeń (Poland). Similar results were also reported by Obarska-Pempkowiak (2013), Gajewska and Obarska-Pempkowiak (2011), and Gajewska et al. (2014). The main reason for the high total nitrogen concentration is lower water consumption (approximately 80 l/day/pe instead of the 150 l/day/pe used for dimensioning of the systems).

18.6.1.4 Total Phosphorus

The average concentration of total phosphorus obtained in facility A, 13.6 mg l⁻¹, was exactly the same as that reported by Wiejak (2013). The average concentration of total phosphorus in wastewater at site B was even higher – 22.5 mg l⁻¹. According to Józwiakowski (2012), the concentration of phosphorus in raw wastewater is variable and can range from 29.6 to 46.4 mg l⁻¹.

18.6.2 *Outflow*

18.6.2.1 TSS

The average concentrations of total suspended solids in the effluent from sites A and B were 9.7 mg l^{-1} and 11.6 mg l^{-1} , respectively (Table 18.3). This concentration is 4–5 times lower than the limit specified for TSS in the Regulation of the Minister of the Environment [2014].

18.6.2.2 BOD₅ and COD

The average value of BOD₅ in the effluent from site A was 12.2 mg l^{-1} , while the minimum and maximum values were 4.4 mg l^{-1} and 25.7 mg l^{-1} , respectively (Table 18.3). These values were all below the permissible limit of 40 mg l^{-1} . At site B, the BOD₅ concentration was even lower, more than 8 times lower than the limit specified in the Regulation of the Minister of the Environment [2014].

The COD concentration in the discharge flowing out from the P filter at site A ranged from 31 to 53 mg l^{-1} (Table 18.3), with an average value of 41.7 mg l^{-1} . The average concentration of COD in the effluent at site B was 62 mg l^{-1} (Table 18.3) with a maximum value of 134 mg l^{-1} (Table 18.3). These concentrations were only slightly lower than the permissible value given by the Regulation of the Minister of the Environment [2014]. This may be due to the short time of operation of this facility. However, in 2015 no COD concentrations in the outflow exceeded the maximum standard limit for COD (150 mg l^{-1}).

18.6.2.3 Total Nitrogen

The concentration of TN in the effluent from site A varied from 40 to 80 mg l^{-1} with an average of 56.3 mg l^{-1} (Table 18.3). Despite the use of a three-stage system with a P filter the facility failed to completely remove nitrogen from wastewater. The limit for TN in wastewater is 30 mg l^{-1} , but only for wastewater entering lakes and their tributaries or reservoirs located on flowing waters [Regulation of the Minister of the Environment 2014]. These requirements were met by the HTW at site B where the average concentration of total nitrogen in treated wastewater was 9.6 mg l^{-1} . The limit values of this parameter ranged from 1.3 to 14.4 mg l^{-1} .

18.6.2.4 Total Phosphorus

The concentration of total phosphorus in the effluent from site A varied between 0.07 and 0.45 mg l^{-1} , with an average value of 0.21 mg l^{-1} (Table 18.3). The average concentration of total phosphorus in the effluent from site B was 0.37 mg l^{-1} and

varied between 0.08 and 1.20 mg l⁻¹ (Table 18.3). For both HTWs, the standard deviations were very low indicating stable removal and the concentration of TP in discharged was much lower than the required standard of 5.0 mg l⁻¹ according to Regulation of the Minister of the Environment (2014). This requirement needs to be fulfilled for settlements with $p_e < 2\ 000$ in case of direct discharge of wastewater into lakes and their tributaries and reservoirs situated on flowing waters. The surveyed systems in the NPs are not subject to this regulation, but the HTWs are located in protected areas and thus should ensure the highest possible protection against eutrophication.

18.7 Efficiency of Organic Matter and Biogenic Compounds Removal

The average TSS removal efficiency was similar at both sites, 98 % at site A and 97 % at site B (Fig. 18.5). Similar high efficiency of TSS removal has been widely confirmed by many authors all over the world for different types and configuration of TWs (Kadlec and Wallace 2009; Vymazal 2010, 2011; Gajewska and Obarska-Pempkowiak 2011). For example, Singh et al. (2009) found that a treatment plant in Nepal eliminated 96 % of TSS, while Seo et al. (2009) reported treatment efficiency of 99 % in South Korea.

The effectiveness of BOD₅ reduction at sites A and B amounted to 98 % and 99 %, respectively (Fig. 18.5). High removal efficiency (99 %) of easily degradable organic matter was reported by Seo et al. (2009) in the facility in South Korea. Józwiakowski (2012) reported removal efficiency of 96 % in the treatment plant in Janów, Poland. Five HTWs investigated in North Poland by Gajewska and Obarska-Pempkowiak (2011) exhibited BOD₅ removal efficiency between 78 % and 96 % depending on the applied configuration. Single family TWs with different configurations investigated by Obarska-Pempkowiak et al. (2015) removed between 64 % and 92 % of BOD₅ during the first 2 years of operation, after which the efficiency increased to over 80 % for all facilities.

The test results presented in Fig. 18.5 confirm that hybrid systems in the RNP provide high COD removal efficiency. COD removal efficiencies at sites A and B were 96 % and 95 %, respectively. Seo et al. (2009) observed 98 % removal efficiency for COD in hybrid wetland plants in South Korea while Melian et al. (2010) observed removal efficiency of 80 % in the Canary Islands. Józwiakowski (2012) observed COD removal efficiency of 94 % in a treatment plant at Janów, and Gajewska and Obarska-Pempkowiak (2011) reported the removal of COD in HTWs in Poland in the range of 75–95 %.

Our study shows that the total nitrogen at site A was removed with 46 % efficiency which is much lower than the removal efficiency at site B (92 %; Fig. 18.5). Gajewska et al. (2004) reported total nitrogen removal efficiencies of 64 % and 34 %, respectively, for HTWs in Waizenfeld and Wiedersberg. Krzanowski et al. (2005)

reported TN removal efficiency during the growing and post-growing seasons at 71 % and 63 %, respectively, in a multistage TW in Muszynka. Gajewska and Obarska-Pempkowiak [2011] found wide variability in efficiency of TN removal for five HTWs in Poland, from 23 to 80 % depending on the working conditions of the facility and the season (removal efficiency was up to 12 % higher during the growing season). For a single family TW in the Kaszuby Lake district in Poland, the efficiency of TN removal in the first 2 years of operation varied from 55 to 77 % (average 60 %) and increased to 73–84 % (average 75 %) in the third year (Obarska-Pempkowiak et al. 2015).

Phosphorus removal efficiencies at sites A and B were 99 % and 98 %, respectively. The values were almost identical in both analyzed P- filters applied to phosphorus removal from wastewater. Moreover, the removal values were stable during the monitoring period, proving the high efficiency of the applied carbon-silica rock technology for phosphorus removal. The available literature results from hybrid TW systems throughout the world reveal that the average total phosphorus removal efficiency in these facilities is between 70 and 89 % (Krzanowski et al. 2005; Seo et al. 2009; Sharma et al. 2010). Józwiakowski (2012) achieved similar total phosphorus removal efficiency (in the range of 89–99 %) in the facility in Janów. Gajewska and Obarska-Pempkowiak (2009) recorded lower average efficiency of 47 % for phosphorus removal in five facilities filled with gravel.

The results obtained confirm that HTWs built in the NPs in Poland according to the assumed concept and construction ensure from the very beginning good treatment efficiency of organic matter, suspended solids, and biogenic compounds and consequently meet the requirements imposed by the Regulation of the Minister of the Environment (2014).

18.7.1 Efficiency of Microbiological Contamination Removal

Effectiveness in reduction of different bacteria groups at site A in February ranged from 81.54 to 99.46 %. In the treated effluent from site A the content of coliform bacteria (as the most probable number of bacteria – MPN) averaged 2.4×10^4 MPN 100 ml^{-1} and the fecal coliform content was 7×10^2 MPN 100 ml^{-1} (Table 18.4).

In September, effectiveness of the bacteria reduction at site A increased, reaching almost 100 %. The average elimination of coliform bacteria amounted to 99.90 %, while fecal coliform removal amounted to 99.99 %. The coliform content in treated wastewater was 2.4×10^4 MPN 100 ml^{-1} and the fecal coliform content was 5×10 MPN 100 ml^{-1} (Table 18.4).

Site B showed similar efficacy as site A. The efficiency of removal of both coliform and fecal coliform bacteria at site B in February was over 99.99 % (Table 18.4) with <5 MPN 100 ml^{-1} in the outflow. In September, the treatment exhibited a slightly lower effect on the elimination of bacteria, with removal efficiencies of

Table 18.4 The effects of the removal of microbiological contamination in the analyzed HTWs in the national parks in Poland

Parameter	Site A			Site B		
	C _{in}	C _{out}	E [%]	C _{in}	C _{out}	E [%]
	18.02.2015					
Total coliforms [MPN 100 cm ⁻³]	1.3×10 ⁵	2.4×10 ⁴	81.54	7×10 ⁸	<5	99.99
Fecal coliforms [MPN 100 cm ⁻³]	1.3×10 ⁵	7×10 ²	99.46	7×10 ⁶	<5	99.99
	04.09.2015					
Total coliforms [MPN 100 cm ⁻³]	2.4×10 ⁷	2.4×10 ⁴	99.90	7×10 ⁶	7×10 ³	99.90
Fecal coliforms [MPN 100 cm ⁻³]	2.4×10 ⁷	5×10	99.99	7×10 ⁵	2.4×10 ³	99.66

99.90 % and 99.66 % respectively for coliform and fecal coliform bacteria. As a consequence, the numbers of coliform and fecal coliform bacteria were higher in comparison to February, amounting to 7×10³ MPN 100 ml⁻¹ for coliform bacteria and 2.4×10³ MPN 100 ml⁻¹ for fecal coliform bacteria. According to Talarko (2003) the removal efficiency of coliform bacteria in the soil-plant filters is about 99%. Bergier et al. (2002) reported fecal bacteria removal efficiency of 98.9 % for constructed wetlands treating domestic wastewater. Lalke-Porczyk et al. (2010) reported slightly lower efficacy of fecal coliform elimination in reed and willow beds (94.51 % and 92.07 %, respectively).

Both treatment plants in the NPs showed almost 100 % removal of bacteria from raw wastewater. Removal of bacteria is extremely important for protecting areas like national parks where the environment should not be exposed to pathogenic microorganisms. Excessive inflow of bacteria into the soil or water could have detrimental effects on the existing ecosystem. In addition, the treatment plants do not negatively affect the environment in which they are located.

18.8 Conclusions

18.8.1 General

As a result of the application of hybrid wastewater treatment wetlands in the National Parks the following effects could be achieved:

1. MATERIAL EFFECT – construction of four highly effective HTWs.
2. ECOLOGICAL EFFECT – treatment plants will provide long-term protection of the environment in a highly protected area and mitigate point source emission of pollutants.
3. LANDSCAPE EFFECT – treatment plants are well integrated into landscape in the national parks and only native plant species are used.

4. **ECONOMIC EFFECT** – treatment plant maintenance costs will be significantly lower than those for septic tanks and previously used activated sludge treatment plant. It is anticipated that at facilities A and D (due to the use of photovoltaic cells) the operating costs of the treatment plants will be limited only to sewage sludge disposal once a year, while in two other HTWs it will be necessary to apply electricity to power the pump. Annual cost of working pump is estimated at 20–30 PLN (6–8 EURO).
5. **EDUCATIONAL EFFECT** – the construction of ecological walkways at facilities A and D and the opportunity to observe the functioning of other facilities will help improve knowledge and awareness of high school students and other tourists visiting both national parks.

18.8.2 Detailed

1. Hybrid Treatment Wetlands applied in the National Parks in Poland ensured from the very beginning very effective removal of TSS (over 96 %) and organic matter (BOD_5 : 97–99 % and COD: 94–95 %)
2. The HTW at site A provided total nitrogen removal efficiency of 45.7 %, while the HTW at site B provided significantly higher removal, up to 92.1 %.
3. The applied configuration of HTWs with the P filters as the last stage of treatment provided more than 98 % removal of phosphorus.
4. Wastewater effluent met all the requirements imposed by the Ministry of the Environment [2014].
5. HTW systems applied in the NPs provided almost 100 % removal efficiency of coliform and fecal coliform bacteria. The numbers of monitored bacteria in treated wastewater were usually very low and had no negative impact on the natural environment in the NPs.
6. The results presented in this paper have direct relevance to the design and construction of high-efficiency hybrid treatment wetlands (HTWs).
7. HTWs for wastewater treatment can be used on a larger scale in rural areas, especially in protected areas and valuable landscapes. They represent a valuable alternative to conventional WWTPs or septic tanks (cesspools).

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Chapter 19

Global Warming: Confusion of Cause with Effect?

Marco Schmidt

Abstract The New Water Paradigm opens up an alternative approach to the climate debate. Rather than explain drought as a consequence of global warming by carbon emissions, this new approach addresses landscape dehydration as a key cause than rather key effect of global warming. The argument is based on the fact that solar energy is converted and a cooling occurs when water evaporates on Earth's surface and water vapour condenses in the atmosphere as clouds.

However, modern urban expansion, global deforestation and desertification reduce evapotranspiration: In turn, reduced evaporation results in increasing short-wave solar radiation which is converted to long-wave thermal emissions and sensible heat. The result is that higher surface temperatures creating heat island effects over cities, contribute to local, regional and ultimately, global climate change challenges. Rainwater harvesting promises a major mitigation strategy against increased temperatures and drought. Urban water management can be enhanced by the ecological design of green roofs, evaporative facades, and ground permeable surfaces combined with vegetation. This paper demonstrates the application of these New Water Paradigm principles in Germany.

Keywords Evaporation • Evapotranspiration • Global warming • Vegetation • Urbanization • Water cycle

19.1 Introduction

The New Water Paradigm explores evaporation challenges and opportunities, leading to a new approach in the climate debate. Rather than explaining drought as a consequence of global warming caused by carbon emissions, it addresses the loss of 800 km² of vegetated surfaces as a key cause rather than key effect of global warming. The

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paradigm is designed to reduce the negative environmental impacts of urbanization which form of land use is expanding at 150 km² per day. Rainwater management in cities usually means a discharge into drains. Increased city temperatures resulting from radiative heat island effects can be considerably reduced by water evaporation (Schmidt 2009; SenStadt 2010). The new paradigm also reduces flooding and drought.

On a global scale, evaporation lowers temperatures due to the phase change of water into its gaseous state of water vapor. Therefore, the daily loss of vegetation of 800 km² globally, 525 km² of it represents deforestation (Hansen et al. 2013), damages regional rain cycles leading to a marked shift in average planetary temperatures. At the same time, reduced photosynthesis by plants decreases O₂ output and increases CO₂ levels in the atmosphere. Additionally, the loss of fertile agricultural land (GTZ 2007; Schmidt 2010a) means major food supply challenges. Rainwater harvesting for irrigation and evaporative cooling from soil and plants, therefore represents a major strategy against global warming from a social justice perspective.

19.2 Water and the Global Energy Budget

Evaporation of water is the largest and most important component of energy conversion on Earth. A large portion of rainfall on our surface depends on the rain that evaporates locally. A reduction in evaporation increases the conversion of short-wave global solar radiation to long-wave emissions and sensible heat. A reduction in evaporation on land translates to a reduction in overall precipitation, effecting a further reduction in evapotranspiration, thus creating a dynamic and circular systemic effect (Schmidt 2010b).

Rather than defining evaporation as a water loss, the new paradigm (Kravčik et al. 2007; www.waterparadigm.org) defines evaporation as a source of precipitation. This new perspective fundamentally changes our understanding of global warming. Drought is conventionally expressed as a result of rising global temperatures, but by the New Water Paradigm increased aridity is the cause, not the effect, of global warming. Intensive land use patterns cause local environments to dry out (Ripl et al. 2007; Kravčik et al. 2007), and at the same time increase temperatures.

The tremendous problems resulting from global climate and water challenges are related to unsustainable land use patterns everywhere. For instance in Germany, even though population density may not increase, urbanization continues to grow at a rate of nearly 1 km² daily (UBA 2008). This results in an annual reduction of 200 mm evaporation and releases sensible heat and thermal radiation of more than 50,000 GWh (0.2 m³ year⁻¹ * 700 kWh * 1 Mio m² * 365 days). In winter, the warming may not be experienced as a negative effect, but certainly it disturbs the stability of weather patterns and cooling in spring, summer and fall seasons. The associated loss of evaporative vegetation further impacts on hydrological processes, causing extreme storms, floods, drought and desertification.

The reduction of evaporative cooling at the Earth's surface increases two components – thermal radiation from long wave emissions and sensible heat. Leaving aside regional climatic differences, Fig. 19.1 using published data from (

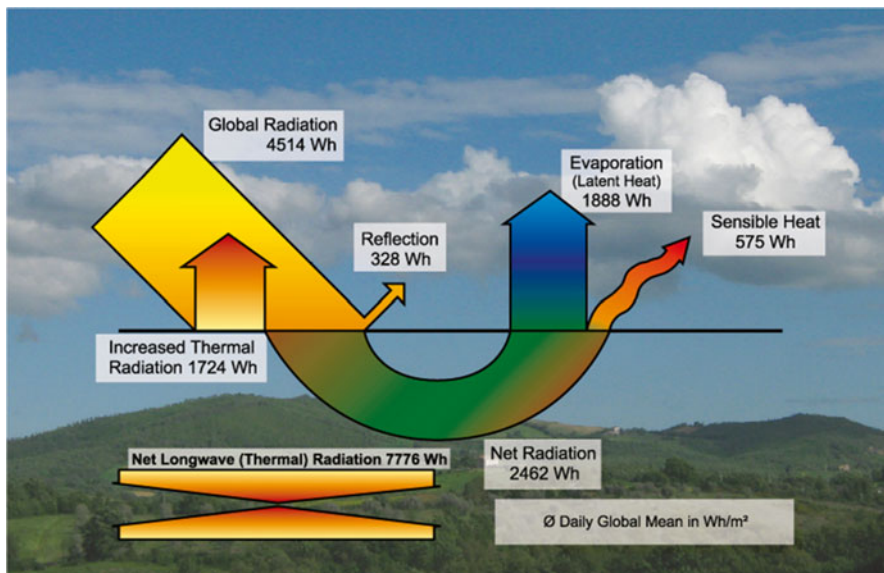


Fig. 19.1 Average global daily radiation budget of one m² worldwide (Schmidt et al. 20010a) (Energy data based on www.physicalgeography.net)

[icalgeography.net](http://www.physicalgeography.net); Trenberth et al. 2009), indicates an average energy flux of one square meter per day across the Earth’s surface. The components of incoming solar radiation break down as – 7.3% reflected and 38% directly converted to thermal radiation due to the increase of surface temperatures. The total long-wave (thermal) radiation consists of atmospheric counter-radiation (7776 Wh m⁻² d⁻¹) and the thermal radiation of the surface of the Earth (7776+ 1724 Wh m⁻² d⁻¹). Other available figures on the global mean energy budget (Trenberth et al. 2009), combine both components, which is scientifically misleading. In Fig. 19.1, the components are presented separately because the process is one of dynamic interaction. All surfaces above –273 °C emit and receive long-wave radiation at the same time.

Net radiation can be either converted into sensible heat (575 Wh m⁻² d⁻¹) or consumed by evaporation, a conversion into latent heat. Therefore, with 1888 Wh m⁻² d⁻¹, the energy conversion by evaporation at 42% from incoming short-wave radiation, is the most largest component of the Earth’s energy budget. Furthermore, evaporation reduces the long-wave thermal radiation due to the decrease in surface temperatures.

19.3 From Global to Local Scale

With regard to Fig. 19.1, the global energy budget is dominated by evaporation and condensation. Urbanization results in huge changes to the small water cycle. Hard materials and surfaces in urban areas absorb and re-radiate solar irradiation so

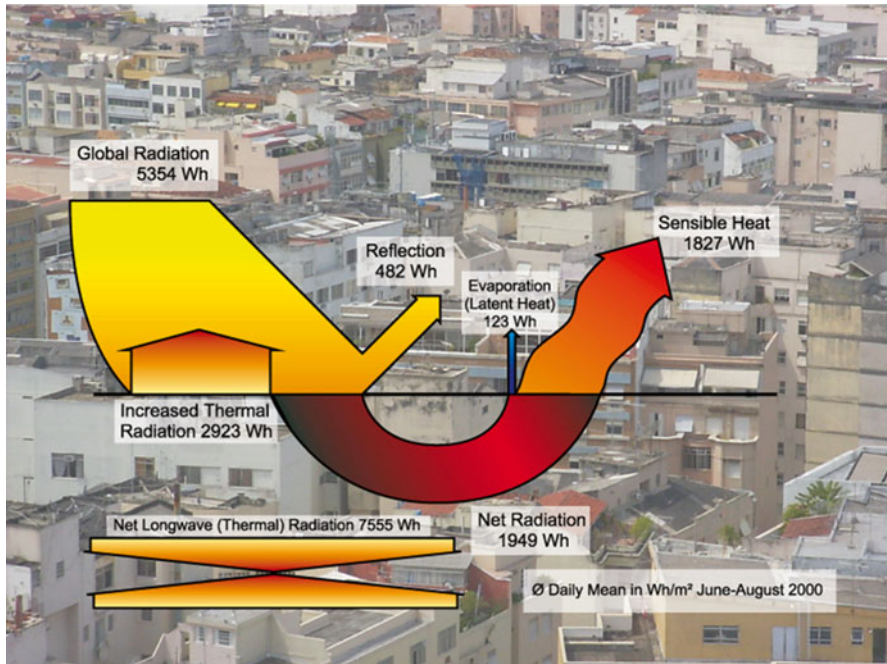


Fig. 19.2 Radiation balance of a black asphalt roof as an example for urban radiation changes (Schmidt 2005)

increasing their heat capacity. The main forces increasing urban heat island effects are vegetation removal and paved surfaces. They influence the urban microclimate through a change in radiation components. As a result, air temperatures inside buildings also rise and leading to greater energy consumption from air conditioning. This worsens the situation of the urban heat island effect by a release of additional heat (Schmidt 2003). To exemplify radiation in urban areas, Fig. 19.2 illustrates the radiation balance of a black asphalt roof. Because rainwater is funneled into sewer systems, most of the net radiation from the urban setting is converted to sensible heat rather than evaporation. Higher surface temperatures also increase thermal radiation.

The option to green buildings is a logical solution to create healthy and sustainable air temperatures in cities and improve microclimate. Vegetation on and around buildings converts solar radiation into latent heat by evapotranspiration. The conventional approach to green buildings focuses mainly on energy conservation and ventilation requirements to reduce CO₂ emissions rather than on the strategic use of vegetation to promote the evaporation process.

According to measurements taken at the UFA Fabrik case study in Berlin, a vegetated roof covered with 8 cm of soil compared to an asphalt roof in the same overall environment transfers 58 % of net incident radiation into evapotranspiration during the summer months (Fig. 19.3). The annual average energy conversion of net radiation into evaporation is 81 %, the resultant cooling-rates are 302 kWh m⁻² year⁻¹

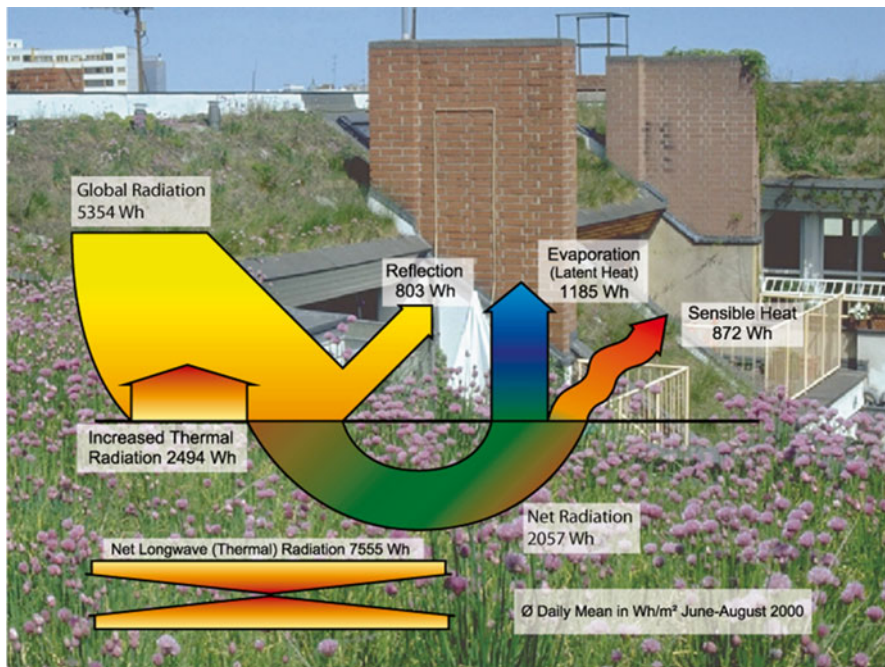


Fig. 19.3 Extensive green roofs transfer 58 % of net radiation into evapotranspiration during the summer months, UFA Fabrik in Berlin, Germany (Schmidt 2005)

with a net radiation of $372 \text{ kWh m}^{-2} \text{ year}^{-1}$ (Schmidt 2005). The asphalt roof and the green roof in Figs. 19.2 and 19.3 have been monitored at the same time in the same location and are directly comparable.

19.4 Global Warming: Confusion of Cause with Effect?

Global temperatures are strongly affected by vegetation, evaporation and condensation. Vegetation also relates directly to O_2 and CO_2 in the atmosphere through photosynthetic processes. Carbon emissions are not responsible for the increase of CO_2 in the atmosphere; the cause of the CO_2 rise is a reduction in photosynthesis. The correlation between rising global temperatures and increasing CO_2 is not a direct one.

Commenting on the social construction of public policy, Salleh (2010, 2016) has noted a trend to carbon reductionism among neoliberal governments and scientific establishments. This reductive thinking translates physical units into economic measurements leading both politicians and global climate activists away from holistic, integrative, environmental solutions for climate change. The turn away from systemic cause-effect dynamics must be corrected if global warming is to be mitigated.

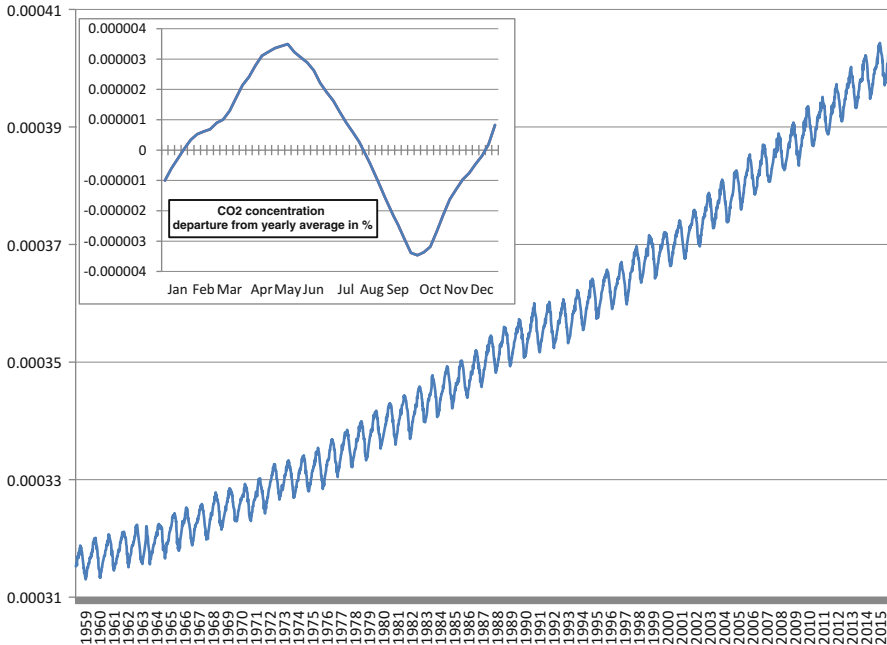


Fig. 19.4 Change in concentration of carbon dioxide in Earth's atmosphere (Keeling Curve), measurements taken at the Mauna Loa in Hawaii. Data from Dr. Pieter Tans, NOAA/ESRL and Dr. Ralph Keeling, Scripps Institution of Oceanography; <http://scrippsco2.ucsd.edu>

Climate change mitigation measures based on carbon reductionism like using biomass for heating may have the opposite effect than intended. Likewise, trading in carbon emissions is more beneficial to market economics than ecological science. At an epistemological level, ecological cycles cannot be understood by linear reasoning but need to be explained as dynamic interactive processes.

In terms of real measurements of CO₂ in the atmosphere, the longest time period of data is available from Mauna Loa Observatory in Hawaii (Fig. 19.4). We have to consider a continuous increase since the 1960s from 310 ppm (0.031 %) of up to 400 ppm today (0.04 %). Looking closer at the graph in Fig. 19.4, we find an annual cycle. As argued correctly in Al Gore's movie "The Inconvenient Truth" and further popular publications, this annual cycle is related to the change in photosynthesis of vegetation in the northern hemisphere. CO₂ reduces in the spring in April, whereas in October, when plants loose their leaves, CO₂ level again increases. The annual CO₂ cycle clearly expresses the photosynthesis process, but the overall increase since 1960 is being attributed to anthropogenic carbon emissions. This is a misinterpretation. Already Keeling 1960 concluded that "since the seasonal variation in concentration observed ... is several times larger than the annual increase, it is as reasonable to suppose, however, that a small change in the factors producing this seasonal variation may also have produced an annual change ...".

The reason why anthropogenic carbon emissions are a non-satisfying explanation of CO₂ increase involves ecological dynamics. The dynamic cycle of CO₂ in the

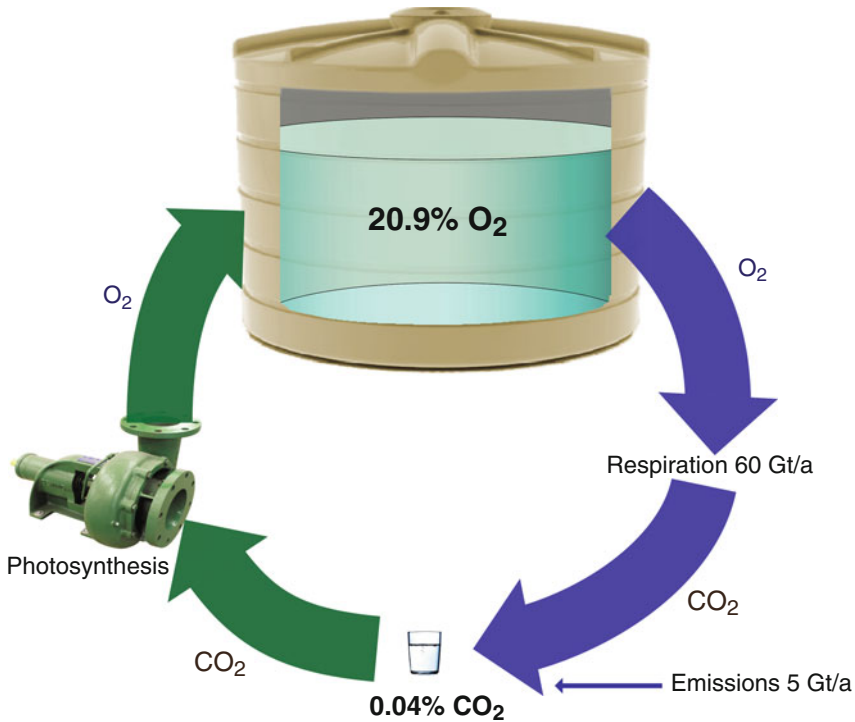


Fig. 19.5 Relation between O₂ and CO₂ by analogy with two types of water storage

atmosphere has its counterpart in oxygen O₂. The rate of CO₂ is 0.04%, whereas O₂ has a concentration of 20.9%. O₂ has a much higher energetic level and humans gain their energy by breathing O₂ and releasing CO₂.

To illustrate the ecological cycle in a dynamic way, we might explain the relation between O₂ and CO₂ by analogy with two types of water storage, a small glass which represents CO₂ on the ground floor of a building and a huge tank on top of the building which represents O₂ (Fig. 19.5). The large tank leaks a bit, converting O₂ into CO₂, in ecology this we call respiration. In between the elements we have a huge pump that puts water out of the small glass into the huge tank above. The pump represents the photosynthesis process. The pump is largely oversized, the relation of CO₂ and O₂ is 0.04–21% instead of say 10% for both elements. Therefore it is not relevant how much CO₂ is put continuously into the glass. The problem therefore is not CO₂ emission, how much additional water is released into the small glass of water on the ground floor, the problem inheres in the performance of the pump! Therefore globally, the relation between CO₂ and O₂ will always favor CO₂. When the pump finally fails, the Earth will end as it started several million years ago at a range of fully CO₂ and zero O₂.

It is reasonable to acknowledge that emissions play a role in the dynamic cycle at a relation of 10% of CO₂ and 10% of O₂. But in a relation of 0.04–21% the dominant factor for the change in the dynamic process is the reduction in photosynthesis, not any emissions.

19.5 Sustainable Water Management – Case Studies

Implementing the New Water Paradigm at a local level requires rethinking urban planning and water management infrastructures. With regard to the urban heat island effect and the issue of global warming, urban planners, architects and landscapers need to consider the natural water cycle, including evaporation, condensation and precipitation (Milosovicova 2010). The conventional management of water discharge, which was implemented for over a 100 years, nowadays bears disastrous environmental effects on both surface water quality and on the climate. More recently, rainwater infiltration has been a popular strategy in Germany. However, in spite of the great benefit of preventing negative impacts to surface waters, infiltration does not fully rectify the natural water cycle. Urban areas are not characterized by reduced infiltration rates (SenStadt 2013). The missing hydrological component is evaporation.

In the catchment area of Berlin/Brandenburg, about 80% of precipitation is converted to evaporation, while groundwater recharge and runoff together represent 20%. Urban areas are characterized by completely paved or semi-permeable surfaces with little to no vegetation. The semi-permeable surfaces allow much higher groundwater recharge compared to naturally vegetated areas (Schmidt et al. 2005), as they over-compensate for infiltration with reference to completely paved surfaces. Therefore, in the interest of effective environmental care taking, the provision for evaporation rather than infiltration needs to become a primary task.

Through urban design strategies such as green facades, green roofs and stormwater ponds, rainwater harvesting cisterns can play supportive roles as adaptation and mitigation strategies against the urban heat island effect and global warming. By applying the New Water Paradigm (see www.waterparadigm.org) small local water cycles are restored. Harvesting rainwater for evaporation should be a first priority in urban areas. Already several projects have been established in Berlin with a focus on rainwater harvesting and evaporation, of which two are presented here – UFA Fabrik and Potsdamer Platz.

19.5.1 Case Study 1: UFA-Fabrik in Berlin-Tempelhof

The Cultural Center UFA-Fabrik in Berlin-Tempelhof, home to various urban ecology initiatives (see: www.ufafabrik.de) includes an integrated rainwater management project. As a first measure, most of the roofs were vegetated from 1983 to 1985 (Figs. 19.6 and 19.7). In 1994, a rainwater harvesting system was introduced. As a result, water from conventional and green roofs was able to be stored in a former underground waterworks station, along with runoff from street level.

The rainwater system at UFA-Fabrik has a total storage capacity of 240 cubic meters in two cisterns. This is equivalent to 40 mm or 6.7% of annual precipitation of the catchment area. The system collects primarily first flush stormwater. By



Fig. 19.6 On top of a Greened roof at UFA-Fabrik Berlin-Tempelhof (Photo author)

capturing the pollutants and nutrients associated with the first flush, UFA-Fabrik with a separated sewer system, provides increased ecological benefits by directing polluted runoff to a modified constructed wetland for treatment. Collected rainwater is used to flush toilets and for irrigation. About 75 % of rainwater used in the summer month is linked to irrigation. This rate of use, the large storage capacity for rainwater, and the greened roofs represent best management practice for the New Water Paradigm.

The constructed wetland (Fig. 19.8) has a size of 25 square meters for a mean treatment of 10 cubic meters per day. This represents a mean treatment value of 400 l of rainwater per day per square meter. Due to the large use of rainwater for irrigation in summer time, the amount of treatment in winter is much less, this corresponds to the treatment efficiency of a constructed wetland during low temperatures. The soil type has been changed twice over the lifetime of the project to improve the treatment efficiency. In preparation for the second case study Potsdamer Platz, several different types of substrate were tested on-site (Fig. 19.8). The substrate which performed best, a mixture of gravel, zeolithe and expanded slade, was selected (see next case study). Soil clogging was dominating factor during the testing period. Finally a grading of 2–4 mm was selected to fulfill the requirements on overall performance.



Fig. 19.7 Green roof combined with photovoltaic panels at UFA-Fabrik Berlin-Tempelhof (Photo author)

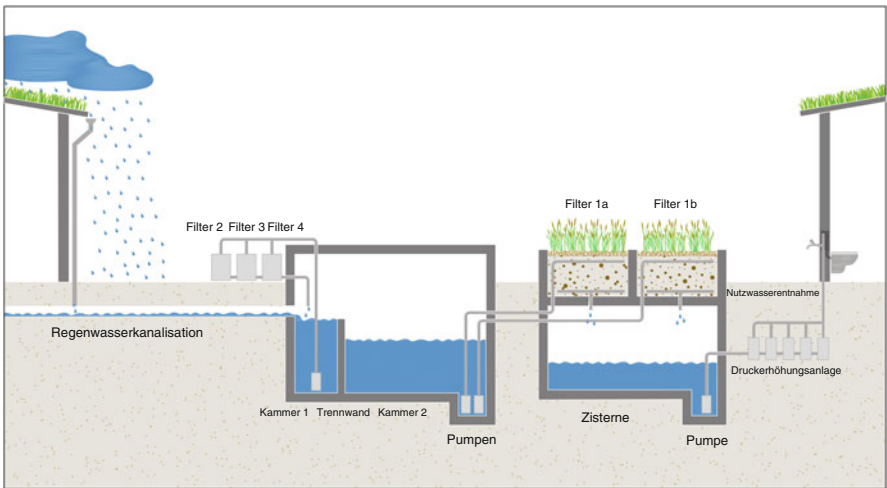


Fig. 19.8 Rainwater harvesting including constructed wetlands for treatment at UFA Fabrik Berlin

19.5.2 Case Study 2: DCI Berlin, Potsdamer Platz

The large constructed wetland (Fig. 19.9) and so called urban lake (Fig. 19.10) at Potsdamer Platz are part of an overall concept of sustainable rainwater management set up in 2000. The city administration had demanded a reduction of rainwater runoff into the combined sewer system by 99% of its intensity during stormwater



Fig. 19.9 Constructed wetland for rainwater treatment of roof runoff at Potsdamer Platz, Berlin (*left*) and its construction in 1998 (*right*) (Photo author)



Fig. 19.10 Rainwater retention by an artificial urban lake at Potsdamer Platz, Berlin (Photo author)



Fig. 19.11 Green roofs at Potsdamer Platz during construction in 1998 (Photo author)

events. The goal was to reduce overload of the combined wastewater sewer. Three measures of sustainable water management were integrated to fulfill this (Schmidt and Teschner 2000). A co-operation between Atelier Dreiseitl, Krüger and Möhrle and TU Berlin combined green roofs (Fig. 19.11), rainwater harvesting and a large urban lake of 12,000 square meters to retain the roof runoff of 46,000 square meters from 17 buildings.

The main function of the constructed wetland is the reduction of nutrients in the urban lake to avoid algae growth (Teschner and Schmidt 2000; Teschner 2005). Water continuously circulates through the gravel filter. Tests for 6 different soil types have been carried out in facilities at www.ufafabrik.de (first case study). Soil clogging was one of the first problems occurring after 4 weeks. The artificial substrate, consisting out of zeolite, expanded slate and gravel, was therefore selected at larger aggregates of 2–4 mm. Water quality in the cascade roofs – cistern – urban lake – circulation tank retains 99 % of its phosphorous load as well as nitrogen (Fig. 19.12). Remaining concentration of phosphorous is less than $15 \mu\text{g l}^{-1}$ which represents an oligotrophic surface water. Interesting is the phosphorous peak around New Year 2000 due to fireworks (Fig. 19.12). Nitrogen concentration reduces in the cascade cistern – urban lake from 2000 to around $300 \mu\text{g l}^{-1}$ caused by biologic processes in the tank. Compared to processes in treatment plants, treatment efficiency improves with a long retention time of rainwater in the tank. Additional technical filters in the building complex like a micro sieve give support to the cleaning process. The rainwater concept at Potsdamer Platz is based on a new approach of sustainable water management. Instead of common runoff management based on sewer systems or infiltration into the ground water, this system provides evaporation and rainwater use for toilet flushing.

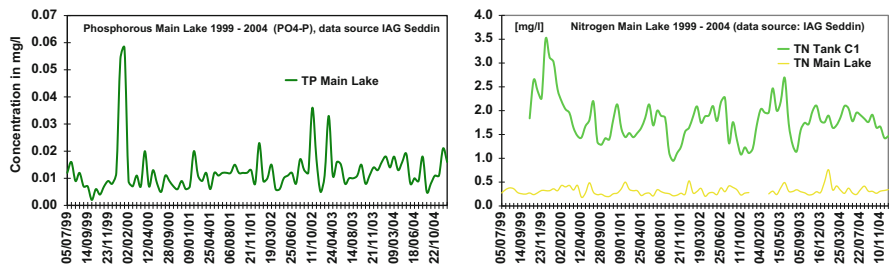


Fig. 19.12 Phosphorous concentrations in main lake (*left*) and nitrogen concentration in rainwater tank and artificial urban lake (*right*) (Source of data: DCI; Meisel, IAG Seddin)

Green roofs are a basic measure for the whole water management concept. Evaporation rates are already 70% of annual precipitation. 80% of polluting nutrients are filtered out by soil, moss and weeds. Several different types of substrate were tested at the Technical University of Berlin for best performance on nutrient removal before being spread on all 17 buildings at Potsdamer Platz.

19.6 Conclusion

Attention to vegetation, climate and water seriously challenges conventional climate measures. Rather than focusing on greenhouse gas emissions to address climate change, the emphasis should be on planting trees and improving soils to store more rainwater and by this means to recover the small water cycle of evaporation, condensation and precipitation.

It is not generally realized that the correlation between CO₂ in the atmosphere and global temperatures as shown in ice core drillings and used as a basis for the climate debate is an indirect correlation. Vegetation and its capacities for evaporation and photosynthesis is the driving force behind carbon increase and behind increased temperatures at the same time.

The current increase in CO₂ and other greenhouse gases has no influence on the energy budget of the Earth. Temperatures and atmospheric counter radiation are fully dominated by evaporation and condensation. Condensation takes place inside the atmosphere above the heavier CO₂ gas levels. Thus, the much publicized greenhouse gases actually have an opposite effect to what is claimed regarding the energy release of condensation as part of the water cycle. For the large amount of latent heat released in the atmosphere greenhouse gases reduce counter radiation, therefore reduce the greenhouse effect. All in all, the absorption of a certain wavelength does not mean, that increased concentrations of greenhouse gases increase the long wave radiation of the atmosphere. What has to be considered also, is the question of at which altitude of the atmosphere the energy of the condensation process is released.

The global climate is dominated by evaporation and condensation of water. In the large component of condensation that makes clouds, energy is released as long-wave radiation inside the atmosphere. In addition to the release of 700 kWh per cubic meters of water in the phase change from water vapor to the liquid state, 91 kWh of energy is also released in the phase change of water into ice.

Rainwater harvesting measures which focus on evaporation rather than infiltration have considerable potential to decrease the environmental impacts of urbanization. Under the New Water Paradigm, global climate change is attributed to reduced evaporation (Kravčík et al. 2007). The popular focus on reducing greenhouse gas emissions to mitigate global warming misconstrues of environmental processes. Computer simulations of global climate change continue to neglect the fundamental driving forces of the global climate: transpiration by vegetation and evaporation from land. The importance of plant photosynthesis cannot be underestimated.

The research summarized above proves that evaporation of water is the cheapest and most effective way to cool a building and a city. It is known that one cubic meter of evaporated water consumes 700 kWh of heat. Thus, instead of energy expensive conventional air conditioning systems, which produce additional heat and release it outside the building, the evaporation of water simply consumes heat at low operating costs.

No drop of water should be funneled into sewer systems or pumped into oceans without being used first for irrigation and further evaporation. The worldwide destruction of vegetation, particularly in North Africa and the Middle East has caused droughts and climate change. With the New Water Paradigm, the critical concept is that global dehydration and drought is what causes climate change, not that climate change is causing drought. The correlation between CO₂ and global temperatures in the past was always merely an indirect correlation with vegetation and water. Rectifying climate change environmentally will mean focusing on precipitation, vegetation and soils.

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Chapter 20

Abundance and Diversity of Taxa Within the Genus *Potamogeton* in Slovenian Watercourses

Mateja Germ, Urška Kuhar, and Alenka Gaberščik

Abstract This contribution presents the species abundance and diversity of the taxa of genus *Potamogeton* in selected Slovenian watercourses. A total of 40 watercourses, divided into 1231 reaches were surveyed and 8 species of the genus *Potamogeton* were assessed. Among these, *Potamogeton nodosus* was the most abundant, followed by *P. pectinatus* and *P. natans*. *P. crispus* was found to be the most frequently encountered species found in 251 reaches. Three hybrids, *P. × cooperi*, *P. × salicifolius* and *P. × zizii* were also found, each in only one or two watercourses. Pondweeds accounted for 19.5 % of the total relative abundance of all macrophytes in the watercourses that were surveyed. Canonical correspondence analysis reveals that land use, structure of riparian zone, substrate quality and flow dynamics exert a significant effect on distribution and abundance of pondweeds. *Potamogeton* species that were detected have different indicator values.

Keywords Genus *Potamogeton* • Hybrid • Abundance • Watercourses • Environmental parameters

20.1 Introduction

Macrophytes are involved in energy flow, nutrient cycling, habitat provision and sedimentation processes (Baatrup-Pedersen and Riis 1999). The diversity, abundance and distribution of macrophyte communities are affected by various environmental factors (Kuhar et al. 2007; Dar et al. 2014; Kennedy et al. 2015), but primarily by changes due to anthropogenic activity (Hrivnák et al. 2009).

Pondweeds (genus *Potamogeton*) are a widespread genus of macrophytes experiencing different environmental conditions, colonising a variety of habitats from

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running to stagnant waters and exhibiting different growth forms (Preston 2003). Different species colonise watercourses with different water regimes and different nutrient content (Germ et al. 2003, 2008; Kržič et al. 2007; Kuhar et al. 2007), and consequently they have also different indicator values (Kuhar et al. 2011). Interspecific hybridization is frequently encountered within the genus *Potamogeton* (Preston 2003) and is very important, because it is one of the main sources of the taxonomic complexity of the genus *Potamogeton* (Kaplan and Fehrer 2011). The *Potamogetonaceae* comprises a diverse family of aquatic plants with three genera, the genus *Potamogeton* with about 72 species and 99 hybrids, the monotypic genus *Groenlandia*, and the genus *Stuckenia* with 7 species and 6 hybrids (Kaplan et al. 2013). In Slovenian flora (Martinčič et al. 2010) the *Potamogetonaceae* family consists of two genera, *Potamogeton* and *Groenlandia*, and 20 species the majority of which are characterised as standing or slow flowing water species. No hybrids within the genus *Potamogeton* are recorded on lists of Slovenian flora.

The aim of this study is to (1) examine the presence and abundance of different taxa of the genus *Potamogeton*, including hybrids in Slovenian watercourses, (2) describe their prevailing abiotic habitat characteristics and (3) identify macrophyte species with which the representatives of the genus *Potamogeton* co-exist.

20.2 Materials and Methods

20.2.1 Study Area

The survey of *Potamogeton* taxa together with other macrophytes was carried out in 1231 reaches of a total of 40 watercourses. We investigated rivers and streams which belong to three Slovenian hydro-ecoregions, the Dinarides, the Pannonian lowland and the Po lowland. 35 watercourses were examined in the entire length and 5 watercourses were partially examined.

20.2.2 Macrophyte Survey

In the majority of watercourses, subsequent reaches of different lengths were surveyed. The surveys were performed from the bank of the stream or from a boat using a rake with hooks to sample plants. Macrophyte species abundances were estimated using the five-degree scale: 1=very rare; 2=infrequent; 3=common; 4=frequent; 5=abundant, predominant (Kohler 1978; Kohler and Janauer 1995) and the relative abundance was calculated using the methodology proposed by Pall and Janauer (1995).

20.2.3 *Environment Assessment*

The environmental condition of watercourses where the macrophytes were encountered was assessed using the modified Riparian, Channel and Environmental (RCE) Inventory proposed by Petersen (1992). This consists of 11 parameters, each describing 4 levels of environmental gradient. The parameters include land-use type beyond the riparian zone, characteristics of the riparian zone (width, completeness and vegetation type) and morphology of the stream channel (bank structure and undercutting, occurrence of retention structures and sediment accumulation, type of stream bottom and detritus, and dynamics of the flow (occurrence of riffles, pools and meanders)) (Germ et al. 2000; Kuhar et al. 2007).

20.2.4 *Statistical Analysis*

Canonical correspondence analysis (CCA) was used to assess the relationships between plant species composition and abundance and environmental parameters, which were coded numerically from 1 to 4. Forward selection was used to determine the contribution of each parameter to the variance in species composition. The statistical significance of environmental parameters was tested by a Monte Carlo permutation test. Analyses were performed using Canoco for Windows Version 4.5.

20.3 *Results and Discussion*

Eight species of the genus *Potamogeton* were determined in the survey of 40 Slovenian watercourses (Table 20.1). Among them *P. nodosus* reaches the highest relative abundance (5.9 %) followed by *P. pecinatus* and *P. natans* with 3.8 and 2.9 % relative abundance respectively (Fig. 20.1). *P. crispus* was the most frequently occurring *Potamogeton* species (Table 20.1), but it was only the 4th species with respect to its relative abundance, which was 2.7 % (Fig. 20.1). Three hybrids were also found, namely *P. × cooperi*, *P. × salicifolius* and *P. × zizii*. Each of these was found in only one or two watercourses. *P. × zizii* grows in Rivers Mali Obrh and Rak, and *P. × salicifolius* was found in the River Rak and River Kolpa, and *P. × cooperi* in the River Dravinja. The hybrids *P. × cooperi* have been recorded in countries mainly in the northern half of Europe, while *P. × salicifolius* was reported in several European countries and in Siberia (Kaplan et al. 2013).

CCA revealed that 10 out of the 11 environmental parameters that were examined, namely land use pattern beyond the riparian zone, stream bottom, width and completeness of the riparian zone, type of riparian vegetation, retention structures, sediment accumulation, occurrence of riffles, pools and meanders, presence of detritus and bank undercutting exert a significant effect on the presence and

Table 20.1 *Potamogeton* taxa found in the survey of the Slovenian rivers in 1231 reaches. Their abbreviations, habitus and frequency of occurrence

Taxon	Abbrev.	Habitus	Relative frequency
<i>Potamogeton crispus</i> L.	Pot cri	Broad leaved, submerged	251
<i>Potamogeton nodosus</i> Poir.	Pot nod	Floating leaved with submerged leaves	242
<i>Potamogeton perfoliatus</i> L.	Pot per	Broad leaved, submerged	201
<i>Potamogeton pectinatus</i> L. (syn. <i>Stuckenia pectinata</i> L.)	Pot pec	Narrow leaved, submerged	138
<i>Potamogeton natans</i> L.	Pot nat	Floating leaved with submerged phylloids	128
<i>Potamogeton lucens</i> L.	Pot luc	Broad leaved, submerged	88
<i>Potamogeton berchtoldii</i> Fieber	Pot ber	Narrow leaved, submerged	66
<i>Potamogeton trichoides</i> Cham & Schldtl.	Pot tri	Narrow leaved, submerged	10
<i>Potamogeton</i> × <i>zizii</i> W.D.J. Koch ex Roth (<i>P. gramineus</i> L. × <i>P. lucens</i> L.)	Pot ziz	Broad leaved, submerged, occasionally with floating leaves	4
<i>Potamogeton</i> × <i>salicifolius</i> Wolfg. (<i>P. lucens</i> L. × <i>P. perfoliatus</i> L.)	Pot sal	Broad leaved, submerged	2
<i>Potamogeton</i> × <i>cooperi</i> Fryer (<i>P. crispus</i> L. × <i>P. perfoliatus</i> L.)	Pot coo	Broad leaved, submerged	1

abundance of *Potamogeton* species (Fig. 20.2). The distribution of different species along the vectors reveals a variety of habitats that are colonised by different pondweeds. The majority of species in the plot were found close to the intermediate values of the quality gradient of the environmental parameters, but *P. berchtoldii* and *P. trichoides* were found in morphologically highly modified streams flowing through agricultural areas (Kuhar et al. 2007).

We also examined the co-existence of *Potamogeton* species with four of the most abundant non *Potamogeton* macrophyte taxa in the watercourses studied (Fig. 20.3). The CCA plot shows that *P. nodosus* co-exists with *Myriophyllum spicatum*, while its abundance is in reverse relation with the abundance of *Elodea canadensis*. *P. natans* and *P. crispus* co-exist with *E. canadensis*. *P. berchtoldii* often co-exists with *Phalaris arundinacea*, while *P. perfoliatus* shares the habitat with Bryophytes, as in karst streams (Kržič et al. 2007). This plot also revealed that vector “*M. spicatum*” proceeds in an opposite direction to the vector “number of taxa”. It has been shown in previous studies that *M. spicatum* as a species with a wide ecological range negatively affects the populations of other macrophytes (Boylen et al. 1999). *M. spicatum* is the most abundant species in Slovenian watercourses and also interact with the alien species *E. canadensis* as revealed by opposite direction of vectors of both species. Previous analyses showed that in the half of the reaches with *E. canadensis*, *M. spicatum* was not present whilst, when both species were present, *E. canadensis* was more abundant (Kuhar et al. 2010). *P. berchtoldii* and *P. trichoides* are found in

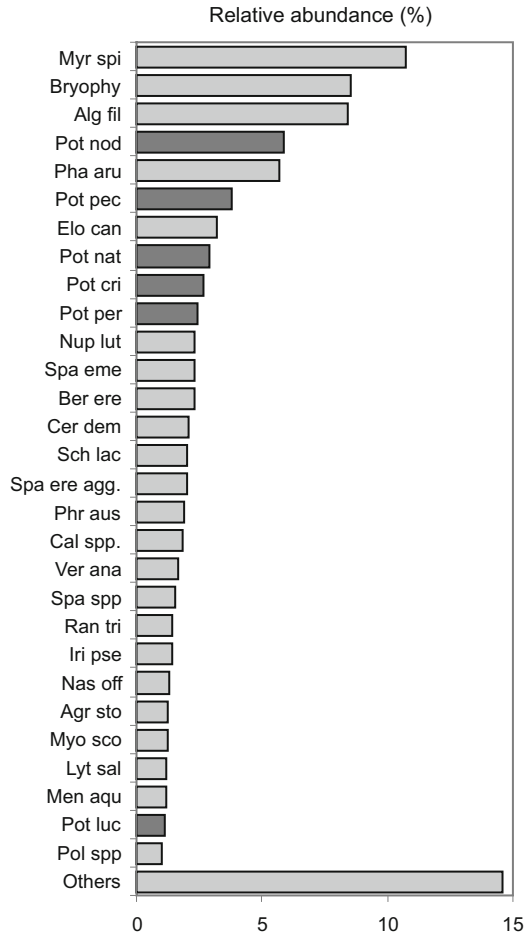


Fig. 20.1 Relative abundance of macrophytes and representatives of the *Potamogeton* species (dark grey column) in Slovenian rivers. Only species having a relative abundance above 1 % are shown and species with lower values are presented cumulatively as “Others”. Abbreviations for *Potamogeton* species are in Table 20.1. Other species: Agr sto – *Agrostis stolonifera*, Alg fil – filamentous algae, Ber ere – *Berula erecta*, Bryophy – Bryophyta, Cal spp – *Callitriche* spp., Cer dem – *Ceratophyllum demersum*, Elo can – *Elodea canadensis*, Iri pse – *Iris pseudacorus*, Lyt sal – *Lythrum salicaria*, Men aqu – *Mentha aquatica*, Myo sco – *Myosotis scorpioides*, Myr spi – *Myriophyllum spicatum*, Nas off – *Nasturtium officinale*, Nup lut – *Nuphar luteum*, Pha aru – *Phalaris arundinacea*, Phr aus – *Phragmites australis*, Pol spp – *Polygonum* spp., Ran tri – *Ranunculus trichophyllus*, Sch lac – *Schoenoplectus lacustris*, Spa eme – *Sparganium emer-sum*, Spa ere agg – *Sparganium erectum* agg., Spa spp – *Sparganium* spp., Ver ana – *Veronica anagallis-aquatica*

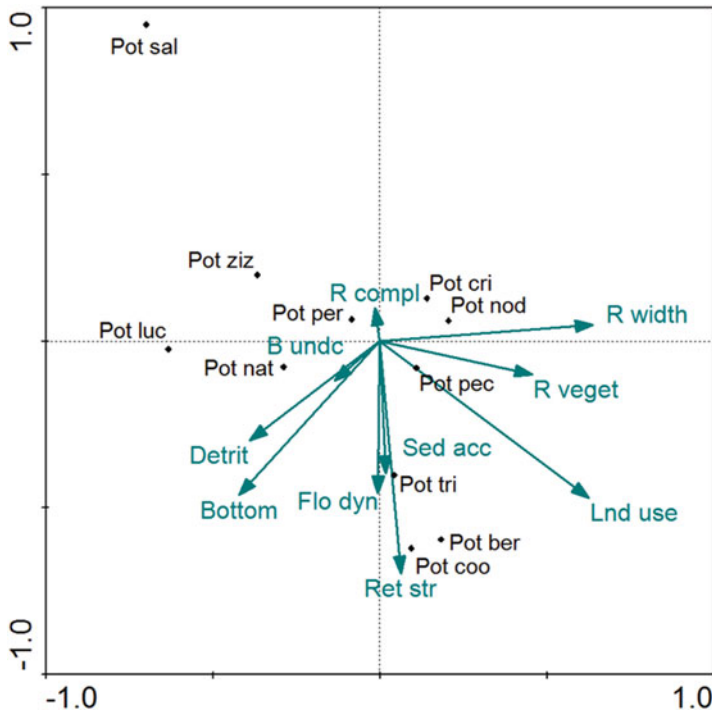


Fig. 20.2 CCA ordination diagram showing the relationship between *Potamogeton* species and environmental parameters. The quality of the environmental parameters decreases in the direction of the arrows. The origin of vectors presents the middle of the quality gradient of the environmental parameter. Codes used for taxa are given in Table 20.1. Codes for environmental parameters are as follows: *Lnd use* land use pattern beyond the riparian zone, *R width* width of the riparian zone, *R compl* completeness of riparian zone, *R veget* vegetation of riparian zone, *Ret str* retention structures, *Sed acc* sediment accumulation, *B undc* bank undercutting, *Bottom* stream bottom, *Flo dyn* flow dynamics (occurrence of riffles, pools and meanders), *Detrit* detritus

the species rich habitats, while the others thrive in less diverse communities. This is possibly due to small growth forms of the former and usually much larger growth forms of the latter.

Different tolerance to environmental parameters in different species is related to their indicator value. According to Slovenian Macrophyte Index (RMI) that is based on the level of natural quality of the stream namely on the ratio of natural areas in the sub-catchment (Kuhar et al. 2011), *P. × salicifolius* is classified into group A indicating reference sites, *P. crispus* and *P. lucens* are classified into group AB indicating reference and moderately loaded sites, *P. pectinatus* is in group B, indicating moderately loaded sites, and *P. nodosus* is in group BC, indicating moderately and heavily loaded sites. *P. natans* and *P. perfoliatus* were classified in group ABC, having no indicator value, because the two species have wide ecological valence (Preston 2003). With respect to the trophism, Haslam (1987) pointed out species that are frequently found in eutrophic habitats, i.e. *P. crispus* and *P. lucens*. The

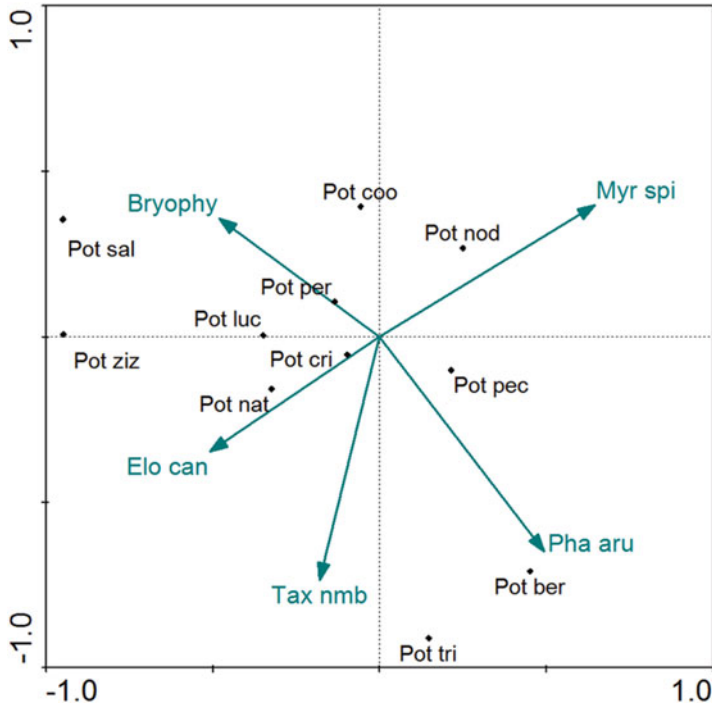


Fig. 20.3 CCA ordination diagram showing the relationship between *Potamogeton* species and macrophytes with high relative abundance and total number of macrophyte taxa. The abundance of macrophytes increases in the direction of the arrows. The origin of the vectors represents the middle of the abundance gradient. Codes used for taxa are given in Table 20.1 and in caption of Fig. 20.1. *Tax nmb* number of taxa

same holds true for *P. nodosus* (Preston 2003). Schneider and Melzer (2003) report that *P. natans* is characteristic of waters with medium nutrient load, while *P. lucens* is found in waters with medium to high nutrient loads. *P. crispus* and *P. pectinatus* can tolerate high levels of phosphorus and ammonia (Dykyjová et al. 1985), while Mackay et al. (2003) report that *P. crispus* is associated with waters with low total phosphorus and intermediate to low alkalinity. Our results and those of different researchers examining the relation of pondweeds with nutrients are inconsistent. It is likely that a combination of different environmental factors affect their tolerance to different nutrient levels.

In Europe, there is latitudinal gradient in hybrid diversity and number of localities with hybrids (Kaplan et al. 2013). Relatively high numbers of *Potamogeton* species and a finding of only three hybrids in studied watercourses is possibly a consequence of the high morphological variability of Slovenian watercourses where habitat conditions vary significantly along the stream and pronounced changes and unevenness of macrophyte communities are common (Kržič et al. 2007; Kuhar et al. 2010). The reason for this is outstanding heterogeneity of Slovenian landscape with

variable geomorphology and steep altitudinal gradients, variable vegetation on land which is 60 % forested, and a variable climate with a steep gradient of precipitation rate in a relatively small area (ARSO 2001).

20.4 Conclusions

Evidence was found for eight different *Potamogeton* species in 1231 stretches of 40 watercourses that were surveyed. *P. crispus* was the most frequently occurring species, while *P. nodosus* was the most abundant. Only three hybrids, *P. × cooperi*, *P. × salicifolius* and *P. × zizii*, were found to be distributed in limited parts of the watercourses and reaches examined. The relatively high number of *Potamogeton* species and the low number and occurrence of hybrids is possibly a consequence of the high morphological variability of watercourses in a highly heterogeneous Slovenian landscape. The detected taxa thrive in a variety of habitats but not in extremely disturbed habitats.

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