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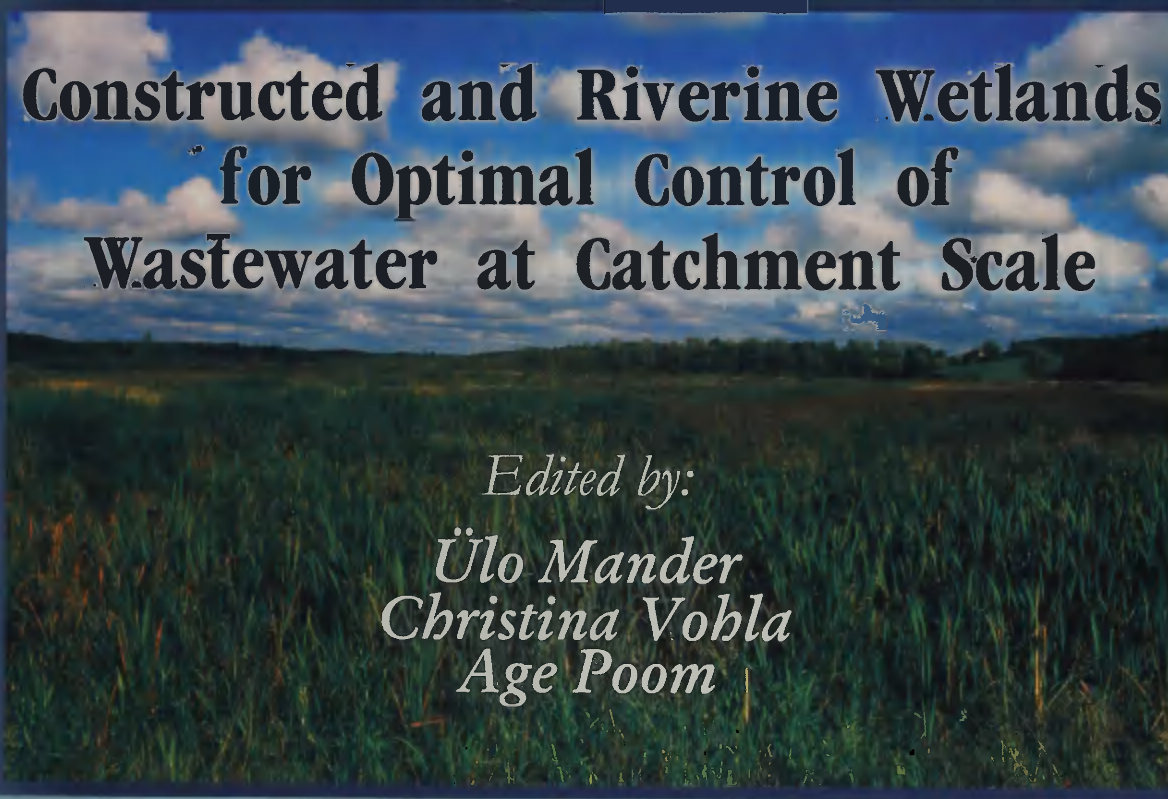

Constructed and Riverine Wetlands for Optimal Control of Wastewater at Catchment Scale

Edited by:

Ülo Mander
Christina Vohla
Age Poom

Tartu, 2003

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International Conference

Constructed and Riverine Wetlands for Optimal Control of Wastewater at Catchment Scale

Conference proceedings

EU 5th FP RTD **PRIMROSE** "**PR**ocess Based **I**ntegrated **M**anagement of Constructed and **R**iverine Wetlands for **O**ptimal Control of Wastewater at Catchment Scal**E**" (EVK1-CT 2000-00065)

LIFE Environment Project "Sustainable Wastewater Purification in Estonian Small Municipalities" (ENV/EE/00924)

Norwegian Centre for Soil and Environmental Research (Jordforsk), Ås, Norway
Institute of Geography, University of Tartu, Estonia

Edited by:
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Introduction

Welcome to the International Conference on *Constructed and Riverine Wetlands for Optimal Control of Wastewater at Catchment Scale* jointly organised by the Norwegian Centre for Soil and Environmental Research (Jordforsk), Ås, Norway, and the Institute of Geography, University of Tartu, Estonia. It is a concluding scientific meeting of the EU 5th FP RTD project PRIMROSE "**PR**ocess Based **I**ntegrated **M**anagement of Constructed and **R**iverine Wetlands for **O**ptimal Control of Wastewater at Catchment **S**cal**E**" (EVK1-CT 2000-00065), and a meeting of the EU LIFE Environment Project "Sustainable Wastewater Purification in Estonian Small Municipalities" (ENV/EE/00924), which is extended by the keynote papers from outstanding specialists in the field of treatment wetland studies and volunteered papers and posters. The conference will be held from 29 September to 2 October in Tartu, Estonia.

The main objective of the PRIMROSE project is to develop the design, dimensioning, and location of constructed wetlands on the base of water purifying processes. These processes – which are highly dependent on water movement patterns in the wetland – include sedimentation of suspended particles, phosphorus adsorption in soil, and the nitrification-denitrification of nitrogen. In natural conditions various factors are in complex interaction, making it quite difficult to predict water purification in wetlands. Hence the need for tools permitting the better estimation of wetland performance is obvious. One aim of the PRIMROSE project is to develop mathematical models for this purpose. Moreover, one objective is to use a statistical approach for the modelling of nutrient retention in the channel network and large natural wetland units in catchments.

The information needed in the modelling work is acquired by reviewing the literature and through research performed in various types of wetlands. The research work includes not only inflow-outflow monitoring but also measurements of purification processes and wetland hydraulics. There are a total of 17 study sites in the participating countries. The developed tools are tested in different catchments in participating countries. PRIMROSE consists of seven work packages, and their participants come from six European countries: Norway, Sweden, Finland, Estonia, Austria and Poland. The Jordforsk Institute in Norway is the co-ordinator of the project.

The aim of the EU LIFE Environment Project "Sustainable Wastewater Purification in Small Estonian Municipalities" is to reduce the N and P content in purified water, using soil as a filter. In addition, two sustainable wastewater purification systems, a short rotation willow coppice plantation-wastewater infiltration field in Kambja, Tartu County, and a hybrid treatment wetland system in Lääne-Viru County have been established as a means of addressing local environmental and energy supply problems in rural areas of Estonia.

The aim of the conference is to provide participants with new and innovative ecological methods for wastewater treatment, which will help reduce pollution. These ecological methods are preferable in areas with sensitive waterbodies and endangered groundwater resources. They are used for the treatment of wastewater from point pollution sources (settlements, farms) as well as diffuse sources such as agricultural fields, mining and peat production areas. New results will be presented on the use of models and GIS-based approaches for better water pollution control to fulfil requirements stated by the EU water framework and nitrate directive.

During the Conference excursion on 1st October to the Estonian PRIMROSE project sites (the Kodijärve hybrid wetland system in Tartu County, the Tänassilma riverine wetland ecosystem and the Kõo hybrid wetland system in Viljandi County, and the Põltsamaa free water surface wetland system in Jõgeva County), the participants will become acquainted with the design, performance problems, and management problems of these various types of treatment wetland systems. The post-conference excursion from 3rd to 5th October to western Estonia and Saaremaa Island will allow the visiting of several treatment wetland systems but also protected natural wetlands.

In the proceedings we offer extended abstracts of 46 oral presentations and 27 posters presented during the Conference. Contributions to this conference represent 20 countries. In addition to presentations from PRIMROSE-project partner countries Norway, Finland, Sweden, Austria, Poland and Estonia, contributions from Australia, Belgium, Canada, China, Czech Republic, Denmark, Germany, Lithuania, Russia, Spain, the Netherlands, Ukraine, the United Kingdom and the United States of America are included. The volume of preprint proceedings consists of keynote papers and volunteered papers on the following topics: modelling of nutrient reduction in wetland ecosystems, hydrological aspects in free water surface constructed wetlands for wastewater treatment, patterns and processes in subsurface flow constructed wetlands for wastewater treatment, phosphorus sorption and ecotoxicological properties of wetland soils and filter media, and several case studies on wetlands treating wastewater from various sources.

The compilation of this volume is the result of the efforts of a number of people. We wish to acknowledge assistance of the Organising Committee and Program Committee, which helped to lay the groundwork for the conference, plan the program, and identify highly qualified speakers to invite. Our special appreciation goes to Ms. Christina Vohla and Ms. Age Poom, students of the University of Tartu, who worked effectively and tirelessly with each of the 73 abstracts during the preparation of this volume. Mr. Alexander Harding provided useful help in correcting the English. We also acknowledge the financial help of the Board of the IALE, the Swedish Environmental Protection Agency, the Swedish organisation FORMAS, and the hosting Universities, the University of Stockholm and the University of Tartu.

Any opinions, findings, conclusions or recommendations expressed in this publiccation are those of the authors and do not necessarily reflect the views of the Norwegian Centre for Soil and Environmental Research or the Institute of Geography at the University of Tartu.

Have a useful and fascinating conference!

Bjørn Kløve
Ülo Mander

International Conference on
Constructed and Rivereine Wetlands for Optimal Control of Wastewater at Catchment Scale
Tartu, Estonia: September 29 – October 2, 2003

Jointly organized by:

Norwegian Centre for Soil and Environmental Research (Jordforsk), Ås, Norway
Institute of Geography, University of Tartu, Estonia

as conclusing scientific meeting of the EU 5th FP RTD PRIMROSE "**PR**ocess Based **I**ntegrated **M**anagement of Constructed and Riverine Wetlands for **O**ptimal Control of Wastewater at Catchment **S**cal**E**" (EVK1-CT 2000-00065), and a meeting of the EU LIFE Environment Project "Sustainable Wastewater Purification in Estonian Small Municipalities" (ENV/EE/00924)

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Gaseous emissions from constructed wetlands and (re)flooded meadows

Jürgen Augustin

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Fens (minerotrophic mires) comprise an area of approximately $500*10^3$ km^2 worldwide and therefore belong to the most important wetlands of the world. In Germany there are $1,0*10^6$ ha of fens, and 45% thereof are located in Northeast Germany. Under natural, unimpaired conditions they essentially function as a sink and accumulate enormous C- and N-quantities, e.g. up to 630 t C and up to 120 t N per hectare, 10 to 100 times more, than mineral soils contain (Succow und Joosten, 2001). However, due to intensive cultivation the role of the fens in C and N cycles drastically changed of the last decades. Drainage and aeration of the peat substrate leads towards an acceleration of the C/N mineralisation processes, to the fast decomposition of the accumulated peat, the overall effect being a transformation of these locations from an C and N sink to a C and N source. This process is of global importance because today more than 50% of peatlands in north and central Europe as well as in south Asia are used in agriculture. In northeast Germany more than 95% of drained fens are used for agricultural production, primarily as meadows (Succow und Joosten, 2001). Since peat mineralisation is strongly connected with the formation and release of the trace gases carbon dioxide (CO_2) and nitrous oxide (N_2O) it was frequently postulated that drained fens contributes substantially to the so-called anthropogenic greenhouse effect (Augustin, 2001). However, changes in socio-economic conditions increased the chances for peatland restoration and/or for the introduction of new peatland. The re-establishment of the original sink function for C and N clearly increased in northeast Germany since beginning of the 1990's, particularly with the reflooding of the drained fens and a perspective use as constructed wetland. It is expected that this will re-establish the sink function and decrease the emission greenhouse gases (Succow und Joosten, 2001). Only recently, e.g. from the mid-1990's on, there is quantitative experimental information available about the intensity of the C and N turnover processes and resulting emission of greenhouse gases on drained fens. One open question was whether these locations do reconvert themselves after reflooding to an extremely strong source of methane (CH_4), a trace gas particularly important in connection with the anthropogenic greenhouse effect (Kim *et al.*, 1998). This process could counteract the positive climatic effect of a decreased CO_2- and N_2O release. The goal of our investigations for several years was to quantify the influence of (reflooded) fen use on the emission of nitrous oxide, methane and carbon dioxide, and to use these results to investigate the possible climatic impact of the northeast German fens. For methodical reasons our measurements first concentrated on the determination of the nitrous oxide and methane emission.

The gas flux measurements were conducted on several field experiment sites covering a range of soil and land use conditions: Sernitz-Welse-Valley (Brandenburg, site Biesenbrow): reflooding of strongly degraded and drained fen meadow with purified wastewater (treatments: drained fen meadow, reflooded meadow with common reed as an constructed wetland). Gumnitzniederung (Brandenburg, site Müncheberg): drainage of an alder swamp forest on a fen mire (treatments: deeply drained/strongly degraded, shallow drained/weakly degraded). Friedländer Große Wiese (Mecklenburg-Vorpommern, site Heinrichswalde): reflooding of a degraded fen meadow (treatments: drained. moist, reflooded). Rhin-Havelluch (Brandenburg, site Paulinenaue): different agricultural use of a degraded fen meadow (treatments: meadow without N fertilization, meadow with high rates of N fertilization (calcium ammonium nitrate), arable land use). Lange Dammwiesen (Brandenburg, site Strausberg): undisturbed, growing fen peatland (treatments: spring mire with sedges). On these experimental sites, short-time trace gas flux measurements (1 h) have been carried out by the closed chamber-method (covering gas collection enclosure: PVC, height 50 cm, diameter 50 cm, volume 65; or enclosure with gas circulation: height 1.8 m depending on vegetation). Gas samples have been taken by means of evacuated (gas sealed) flasks (100 ml). The trace gas concentrations have been determined in the lab by using automatic gas chromatographic systems (detectors: ECD and FID, modified according to Loftfield *et al.*, 1997). The emission rates of trace gases have been calculated as the difference of gas concentrations between the beginning and end of measurements, corrected for the area and volume of the chamber.

The overall conclusion from the experiments is that besides the depth of the ground water table the trace gas fluxes are affected in a very complex way by a number of factors (Table 1). As expected, drained fens sites therefore frequently places a strong source of nitrous oxide, at the same time as a weak methane sink in addition. Extremely high nitrous oxide missions resulted on freshly drained peatlands, and on older drained fens after extremely high N fertilizer rates and after grassland ploughing (fallow land). However at constantly low ground water table fens that are used extensively are a weak source of nitrous oxide only. Reflooding of drained and degraded fens caused a further decrease of the nitrous oxide emissions; the same is true for virgin fens.

Methane emissions, however, reacted always opposite to rising groundwater levels. Only small methane fluxes could be observed in the shallow drained alder forest and on the rewetted fen meadow. Complete reflooding always caused a drastic increase of the methane emissions. Similarly, the unimpaired peatland showed very high methane fluxes. In general, these results agree with investigations at other European mire sites, e.g. drained mires in Western Germany (Flessa *et al.*, 1998), the Netherlands (Velthof *et. al.*, 1996, Van den Pol-van Dasselaar *et al.*, 1998), and Finland (Nykänen *et al.*, 1995, Maljanen *et al.*, 2003). To our knowledge, these are the only data available for reflooded fen mires. It has to be stressed that generalizing *quantitative* statements concerning the effects of cultivation are hardly possible. This is due to the high spatial and temporal variability of the gas emissions (Nykänen *et al.*, 1995; Van den Pol-van Dasselaar *et al.*, 1998; Augustin, 2001) that are caused by the interactions of the anthropogenic and natural processes underlying trace gas emissions.

In order to determine the climatic relevance of drained and reflooded fens data published by Mundel (1976) and Armentato and Menges (1986) were included, respectively, the latter investigating virgin peatlands. Our results and other published research strongly suggest that reflooding leads to a reduction of the climatic impact of the fens, irrespective of the strong rise of the methane emissions (Table 2). The key processes and – at the same time – the weakest data sets are related to CO_2 emission because they have the strongest influence on the climatic relevance of distinct fen sites. Therefore, it is of highest importance and priority to determine the net CO_2 fluxes comprehensively and precisely. The span of data available may not only indicate spatial and temporal variation of the processes, but may suggest that fen types can exhibit a broad spectrum of climatic relevance. Therefore, realistic estimates on a regional and global level can only be expected only if the individual characteristics of a fen types are connected to regional characteristics.

If this is done for the northeast German fens, the emissions of these ecosystems at present amount to approx. 2.474.5 kt CO_2-C (net), 8305.5 t CH_4-C and 4231.7 t N_2O-N per year. This means that 2.9% and approx. 5,8% at the total German nitrous oxide emissions (145 kt N_2O-N per year, Federal Office for Environment Protection 1998) would be caused by northeast Germans fens and all German fens, respectively. Considering the small portion of the fen area of 1.5% and 3% (Statistisches Bundesamt, 1996), fens are more than proportionally involved, and they represent a regionally important source of nitrous oxide. On the other hand, at present they play no role as source of methane, regardless of the very high emissions from (the few) natural and reflooded fens, e.g. 0.2% of the German methane emissions (3543 kt CH_4-C, Umweltbundesamt, 1998). This suggests that an increase in reflooded areas or constructed wetland will not result in a relevant source of. The net CO_2 emission from fens as compared to the total of Germany (despite the uncertainties already mentioned) amount to only 1% of 250000 kt CO_2-C (Umweltbundesamt, 1998).

Table 1. Typical annual N_2O and CH_4 gas flux rates of northeast German fen mires.

Experiment	Gas emissions: Nitrous oxide ($kg\ N_2O\text{-}N*ha^{-1}*a^{-1}$)	Gas emissions: Methane ($kg\ CH_4\text{-}C*ha^{-1}*a^{-1}$)
Reflodding by purified waste water (Sernitz-Welse-Valley)		
Meadow (drained)	0.4	–0.2[2]
Constructed wetland (reflooded)	0.00	640.0
Reflooding of degraded grasland (Friedländer Große Wiese)		
Drained (60 cm WBS)[1]	0.3	1.3
Moist (10–40 cm WBS)	0.1	18.3
Reflooded (0–10 cm WBS)	0.1	521.2
Drainage of an alder swamp forest (Gumnitz)		
Deeply drained/strongly degraded	26.9	–1.4
Shallow drained/weakly degraded	0.8	1.7
Differently peatland use (Rhin-Havelluch)		
Meadow, without N fertilization	1.0	–0.3
Meadow with 480 kg N per ha	17.2	–0.6
Arable land	34.1	0.05
Undisturbed fen peatland (spring mire, Lange Dammwiesen)		
Sedges	0.3	179.0

[1] WBS = water table below soil surface
[2] negative values means methane uptake by soil (sink function)

Considering the lack of knowledge about the release of the three trace gases with impact on the climate it is strongly suggested that (i) the spectrum of the locations must be extended, and (ii) that virgin fen sites and alder swamp forests are included in the studies because only incomplete and inconsistent information is available.

Table 2. Assessment of the contribution of different fen use to the greenhouse effect (cumulative impact of trace gas emission on the radiative balance (exemplary examples, for details see Augustin, 2001).

Greenhouse gas	Drained fen mires	Reflooded fen mires
	CO_2-C equivalents (kg CO_2-C*ha^{-1}*a^{-1})[1]	CO_2-C equivalents (kg CO_2-C*ha^{-1}*a^{-1})[1]
CO_2	2900 to 6700	–140[2 3] to –2250[2 3]
CH_4	–12[1] to 29	24 to 4585
N_2O	40 to 3605	0 to 107
Cumulative impact on the radiative balance	–2226[1] to 4552	2928 to 10334

[1] relative global warming potential of trace gases with reference to CO_2: 1 kg CO_2-C = 1; 1 kg CH_4-C = 8,8; 1 kg N_2O-N = 134
[2] negative numbers means a contribution to the reduction of the greenhouse effect
[3] estimated values, taken from undisturbed fens, since there are no measured values for reflooded sites

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The design and performance of vertical flow and hybrid constructed wetland systems

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Abstract

The paper reviews the development of the Vertical Flow [VF] Reed Beds / Constructed Wetlands over the past 18 years. The performance of VF systems and their use within Hybrid Systems is analysed by reference to a number of short case studies.

Hybrid Systems have been shown to produce extremely high quality effluents. VF beds have been gradually improved over the past 15 years and have been shown to achieve nitrification as well as good BOD_5 removal. They can provide very attractive options for the treatment of the wastewater from single houses, small groups of houses or remote properties such as hotels or visitor centres. There has however always been a concern that the VF systems would suffer from surface clogging and hence flooding.

The importance of the selection of the gravel or sand to use as bed media is emphasised since it affects both the Hydraulic Loading Rate (HLR) and the Oxygen Transfer Rate (OTR) that is achievable from the design. The Oxygen Transfer Rate achieved is absolutely critical to the sizing of the systems. The author reviews the reported OTRs and comments on the existing equations proposed for calculation of the area of beds.

Older VF systems used a set of parallel beds that were dosed in rotation and then rested for a period of days because there was considerable concern [based on previous experience] that they would become clogged. In the past 10 years a number of new designs have been built which make use of a single bed. The Hydraulic Loading Rate and the selection of the bed media which are critical to the design and hence successful operation of these 2nd generation Compact VF beds are described. It is now possible to achieve full nitrification as well as BOD_5 and TSS removal in VF beds sized at $2m^2$ PE^{-1} or less when treating domestic sewage.

If the appropriate design additions are made then more than 75% removal of Total N is possible by using denitrification techniques to remove the NO_3N as well as the NH_4N. Where a sacrificial bed of the appropriate media is added into the flowsheet then substantial P removal should be possible.

It is now possible to obtain a very high quality of effluent from VF beds alone with typically <10 mg BOD_5 l^{-1}, <10 mg TSS l^{-1} and <2 mg NH_4N l^{-1}. As a result it seems likely that compact VF beds may take over some of the role previously played by Hybrid Systems.

Key words

BOD_5 removal, constructed wetlands, denitrification, hybrid systems, hydraulic loading rate, media size, nitrification, nitrogen removal, oxygen transfer rate, secondary treatment, sewage treatment, single house treatment, small systems, tertiary treatment, vertical flow, wastewater treatment.

Treatment performance of multistage wastewater constructed wetlands in Norway

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Introduction

The first constructed wetland for treatment of domestic wastewater was built in 1991. In order to enhance removal of both organic matter and nutrients, multistage systems with aerobic pre-treatment were built. Since then numerous systems have been built, and today the concept of a constructed wetland with aerobic pre-treatment is becoming a widespread method for wastewater treatment in rural areas. The reason for this is that the systems have proven high performance in cold climate and require low maintenance. The success of the system is based on adapting the design to the climatic conditions so local performance requirements are met. The paper describes the general design of the systems and discusses development, function and sizing of the various components. Finally, reuse of the filter media is considered.

Design and performance

The general concept (Figure 1) consists of pre-treatment of the wastewater in a septic tank, pumping to a vertical down-flow aerobic biofilter followed by a subsurface horizontal-flow porous media filter. The biofilter may be integrated or located separate from the horizontal flow section. The wetland section is usually vegetated with common reed (*Phragmites australis*). Evaluation of the role of plants in these systems, both in field and mesocosm scale systems, showed that the root-zone had a positive effect on N-removal, but no significant effect on P and BOD. Some of the later systems have therefore been built with grass over an insulating soil cover. The grass-covered systems do not fulfil the strict definition of a wetland, although the filter is water saturated. The systems built in Norway treating both traditional domestic sewage (black- and greywater) have a total surface area/person varying from 8–12 m^2 and according to present guidelines 8–10 m^2 is recommended. The depth of the horizontal subsurface flow constructed wetland part (HSF) in existing systems is between 0.8–1.2 m. The guidelines recommend a minimum of 1 m. The reason is the cold climate and the need to meet phosphorus discharge consent of 1 mg l^{-1} without frequent change of the P-saturated filter media. The final geometry (length, width, depth) of a system is based on hydraulic considerations. For systems treating greywater only the recommended surface area is 2–3 m^2/person. All systems in Norway are built with a pre-treatment filter. Some systems use sand in the horizontal flow section, but the majority of the systems use light weight aggregates (LWA) both in the pretreatment section and the horizontal flow section. The overall treatment performance for the systems (Table 1) generally exceeds 80% for BOD_7, 90% for phosphorus and varies from 40–60% for removal of total N. For the indicator bacteria, termotolerant coliform bacteria (TCB), the concentration in the effluent normally is below 1000/100ml and many systems consistently show <100/100ml (Table 1).

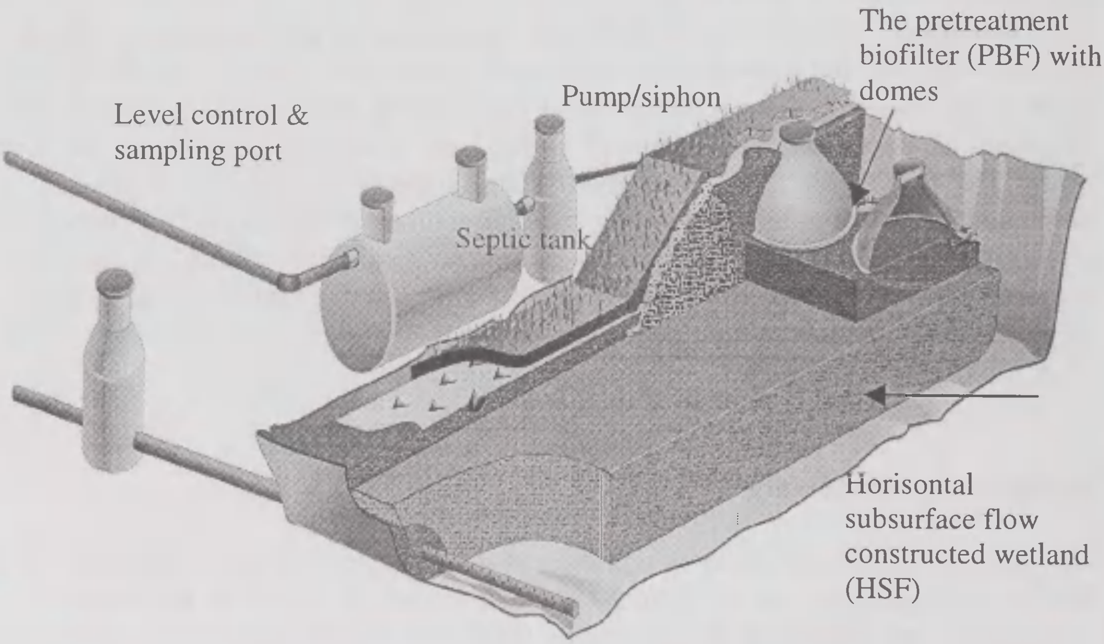

Figure 1 The last generation constructed wetlands for cold climate with integrated pre-treatment biofilter (PBF) in Norway.

Table 1 Removal efficiencies (%)[a] and outlet concentrations (mg l^{-1}) in 13 constructed wetland systems in Norway. Results include the pretreatment biofilters.

System (p) [b]	Syst.	Built	TP		TN		COD		BOD_7 [c]		TOC		TCB [d]
	no.	year	%	C_{out}	%	C_{out}	%	C_{out}	%	C_{out}	%	C_{out}	
Haugstein (7)	1	1991	97	0.3	64	40	75	52	80	15			<50
Tveter (7)	2	1993	96	0.4	41	49	69	123	84	21			<50
Østegården (8)	3	1993	93	0.6	79	23	41	143	90	22			<50
Fagernes (8)	4	1996	98	0.1	60	17	88	34					
Lilleng (60)	5	1997	95	0.1	53	25					90	9	<50
Bromølla (40)	6	1998	98	0.1	47	23			98	5			<10
Bogstad (56)	7	1999	98	0.05							80	22	<10
Holt farm (30)	8	1999	98	0.01							90	10	
Tyrili (50)	9	2000	97	0.3							70	38	
Dal skole (39)	10	2000	98	0.1	69	9	82	24					
Kaja* (48)	11	1997	89	0.1	72	2.5			93	5			<100
Torvetua* (140)	12	1998	79	0.19	60	2.2	82	62,0					
Klosterenga*(100)	13	2000		0.2		2.5		19,0					

*) Greywater systems
a) Based on arithmetic mean of concentrations, n=5-50 grab samples.
b) No. of persons (p) served in the parenthesis
c) 7-day BOD is standard in Norway
d) Termotolerant coliform bacteria, TCB per 100 ml.

The pre-treatment biofilter

The pre-treatment biofilter (Figure 1) has a standard depth of 60 cm and a grain size within the range 2–10 mm is recommended. The pre-treatment biofilter accounts for about 70% of the BOD and SS removal in full-scale systems (Table 1) in addition nitrification is achieved. The best performance regarding nitrification and BOD removal are achieved in systems with maximum hydraulic loading rate 20–30 cm d^{-1} and > 12 doses per day. Surprisingly these biofilters also removed 20–40% of the total N. This effect may be explained by denitrification in anoxic sites in the filter. Using 2–4 mm LWA, near complete nitrification of septic tank effluent (STE) could be achieved at loading rates up to 30 cm d^{-1}.

The pre-treatment biofilters have the ability to reduce the number of indicator bacteria by 2–3 logs or more. The keys to obtaining high purification performance in a single-pass coarse media biofilter is even distribution of the effluent over the filter surface, the number of doses per unit time, and the volume per dose. A system where a small pump fed spray nozzles suspended over the filter media (Figure 1) has been developed. Siphons or other dosing devices may also be used if the elevation difference between the dosing chamber and the pre-treatment filter is sufficient. A maximum loading rate of 30 cm d^{-1} is suggested in the Norwegian guidelines. However, new findings indicate that higher loading rates may be used especially for greywater.

The horizontal subsurface flow section (HSF)

The main purpose of the horizontal subsurface wetland (HSF) in the Norwegian systems has been to reduce phosphorus. In addition, sites for denitrification are provided and the hygienic quality of the effluent is improved. In vegetated systems evapotranspiration reduces and even eliminates the discharge in dry summer periods. The volume of the HSF has been based on estimates of the P-sorption capacity and the geometry on hydraulic properties of the porous media. P-sorption capacity is measured by batch experiments in the laboratory. However, the correlation of the sorption values derived from simple batch experiments using phosphate solution to sorption in full-scale systems receiving wastewater is difficult. Several authors have found that soils below wastewater infiltration systems sorb more phosphorus than the adsorption maximum based on batch experiments indicate. This has not been documented for wetland systems. For man made media as LWA it has ben suggested that the actual P-sorption capacity in full scale subsurface-flow constructed wetland systems is lower than the P-sorption capacity measured in batch experiments. Despite several attempts to predict the long term P-sorption capacity of CWs no universal method exists.

The natural material with the highest potential sorption capacity tested in Norway is the shellsand.

Shellsand is a natural carbonatic material consisting mainly of $CaCO_3$ and $MgCO_3$ and has a P-sorption capacity of 17g P kg^{-1}. Iron rich sand, which is used in many systems, has a sorption capacity of more than 1g P kg^{-1}. The highest measured P-sorption in LWA material is 12 g kg^{-1} (Norwegian Filtralite-P).

In Norway the sizing of constructed wetlands with respect to P-removal is based on sorbing 90% of the P-emitted over a 15-year period. Using the maximum sorption values from the batch experiments, the amount required of shellsand and Filtralite-P would be 0,53 and 1,25 m^3 per person, respectively. One person emits 0.6 kg P yr^{-1} and the bulk density of shellsand is 1.0 and FiltraliteP 0.6. However, the sorption in full scale systems have not yet reached the values measured in batch experiments. The sizing of HSF´s according to P-sorption therefore involves conservative estimates of the P-sorption capacity. When using iron rich sand, shell sand, and LWA 7–9 m^3/person is recommended in the Norwegian guidelines.

All systems where indicator bacteria are investigated meet the European standard for swimming water quality of < 1000 TCB 100 ml^{-1} (Table 1). It is not yet clear what is the main mechanism for bacteria removal. In systems using the Filtralite-P high pH will contribute to the bacteria dieoff the first years. However, systems where the pH is near neutral (7–8) also show excellent bacteria removal (system 1,2,3 and 11 in Table 1). System 1 and 3 both contain a sand section that have pore sizes small enough to suggest straining as a removal mechanism. The grain size of the porous media in system 2 and 11 is to large (1–4 mm LWA) to explain the bacteria removal as straining. It is known that the bacteria removal may be positively correlated to P-removal. The media used in the Norwegian CWs all have high P-removal capacity. Plants may have an effect on bacteria removal by improving the environment for ciliated protozoa, which favour feeding on *Escherichia coli*. For system 11, which have a LWA filter media with low P-sorption, the bacteria removal has improved over the first 3 years. This correlates to the root and plant development.

Seasonal variation

Table 1 shows the overall average treatment performance. Although the influent COD and total-N concentrations show a great variation the constructed wetland system produces a consistent effluent of low variation. The seasonal variation is also small and there are no significant differences between cold and warm seasons. Similar results are found for other cold climate constructed wetlands with a pretreatment biofilter. The phosphorus effluent concentration has also been consistently low and that there is no significant seasonal variation.

Reuse

When saturated with phosphorus, the porous media can be used as fertiliser. Investigations have shown that accumulated P in Filtralite-P has the same growth effect as P in mineral fertilisers. However, if heavy metals from wastewater are sorbed to the filter media agricultural reuse may be limited. Measurements of the heavy metal content in two constructed wetlands (sand/LWA and LWA) treating domestic wastewater for 8 and 7 years respectively, have been performed. The sand accumulated all of the investigated heavy metals except for Cd. The LWA filters accumulated Zn, Ni, Cr, Co and V. Both for the sand and the LWA filters the increase in heavy metals was small compared to the initial content in the filter media. This indicates that the sorption capacity is low. There was no systematic enrichment gradient through the filters (vertical or horizontal), which indicates that the sorption capacity was near saturation.

The heavy metal concentrations in the wetlands' filter media were low compared to the Norwegian Standards for allowable maximum levels of sewage sludge applied to agriculture. These results indicate that heavy metal content will not limit the reuse of the filter media as fertiliser.

Conclusion

Constructed wetlands with pretreatment biofilters using porous media with high phosphorus sorption capacity have consistently removed >90% of the phosphorus for more than 10 years. The nitrogen removal is in the range of 40–60% and the effluent meets European standards for swimming water quality with respect to indicator bacteria. Constructed wetlands with pretreatment biofilters produce an effluent quality that is independent of season. Pretreatment biofilters that nitrify and reduce BOD are a necessary component of cold climate constructed wetlands. Better prediction of the phosphorus removal and further optimising of the pretreatment biofilter will yield more compact systems.

Nutrient removal in treatment wetlands

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Treatment wetlands address many pollutants, including nutrients, comprised of nitrogen and phosphorus compounds. The targets for small communities are often biochemical oxygen demand (BOD) and total suspended solids (TSS), but increasing concerns for the eutrophication of receiving surface waters have more recently expanded treatment goals to include nutrient removal. Several variants of wetland systems are in use, including surface and subsurface flow, and a wide variety of substrate and vegetation choices. Several types of wastewater are of concern. In addition to domestic wastewater treatment, animal wastewaters, landfill leachate and some industrial effluents have emerged as nutrient discharges that contribute to surface and ground water pollution. Diffuse pollution from agricultural and urban sources contributes higher volumes at lesser concentrations.

Traditional treatment plants can provide nutrient removal, but at considerable cost. Nitrification is the easiest process to add to secondary treatment facilities, but denitrification is very expensive. Phosphorus removal requires either additional works for biological removal, or great cost for chemicals if precipitation is used. Further, traditional treatment plants are difficult to operate, which places them out of reach of small communities. Conventional municipal facilities are nearly impossible to implement for non-point sources, such as urban and agricultural runoff. As a consequence, either passive or managed treatment wetlands achieve prominent consideration by a process of elimination.

Nitrogen removal

Water quality improvement in constructed wetlands typically includes some measure of nitrogen conversion and removal. The several nitrogenous chemical species are inter-related by a reaction sequence, and biological transformations from inorganic to organic compounds and back from organic to inorganic. Important processes that transform nitrogen from one form to another include: (1) ammonification (mineralization); (2) nitrification; (3) denitrification; (4) plant and microbial uptake, (5) sorption of soluble nitrogen on substrates. Ammonia volatilization and nitrogen fixation are of lesser importance.

Water entering a treatment wetland may contain differing proportions of various nitrogen species: oxidized (nitrate plus nitrite, NO_3-N + NO_2-N = NO_x-N), ammonia (NH_4-N), and organic nitrogen (ON). Combinations are: total Kjeldahl nitrogen (TKN = NH_4-N + ON) and total nitrogen (TN = TKN + NO_x-N). These may undergo a sequential conversion process: mineralization, in which ON is converted to NH_4-N; nitrification, in which NH_4-N is converted to NO_x-N; and denitrification, in which NO_x-N is converted to nitrous oxide and dinitrogen gases. The transfers are all microbially mediated, and are commonly assumed to be the dominant processes.

Nitrification requires oxygen, or other equivalent electron acceptors. The transfer of this limiting reactant restricts the rate of nitrification in most wetland configurations and types. Denitrification requires a carbon supply, which may be insufficient in some wetland types.

Several wetland modifications enhance N removal. Vertical flow treatment wetlands (vegetated intermittent sand filters) provide extra oxygen for nitrification on a cyclic basis. Paired SSF systems that are filled and drained on a recurrent basis are termed reciprocating wetlands, which can thus deal with uniform flow. Bottom aeration of subsurface horizontal flow wetlands offers the opportunity to enhance oxygen transfer to the root zone.

Phosphorus removal

Phosphorus movement in treatment wetlands is influenced by hydrologic, soil and biotic processes. Surface water movement is the basis for advective transport into, through and out of FWS ecosystems. However, there is often the potential for vertical flows into and out of shallow groundwater beneath the wetland. Within the wetland, sheet flow is typical; but vegetative zonation and topography nearly always create preferential channels. Particulate forms may settle and become trapped in the litter and floc layers on the wetland bottom. Sorption provides removal until soils become saturated. Biological processes at several scales utilize and convert phosphorus. Microorganisms and algae can rapidly incorporate available P into their tissues. Residuals from the biogeochemical pathways form new sediments and soils in the wetland. These accretions are the long-term sink for P in the FWS wetland. However, this accretion is relatively inefficient, thus restricting FWS systems to low loadings. Accordingly, FWS wetlands are appropriate for P removal from tertiary effluents, or runoff.

SSF systems often target BOD and TSS in strong effluents, and are cones-quently the recipients of high phosphorus loadings. Accordingly, P removal in these wetlands is dominated by sorption phenomena, and the frequency at which that sorption capacity must be restored. Designs focus on substrate selection and management, because without such attention, the SSF wetland can remove P for only a short portion of its early life.

Phosphorus sorption and desorption control the early phase of operation for both types of wetlands. However, FWS systems are constructed on soils that reflect a past history of phosphorus conditioning. Wetlands built on agricultural soils may have to deal with large stores of soil P that may temporarily result in high concentrations in floodwaters. Concentrations start out high and reduce to a long-term stable condition, a process termed stabilization.

Biogeochemical cycles

Plants utilize phosphorus, nitrate and ammonium, and decomposition processes release nitrogen and phosphorus back to the water. The two easily assimilated forms of nitrogen are ammonium and nitrate nitrogen. For nitrogen, 90% of animal waste wetlands in North America are loaded at more than three times the agricultural agronomic rate, and 77% of FWS municipal wetlands are loaded at less than the agricultural agronomic rate. For phosphorus, 100% of animal waste wetlands are loaded at more than 20 times the agricultural agronomic rate, and 30% of FWS municipal wetlands are loaded at less than that rate. In lightly loaded systems, vegetation lays claim to a significant share of the available ammonia during the growing season. Rate constants are high in spring, during the growing season, and drop to low values in fall, despite the fact that spring and fall temperatures are roughly the same. This seasonal cycle renders a modified Arrhenius (theta factor) calibration ineffective. In heavily loaded systems, the preponderance of ammonia removal is due to microbial processes, and is primarily temperature-driven. Under these conditions, the theta-factor temperature correction applies.

Modelling

Wetland data often support only simple design models, such as a first order, areal net uptake of phosphorus (Kadlec and Knight, 1996), characterized by the first order constant "k". This rate is combined with the appropriate water mass balance and internal mixing model to compute the performance of the entire wetland. Internal hydraulics may often be represented by N tanks-in-series model. The end result is a first generation formula relating inlet (Ci) and outlet (Co) concentrations to detention time (τ) and depth (h):

$$\frac{C_o}{C_i} = \left(1 + \frac{k\tau}{hN}\right)^{-N} \tag{1}$$

Many wetland variables can affect k-values, including vegetation type and density, substrate P binding capacity and its extent of saturation, temperature and season. Annual performance averages remove the effects of seasonal cycles, but other sources of variability remain. Those are illustrated for some large sets of FWS wetlands (Figures 1, 2, 3), for ammonia, nitrate and phosphorus, respectively.

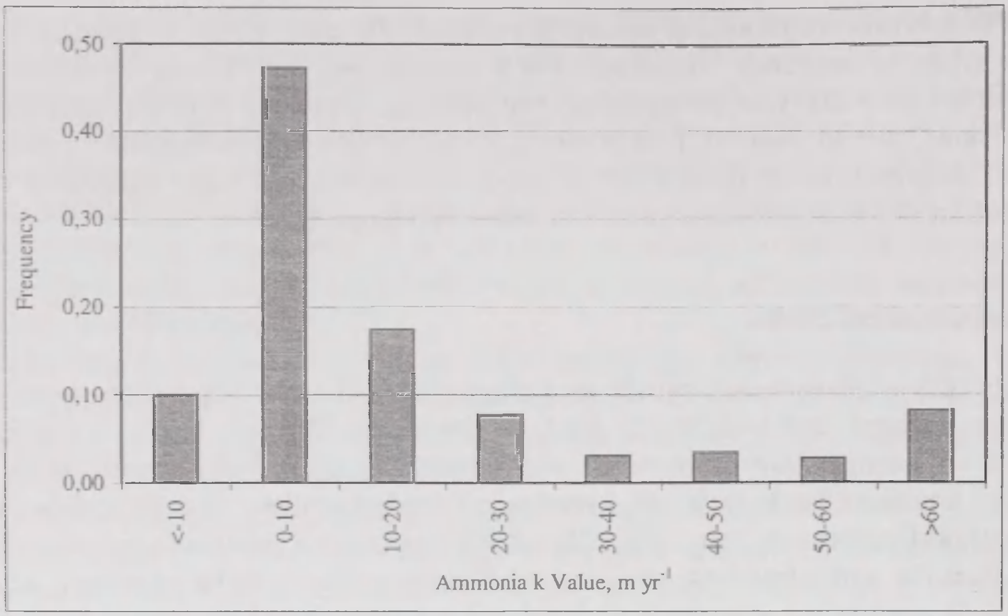

Figure 1. First order apparent rate constants for ammonia removal in FWS wetlands. These are annual averages, with one datapoint for each wetland for each year. There are 174 wetlands, 351 wetland-years. Median = 8.0.

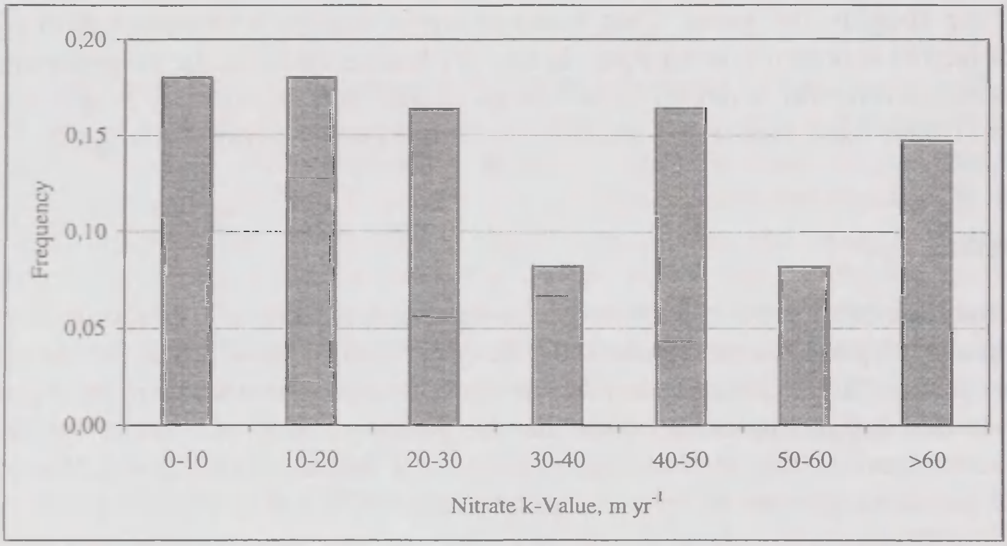

Figure 2. First order apparent rate constants for nitrate removal in FWS wetlands. These are annual averages, with one datapoint for each wetland. There are 61 wetlands. Median = 29 m yr^{-1}.

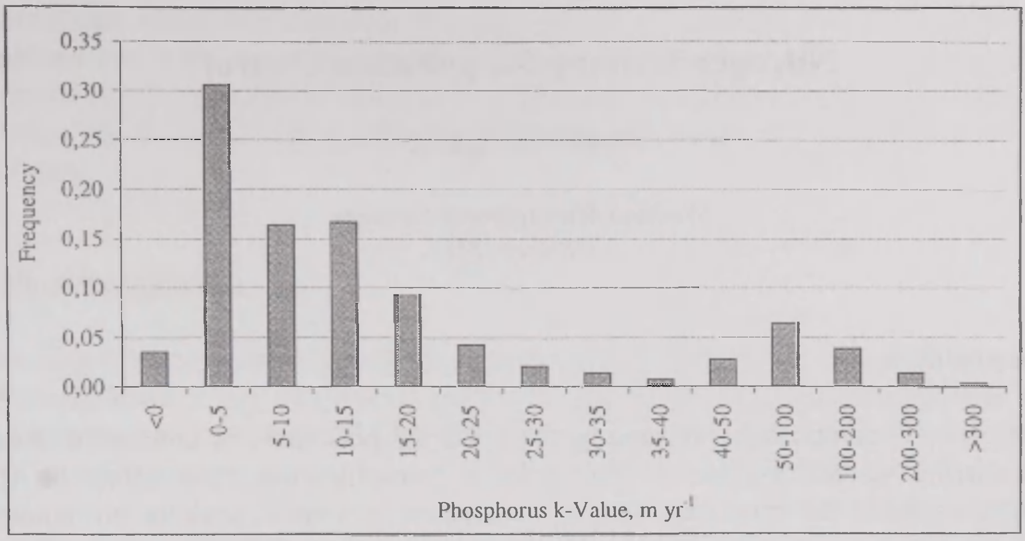

Figure 3. First order apparent rate constants for phosphorus removal in FWS wetlands. These are annual averages, with one datapoint for each wetland. There are 281 wetlands. Median = 10.0 m yr^{-1}.

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Nitrogen farming for pollution control

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The problem

Nitrogen (N) compounds are among the principal pollutants of concern in fresh and marine waters because of their role in eutrophication, their effect on the oxygen content of receiving waters, and their potential toxicity to aquatic invertebrate and vertebrate species. The nitrogen content of the streams and rivers of the Midwestern United States is of particular importance at this point in history, because of hypoxia in the Gulf of Mexico, together with the associated ecological and economic consequences (Dias and Solow, 1999). The size of the "dead zone" in the Gulf ranges up to 20,000 km^2.

Both point and non-point sources contribute to the nitrogen content of waters within the Mississippi River drainage basin. About 60% of the water-borne total nitrogen is in the form of nitrate (Goolsby and Battaglin, 2000). The source of the nitrogen is about two-thirds from agriculture, and one-third from other sources, including urban runoff, atmospheric deposition, and point sources. Numerous municipal and industrial wastewater treatment plants discharge into waters of the basin, typically as a wholly or partially nitrified effluent, which comprise about 10% of the total N load. The result is a surface water nitrate concentration, in the upper portions of the basin, of about four mg N l^{-1}.

Federal and State officials have agreed on an action plan that includes a component intended to promote restoration and enhancement of natural systems for nitrogen retention and denitrification (USEPA, 2001). The premiere natural system that has the capability to effectively remove nitrate from surface water is the free water surface wetland, based simply on its ability to place contaminated water in intimate contact with the biogeochemical cycle that removes N. More than half of the pre-settlement wetlands in the upper Mississippi basin have been lost to drainage (Dahl, 1990). It is therefore synergistic to restore wetlands that are positioned to effectively function for nitrate reduction. The goal of the Action Plan is to encourage actions that are voluntary, practical and cost effective (USEPA, 2001). Given that lands positioned properly to aid in N reduction are mostly in private ownership, and currently utilized for agriculture or other private endeavors, it is clear that reallocation of use must be an attractive alternative for those private landowners. Accordingly, a further goal of this paper is to set forth preliminary economic concepts that can satisfy the goal of the Action Plan. In combination with the selection of the most logical ecosystem (wetlands), this redirection of agricultural lands to N reduction is herein referred to as nitrogen farming (N Farming) (Hey, 2002), and is taken to mean the reduction of nitrate nitrogen.

Part of the solution

The idea of treatment wetlands for runoff water quality improvement is not new. Wetlands form a key element in the US Department of Agriculture nutrient and sediment control system (NSCS) (DuPoldt *et al.*, 1991). The Des Plaines River Wetland Demonstration project documented the use of restored and constructed wetlands, over several years, for river water quality improvement (Sanville and Mitsch, 1994). Over 40,000 acres of wetlands have been, or are being, built in south central Florida for control of agricultural phosphorus runoff (SFWMD, 2003). Nevertheless, real and perceived issues of scale-up, site specificity and technology transfer remain. This paper therefore examines the conditions and settings of in-basin demonstration projects.

Fortuitously, a large technical database on wetland performance has been accumulated in the broader context of the use of treatment wetlands (Kadlec and Knight, 1996). In combination with numerous recent detailed studies of wetland biogeochemistry and ecology, a firm basis for the design of demonstration projects is available. Marshes are effective for denitrification, with first order areal annual rate constants centered on 34 m yr^{-1}. Performance improves at higher water temperatures, with a modified Arrhenius temperature factor of 1.090. Performance also increases with increasing hydraulic efficiency, created by prevention of short-circuiting, and reflected in values of the tanks-in-series parameter $N>5$. Higher efficiencies are associated with submergent and emergent soft tissue vegetation, and lower efficiencies with unvegetated open water and with forested wetlands. Hydraulic loadings of 2–7 cm d^{-1} can produce 30% nitrate load reductions, over the temperature range 6–20 °C. Carbon availability limits denitrification at high nitrate loadings, however, wetlands produce carbon in sufficient quantities to support the loads anticipated in the upper Midwest. The conversion of agricultural lands to treatment wetlands focused on nitrate reduction is termed nitrogen (N) farming (Hey, 2002).

A key feature of treatment wetlands is the ability to manage the system for either concentration reduction or for mass removal, but one at the expense of the other (Trepel and Palmeri, 2002). The design models assume a direct proportionality: doubling the concentration doubles the removal rate. As a result, removal rates decrease as water passes through the treatment wetland, and nitrate concentrations are reduced (Figure 1). However, the actual mass of nitrate-nitrogen that is removed increases with increasing hydraulic loading. Thus increasing hydraulic loads result in more tons removed, but at the expense of a lesser concentration reduction.

The loading and detention required to achieve the USEPA 30% goal serve to set the wetland area needed. As a benchmark reference example, consider the task of treating 1% of the Illinois River, which is approximately 8.89 $m^3 s^{-1}$ originating from 74,900 hectares (Hey, 2002). On average, 70% of the nitrate will remain. Presuming a rate constant of 35 m yr^{-1} (corresponding to T=20°C) and N=4, the allowable hydraulic loading is 6.8 cm d^{-1}. The model calculation shows a required area of 1,125 ha (2,778 acres), which is 1.5% of the contributing watershed. However, the mean annual temperature is lower than 20°C, and flows arrive predominantly in the spring and fall cool seasons.

Demonstration project considerations

One or more treatment wetland projects are needed within the Mississippi drainage basin, to demonstrate the technology and further illuminate design and operating issues. One can argue this need to be large-impact projects, with a view to addressing scale-up from the mostly small prototypes discussed in this paper. A decade of experience in south Florida, with building and operating several wetlands of more than a thousand ha each (range: 600–14,000 acres), relieves a new project of most of the concerns of physical scale-up (SFWMD, 2003). Accordingly, the demonstration may be configured at a scale that deals with a significant portion of the sub-basin in which it is located. Although there is no precise size for a prototype demonstration, a target of about 1000 ha seems reasonable.

Ideally, this demonstration N Farm would receive and discharge water by gravity. Water availability should support hydraulic loading of the wetland at 2–10 cm d^{-1}, which translates to about 2–10 $m^3 s^{-1}$. It would be sited to access waters with relatively high nitrate concentrations, and have the capability of selectively treating high nitrate episodes. Appropriate steps should be taken to ensure effective use of the wetland area, including internal compartmentalization and flow distribution.

The success of N farming hinges on social and economic factors as well as the technology itself. It is here presumed that treatment wetland technology is demonstrated to be effective in some non-trivial subset of the overall nitrate pollution problem of the Mississippi basin. Many tens of thousands of acres of land are be reconfigured and placed in nitrogen farming. That land is likely to be predominantly in private ownership. The owners are putatively unwilling to have their lands pass into government ownership. Therefore, most N farm projects are likely to be in private ownership. Public (government) projects may also be logical in some circumstances, but those will not be discussed further here. It is necessary to define economic and regulatory terms of reference that allow N farming to go forth in the private sector.

Economics must be such that it is attractive for a landowner to go into the business of N farming. The component pieces of the economic structure are: (a) land use charges, (b) capital construction costs, (c) debt service, (d) operations, maintenance and monitoring costs, (e) income from sale of products, and (f) incentives for optimal operation. This structure is not far different from that associated with agriculture. Property and equipment must be purchased, usually with attendant finance charges. There are labor and material charges involved in raising and harvesting the crop. The net profit forms the income stream of the owner. If yields can be increased through optimization, that enhances the income stream.

In addition to economics, there are significant issues related to regulation and land use. It is commonly the case that discharges through structures to waters of the USA are subject to permit requirements. Permits that contain fixed concentration limits for nitrogen or any other waterborne constituent would place the N farm operator at risk that would likely be unacceptable. There are, however, generally accepted alternatives, such as technology-based standards that could circumvent that difficulty.

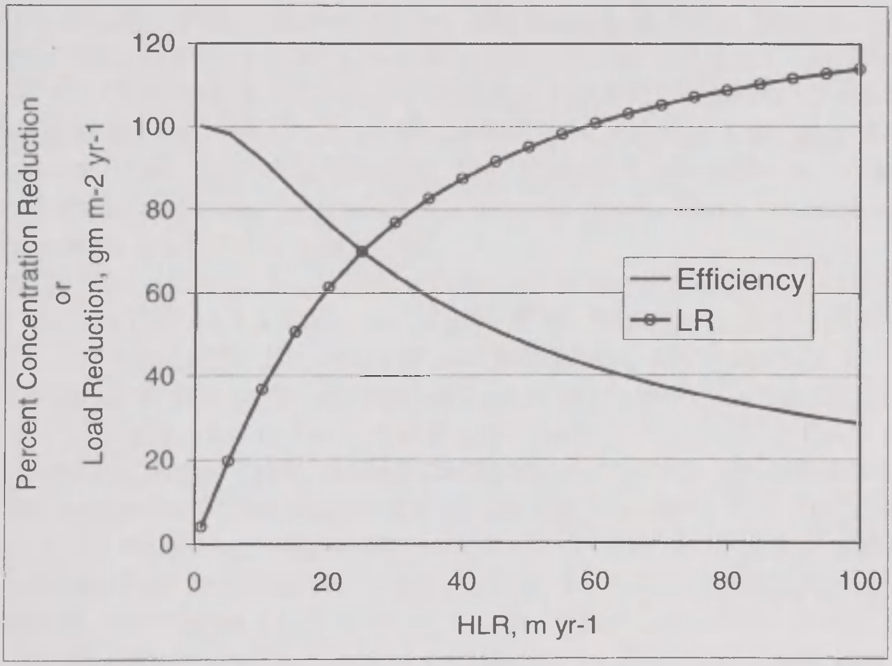

Figure 1. Concentration reduction and load reduction as a function of hydraulic load for a hypothetical nitrate treatment wetland. Parameters: $C_i = 10$ mg l^{-1}, $k = 35$ m yr^{-1}, $N = 4$.

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Efficiency and optimisation of nutrient elimination in wastewater lagoons with regard to river catchment pollution

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Introduction

Wastewater lagoons are one of the oldest and most traditional forms of wastewater treatment; they are a widespread method of wastewater treatment all over the world. As a widely employed method, they make use of biological self-cleaning processes. Due to their comparatively high space demand, these treatment methods are predominantly applied in rural areas. In the past, they were often regarded as temporary solutions but lately scientists are in agreement that these cost-efficient, low-maintenance and rather effective methods are a veritable alternative to the classical "technical" methods in many areas of application. Being a treatment method with a high retention time, lagoon plants achieve stable purification results. The efficient carbon degradation in these systems has been documented extensively. In Germany, effluent values of <25 mg l^{-1} for BOD_5 and <90 mg l^{-1} for COD can be observed during the treatment of domestic wastewater in lagoons (Bucksteeg, 1987). This corresponds to an average efficiency degree of approximately 90%. As the nitrogen and phosphorous effluent values are commonly no legal monitoring parameter in small plants, there are relatively few detailed analyses available in this regard.

Data of the average N_{total}-degradation in unaerated lagoon plants varies between 70% and 80% (Racauld *et al.*, 1995: 67%; Schleypen, 1987: 75%; Garcia *et al.*, 2000: approx. 70%; Barjenbruch and Brockhaus, 2001: approx. 80%). One has to bear in mind that in the effluent of lagoon plants nitrogen occurs mainly as ammonium nitrogen, due to the limited nitrification. Therefore, NH_4-N effluent values exceeding 20mg l^{-1} are frequent (Schleypen and Wolf, 1983; Racauld et al., 1995). As ammonium discharges are a crucial parameter for the ecological situation of the receiving waters, lagoon plants should be equipped particularly with regard to their nitrification. In regard to the phosphorous-elimination in lagoon plants, Schleypen (1987) states for P_{total} for maturation ponds (lagoon plants with very low pollution loads) an efficiency of 63% in summer and 19% in winter. Racauld et al. (1995) report an average P_{total} retention of 67% in lagoon plants.

Based on the examination of 2 lagoon plants in Lower Saxony, this paper will document the nitrogen and phosphorous elimination in relation to the lagoon surface area. Furthermore, the N and P discharges from lagoon plants into the environment will be estimated and different arrangements to reduce these discharges will be discussed. In order to evaluate the relevance of these nutrient loads for the receiving waters, they will be related to the calculated discharges of other surfaces (e.g. instance agriculture/pastures) of the considered catchment areas.

Lagoon plants examined and methods used

Wastewater lagoon "Gross Mahner"

The combined wastewater of the village of Groß Mahner (700 PE) is treated in a lagoon plant with five ponds (Figure 1). The ponds are built in a row and integrated nature-based into the rural environment. Their overall surface area is 10.440 m² and hence corresponds to a dimensioning value of 15 m² PE^{-1} which is the current state-of-the-art for lagoon ponds for combined wastewater treatment. The average pond depth is 1,20 m. The combined wastewater is fed directly into the lagoon plant, without any retention of solid matter by mechanical pre-treatment (screen, sieve, etc.). Only the influent area is separated by a scurnboard, thus forming a separate settling zone. The purified wastewater is discharged via an approximately 1,5 km long ditch into the river Warne.

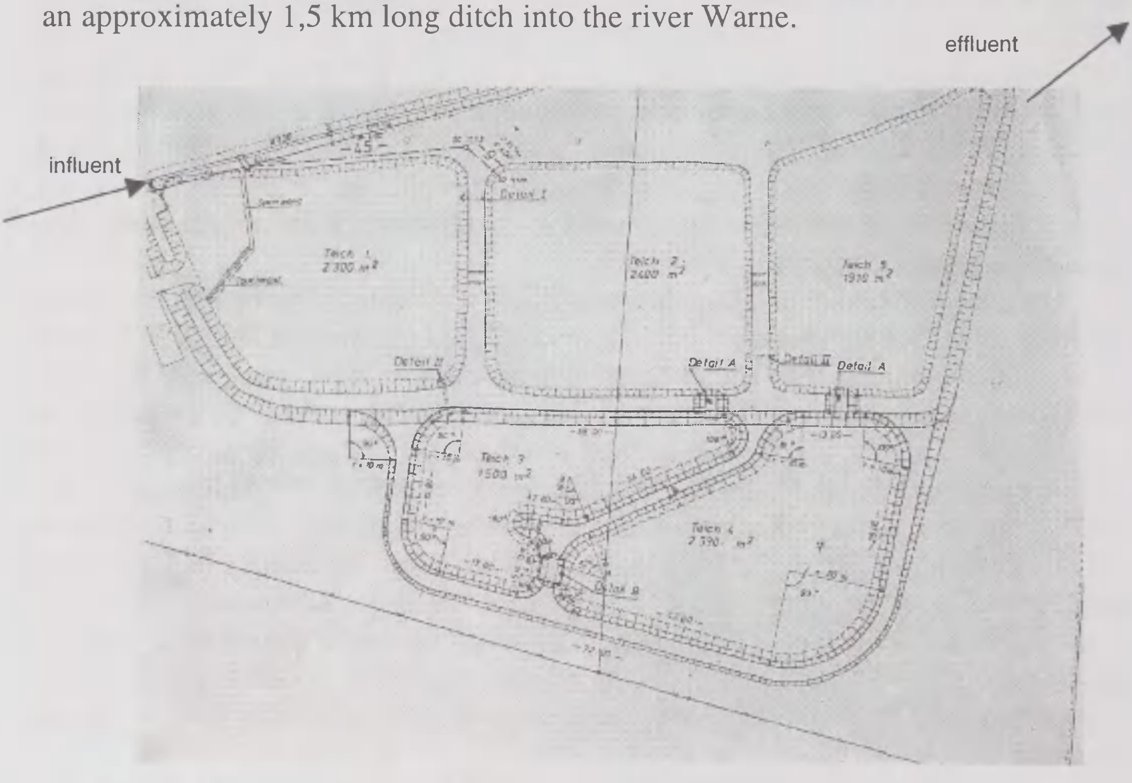

Figure 1. Wastewater lagoon "Gross Mahner".

Wastewater treatment plant "Ettenbuettel"

For decades, the combined wastewater in the village of Ettenbuettel (1.000 PE) had been treated in non-aerated wastewater lagoons. The plant consists of three ponds with an overall surface area of approx. 6.000 m²; the average depth is 1,20 m. Originally, the ponds had been dimensioned for 10 m² PE^{-1}; yet due to population growth, the purification performance of the plant decreased more and more. The ammonium effluent values increased to approx. 15 mg l^{-1} NH_4-N (with peak values of up to 25 mg l^{-1}). This meant that in the middle of the 1990s the plant had to be redeveloped. Because vertical-flow reed beds were known as very efficient regarding nitrification (Laber *et al.*, 1997; Cooper *et al.*, 1997; Platzer, 1998), the existing old lagoon plant "Ettenbuettel" was combined with a vertical-flow reed bed.

Figure 2 shows the general design of the combined plant at "Ettenbuettel." The first stage is the settlement pond, the second step is a facultative lagoon. The effluent of this lagoon is conveyed by a pump into the vertical-flow reed bed (VF), which is divided into two parts of equal size.

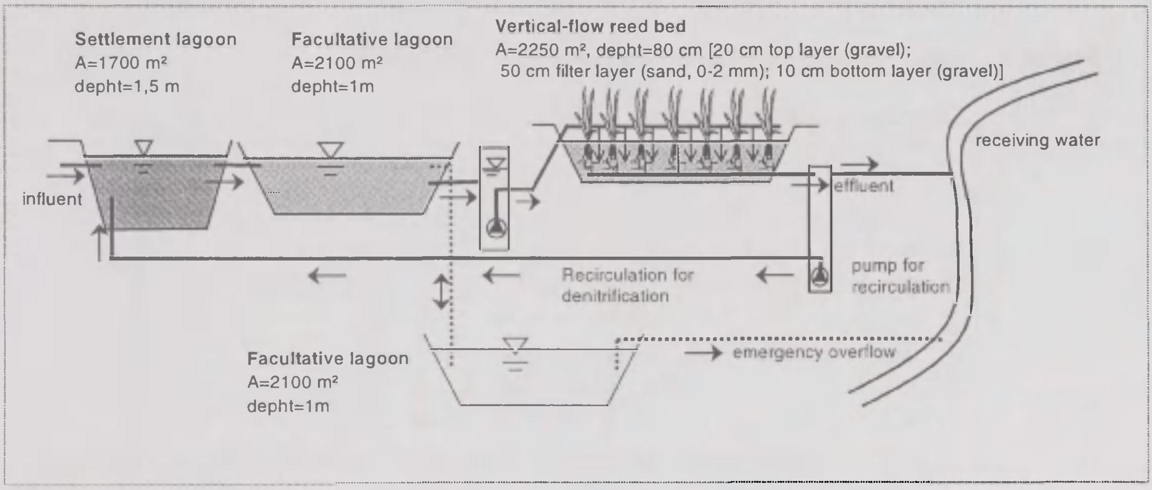

Figure 2. Design of the combined treatment plant "Ettenbuettel".

The size of the reed bed was designed for 2.25 m^2 PE^{-1}, with the filter being fed intermittently. The treated wastewater is either discharged from the VF into the receiving water or can be pumped back into the settlement pond. Apart from intermediate storage, the recirculation serves to achieve a better purification, particularly a better denitrification. A third lagoon provides a sufficient buffer volume for the necessary water-amount management. During the first 2 years of the filter's operation (10/99–10/01), a wide range of hydraulic loads were tested and the purification results documented.

Results and conclusions

Purification capacity of the lagoon plants

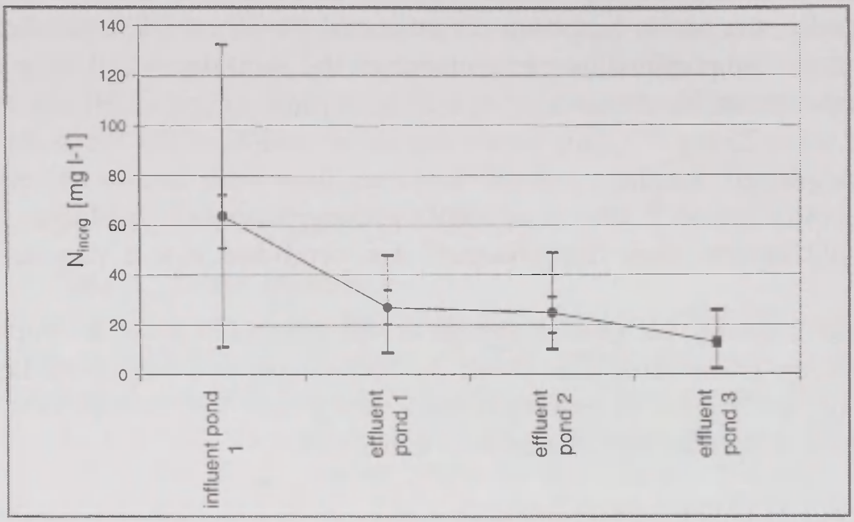

Figure 3a. Reduction of N_{inorg} in the lagoon systems of "Ettenbuettel".

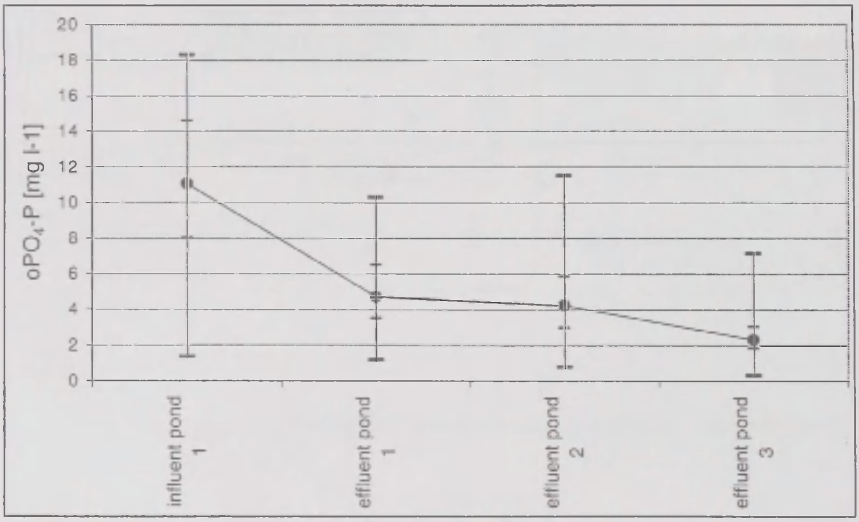

Figure 3b. Reduction of oPO_4-P in the lagoon systems of "Ettenbuettel".

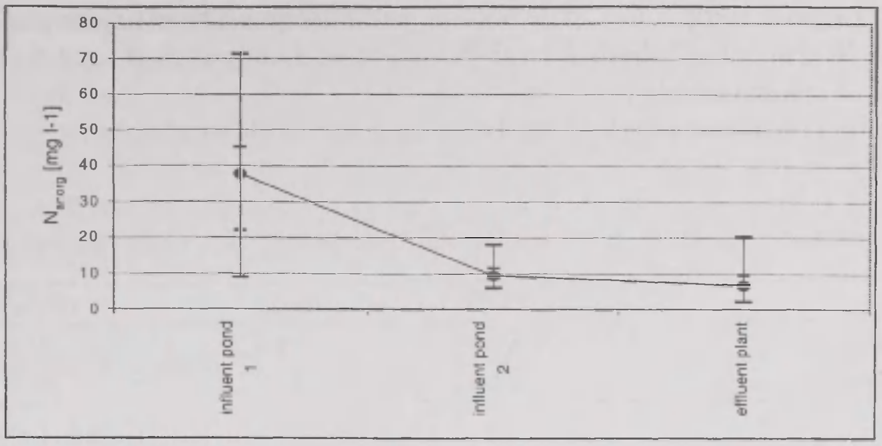

Figure 3c. Reduction of N_{inorg} in the lagoon systems of "Gross Mahner".

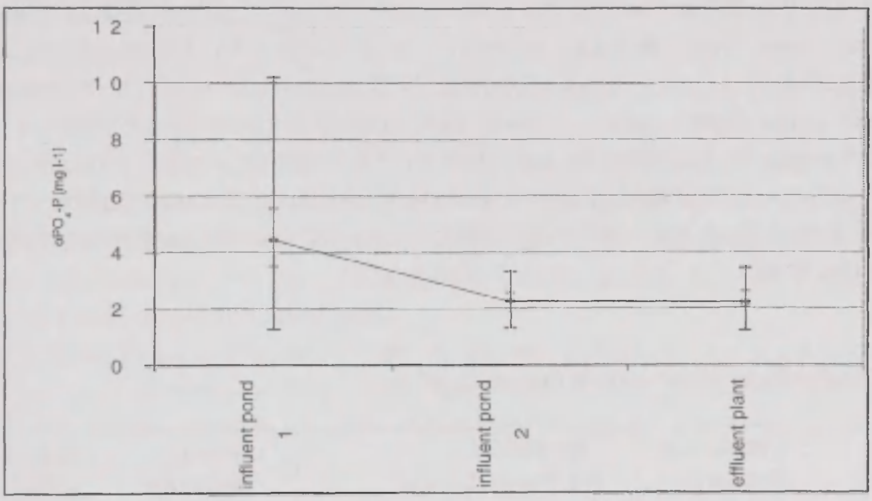

Figure 3d. Reduction of oPO_4-P in the lagoon systems of "Gross Mahner".

Figures 3a, 3b, 3c and 3d show the statistical analysis of the N_{inorg} and oPO_4-P concentrations in the two lagoon systems (max, 75%, 50%, 25%, and min). The results of the plant "Ettenbuettel" refer here exclusively to the old lagoon plant and thus to the state before the extension by the vertical-flow reed bed. At the time of writing this abstract, the results for N_{org} and P_{total} were not yet available in a sufficient amount. Therefore, at this stage N_{inorg} and oPO_4-P are used for the general estimation of the purification capacity. For a succeeding paper, these data will be added.

It becomes obvious that the first lagoon (specific surface 1.7 m² PE^{-1} in "Ettenbuettel" and 3,3 m² PE^{-1} in "Gross Mahner") alone has a high efficiency in reducing organic compounds, phosphorus, and nitrogen. The efficiency of ponds 2 and 3 in "Ettenbuettel" (both 2.1 m² PE^{-1}) and of Ponds 2, 3, 4, and 5 in "Gross Mahner" is comparatively low.

The total efficiency in % of the two plants is as follows (Table 1):

Table 1. The total efficiency of the treatment plants "Ettenbuettel" and "Gross Mahner" in %.

	COD	N_{inorg}	oPO_4-P
Ettenbuettel	89,4	79,6	79,1
Gross Mahner	94,6	82,2	58,5

Nutrient discharge in the examined areas

Table 2 shows the discharge of nutrients of the two examined areas into the receiving waters of the considered catchment areas. The discharge is divided into the effluent from "natural area" and into the effluent from the wastewater lagoon plants "Ettenbuettel" and "Gross Mahner." The estimation of the discharge from urban and agricultural areas, forests and pastures (here combined as "natural area") was made by the specific land-use in the examined areas. Empirical data of our work-group "Management of Catchment Areas – Quantification of Diffuse Sources," which had been recently determined for similar regions, were used as area-specific loads.

Table 2. The discharge of nutrients in the examined areas.

All data in [kg a^{-1}]		Theoretical discharge „natural area" (urban, agriculture, forest, pasture, etc.)	Theoretical discharge due to wastewater „without treatment" (inhabitant specific load, TKN=11 g PE^{-1}*d P =1,8g PE^{-1}*d)	*Efficiency*	Actual discharge due to wastewater „with existing lagoon plants"	Rate of the actual discharge of the theoretical discharge from the "natural area"
Etten-buettel	TKN	676,08	238,50	80%	47,70	7%
	P	16,74	39,00	80%	7,80	46,6%
Gross Mahner	TKN	2437,12	4496,80	82%	809,42	33,2%
	P	117,70	735,84	58%	309,05	262,6%

On the basis of the efficiency determined for the two wastewater lagoon plants, the actual nutrient discharge was calculated. It became apparent that in spite of the relatively high and stable efficiency of the lagoon plants there is still a considerable discharge of nutrients into the receiving waters. Further improvements could be achieved by optimisation measures, such as the topping of a sufficiently dimensioned vertical-flow reed bed for nitrification and the installation of a recirculation to the first pond for denitrification.

Options of optimisation

Nitrification and nitrogen elimination

One option for the optimisation of lagoon plants in regard to the nitrogen degradation was realised at the plant in “Ettenbuettel” by extending the plant with a vertical-flow reed bed (cf. Figure 2). The major target of this extension was the improvement of the nitrification, not that of the N_{total} elimination. By recirculating the nitrified discharge of the reed bed into Pond 1 (settlement lagoon), however, the option of an improved N_{total} elimination is given in this combined plant. The reed bed in “Ettenbuettel” is designed too small for a continuous recirculation, so that during the two-year examination period the recirculation was able to run only occasionally. From the achieved results it can be inferred that the denitrification potential of Pond 1 must be estimated as very high. Even at recirculation ratio >1, a complete nitrification of the recirculated nitrate could be achieved. Higher recirculation ratios could not be tested.

With a 80% N_{total} elimination rate in lagoon plants it can be stated that in combination with a vertical-flow reed bed with recirculation (RV=1) a N_{total} elimination of 90% should be achievable (the vertical filter should then be dimensioned for at least 3 m^2 PE^{-1}).

P elimination

An optimisation of the examined plants with regard to the reduction of the P discharge can only be achieved by using filters and through the use of the specific properties of the filter material. Lagoon plants alone cannot be optimised in this respect. Analyses by Rustige and Platzer (2002) principally show that the retention of phosphorous in reed beds is favoured by a high reaction volume and increased contents of iron and aluminium hydroxides and calcium in the filter material. They show also a very good retention capacity at the start of the operation which decreases as the operation time increases. However, reliable prognoses of the P retention performances are currently not possible; hence soil filters can presently not be dimensioned for continuous P retention.

If there are particular demands on the phosphate effluent concentration, the above-mentioned authors recommend building topped P retention filters. As filter medium, fresh used waterworks gravel and granulated blast furnace slags have proven their worth. When the filter is exhausted, i.e. when the filter material is loaded with phosphate, it can be removed and replaced by fresh gravel.

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Overview on hydrological studies and retention processes measurements at PRIMROSE study sites

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Abstract

The presentation will give an overview of the studies carried out in the European Community funded project called PRIMROSE (**PR**ocess Based **I**ntegrated **M**anagement of Constructed and **R**iverine Wetlands for **O**ptimal Control of Wastewater at Catchment **S**cal**E**). The focus of research was on re-use of soils saturated with P, hydrology and hydraulics, retention processes, and on modelling of wastewater retention. Also, an overview will be given on different types of pilot wetlands that were used in various studies within the framework of the PRIMROSE project. This overview will be presented using the PRIMROSE database on constructed wetlands.

The sites studied consisted of various types of ponds, subsurface flow (SSF) filters and peatlands treating wastewater mainly from municipal sources but also from agricultural runoff, landfills, and peat mining areas. The study sites were located in Norway, Sweden, Finland, Estonia and Poland. Laboratory studies on pond hydraulics were carried out in Austria. The soils from all sites studied were collected in first part of the project and analysed in Norway using different tests to determine agricultural re-use possibilities of these soils.

The hydraulics at the sites was studied using tracers and isotopes in various ways. Break Through Curves (BTC) were determined from various sites. This method seemed somewhat difficult to apply at some sites. Main reasons are: (1) difficulties in measuring a proper water balance, (2) a very long residence times, and (3) formation of density currents. A method was developed to test when density currents occur and to obtain an estimation of maximum input concentration of salts. New methods including isotopes and tracer measurements in groundwater were used in Finland to determine flow depths in peat-based systems.

Several processes occur in wetlands which effect wastewater removal. The phosphorus removal properties of soils were studied by measuring P uptake in batch tests to obtain P-adsorption isotherms and to compare these isotherms to soil

chemical properties. P-transport was also studied in column experiments and in large laboratory tanks. Nitrogen processes were studied by measuring different components of nitrogen flows including gas emission measurements using closed chamber method (for N_2O and CH_4 fluxes) and He-O method (for N_2 fluxes). Removal of pathogenic bacteria was measured using normal indicator bacteria e.g. *Escherichia coli.*

The modelling of the wastewater transport and different ways of retention was carried out in all countries. The approaches included groundwater modelling techniques and Artificial Neural Networks (ANN) as well as analytical approaches. The models were used in connection with tracer test e.g. to obtain the distribution of flow and P in wetland filters.

Temperature, plants, and oxygen: how does season affect constructed wetland performance?

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Abstract

The influence of temperature and plant-mediated oxygen transfer continues to draw attention from researchers, practitioners and regulators interested in the use of constructed wetlands (CWs) for wastewater treatment. Because the vast majority of research on constructed wetland performance has been conducted during periods of active plant growth, the true influence of temperature, season, and plant species selection on CW performance has not yet been evaluated adequately. In this paper, we briefly summarize changes in the understanding of these influences on wetland performance, and suggest that temperature and oxygen transfer are not separable in that both factors respond to seasonal cycles. We further speculate that the net effect of seasonal variation in these factors is such that plant-mediated oxygen transfer affects water treatment most in winter. Results of controlled-environment experiments conducted at Montana State University support these perspectives. Different plant species' capacities to oxidize the root zone responded differently to seasonal cycles of growth and dormancy, and species' effects on wastewater treatment were most pronounced in winter.

Introduction

Many early reports and design manuals assumed a positive relationship between temperature and carbon and nitrogen removal efficacy in constructed wetlands and some data support this supposition (detailed references can be found in Allen *et. al.*, 2002; Riley *et. al.*, 2003; Borden *et. al.*, 2001; Kowles, 2001; Hook *et. al.*, 2003; Stein *et. al.*, 2002). This reasoning was based on the well-documented fact that microbial growth rates and rates of treatment processes assayed *in vitro* decrease sharply with decreasing temperature. However, results of many other studies suggest that CW performance does not respond to temperature as expected. Reviewing available data, Kadlec and Knight (1996) found there was little, if any, influence of temperature on overall CW performance, and Kadlec and Reddy (2001) concluded that if any relation exists, it is negative: *i.e.* performance decreases as temperature increases.

The degree to which plant-mediated oxygen transport influences processes in the microbially active root zone of subsurface flow wetlands has been debated extensively. Assuming aerobic metabolic pathways were responsible for observed organic carbon removal, several earlier reports estimated high rates of oxygen transport, fostering a general view that wetland plants play a strong, positive role in wastewater treatment. However, further research indicated poor oxidation of reduced nitrogen compounds, evidence of the importance of alternative anaerobic metabolic pathways, and a largely anaerobic root zone. Therefore, recent publiccations have downplayed the importance of wetland plants to oxygen transport and wastewater treatment, relegating their benefits primarily to their ornamental, wildlife habitat, and thermal insulation value.

We believe that the physiological response of some plant species to seasonal dormancy and lower temperature permits increased oxygen transfer to the root zone of subsurface constructed wetlands, while the potential for plants to enhance treatment is more limited during periods of active plant growth and higher temperature. Assuming oxygen availability frequently limits the rates of microbial processes in wetlands, winter-elevated oxygen availability could offset the reduction of microbial activity due to cold temperatures. Below we summarize research conducted with co-workers at Montana State University indicating that plant effects on root zone oxidation and wastewater treatment vary both seasonally and between species. Based on these studies, we argue that effects of plant species selection on CW performance is likely to be important in regions subject to extended periods of low temperatures and plant dormancy.

Our studies

Starting in 1997, we conducted a series of greenhouse and laboratory studies of seasonal variation in model CW systems receiving various synthetic wastewaters simulating municipal wastewater (Allen *et al.*, 2002; Riley *et al.*, 2003; Hook *et al.*, 2003; Stein *et al.*, 2002), metalliferous mineland runoff (Borden *et. al.*, 2001; Stein *et. al.*, 2002), and solvent-contaminated laboratory effluent (Kowles, 2001). Within each set of experiments, influent concentrations of contaminants were uniform across seasons, reducing statistical 'noise' compared to studies of operational CW systems with varying wastewater composition and allowing controlled evaluation of seasonal variation related to temperature and plant growth. Each set of experiments compared several plant species across seasons, with conditions ranging from warm temperatures (typically 24°C), long days, and active plant growth to cold temperatures (typically 4°C), short days, and plant dormancy and senescence.

Allen *et al.* (2002) showed that COD removal in batch-operated wetlands displayed no differences between plant species when operated at 24°C (during active plant growth), but at 4°C (during plant dormancy) there were strong statistical differences among species (*Carex rostrata* > *Scirpus acutus* > *Typha latifolia* > unplanted controls). Only *Typha* and controls showed decreased performance at 4°C versus 24°C; *Carex* wetlands actually showed improved performance at 4°C. Differences in COD removal were explained by differences in measured redox potential, which were corroborated by differences in sulfide and sulfate concentrations indicative of anaerobic or aerobic conditions; Eh was uniformly low for all species and controls at 24°C but increased for planted treatments at 4°C, especially *Carex* and *Scirpus* wetlands. It appears that anaerobic pathways were mainly responsible for organic carbon removal during warm periods regardless of presence or species of plant. During winter (and plant dormancy) *Typha* wetlands and unplanted controls remained strongly anaerobic and displayed the expected performance decrease due to low temperature, whereas *Carex* and *Scirpus* wetlands developed a more aerobic environment and did not show decreased performance. Redox, sulfate, and COD removal data for *Carex* and *Scirpus* wetlands all suggested at least a partial shift to a more efficient aerobic metabolism during winter. We speculate that in the presence of *Carex* and *Scirpus,* the expected decrease in microbial activity due to low temperature was offset by the increased efficiency of aerobic metabolism, resulting in a lack of temperature dependency in overall performance.

The influence of plant-mediated oxygen transport on wastewater treatment appears to depend on the total demand for oxygen by plants and microbes, which depends on the organic carbon load to the system as well as temperature and plant growth or dormancy. Riley *et al.* (2003) concluded that seasonal effects on ammonium removal in batch-loaded *Carex* wetlands were influenced by carbon load and temperature; in summer at 24°C increased levels of organic carbon caused decreased ammonium removal, but in winter at 4°C ammonium removal was higher in wetlands with organic carbon. We speculated that in winter, increased oxygenation, as observed by Allen *et al.* (2002), was sufficient to support both hetrotrophic carbon oxidation and nitrification, while in summer competition for oxygen with hetrotrophs limited nitrification. Nitrate levels were higher in winter, but not significantly, suggesting denitrification was significant in all seasons.

Removal of some metals from wastewater can occur through precipitation of metal-sulfides. Sulfate reducing bacteria will utilize sulfate as an electron acceptor in the breakdown of organic carbon. Because the energy yield of this process is much lower than for aerobic breakdown of carbon, these microbes cannot compete with aerobic heterotrophs when oxygen is available and prefer anaerobic conditions. Consequently, sulfate reduction and metal-sulfide precipitation may be limited by insufficient carbon, excess oxygen, or low temperatures. Borden *et al.* (2001) showed that sulfate reduction was limited by organic carbon availability at 24°C (during active plant growth) in *Scirpus*, *Typha* and unplanted wetlands operated in either batch or continuous-flow mode. Sulfate reduction was only mildly inhibited at 4°C compared to 24°C in unplanted wetlands; however, cold temperature inhibition was greater for *Typha* wetlands and especially strong for *Scirpus* wetlands. Measured redox potentials were low for all treatments at 24°C, but were elevated for *Typha* and, especially, *Scirpus* wetlands at 4°C. In all seasons, virtually all available organic carbon was consumed (COD values approached zero), indicating similar conditions of carbon supply and limitation. We concluded that inhibition of sulfate reduction in planted wetlands during winter was most likely due to relatively aerobic conditions. As in the Allen et al. study (2002), a plausible explanation is that increased plant-mediated oxygenation during winter dormancy increased aerobic microbial utilization of available carbon, causing a shift in the dominant metabolic pathways of wetland microbial community. In this case, the consequences were greater inhibition of sulfate reduction than expected to result from low temperature and less effective precipitation of zinc when plants were present.

Kowles (2001) simulated treatment of wastewater from an instructional laboratory building to evaluate a potential on-site greenhouse CW system. Batch incubation experiments compared five plant treatments (unplanted control, *Juncus effusus*, *Carex lurida*, *Pontederia cordata*, and *Iris pseudacorous*) and three non-halogenated, polar, organic solvents (1-butanol, acetone, tetrahydrofuran) during summer (24/16°C day/night) and winter (13/7°C). Overall performance and differences between summer and winter performance depended strongly on presence and species of plants, with *Juncus effusus* providing the best overall performance and the least loss in performance during winter. Observed concentrations of sulfide, sulfate, and intermediate products of solvent degradation suggested that performance differences were partly due to differences in root zone oxygenation, while estimates of solvent loss via the plant transpiration stream suggested that this pathway also contributed to differences among species.

Discussion

Based on results of Allen *e al.* (2002) and additional observations of seasonal patterns in the same experiment, Hook *et al.* (2003) drew three main conclusions, which have generally extended to our later studies. First, seasonal variation in water treatment performance is modified strongly by the presence and species of plants. Depending on the wastewater type and plant species, contaminant removal may be more effective, equally effective, or less effective in winter as summer. Second, effects of plants on seasonal performance patterns appear to be explained largely by differences in root-zone oxidation. Third, the effects of plants on performance are frequently greatest during the coldest periods, during dormancy, implying that plant species selection may be more important to cold-season than to warm-season performance.

Seasonal differences in root zone oxygenation may result from variation in root respiration driven by temperature and cycles of growth and dormancy (Allen *et al.*, 2002; Hook *et al.*, 2003). Limited published research suggests root respiration is reduced during periods of plant dormancy and that oxygen transfer via plants is possible year-round. Consequently, variation in root respiration would be expected to influence root-zone oxygen supply. With high temperature and active plant growth, root respiration would consume the bulk of oxygen transferred to roots; with low temperature and plants dormant, reduced internal oxygen consumption would allow greater oxygen leakage from roots. Thus, active plant growth and warm temperatures might favor anaerobic microbial metabolism, while plant dormancy and cold temperatures might favor aerobic microbial respiration. Considering that differences in efficiency between aerobic and anaerobic metabolism are similar or greater than differences in metabolism expected to result from seasonal variation in temperature, this hypothesis offers one possible explanation for the limited effect of temperature on wetland performance reported in the literature. Other explanations including variable loading rates and carryover of organic matter between seasons may also apply and are not mutually exclusive (Kadlec and Knight, 1996; Kadlec and Reddy, 2001).

There is little information on the relative performance of different plant species in cold-region subsurface wetlands. Most research involves just a handful of species, and direct comparisons between species have focused mostly on growing season performance at relatively warm temperatures. However, our results suggest that the influence of plants might be greater during periods of plant dormancy and that differences between species might also be greater at this time. As a result, plants are potentially as important, or more important, to subsurface wetland performance during the winter as during the growing season.

The plant effects described here may be more pronounced in batch-loaded than continuous-flow CW systems. Stein *et al.* (2002) hypothesized that batch hydraulic loading ensures that the entire microbial population will be exposed episodically to decreasing organic carbon concentrations. This decrease in oxygen demand in turn can allow root zone redox potential to increase over the course of a batch incubation, particularly when plants are dormant and oxygen transfer rates apparently are higher. Temporal variation in oxygen supply and redox potential may select for more robust, aerobically facultative biofilms capable of rapid sequestration of COD and nutrients for utilization later in the batch cycle.

Conclusions

Based on results of our studies and information available from others, we believe that plant-mediated oxygen transport to the rhizosphere and competition between plant tissues and microbes for available oxygen varies by species, and the balance between these processes varies during the season. In the presence of some plants, but not others, this tends to shift microbial metabolic pathways from predominantly anaerobic during periods of active plant growth to an increased contribution by aerobic respiration during plant dormancy. However, it is likely that overall organic carbon load and mode of hydraulic operation influence this potential shift; high carbon loading and continuous-flow operation likely dampens seasonal and plant effects. Practical consequences of differences in root zone oxidation will depend on wastewater type and the importance of oxidative and reducing processes to treatment. Better understanding of seasonal variation in processes responsible for constructed wetland efficacy should lead to improved design and management of these systems, making them viable technologies for wastewater treatment in cold temperate climates. To achieve this understanding, it is critical to recognize that seasonality comprises much more than temperature.

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Removal of enteric bacteria in constructed treatment wetlands with emergent macrophytes

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Constructed wetlands (CWs) with emergent macrophytes either with free water surface (FWS) or with sub-surface flow (horizontal – HSF and vertical – VF) are predominantly designed to remove organics (BOD and COD) and suspended solids. More recently, these wetlands in combination (hybrid CWs) have been designed for nutrient removal. The removal of microbiological pollution is never a primary target for treatment using constructed wetlands. However, wetlands are known to act as excellent biofilters through a complex of physical, chemical and biological factors which all participate in the reduction of the number of bacteria. Physical factors include mechanical filtration and sedimentation, chemical factors include oxidation, UV radiation, exposure to biocides excreted by some plants and adsorption to organic matter. Biological removal factors include antibiosis, predation by nematodes, protists and zooplankton, attack by lytic bacteria and viruses and natural die-off (Seidel, 1976; Gersberg *et al.*, 1989).

Domestic and municipal sewage contains various pathogenic or potentially pathogenic microorganisms which, depending on species concentration, pose a potential risk to human health and whose presence must therefore be reduced in the course of wastewater treatment (Hagendorf *at al.*, 2000). Measurement of human pathogenic organisms in untreated and treated wastewater is expensive and technically challenging. Consequently, environmental engineers have sought indicator organisms that are 1) easy to monitor and 2) correlate with population of pathogenic organisms. No perfect indicators have been found, but coliform bacteria group has been long used as the first choice among indicator organisms (Kadlec and Knight, 1996). Coliforms are usually monitored as total or fecal coliforms. The fecal streptococcus (FS) group is also used frequently to confirm fecal contamination. Coliform bacteria and fecal streptococci are excreted as fecal constituents. Total coliforms (TC), however, are ubiquitous in surface waters, and they include many bacteria from the family *Enerobacteriaceae* that are not derived from human or other animal pollution sources. Contrary to FS the detection of coliforms in wastewater indicates only a possible contamination by feces, as these organisms are capable not only of surviving but also in some cases, of multiplying in water and on soil particles and plants Thus, the total coliform measurement is the least specific indicator for providing evidence of human fecal contamination. The fecal coliform (FC) group is composed largely of fecally derived coliforms (mostly genera *Escherichia*, *Klebsiella*, *Citrobacter, Enterobacter*), but it also includes free-living bacteria and bacteria from other warm-blooded animals including birds and mammals. Thus although the fecal coliform measure is a better indicator of human fecal contamination than total coliform, it is by no mean specific (Kadlec and Knight, 1996; Dufour, 1977). However, it has been confirmed that bacterium *Escherichia coli* is unanimously derived from human feces. The detection of FS in wastewater indicates a direct fecal contamination of the water, as FS are not or not significantly capable of multiplying following excretion by humans. Because fecal streptococci bacteria are more resistant to environmental stress (*e.g.* temperature, chemical agents) than fecal coliforms, they are used as a second indicator of fecal contamination and may be a better indicator of the presence of the longer-living viruses originating in wastewater (Clausen *et al.,* 1977). However, it has also been reported that despite higher resistance towards certain environmental stress factors FS usually survive in water shorter period than bacteria of the family *Enerobacteriaceae.* Therefore, FS are considered as indicators of "fresh" pollution. Another indicator of fecal contamination is *Clostridium perfringens*, anaerobic spore-forming bacterium, which is always present in human feces. *C. perfringens* spores are very resistant and survive in water longer that coliforms.

The literature survey revealed that removal of total and fecal coliforms in constructed wetlands with emergent macrophytes is high, usually 95 to >99%. This removal effect is slightly superior to activated sludge process and trickling filters where typical reductions are 90–99% (Miescier and Cabelli, 1982; Crook, 1990) and comparable with slow sand filtration (Verlichini and Masotti, 2000). Removal of fecal streptococci is lower, usually 80–95% based on literature survey. Because bacterial removal efficiency is a function of inflow bacteria number, the high removal effects are achieved for untreated or mechanically pretreated wastewater. Results from operating systems around the world suggest that enteric microbe removal efficiency in CWs with emergent macrophytes is primarily influenced by hydraulic loading rate (HLR) and resultant hydraulic residence time (HRT) and the presence of vegetation. In Figure 1, an example of HRT influence on bacteria removal is presented. The effect of HRT is very simple – the longer HRT the longer the bacteria are exposed to unfavourable conditions causing a natural die-off. It seems that CWs with HLR < 5 cm d^{-1} and HRT of at least 5 days provide a very high bacteria removal.

There is growing evidence that CWs with macrophytes are more effective in bacteria removal as compared to unplanted beds or unplanted ponds (Figure 2). This phenomenon may be caused, first of all, by two factors: 1) presence of oxygen in water column of FWS or rhizosphere CWs with sub-surface flow and 2) presence of plant exudates with antimicrobial properties. Enteric bacteria are either facultative or obligate anaerobs and thus the presence of oxygen create unfavourable life conditions for these organisms.

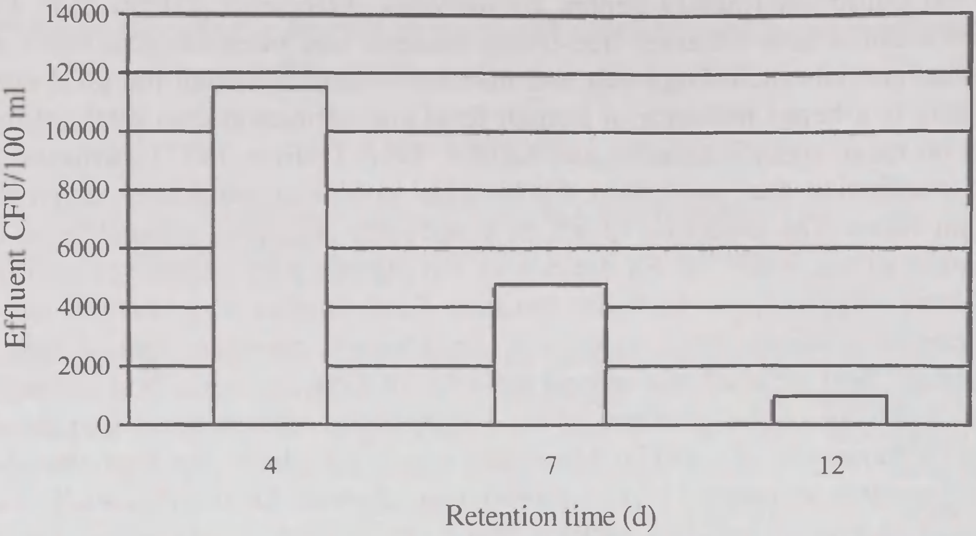

Figure 1. Fecal coliforms in the effluent of FWS constructed wetland with floating mats of *Cyperus papyrus* at Jinja, Uganda under various HRT (data from Okurut and van Bruggen, 2000).

In FWS wetlands, oxygen concentrations in water could be quite high because of photosynthetic activity of algae, in CWs with subsurface flow oxygen leaks from roots and rhizomes of macrophytes. In addition, in VF wetlands an intermittent feeding allows more oxygen to diffuse into beds.

It has been shown that root excretions of certain aquatic macrophytes including *Scirpus lacustris* and *Phragmites australis* kill fecal indicators and pathogenic bacteria (Sediel, 1976; Gopal and Goel, 1993; Neori *et al.*, 2000). Additionally, the enhanced development of populations of bacteria with antibiotic activity (*e.g. Pseudomonas*) in the rhizosphere (Broadbent *et al.*, 1971) may also account for coliform die-off.

It has been shown that in HSF CWs there is little seasonality and removal of bacteria is steady because temperature in the bed fluctuates only slightly during the year (Watson *et al.*, 1987; Vymazal *et al.*, 2003). In FWS, the temperature fluctuation of water during the year is much higher but the effect of this fluctuation is not unanimous. Increased temperatures can prolong bacteria survival in the environment but on the other hand higher temperature favours growth of their predators and also growth of algae resulting in oxygen production. Physical processes, such as sorption or settling, are not particularly temperature-sensitive. Annual irradiation patterns mimic the annual temperature cycle, and hence ultraviolet-induced mortality should be higher at higher temperature. In constructed wetlands with horizontal subsurface flow, *i.e.* systems with no free water surface, ultraviolet-induced bacteria removal is negligible.

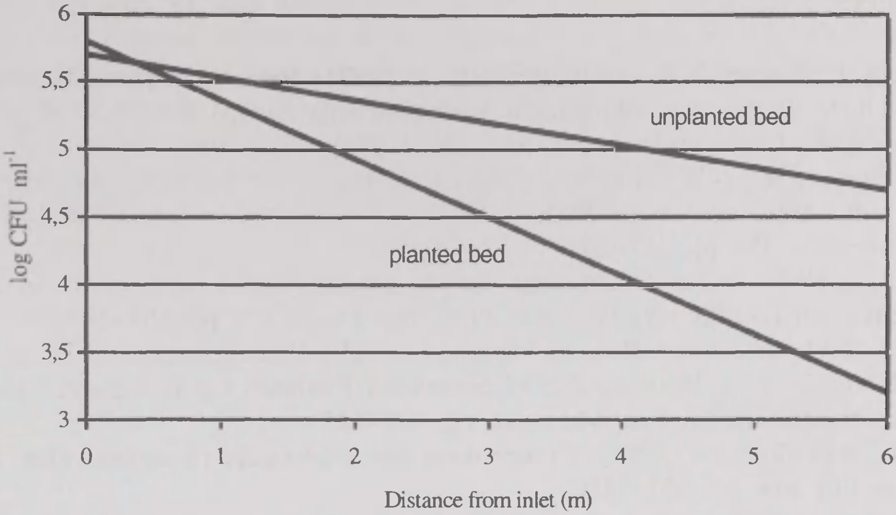

Figure 2. Relationship between *E. coli* numbers and the distance from the inlet of HSF CWs at Audlem, UK (data from Warren *et al.*, 2000).

Removal of enteric bacteria follows approximately a first-order relationship (Equation 1) as long as inflow bacteria populations are high:

$$C_o/C_i = \exp(-k/q) \text{ or } k = q \ln(C_i - C_o) \qquad (1)$$

where C_o and C_i are bacteria concentrations (#/100 mL) in the outflow and inflow, respectively, k is areal first-order rate constant (m d^{-1}) and q is HLR (m d^{-1}). Results from more than 50 constructed wetlands around the world revealed that k values (m d^{-1}) vary widely in the range of 0.032–3.46, 0.007–3.12 and 0.066–2.25 for total coliforms, fecal coliforms and fecal streptococci, respectively.

Acknowledgements

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CONTRIBUTED PAPER SESSIONS

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Phosphorus sorption by Filtralite-P – small scale box experiment

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Introduction

Phosphorus is often the major limiting nutrient in freshwater systems. Since there is no important gaseous component in the biogeochemical cycle, phosphorus tends to move to the sediment sink in natural systems and become scarce in the ecosystem (EPA, 2000).

Several authors (Drizo *et al.*, 1997; Johansson, 1998; Cheung, 2000; Arias, 2001; Drizo *et al.*, 2002) have been trying to obtain the long term P sorption capacity of different porous medias with different methods to calculate the life-time of these natural systems. While others (Lookman *et al.*, 1996 and Monterroso Martinez *et al.*, 1996) tried to find relationship between the properties of these medias and P sorption. To take one step forward on this field a box experiment had been designed. The idea was to take a box – as one cell or a small part of the full-scale horizontal flow wetlands – to model the P sorption processes and kinetics.

The main objective of this study was to investigate the nature of phosphates accumulated in the filter material and the kinetics of reactions of orthophosphate ions with compounds of the filter media to be able to calculate the long term P sorption capacity of these wetland systems.

Materials and methods

Filtralite-P

The Filtralite-P is an expanded clay product especially made to sorb phosphorus. The clay particles were fed into a long rotary kiln, where the clay is expanded at a temperature about 1200 °C (Jenssen *et al.*, 2002; Optiroc, 2003). The expansion creates high porosity (65%). Its dry particle density has a range of 600–800 kg m^{-3} with a corresponding dry bulk density of 300–550 kg m^{-3}, and a saturated hydraulic conductivity of 100 m day^{-1}. (Optiroc, 2003). Filtralite-P consists of crushed particles in the range between 0–4 mm. This leaves the inner structure of the Filtralite-P exposed with a large surface area that enhances the removal of pollutants. Khadhraoui *et al.* (2002) showed that the increase of the amount of P removal could be positively correlated with the increase of the specific surface area.

Filtralite-P was filled in transparent- boxes with an outer dimensions of 26 cm * 12 cm * 7 cm (Figure 2). Each box contained almost 1.64 liter of the Filtralite-P with an average weight of 953.35 ± 0.01 gm. Initially, deionized water was used only. Starting from day 22, five different concentrations of phosphorus (15, 10, 5, 2, and 0 ppm) with 5, 2,5, and 1,25 l d^{-1} loading rates were used. The P solution was prepared by mixing KH_2PO_4 with deionized water to reduce potential biological activities that could disturb the sorption mechanisms. The stock of each concentration was prepared for a period of almost one week and dosed from a 45 liter containers for the 10, 5, 2, and 0 concentrations and from a 200 liter container for the 15 ppm concentration.100 ml samples were collected each second day to measure the total phosphorus content in the outlet, as well as the EC, pH, Calcium and Magnesium. Not only the P removal capacity of the material, but also the relationship between the different parameters and P removal and degree of saturation was investigated. P extraction was conducted throughout 2 boxes with 1,25 l d^{-1} loading rate and with 15 and 2 ppm inlet P concentrations in 27 locations after 1,5 years of operation. Maximum sorption capacity and P fractionation was studied by batch experiments.

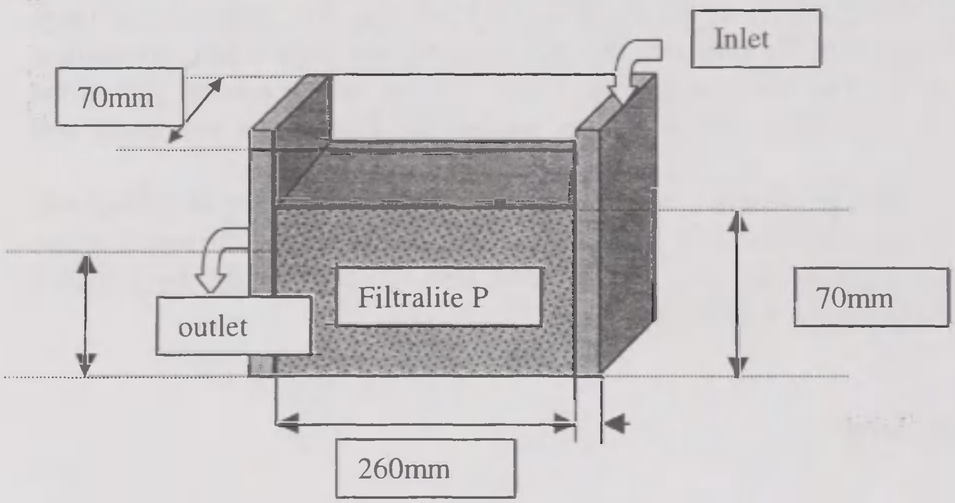

Figure 1. Illustrative sketch of the boxes used in the experiment.

Analytical methods

The PO_4-P was determined by mixing 10 ml of each sample with 0.4 ml of ascorbic acid and 0.4 ml of molybdate and measures its' absorbance by Spectrophotometer (model/Stasar II). 10 ml standard phosphorus solution (1 ppm concentration) was measured before each analysis as a reference for the phosphorus content in the samples. Calcium and magnesium concentrations were determined by taking 0.1 ml of each sample and dilute it up to 3 ml with distilled water, then mix it with 0.1 ml lanthanum chloride (10% w v^{-1} La, and 26.6% w v^{-1} $LaCl_3 \cdot 7H_2O$) and measured by the atomic absorber (model/ PERKIN-ELMER 2380) (NSF, 1984). P had been extracted with H_2SO_4 (12 N) from the inlet, middle and outlet section of 2 different boxes after one and a half years of operation. One section consisted of 9 different spots. The extraction was conducted by using Norwegian Standards.

The P fractionation of the material was carried out by a method used by Zhu *et al.* (1998). One gram of each sample with 40 ml of extractant was shaken during the extraction period. Extractant from each step was filtered through a 0,45 micrometer membrane and analyzed for PO_4-P by the same method mentioned above.

Results and discussion

The outlet P concentration was increasing by time (Figure 2 and Figure 3). The boxes with high inlet P concentration and loading rate reached the saturation level relatively fast (after about 150 days of operation), while the boxes with low inlet P concentration (5,2 ppm P) and low hydraulic loading (1,25 l d^{-1}) are still not completely saturated even after 1,5 years of operation, 3,53 ppm P and 1,39 ppm P respectively.

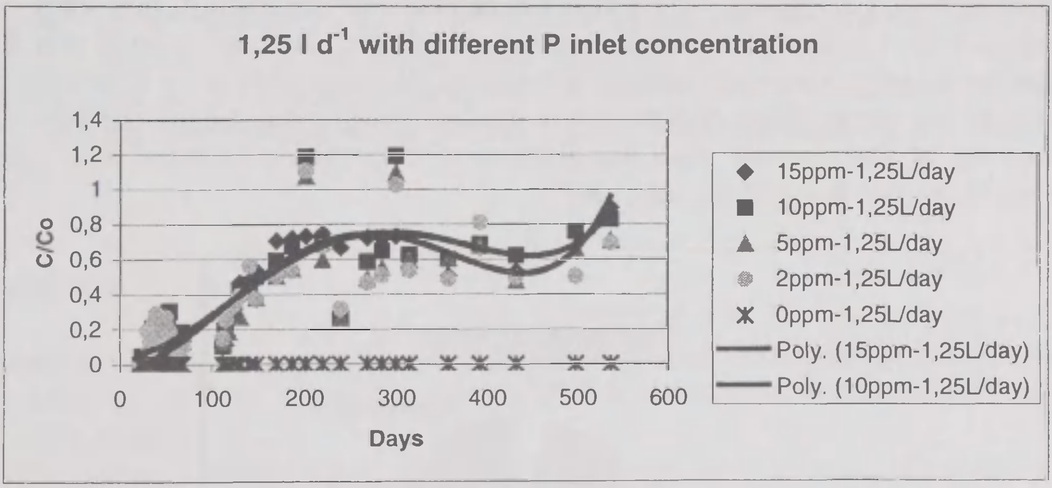

Figure 2. The level of P saturation in the boxes with 1,25 l d^{-1} loading rate vs. time.

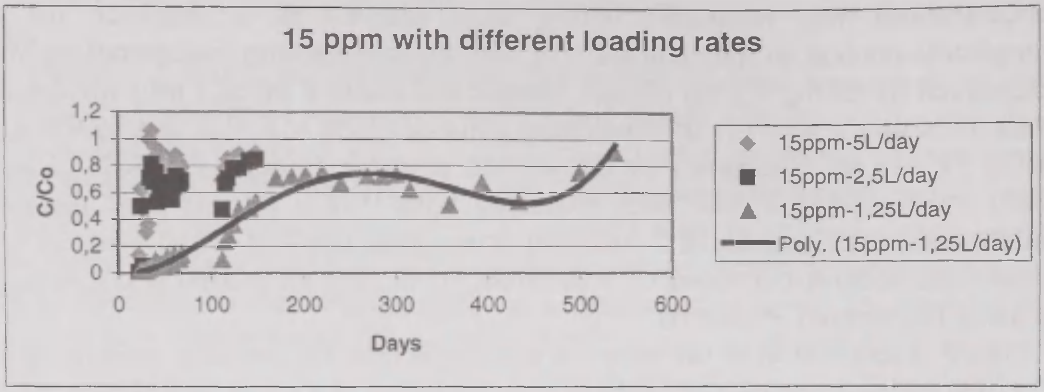

Figure 3. The level of saturation in the boxes with 15 ppm inlet P concentration vs. time.

This is obvious from the boxes with 15 ppm inlet P concentration and with 5, 2.5 and 1.25 $l\ d^{-1}$ loading rates; whenever the loading rate increases, the outlet P concentration was increasing due to the more loaded mass of P into the box. Regarding the different inlet P concentration with the same loading rate, it showed also the same behaviour, where the outlet P concentration increases as the inlet P concentration increases.

We assumed that Ca bounded P would be the main P pool because of the high pH and high Ca content of the media. Fractionation studies have been carried out to confirm our assumptions. Surprisingly we found that the Ca pool contribute only 11.1% of the total P removal (Figure 4). The majority of P (56%) was bound to unidentified compounds of the material constituent. The fractionation study did not include the identification of organic phosphates since we assume that the system did not contain any organic substances. However with using P solution to imitate the composition of wastewater we can easily overestimate the sorption capacity of the material with the absence of competition of other anions for adsorption (van Riemsdijk *et al.*, 1977).

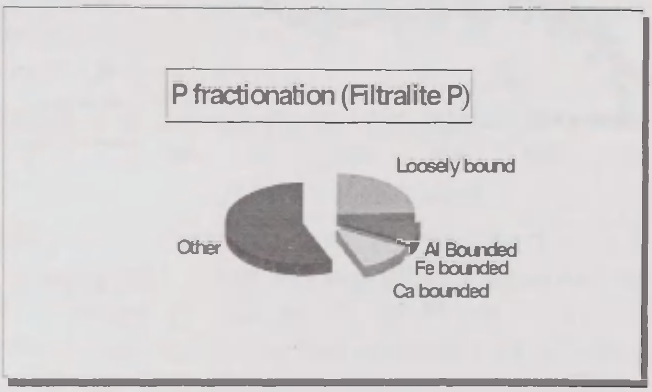

Figure 4. P fractionation of the Filtralite-P.

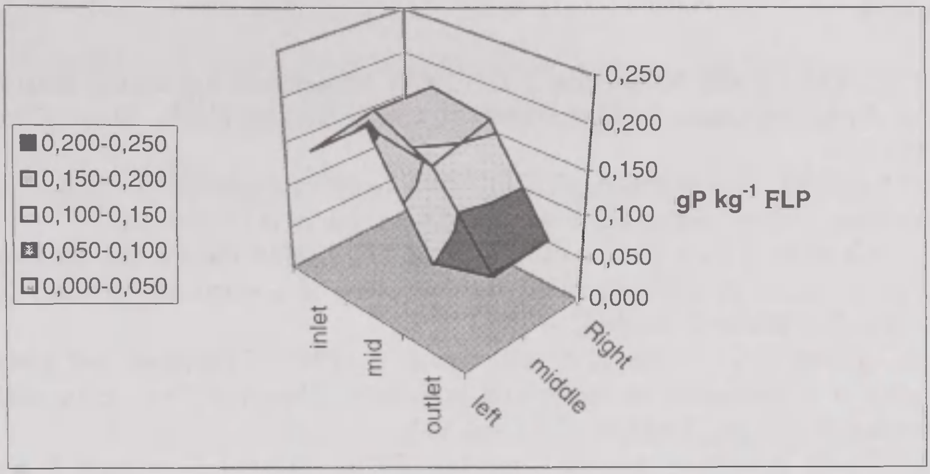

Figure 5. Distribution of the P concentration (g P kg^{-1} Filtralite-P (FLP)) in the box.

The distribution of P shows how the P accumulated inside the boxes after 1,5 years of operation. The background P concentration of the material was measured to 38 mg P kg^{-1} Filtralite P. The total P sorbed by the material was calculated by subtracting this value from the measured total P values.

Figure 5 shows that the sorbed P accumulated within the inlet section of the box and gradually decreased towards the outlet. The distribution pattern also shows a gradually increase towards the bottom of the boxes.

The fractionation study showed that the major part of the P could not be extracted by the method we used. In this case we are not able to calculate the expected lifetime of the on site systems based on this data since we would underestimate the P sorption capacity of the material. So for this purpose we rather use the mass balance of the boxes ($M_{sorbed\ P} = M_{inlet}-M_{outlet}$) for the whole period.

Conclusion

Filtralite P has shown a good P removal capacity during the experimental period. The removal rate was dependent on the inlet P concentration, Ca content and hydraulic loading.

The sorbed P accumulated within the inlet section of the boxes and gradually decreased towards the outlet zone. The fractionation study showed that the Ca and Al were the major identified chemically P sorption pools in the boxes. But further digestion of the material should be carried out using stronger acidic extractants in order to characterise the unidentified compounds of the P sorption.

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Ecotoxicological and leaching properties of sediments and filter-media from constructed wetlands

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Summary

The results from the ecotoxicological tests show that mixing of 10–25% of sediments or filter-media from constructed wetlands (CW) into soil has in most cases a beneficial effect on plant growth or soil living *Enchytraeidae*. The observed biological effects of CW-materials in soils cannot be explained by the content of inorganic and organic contaminants alone, but are a combination of nutrient and contaminant content, as well as the texture of the materials.

Ecotoxicological testing of CW-materials have proven useful for elucidating both negative and positive effects of CW-materials in soils. The methodology has to be improved to better separate the effects of nutrients and texture from possible long-term detrimental effects.

The leaching of nutrients and inorganic contaminants from the CW-materials varies considerably. Results indicate that both pH and the redox-potential change during the leaching process, showing that long-term leaching experiment are important for proper CW-material characterisation.

Introduction

Depending on the wastewater source and type of CW, we have shown that filter-materials and sediments from CWs contain varying amounts of nutrients, organic matter and inorganic and organic contaminants. The strong acid extractable concentration of e.g. cadmium, lead, mercury, copper and zinc has been shown to be low in most of the materials. So has the concentration of polycyclic aromatic hydrocarbons (PAH), phthalic esters, nonylphenol (NP) or nonylphenol ethoxylates (NPE) and linear alkylbenzensulphonates (LAS). The "total" concentrations of each of the inorganic and organic contaminants are generally lower than threshold values for toxic effects established for these contaminants in the EC or national legislations concerning land application of sewage sludge or compost. Existing legislations therefore do not restrict the re-use options of most CW-materials. The low concentrations of measured contaminants, however, can not exclude the possibility that CW-materials may have detrimental effects on soil function or its biological diversity. Neither can negative effects due to leaching bee excluded.

The toxic effect of CW-materials after application to soils can be determined by measuring the effect on different trophic levels in soils. The major advantage of performing toxicity tests on these materials is that the effect of all contaminants which is toxic for the test organisms is measured, including those contaminants that are not determined in the chemical analysis. In this project sub-lethal toxicity tests with ryegrass (*Lolium perenne*), an enchytraeid worm (*Enchytraeus crypticus*) and soil nitrifying bacteria were therefore performed.

The leaching properties of the CW-materials with respect to nutrients and heavy metals were also determined by performing a batch extraction and an up-flow percolation test.

Methods

CW-samples

Leaching and ecotoxicological tests were performed on filter-materials and sediments from twelve different CWs (Table 1). Materials were sampled in wetlands with combined overflow and groundwater flow (OGF), groundwater flow in porous media (GWS) and free water surface systems (FSW) with or without vegetation.

The chemical and physical composition of the materials is presented in Amundsen *et al*. 2003 (poster at this conference).

Biological tests

In the plant and *Enchytraeidae*-tests all CW-samples were tested at five "concentration" levels (values are % CW-sample (w/w) in reference soil): 0, 10, 25, 50, and 100. Three replicates were used per test concentration. For each concentration level, the sample was mixed thoroughly and the water content adjusted to approx. 65% of the water holding capacity (WHC). In the plant test the seedlings were cut and immediately weighed to the nearest 0.001 g (fresh weight) after 21–24 days of exposure. For the effect estimations, the mean fresh weight of seedlings in each replicate were used (the test procedure was a modification of OECD Guidelines 208: Terrestrial plants, Growth test (OECD, 1984)). In the *Enchytraeidae* test the number of surviving adults and number of juveniles was counted manually after 21 days of exposure (test procedure similar to *Enchytraeidae* reproduction test. OECD Guidelines 220, Draft guideline, March 2000). All test results were in accordance with validity criteria defined in the test standards.

In the nitrification test the CW-materials were mixed with reference soil at test concentrations of 10% and 50% (w/w) and the samples moistened to approximately 60% of the WHC. The potential ammonium oxidation rate (PAO) was assayed as accumulated nitrite according to a short incubation, chlorate inhibition technique. As a reference soil, a sandy loam was used.

The Microtox Acute Toxicity Test (which is a 15 minute exposure test with luminescent bacteria) was used for screening of the toxicity of the LS 0.1-water extracts from the column leaching experiment (see description below).

Leaching experiments

Batch extraction tests were performed by extracting one hundred grams of CW-material with 1000ml purified water in a polyethylen bottle for 24 hours and the liquid-solid separation was done by filtration (0.45μm membrane filter).

In the column experiment the CW-materials were packed in PE-columns (height: 30cm; width, 7,5cm). After saturation of the CW-materials for 72-hours, eluates were collected at the top of the column (up-flow water flow) and filtered. The liquid/solid (LS)-eluate fractions 0.1, 0.2, 0.5, 1.0, 2.0, 5.0 and 10 were collected during the test. The content of nutrients (total P, PO_4^{3-}, total N, NH_4 and NO_3), dissolved organic carbon (DOC), macro and trace elements, as well as pH and conductivity, was determined in the filtered eluates from both leaching experiments.

Results and discussion

Biological tests

The dose-response relationships of the plant-tests could be divided into two main groups: Samples with no significant effects on growth (Kompsa, Tänassilma and Mniow) (Figure 1), and samples with significant positive effects on growth at the 10–50% exposure levels (Figure 1) (the rest of CW-materials with exception from Vassum).

Only one sample (Vassum) had a significant ($p<0.05$) negative effect on plant growth at the 100% exposure level (Table 1; Figure 1).

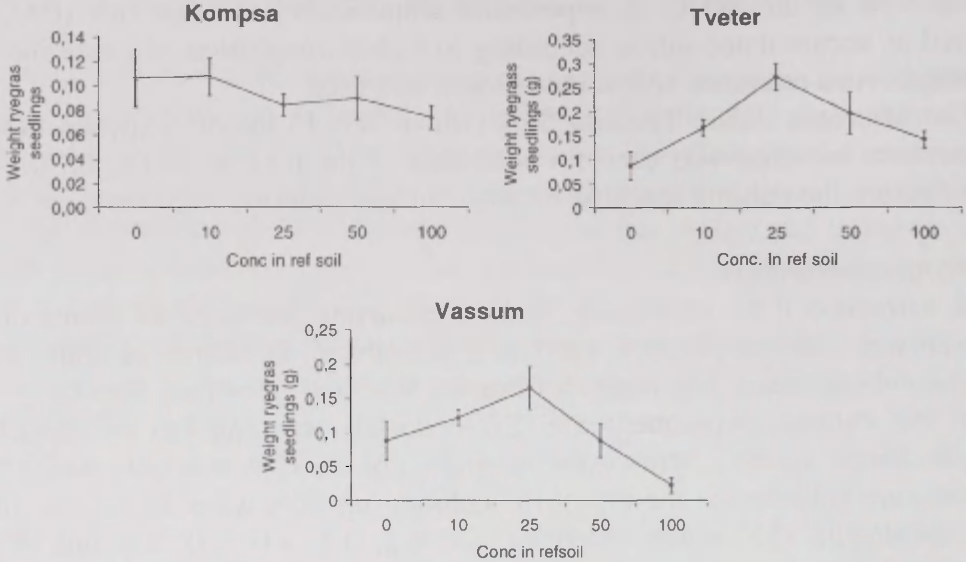


Figure 1. Dose-response relationship for the growth of ryegrass in mixtures of various CW-materials and reference soil (amount of CW-materials equals 0, 10, 25, 50 and 100%).

Table 1. Summary of results for the toxicity tests with the highest concentration of filter samples that was tested (50% is the highest test concentration for nitrification, 100% for the other tests). Observed effects on the various organisms and endpoints are categorised as follows: Negative effects (slight "-", distinct "- -", extreme "- - -") and positive effects (slight "+", distinct "+ +", extreme "+ + + +"). Effects that are non-significant at the 5% confidence level (ANOVA, Dunnett's) are written in parentheses.

Wetland	Enchytraeidae	Nitrification (50% level)	Ryegrass
Kompsa	(+)	--	(-)
Lakeus	0	++	(+)
Hovi	- - -	+	0
Tveter	- - -	++	(+)
Spillhaug	- -	0	(+)
Ski	+ +	+++	(-)
Kodijärve	(-)	++	(-)
Põltsamaa I	- -	++++	(-)
Tänassilma	(-)	++++	(+)
Mniow	- - -	--	0
Haugstein Fe-filter	0	--	0
Vassum	(-)	++	- -

For enchytraeids, a strong positive effect on reproduction was observed for the sample from Ski, while the samples from Hovi, Tveter, Spillhaug, Põltsamaa I and Mniow gave a reduction in reproductive output at the 100% level (Table 1). At the lowest concentration level tested (10% sample in reference soil) no toxicity was recorded for any of the samples tested.

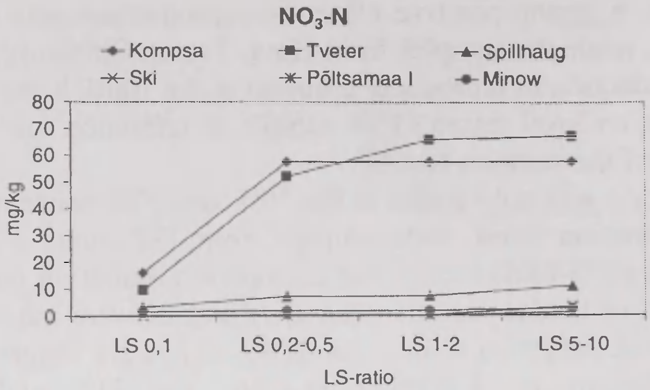
Soil nitrification was only tested at the 10% and 50% concentration levels. At the 50% concentration level, the samples from Põltsamaa I and Tänassilma showed a very large (7-fold) increase in ammonium oxidation potentials (AOPs), and for seven out of twelve samples, the AOP-values were more than two times that of the controls. Negative effects on AOPs were only observed for materials from Kompsa, Mniow and Haugstein Fe-filter. The 50% AOP-values for the samples from Tveter and Vassum were lower than 10% AOP-values, suggesting that some negative effect are associated with the samples at higher concentration levels.

When CW-materials were mixed with soil at low concentrations (10%) the materials gave positive or no biological effect on the test species applied in this project. This indicates that based on the results presented here, the beneficial effects of these materials at low doses in soils may be exploited without the concern for possible negative effects.

At higher concentrations in soils (25–50%) some of the CW-materials have negative effects on soil organisms.

Leaching experiments

The composition of the eluates from the CW-materials varies considerably. The pH, conductivity and dissolved organic carbon vary in the range 5.9–9.7, 65–2800 μS/cm, and 4,4–51,8 mg/l, respectively. Also the concentrations of total phosphorous, PO_4, NH_4 and NO_3, as well as inorganic contaminants, vary considerably between eluates from the CW-materials. The quantity of water-soluble species in the eluates depend on the composition of the wetland inlet water, the biological processes going on in the wetland and the sorption properties of the CW-material. There is, however, no significant correlation between the leachable fraction of heavy metals and the Aqua Regia extractable content.



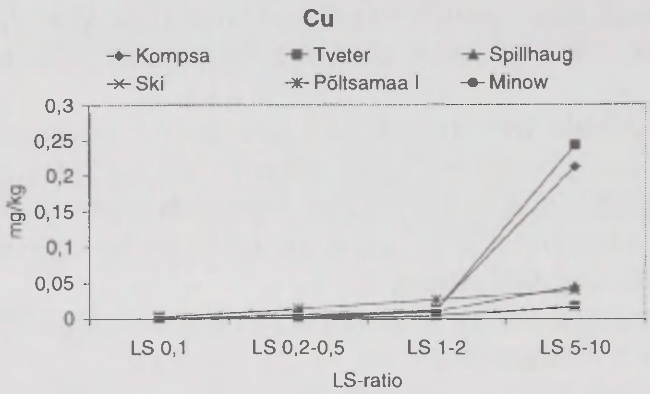


Figure 2. Accumulated leaching of NO_3-N and Cu during column leaching experiment for some selected CW-materials.

In general, the leaching of nutrients and inorganic contaminants decreases with time and the accumulated leaching from the CW-materials increases slowly during the leaching period (Figure 2). For some CW-materials, however, the leaching of nutrients and inorganic contaminants increase abruptly during the last stage (Figure 2), indicating that changes might occur in the CW-materials that influence the leaching process. E.g. for CW-materials having a high the content of organic matter and sulphur, the redox-potential seems to decrease markedly during the process.

The toxicity of the LS 0.1-water extracts from the column leaching experiment was only observed in the eluates from Põltsamaa I (EC_{50} 7.4) and Mniow (EC_{50} 21.7). The low observed toxicity is in good agreement with the low level of inorganic contaminants that was found in the eluates.

A Helium incubation method to determine flux rates of CH_4, CO_2, N_2O, and N_2 of drained and flooded wetland soils

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Until today one knows only very few about the actual C/N turnover and transfer rates of N rich wetland sites like reflooded fens, riparian buffers zones, and constructed wetlands. That applies particularly to flooded conditions and the denitrification process. Very little information is available about the actual N_2 emissions from all wetland sites. However, investigation of the denitrification potential indicates that the total N emissions (N_2O + N_2) might substantially exceed the N_2O emissions. At least for fens with a high water table, potential denitrication rates in the range of 70–200 kg N ha^{-1} a^{-1} have been reported (Terry and Tate, 1980; Lippold *et al.*, 1986; Eschner and Lorenz, 1989). As is generally known during denitrification not only N_2 but also the greenhouse gas N_2O is formed. For example, there is some evidence that constructed wetlands polluted with nitrate containing wastewater (Frey *et al.*, 1999, Tai *et al.*, 2002) and drained fens (Merbach *et al.*, 2001) could be a significant source for N_2O. The main cause for the big knowledge deficits is the insufficient suitability of current methods to measure the actual formation and emission rates of N_2O and N_2 especially under anoxic conditions. That is in particular valid for the acetylene blockage technique, which is often used for this purpose. Especially under flooded conditions it is not suitable for measuring reasonable denitrification rates (e.g., Rolston, 1986; Bollmann and Conrad, 1997). To be very much more reliable the ^{15}N gas flux and technique proved meanwhile (Chen *et al.*, 1998, Augustin *et al.*, 2002). However, labelling of soil as prerequisite for measuring N gas fluxes by ^{15}N technique is especially in peat soil (high N content) extremely expensive. Moreover, it not of clear, whether the input of relatively high amounts of labelling material as it is necessary here, influenced the processes of N turnover and N gas emission.

Therefore we try to develop an alternative approach, which permits us to measure non-falsified, actual denitrification rates in a fast and a little expensively way under oxic and anoxic conditions. Basis for the development was the old idea, allowing the direct measurement of the N_2O and N_2 evolution after replacing the N_2 of the atmosphere (soil and air) by a novel gas like Helium in closed incubation systems (e.g., Ackermann *et al.*, 1972). Contrary to the procedures already described in the literature (Scholefield *et al.*, 1997; Butterbach-Bahl *et al.*, 2002; Cardenas 2003) our system should be used also under flooded conditions. Since wetlands exhibit a high potential climatic relevance because of its high C and N-supplies, also the others, here particularly important greenhouse gases like carbon dioxide (CO_2) and methane (CH_4) should be also seized within the measurements. In order to guarantee that one receives genuine measured values with the procedure also under these conditions it was implemented, test series with an inert and sterilized soil substrate (quarry sand silica dust mixture) under a broad soil moisture spectrum were conducted.

In the following selected results are presented for the examination of the Helium incubation method and for the investigations on fen peatland sites, riparian buffer zones, and stream sediments accomplished with its assistance.

Table 1. Some characteristics of the sites and/or cores investigated.

Origin place	Substrate Type	Soil dry matter %	Total N %	NH_4^+ mg/100g (average)	NO_3^- mg/100g (average)	Total C %
Lab mixture	quarry sand/silica dust	variable	0	0	0	0
Stream bed	stream	21.0	0.57	2.9	0.08	
without N pollution	sediment	14.5	1.35	5.3	0.11	7
with N pollution						15
Fen mire alder forest	Eutric	variable	1.6	0.43	2.80	
strongly degraded	Histosol	variable	1.9	0.46	1.40	20.1
weakly degraded						39.5
Riparian alder forest	Molli-					
river bank (M)	Eutric	variable	0.4	1.88	0.09	4.0
alder site (K)	Gleysol	variable	0.8	0.33	0.14	4.5
meadow (S)		variable	0.3	0.61	0.11	5.3

For gas flux measurements by means of the helium incubation method soil cores of the origin were used. **Gumnitz** (Brandenburg/Germany): drainage of an alder swamp forest on a fen mire (variants: deeply drained/strongly degraded, shallow drained/weakly degraded). **Porijõgi Valley** (south Estonia): effect of different positions into a riparian grey alder zone (variants: river bank, alder site, meadow). **Erpe Valley** (Brandenburg/Germany): effect of stream N pollution by sewage plant (variants: without N pollution/ above sewage plant, with N pollution/ below sewage plant. Quarry sand/silica dust (artificial lab mixture): effect of different soil moisture conditions in an inert and sterilized quarry sand/silica dust mixture (3:1, sand particle size 0.2–0.6 mm, silica particle size < 0.02 mm) of the emission of (abiotic) N_2 derived from soil air (variants: moist, wet, flooded, inundated). In the Table 1 some further information of the substrates is shown. While the soil samples of the field sites were undisturbed, it concerned with the sand/dust mixture disturbed samples.

For the measurement of the gas fluxes the soil cores (diameter 6.8–7.5 cm, height 6 cm) were sampled in Estonian and German study sites at certain times in the process of the years 2001 up to 2003, respectively. At the laboratory all types of samples was introduced in special gas-tightly closed incubation vessel. For the determination of actual gas flux rates following procedures was used: 1) removing of N_2 by 3 subsequent slight evacuation/flushing cycles with the artificial gas mixture (21,3% O_2, 78,6% He, 337 ppm CO_2, 374 ppb N_2O, 1882 ppb CH_4 and approx. 5 ppm N_2); 2) establishing a new flow equilibrium by continuous flushing the vessel headspace with the artificial gas mixture at 10 ml per minute for 16 hours; 3) measuring the N_2 and the greenhouse gas concentration in the continuous gas flow via sample loop and gas chromatographs in the continuous gas flow (start value): N_2 by a molecular sieve column and a thermal conductivity detector (TCD); N_2O and CO_2 by a electron capture detector, CH_4 by a flame ionisation detector, all three with a Porapak Q column; 4) closing the incubation headspace for one hour to accumulate the emission of N_2 and the greenhouse gases; 5) measuring again the gas concentrations in the incubation headspace (final value); 6) make the difference of the gas concentrations (final accumulation value minus start continuous flow value) as the basis for the calculation of the emissions rates (Loftfield *et al.*, 1997; Scholefield *et al.*, 1997).

Table 2. N_2 flux gas rates of a sterilized quarry sand/silica dust mixture at different soil moisture conditions.

Variant (soil moisture)	Water filled pore space (%)	N_2 flux rates ($\mu g\ N_2\text{-}N^*\ m^{-2}{*}h^{-1}$)
Moist	38	0
Wet	74	0
Flooded	100	68
Inundated	100 (water table 2 cm above surface)	61

As clearly follows from the investigations at the inert sand/dust mixture, the (abiotic) N_2 derived from soil air can be completely removed also in case of high soil moisture with the help of the evacuation/flushing cycles (Table 2). Only during complete flooding still another small abiotic N_2 emission is to register. In addition, these disappeared after further 24 hours of the incubation (data not shown). During the following investigations in flooded substrates the measured values were always reduced by the amount of the abiological N_2 release determined here, in order to determine the real denitrification values.

Table 3. Gas flux rates as a function of site and date.

Site/date (month/year)	CO_2 CO_2-C* m^{-2}*h^{-1}	CH_4 CH_4-C* m^{-2}*h^{-1}	N_2O N_2O-N* m^{-2}*h^{-1}	N_2 N_2-N* m^{-2}*h^{-1}	N_2O-N/N_2-N ratio
Fen mire alder forest					
strongly degraded 5/01	–	4.1	1.2	143.2	127.7
weakly degraded 5/01	–	–	–	–	–
strongly degraded 5/02	15103	–1.8	3.3	327.0	98.6
weakly degraded 5/02	18098	–3.2	3.1	1559.9	507.4
strongly degraded 5/03	14191	–1.8	5.3	366.6	58.0
weakly degraded 5/03	23489	8.6	94.9	1056.8	22.3
Riparian alder forest					
river bank 10/01	13046	394.0	38.2	721.3	30.5
alder site 10/01	12934	11.4	159.6	1613.3	16.4
meadow 10/01	19086	0.0	128.1	1347.6	90.9
river bank 10/02	2557	–3.1	1.8	644.7	454.4
alder site 10/02	17547	–4.2	3.5	785.8	886.2
meadow 10/02	6866	–5.6	1.4	369.8	363.5
river bank 7/03	24546	–6.5	1.7	569.4	432
alder site 7/03	14448	1.5	2.0	1455.2	731
meadow 7/03	26941	–6.5	4.3	2153.6	688
Stream bed					
without N pollution	3598	1.7	2.3	493.0	244.4
with N pollution	5651	19.0	49.2	1408.2	53.5

The gas flows measured for the soil samples of the different field sites are generally characterized of extremely high temporal variability (Table 3). That applies also to the N_2O-N to N_2-N relationship. It was very surprising that compared with the nitrous oxide emissions always very many higher N_2 emissions were to be found. However, similar relationships were observed in lab model experiments with reflooded fen substrates, determined by the ^{15}N gas flux method (Augustin *et al.*, 2002). Obviously the N_2 formation via denitrification actually represents the main source for the N-discharge for the examined ecosystems. Moreover, these findings make clear, how important the correct determination of the N_2 fluxes for of realistic N-balances is in case of eutrophic wetlands. Much points on the fact that the variation of the gas emission is not to always due to the effect of an individual factor, separating a whole factor complex. For example, in the context frequently as particularly importantly outstanding groundwater table had hardly an influence on the emissions, as the comparison between the soil samples of the fen and riparian zone with the flooded sediments shows. On the other hand, an increased supply of easily convertible N compounds apart from the denitrification seems to promote as such above all the nitrous oxide formation, clearly visibly becoming at the strongly narrowed N_2O-N to N_2-N relationship. Such conditions were obviously found in the riparian alder forest sites during October 2001, in that weakly degraded fen during July 2003, and in the N polluted sediment. In order to clarify this assumption, special investigations for the N turnover processes in the substrate are necessary. The same applies to clarifying the role of the easily available soil carbon C, since it often plays a key role in regulating kind and intensity of the denitrification process (Luo *et al.*, 1999).

In summary, the He atmosphere incubation technique is a suitable technique for determination actual N_2 and greenhouse gas flux rates from flooded wetland soils. With this method a long term monitoring of actual denitrification rates from different field sites seems to be possible. It is disadvantageous that it is suitable only for the quantification of high N_2 fluxes. Reason for it is the small sensitivity of the TCD detector used. For the future is planned to replace it by substantially more sensitive pulsed He ionisation detector (e.g., Butterbach-Bahl *et al.*, 2002). However, further validation of the method is necessary by simultaneous measuring of the denitrification rates by means of alternative techniques like the ^{15}N gas flux method. For better interpretation and prognosis of the gas flux data it appears urgently necessary to combine emission measurements with C/N turnover process studies in the soil itself.

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Sustainable water management and wastewater purification in tourism facilities

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Abstract

A great number of tourism facilities of various types in different locations throughout Europe do not yet have an adequate sewage treatment system or need to improve the existing one, solving the related problems, e.g. high seasonal fluctuation in the wastewater flow, lack of water, low maintenance capabilities of owners, and natural environment deserving special protection. ˝"SWAMP" – **S**ustainable **Wa**ter **M**anagement and Wastewater **P**urification in Tourism Facilities, a project under the Energy, Environment and Sustainable Development Programme of the 5th Framework Programme of the European Community develops sustainable water management concepts and tests them on concrete examples with partners in Austria, Italy, Latvia and Germany. SWAMP means an efficient water use, recycling of nutrients and a cost effective wastewater treatment by constructed wetlands. Project work is divided up into design of variants of sustainable alternatives at each tourism site, realisation, monitoring and assessment of a practicable variant and contribution to European guidelines and national proposals. Promoting and dissemination of sustainable technologies are made by marketing agencies in each partners country.

Wastewater and sustainable water management in tourism facilities

Water management in tourism facilities is of particular concern throughout the world. In fact, tourism industry is more and more attracted by isolated virgin locations where neither water supply nor wastewater collection is available. A welcomed development beside these aspects is a growing tendency of tourists to consider the environment quality/performance when choosing an accommodation facility. Sensitivity to environmental matters and relating requirements are increasing in all segments of the conventional tourism market.

The main innovation of SWAMP is to consider the wastewater as part of the whole water consumption process. A sustainable water and wastewater management means using water efficiently, avoiding hazardous substances, recycling nutrients, treating wastewater cost-effectively and reclaiming the treated outflow for irrigation purposes.

Objectives of the SWAMP-project

SWAMP aims at developing an economically feasible and technically satisfying wastewater management systems for tourism facilities with high fluctuation. Sustainable water management concepts will be developed and tested on 13 concrete examples in four European countries: Austria, Germany, Italy and Latvia (Table 1). This will be achieved by the following work packages:

WP 1: Audit of each participating facility and development of sustainable wastewater concepts for 13 typical tourism sites with capacities from 50 to 1.200 PE in various climates of Europe

WP 2: Realisation of one practicable variant at each location

WP 3: Monitoring of pilot plants focusing on operation, social acceptance, economical advantages and of innovative sanitation appliances and treatment efficiency of constructed wetlands

WP 4: Contributions to common European guidelines and elaboration of national proposals focusing on technical rule, cost-effectiveness and ecological benefits

WP 5: Promotion and publication of the applied technology by marketing agencies in each partner country

Innovative approaches

The project will advance the state of the art of treating wastewater in combination with constructed wetlands technology by following three innovative approaches.

Least-cost planning

An audit of the water flow is necessary in order to optimise the layout of a treatment facility. An integrated view of used and treated water will lead to technically adapted and cost-effective solutions. Less water consumption will reduce costs for water supply and wastewater treatment. A decrease of wastewater discharge will minimise pollution of wastewater effluents and expenditures for water protection

Table 1. Components of sustainability and techniques in the SWAMP-projects.

	SWAMP-Projects	Save Water			Reclaim Nutrients				Reuse
	Sustainable Water Management and Wastewater Purification in Tourism Facilities	Water saving technologies	Waterless urinals	Vacuum toilets	Separate grey- and blackwater	Use sludge in agriculture	Urine separation	Natural treatment plants	Irrigation
Austria	Klug-Veiti							X	
	Weissmann	X	X					X	
	Fleschwitt							X	
Italy	Fattoria Baggiolino	X						X	X
	Parco del gigante							X	
	La Cava	X			X			X	X
	Relais Certosa							X	X
Germany	Park Moränasee	X				X		X	X
	Burg Lenzen	X	X	X			X		
	Stranddorf Augustenhof	X					X		
	Kulturraum Sammatz	X				X		X	
Latvia	Art museum Pedvāle	X				X		X	
	Mazais Krogs	X				X		X	
	Mežezers	X				X		X	

Treating wastewater as a resource

Normally wastewater is considered as a substance to be disposed of as soon as possible. In this part this attitude has led to expensive and energy intensive treatment plants with a negligible reuse of nutrients. Separation of sewage into its components black water, urine and grey water offers new possibilities for treatment and reclamation of wastewater. Natural treatment plants, e.g. ponds, reed beds, constructed wetlands as an efficient method for wastewater purification have to be adapted to these new challenges. One innovative approach of SWAMP is to implement these techniques and to test them in routine operation.

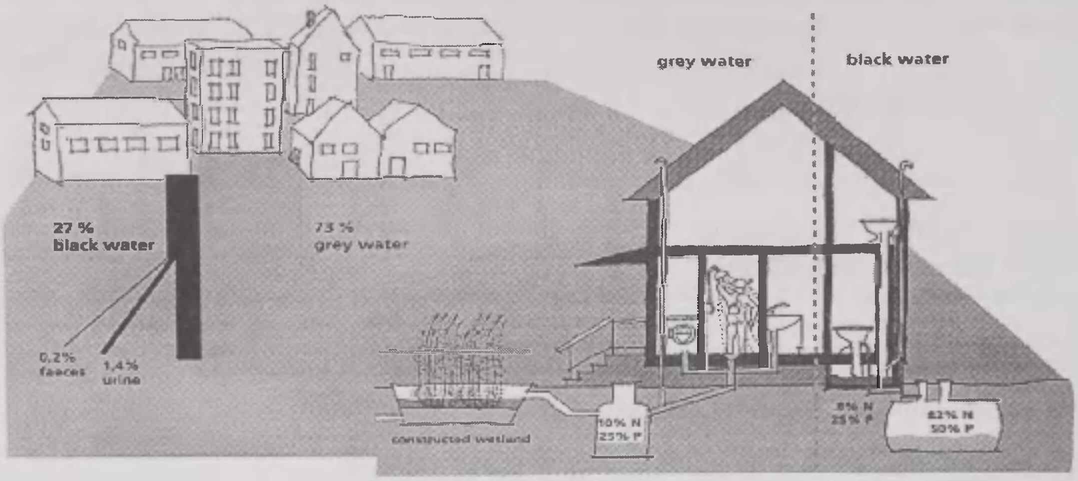

Figure 1. Percentage of Sewage Components of European Households.

New sanitation appliances
Nowadays some special sanitation appliances are available on the market. Water saving and wastewater avoidance by water saving armatures, water flow-restriction, limitation devices for toilet flushing, waterless urinals, vacuum toilets and separation toilets decrease the drinking water consumption. Reduced waste-water quantities lower investment costs and increase the effectiveness of the treatment plants. Vacuum systems in combination with separation toilets allow the separation of faeces (black water) and urine. Urine contains nutrients as N, P, K that can be utilised as fertiliser in agriculture.

There is a need to gather experiences of a broader use of such appliances, with respect to cultural particularities, as water- and sanitation-related hygiene practices are very individual and related to education. A further innovative approach of the SWAMP-project will be to start cautiously but seriously a public discussion on sustainable wastewater management.

Guidelines and scientific objectives

European guidelines and national guidelines for sustainable water and wastewater management and for reed bed treatment systems in tourism facilities for the involved countries are prepared. The overall objective of preparing guidelines implies several other scientific and technological subordinate objectives, as the development of

- a cost-effective water management;
- reclamation concepts for treated wastewater;
- separation technologies in sanitation;
- reed bed treatment systems (or constructed wetlands) for treating wastewater under respect of tourism facilities in remote areas.

The work will be accompanied and completed by marketing agencies in the four countries promoting and disseminating the new technologies.

Appendix 1

SWAMP-partners

AEE Intec, Gleisdorf, Austria
Ambiente Italia s.r.I., Milano, Italy
- Carl Bro Latvia, Riga, Latvia

Ingenieurbüro AWA, Uelzen, Germany
IRIDRA Srl, Firenze, Italy
Ökologisches Projekt, Graz, Austria
Sia Aprite, Cesis, Latvia
Target GmbH, Hannover, Germany

Figure 2. The location of the SWAMP-partners.

Nutrient removal parameters in combined bioreactors and wetland systems

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Abstract

Small plants for domestic wastewater treatment, designed to achieve cost efficient nutrient removal, are described and evaluated. The 6 plants included in the study have a common general layout: a 3-compartment septic tank, an aerated biofilm reactor, sedimentation with sludge return to the septic tank and a final sub-surface wetland treatment. Differences between the plants, such as the biofilm media, process control strategies and the wetland design, are also evaluated. Removal efficiencies obtained include: 96–99% BOD_7, 72–88% COD, 92–96% SS, 80–99% P, and 37–86% N. Approximately 99.9% removal of thermo-tolerant fecal coliform bacteria is typically obtained. Most P is removed in the wetland part of the plants. Most of the nitrogen is removed prior to the wetland. High total nitrogen removal (>80%) is obtained in plants in which aeration and sludge return is optimized for this purpose. Models are applied to generalize the results and design parameters are determined. Phosphate removal coefficients is similar to typical literature values, and the plants are capable of handling high loads. The nitrogen removal capacity in the wetland parts of the plants were significantly lower than typical literature levels, while the plants total nitrogen removal capacity is comparable to other high efficiency natural treatment systems.

Results

Data reported are based on measurements of samples from the inlet to aerated biofilm reactors; from sedimentation tank outlets; and from the outlet from the constructed wetlands; during a test period of 1.5–3 years in 6 Norwegian treatment plants, P1-P6. The hydraulic loading, Q, varied between 200 and 4200 l d^{-1}, at an average around 1000 l d^{-1} (1 m^3 d^{-1}). The mass loading measurements are used to calculate person equivalent, PE, loading. The hydraulic loading on a person equivalent basis is on average ~150 l PE^{-1} d^{-1}. Wash-water from milking machines (dairy farming) is treated along with domestic sewage in plants 3 and 4 causing the mass loading on P3&4 to be significantly higher than the rest.

The removal of pollutants, measured as g d^{-1}, through the bioreactor and sedimentation parts of the plants, are presented as % reduction compared to inlet concentrations in Table 1 (number of samples for each value; $14 < n < 23$). The results are presented as the average of pairs of plants of similar design. Standard deviation for the samples are also presented. Total % reduction of the various pollutants, measured as g d^{-1}, through the bioreactors, sedimentation and wetland (number of samples for each value; $14 < n < 23$), is presented in Table 2. The results are presented as the average of the same pairs of plants of similar design as above.

All plants remove >96% BOD, with consistently 99% removal in P3&4 plants (effluent concentrations < 4 mg l^{-1}). The removal in the reactor part of the plants vary more, with 83% in P5, while P3&4 remove 99%. P3&4 also remove COD more efficiently (90%) and consistently compared to P1&2, which remove 78%, while P5&6 remove 62%. All COD removal in P1–4 is obtained in the reactor part while approx. 10% of the COD removal in plants P5&6 occur in the constructed wetlands.

Total nitrogen removal varied from >80% in P3&4 to less than 40% in P1&2. Most of the nitrogen was removed in the reactor part of the processes and 18–35% in the wetlands. A removal efficiency of 75–94% for ammonia was measured, of which just a few percent occurred in the wetlands. The operation of the aeration equipment greatly affected the nitrogen removal rates.

Total phosphorous removal was consistently well above 80%, of which 92%, 62% and 55% occurred in the wetlands in P5&6, P3&4 and P1&2, respectively. A slight reduction with time in P removal in the wetlands is observed. Suspended solids, SS, removal of 93–95% is measured in all plants, of which >80% occurred in the reactor parts of P1–4 while most SS removal occurred in the wetlands in P5&6.

Around 99.9% of the thermo-tolerant coliform organisms (fecal coliform; FC) were typically removed in these plants, yielding an average effluent quality close to the Norwegian “recreational (swimming) water quality standard” of maximum 100 FC/100ml.

Modelling

A further evaluation of the treatment wetlands is based on the “k – C*” mass balance design method of Kadlec and Knight (1996) for both surface and subsurface flow constructed wetlands. This model with recommended parameter values is applied to compare the performance of the tertiary treatment wetlands in this study to more conventional wetland systems reported in the literature.

Phosphorus

Literature indicates that phosphorus removal efficiency is strongly dependent on loading rate with 65 to 95% removal at loading rate less than ~0.14 kg ha^{-1} d^{-1}, while removal efficiency decreases to 30 to 40% or less when phosphorus loadings are greater than ~0.3 kg ha^{-1} d^{-1} (Faulkner and Richardson, 1989). The results from the case study in this chapter suggest that such load limitations as observed by Faulker and Richardon are not valid in cases with advanced treatment prior to the wetland. The relationship between mass loading and removal rates of phosphorus is represented in Figure 1. Approximately 90% P removal was consistently obtained in all plants, at loads up to 10 times the level for which Faulker and Richardson (1989) maintained high removal. It is therefore concluded that much smaller and more efficient wetlands can be applied for P removal following advanced secondary treatment.

The data from the case study is compared to the "k – C*" model in Figure 2. The ratio of the plant areas divided by Q, i.e. specific area load, versus removal of phosphorus is represented in Figure 2. The data fit quite well with the model applying the coefficients suggested by Kadlec and Knight (1996) for phosphorus removal. According to this, it can be supposed that the coefficients for the case of phosphorus, are also applicable for tertiary treatment wetlands with high loads.

Nitrogen

Data from 200 working treatment wetland systems in North America indicates that nitrogen removal efficiencies decline at mass loading rates above 20 kg ha^{-1} d^{-1}. Total nitrogen removal up to 79% are reported at higher nitrogen loading rates of up to 44 kg ha^{-1} d^{-1} from some systems in US (Watson *et al.*, 1989). Data from the case study show that the mass loads and the removal rates are much lower than the above mentioned US data. The mass loading values for the plants are in the range proposed by Knight *et al.* (1993), i.e., 10 kg ha^{-1} d^{-1} for mass loading rate, while the removal rates are significantly lower than the 5 kg ha^{-1} d^{-1} observed by Knight *et al.* (1993). The discrepancy between the literature data and the results from the wetland preceded by advanced secondary treatment can be partly explained by the fact that a large portion of the nitrogen in the wastewater has already been removed in the bioreactors. A closer look at the data, however, reveal that the plants (P1 and P2) with the lowest nitrogen removal in the wetland also has the lowest removal in the bioreactors, suggesting that the more advanced control in reactors P3–6 compared to P1-2 is also beneficial for the downstream wetland process.

The data from the case study is compared to the "k – C*" model in Figure 3. The ratio of the plant areas divided by Q, i.e. specific area load, versus removal of nitrogen is presented in this figure. It can be observed that the curve that represents the model using k=27, as recommended by Kadlec and Knight (1996) does not correspond to the evaluated plants in this chapters case study (represented as P1, P2, ... P6 in the Figure 3). The simulations using the recommended value for total nitrogen for the Czech SSF systems (Kadlec and Knight, 1996), i.e. k= 10.2 m yr^{-1} are also significantly different from the data. The removal rates of nitrogen in the tertiary wastewater treatment constructed wetlands case is reasonably modelled using k=2.

The total nitrogen removed in the best plants (most advanced process control) used as case studies in this chapter, including primary, secondary and tertiary treatment, exceeds that which, according to Kadlec and Knight (1996), is obtainable in conventional natural treatment processes at similar loads. The case study results are also comparable to fully manned complex activated sludge nutrient removal plants. It is therefore concluded that combined plants presented in this paper is an efficient nitrogen removal concept requiring low cost, maintenance and operational resources.

Table 1. Removal efficiency (%) and standard deviation (SD) of pollutants in the bioreactor and sedimentation.

	P1&2		P3&4		P5&6	
	%	SD	%	SD	%	SD
BOD_7	95.5	1.5	99	0	87.5	4.5
COD	78.5	6.5	90.5	0.5	62.5	3.5
Tot-N	29	5	68	0	40	6
NH_4	74.5	12.5	93	1	72	6
Tot-P	39	17	36.5	2.5	7	11
SS	78.5	6.5	77.5	3.5	29	7

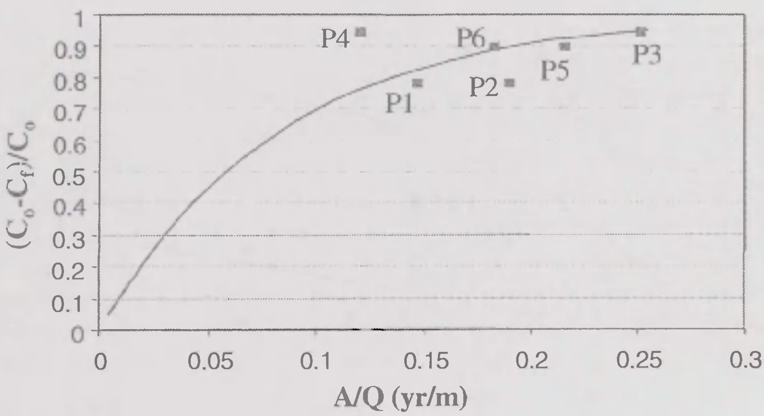

Figure 1. Phosphorus removal and mass loading rates in six wetlands (Case P1-P6).

Table 2. Removal efficiency (%) and standard deviation (SD) of pollutants in the total treatment system.

	P1&2		P3&4		P5&6	
	%	SD	%	SD	%	SD
BOD_7	97.5	1.5	98.5	0.5	97.5	0.5
COD	80.5	7.5	87.5	0.5	73	1
Tot-N	38	1	83.5	2.5	61	3
NH_4	81.5	0.5	94	2	75.5	8.5
Tot-P	86.5	6.5	96.5	2.5	90	4
SS	94.5	1.5	95	0	93	1

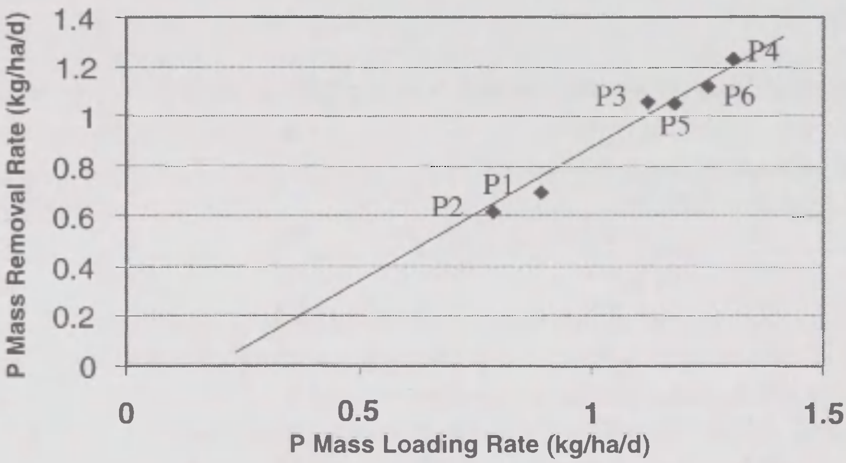

Figure 2. Removal of phosphorus and specific area load (A Q^{-1}), modelled and measured.

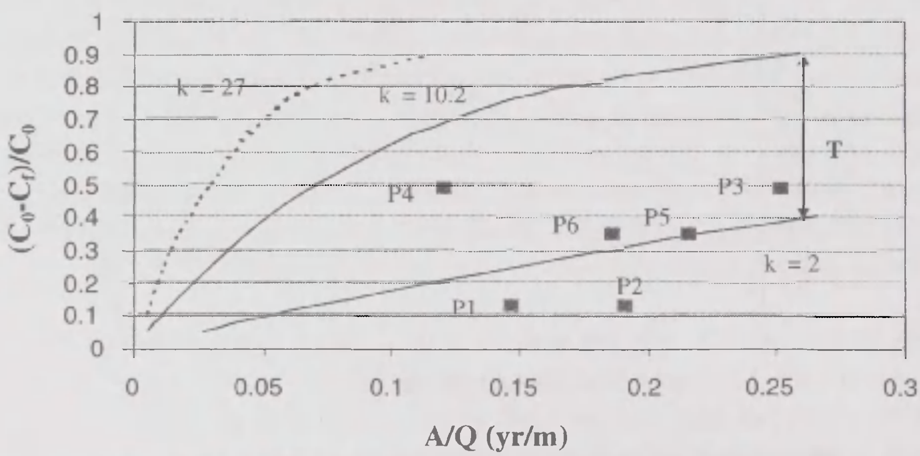

Figure 3. Removal of phosphorus and specific area load (A/Q), modelled and measured.

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Biochemical degradation involved in the removal of organic matter in horizontal subsurface flow constructed wetlands

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Abstract

This study presents an evaluation of the effects of hydraulic loading rate (HLR), aspect ratio, size of granular medium and water depth on the performance of horizontal Subsurface Flow Constructed Wetlands (SSFCWs), moreover, the biochemical reactions involved in the removal of organic matter are also evaluated. Experiments have been carried out in a pilot plant that treats treats in part the urban wastewater of the housing development of Can Corró (200 PE), Les Franqueses del Vallès (Barcelona, northeastern Spain). The climate at the site is Mediterranean with an average annual temperature and annual accumulated rain of 13.5°C and 460 mm respectively.

The pilot plant is constituted by 8 parallel horizontal SSFCW (Figure 1), all the beds have approximately the same surface area (54–56 m^2) and their aspect ratio (length to width) varies in pairs. The two of the type named A have an aspect ratio of 1:1, B of 1.5:1, C of 2:1 and D 2.5:1. Furthermore, the size of the granular medium inside of each pair also varies. Thus, beds of type 1 contain a coarse granitic gravel (D_{60}=10 mm, C_u=1.6) while type 2 contain fine granitic gravel (D_{60}=3.5 mm, C_u=1.7). Beds of the type A, B and C have a slope that ranges from 0 to 1%, while the slope in beds of the type D is approximately 2.5%. Differences in slope are due to construction matters. Furthermore, beds of the type D were constructed shallower in order to assess the effect of water depth on treatment efficiency. The water level has been adjusted during experimentation to 0.05 m under the gravel surface in all beds, resulting in an average water depth of approximately 0.5 m for types A, B, and C, and of 0.27 m for type D. Taking into account the surface area, water depth and porosity, the nominal volume of the beds is approximately 10.4 m^3 for types A, B and C, and 6.0 m^3 for type D. All beds have 3 perforated tubes (0.1 m in diameter) inserted in the middle part of the gravel and uniformly distributed throughout the length of the bed, which allow to obtain intermediate samples and to check the water level.

The system was designed to meet secondary effluent water quality requirements according to European Directive 91/271 (Council of the European Communities, 1991) and the design HLR was 45 mm d^{-1}. However, at the beginning of the operation of the beds, the system was not able to meet the requirements with this HLR, so we selected lower HLRs for experimentation. For this study, three different total HLRs were used: 20, 36 and 45 mm d^{-1}. Sampling campaigns were conducted approximately on a weekly basis from June 2001 to January 2002. A total of 59 campaigns were carried out, of which 21 correspond to an HLR of 20 mm d^{-1}, 22 to 36 mm d^{-1}, and 16 to 45 mm d^{-1}.

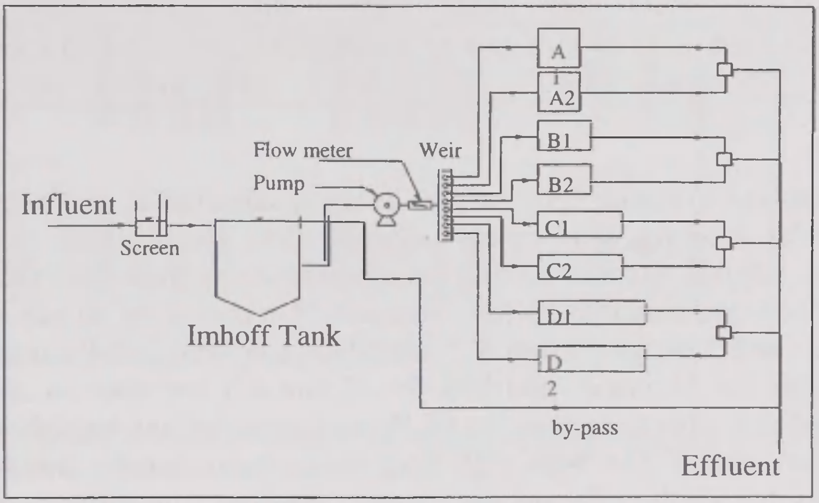

Figure 1. Schematic diagram of the HFRB pilot plant at the Can Suquet housing development, Les Franqueses del Vallès, Barcelona.

Average COD and BOD_5 removal ranged between 50 and 85% and depending on the bed (Table 1). Beds A, B and C had very similar average COD and BOD_5 removals, ranging between 58 and 61% and 51 and 56% respectively. Beds of type D had the higher average COD and BOD_5 removals, particularly bed D2. In fact, only bed D2 showed statistical differences from the averages of the all other beds ($p<0.05$) in the effluent COD and BOD_5 concentrations when a pairwise comparison between beds was carried out.

Table 1. Overall averages and standard deviations of the water quality parameters in the influent and effluents of all the beds. Average percentage removals are shown in parenthesis. (a) means there was no removal.

Parameter	Influent	A1	A2	B1	B2	CI	C2	D1	D2
COD, mg l^{-1}	239 ± 87	97 ± 36 (58)	90 ± 34 (61)	99 ± 36 (57)	87 ± 31 (62)	93 ± 32 (61)	87 ± 32 (61)	67 ± 29 (71)	46 ± 24 (80)
BOD_5, mg l^{-1}	143 ± 48	67 ± 30 (54)	66 ± 26 (53)	70 ± 29 (51)	61 ± 27 (56)	66 ± 30 (54)	66 ± 28 (53)	40 ± 23 (73)	21 ± 19 (86)
NH_3, mg N l^{-1}	55 ± 12	44 ± 11 (19)	42 ± 9.7 (22)	45 ± 10 (26)	40 ± 10 (26)	43 ± 10 (21)	39 ± 10 (27)	36 ± 107 (35)	24 ± 12 (58)
DRP, mg P l^{-1}	9.5 ± 3.0	9.5 ± 3.1 (a)	9.4 ± 2.5 (a)	9.4 ± 2.7 (a)	9.2 ± 2.3 (a)	9.2 ± 2.3 (a)	8.8 ± 2.2 (a)	8.7 ± 2.8 (7)	7.3 ± 2.4 (19)

As can be observed in Figure 2, of the four factors studied (HLR, aspect ratio, size of the granular medium and depth), only the first and the last yield clear differences in effluent concentrations of the contaminants measured. HLR has a significant effect on effluent COD (on average 67% removal for 20 mm d^{-1}, 62% for 36 mm d^{-1} and 57% for 45 mm d^{-1}) and BOD_5 (on average 64% removal for 20 mm d^{-1}, 58% for 36 mm d^{-1} and 54% for 45 mm d^{-1}), but none (or an unclear effect) on nutrients. This is because the HLRs used were not low enough to reduce nutrients to any extent. The beds with finer media have slightly lower effluent concentrations for all the contaminants tested. However, the differences are in general small (on average 83 and 71 mg l^{-1} for COD, 56 and 50 mg l^{-1} for BOD_5, 38 and 33 mg N l^{-1} for ammonia, and 8.4 and 7.9 mg P l^{-1} of ortophosphate for the coarse and the fine media respectively).

The results of this study indicate that water depth is a very important factor for contaminant removal in horizontal SSFCWs. As can be seen in Figure 2, for all contaminants tested the shallower beds (type D) produced systematically lower effluent concentrations. In these beds it was found that denitrification was responsible for 20 to 30% of organic matter removal, but this process was not apparently present in the other beds. The amount of organic matter removed by sulfate reduction (Figure 3) was higher in beds of types A, B and C (10–40% of the total organic matter) than in beds of type D (10–20%).

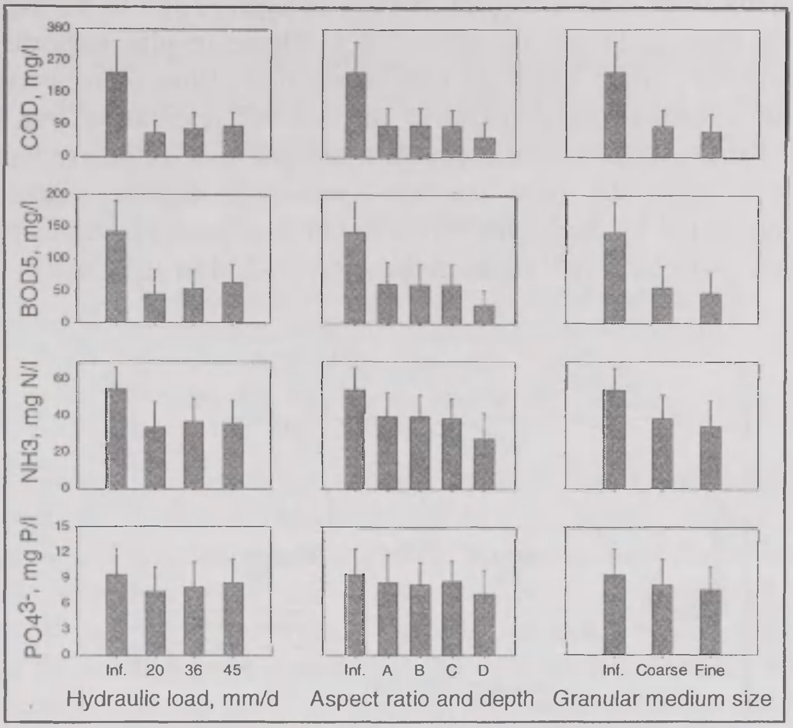

Figure 2. Average and standard deviation of the influent and effluent's COD, BOD_5, ammonia and DRP concentrations grouped according to the parameters tested.

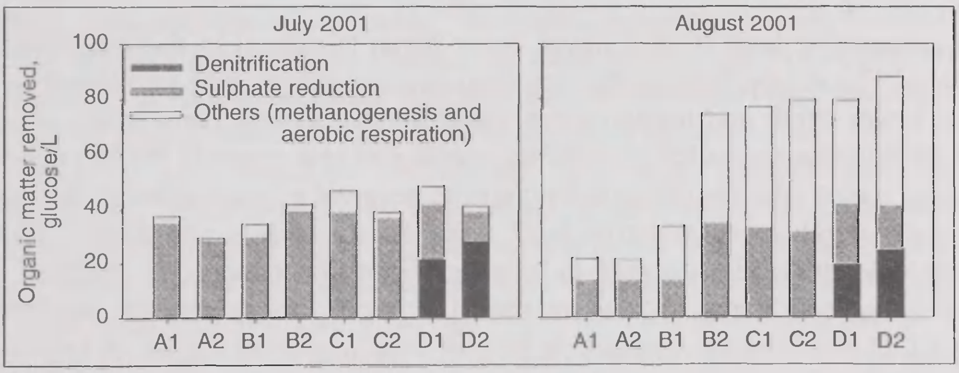

Figure 3. Estimated amount of soluble COD removed by different biochemical pathways (denitrification, sulphate reduction, methanogenesis and aerobic respiration) in 2 campaig in each bed for the campaigns of July and August of 2001.

Methane emissions were clearly higher in beds of type A (15 to 47 mg $m^{-2}.d^{-1}$) than in beds of type D (1 to 11 $mg/m^{-2}.d^{-1}$). These results indicate that in shallower beds prevail more oxidized conditions that allow more energetically favorable reactions such as denitrification and aerobic respiration (we have no estimate for the former reaction). In contrast, in deeper beds sulfate reduction and fermentation are almost the only reactions removing organic matter. These statements are supported by the higher redox potential (Figure 4) values measured in D-type beds (–338 to –351 mV) than in the others (–358 to –390 mV).

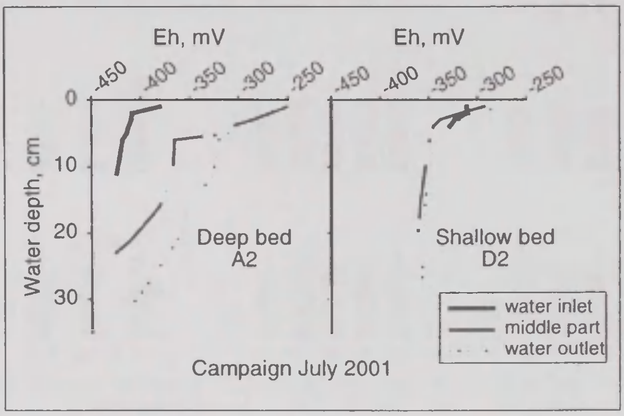

Figure 4. E_H profiles obtained from the perforated tubes inserted in beds A2 and D2 in July 2001 campaign. Note that the profiles do not reach the bottom of the beds due to mechanical problems with the arm used to introduce the E_H probe.

Higher removal in beds with a lower water depth is related to the less reducing conditions. The reason behind the less reducing conditions may be related to the fact that lower depth and higher water speed (the same flow for a lower perpendicular section) increases the re-aeration coefficient that controls the oxygen flux from air to water. The results indicate that the removal of contaminants decreases as the water depth increases from 0.27 m to 0.5 m. In July and August (2001) campaigns, denitrification seemed to be a significant mechanism for soluble COD removal in type D beds, but not in the other beds. Moreover, the amount of soluble COD removed by sulphate reduction was in general higher in type A, B and C beds than in type D beds. Thus, the higher E_H values found in type D beds are related to the fact that the mechanisms that remove soluble COD are different or have a different relative importance.

The behaviour of electron acceptors and the estimation of the relative importance of the biochemical pathways suggest that, as the water depth decreases, more varied reactions occur. In fact, denitrification was estimated to be a significant reaction for the removal of organic matter in shallower beds, while it did not occur in deeper beds.

Empirical evaluation of modelling parameters in performance assessment of wetland systems for catchment scale pollutant management

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A comparison of two of the regionally most widely used options for pollutant management in stormwater was assessed in a study by the Centre for Water and Environmental Technology in 1997/1999. The performance of a linear constructed wetland at Plumpton Park and a detention pond at Woodcroft Estate for the reduction of stormwater pollutant loads to receiving waters was assessed. The results of these studies indicated that the constructed wetland was significantly more effective than the pond in retaining nutrients, sediment and bacteria. These and several other studies of the sites, however, primarily utilised routine-interval pollutant concentration measurements only. Further, hydraulic flow estimates, where considered at all, were based on catchment scale assumptions with no direct measurement of flow.

In this current study, flow monitoring was carried out using water depth and velocity measurement to calculate flow and autosamplers were used to take discrete samples during storm events. The more direct flow monitoring and storm event related pollutant sampling was used to evaluate modelling assumptions used in previous assessment of performance of the wetland system.

Plumpton and Woodcroft Estate are two recently established residential developments approximately 40 km northwest of Sydney, which produce large volumes of stormwater with high suspended solids and nutrient concentrations during storm events. The 0.45 ha constructed wetland system at Plumpton Park was completed in 1994 within the existing 75 ha residential catchment. The soil landscape for each of the systems is typified by hard setting clays that are slightly saline and acidic with occurrences of soil which has a high potential for erosion along the watercourses. Both catchments are substantially urbanised and are reasonably stable, despite the high sediment loads. Both the Plumpton Park and Woodcroft Estate constructed wetlands provide amenity and a recreational facilities for the local community, as well as demonstrating a structural series treatment train with in-line gross pollutant traps, sediment traps and constructed wetland components. The sites are typical of urban developments in the region, with peri-urban siting of housing estates often exacerbating problems of both point source and diffuse source pollutant management.

Stormwater from Plumpton Park flows into Bells Creek and then into Eastern Creek, which in turn joins South Creek and flows into the Hawkesbury River. In the upper reaches Bells Creek has been converted into a formal drainage swale and low flow pipe but the lower reaches have not been modified and flow through rural land. The extent of formal channels and efficient stormwater conveyance systems in the upper reaches tend to increases the "shock load" of pollutants on the river system, which is already degraded and prone to algal blooms.

Water quality objectives: have been identified as needing to meet the basic criteria for secondary contact recreation, with primary contact a long-term objective. These objectives require improved understanding to better manage the pollutant inputs of increasing volumes of both stormwater runoff and sewer overflow inputs to regional surface waters.

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Factors for the spreading of constructed wetlands in developing countries: a strategy for research and action

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Introduction

One of the major problems facing developing countries today is the lack of reliable wastewater treatment systems. However, simply importing modern sewage systems from developed countries is often unrealistic, going beyond the limited capabilities of developing countries. Other, more appropriate solutions must be found for these parts of the world.

Constructed wetlands (CWs) are innovative wastewater treatment systems based on a natural process, presenting developing countries with a realistic alternative to conventional wastewater treatment technologies.

However, progress in the implementation of CWs has been slow at best. A survey of experts has been conducted, people with practical experience in the planning or construction of such facilities in developing countries, to identify possible reasons for the delay of implementation.

Based on an evaluation of relevant literature, a questionnaire was developed and sent to 449 persons and institutions.

Results

Thirty-eight of the questionnaires returned were taken into account in the qualitative evaluation. The experts who took part in the survey had been involved with the subject matter for an average of just under five years. The study drew on a base of experience comprising 171 successfully concluded projects in 27 countries.

Evaluation revealed a striking correlation between the answers given and the origin of the questioned expert. In many ways, those from developing countries had substantially different experiences than those from industrialized countries.

"Northern researchers" cooperate with other partners, are faced with other difficulties during implementation, and have other criteria for the construction of CW than their colleagues from developing countries. This realization underscores the importance of the so-called "soft factors" in technology transfer. If these are not taken into account, substantial frictional loss can be the result.

The experts surveyed and the relevant specialist authors are fundamentally in agreement that Constructed Wetlands are essentially suited to the infrastructure of developing countries.

From our analysis of data from the survey, we know that:

- Horizontal filters are the most commonly used form of CW systems.
- The main treatment goal is the reduction of organic loading, followed by the removal of pathogens.
- Most of the experts surveyed agreed that CWs in developing countries have not gone past the state of research plants.
- Construction of CWs is not initiated by nations, developed or developing, but by committed individuals (the experts who took part in the survey).
- The state plays a key role in developing countries. Most commonly, the government or an authority decide whether a CW gets built or not. Followed by the state the local population, NGOs or universities are considered as decision-makers.
- The most important criteria in favour of a CW, in order of descending importance, are: simple operation and maintenance, system operates without electricity, simple construction, cost effectiveness, no need for highly qualified personnel for operation, and use of locally available materials.
- Lack of awareness is the most common reason for deciding against the implementation of a CW.
- Implementation usually takes place in cooperation with a local university.
- The most crucial difficulties during implementation are financial difficulties, lack of awareness, no sewerage system, high demand of land area and carelessness in operation and maintenance.
- The first installed CW has a substantial multiplier effect in that particular country.
- Plants harvested from CW are used, if at all, primarily as building material or animal fodder.
- The selection of plants suited to that purpose is currently very limited.
- The treated wastewater is still seldom used for agricultural irrigation.
- Most experts surveyed were of the opinion that a financial involvement of the local communities would lead to improved care of the CWs.

Recommendations for improving the implementation of CWs

In addition to the general guidelines appropriate to all forms of technology transfer, we suggest certain recommendations for action in the specific case of CWs, based on the results of our survey.

Consolidation of baseline informations and information exchange
As a basis of information for future operations, a competent, **coordinating committee** (based on the example of such groups as the IWA "Specialist Group on Use of Macrophytes in Water Pollution Control") should collect, evaluate and publish data on CWs in developing countries worldwide.
A publicly accessible **internet website** dedicated to CWs would be an excellent idea (the website of the above mentioned IWA Specialist Group furnishes a good example of the possibilities). This would provide a platform for the rapid spread of information and intensified exchange of views among all those involved in the area of CWs.

The internet, however, cannot replace personal contact between the individuals involved. For this reason, the offering of **international conferences** on the subject must be expanded. These should take place in developing countries where at all possible.

Acceptance of the decision-makers
Demonstration sites have a strong multiplication effect in developing countries and should be pushed for this reason. The decision-makers would gain insight into the function of CWs and presumably be more open to this technology.

For the local population, the **use of harvested plants** could be a decisive stimulation for the implementation of a CW. In addition, this could provide motivation for the appropriate maintenance of the facilities. There is a need for research to identify appropriate plant varieties and explore their marketing potential.

The **financial involvement** of users in the cost of investment leads to better maintenance of the CWs in the long term. The cost of construction should only be partially covered by development aid. The rest would be provided by the users themselves (backed up if necessary by small loans). Running costs for operation and maintenance could be covered from the proceeds from the sale of harvested plants.

The purified wastewater should be used for **irrigation in agriculture**. The use of this water, thanks to its substantial hygienification, is quite safe for farmers. With the appropriate procedure, remaining nutrients in the wastewater can be utilized as an alternative to costly artificial plant fertilizer.

Development cooperation

The development of special information campaigns and **training programs** for senior administrators and other public officials must be given highest priority. For the long-term support of this technology, intense lobbying of the appropriate ministries and authorities will be necessary.

Intensified **communication** in all directions (north-south, south-south, south-north, as well as north-north) is indispensable!

Our study has shown that CWs are quite well suited to the structures of developing countries. Nevertheless, neglect of complex socio-economic factors can lead to substantial difficulties in the process of implementation.

In order to encourage the trend toward more CWs in developing countries, the course of action recommended here should be actualised in cooperation with all the persons and institutions involved.

New directions in the development of natural systems for sewage purification

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Introduction

The new water law for all states of the EU, passed in 2000 by the European Parliament, is the result of concern about the present and future state of water in Europe. Simultaneously, proper water management must be based on well-considered and modern sewage management. The analysis of existing problems of sewage management in rural areas and the protection of ecologically-valuable terrain shows that this should determine the directions of development of sewage management for these regions. Natural systems of sewage purification, with appropriate supply technology and construction, may be the solution. They have much better ecological effects than conventional sewage treatment facilities.

The former facilities have already been in use for a long time and numerous solutions and ideas connected with sewage flow and purification technology have been applied to them.

Below are presented 4 different natural systems of sewage purification.

Pond treatment plants

Correctly designed artificial ponds with a surface which does not exceed 15 m^2 PE^{-1} were the first and simplest systems based on biological purification with heterotrophic micro-organisms, water plants and hydrophilous plants. Mechanical purification, realized with the detritus tank, for example, should be a supplementary element of this system.

These sewage treatment facilities use the natural self-cleaning processes which take place in the aquatic environment and, under suitable conditions, cause the development and growth of hydrophytes. This fosters oxygenation and reduction processes, which together with processes of sorption, sedimentation and assimilation, allow the removal of much pollution from the environment, if properly realized.

Such facilities may consist of a sediment tank used for mechanical-biological purification and possessing a capacity of about 1 m^3 PE^{-1}. It may also consist of a pond with a surface of about 10 m^2 PE^{-1} with banks consolidated and planted with hydrophilous flora that is isolated from the soil with a geo-membrane.

Supervisory sewage may also be used. Water that has already been treated goes through it to reach the reservoir.

Treatment plants with horizontal flow of sewage

These facilities are based on the experience obtained while building and using pond treatment plants. Therefore, in order to increase the efficiency of purification, they are planted with hydrophilous flora, usually reeds.

In these plants, water level may be above ground level and may flow above the sedimentary layer. Transverse dams or serpentine-shaped ditches are built in such facilities to control the flow, which is often distorted by plants themselves.

These facilities also require mechanical extra-purification as well as the plugging of subsoil and slopes with foil. The flow is aided by the designed gradient of the bottom (up to 0.5%) and the adjustment of the level of sewage outflow.

Another trend in the construction of such plants was, apart from the planting of aquatic vegetation, the filling of the pond with a suitable material (e.g. sand or gravel) of a high coefficient of hydraulic conductivity.

The exploitation of such facilities should also be combined with preliminary extra-purification in a settling tank with a capacity of about 1.5 m^3 PE^{-1} and plugging of the subsoil with the help of a geo-membrane. An individual filter-lode planted with uliginous vegetation, on the other hand, has a surface of 10 m^2 PE^{-1}. Unfortunately, the fundamental disadvantages of such a facility are the hydraulic problems connected with clogging.

Sewage plants with perpendicular sewage flow

The next type of sewage purification after the mechanical extra-purification in settling tank are aquatic plants with the perpendicular sewage flow.

This kind requires a pool with plugged subsoil and slopes that is filled with a filter material consisting of several layers. Sewage is periodically provided, which requires the use of an intermediate pumping station. The sewage from the settling tank, reaches the intermediate pumping station and, then, a strainer core where they are distributed over the surface by a system of pipes. Later, the sewage swims vertically down through the layer which is overgrown with reed .The sewage flows out into the control basin due to the help of collecting drainage laid on the bottom of the strainer core.

The advantage of the facilities is not only effective purification, but also smaller surface, ranging (depending on needs) from 3 m^2 PE^{-1} to 6 m^2 PE^{-1}.

Vegetation pond sewage plants (the Polish system)

These systems are the most recent of all. They combine the best features of the pond and aquatic plants.

The applied technology is based on multistage mechanical and biological purification processes. These processes take place in the settling tank, the vegetation strainer core and the denitrification pond.

As usual, the first stage of purification is the settling tank, which has a capacity of about 0.5 m^3 PE^{-1} and in which mechanical-biological purification takes place. Then sewage is directed to the vegetation strainer core, with a surface of about 2 m^2 PE^{-1}. Perpendicular sewage flow is used, which is similar to the aquatic plants. Nevertheless, this differs in terms of filling material and vegetation.

Having passed through the strainer core, the sewage flows into the denitrification pond for a surface of about 2–3 m^2 PE^{-1}, in which it is necessary to use supplementary purification. Eventually the sewage passes through a control basin to a receiver set.

Both the pond and the strainer core are planted with hydrophilous vegetation.

The essential component for purification is the vegetation strainer core in which organic compounds are reduced, ammonium nitrogen is nitrified, partial denitrification occurs, phosphorus and pathogenic organisms are removed and heavy metals are retained. Apart from its small surface and relatively low exploitation costs (in comparison to conventional facilities), the big advantage of this type is a high level of elimination of organic substance and biogenic compounds. The research, which continued over a period of several years, yielded very satisfactory results.

Below is a presentation of the reduction of organic substances in reference to BOD_5 (Figure 1), the reduction of general nitrogen (Figure 2) and phosphates (Figure 3).

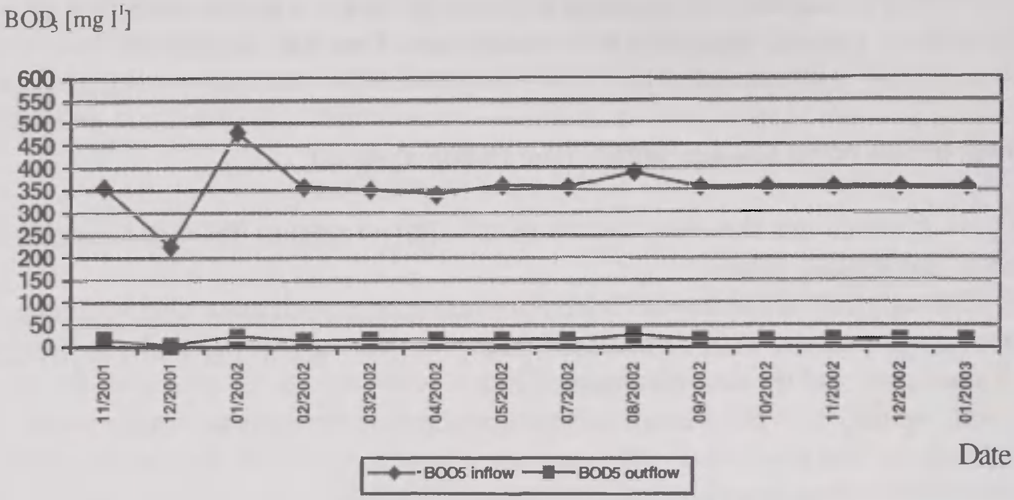

Figure 1. The reduction of organic substance in reference to BOD_5.

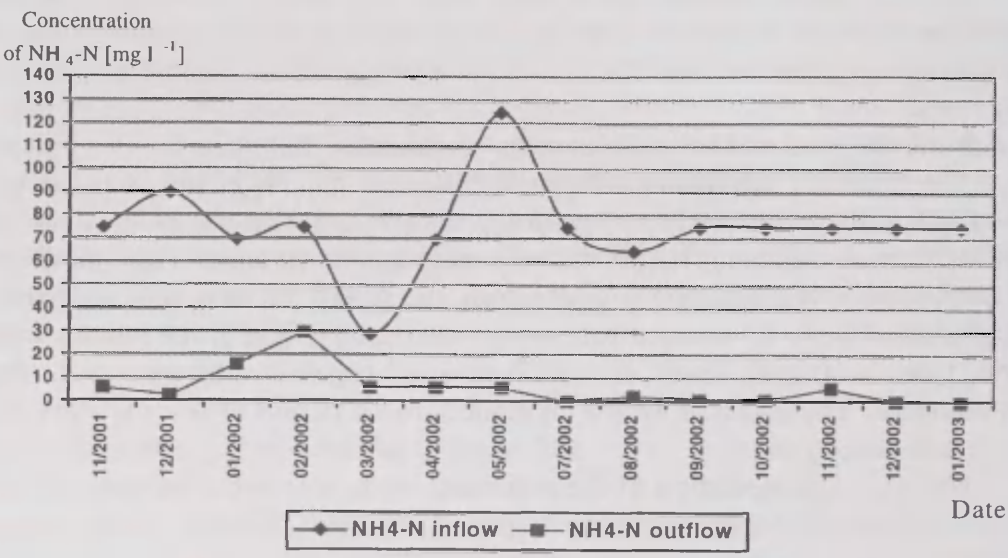

Figure 2. The reduction of ammonia nitrogen.

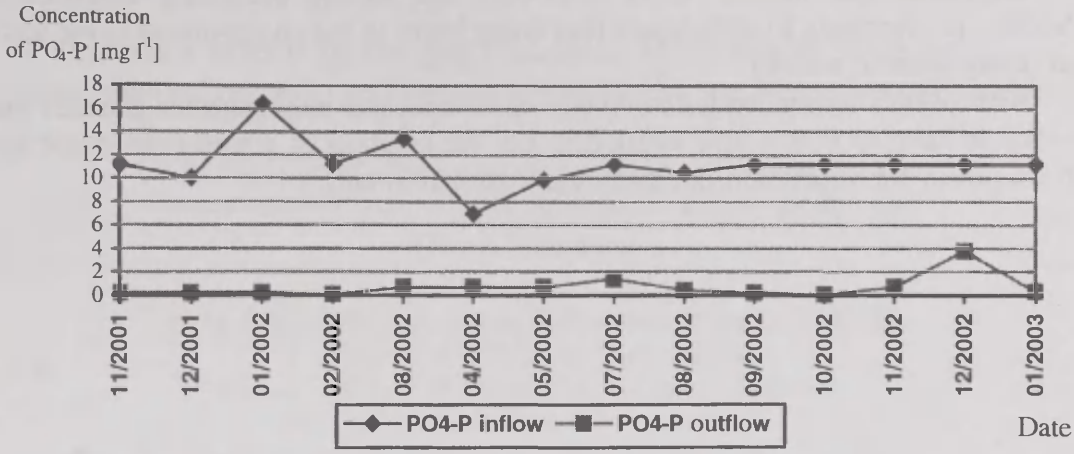

Figure 3. The reduction of phosphate phosphorus.

The mentioned results prove that the vegetation-pond (the Polish type), with the elimination of organic substances in reference to BOD_5 at a level of 95%, general nitrogen at 78% and phosphates at 94% may be the kind that will soon become the new standard in sewage purification in the countryside and in ecologically valuable areas.

Conclusions

Vegetation-pond sewage plants, the latest achievement in environment engineering, point a new direction in the development of natural systems for sewage purification. They combine many advantages i.e. high effectiveness in the elimination of organic substances and biogenic compounds, low cost, the possibility of utilizing of treated water (e.g. to irrigate fields) and increasingly widespread acceptance in society.

Aquatic plants are versatile: they are used not only to treat sewage from farms and homes but also sewage of industrial origin, rainfall or even concentration silage liquid. It is very important that it be possible to apply these structures successfully as elements of the supplementary purification of treated water coming from a conventional plant, at the same time increasing the value of this water and limiting the possibility of the receivers' eutrophication.

Natural treatment plants generate increasing interest, not only among experts, but also among potential users. The social consciousness of the need to preserve the environment will increase in parallel with the quantity of built objects.

Drainless and usually leaky reservoirs are slowly becoming insufficient. Society is beginning to understand that doing harm to the environment is the same as doing harm to oneself.

This is the reason why ecologists, environmental engineers, authorities and officials have to face a new challenge, i.e. the creation of a well-considered and fresh policy for vegetation-pond sewage treatment plants.

Factors affecting the purification efficiency of wetlands constructed on minerothrophic peatlands in Northern Europe

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Abstract

It has been possible to decrease effectively the loading of suspended solids (SS), N, P, Fe and organic matter imposed to watercourses by means of wetlands constructed on minerotrophic peatlands in northern Finland. The method, which involves conducting wastewater across an intact mire surface surrounded by ditches, is widely used in Finland to decrease the loading from peat mining areas. Modifications of the method have also been used in forestry and in the purification of household wastewaters.

The present instructions for the construction of wetlands on minerotrophic peatland areas are based mainly on the properties of the Kompsasuo wetland. This model wetland of the method is situated in northern Finland in the minerotrophic peatland area surrounded by coniferous forests of the mid-boreal type. The size of the wetland is approx. 2.4 ha, consisting 4.8% of the catchment area. The wetland is a pine mire, and the typical vegetation at the present includes *Menyanthes trifoliata*, *Carex sp.* and *Potentilla palustris*. The prevailing peat type of the wetland is *Sphagnum-Carex* peat. The wetland was constructed in 1987 for the purification of the drainage waters of the Kompsasuo peat mining area. Waters from the peat mining area flow first to a sedimentation basin and thereafter via an outlet ditch to a distribution ditch located in the upper part of the wetland. Waters from the wetland flow via collection ditches to an outlet ditch below the area. So far the wetland has been in use for 16 years.

The purification efficiency of the Kompsasuo wetland has been monitored during the time of its usage. All the time the efficiency has been high. In 1996, when the wetland had been in use for ten years, the reductions achieved were 29% for SS, 11% for organic matter, 60% for tot. N, 84% for inorganic N, 60% for Total P, 74% for PO_4-P and 59% for Total Fe. The data on the purification efficiencies is based on the decreases in pollutant amounts between the inflow and outflow waters. All the time there has been continuous measurement of hydraulic loading to the area. In 2001–2002 the purification efficiencies have been studied in the Primrose project.

The purification of water in the wetland is the result of many physical, chemical and biological processes. The most important processes leading to nutrient reductions are PO_4-P and NH_4-N sorption to peat-soil, and denitrification. According to the present knowledge also denitrification process demands solid living base for bacteria. Thus denitrification bacteria can be supposed to live mainly on the surfaces of soil particles. The peat-soil of the wetland seems thus to have an important role in water purification in many ways. It is known that also vascular plants prefer nutrients bound on soil in assimilation.

Phosphate sorption onto peat-soil has been studied by conventional adsorption isotherms. The ability of peat-soil to retain PO4-P increases with its increasing Fe content. It can be supposed that the peat-soil of the wetland could gradually be saturated with PO4-P, because only a small amount of PO4-P adsorbed on the peat is assimilated by the plants growing in the area. The ability of peat-soil to retain $NH4^+$ has been estimated using the values of cation exchange capacity. As compared with the actual decreases in inorganic N achieved in the wetland, the ability of peat-soil to retain $NH4^+$ is small. This indicates that nitrification-denitrification processes have an important role in the inorganic N reductions. The NH4-N retained by peat is probably an important substrate for the nitrification-denitrification process, and it is also assimilated by the vascular plants in the area. $NH4^+$ retention onto the peat-soil can be supposed to have an important indirect effect on the functioning of the wetland: it makes the detention time of NH_4^+ in the wetland longer so that nitrification- denitrification processes and plants have more time to remove inorganic N from the system.

As final nutrient sinks vascular plants have only a small role in the Kompsasuo wetland. The main role of vegetation in the wetland is probably the aeration of the rhizospere, which increases the thickness of the acrotelm, charaterized by the most intensive chemical and microbiological retention processes in the peatlands. Plants also release dissolved organic matter for bacterial denitrification. Plant litter can also provide a substrate for microbial processing of nutrients.

Knowledge on the processes affecting the functioning of the wetland constructed on minerotrophic peatland area indicates that in order to get good purification results with the structure it is important to use the purification properties of the peat-soil as effectively as possible. There should be as efficient contact between the peat-soil and the water to be purified as possible. In this respect the hydraulic loading imposed on the wetland is the most important factor affecting the purification results achieved. Because of the rapid decrease in peat-soil hydraulic conductivity with increasing depth in peat-soils the water flows mainly across the surface of the wetland, and there is almost no contact between the peat-soil and water during the times of high hydraulic loading. The other dimensioning factors of a wetland that are connected to the efficient use of the peat-soil layer, are the size of wetland, the utilization rate of wetland area and the gradient of the wetland. There should also not be any bypass flows in the wetland area. All these dimensioning factors can be affected by appropriate planning and construction measures.

The gradient of a wetland should be about 1%. If the area is too flat, water forms anaerobic pools and Fe and P are leached downstream. The P removal in the wetlands constructed on peatlands has been attributed largely to chemical sorption on the peat soil. Also the long-term storage of P in these wetlands seems to depend largely on this chemical sorption, because P retention by vegetation, the other possible main sink of P, has been estimated to have a small role in the P retention. These results indicate that the possible mobilization of P from the wetland soil in anoxic conditions can have an important role in determining the level of the final result of purification. The forms and amount of mobilizable P in the peat-soil of the Kompsasuo wetland are now studied in the PRIMROSE project. Also the effect of the possible P mobilization on the final purification result of P achieved with the wetland will be estimated.

The possible contact of water with mineral soil can be prevented by a peat-soil layer thick enough. According to the experiences the thickness of peat-soil layer in the wetland should exceed 0.5 meters. In a thinner peat layer the possibility of the contact between water and mineral soil below the peat layer increases, which in the mires in question leads to increased outflow of P and Fe.

In order to reduce the SS loading imposed to the wetland, and to increase the purification efficiency of SS, there should be a sedimentation basin above the wetland.

One of the questions still remaining on the use of wetlands constructed on peatland areas is the estimation of the actual service life of the system and the factors affecting the service life. Also this question will be discussed.

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The nutrient removal capacity of short rotation willow forest – Estonian case study

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Introduction

Short rotation forests (SRF) have gained attention as a potential renewable energy resource. At temperate latitudes the productivity of an SRF is most often limited by nutrient and water availability (Perttu, 1988, Lindroth and Båth, 1999). In addition to mineral fertilisers, both municipal wastewater and sludge from wastewater purification facilities have been used in order to increase the wood productivity of SRFs. Municipal wastewater typically contains nutrients in almost ideal proportions for the growth of *Salix* SRFs (Perttu, 1988). There have therefore been a growing number of attempts to use SRF for dual purposes – wastewater purification (or municipal sludge neutralisation) and renewable energy production.

The first willow SRF plantations in Estonia were established in 1993 using clones of *Salix viminalis* and *Salix dasyclados* selected within the Swedish Energy Forest Program (Koppel *et al.*, 1996). In 1995 a small-scale experimental surface-subsurface wastewater purification system was launched at Aarike with high hydraulic and nutrient load (Kuusemets and Mauring, 1996). In 2000, experiments with the use of municipal sludge were started at Nõo plantation. In 2003 three full-scale vegetation filter systems were established for wastewater purification in two Estonian municipalities.

The aim of this paper is to give an overview of the experiments carried out in Estonia for wastewater purification and municipal sludge utilisation employing SRF and to analyse the corresponding nutrient removal capacity.

Material and methods

The experiments were carried out in Aarike and Nõo plantations. The data on the production capacity of Saare plantation are used as background information. Cuttings of selected clones were planted in double rows with a planting density of 2 plants per m^2. Detailed descriptions of the experimental plantations can be found in earlier publications (Koppel *et al.*, 1996; Kuusemets and Mauring 1996; Kuusemets *et al.*, 2001).

Shoot production was estimated using the allometric relations between shoot diameter at 55 cm above ground and shoot dry mass (Heinsoo *et al.*, 2002).

In Nõo plantation, which was established in 1995 and harvested in winter 1999, half of the area (0.44 ha) was treated in 2000 with municipal sludge originating from the Tartu Water Purification Station. The amount of applied sludge was 18 t (dry matter content 35%), which corresponds to the following amount of elements (kg ha^{-1}): N- 304, P- 217, K- 46, Ca- 709, Mg- 125. Biomass production was estimated after two vegetation periods. Phosphorus and nitrogen leaching was estimated using 8 lysimeters (0.6 m^2 in area) placed at 0.1 and 0.4 m from the soil surface, both in treated and control plots.

Shoot nitrogen content was estimated using the Kjeldahl method in the Plant Biochemistry laboratory at the Estonian Agricultural University. The average shoot samples were taken along 0.5 m sections of the shoot in September, at the end of the vegetation period, oven dried and ground in a mill. The shoot sample consisted of both wood and bark and thus represented the harvestable biomass.

Wastewater samples were taken 3 to 7 times per year from inflow and outflow from 1995 to 1997 in the Aarike purification system and analysed in the South Estonian Laboratory of Environment Protection using standard methods (for details see Kuusemets *et al.*, 2001).

Results and discussion

The production of different willow clones varies between plantations and years. In general, however, the shoot productivity of willow SRFs is almost doubled by the application of mineral fertilisers or municipal sludge (Table 1). In constructed wetlands the annual average plant growth is close to the figures for fertilised plots on mineral soil. The average and maximum productivity figures are close to the corresponding figures for Sweden. The most suitable clones for the local climate could produce 12 t of dry matter per year per ha.

The average purification efficiency (reduction of concentration between water inlet and outlet) of the Aarike constructed wetland was 32% for total N, and 14% for total P for the period 1995–1999 (Kuusemets *et al.*, 2001). Rough calculations indicate that the system removed 35 kg N and 2.1 kg P per year.

As detected by the lysimeters, sludge application did not cause nutrient leakage during the first vegetation period at Nõo.

In order to calculate the possible removal of N and P from the SRF plantation on the basis of the harvestable yield, some simplifying assumptions were made. The average shoot N and P content for two *Salix* clones was 0.74 and 0.07%

respectively. In the case of annual harvestable production of 12 t ha^{-1}, nutrient removal is 88 and 8.4 kg of N and P.

Table 1. Shoot productivity of Estonian willow SRF.

Plantation	Treatment	Rotation	Average plant productivity, g DM y^{-1}	Clone (a)
Aarike	surface-subsurface constructed wetland	1st, 4 years	520	S.v. clone 78183
		2nd, 4 years	750	S.d. clone 81090
Nõo	municipal sludge	2nd, 2 years	560	S.v. 3 clones
	control		284	S.v. 3 clones
Saare	fertilised	1st, 4 years	565	S.v. clone 78183
	fertilised	1st, 4th year	950	S.v. clone 78183
	fertilised	1st, 4 years	730	S.d. clone 81090 (c)
	fertilised	1st, 4th year	1200	S.d. clone 81090 (d)

(a) – clone numbers are given according to the Swedish clone numbering system; S.v. =Salix viminalis, S.d.= S. dasyclados; (c) – the clone with the highest productivity; (d) –highest annual productivity.

The results at Aarike show that only 3% of removed N and 5% of P were stored in harvestable biomass. The rest is either stored in the system or in the case of N, lost by denitrification. In the SRF-based purification system established in 2003, the nutrient load corresponds to 100 population equivalents per ha. We assume that the calculated nutrient load (440 kg N, 60 kg P and 100 kg K) is optimal for maximum willow growth and the avoidance of nutrient leakage into the groundwater.

Acknowledgements

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Closure with ecological engineering of a remote Cu/Zn concentrator: overview of 10 years R&D field program

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A 15-year field investigation to rehabilitate a small (0.75 million t) remote mining property using Ecological Engineering is described. The effluents from mill and tailings deposit (41% pyrite, 4% pyrrhotite) represent 15 t y^{-1} of zinc, the major contaminant. Other metals commonly present in AMD are also discharged into a 1-million m^3 lake, to protect the larger surrounding lake. In Figure 1 below, the major pathway of the water and contaminant sources are given.

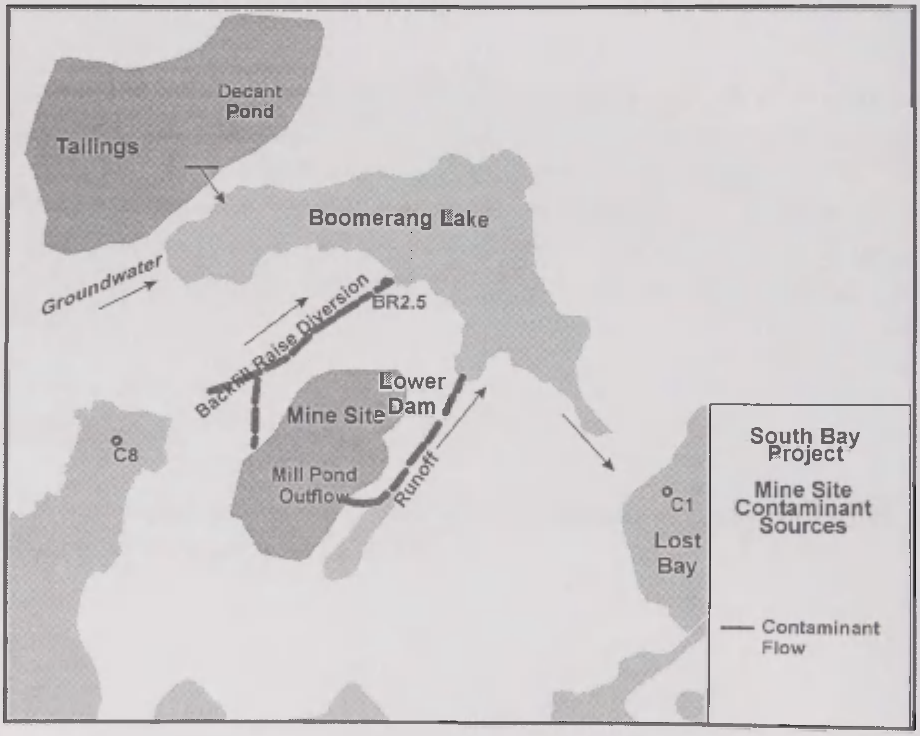

Figure 1. The major pathway of the water and contaminant sources in South Bay area.

The annual precipitation records from a weather station located in Red Lake, about 100 km to the west of the mine site, are used to derive land and lake run-off estimates for the areas where either clean or contaminated water reaches the lake. The total drainage area for Boomerang Lake is 107 ha, where 70 ha produce fresh uncontaminated water to the Boomerang Lake system (Figure 1). In Table 1, the estimated annual flows of clean and contaminated run-off are given along with the groundwater contribution from the tailings. The groundwater contribution to the lake is derived from the groundwater flow model which was constructed using Visual Modflow, a program developed by Waterloo Hydrogeologic.

Table 1. Water Sources for Boomerang Drainage Basin.

Water Sources for Boomerang	Drainage Basin	Total Runoff: Total ha	Land ha	Lake ha	Flow $m^3 y^{-1}$
	Clean Runoff	70	70	–	192,500
	Dirty Backfill Raise (BFR)	12.7	12.7	–	34,788
	Mill Pond Mine Site	11.97	10.7	12.7	31,536
	Mill Pond Outflow	12.3	12.3	–	33,835
	Ground Water	–	–	–	22,142
Entire Boomerang Drainage Basin		130.6	107	23.6	335,497

The small lake which was contaminated since mining started in 1971 is serving as a polishing pond. Ecological Engineering measures consisted of adding brush, phosphate, and calcium nitrate to the lake water and or the sediments. The biological activity, enhanced by these measures, retains 80% of the zinc load within the polishing pond.

One example is given of the approach to derive an assessment of the remediation measures and the pathway of metals from the contaminants source to the lake is given in Figure 2. The measured zinc concentration found in the lake remained relatively constant between 1986 and 1993, varying between 8–10 mg l^{-1}. However, the calculated zinc concentrations, based on mass balance, showed a rather steady increase over the same period would have been expected. More zinc was entering the lake, but was not leaving at the outflow. Thus is it expected to find the zinc in the sediments of the lake.

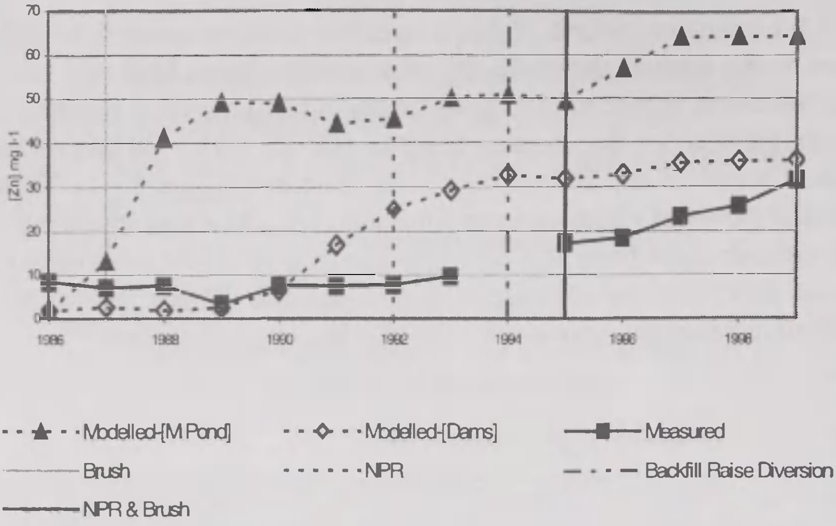

Figure 2. Boomerang Lake Performance Zinc Concentration, 1986–1999.

Zinc precipitates as zinc hydroxide and zinc carbonate at pHs between 6.5 and 7.5, whereas the current pH of the lake is around 3.0. This suggests that the zinc is adsorbed or co-precipitated with the iron hydroxide. The zinc associated with the solids collected in the sedimentation traps suggests that only 0.2 to 0.4% is contained as part of the precipitate. An aquatic moss, *Drepanocladus fluitans*, which is growing as a periphytic cover from transplanted moss bags, over the sediment after the addition of phosphate, extends over the entire lake. It is expected that is could contain a maximum of 12 kg of zinc given its concentration of 0.9% (dry weight). The major location of the remaining Zn are the sediments, where the average concentration of zinc is 0.8% with maximum reported 3.5% based on 49 sediment samples.

The increase in "resident" zinc (Figure 2) following the initial brush cutting introduction is evidence that the brush cuttings may have influenced the amount of "resident" zinc. The retained zinc is also evident if the modeled concentrations between Mill Pond outflow and the Dams are compared, representing the retention of zinc within the small, densely vegetated drainage area between the mill site and Boomerang lake (Figure 1). The only reasonable explanation is that increased surface area supplied by the brush enabled a significant increase in the algal and moss biomass in the lake, and this in turn assists in particulate retention, of which there are 140 to 191 t precipitated in the lake per year.

This paper will provide a review of the biological polishing capacity of this acidic lake and the effect of the remediation measures will be quantified. A mass balance of contaminant input to the lake and the sediment sink will be provided for 15 years of monitoring. Proposed mechanisms of transport of the metals from the water to the sediments will be discussed.

Increasing the natural values of treated wastewater: WATERHARMONICA, a way to transfer treated wastewater into a usable surface water

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The "tourist island" Texel is interesting for testing new policy plans for integrated water management. In this policy plans treated wastewater is being considered as a commodity instead of a burden, a good source of fresh water and nutrients. Moreover, it is demonstrated that a constructed wetland can be beneficial, when combined with the surroundings it can function as a recreation area, increases natural values and can be used for recycling nutrients. It can also function as a "water harmonica" between a sewage treatment plant and the surface water, being a part of closing water and nutrient cycles. The full-scale Everstekoog constructed wetland has already proven that a clever combination of an oxidation ditch and a constructed wetland is a cost-effective way to change treated sewage into "living" water suitable for various purposes. High numbers of *Daphnia* in a basin, despite low algae concentrations, led to the idea to grow *Daphnia*, which appeared to live on small sludge particles from the sewage treatment plant, incorporated in the so called "Kwekelbaarsjes system". It comprises a proposals for a step-wise 'food-chain type' system to increase the ecological value of effluents from oxidation ditches to enhance the natural values on the island, to "produce food" to improve the food situation for fish and subsequently birds like Spoonbills, which feed on small planktivorous fish, to change the effluent of the sewage treatment plant in a "living water" and to use this improved effluent as a lure flow for a fish trap to siphon fish from the sea across a high Dutch dike. The work on Texel has led to a renewed interest in various types of constructed wetlands in The Netherlands. Moreover it led to a practical innovation, which we called the Waterharmonica (project site www.waterharmonica.nl).

The WATERHARMONICA is a way of thinking to fill the gap between treated wastewater and surface water (Figure 1). It is a way to transfer well-treated wastewater in "healthy and useable" surface water. Wastewater was "the best water you had", as it was expensive drinking water, only used to transport waste.

The WATERHARMONICA aims on new integrated ecological engineering processes, by optimising multifunctional constructed wetland processes, not only used for after treatment of the clear effluent from sewage treatment plants, but also to “biological” revitalize the water. The combination with cultivation of biomass, like Daphnia and fish, aimed on strengthening of natural values by using the nutrients from the wastewater makes it a new practical form of ecological engineering.

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Phosphate sorption properties of northern wetland soils

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Introduction

Different types of constructed wetlands are used to diminish nutrients from different type of wastewaters. The major phosphorus removal processes in constructed wetlands include chemical sorption on wetland soil/sediment, P sedimentation with suspended solids and P assimilation by vegetation and micro-organisms. The chemical sorption of phosphorus on wetland soil/sediment is considered to have an important role in phosphorus retention in the wetland in a long term. There is, however, variation in the P sorption capacity of wetland soils/sediments, caused mainly by changes in their geochemical properties. The aim of the study is to present factors effecting P sorption in soils/sediments taken from 20 constructed wetlands purifying wastewater.

Material and methods

The study sites (Table 1) situate in northern and middle Europe and they include three types of constructed wetlands (CW): nine FSW wetlands with free surface water flow, four OGF wetlands with a combined overland and groundwater flow and seven SSF wetlands with sub-surface flow. Most (14) of the wetlands are used in the purification of municipal wastewater. There are also wetlands for the purification of agricultural (1) and peat mining waters (1), landfill leachates (3) and road runoff (1). Altogether seven different soil, sediment and filter media types were studied (peat, clay, sand, coarse sand, gravel, LECA and settled suspended solids).

One composite sample for P sorption experiments was taken from the soil/sediment close to the inlet of each wetland. If the wetland had different kinds of soils/sediments or filter materials, a composite sample from each of these materials was taken. At the wetlands constructed to peatland, Kompsasuo, Ruka and Skalstuggu, different parts of the wetlands were studied. At the Kompsasuo the samples were taken from five parts of the wetlands, from the reference area outside the wetland and from the sludge of the sedimentation basin before the wetland. At the Ruka wetland the samples were taken from four parts of the wetland. For the sampling of these wetlands three sub-samples were taken from each of these parts and combined. All the samples were taken from the 30 cm surface layer. At Skalstuggu the samples were taken from the wetland and reference area.

Table 1. List of wetland systems studied. The wetland type indicates the following types: OGF (Overland and groundwater flow), SSF (Sub-surface flow), FSW (Free surface water flow), FSW2 (Free surface water flow, with vegetation)

Wetland	Location	Wetland type	Soil type	Waste water source
Kompsasuo	Finland	OGF	peat[1]/sediment[2]	peat mining
Lakeus	Finland	FSW	sediment	municipal
Ruka	Finland	OGF	peat	municipal
Hovi	Finland	FSW	clay	agricultural
Tveter	Norway	SSF	leca	municipal
Spillhaug	Norway	FSW	sediment	landfill
Skjønnhaug	Norway	FSW	sediment	municipal
Bogstad	Norway	SSF	leca	municipal
Alhagen	Sweden	FSW	clay	municipal
Kodijärve	Estonia	SSF	coarse sand	municipal
Aarike	Estonia	OGF	coarse sand	municipal
Põltsamaa	Estonia	FSW	sediment	municipal
Tänassilma	Estonia	FSW	peat	municipal
Nowa Slupia	Poland	SSF	sand/gravel	municipal
Mniow	Poland	FSW	sand	municipal
Esval	Norway	SSF	gravel	landfill
Bølstad	Norway	SSF	gravel/leca	landfill
Haugstein	Norway	SSF	Fe-rich sand/leca	municipal
Ås	Norway	FSW	sediment	road runoff
Skalstuggu	Norway	OGF	peat	municipal

[1] From wetland
[2] From sedimentation basin

The geochemical properties analysed from the soil/sediment samples were TOC, organic SS, S, V, Co, Ba, Mo, Ti, Hg, Cd, Pb, Zn, Cu, Ni, Cr, Mn, As, Al, Fe, Tot.N, Tot.P, NH_4-N, NO_3-N, K, Ca, Mg and Na. Also the texture (clay, silt and sand) of the material was determined. From the Kompsasuo, Ruka and Skalstuggu wetlands, pH, total P, K, Ca, Mg, Na, Mn, Al and Fe were analyzed. Interrelationships between the P sorption capacity and the geochemical properties of the material were studied by Spearman rank correlation coefficients.

P sorption capacities of the soil/sediment samples were studied by adsorption isotherm analysis. Samples from the wetlands were dried in room temperature and homogenized. Peat samples from the Kompsasuo and Ruka wetlands were analyzed as wet samples. The samples (three replicates) were weighed (4–6 g dried sample, except from Skalstuggu 1-2 g; 4–6 g wet sample) and placed in closed polyethylene bottles. For each isotherm, a set of six samples was shaken for one hour with 40 ml of a standard KH_2PO_4-solution (0, 0.5, 2.0, 4.25, 6.0, 8.0 mg P l^{-1}), allowed to stand for 23 hours, hand shaken for 10 minutes and allowed to stand for 30 minutes. The samples were filtered through 1.2 μm (Schleicher & Schuell GF52) and 0.2 μm (Nuclepore) membranes. 10 ml samples were diluted to 50 ml, 0.5 ml H_2SO_4 was added, and the samples were analysed for PO_4-P by a molybdenum blue-ascorbic acid method (National Board of Waters 1981, SFS 1986). All ortho-phosphate lost from the liquid phase was assumed to be adsorbed. Adsorption to sediment or filter material was calculated in mg P g^{-1} dry material. For this calculation the dry weights of the wet peat samples were determined by drying them for several days at 105 °C.

The relationship between the equilibrium PO_4-P concentration in solution and the PO_4-P amount sorbed on the matrix surfaces was described by linear Freundlich adsorption isotherms. Adsorption isotherms describe the equilibrium relationship between the amounts of adsorbed and dissolved species, and they have been used extensively to study reactions of phosphate between particle surfaces and solution.

The linear Freundlich equation may be written $v = K_d\, c_d$, where
v = amount of phosphorus adsorbed (mg g^{-1})
K_d = relationship between phosphate adsorbed and in water (L g^{-1}).
c_d = P concentration in the final equilibrium solution (mg L^{-1})

For each wetland the mean of K_d-value was calculated from the K_d-values in different equilibrium concentrations and the mean was used when comparing sorption in different wetlands. The mean K_d-value for Kompsasuo and Ruka wetlands has been calculated from the K_d-values of different parts.

Results and discussion

The average K_d-values in different wetlands varied greatly (Table 2). In the inlet of some wetlands the soil had become fully saturated by P in wastewater, whereas in other wetlands the sorption capacity was still high. Most wetlands had been operated for a relatively long period of time (except for Skalstuggu) and saturation would have been expected. The saturation is indicated by high equilibrium concentration changes after added PO_4-solutions.

Table 2. The average of relationship (K_d) between phosphate adsorbed on matrix and in water in studied equilibrium concentrations for the studied wetlands.

Place		K_d
Kompsasuo	sedimentation basin	0.166
	wetland	0.199
	reference area	0.035
Lakeus		2.197
Ruka		4.046
Hovi		0.118
Tveter		–0.002
Spillhaug		0.907
Skjønnhaug		0.971
Bogstad		0.040
Alhagen		1.258
Kodijärve		0.0001
Aarike		0.003
Põltsamaa		0.033
Tänassilma		0.027
Nowa Słupia		0.037
Mniow		0.029
Esval		2.617
Bølstad	gravel	–0.0003
	LECA	0.001
Haugstein,	Fe-rich sand	–0.001
	LECA	0.001
Ås		0.149
Skalstuggu	wetland	0.011
	reference area	–0.021

In most of the wetlands, phosphorus sorption increased with the increasing equilibrium concentration of PO_4-P. However, in some wetlands which had high contents of ammouniumoxalate extractable Fe in the soil, the Freundlich or the Langmuir equation suited poorly as all the PO_4-P added was retained in these wetlands. A possible reason for this could be that PO_4-P was retained by other processes than adsorption e.g. precipitated by Fe+ and Al+.

The results indicate that the wastewater composition affects the K_d-value and can maintain higher P-sorption than what is estimated from P adsorption analysis on soil. The highest K_d-values (1–4) were found in wetlands Lakeus, Ruka, Skjønnhaug, Alhagen receiving wastewater from municipal wastewater treatment plants using chemical precipitation resulting in remnants of Fe/Al in wastewater. The same seems to be the case for the wetlands of Spillhaug and Esval where metal rich water is leaching from the municipal landfills. Also, at the peatland treatment system of Skalstuggu similar result was observed. There the P-sorption was higher at the treatment wetland than at the reference area (area not affected by wastewater) due to the uptake of Ca. The results imply that even wetlands with fairly low P-sorption capacity can be used for P-removal if the wastewater itself supplies the wetlands with sorbing anions.

A low K_d-value (below 0.01) was found at several wetlands. The lowest values are mainly due to saturation of the soil in the inlet by phosphorus loading. The LECA filter at Tveter and sand filter of Haugstein have already been operated for about 10 years and saturation has occurred at the inlet. For these sites the results indicate that no further P-uptake in the inlet can be achieved by processes other than sorption, e.g. precipitation.

The comparison of the K_d averages of studied wetlands with the geochemical properties of wetland soils showed that those wetlands with the highest contents of Fe and Al correlated significantly with the average K_d-values ($r=0.591$, $p=0.005$, and $r=0.451$, $p=0.040$, respectively, $n=21$). Also the proportion of silt and clay in wetland texture correlated positively with the P sorption capacity ($r=0.582$ $p=0.014$ and $r=0.690$ $p=0.002$, respectively, $n=17$), but sand as filter material correlated negatively ($r=-0.657$ $p=0.004$, $n=17$).

The desorption of P in the reference concentration (0 mg P l^{-1}) varied between the soils. However, for most soils, only a small portion of P was desorbed. This partly indicates that the sorption process is not reversible due to precipitation on Fe, Al and Ca. Thus, it also indicates that if P concentration in the wetland inlet is temporarily lowered the wetland will not leach P.

Acknowledgements

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Passive treatment of contaminated water from uranium mining and milling

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Treatment of contaminated water is a key issue in the remediation of the legacies left behind by former uranium ore mining and processing operations in the German federal states of Saxony and Thuringia.

To minimize expenditures for the separation of contaminants while maintaining compliance with regulatory quality standards, passive processing technologies are being developed and implemented.

To treat a partial flow (0.5 m^3 h^{-1} to 2 m^3 h^{-1}) of the food water from the Pöhla/Tellerhäuser mine, a constructed wetland (CW) was put into operation for continuous test work in summer 1998. Relevant components of the neutral flood water include Radium-226 (ca 4 Bq l^{-1}), Iron (ca 9 mg l^{-1}), and Arsenic (ca 2.5 mg l^{-1}).

To achieve a stable separation of the pollutants in the pilot water treatment plant, detailed studies were carried out on physical-chemical processes like passive oxidation/precipitation and adsorption as well as on microbiological processes and accumulation of pollutants by macrophytic algae and helophytes.

The CW system consists of a ventilation cascade, five reaction basins and a post-processing phase using reactive material. Specific features (plant-carrying floating mats, coco mats placed vertically to the circulating water flow, etc.) were installed to serve the establishment of microorganisms and to retain finely dispersed particles.

During 5 years of operation, mean separation rates achieved were 98% for Iron, 83% for Arsenic, and 73% for Radium-226. Operation of the pilot water treatment plant is stable and reliable.

In the light of the findings and experiences from that pilot plant, a large-scale installation was designed to ensure the long-term treatment of the entire mine water flow at a rate of approx. 20 m^3 h^{-1}.

This large-scale water treatment plant will go on-stream in 2003.

Deceleration of runoff in free water surface wetlands

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Introduction and objective

Constructed wetlands for the treatment of agricultural runoff have become more an more common in recent years (Uusi-Kämppä *et al.*, 2000; Woltemade, 2000). In terms of retention performance, previous experiences (*e.g.* Uusi-Kämppä *et al.*, 2000; Koskiaho *et al.*, 2003) suggest that a constructed wetland's area in proportion to its watershed's area is the key design factor, as it is directly related to the nominal residence time of water (t_n). In theory, the longer the t_n, the more there is time for the water purifying processes. However, wetland must also have a good hydraulic efficiency in order to make actual use of the t_n. Wetlands with similar storage volumes may, according to their shape and layout, vary highly with regard to their hydraulic efficiencies (Persson *et al.*, 1999). The aim of this study was to investigate the flow behaviour in two differently dimensioned and shaped Free Water-Surface type of constructed wetlands (FWSs), both of which were situated in agricultural watersheds in southern Finland.

Material and methods

The most important properties of the investigated FWSs are presented in Table 1.

The flow simulations were made with a two-dimensional hydrodynamic model RMA2 (King *et al.*, 1975). Input data consisted of (i) digital meshes of the FWSs, i.e. datasets of x-, y-, and z-coordinates depicting their physical dimensions, (ii) inflows representing typical spring flood situations, and (iii) corresponding water surface elevations at the outlets. According to our own flow measurements, and the suggestions of Seuna (1983), steady inflow rates of 29 l s^{-1} and 153 l s^{-1} were chosen for the RMA2 simulations for Hovi and Alastaro, respectively. These values represent 'critical' situations when $t_n s$ are at shortest and big portion of the annual load transported through the FWSs. The use of lower flow rates like annual means would have yielded less relevant information. Manning's n values were used as roughness coefficients. Tracer tests were simulated with the RMA4 model (King *et al.,* 1973) using the flow fields obtained from the RMA2 simulations. RMA4 computes concentrations within two-dimensional computational mesh domain by simulating the depth-average advection-diffusion process in an aquatic environment. RMA4 inputs included diffusion coefficients for each material type defined in the meshes, and inlet concentrations at the beginning of the simulations.

Table 1. Descriptions of the FWSs in study.

	Hovi	Alastaro
FWS area (ha)	0.60	0.48
Watershed area (ha)	12	90
FWS area/Watershed (%)	5%	0.5%
Year of construction	1998	1996
Water volume at flood (m^3)	4000	3500
Vegetation during the study period	Relatively sparse (appr. 44 g m^{-2}) due to the young age of the FWS. Main species: *Typha latifolia, Scirpus sylvaticus, Alisma plantago-aquatica.*	Well-developed (appr. 91 g m^{-2}). Main species: *Typha latifolia, Alisma plantago-aquatica.*
Morphology of the FWSs	Deep part (>1.5 m) at the beginning. Becomes shallower (<0.5m) towards the outlet. An islet in the middle of the FWS. Two baffles create a form of the letter "S".	Deep (1.2 m) and shallow (0.3 m) parts are divided by a levelling terrace. Two islets in the shallow part. Rectangular, elongated shape with inlet and outlet at different sides of the FWS.

Hydraulic efficiency (λ) of the FWSs was examined by similar approach to that presented in Persson *et al.* (1999). The λ values were calculated by Equation (1).

$$\lambda = \frac{t_p}{t_n} \quad (1)$$

where t_n was the nominal residence time of water (Volume divided by inflow) and t_p was the time elapsed from the moment of tracer input until the moment of maximum tracer concentration observed at outlet during the RMA4 simulations. The simulation periods (120 hours for Hovi and 20 hours for Alastaro) were chosen to well exceed the $t_n s$ of the FWSs. Flow fields for the RMA4-simulations were obtained by repeating the RMA2-simulations for these periods (40 time steps for each FWS).

Flow was measured at the outlets of both FWSs by the means of v-notch weirs and continuously drawing water stage recorders. In Hovi, similar equipment was at the inlet ditch, as well. This gave an opportunity to simultaneously examine the inflow and the outflow, and thereby obtain apparent evidence of runoff deceleration.

Results and discussion

The RMA2 simulations made for Hovi revealed that the flow velocity was at its slowest (ca 0.5 mm s^{-1}) in the deep part and gradually increased until the bend between the two shallow regions, being there 4 mm s^{-1}. This is still clearly below 21 mm s^{-1}, which according to Huisman (1973) is the critical velocity for the detachment of clay (Ø < 0.002 mm) particles. Hence, there is virtually no risk of resuspension from the bottom of the Hovi FWS even during flood periods. The positive effect of the baffles on the hydraulic efficiency of the Hovi FWS was demonstrated in Koskiaho (2003). The effect of the Hovi FWS on flow deceleration is clearly illustrated in the hydrographs in Figure 1. The inflow curve has markedly higher peaks and faster post-peak decreases than the smoother shaped outflow curve.

The flow velocity simulated for the Alastaro FWS exceeded the abovementioned critical value (21 mm s^{-1}) in ca 25 m^2 areas right after the inlet ditch and just before the outlet weir. This indicates the risk of resuspension in the Alastaro FWS. Persson *et al.* (1999) found that the inlet and the outlet being on different sides of an FWS is beneficial in terms of hydraulic efficiency. Unexpectedly, however, no convincing evidence for this was found in Koskiaho (2003), where the hydraulic efficiency of the Alastaro FWS proved to be quite insensitive for the changes of position or width of the inlet. This was presumed to be due to (i) favourable length to width ratio (4:1), and (ii) levelling effect of the difference in elevation between the deep and shallow parts of the FWS.

Comparison of λ values (Table 2) suggests that the Alastaro FWS is hydraulically not as efficient as the Hovi FWS. However, it cannot be judged very poor, either. Indeed, more likely reason for the drastically weaker retention performance than in Hovi is its smaller area in proportion to its watershed (Koskiaho *et al.*, 2003). Many of the features of the Alastaro FWS like elongated shape, submerged terrace, and the islets have been found to improve the λ values of pond layouts (Persson *et al.*, 1999). For the design cases with small FWS-to-watershed area ratio a simple, elongated shape without terrestrial elements (like baffles) that inevitably occupy their part of the already small FWS volume, could a beneficial solution. Yet, this does not mean that a dull, rectangular pond remains the only choice. At least aesthetically better outcome could be – without compromising the original idea of simplicity – achieved e.g. by applying mildly meandering shorelines and smoothly alternating zones of open-water and vegetation. Obviously, in the cases with generous FWS-to-watershed area ratio, like in Hovi, the loss of CW volume can be well afforded and the baffles are highly recommended.

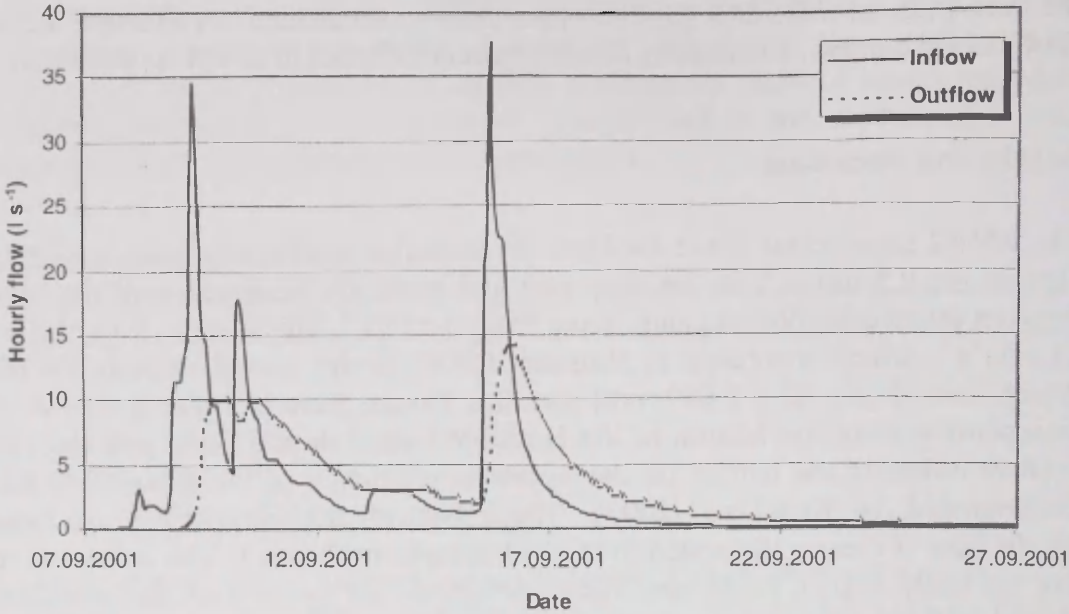

Figure 1. Inflow and outflow (hourly means) in the Hovi FWS in September 2001.

Table 2. Hydraulic efficiency (λ) of the Hovi and Alastaro FWSs.

FWS	Nominal residence time of water (t_n)	Residence time of tracer peak (t_p)	λ
	hh	Mm	
Hovi	39:10	25:30	0.65
Alastaro	6 :19	3 :15	0.52

Conclusions

FWSs offer a cost-beneficial option to capture part of the agricultural diffuse loading by slowing flow velocities during the critical floods. When optimum retention performance is aimed for, an adequate FWS-to-watershed area ratio (at least 2%) and a hydraulically efficient layout together form a solid basis for FWS design. A useful design option is to use baffles for directing the flow to maximally exploit the pond area. Baffles not only improve the hydraulic efficiency but also – like islets – enliven local landscape and increase biodiversity. However, for the cases of exiguous FWS area simpler shapes may be more suitable by providing fairly high hydraulic efficiency without the loss of volume.

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Convective O_2 transport in a constructed free water-surface wetland and its ANN-based modelling

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Introduction

Due to the slow diffusion of oxygen (O_2) in an aqueous solution, the deep waters and the inundated soil in wetlands often become anaerobic (Mitsch and Gosselink, 1986). However, there are mechanisms that transport O_2 from the atmosphere into the deeper parts of a wetland. These include O_2 release from the roots of emergent macrophytes and convective O_2 transport. The latter phenomenon has been recognised in lakes (Horsch and Stefan, 1988) as well as in shallow wetlands (Arnold and Oldham 1997). Since anaerobia is known to trigger many undesired processes in aquatic ecosystems (e.g. phosphorus release from the sediment and emissions of greenhouse gases), convective O_2 transport is worth consideration. The aim of this study was to find out whether, and to what extent, convective O_2 transport occurs in a Free Water-Surface (FWS) type of constructed wetland. Another objective was to examine the appropriateness of Artificial Neural Networks (ANNs, for ref. see Task Committee, 2000) for the modelling of convective O_2 transport in FWS wetlands.

Study site and measurement programme

The FWS wetland at Hovi (Figure 1) was constructed in the south of Finland (60°25′ N, 24°22′ E) in 1998 and has a watershed of 12 ha (Koskiaho, 2003). It receives diffuse loads from arable fields, with no point sources. The wetland pond has a surface of some 6000 m² at elevated water-levels (*i.e.* 5% of the catchment area) and comprises a deep part (some 120 cm depth) close to the inflow section, followed by a vegetation zone (of some 65 cm depth) and a shallow part (50 cm depth and less). In the period addressed here (2001), prominent macrophyte species at Hovi were cattail (*Typha latifolia)* and club-rush (*Scirpus sylvaticus*), among others (see Koskiaho, 2003, also for further detail). Instrumentation with measuring devices formed part of the research effort by the Finnish Environment Institute (SYKE).

Hourly data measured in the deep part comprised water temperatures at 5 cm, 38 cm, 72 cm, 105 cm and 115 cm from the bottom, turbidity 105 cm from the bottom and O_2 saturation some 20 cm above bottom. Sensors placed in the vegetation zone recorded temperatures at four different depths, together with turbidity and O_2. Rates of inflow and outflow were recorded continuously, and air temperatures were available on an hourly basis. In addition, values of wind speed and direction at 3 m height above ground were recorded for part of the measuring period.

Figure 1. An aerial photograph depicting location of the measurement devices in the Hovi wetland.

Hydromechanics of convective O_2 transport and results of the measurements

The instrumentation of the deep part and the vegetation zone as described in the previous section was well suited to capture events of convective O_2 transport from the surface to the bottom ('sinking plumes'). Such events occur when heavier water overlies lighter (less dense) water lower down in the vertical profile. Such unstable layering may form as the surface water cools during the night and, thus, becomes denser and eventually heavier than the water deeper down and closer to the bottom. Conversely, heating of the bottom would also result in a hydromechanically unstable situation (rising thermals). At night, in the absence of photosynthesis, re-aeration usually becomes the main source of O_2 in the wetland pond, but this mechanism primarily affects the upper part of the water body close to the air-water interface. To become effective close to the bottom as well, some transport mechanism is needed. Theoretically, there is the presence of diffusive fluxes, but these are weak: coefficients of molecular or Fickian diffusion have an order of magnitude as low as 10^{-9} m² s^{-1}, and at Reynolds numbers of, typically, 1000 and below, turbulent diffusion cannot be strong either. Yet, in spite of the weakness of the above transport mechanisms, the measurements at Hovi documented pronounced increases in O_2 saturation near the bottom under certain circumstances. Between 30 July and 7 September, there were 34 nightly periods when surface water became colder than bottom water. These have been termed "convection periods" here. An example of a convection period can be seen from Figure 2. In the deep part, clear increases of O_2 saturation were noted in 74% of these periods. In the vegetation zone, increases were not only noticed less frequently (44% of the periods), but also more equivocally and with weaker rates. One obvious reason for this was that the average initial O_2 saturation, *i.e.* the saturation at the beginning of a convection period, was clearly higher in the vegetation zone (71%) than in the deep part (43%). This difference may be due to the transport of O_2 through the emergent cattails into their roots, and subsequently to the deep water in the vegetation zone.

Figure 2 shows an interval with denser surface water overlying lighter water close to the bottom. This relative position of layers is unstable, and tends to trigger a convective flow (Tritton, 1991), which will start as soon as this instability overrides the counteracting influences of viscosity (resistance to the flow) and heat conduction (aiming at a removal of the temperature difference). A measure of the tendency towards convection is the Rayleigh number Ra, defined as:

$$\mathrm{Ra} = \frac{g \cdot \alpha_V \cdot (T_2 - T_1) \cdot h^3}{\nu \cdot \kappa} \tag{1}$$

with *g* the acceleration of gravity, α_V the coefficient of expansion, T_2 and T_1 the water temperatures at bottom and surface, respectively, h the water depth and ν the kinematic viscosity of the fluid (water). κ stands for the thermal diffusivity (here: of water), to be computed from $\kappa = \frac{\lambda}{\rho \cdot c}$, with λ the thermal conductivity, ρ the density and c_p the specific heat at constant pressure. Linear stability theory predicts that the onset of convection is characterized by Ra = 1708, and experimental evidence supports this prediction, so that a critical (minimum) Ra number for convection can be given as (Tritton, op.cit.):

$$\mathrm{Ra}_{crit} \approx 1700 \tag{2}$$

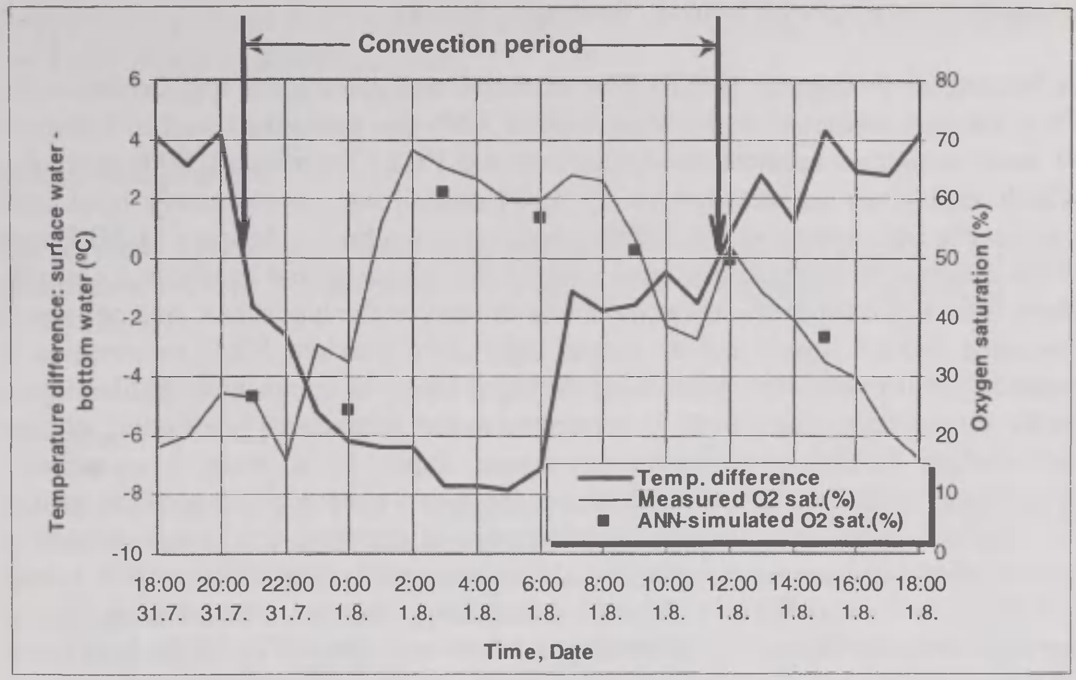

Figure 2. The convection period determined by vertical temperature differences, and ANN-simulated (validation only) versus measured O_2-saturation for 31st July – 1st Aug, 2001 in the Hovi wetland.

Substituting the values of water for the expansion coefficient, the thermal diffusivity and the kinematic viscosity, i.e. $\alpha \approx 18 \cdot 10^{-5}\ \mathrm{K}^{-1}$, $\kappa \approx 0.14 \cdot 10^{-6}\ \mathrm{m^2/s}$ and $\nu \approx 1.3 \cdot 10^{-6}\ \mathrm{m^2/s}$, one obtains Ra ≈ 10^7 per Kelvin (or °C) of temperature difference T_2-T_1 in case of water depth h = 10 cm. A water depth of h = 1.2 m (and this is about the value of Hovi's deep part, see above) each degree centigrade of temperature difference is associated with Ra ≈ 10^{10}. In consequence, the critical Ra-number is exceeded practically as soon as the surface temperature falls below the bottom temperature, and convection sets in almost immediately. This agrees with previous research on convection in paddy fields (Mowjood *et al.*, 1997), reporting convective flows to start at temperature differences below 1 °C.

This rapid onset of convection does not, however, mean an immediate rise in O_2 saturation near the bottom of the deep part at Hovi. Figure 2, for instance, shows the reaction of the O_2-saturation curve to be somewhat delayed, and such a lag between the onset of convection and the rise in O_2 content near the bottom is not unexpected, as the O_2 enriched particles at or near the water surface, which are moved downwards by the convective flow, have to travel a distance of the order of 1 m (deep part) at a fairly low velocity.

Modelling by Artificial Neural Networks (ANNs)

A Multilayer Perceptron (MLP) type of ANN was chosen for application to the Hovi datasets described above. This type of ANN has performed well in a number of water resources applications before, see e.g Task Committee (2000), a source, which also gives an introduction to ANN modelling. An extremely brief (and necessarily incomplete) outline of the model is given here as follows: A MLP type ANN consists of single nodes (also called 'neurons') arranged in layers. Typically, there is an input layer, the neurons of which receive the input data, one or several so-called 'hidden layers' and an output layer. In a standard MLP, information is passed forward unidirectionally from the input layer via one or more hidden layers to the output layer. Each node is connected to the subsequent layer only, with no information feedback to preceding layers. Input to a node from another (preceding) node is weighted, and the net input to a neuron consists of the sum of all weighted inputs from the individual neurons of the preceding layer (a threshold value called bias may or may not be subtracted as well). The resulting real-valued net input is then subjected to a so-called activation function, which transforms the net input into the output of the neuron to be, in turn, passed on to the next layer. The sigmoid (logistic) function is frequently used as activation function, and this is also the case in this application. Values of interneuron connection weights (and, possibly, bias) are determined in the course of model calibration (called network training, here: error backpropagation), model validation (on previously unused data) takes place in the course of 'network testing'.

Several series of ANNs of different architecture were developed to study their potential to 'learn' convective O_2 transport in the wetland. Model input consisted of hourly values of temperatures (115 cm and 5 cm above bottom), O_2 saturation [%], wetland inflow [$l\ s^{-1}$], wetland outflow [$l\ s^{-1}$] and air temperature for the period between 28th June and 31st October, 2001, with the time of day added as a further input to characterize diurnal variations certainly present in the processes. The best 3-layer ANN (input layer, 1 hidden layer, output layer) addressing this input data set was identified as a 7-8-1 MLP (7 input nodes, 8 hidden nodes, 1 output in the form of O_2-saturation 4 hours ahead), with the best choice internal parameters 'learning rate' being 0.03 and 'momentum' 0.0. With the maximum value of each variable (*i.e.* the aforementioned 7 input and 1 output variables) scaled to 0.9 (as is usually done in ANN development), the above 7-8-1 MLP achieved 77.1% correct testing results at a level of error margin 0.1. Data separation into training and testing patterns was performed by making every third record a member of the testing set (with the other ca. two-thirds left for training). Maximum values were moved to the training set in any case so as to avoid extrapolation by the ANN (which cannot be expected to reproduce patterns it has had no change to 'learn'). Evaluated specifically for convective phases in the wetland, results were favourable and similar to those shown in Figure 2. The comparisons indicate that a three layer MLP is quite capable of 'learning' convective O_2 transport in a wetland pond.

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Spatial modelling and patter analysis of nutrient reduction in an estuary wetland in northeastern China

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The eutrophication problem of coastal seawater has been a serious hazard in the last two decades in Eastern China. This problem is mainly caused by nutrients such as nitrogen and phosphorous from inland non-point sources. The purification function of natural wetlands at large river deltas provides a potential solution to cut down nutrient input into the sea. The objective of this research is to develop a model to simulate the ability of nutrient reduction in the wetland system, so as to evaluate to what extent a natural wetland can be used as a treatment system for nutrient-enriched river water.

The study area is in the Liaohe Delta in northeastern China, which is home to the largest reed marsh in the world, with an area of over 80,000 ha. The climate here is temperate monsoon, i.e. hot and rainy in summer and cold and dry in winter. Spring is even drier due to the windy weather. As in the case of many other wetlands, it is a paradise for many species of wildlife, especially birds. Nevertheless, this piece of land has been under great pressure due to the rapid development of agriculture, aquaculture and the oil industry. The natural wetlands area has been shrinking because of conversion to other forms of land use. The reed marsh itself has also become semi-natural, with irrigation in spring and harvest in winter. However, extensive management in the reed system provides a good chance to balance the conflict between human activity and nature conservation. If the natural wetlands along the coast are used for nutrient reduction, advantages will be gained for both the wetlands and the nearby marine environment.

The natural wetland is a multi-functional landscape in terms of biomass production, nature conservation, water regulation, soil formation, coastline stabilisation and pollutant purification. The reed and canal system has a high reduction rate for many pollutants such as COD, nitrogen and phosphorous. In addition, the reed marsh can also be used as a treatment system for oil drilling water. Therefore, wastewater irrigation in the reed field should be encouraged to solve the water shortage problem in the spring, increase reed productivity and prevent coastal pollution. Furthermore, the *Suaeda heteroptera* community has great potential to be used as a treatment system for exchanged water from breeding ponds.

Two different models have been established to simulate the nutrient reduction and its distribution in the reed-canal system of the Liaohe Delta. One is solely based on field data, which is more complicated and dependent on hydrological data. The other is partly based on Mander & Mauring's regression model, which is more general and demands less data. The simulation results from the two models are comparable. Although the model based on field data is slightly superior for the Liaohe Delta, Mander & Mauring's model was preferred because of its generality and simplicity. Finally, the field data-based non-linear regression model for the canal system was used, and Mander & Mauring's linear regression model for the reed system was adopted. The algorithms used are:
Canal system:

$$C_{(x,y)} = -\mathrm{A}^* \ln(dist.) + \mathrm{B} \quad (\mathrm{A}>0, \mathrm{B}>0) \tag{1}$$

where $C_{(x,y)}$ is the nutrient concentration value (in mg/l) of a canal point at a certain distance *(dist.)* from the pumping station. A and B are linearly related to the nutrient input concentration (*inload* in mg l^{-1}) at the pumping station:

$$\mathrm{A} = \mathrm{C}_1 * inload + \mathrm{C}_2 \tag{2}$$
$$\mathrm{B} = \mathrm{C}_3 * inload + \mathrm{C} \tag{3}$$

The values for $C_1 \sim C_4$ are obtained for nitrogen and phosphorous on the basis of the experimental data gathered in the field.
Reed system:

$$Y = \mathrm{A}^*X + \mathrm{B} \tag{4}$$

where Y is the retention, and X is the input load, both in g $m^{-2}.d^{-1}$ for nitrogen and mg $m^{-2}.d^{-1}$ for phosphorous. Constants A and B are obtained on the basis of published data from more than 40 study sites all over the world (Mander and Mauring, 1994).

According to the results of the simulation, there is a "mutual compensation" for the nutrient reduction in the reed and canal systems, so that the total reduction rate remains relatively stable in spite of the input concentration change at the pumping station. It is 66% for total nitrogen and 90% for soluble reactive phosphorus. In combination with the canals, the present 80,000 ha of reed can remove about 3.2–4.0 tons of nitrogen and 80 tons of soluble reactive phosphorous during the annual irrigation period. Nevertheless, this is only 1/10 of its total reduction capacity, with water being the limiting factor.

To study the effect of landscape pattern on nutrient reduction, four spatial combinations of reed, canals and pumping stations are designed.

Canal density: no canal, ¼ present, ½ present, present and double present canal density. Size of reed area: ¼ present, ½ present, ¾ present and present reed area.

Reed shrinking pattern: shrinkage, perforation, fragmentation and bisection. Position of pumping station: 2 cases on the border of the reed area and 3 cases at some point inside the reed area.

By altering one factor while keeping the others stable, the nutrient reduction corresponding to each case was calculated using the model developed above. The results of the simulation indicate that each factor causes a less than 10% deviation in the total rate of nutrient reduction, although the absolute quantity of reduction can vary. If the reed area is stable, it is preferable to retain a low canal density, and keep the pumping station near the border of the reed area. Generally speaking, a smaller reed area is more efficient for nutrient reduction than are larger, scattered ones. The reduction rate sequence for different reed distribution patterns is: Shrinkage > Perforation > Fragmentation > Bisection. The shrinkage pattern of land transformation for the reed is most recommended in maintaining a high reduction rate for the nutrients. The present reed area can receive at least 4 times more water in spring.

The relationship between landscape structure and nutrient reduction is further measured with the help of certain landscape indices. A couple of indices are selected and calculated corresponding to each scenario of spatial patterns designed in the above section. The correlation coefficient between the value of indices and nutrient reduction rates is calculated for each scenario. The results indicate that not all of the landscape indices are closely related to the wetland system's nutrient reduction. Therefore the ability of landscape indices to characterise the effect of pattern change on nutrient reduction is relatively limited. Redundancies also exist among similar indices. Landscape indices should be chosen in accordance with the purpose of the study, based on the criteria of simplicity, generality and ecological significance.

The research work is a combination of landscape ecology, wetland ecology and GIS technology. The spatial model developed is also applicable to other areas with similar situations. The results will contribute to sustainable landscape planning in the study area. The final conclusion is that natural wetlands have great potential to be used for the reduction of nutrient input into the sea.

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Tracer experiment with ^{15}N-enriched nitrate, ^{32}P-labelled phosphate and tritiated water in Ekeby treatment wetland, Sweden

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Introduction

Run-off water from agricultural areas and municipal wastewater bring nutrients (phosphorus and nitrogen) to marine environments and can cause algae blooms. Eutrophication of the Baltic Sea causes algae to accumulate in bottom sediments and contributes to dead, oxygen-depleted bottoms (Ambio, 1990; Petterson and Boström, 1990). In the south part of the Baltic Sea nitrogen was recognised as probably the most limiting nutrient for eutrophication (Granéli *et al.*, 1990).

Sewage treatment plant and treatment wetlands have shown to play an important role in reducing the nutrient content of recipient waters. Particularly wetlands have the potential to offer sufficiently long residence times needed for reducing both the phosphorus and nitrogen load. Therefore, it is of great importance to develop new, design models from which it is possible to optimise the nutrient removal efficiency by an effective use of land area with respect to through flow and nutrient load, the shape of wetlands, vegetation type and its distribution. Design approaches are presented among others by Kadlec and Knight (1996), Kadlec (1999) and Wörman and Kronnäs (2003).

A crucial point in model development is to access high quality data of the functional characteristics of wetlands. This is not always possible to achieve with standard monitoring data, due to the temporal variability in the in- and out-loads of the wetland and the internal circulation of nitrogen and phosphorus, which also varies temporally with changes in whether conditions as well as spatially. Tracer experiments, on the other hand, offer a possibility of artificial production of in- and output signals (of the tracers) that are more clearly defined and, therefore much easier to interpret in terms of model parameters. The experimental data can also reveal the mass fractions of nutrients that are removed in the wetland.

Tracer experiments with "inert" solutes are routinely used in wetland technology (Kadlec, 1999). However, surprisingly few tracer experiments in treatment wetlands with phosphorus or nitrogen isotopes are reported in the scientific literature.

The overall objective of this study is to present a simultaneous tracer experiment in Ekeby treatment wetland in Eskilstuna, Sweden, with tritiated water (HTO), ^{15}N-enriched nitrate (N-15) and ^{32}P-labelled phosphate (P-32). The use of both the conservative tracer, HTO, and reactive tracers, N-15 and P-32, facilitate a discrimination of the hydraulic processes and biogeochemical reactions. The wetland consists of several parallel series of treatment ponds and the test was carried out in one of the ponds. This paper describes mainly the results related to tritiated water and P-32.

Experiment – wetland characteristics and methods

Ekeby wetland is a constructed wetland that receives the effluent water from the municipal sewage treatment plant in Eskilstuna 120 km west from Stockholm, Sweden. The annual flow through the treatment plant and the wetland is 15,400,000 m^3 and serves 76 500 connected persons. The annual load to the wetland is 66.3 tons of total phosphorus, 420 tons of total nitrogen and the total open water area is about 30 hectares. Main vegetation in the wetland consists of emergent Reed Grass (*Glyceria maxima*) and submersed Canadian Pond Weed (*Elodera Canadensis*).

The wetland consists of 8 ponds. The treated wastewater from the treatment plant flows into a channel that distributes the water flow on five parallel ponds. Thereafter, the water flows into another distribution channel that separates the water on another three ponds. After the three downstream ponds the water ones again are collected in a channel before it flows into the Eskilstuna River. Eskilstuna River ends in Lake Mälaren, a lake that is connected with the Baltic Sea, via Stockholm.

On the 22nd November 2002 a tracer mixture was injected in the inlet pipe that connects the distribution channel with pond one in Ekeby wetland. Pond one has a surface area of 2.6 hectares and is the first of the first five parallel ponds and is situated closest to the sewage treatment plant. The second inlet tube was closed during the injection and the discharge through the pond increased from 84 l s^{-1} at the time of the injection to 150 l s^{-1} on the 26th of November and then decreased to 84 l s^{-1} on the 10th of December. The mixture consisted of 18 GBq of P-32, 3 kg N-15 (10Atom% KNO_3-1^5N) and 74 GBq of HTO. P-32 was injected in ionic form (PO_4^{3-}) and N-15 as nitrate (NO_3^-). The injection was performed at a fairly constant rate during 5 h and 15 min.

Water samples were taken by means of auto-samplers at five points in the pond, at the inlet and outlet and at the three small islands that lie in a transversal line across the pond about 1/3 of the way from the inlet to the outlet. The last water sample was taken to weeks after the injection at the outlet. The analyses of

P-32 and HTO were performed in a beta counter and corrected for natural background and radioactive decay.

After the tracer pulse had passed through the pond, samples from sediment and vegetation were collected. Sediment samples were taken from pond bed sediment before and after the small islands and vegetation samples from selected locations near two of the islands. Filtration of water samples was performed in the laboratory for determination of the partitioning of particulate and dissolved (< 0.45 μm) fractions of N-15 and P-32.

Model and evaluation methods

A mass balance of P-32 was undertaken based on the known injected activity, **M**, and a numerical integration of the discharge from the wetland pond according to

$$M_{eff} = \int_0^{\infty} C_{out} Q dt \approx \sum_{i=1}^{N} C_{out,i} Q_i \Delta t_i \tag{1}$$

in which Q is the water flow [m^3/s], Δt is a time interval [s], C_{out} is concentration [Bq/l] and i is an index of the time discretisation.

The phosphorus transformations in the wetland are also studied using a model framework that decouples hydraulics and chemical transformation of solutes in two separate functions that can be evaluated independently using HTO and P-32 or N-15 (see e.g. Kadlec and Knight, 1996; Levenspiel, 1999; Fogler, 1999). The coupling can be performed under the assumption of no mixing between flow paths. The method can be regarded as an evaluation of the average response at the exit concentrations of several pathways of different residence time τ. Generally, the phosphorus concentration at the outlet of the wetland can be written:

$$C_{out}(t) = \int_0^{\infty} C(t,\tau) f(\tau) d\tau \tag{2}$$

in which f(τ) is the probability density function (PDF) of the residence time for water between the inlet to the outlet and C(t,τ) is the variation of the concentration with time t for a specific transport pathway (denoted by τ). The concentration function is derived in Laplace space based on a model concept similar to that of Wörman *et al.* (2002) and the solution is inverted using a numerical routine. The water residence time was modelled based on the commonly used “tanks-in-series” (e.g. Kadlec and Knight, 1996).

Results

From the breakthrough curves shown in Fig. 1, the mean residence time of water in pond 1 can be estimated to be about 3 to 3.5 days. This agrees well with the mean residence time estimated as the volume divided by the through flow, which results in about 3.2 days.

The total injected activity of P-32 was 18.0 GBq and about 13.7 GBq was recovered at the outlet during the investigation period ending 10 days and 16 hours after the start of the injection. This implies that 24% of the phosphate solution was removed in the November – December period in which the experiment was performed. An analysis of regular monitoring data shows that the annual removal rate of total phosphorus in the entire wetland for year 2002 (each flow line passes two ponds in series) is 57%.

Tentative results from the bed core samples indicate that a major fraction of the P-32 was removed by accumulation in the bed sediments. Probably, the most important mechanism for this removal is adsorption onto particulate matter and deposition as well as a certain exchange in dissolved phase directly with the bed sediment and adsorption onto the bed matrix. Analyses of vegetation material also show that a certain degree of P-32 accumulation onto vegetation and suspended solids attached to vegetation has occurred.

On a short time-scale, the removal of P-32 can be seen as irreversible. A simple representation is to use a first order reaction for the removal rate from the water to the bed sediment under the water travelling through the wetland. The result in terms of the through flow of P-32 can be expressed as $M_{eff}/M_{in}=Exp(-k <\tau>)$, where k is a rate coefficient $[s^{-1}]$ and $<\tau>$ is the mean residence time (~3.2 days). Since, $M_{eff}/M_{in}= 0.76$, we find that $k = 0.086\ days^{-1}$.

The model evaluation based on (2) starts with fitting of the "tank-reactor series" model to the observed breakthrough curve of HTO and the resulting $f(\tau)$-function is used as a basis for evaluation of $C(t,\tau)$ in (2). The fitting is done "by-eye". The P-32 curves are shown in Figure1.

Conclusions

A prototype scale tracer test with HTO, P-32 and N-15 was successfully performed in Ekeby treatment wetland, Eskilstuna, Sweden. The quality of the data allows for a fairly reliable evaluation of different model concepts that can be used to interpret treatment processes and optimise the use of the wetland.

About 24% of the injected P-32 was removed on the time frame investigated in this study. Since, the main chemical form of P was phosphate it is likely that adsorption to particulate matter and subsequent accumulation in bed sediments is responsible for the removal.

A simple modelling interpretation indicated a need to separate the hydraulic response from the chemical reactions. The use of both an inert and reactive tracers made it possible to evaluate a process based model that can be used to optimise treatment with respect to shape of wetland and promotion of adsorption processes.

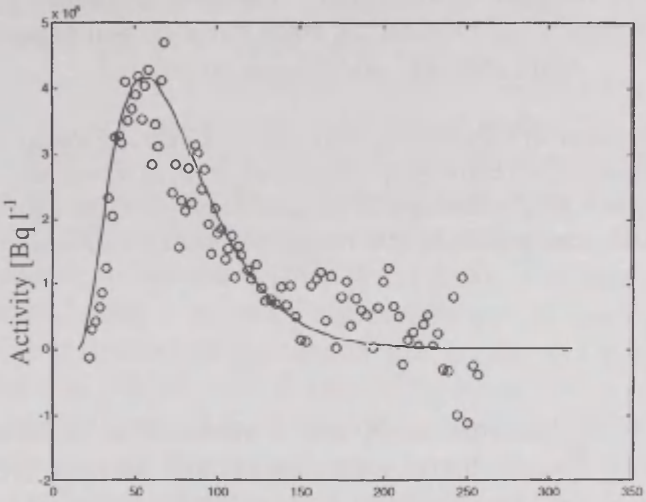

Figure 1. Breakthrough curves of P-32 at the outlet according to observations (circles) and model representation using (2).

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Nitrogen and phosphorus budgets in a horizontal subsurface flow wastewater treatment wetland

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Introduction

Although the main mechanisms of N and P removal in constructed wetlands (CWs) are quite well known, their quantification and budgets are unclear. In N removal, emissions of N_2O and N_2 (from both denitrification and nitrification) are believed to play the most important role (Vymazal *et al.*, 1998). Phosphorus is adsorbed in the filter material or precipitated, although the main issues are the longevity of sorption capacity and the recycling of the saturated material (Zhu *et al.*, 1998).

Few data are available concerning N_2O fluxes from CWs. This is mostly about the contribution from free water surface CWs (Wild *et al.*, 2002), and only two works (Fey *et al.*, 1999; Tanner *et al.*, 2002) consider N_2O fluxes from horizontal subsurface flow (HSSF) CWs. A few investigations have been carried out on N_2 emission from soils (Scholefield *et al.*, 1997), but we found no publications on N_2 fluxes from CWs. Several investigations have demonstrated that the assimilation of nutrients in plants in CWs plays a minor role. It is usually less than 10% of the nitrogen and in most cases less than 5% of the phosphorus removed in a constructed wetland (Brix, 1994). Microbial immobilization of N and P has been estimated to be at the same level as plant assimilation (Geller *et al.*, 1991).

There are only a few examples in the literature of long-term nitrogen and phosphorus budget in CWs, in which the main fluxes and pools have been measured (Kadlec and Knight, 1996; Vymazal *et al.*, 1998). Therefore, the main objective of this study was to determine the budget of both nitrogen and phosphorus in the Kodijärve HSSF CW, measuring the most important pools and fluxes of these elements: accumulation in soil and plants, immobilization by microorganisms and gaseous fluxes of N.

Materials and methods

Site description. The Kodijärve (HSSF) planted sand filter (in South Estonia, constructed in October 1996) purifies wastewater from a hospital for about 40 population equivalents (p.e.). Before entering the wetland system, wastewater flows through a dual-chamber septic tank. The system consists of two beds (chambers) each measuring 25×6.25×1 m, which are filled with coarse sand. The wastewater enters the system through a splitter chamber that divides the water over two 90° triangular measuring weirs into the beds where it seeps into the sand through the inlet pipes installed in the peripheral sites of the beds. The outlet pipes installed in the centre collect the purified water at the outlet well where the stand-pipes allow regulation of the water table in the beds. The base and sides of the chambers are isolated with a polyethylene membrane. From the outlet well the water flows through a channel to the natural reed stands on the lakeshore. In May 1997, the right bed was planted with *Typha latifolia* (cat-tail) and the left with iris plants (*Iris pseudacorus*) and reeds (*Phragmites australis*). Since the planting, the *Typha* and *Iris* have almost disappeared, and the right bed is dominated by wood club-rush (*Scirpus sylvaticus*) while reed is dominant in the left bed. Also, *Urtica dioica* (nettle), *Epilobium hirsutum* (hairy willow-herb) and single individuals of other species are presented in the left bed. The left bed, which has finer filter material, has more surplus moisture conditions and is hereafter referred to as the wet bed. The right bed, which has coarser material and drier conditions, is called the dry bed.

Water sampling and analysis

During each gas sampling session we measured the pH, temperature, conductivity, dissolved O_2 and redox potential in the 50mm polyethylene sampling wells installed in both beds (1–9 in the right bed and 10–18 in the left bed) using Mettler-Toledo and Evikon portable equipment. We also measured the water table, and took samples from piezometers and the inflows and outflows of both beds for further analyses for BOD_7, NH_4-N, NO_2-N, NO_3-N, total N, PO_4-P and total P (all according to APHA, 1989) in the lab of South Estonian Environmental Research Ltd.

Gas and soil sampling and analysis

Two methods – the “closed chamber” (“closed soil cover box”) method (Hutchinson and Livingston, 1993) and the He-O method, after Scholefield *et al.* (1997) were used to measure N_2O, N_2 and CH_4 emission. Gas samplers (closed chambers) were installed in 5 replicates above the inlet and outlet pipes of both beds. Gas sampling was carried out once a month in October and November 2001 and in March, May, June, July and September 2002. The trace gas concentration in the collected air was determined using the gas chromatographic system in the lab of the Institute of Primary Production and Microbial Ecology, at the Centre for Agricultural Landscape and Land Use Research (ZALF), Germany. Intact soil cores for the use of the He-O method were sampled from the topsoil (0–10 cm) in closed chamber locations, after gas sampling was completed. Soil samples were weighed, kept at a low temperature (4°C), and transported to the laboratory of ZALF. A detailed description of the gas analysis is given in Mander *et al.* (2003). Simultaneously, the NH_4-N and NO_3-N concentrations in soil samples was analysed using the Kjeldahl method (APHA, 1989). In addition, in every October since 1997, complex soil samples have been taken at 3 different depths (0–10, 30–40 and 50–60 cm) around the sampling wells to analyse for Kjeldahl N, lactate soluble P and organic C (ignition loss) in the Laboratory of Plant Biochemistry of the Estonian Agricultural University (LPB-EAU; Mander *et al.*, 2001).

Phytomass sampling and analysis

The biomass samples were collected from 5 plots of both beds in July 2001 and 2002, during the maximum flowering time of the dominant plant species. Sample plots were chosen in typical areas of the community. In the dry bed all plots were dominated by the wood club-rush, while in the wet bed three plots were dominated by wood club-rush and two plots by reed. The above-ground biomass and the litter from the same year were collected from quadrates 0.5x0.5 m. Below-ground root biomass was collected from soil cores taken by auger in 10 cm layers to a depth of 50 cm from each sampling plot.

Microbial analysis

In October 2001, complex soil samples for the microbial analyses of soil characteristics were collected with a soil core drill from 0–10 and 20–30 cm of 18 sampling sites. Samples for other analyses were stored in 500 ml plastic boxes at room temperature. Analyses were carried out in the Laboratory of the Environmental Protection Institute of the Estonian Agricultural University. Microbial biomass-N was measured using the Fumigation-Extraction Technique coupled with Ninhydrin–Reactive N detection (Öhlinger, 1996). The amount of P immobilized into the microbial consortia of the soil was determined using the Fumigation-Extraction Technique.

Results and discussion

The most important flux in N budget in 2001 was in N_2 emission (12.4 kg yr^{-1} or 41.7%), followed by microbial immobilization (8.7 kg yr^{-1}), accumulation in soil (4.2 kg yr^{-1}), plant assimilation (4.1 kg yr^{-1}) and N_2O emission (0.35 kg yr^{-1}; Figure 1). Due to difficulties in quantifying annual fluxes of phosphorus, this budget was made for the period 1997–2001 (Figure 2). Major pools of P are:

accumulation in soil as lactate-soluble P via adsorption (46.5 kg; 88.1%), plant accumulation (3.2 kg), microbial immobilization (2.3 kg), and other pools (e.g., precipitation with Ca; 0.8 kg).

Average annual N and P removal from the system was 13.2 and 6.8 g m^{-2} yr^{-1}, respectively. This is a satisfactory value for phosphorus (Kadlec and Knight, 1996), although it can be significantly higher for nitrogen (Vymazal *et al.*, 1998). The main problem is the lack of oxygen in the filter system, which does not allow the nitrification of ammonia into nitrates. To solve this problem, a pre-treatment vertical filter bed must be constructed between the septic tank and HSSF.

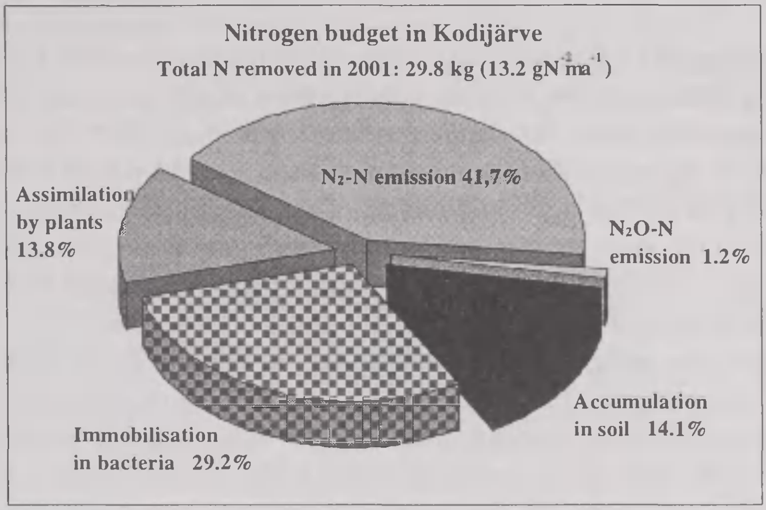

Figure 1. Nitrogen budget in the Kodijärve HSSF CW.

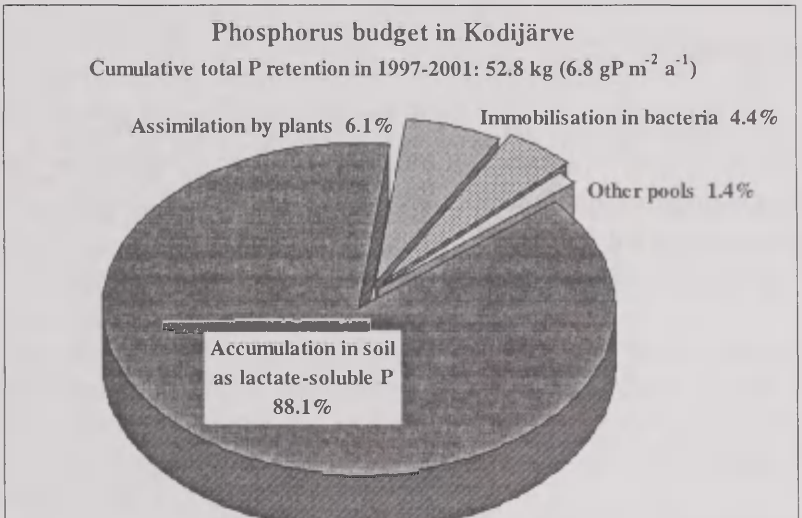

Figure 2. Phosphorus budget in Kodijärve HSSF CW.

In addition, phosphorus adsorption in the sand filter has been decreasing. On the one hand, this is related to decreased initial loading. On the other hand, the increasing outwash of Fe (Mander *et al.*, 2001) may be a sign of decreasing adsorption capacity. Thus an additional sorption system is needed to maintain and enhance P retention.

Plant uptake and microbial immobilisation played a minor role in the budget of both N and P. This has also been reported by other investigations (Brix, 1994).

Conclusions

A large proportion (41.7%) of the total removal of nitrogen in 2001 (29.8 kg N yr^{-1}) was emitted as dinitrogen (N_2), which is harmless to the atmosphere. Only 1.2% of the total N removed was the dangerous trace gas N_2O. We can consider that emission rates of N_2 and N_2O from normally loaded (<0.12 p.e. m^{-2}) HSSF CWs is about 400 and 10 kg N ha^{-1} yr^{-1} respectively. The majority of cumulative P retention in 1997–2001 was due to soil adsorption: 46.5 kg P a^{-1} (88.1%). However, decreasing annual P adsorption shows that planted soil filters such as the Kodijärve HSSF CW can be saturated with P during 5–6 years of operation.

To enhance the efficiency of purification, a vertical flow filter bed at the beginning of the system and another filter bed at the outflow, filled with calcareous fly ash from the oil-shale burning process, were established in August 2002. The preliminary results show better performance in both nitrogen removal (due to better aeration and increased nitrification) and phosphorus (sedimentation in the calcareous filter).

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Gaseous fluxes from subsurface flow constructed wetlands for wastewater treatment

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Introduction

Relatively few studies have been carried out on N_2O and CH_4 fluxes from constructed wetlands (CWs) for wastewater treatment. Most of the data are available on free water surface (FWS) CWs' contribution to N_2O (Bachand and Horne, 2000; Lund *et al.*, 2000; Spieles and Mitsch, 2000; Wild *et al.*, 2002) and CH_4 (Tanner *et al.*, 1997; Tai *et al.*, 2002; Wild *et al.*, 2002) emissions. Only two works (Fey *et al.*, 1999; Tanner *et al.*, 2002) considered the N_2O fluxes from subsurface flow constructed wetlands. We have not found any published materials on N_2 emissions from CWs.

The main objectives of this research were: (1) to measure – using two different methods – the "closed chamber" method and a new He-O method – N_2O, N_2 and CH_4 fluxes from two subsurface flow CWs for municipal wastewater treatment, (2) to compare N_2O, N_2 and CH_4 fluxes and their global warming potential (GWP) from CWs.

Materials and methods

We measured the gas emissions in two types of constructed wetlands located in the southern part of Estonia: (1) in the Kodijärve horizontal subsurface flow (HSSF) planted sand filter (constructed in October 1996; located in Tartu County)) that purifies the wastewater from a hospital for about 40 persons (see Mander *et al.*, 2001 for detailed description), and (2) in the hybrid treatment wetland system in Kõo, Viljandi County, which consists of a two-bed vertical subsurface flow filter (VSSF; 2×64 m^2, filled with crashed limestone, ø 5–10 mm, planted with common reed, *Phragmites australis*), a horizontal subsurface flow filter (HSSF; 365 m^2, filled with 15–20 mm crushed limestone, planted with cat-tail (*Typha latifolia*) and reed, and two FWS beds (3600 and 5500 m^2, planted with *T. latifolia*). The Kõo hybrid system was constructed in 2000 for the purification of the raw municipal wastewater generated by about 300 population equivalents (PE).

For the measurement of N_2O, N_2 and CH_4 emission two methods – the "closed chamber" ("closed soil cover box") method (Denmead and Raupach, 1993; Hutchinson and Livingston, 1993) and the He-O method (Butterbach-Bahl et al., 1997; Scholefield et al., 1997) were used. The latter was used especially for the measurement of N_2 fluxes. In Kodijärve, gas samplers (closed chambers) were installed in 5 replicates on the inlet and outlet pipes of both beds. In the hybrid wetland system in Kõo, 10 gas samplers were installed in the vertical flow filter (4 in each bed) and 15 in the horizontal flow filter (5 on two inlet pipes and 5 on the outlet pipe). At the end of the 1 hr measuring period gas samples were taken from the enclosures of samplers by previously evacuated gas bottles. Gas sampling was carried out on the following time schedule: once a month in October and November 2001, and March, May to December 2002, and January to March 2003.

Intact soil cores (diameter 6.8 cm, height 6 cm) for use with the HeO method were sampled from the topsoil (0–10 cm) at gas sampler (closed chamber) sites, after gas sampling was finished, in the following order: (1) in Porijõgi – in September 2000, October and November 2001, and March 2002; (2) in Gumnitz – in September 2000, January, May and October 2001, and May 2002; (3) in Kodijärve and Kõo – in October and November 2001, and March 2002. Soil samples were weighted, kept at low temperature (4°C), and transported to the laboratory of the ZALF. At the lab, intact soil cores were introduced in a special incubation vessel sealed using gas. The procedures used for the determination of the actual gas emission rate are described by Mander et al. (2003). Simultaneously, the NH_4–N and NO_3–N concentration in soil samples was analysed using the Kjeldahl method.

Results and discussion

We found a remarkable variability in the average emission rates of N_2O-N, N_2-N and CH_4-C, ranging from 1 to 2600, 170 to 130,000 and –1.7 to 87,200 $\mu g\ m^{-2}\ h^{-1}$ respectively. According to the Duncan test, a significantly higher release of all gases from CWs was observed during the warmer period (Figure 2A-C), although the N_2O flux showed no significant correlation with water temperature. In the microsites above the HSSF inflow in both Kodijärve and Kõo, the time-dependence of N_2 and CH_4 emission was extremely remarkable. The very cold winter of 2002/2003 with air temperatures from –15 to –25°C for almost two months apparently influenced both the efficiency of water purification and also gas emissions. As with purification performance, gaseous emissions in the spring and early summer were significantly lower than in the autumn.

The average flux of N_2O-N from the microsites in the Kodijärve HSSF CW and Kõo hybrid CW ranged from 27 to 370 and from 72 to 500 μg N_2O-N m^{-2} h^{-1} respectively. The average N_2-N flux from the microsites in Kodijärve was 2–3 magnitudes higher than the N_2O flux, ranging from 19,500 to 33,300 μg N_2-N m^{-2} h^{-1}. Differences in individual values of N_2 fluxes from replicate soil cores varied greatly, from 170 to 130,000 μg N_2-N m^{-2} h^{-1}, but the variations were statistically non-significant. In contrast to the N_2 emission, we found significant differences in average N_2O fluxes between the microsites: about 30 kg ha^{-1} yr^{-1} from chambers installed above the inflow pipes and <5 kg ha^{-1} yr^{-1} from chambers above the outflow pipes (Figure 1A). In Kõo, according to the Wilcoxon Matched Pairs Test, significant differences were found between the average dinitrogen emission rates: these values ranged from 3300 to 4680 kg ha^{-1} yr^{-1} above the inflow pipes of the horizontal flow bed, and up to 260 kg ha^{-1} yr^{-1} above the outflow pipes (Figure 1B).

Like the other gases measured, CH_4 emissions showed great variability in both time and space. The average methane emission from the microsites in the Kodijärve HSSF CW and the Kõo hybrid CW ranged from 31 to 12,100 and from 950 to 5750 μg CH_4-C m^{-2} h^{-1}, respectively. These values are 2–3 times higher than have been reported on re-flooded fens (Augustin *et al.*, 1996) or constructed cat-tail wetlands (Wild *et al.*, 2002), but up to 5 times lower than has been observed in FWSW-s for wastewater treatment (Tai *et al.*, 2002). With respect to the differences in CH_4 flux between the microsites, we found quite a similar pattern with nitrogen gas fluxes. According to the Wilcoxon Matched Pairs Test, significantly more methane was released from the microsites situated above the inflow pipes of both HSSF CWs (500–1200 kg ha^{-1} yr^{-1}) than from the microsites above the outlet pipes (<150 kg ha^{-1} yr^{-1}; Figure 1C). This is consistent with the significant rank correlation between the water table and the CH_4 flux (Mander *et al.*, 2003). Likewise, this relationship has often been mentioned in other studies of wetlands (Cao *et al.*, 1998). Similarly, the same authors have found temperature to be an important environmental factor influencing methane emission. In addition, methane release is positively influenced by suspended solids, NH_4-N, total N, PO_4-P and total P, and BOD_7 level in wastewater. At the same time, a significant negative rank correlation was found between CH_4 flux and the NO_2-N and NO_3-N concentrations in water (Mander *et al.*, 2003).

Several investigations have demonstrated that N_2O emission does not clearly depend on soil temperature, and the release of this gas from soil in cold periods can be as high or even higher in winter as it is in summer (Augustin *et al.*, 1996; Fey *et al.*, 1999). In our HSSF CW, the water temperature was always >2°C and the soil temperature above the water table was >0.5°C (Mander *et al.*, 2001), enabling the performance of bacteria responsible for both nitrification and denitrification.

The N_2O emission rate has been found to increase with decreases in the level of the water table (Martikainen *et al.*, 1993). In our earlier investigations we have found a significant correlation between the N_2O flux from all replicate chambers and the level of the water table in both beds. An elevated water table (lower depth in cm) increased the dinitrogen flux, especially above the inflow pipes of both beds. In addition, there was a significant rank correlation between the N_2O and N_2 emission and various physical and chemical parameters of water measured in sampling wells (Mander *et al.*, 2003).

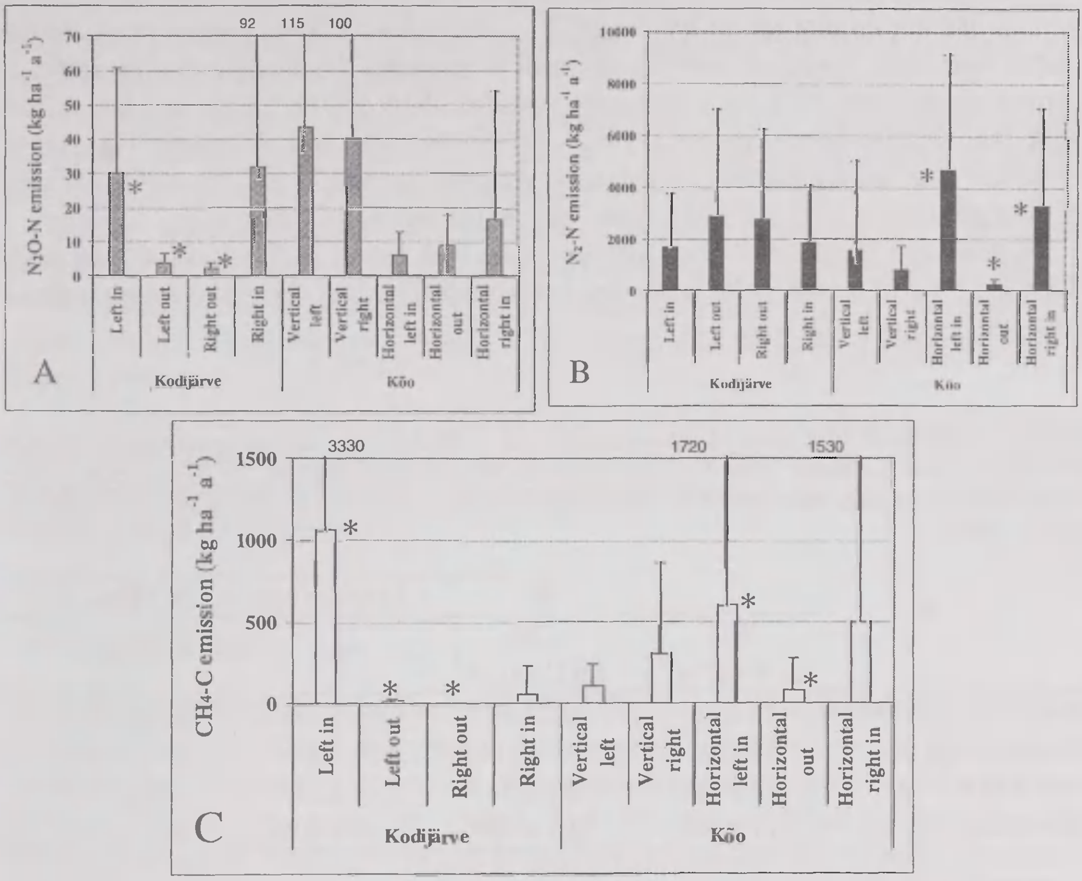

Figure 1. Emission rates of nitrous oxide (A), dinitrogen (B) and methane (C; mean±SD) from microsites in the Kodijärve HSSF CW and Kõo hybrid wetland system. * – significantly differing value (p <0.05) with at least two other microsites according to the Wilcoxon Matched Pairs Test.

Cumulated emission and global warming potential

The cumulated emission of all studied gases varied from 18.4 to 25, from 2326 to 3040 and from 309 to 351 kg ha^{-1} yr^{-1} for N_2O, N_2 and CH_4 respectively. The emission level in Kõo always exceeded the relevant values in Kodijärve, which is probably due to the relatively high loading of the vertical flow system (only two beds of 64 m^2 for about 300 PE). In Kodijärve the nominal loading is only 20–40 PE per 312.5 ha. When properly working, however, the vertical flow system can have a relatively small area, although this seems to enhance N_2O emission.

Regarding CH_4 flux, it is crucial to avoid clogging both vertical flow and horizontal flow filters: this might help with a higher N_2 flux and correspondingly lower N_2O flux; however, it significantly increases methane emissions. Sometimes such clogging took place in all of the CWs studied, which probably led to high CH_4 emission values (Figure 1).

Cumulated greenhouse gas fluxes were converted into CO_2 equivalents using the calculation factors given by the IPCC (1995). This allows one to evaluate the global warming potential (GWP) of studied systems. In Kodijärve, the average nitrous oxide from both beds was quite similar: 4.16 ± 7.68 t CO_2 ha^{-1} yr^{-1} in the right bed and 4.16 ± 4.16 t CO_2 ha^{-1} yr^{-1} in the left bed. Methane flux rates, however, showed significant differences, ranging from 0.74 ± 2.18 t CO_2 ha^{-1} yr^{-1} in the right bed to 13.2 ± 28 t CO_2 ha^{-1} yr^{-1} in the left bed (Table 1).

In Kõo, the highest GWP of nitrous oxide was found in the vertical flow beds (13.4 ± 21.1 t CO_2 ha^{-1} yr^{-1}), while the horizontal flow bed showed a high methane flux (9.73 ± 19.1 t CO_2 ha^{-1} yr^{-1}).

Table 1. Cumulated flux rates of nitrous oxide and methane from various constructed wetlands and riparian alder forests from October 2001 to March 2003 (kg ha^{-1} yr^{-1}; mean±SD). The conversion of the flux rates into CO_2 equivalents is given with 320 for N_2O and 24.5 for CH_4 (IPCC, 1995).

			CO_2 equivalents (10^3 kg ha^{-1} yr^{-1})	
	N_2O kg N ha^{-1} yr^{-1}	CH_4 Kg C ha^{-1} yr^{-1}	N_2O	CH_4
Kodijärve right bed	13 ± 24	30 ± 89	4.16 ± 7.68	0.74 ± 2.18
Kodijärve left bed	13 ± 13	537 ± 1143	4.16 ± 4.16	13.2 ± 28.0
Kõo VSSF #	42 ± 66	211 ± 347	13.4 ± 21.1	5.20 ± 8.50
Kõo HSSF	11 ± 18	397 ± 780	3.20 ± 5.76	9.73 ± 19.1

– average of left and right beds

The results of other investigations using a similar measurement technique have revealed somewhat lower emission values than ours (see Wild *et al.*, 2002 and Augustin *et al.*, 1996).

Conclusions

In both CWs we found a remarkable variability in the average emission rates of N_2O-N, N_2-N and CH_4-C, ranging from 1 to 2600, 170 to 130,000 and –1.7 to 87,200 $\mu g\ m^{-2}\ h^{-1}$ respectively. A significantly higher release of all gases studied was observed during the warmer period, although the N_2O flux showed no significant correlation with water temperature. Apparently, the very cold winter of 2002/2003 with air temperature from –15 to –25°C for almost two months did influence both water purification efficiency and gas emission. As in the case of purification performance, gaseous emissions in spring and early summer were significantly lower than in autumn.

The most intensive flux of N_2O and CH_4 was observed in chambers installed above the inflow pipes of horizontal flow beds. The vertical flow wetland did emit significantly more N_2O than the horizontal flow beds.

Water table increase in the horizontal flow systems may not significantly influence the efficiency of water purification, although it does increase methane emissions by a few magnitudes. Thus it is very important to avoid the clogging of pipes, which is normally guaranteed by the regular cleaning of sediments from the septic tank.

Although the emission of N_2O and CH_4 from CWs was found to be relatively high, their global influence is not significant. Even if all global domestic wastewater were to be treated by wetlands, its share in the trace gas emission budget would be less than 1%.

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Recycling of treated effluents enhances reduction of total nitrogen in vertical flow constructed wetlands

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Abstract

Uncontrolled discharge of nutrient rich wastewaters to the environment is a common source of pollution and it is partly responsible for the eutrophication of surface waters, as well as the degradation of the quality of groundwater. Constructed wetlands have proven to be an effective and affordable alternative to reduce BOD, suspended solids and even to reduce the concentration of nutrients from domestic wastewaters, if the design, operation and establishment of the system meet certain conditions.

Vertical flow constructed wetlands efficiency in producing well nitrified effluent is widely documented (Cooper, 1998; IWA, 2000; Brix et al., 2002). Although, and due to the nature of the nitrification-denitrification process and because the treated effluent is oxygen saturated and with the low availability of carbon in the wetland, the removal of total nitrogen is bound to be limited. We hypothesised that recycling a determined volume of the fully nitrified effluent to the pre-treatment unit (sedimentation tank), where conditions are more favourable for denitrification, the removal of total nitrogen from the wastewater can be enhanced.

We monitored the process in a full scale experimental vertical flow constructed wetland, consisting of a sedimentation tank, two vertical flow beds of 10 and 5 m^2, a phosphorous filtering unit and the necessary pumping equipment and controllers to manage the loading and recycling volumes. The results have shown a correlation between total nitrogen removal and the recycled volume, while maintaining good removal rates of all the typical domestic wastewaters quality parameters. The total-N removal rate reached levels above 60%. These results have been validated in an operational single household autonomous system that has operated with recycling.

The study took place in the vicinity of Århus (DK), between October 2002 and January 2003. The study consisted of separate; three consecutive days grab

sample campaigns, which evaluated the overall performance of the system. Samples were taken from 5 different points located along the water flow line as shown in Figure 1. They were evaluated for conductivity, temperature, dissolved oxygen, pH on site and for COD, Suspended solids (SS), NH_4, NO_2, NO_3 and TKN at the Department of Plant Ecology of Århus University. During all the campaigns, the wastewater treatment system operated with approximately the same inlet flow (aprox 500 Ld^{-1}), while the recycled percentage of treated wastewater varied (0%, 100%, 200%, and 300% respectively).

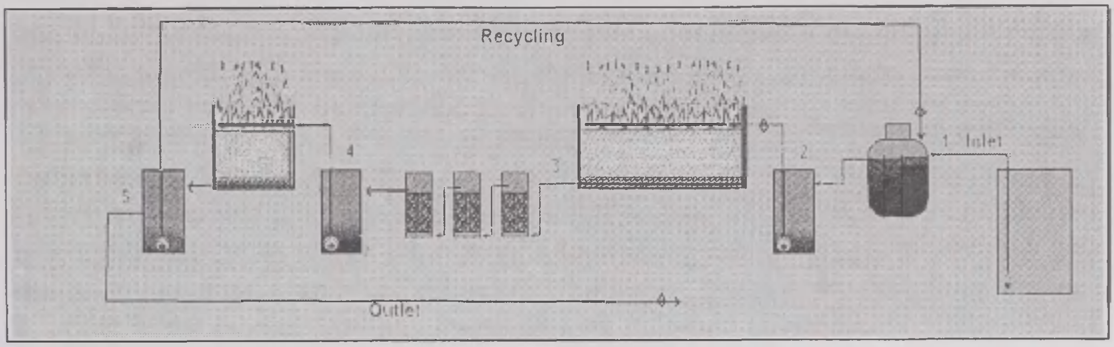

Figure 1. Sampling points along the water flow line of the system: (1) inlet sedimentation tank, (2) outlet sedimentation tank, (3) outlet first bed, (4) inlet second bed and (5) outlet second bed.

Since the system is fed by a pump installed at the inlet of a local municipal wastewater treatment plant, the volume of influent water to the system can be slightly affected by the water level at the pumping well; therefore the loading during the sampling campaigns was not identical, nevertheless it was equivalent. The percentage of recycled treated water was in the range of the planned volumes. The hydraulic area loading to the beds varied according to the increased recycled water and even though high loading rates occurred, especially in the second bed, no clogging was registered during the test. Table 1 shows the average loading and recycling, as well as the corresponding recycling percentage and hydraulic loading rate.

Table 1. Average and standard deviation of inlet measured flows, recycled volumes and surface loadings, for each one of the campaigns.

Campaign	Inlet ($l\ d^{-1}$)	Recycling ($l\ d^{-1}$)	Recycling (%)	Total Q ($l\ d^{-1}$)	Loading CW1 ($cm\ d^{-1}$)	Loading CW2 ($cm\ d^{-1}$)
1	503 ± 25	0	0	503 ± 25	5.6 ± 0.3	11.2 ± 0.6
2	585 ± 42	592 ± 36	101%	1177 ± 75	13.2 ± 0.8	26.2 ± 1.5
3	637 ± 94	1257 ± 55	198%	1893 ± 113	21.1 ± 1.3	42.0 ± 2.5
4	624 ± 12	1855 ± 138	297%	2479 ± 149	27.6 ± 1.7	55.0 ± 3.3

Given that the sampling took place during the cold season, the environmental temperatures as well as the water temperature were low. Water temperatures ranged between 7 and 18°C, and no freezing events within the system were recorded. Dissolved oxygen in the wastewater was affected by the recycled volume; the oxygen saturation percentage measured at the sedimentation tank outlet (place where recycling occurred) varied from 0%, when no recycling occurred to up to 60% when the recycling volume was 300%. The overall removal of COD, and suspended solids (SS) in the system was satisfactory, as expected from a vertical flow constructed wetland. Both parameters were highly influenced by the dilution effect generated by the recycling of treated wastewater to the sedimentation tank and therefore, a high removal percentage was registered in the sedimentation tank. The removal percentage of COD and SS along the system was governed by the influent concentration. Figure 2a and 2b respectively show the removal performance of COD and SS for all the sampling points through all the campaigns.

To evaluate the removal of nitrogen in the system, the different nitrogen species were analysed separately, in order to determine the efficiency of the nitrification-denitrification process in the system. Regarding the nitrification of ammonia, and as expected form a vertical flow constructed wetland; the system was able to nitrify effectively during all the campaigns and consistently produce final effluents with ammonia concentrations below 0.1 mg l^{-1} NH_4-N. If the campaigns are analysed individually it should be mentioned that in the run without recycling there was a generation of ammonia in the sedimentation tank, an elimination of over 80% while trickling through the first bed and reaching a total 99,9% removal after the second bed. For the runs with recycling, the effect of dilution and the presence of dissolved oxygen in the sedimentation tank reduced the concentration of ammonia in the tank for all the cases (aprox 40%, 55%, 70% for each one of the recycling percentages). While in the first bed the measured ammonia concentrations for all the cases was always less than 1.0 mg l^{-1}, and below detection limits after the second bed, when finally disposed to the environment (Figure 2c).

Regardless of the operation mode full nitrification was accomplished in the system; therefore the nitrogen in the system is in the nitrite-nitrate form. In order to evaluate the removal capacity of total nitrogen of the system the denitrification conditions were favoured, by recycling the treated wastewater to the sedimentation tank. For all the campaigns the influent nitrate concentrations in the raw wastewater was low (<1.6 mg l^{-1}). For the run with 0% recycling no nitrate was generated in the sedimentation tank and the low content present in the raw wastewater was denitrified in the sedimentation tank. After the first bed and through the rest of the system, all the ammonia was converted to nitrate but no considerable reduction of nitrate was measured in the system. For the campaigns with recycling, nitrate removal always occurred in the sedimentation tank, although these removal percentages varied proportionally to the recycled volume as follows: 95% when 100% recycling occurred, less than 40% for 200% recycling and less than 5% for 300% recycling. For the campaign with 100% recycling the nitrate generated in the first bed was not eliminated at the end of the system. For the campaign with 200% and 300% recycling, and as mentioned before, the sedimentation tank was not able to eliminate all the NO_3. Additional NO_3 was produce in the bed, but considerable denitrification was also registered in the first bed.

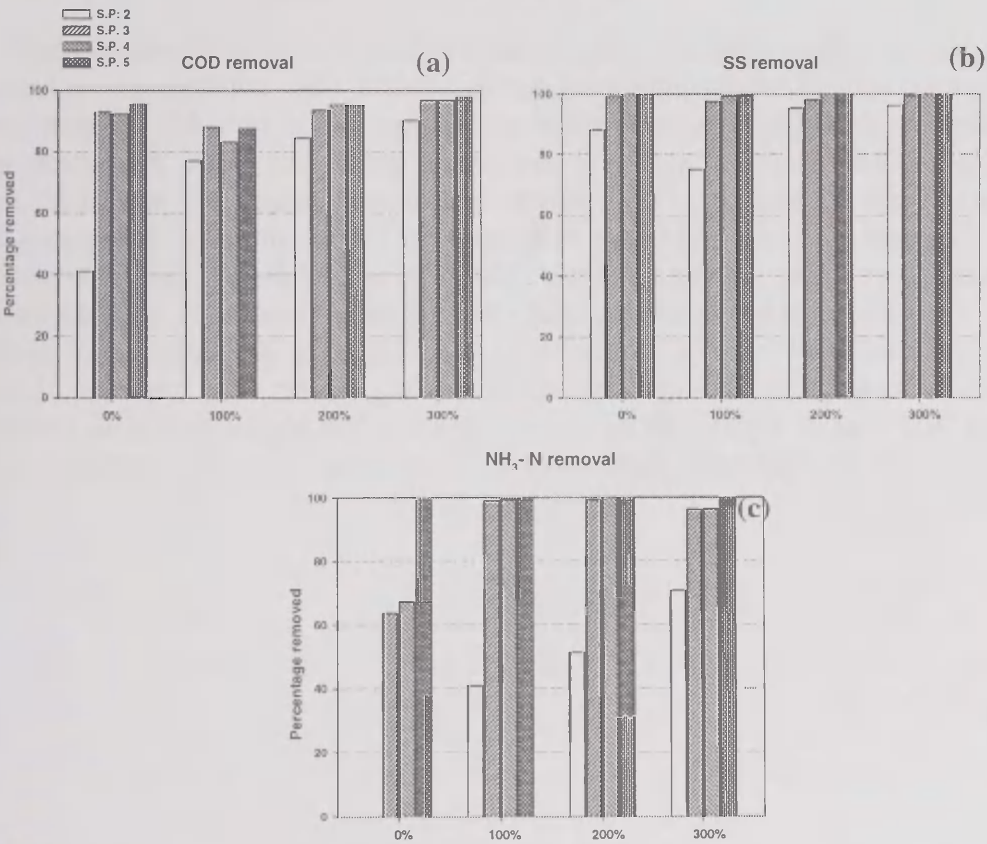

Figure 2. Percentage removals of COD, SS and NH_3 through all the campaigns and at all the sampling points (based on concentrations).

The overall removal of nitrogen in the system was clearly dependant on the recycled volume. During the campaign without recycling the removal of total nitrogen was negligible. Through the runs with recycled effluent the removal was significant and as recycled volume increased so did the removal of total nitrogen. Table 2 shows the average percentage of total nitrogen removed for each of the campaigns.

Table 2. Average of total nitrogen percentage removal measured at the sampling points (based on concentrations).

Sampling points	Percentage recycled			
	0%	100%	200%	300%
2	9	44	59	58
3	2	53	69	67
4	4	49	68	70
5	1	52	66	68

The experiment shows that recycling nitrified effluent to the sedimentation tank is an effective way to reduce total nitrogen in vertical flow constructed wetlands. Considerable denitrification occurred in the sedimentation tank for the recycling campaigns. Denitrification was also registered within the beds. These can be justified by the occurrence of both anoxic and oxygen saturated zones in the the beds. According to our data, the elimination of total nitrogen eliminated is increased as recycling volume increases. Although several details should be considered: a) as higher volumes are recycled, the residence time in the sedimentation tank is reduced, with the possibility of negatively affecting the performance of the primary treatment. b) as higher hydraulic loading occurs, the capacity of the vertical flow bed to nitrify can be compromised. c) the higher hydraulic loading can also affect the hydraulic capacity of the bed to the limit of overloading and flooding.

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Fluctuations in the organic matter decomposition rate and nitrogen speciation in vegetation and outside vegetation seasons in a hybrid wetland system

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Abstract

Municipal sewage from the village of Wiklino near Slupsk (located about 100 km from Gdansk) is discharged to a two-chamber septic tank with a retention time of about 2.3 days. Then the sewage is periodically pumped into the wetland facility. The dose of sewage is approximately 300 dm^3, and the discharge time is 5–6 minutes. The first stage of biological treatment is carried out in a horizontal flow bed (HF-CW I). Then the sewage constantly outflows to two parallel units of the vertical flow bed (VF-CW). The flow of sewage took place by gravitation until modernisation in spring 2000. The next stage of treatment (following VF-CW bed) is the second horizontal flow bed (HF-CW II). A map of the facility is presented in Figure 1.

In spring 2000 modernisation of the sewage supply system in the VF-CW bed was performed. The pressure supply system for discharging sewage to both units of the bed was constructed. Installation of a pump allowed for portion dosing of sewage. Portion supply improves oxygen conditions in the bed. Additionally, valves were installed on the pipes discharging sewage to both units of the bed. This enabled the beds to work in two-week periods one after another. The facility's characteristic operating parameters are given in Table 1.

During the vegetation season, the average amount of discharged sewage was 19.0 $m^3\ d^{-1}$, which was slightly higher than the average amount of sewage discharged outside the vegetation season (18.4 $m^3\ d^{-1}$). The average amounts of sewage flowing out of the facility were equal to 12.9 $m^3\ d^{-1}$ in the vegetation season and 13.9 $m^3\ d^{-1}$ outside the vegetation season.

Mean samples of sewage were collected twice a month before modernisation (period I: from April 1998 to February 2000) and after modernisation (period II: from March to May 2000). The vegetation season lasted from April to October. The duration of the vegetation season was estimated on the basis of the vegetation time of the common reed and air temperature. It was assumed that sewage temperature is not a relevant criterion due to the fact that the temperature of sewage flowing out of the septic tank in winter was elevated. The average air

temperature in the vegetation season was 14°C. Outside the vegetation season it was 3.8°C. The location of sampling points is presented in Figure 1.

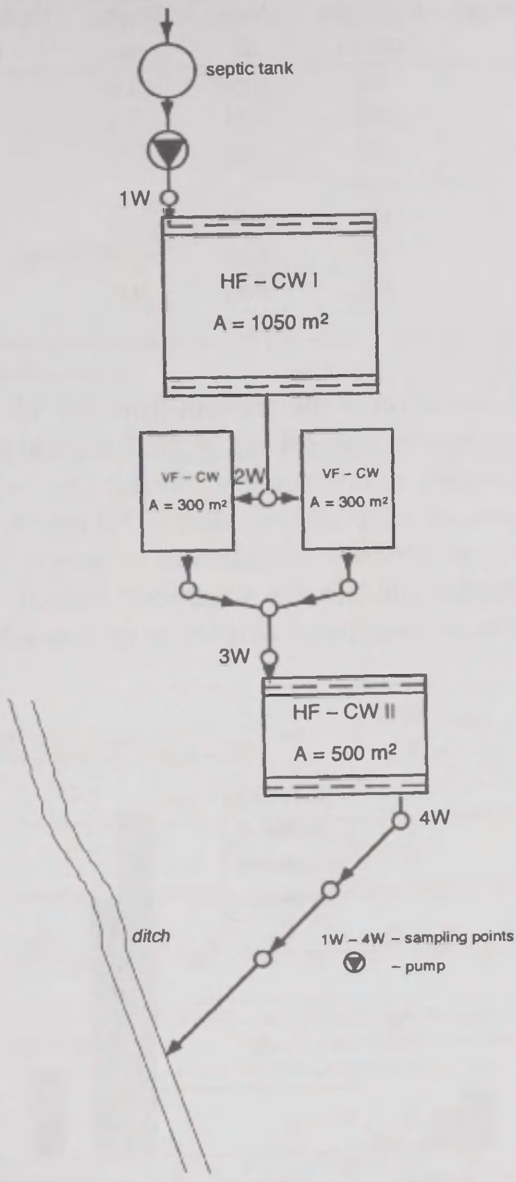

Figure 1. Scheme of wetland facility in Wiklino.

The average concentrations of nitrogen compounds in sewage flowing into and out of the facility are presented in Figure 2. The average concentration of nitrogen compounds in sewage inflowing to the facility during vegetation season were lower than outside the vegetation season.

Table 1. Characteristics of wetland facility. HF – horizontal flow, VF – vertical flow, PE – population equivalents.

Facility	Flow of sewage $m^3 d^{-1}$	Configu-ration	Area, m^2	Depth, m	Hydraulic load, mm d^{-1}	Unit area, $m^2 PE^{-1}$
Wiklino I – before moderni-sation	18.7	HF	1050	0.6	17.9	7.0
		VF	624	0.4	24.1	4.0
		HF	540	0.6	27.5	3.4
Wiklino II – after moderni-sation	18.6	HF	1050	0.6	17.7	7.0
		VF	624	0.4	46.9	2.0
		HF	540	0.6	25.7	3.4

Average nitrate concentration in the effluent from the VF-CW bed was 4.0 mg N dm^{-3} in the vegetation season and 2.4 mg N dm^{-3} outside the vegetation season (Figure 3). Average oxygen concentrations during the vegetation season and outside the vegetation season were almost equal – 3.0 mg O_2 dm^{-3} and 3.1 mg O_2 dm^{-3} respectively. The concentration of dissolved oxygen in sewage after the VF-CW bed was slightly higher outside the vegetation season, probably due to the lower temperature of sewage, resulting in higher oxygen solubility (Figure 3).

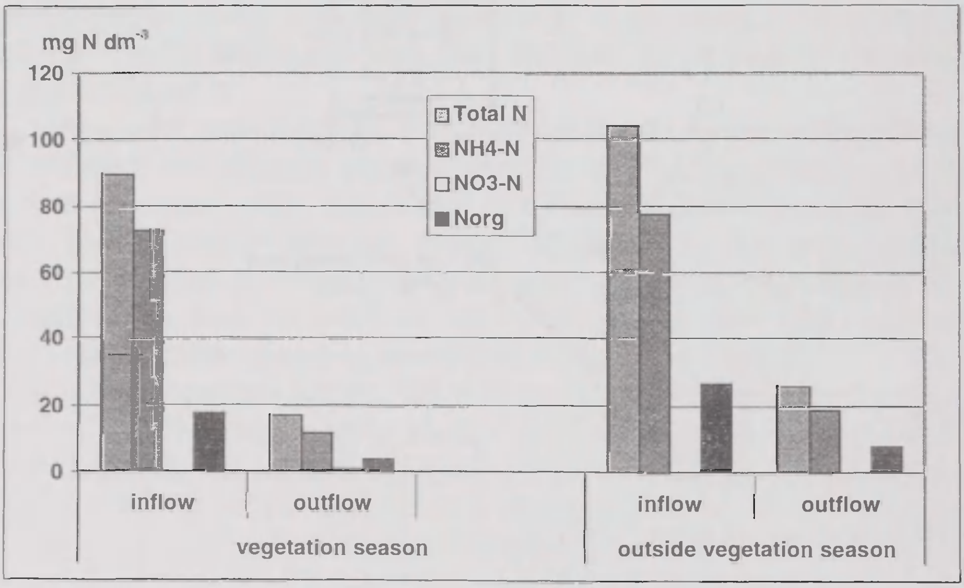

Figure 2. Average concentrations of nitrogen compounds in sewage flowing into and flowing out of the facility.

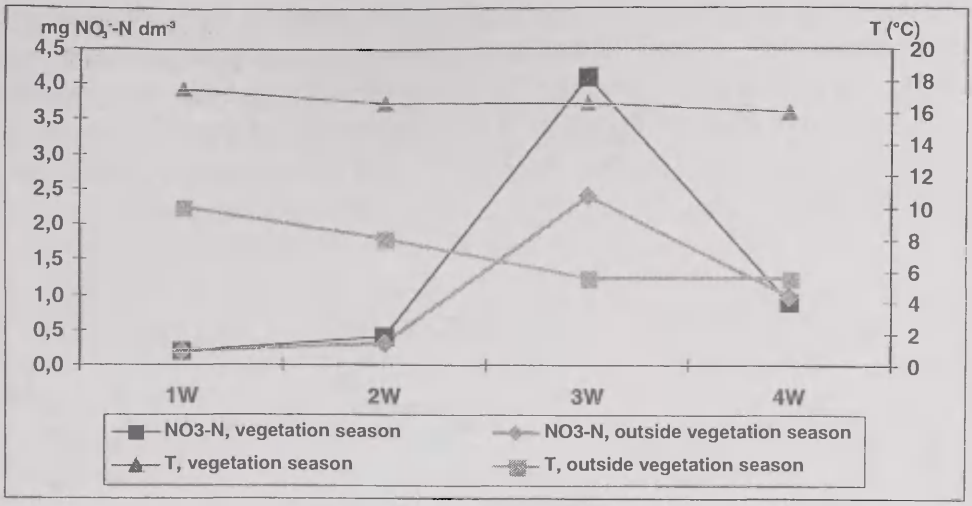

Figure 3. Average concentrations of nitrates and average temperature of sewage after subsequent stages of treatment.

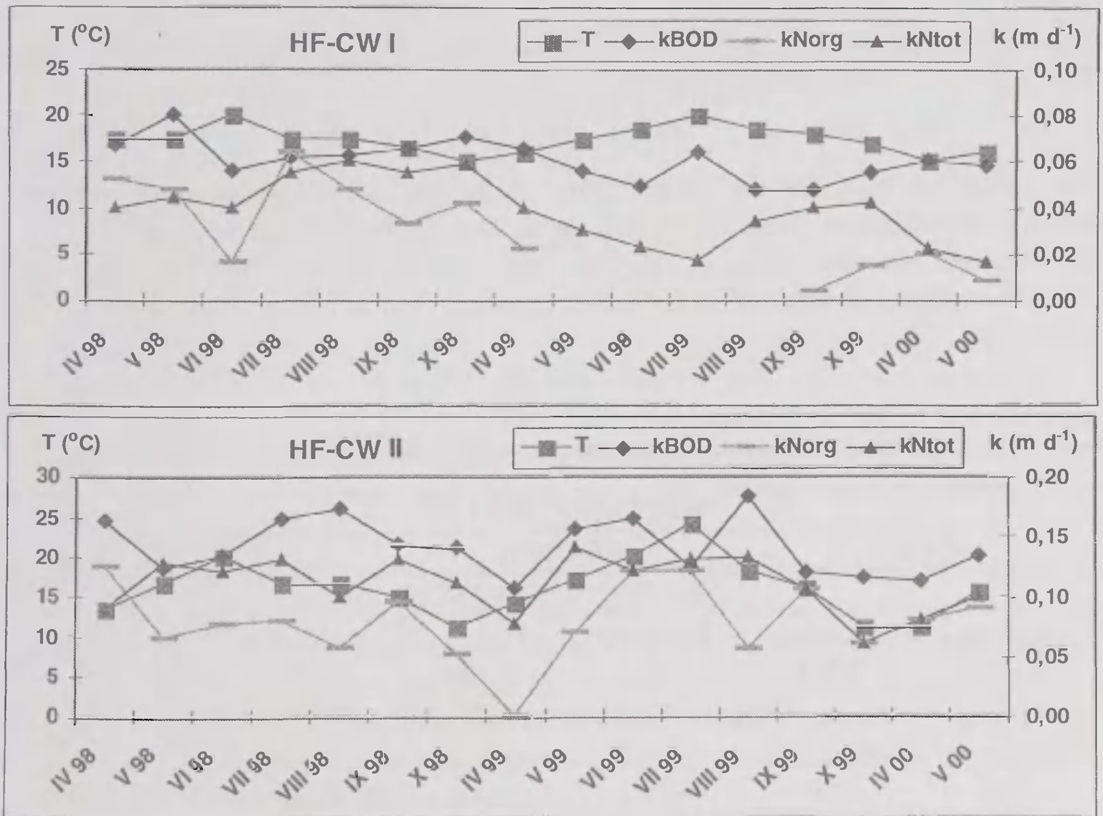

Figure 4. Fluctuations of reaction rate constants k_{BOD}, k_{Norg} and k_{Ntot} and temperature of sewage after HF-CW I bed during vegetation season.

The relationships determined between calculated constant rates of organic substance decomposition, organic nitrogen mineralisation and total nitrogen removal and temperature of sewage for HF-CW I and HF-CW II beds are presented in Figure 4 during the vegetation season. The relationships of modified reaction rate constants k_{BOD}, k_{Norg} and k_{Ntot} for the VF-CW bed for vegetation season are also presented in Figure 5.

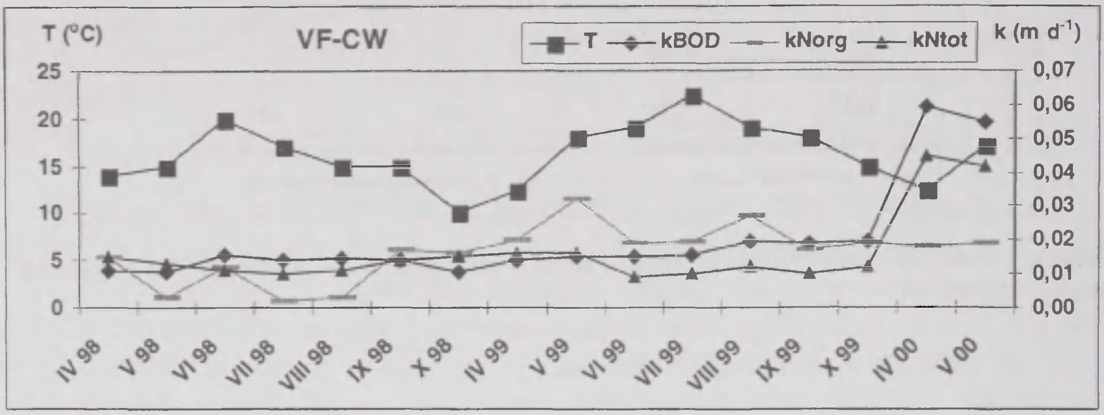

Figure 5. Fluctuations of reaction rate constants k_{BOD}, k_{Norg} and k_{Ntot} and temperature of sewage after VF-CW bed, during vegetation season.

The values of $k_{p(T)}$ for the temperature of 20^0C, taking into account average monthly temperatures, were calculated using the relationship $k_{p(T)} = k_{p(20)}\,(1.1)^{T-20}$. The obtained average values of reaction rate constants for the HF-CW I and HF-CW II beds and corresponding modified constant rates for VF-CW bed are given in Table 2.

Table 2. Average values of analysed reaction rate constants at the temperature 20^0C, d^{-1} and $m\ d^{-1}$.

WETLAND SYSTEMS					
HF-CW I		HF-CW II		VF-CW	
VEGETATION SEASON					
$k_{pBOD(20)}$	$kp_{Ntot(20)}$	$k_{pBOD(20)}$	$kp_{Ntot(20)}$	$k_{BOD(20)}$	$k_{Ntot(20)}$
0.122	0.094	0.27	0.124	0.031	0.025
OUTSIDE VEGETATION SEASON					
$k_{pBOD\ (20)}$	$kp_{Ntot(20)}$	$k_{pBOD\ (20)}$	$kp_{Ntot(20)}$	$k_{BOD\ (20)}$	$k_{Ntot(20)}$
0.071	0.045	0.111	0.062	0.019	0.019

The obtained values of reaction rate constants indicate that organic substance decomposition was the fastest process, while total nitrogen removal took place at a slightly slower rate. Mineralisation of organic nitrogen was the slowest process, both during the vegetation season and outside it. According to Obarska-Pempkowiak *et al.* (2002) it was also proven that a substantial proportion of organic nitrogen was ammonified in the septic tank. Temperature had a negligible influence on the effectiveness of the ammonification process. In the vegetation season the values of $k_{pBOD(20)}$ and $k_{pNtot(20)}$ in the HF-CW I bed were 0.122 and 0.094 m d^{-1} respectively, and were higher than outside the vegetation season – 0.071 and 0.045 d^{-1} respectively. The average values of the constant rates $k_{pBOD(20)}$ and $k_{pNtot(20)}$ were 42 and 52% higher in the vegetation season than outside it.

The average values of modified constant rates of organic substance decomposition, organic nitrogen mineralisation and total nitrogen removal at the temperature of 20°C for the VF-CW beds are presented in Table 2. Average constant rates $k_{pBOD(20)}$ and $k_{pNtot(20)}$ in the vegetation season were equal to 0.031 and 0.025 m d^{-1} respectively, and were higher than outside the vegetation season. Similar results were obtained by Birkedal *et al.* (1993) for 37 VF-CW beds in Denmark. The average value of the modified $k_{Ntot(20)}$ constant rate was 0.0247 m d^{-1} (Birkedal *et al.*, 1993). The values of $k_{BOD(20)}$ obtained during investigations carried out in Austria and Great Britain varied from 0.067 to 0.1 m d^{-1}, and were higher than the values calculated for the VF-CW beds in Wiklino. This could be caused by higher average air temperature (Haberl *et al.*, 1998; Cooper and de Maeseneer, 1996).

Conclusions

1. The amount of sewage discharged into the analysed wetland facility was lower and the concentrations of pollutants were higher in the vegetation season.
2. The temperature of air and sewage had a substantial influence on nitrogen compound removal. The average effectiveness of ammonia nitrogen removal in the vegetation season was about 85%, decreasing to 78% outside the vegetation season.
3. The VF-CW bed in both analysed periods of the year played an important role in oxygen supply to the sewage. Regardless of season and temperature, the concentration of dissolved oxygen in sewage increased from 0.5 mg O_2 dm^{-3} after the HF-CW I bed to about 3.0 mg O_2 dm^{-3} after the Vf-CW bed.
4. The reduced effectiveness of nitrogen transformation processes outside the vegetation season is confirmed by the values of reaction constant rates (about 60% of the values reached in the vegetation season).

Acknowledgements

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Ten years' experience of constructed wetlands in Poland

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Introduction

The treatment of wastewater using plants known as hydrophytes in constructed wetlands created specially for this purpose is a relatively new ecotechnology, which was launched in the 1960s, only to develop and gain wider application throughout the 1980s and 1990s in both Europe and the USA. In Poland one of the first such plants made use of surface flow and a multi-species array of plants to offer a third level of wastewater treatment at a conventional plant in Pruszków (Ozimek and Klekot, 1979). While the effects were spectacular, the method did not gain wider application, as was the case in Germany after the first Seidel tests (Seidel, 1966). The rebirth of interest in non-conventional wastewater treatment methods began in Western Europe at the end of the 1970s, as well as ten years later in Poland, where the political transformation of the country was required to encourage greater interest in the state of the environment. This led to a search for cheap and environmentally-friendly methods with which water pollution might be limited. Hydrophyte-based plants were one such method.

Furthermore, the period in question saw the appearance in Poland of a great many enterprises specializing in the design and building of such treatment plants, capitalizing as they did so on the experiences of the Western European countries and the USA. The first decade of constructed-wetland use in Poland has now passed, and thus the aim of the present work is to summarise Polish experiences in regard to use of this technique.

Natural wetlands

In Poland, natural wetlands (in the broad sense assigned to the term in accordance with the Ramsar Convention) are mainly used in a disorganised manner as receiving waters for sewage. An example might be the lakes of the Mazurian Lakeland, of which more than half received wastewater from towns, villages and tourist centres in the 1970s and 1980s (Pieczyńska and Ozimek, 1976). In many cases such activities led not only to a degradation of the littoral into which the waste went, but also to a limitation of the ranges of certain plants, an impoverishment of species composition, a curtailing of the growing season and a reduction in biomass (Ozimek, 1978; Ozimek and Kowalczewski, 1984; Kowalczewski and Ozimek, 1990). The fauna of the littoral was reconfigured (Pieczyńska et al., 1975), and it was by no means uncommon for the whole lake ecosystem to experience reduced transparency even in the centre, oxygen deficits in deeper parts and the mass appearance of the blue-green algae (as at Lakes Mikolajskie and Niegocin) (Gliwicz et al., 1980). In some cases, the excessive influx of organic matter into the littoral led to its saprotophication and hence to the disappearance of higher plants at the expense of the mass development of algae. On such a scale, field experimentation made it clear that wetlands with submerged macrophytes are not suitable for the treatment of raw wastewater. In such conditions the littoral does not serve as a filter, but rather acts as a trap for allochthonous matter. The effects of such degradation are persistent and difficult to remove, with data showing that a reduction of inputs following the construction of treatment plants does not allow the littoral biocoenosis to return to its initial state.

Other natural wetlands that are only periodically inundated are used in secondary or tertiary wastewater treatment. The effectiveness of the treatment possible in such wetlands depends on a host of factors, such as age, the sorption capacity of the substratum and the plant species composition. The role is not always played properly, such that areas of the kind may often serve as sources, rather than sinks, of biogenic compounds (Mosiej and Renman, 1995).

Thus natural wetlands need to be put to use very carefully, and only to a limited extent. In fact, each site needs to be subject to research specific to it that can forecast likely changes.

Constructed wetlands

There are no Polish data on exactly how many constructed wetlands are in use, or on their types. The number is hard to determine, but estimates that there are 100 such sites may be an underestimation, on account of the fact that quite a few of those in existence are by houses and hence go unregistered (since under the Water Law Act, no permit is required to operate those with a throughput of less than 5 m^3 d^{-1}). The present study is in fact based on data for 90 objects, with the primary source of information being the scientific publications and data from firms installing such systems.

Interest in hydrophyte treatment plants assumed greater dimensions in Poland at the end of the 1980s. A rapid increase in the numbers of such plants being established took place in the years 1991–94. A further phase of interest is likely to arise in connection with EU accession and the further adjustment of the environmental law, which requires the introduction of wastewater treatment plants in all localities with populations greater than 2000.

In Poland, hydrophyte-based treatment plants most often serve villages, small settlements or small towns (in which sewers either have or have not been installed – where they have not, the wastewater is transported in). Individual farms or groups of farms may also have them. The most limited representation characterrises plants of the least and greatest throughput (less than 5 m^3 d^{-1} and 1000 to> 2000 m^3 d^{-1} respectively). In turn, the greatest number of examples are of plants processing some 5–500 m^3 d^{-1}. Hydrophyte-based plants are mainly applied in the treatment of household sewage as the second biological stage, or else as a second plus third or as the third stage at conventional plants. The situation resembles that in other countries, although here greater use is made than elsewhere. To a limited extent the plants of this type are also used in treating specific kinds of wastewater that are precipitation waters, runoff from motorways, leaks from waste dumps and waters running off from a zoo. Finally, they have also been used in cleaning up small watercourses and in protecting drinking-water intakes (Obarska-Pempkowiak *et al.*, 2002), as well as in the utilisation of sewage sludges (Ozimek *et al.*, 2001; Hardej and Ozimek, 2002).

The hydrophyte systems installed most often are those with a subsurface flow and emergent macrophytes, as well as those with surface flow and pleustonic plants. Systems with subsurface flow were put in place in line with guidelines (Cooper, 1990), or else after Kickuth (1984). The largest number of plants in this group comprise those in which there is horizontal subsurface flow, though more objects with mixes of horizontal and vertical flow have appeared recently. Monocultures of *Phragmites australis* are most often used (in 90% of cases), while *Salix vimmalis, Typha latifola* and *Glyceria maxima* are applied less often. It is rare to find multi-species plantings. The plants involved are most often reproduced vegetatively from rhizomes or stems, or more rarely from seedlings grown from seed. The material for planting is of clones best adjusted to local conditions. In exceptional cases, plants are transferred from one region of Poland to another (e.g. reeds from the Bay of Puck to Zambrów). The plants do not represent ballast because they are not cut.

Treatment systems with a surface flow and pleustonic plants are constructed under a “Lemna Corporation” licence. The commonest type in Poland is the “Lemna System”, and indeed the country has more of these than anywhere else. They are employed in the treatment of household sewage. A pond with plants represents tertiary wastewater treatment and are preceded by a septic tank and an aerated pond. The plants introduced from the natural sites most often generate a two-species system comprising *Lemna minor* and *Spirodela polyrrhiza*. The plants are removed from time to time, and their mass represents an unfavourable ballast that is difficult to utilise (Ozimek, 1996).

As constructed wetlands represent a relatively new technology, the determination of their suitability both worldwide and in Poland has most often taken place on the basis of a study of their effectiveness in removing pollutants. The world literature contains hundreds of publications, of which 95% include information on efficiency. This has made it possible to draw conclusions and make generalisations on the basis of these data, which mainly concern constructed wetlands with emergent macrophytes. Against the background of data for temperate-zone countries, it has been possible to show the data for Poland. The Polish experiences confirm the high efficiency with which total suspended matter is removed, along with organic matter liable to biochemical and chemical breakdown. In turn, the effectiveness with which compounds of nitrogen and phosphorus are shown to be removed is low. Further research should focus on an analysis of seasonal changes against the background of those that occur over the long term. Determinations will need to be made of the degree to which the efficiency of wastewater treatment changes as the wetland is used, and of how the substratum and above all its hydraulics change. Is it possible to produce an empirical designation as to when the removal of plants has to begin on account of an excessive amount of organic matter having accumulated?

The Lemna system is not popular in Europe outside Poland, but here it accounts for 42% of all constructed wetlands. As Vymazal (2001) shows, there is a lack of information on the treatment efficiency of operational versions of the duckweed-based system. Data in the literature are lacking, with those available being from analyses carried out by the designers.

Sadly, such work needs to be approached with considerable caution.

The main problems associated with the use of the Lemna systems are the variability of plant biomass and cover from year to year, and the lack of any information on what influences these differing amounts of cover. The work that has been done so far suggests that ponds with duckweed do not raise the effectiveness of wastewater treatment, and do generate unnecessary ballast.

Work carried out by Ozimek and Milewski (unpubl.) makes it clear that hydrophyte–based treatment plants are a source of great interest and are highly estimated. Economics is a leading factor in this. There is a marked need for specialists able to clearly demonstrate the advantages and limitations of the use of such solution to become involved in their popularisation.

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A study on pollution of the Kola River and its outflow into the Arctic Sea: source identification, protocol for monitoring and low cost purification measures

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Abstract

The Kola River in Northern Russia is polluted by sources of complex origin, which brings about a negative effect on water quality and biota in the river ecosystem, as well as in the nearby marine ecosystem of the Arctic Sea. In the framework of a 4-year research project, which was started on December 1 2000, and is partly financed by the European Union (INCO-Copernicus Programme), policies to mitigate pollution and a protocol for more adequate water pollution control are worked out. In addition, guidelines for adequate, cost-effective monitoring will be formulated according to EU water directives. Preliminary results of chemical analyses and bio-indicators showed that the Kola River is less polluted than previously has been assumed. As wetlands appear to be very suitable for wastewater purification in cold climates, possibilities for introducing this low cost technology for the first time in an Arctic climate are studied. A pilot artificial wetland was constructed along the Kola River in the village of Shonguy during the period November 2001 – June 2002. It includes an area with vegetation

and a sedimentation pond. The wetland was put in operation in June 2002, but until December 2002 functioning has been irregular, and apparently the biological parameters had not yet stabilised. Preliminary results showed that it was most effective for suspended substances (more than 70% purification), oil products (more than 80%) and synthetic surface-active substances (more than 70%). The decrease in BOD varied from 50% to 80% and nitrification varied from 0% to 40%.

Introduction

The Kola River in Northern Russia flows via the Kola Bay into the Barents Sea, which is part of the Arctic Sea. The lower part of the river is loaded by pollutants of complex origin, emanating from large pig, poultry, fur and cattle farms, as well as from different industries. In the upper part of the river the steelwork in Olenegorsk and the Monchegorsk nickel smelting-works partly discharge into the Kola river via the large lake Kolozero. In this context it should be noted that the Kola river is a main drinking water source for the 450 000 citizens of the city of Murmansk and that it is an important spawning area for valuable fish species.

The current project is carried out by an international consortium, which has a wide-scale experience in water analysis techniques, formulation of guidelines for cost-effective and adequate river status assessment and monitoring, as well as low-cost, energy saving, technology for water purification.

The objectives of the project are to map the pollution sources of the Kola River in detail and to formulate guidelines for an adequate, cost-effective monitoring, taking into consideration EU water directives. In this context it should also be noted that an interactive computer-based decision support system will be developed which will make it easier for local authorities to integrate environmental considerations into land planning and management practices.

Another objective is to test artificial wetland technology for purifying wastewater, which has been shown to be efficient in cold climates, for the first time under Arctic conditions.

Description of the work

Pollution sources of the Kola River, including ordinary point sources as well as diffuse loading from soils and areas of accumulated wastes, are mapped on the basis of previous studies by the Kola River Environment Programme (KREP) (Bergström and Mikaelsson, 1998; Bergström *et al.*, 1999), and also by additional work. Subsequently, the harmfulness of the different pollution sources will be estimated. For each pollutant source it will be checked whether low cost interventions are feasible to decrease its impact on the environment. In line with the RiverLife project, a three-year EU/Life project which has been carried out in Northern Finland (starting date September 1998), a decision support system including basic elements for effective water pollution control, and guidelines for adequate, cost-effective monitoring will be formulated. This will imply that EU water directives will be implemented in Russia. Moreover, the newest international knowledge on river status assessment and the impacts of harmful substances on aquatic ecosystems will be utilised. Existing data (hydrological, hydrographical, phytoplankton, zoobenthos) will be gathered into computer available databases. Relevant and predictive variables in each dataset will be searched for to widen the general international knowledge on river systems and needs for water pollution control in Arctic rivers.

Two relatively unpolluted rivers in the same region, River Kalix in northern Sweden and River Näätämöjoki in northern Finland are for reasons of comparison also included in the studies

For low cost of wastewater purification an artificial wetland is constructed, based on experiences obtained in the framework of a four-year INCO/Copernicus project in the Ukraine and Estonia (starting date February 1997) (Mander *et al.*, 1999; Stolberg *et al.*, 1999), and also on comparable studies in Canada, Finland and Sweden (Mander and Jenssen, 2002). The purifying capacities of this pilot system, which is for the first time studied in an Arctic climate, will be monitored by analysing the in- and outflows.

Results and discussion

Chemical analyses showed that, in general, pollution of the Kola river is less serious as previously was presumed. When compared to the Kalix river in Northern Sweden, the concentrations of major elements were very similar, with the exception of Na, Cl, Cu, Zn and Ni, which were two to three times higher in the Kola river. The major sources of pollution proved to be the iron ore concentration plant in Olenegorsk in the upper part of the river and the Varlamov, Medvezhiy and the Zemlyanoy creeks in the lower part of the river, which drain off wastewater from various kinds of industries. The pollutants from the plant in Olenegorsk enter the river via Lake Kolozero and the concentrations of Ca, K, Mg, S, Mn, As, Cd, Al, Co, Mo and Sr in this lake are relatively high. Particularly in the suspended phase concentrations of various elements were significantly higher than in the middle part of the river. For example As was 152 times higher, Mg 21 times, Al 16 times, Co and Cu 9 times, Mo and Ba 7 times, and Pb and Cd 6 times higher. The increase in pollutants is especially high in May when the snow is melting. Heavy metal concentrations in the outflow of Lake Kolozero are still two to four times higher than in the middle part of the river.

Pollutants from two large poultry farms near the lower part of the river enter to a certain extent the Medvezhiy and Zemlyanoy creeks. These creeks drain off into in the Kola River, but in general, if no accidents occur, this does not significantly influence the water quality.

The saprophytic microflora in the Kola River varied from 300 to 2800 counts per ml in Lake Kolozero, 2500–3400 counts per ml in the Varlamov creek, and 5100–5300 counts per ml in the Medvezhiy creek. The concentration of saprophytic bacteria in the unpolluted River Näätamöjoki in Northern Finland, which is studied for reasons of comparison, did not exceed 200 counts per ml.

The biological parameters expressed in the Pollution Sensitivity Index (IPS), the Trophic Diatom Index (TDI) and the Generic Diatom Index, showed generally a high or good water quality in the Kola River, with the highest scores in the middle part of the river. In comparison to the River Näätämöjoki, the hydromorphological state of the Kola River was somewhat lower.

In the Kola River basin the proportion of eutraphentic diatom species that indicate nutrient pollution was the highest in the Varlamov and Medvezhiy creeks. As far as saprobity classes are concerned, which indicate the level of organic pollution, the majority of the diatom species in most of the study sites, both in the Kola River and in the River Näätämöjoki, could be classified as oligo- or mesosaprobes. The largest amount of polysaprobes, that indicate an increased level of organic pollution, was observed in the creeks in the lower part of the Kola River.

The number of aquatic macrophytes in the Kola River varied from 2 to 11 in the channel and 37 to 68 at the river margin. In River Näätämöjoki the number varied from 2 to 5 in the channel and 47 to 64 at the river margin. With regard to the river habitat survey the average Habitat Quality Assessment scores (HQA) varied for the Kola River from 36 to 61, and in River Näätämöjoki the scores varied between 40 and 54. The Habitat Modification Scores (HMS) varied in the Kola River between 0 and 7, whereas in River Näätämöjoki, where variation between the sites was relatively small, values of 0 to 2 were measured. Along River Näätämöjoki there is hardly any human disturbance, except for a few cabins on the shore. Construction of the artificial wetland in the village of Shonguy took place during the period November 2001 – June 2002. It includes an area with natural vegetation (mainly consisting of the willow tree *Salix schwerini*) and a sedimentation pond. In the summer season additional willow trees as well as helophytes, such as sedges, were planted in the vegetation unit. The total of the wetland is 2700 m^2 and the slope of the vegetation unit is 5 m. In June 2002 the artificial wetland was put in active operation.

Wastewater from the village of Shonguy is, before entering the artificial wetland, first purified in a nearby conventional waste water plant. The purification process in this plant is not very effective and not up to standard. Before construction of the artificial wetland this partly purified wastewater was directly released into the Kola River. Currently, this wastewater, after treatment in the plant, is via a sedimentation tank led through the vegetation unit and subsequently through the sedimentation unit, which consists of gravel, before it is discharged into the Kola River. During the first months functioning of the newly constructed artificial wetland was irregular. However, in December 2002 it appeared that the biological processes had more or less stabilised. Even at an air temperature of minus 32°C the system worked adequately as far as breakdown of organic compounds was concerned.

The system most effectively purified suspended substances (more than 70% purification), oil products (more than 80%) and synthetic surface-active substances (more than 70%). The decrease in BOD varied from 50% to 80% and nitrification from 0% to 40%. However, the results are very preliminary and it is too early to draw any final conclusions.

Acknowledgements

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Characteristics of constructed wetlands and potential significance for agricultural water protection in practice

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Introduction

Constructed wetlands (CWs) are known to have an ability to retain nutrients and other harmful substances from the waters that flow through them (Mitsch and Gosselink, 2000). Thus, they have been suggested for control measures against non-point source (diffuse) pollution (*e.g.* Leonardson, 1994). Usually, the main objective of these systems is to decrease the loading of the algal-growth fueling nutrients; nitrogen (N) and phosphorus (P), and hence eutrophication of surface waters. The most important nutrient retaining, or removing, processes in CWs are (i) sedimentation of nutrient containing particles onto the CW bottom, (ii) filtration of nutrient containing particles into the water-permeable soil and/or vegetation, (iii) biological nutrient uptake, mostly by macrophytes, (iv) adsorption of dissolved P into the CW soil, and (v) denitrification of N. In order to ensure that these processes actually function – and bring about a desired minimum efficiency for CWs – tools and guidelines are needed for practical CW design.

The objective of this study was to investigate the factors behind the nutrient removal effectiveness of CWs in agricultural catchments, particularly in terms of the CW design and dimensioning parameters. Moreover, the potential significance of CWs in relation to other measures in water protection management at catchment-level was briefly considered.

Material and methods

In this study, materials and research results were collected from various sources. In Finland, retention performance of CWs has been studied in 4 agricultural study sites (Häikiö, 1998; Koskiaho *et al.*, 2003). Flow behaviour in 2 of these CWs has been simulated with 2-dimensional hydrodynamic models (Koskiaho, 2003). A literature review was made in order to widen the perspective in terms of different types of loading (agriculture, point sources, and peat mining). As for agricultural runoff, comparisons of commensurate CWs gave implications of the minimum requirements of CW properties.

Results and discussion

Load reduction effectiveness of constructed wetlands

CW effectiveness is controlled by distribution and quality of input loading, water temperatures as well as the form, size, and soil and vegetation properties of the CW. These controlling factors – save loading and temperature – can be affected by choices of design and construction.

When compared with point source loading, diffuse loading:

- usually consists of lower concentrations and higher amounts of water,
- is governed by stochastic weather events (rainstorms, snow melting *etc.*) which makes it much more difficult to control, and
- takes place in runoff waters of which temperatures are usually lower, particularly in cold, boreal conditions.

Due to all these reasons, the effectiveness of CWs treating diffuse loading is typically weaker than that of the CWs treating point source loading (Table 1).

Table 1. Examples of nutrient removal-% in CWs treating point source and diffuse loading during at least 1-year study periods.

Loading source type	Total N removal (%)	Total P removal (%)	Reference
Point	>57	>97	Maehlum and Stålnacke (1999)
	67	80	Mander and Mauring (1997)
	66	69	Mander and Mauring (1997)
	60	99	Sundblad and Wittgren (1991)
Diffuse	3–15	21–44	Braskerud (2001)[1)]
	–12–36	–6–62	Koskiaho *et al.* (2003) [2)]
	68 [3)]	43	Whigham *et al.* (1999) [4)]
	3	6	Häikiö (1998)

[1)] Range of 4 CWs , [2)] Range of 3 CWs, [3)] NO_3-N, [4)] Average of 7 CWs

Wetland to catchment ratio

Probably the most important single factor with regard to CW effectiveness is the size of a CW, i.e. its area in relation to the area of its catchment (W/C ratio). The relationship between CW effectiveness and W/C ratio has been noted in several recent studies (*e.g.* Kovacic *et al.*, 2000; Koskiaho *et al.*, 2003; Woltemade, 2000). In Figure 1 the dependence is demonstrated by the results from 12 CWs in Finland and USA.

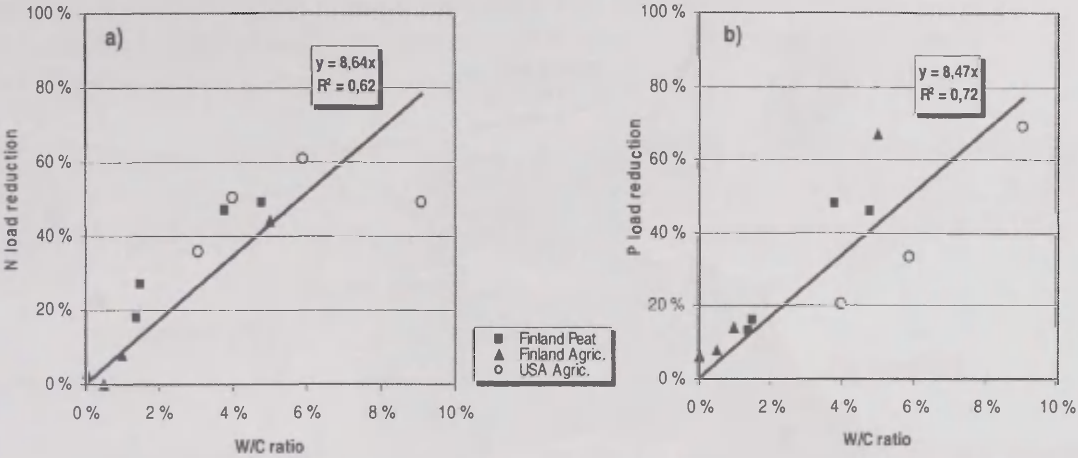

Figure 1. Dependence of nitrogen (a) and phosphorus (b) load reductions on CW to catchment ratio.
Sources : Ihme (1994)
▲ Häikiö (1998), Koskiaho *et al.* (2003)
○ Whigham *et al.* (1999), Kovacic *et al.*(2000)

In spite of the uncertainty that is inevitably involved with such rankings the message is clear: if substantial (say, more than 20%) load reductions are desired, the W/C ratio should be more than 2%. In case of runoff water treatment, dimensioning should be based on input water volumes of the highest annual runoff events because the high water periods account for great deal of annual loading.

Hydraulic efficiency

CW effectiveness does not only depend on W/C ratio but also on hydraulic efficiency, i.e. on how efficiently the through-flowing water utilizes the CW volume (Koskiaho, 2003). Hydraulic efficiency is a very important property of a CW because short-circuiting of flow leaves large CW areas idle in terms of water purifying processes and shortens the WRT of main flow, i.e. the actual residence time. Therefore, a CW planner should not only emphasize the W/C ratio but also be concerned of the matters behind hydraulic efficiency. These include CW form and the elements directing the flow, in a word, CW layout. An example of this was presented in Koskiaho (2003) where flow velocities and WRTs in different CW layouts and in different inflow volumes were investigated with 2-dimensional hydrodynamic models. In the other case of the study baffles (spits of land, see

Figure 2) improved the hydraulic efficiency number (λ) from 0.24 to 0.65 (For λ, theoretical min = 0 and max = 1). Also a deep pond-part (Figure 2) in the beginning or a levelling terrace in the middle of a CW are good design elements by decelerating flow and distributing it evenly to whole width of the CW.

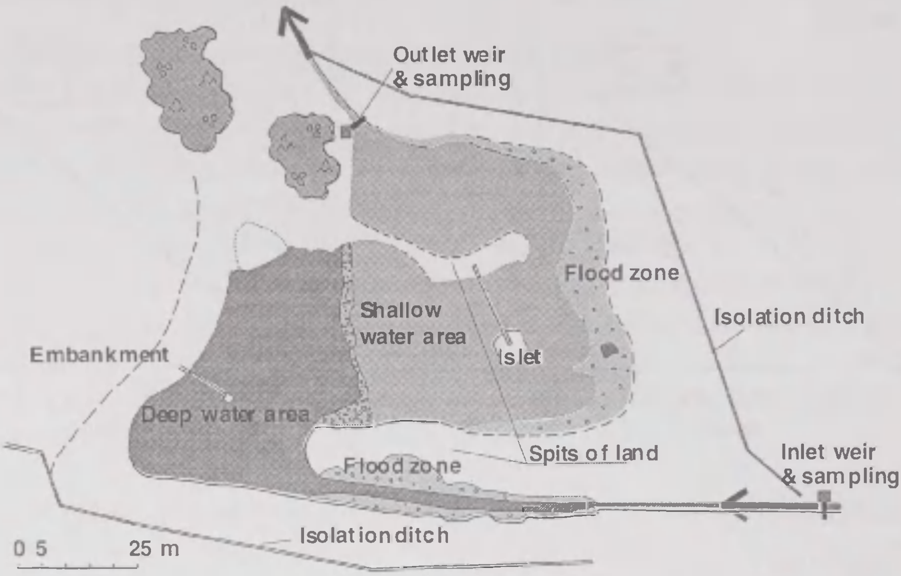

Figure 2. Schematic image of the Hovi wetland.

Other characteristics

Agricultural CWs are often constructed on a former field. In order to avoid the risk of P release, the P-rich top soil should be first removed (Liikanen *et al.*, 2003). The development of vegetation in CWs can be enhanced – or guided towards the desired direction – by seeding and transplanting. On the other hand, solely natural development can also lead to a proper outcome. (Koskiaho *et al.*, 2003).

Potential significance of constructed wetlands in water protection

Achievement of enduring improvement in water quality requires reductions of loading in many polluting sources. In terms of diffuse loading, comprehensive catchment management, *i.e.* reasonable locating of CWs and other water protection measures is the fundamental base, and proper design of single CWs is just a part of this process. In agricultural catchments, the water protection measures may include careful control of fertilizing, reduced tillage cultivation, controlled drainage, and – as last lines of defense – riparian buffer zones and CWs. The choice between the measures should be made according to cost efficiency, *i.e.*

how much load reduction will be achieved by a certain investment (Äijö and Tattari, 2000). As for CWs, the questions regarding locating strategy include:

- How large portion of the catchment at issue should be covered by CWs?
- How many CWs are needed for that?
- How should they be located with regard to the production areas?
- Do the suitable places and their owners' desrire to co-operate coincide?
- How much does the required number of CWs cost?

Comprehensive catchment management calls for decision making tools, like the VIHMA-model being developed at the Finnish Environment Institute. In the VIHMA-model CWs are regarded as one of the potential water protection measures.

Conclusions

Agricultural CWs should be established in places where they are naturally suited for and indeed necessary. As for CW dimensioning, requirements of a desired minimum effectiveness should be fulfilled. The most important single characterristic affecting CW effectiveness is the actual WRT. Because CWs are typically quite shallow, the area rather than the depth determines the CW volume. Hence, rough predictions of CW performance can be made on the base of W/C ratio. To ensure that the minimum effectiveness determined according to the W/C ratio will be achieved, a CW should have a good hydraulic efficiency. Short-circuiting of flow can be prevented by creating deep parts, baffles, and shallow flood-zones. A bypass channel should be constructed for the flow peaks exceeding a pre-defined critical flow. CW vegetation can be established by both seeding and transplanting, preferably with local, natural species.

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Model study of secondary treatment reed beds

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Introduction

Over the last years, several dynamic, compartmental models of root zone constructed wetlands have been presented in literature (Wynn and Liehr, 2001; Senzia *et al.*, 2002). These models explicitly take into account the different processes occurring in constructed wetlands. Simulation results of these models appear quite promising. They however have one major drawback: a large number of parameters need to be estimated. Since little has been published about the values of most of these parameters, calibration must be based on input-output data alone and cannot rely on prior knowledge. Dynamic constructed wetland models are therefore still quite useless as *a priori* design tools. The research presented here aims at extending and calibrating an existing model (Wynn and Liehr, 2001) and at checking whether or not the model output would be accurate enough to use this model as an appropriate design tool.

Material and methods

In August 2002, a detailed data set was collected at a two-stage reed bed of Severn Trent Water Ltd. at Saxby (Leicestershire, UK), designed for 47 PE and in service since 1998. The system consists of two horizontal subsurface-flow beds connected in series, preceded by a conventional septic tank for primary treatment. Each bed has a surface area of 117 m² and an average depth of 0.6 m.

During 2 weeks, the influent and effluent of both reed beds were monitored by means of automated samplers. The samplers were set to collect time-proportional composite samples over 8 hours of the influent and both effluents. Composite samples were preferred because of facilitating the application of mass balances. Samples were analysed at the Severn Trent Laboratories for the common variables COD, BOD, TOC, N, P etc. Simultaneously, meteorological data were collected because of their importance in the water balance. Effluent flow rates of the second bed were registered on a 15 minute base.

Survey results

The average air temperature during the 2-week survey varied between 12 and 30 °C. Some severe rain events occurred which increased the influent flow rate from a base flow of less than 0.1 L s^{-1} to a peak flow of about 15 L s^{-1}. This caused temporary flooding of the beds.

The treatment works nevertheless consistently produced a high quality effluent with total BOD and SS concentrations lower than 10 mg L^{-1} and 30 mg L^{-1} respectively. As an example, the SS concentrations are shown in Figure 1.

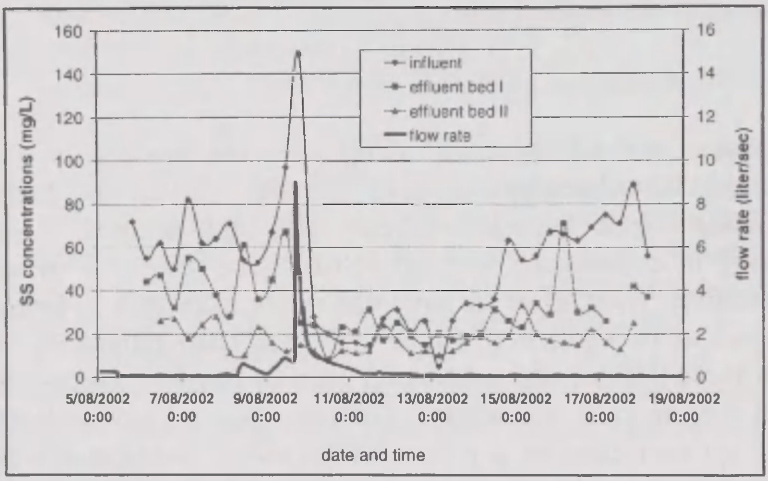

Figure 1. Effluent flow rates and SS concentrations of the secondary treatment reed beds at Saxby during the period 5–19 August 2002.

The average BOD_{total} (BOD_t), $BOD_{filtered}$ (BOD_f), ammonium (NH_4^+-N), Kjeldahl nitrogen (kjN), total oxidized nitrogen TON (= NO_3^--N + NO_2^--N), total nitrogen (TN) and orthophosphate (OP) removal efficiencies (Table 1) can generally be called excellent with reference to reported literature values. SS removal on the other hand seems to be only average.

Table 1. Average concentration-based removal efficiencies (%) at Saxby during the period 5–19 August 2002. Negative values indicate a net production.

	BOD_t	BOD_f	SS	NH_4^+-N	KjN	TON	OP	TN
Bed I	93.8	94.0	38.4	61.2	57.0	–188.6	76.3	47.4
Bed II	53.7	43.5	12.5	32.5	20.7	56.6	–70.1	31.3
Total	97.1	96.6	46.1	73.8	65.9	–25.3	59.6	63.8

These figures also clearly indicate that the beds have enough oxygenation capacity for aerobic degradation and nitrification but also provide enough anoxic regions where denitrification takes place.

Taking into account that the bed operation started more than 4 years ago and that the matrix material consists of gravel rather than sand, phosphorus removal is quite satisfactory. There is no sign of saturation of the sorption sites yet. The net production of phosphorus in the second bed suggests that there is some decay of organic material.

Model study results

For the model study of the Saxby reed beds, the model of Wynn and Liehr (2001) was used as a starting point. This model describes carbon and nitrogen transformations in a subsurface-flow wetland. P transformations are not considered since these are mainly of physical-chemical nature and the main focus of the study was on microbial processes. One important assumption made in the original model is that the SS removal is 100%, which means no particulate substances are leaving the bed. This assumption was based on the fact that effluent SS levels are normally very low. For the Saxby case however, effluent SS concentrations are not really negligible. On the other hand, the difference between total and filtered BOD and TOC in the effluent was mostly very small, which means that the 100% removal assumption did not jeopardise the model predictions for BOD. For a comprehensive explanation of the model, the reader is referred to the paper of Wynn and Liehr (2001).

Originally, the model was written in STELLA© code (High Performance Systems Inc). The simulations for this study were carried out in WEST© (Hemmis NV). Since WEST© works with the MSL language, the model had to be re-implemented. During this re-implementation, some minor model flaws were rectified (De Wilde, 2001; De Moor, 2002).

The model requires 9 inputs, 6 regarding the influent (flow rate, BOD, Organic-N, NH_4-N, NO_3-N and dissolved oxygen) and 3 regarding external influences (day length, air temperature and precipitation). There are 6 standard model outputs consisting of the same variables as the influent input. One can however also keep track of certain model variables like plant growth, peat accumulation, evapotranspiration *etc.* if that is of interest.

To simulate intermediate flow behaviour, two completely mixed tanks in series were used to represent one reed bed.

Although Wynn and Liehr (2001) obtained good results with their long-term, low-frequency dataset, the model results for the Saxby case were not satisfying at all (Figure 2a). Two possible causes for this discrepancy are:

1. *Time steps*: Wynn and Liehr used a dataset that consisted of biweekly measurements of C and N components (grab samples). To obtain daily inputs for the model, interpolations between the datapoints were made. This is totally unlike the Saxby dataset (8-hour composite samples) and will influence the model output.
2. *Model uncertainty*: it is quite possible that certain processes occuring in wetlands are not included in the model. Due to external conditions, these processes might have been of minor importance in the Wynn and Liehr case, but of bigger importance in the Saxby case.

Several adaptations of the Wynn and Liehr model resulted in better model predictions for the Saxby case. Our adapted model was among others extended with a Freundlich sorption equation for ammonium, as described in McBride and Tanner (2000). The microbial kinetics was also restructured as pure Monod kinetics with switching functions instead of applying a minimum conversion. These model extensions together with some parameter adjustments caused the model output to reasonably reflect the real effluent concentrations of the first reed bed (Figure 2b).

For validation purposes, this extended model was run again with the dataset of the second reed bed. N removal was not adequately predicted which does not immediately imply that the model is not correct. Indeed, some parameters and initial conditions can be different for bed I and bed II. Because plants and micro-organisms in the second reed bed are subjected to smaller loads, several authors (*a.o.* Kadlec and Knight, 1996) have proven that growth rates are lower. Hence, new simulations with lower growth rates were performed and these gave acceptable results (Rousseau, 2002).

The only characteristic the model underestimates, was the buffering capacity of the reed beds. When high influent concentrations occured, the model occasionally predicted peak effluent concentrations that in reality were not observed. This might suggest that some more mixing in the reed beds occured than was modelled, or that other sorption or temporary storage mechanisms are active.

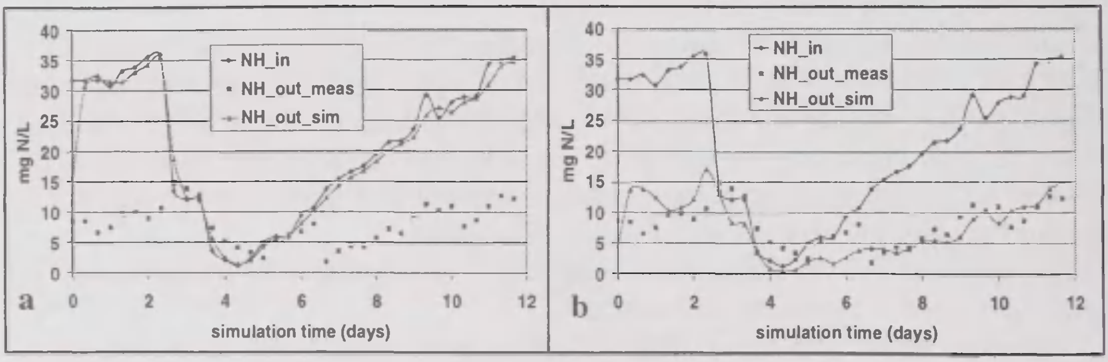

Figure 2. (a) Influent, measured effluent and simulated effluent NH_4-N concentrations in the first reed bed at Saxby with the original model and original parameter settings. (b) Influent, measured effluent and simulated effluent NH_4-N concentrations in the first reed bed at Saxby with the adapted model and after calibration.

Conclusions

- The extensions of the Wynn and Liehr model and especially the NH_4^+-sorption submodel significantly enhanced the simulation results.
- Further calibration and validation with other datasets are needed to improve the model predictions and to reduce parameter uncertainty.
- The different parameter settings for the first and the second reed bed suggest that at this stage, it might be unwise to use this model as a design tool for constructed wetlands.

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Alternative models for nutrient reduction in surface treatment wetlands based on limited data access

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Surface treatment wetlands (STWs) are gaining ground to complement conventional sewage treatment systems. Being considered as having relatively long life spans, STW's long-term performance is in need of thorough analysis. Although the performance will reach, a steady state when the adaptation phase has settled, random (usually unexpected disturbances), seasonal and climate effects will most likely obstruct such analyses.

Currently used models for prediction of STW performance are either generally complicated, with extensive data requirements, or quite simple. The combination of seasonal and random effects with extensive data requirements contradicts the relatively poor databases collected in standard programmes for STW surveys. Advanced models are therefore mainly applicable in experimental STWs, whereas simple models often are used in the evaluation of standard STWs. Data scarcity has hence been an important constraint in the development of models for reliable evaluation of standard STW performances. By modifying simple models, based on hydrological and biochemical processes, the evaluations of trends, reduction rates, efficiency of parameters and variables are to some extent possible. These simple models have been expanded into dynamic models that enable further evaluations. The dynamic model is a useful tool to help managing STWs. It can also be used to estimate the effects of single events and long-term changes.

County authorities regulate Swedish programs of STW surveys. Since the programs differ with respect to sampling frequency, including variables, periodicity of missing data, etc., the compatibility of data between STWs is generally poor. In this study, data from surveys of four Swedish STWs have been used in an effort to develop "semi-advanced" models for the evaluation of STW performance. There are significant differences in loading, climate and design between the STWs.

Due to the sparse data and complexity of the STWs, monthly averages are used to evaluate the performance. By averaging over a period longer than the hydraulic retention time, rapid temporal effects are suppressed. Different combinations of models and variables have been tested to identify the most suitable predictors describing the STW performances. STW water temperature was found to be a good predictor, although the variable often causes systematic errors since intra-STW processes are neglected. An equally good prediction is accomplished by using a seasonal variable instead of temperature. This variable describes the combined effect of biota and temperature variation but loses some of the rapid temporal effects. When the model contains a variable describing hydraulic load, the resulting model allows evaluation of long-term trends in STW performance. The long-term trend can be divided into a mean trend and an annual variation trend. This is particularly applicable when newly established STWs are addressed and for evaluation of seasonal timing of effluent composition.

The parameters of STW performance estimated with the simple models discussed above are possible to use as input to dynamic models. These quite highly resolved models are validated with the help of the original data. The resulting models are superior when effects of single variables are analysed. In cases where influent and effluent flows are simultaneously surveyed, it is possible to estimate the effects of varying sewage dilution and concentration.

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Salinity-induced density stratification in low Reynolds number free surface flows

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Introduction

Salt tracer experiments are a fairly widespread method of obtaining information on flow and transport processes in constructed wetland ponds. The removal efficiency and effectiveness of these wetland ponds is, for instance, strongly rated to the distribution of flow paths, which – in turn – is reflected by the breakthrough curves of an ideal tracer injected into the flow. Salt, however, is not an ideal tracer, as solutions of elevated salinity are distinctly heavier than water. In most hydrologically motivated salt injections this particular deviation from ideality does not usually compromise the results obtained, as turbulent flows in river and brooks effectively counteract the formation of stable layers within the flow. In contrast, flows in constructed wetland ponds are characterized by low to very low Reynolds numbers, which are close to and, at times, even within the laminar flow regime. It is under these conditions that the release of a salt tracer may entail strong density effects, distorting breakthrough curves and threatening their information content. In these circumstances the availability of a tool that permits an intended salt tracer experiment to be judged at the planning stage already, is clearly desirable. The presentation of such a criterion forms the subject of this contribution.

Derivation of criterion

The stability of a shear flow in the presence of density differences can be expressed by a Richardson number (Miles, 1962; Rutherford, 1994), the 'bulk' version of which reads:

$$\mathrm{Ri}_b = -\frac{g}{\rho} \cdot \frac{\Delta\rho / \Delta z}{(\Delta u / \Delta z)^2} = -\frac{g \cdot h}{\rho} \cdot \frac{\Delta\rho}{(\Delta u)^2} \tag{1}$$

with ρ as the depth-averaged density, g gravitational acceleration, z the vertical coordinate (positive upwards) and u the velocity component in streamwise direction. In weakly stratified flows, surface or bottom water densities can be used in place of the depth-averaged quantity, if convenient. h denotes water depth and Δ a difference between surface and bottom values.

The Richardson-number limit between stable shear flow (with density stratification) and unstable, well-mixed conditions can be given as some 0.25 in agreement with literature (Miles, 1962; Rutherford, 1994; Boehrer *et al.*, 2000). Consequently, a criterion of negligible density effects reads: $\mathrm{Ri}_b \leq 0.25$. This is straightforward, but impractical from the viewpoint of experimental design, as the bulk Richardson-number is unknown a priori. To overcome this difficulty, $\Delta\rho/\rho$ and Δu must be estimated. Injection of salt tracer mass M_0 in the form of a pulse of duration Δt into a flow of rate Q will be associated with the initial concentration

$$C_0 = \frac{M_0}{Q \cdot \Delta t} \tag{2}$$

if full initial mixing of the tracer with the wetland inflow can be assumed. After the injection has stopped, the inflow again consists of water without added salt, so that the concentration C above background drops to zero again, *i.e.* C = 0. During the salt tracer release, a near-constant vertical concentration profile close to the inflow cross-section of the wetland pond develops due to the assumed full vertical mixing, with a concentration close to the value given by Equation (2). After the end of the pulse, the current will move fresh inflow water (without added tracer now, *i.e.* C=0 again) over the salty water lower down in the profile, as the streamwise flow velocities are higher in the upper part of the flow. Thus, in this approximate model of the density stratification, there will be a concentration of C=0 at the surface and $C=C_0$ (from Equation 2) at the bottom. This concentration difference is associated with a corresponding density difference, which can be derived from a density formula given by Crowley (1968). In terms of relative density differences, one obtains:

$$\frac{\Delta\rho}{\rho} \approx -10^{-6} \cdot (0.802 - 0.002 \cdot T) \cdot C_0 \tag{3}$$

with C_0 as the salt concentration in mg l^{-1} and T the temperature in °C. While Crowley (1968) presented his formula in an oceanographic context (so that 'salinity' must be taken to mean primarily NaCl), the relationship also works well with other salts, like KBr.

The only quantity in Equation (1) left to be expressed is now Δu. Both from literature (Hammer and Kadlec, 1986) and from a survey of characteristic data of the constructed wetland ponds studied in the PRIMROSE project, an upper limit of Reynolds number $\mathrm{Re} = \bar{u} \cdot h/\nu \approx 1000$ (with $\bar{u}$ as average velocity and ν the kinematic viscosity of water) appears correct in the large majority of cases, and the assumption of a laminar or near-laminar transitional flow regime is justified here. For the laminar regime, the velocity distribution is parabolic and can be given as:

$$u(z)=\frac{24\cdot g\cdot S}{K\cdot \nu}\cdot z\cdot\left(h-\frac{z}{2}\right) \tag{4}$$

with S denoting friction slope and K a roughness parameter. The velocity difference is $\Delta u = u(z=h)-u(z=0)=u(z=h)$. The resulting relationship for Δu can, in turn, be expressed by means of discharge per unit width q, a known or at least easily measurable quantity, and one obtains $(\Delta u)^2 = 9\cdot q^2/(4\cdot h^2)$. Substitution of this relationship together with Equation (3) into Equation (1) after rearrangement finally yields the desired criterion for the laminar flow case:

$$C_{max}\,[mg/l]=\frac{10^6}{0.802-0.002\cdot T[°C]}\cdot\frac{9}{4}\cdot\frac{q^2\cdot Ri_{b,max}}{g\cdot h^3} \tag{5}$$

with $Ri_{b,max} = 0.25$ and C_{max} as the maximum of C_0 without pronounced density effects. A plot of C_{max} versus Re-number for various water depths is shown in the following Figure 1:

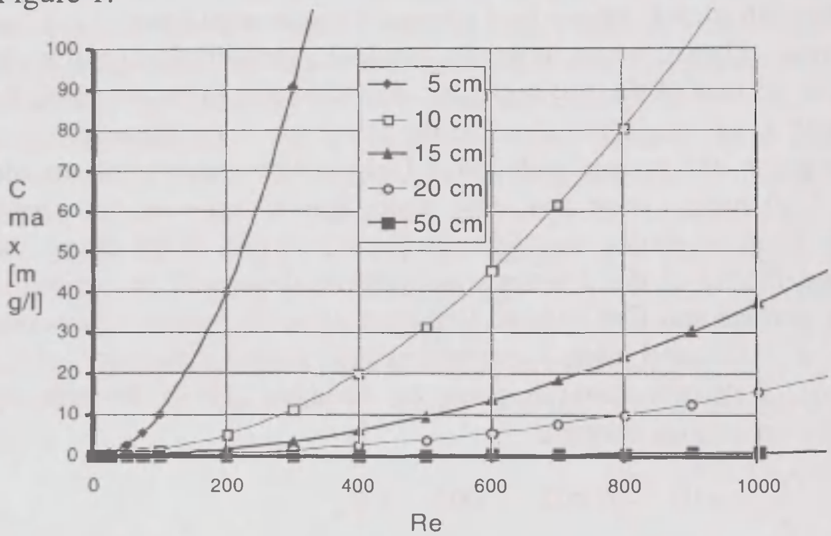

Figure 1. C_{max} after initial mixing (not causing strong density effects) versus Re-number.

Laboratory experiments

Salt-induced density stratification was studied experimentally in a 1.5 m wide and approximately 40 m long, straight laboratory flume of rectangular cross-section. Experiments were conducted at Re-numbers close to 1000, *i.e.* the upper limit of the range mostly encountered in real world constructed wetland ponds, as indicated above. Water depth was fixed at h = 20 cm and flow rate Q at 1.6 l s^{-1}. The study was conducted with NaCl and KBr as salt tracers. Velocities were measured by a 3D Acoustic Doppler Velocimeter (ADV), and chloride / bromide concentrations were obtained from recorded electric conductivities (with probes mounted at 1.5 cm, 5 cm, 10 cm and 15 cm above bottom, at x = 34.6 m downstream of the injection).

Measurement of the vertical velocity profile revealed the velocity distribution to be somewhat flatter than its theoretical fully laminar counterpart. This is attributed to the presence of a transitional regime, which agrees with the Reynolds number being right in the interval between 500 (upper limit of laminar range) and 2000 (lower limit of turbulent regime). From the viewpoint of the criterion, equation (5), a flatter velocity distribution results in lower values of $(\partial u/\partial z)^2$ as compared to the theoretical ones, and the criterion (5) must, therefore, be expected to slightly overestimate C_{max} in such cases.

8 experimental runs were conducted at Re ≈ 1000, with 6 of them performed as short pulse injections (injection interval around 10 min) and two more in a steady state mode, continued at a constant injection rate until equilibrium conditions were reached. The steady state runs expectedly showed much less tendency towards stratification. For the pulse injections, evaluation of criterion (5) indicated a limiting concentration C_{max} after initial mixing of some 15 mg l^{-1}. The salt mass released varied between 13.7 g and 5.56 kg, resp., with associated values of initial concentration between 14.3 mg l^{-1} and 6.82 g l^{-1}. Breakthrough curves were measured at the depths given above, and the differences in respective areas under the curves (the zeroth moments) were used as a measure of the strength of the density stratification (in a well-mixed situation, these zeroth moments should be equal). In the majority of the cases, strong density stratification developed, and its presence was predicted correctly by the criterion Equation (5). A borderline case arose out of the injection of 13.7 g NaCl (corresponding C_0 = 14.3 mg l^{-1}), just below C_{max} = 15 mg l^{-1}. Thus, criterion (5) predicted only a negligible density effect, whereas comparison of the zeroth moments at different depths indicated a weak but still discernible stratification (with normalized zeroth moments deviating between +27% and –21% from their depth average). As explained before, this is attributed to the flatter velocity distribution at Re ≈ 1000 as compared to the fully laminar profile.

Conclusion

A criterion has been presented that permits experimenters to judge the importance of density effects to be expected in the course of salt tracer experiments in the low Reynolds number flows typical of constructed wetland ponds. Results of this study show that stringent upper limits to the injected mass of salt must be observed, if strong density effects are to be avoided.

Acknowledgement

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Wastewater purification efficiency in constructed wetlands with surface and subsurface flows

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Constructed wetlands for wastewater treatment are commonly used for the purification of domestic wastewater in rural areas, because of the lower costs of capital expenditure and exploitation. Such treatment for domestic wastewater was constructed in 1975 in Kopaszewo (Agroecological Landscape Park, in West Poland-Lowland). The studied set consisted of a screen bar, Imhoff tank, trickling filter and secondary clarifier. The Imhoff tank took the form of a concrete cylinder of 4.0 m diameter and volumes $V_1 = 12.0\ m^3$ (trough) and $V_2 = 30.0\ m^3$ (open digester) situated below ground. The trickling filter was a cylindrical reservoir made of reinforced concrete, with a diameter of 6 m and height of 2.5 m, and was situated on the ground and filled with coke. The secondary clarifier was also cylindrical, had a diameter of 2.0 m, and a volume of 9.8 m^3. The plant was built in 1997 by construction of a reed bed and a reed pond as tertiary treatment stages (Figure 1). The reed pond was constructed above ground level within earth dikes. This pond was sealed with PVP foil. Its bottom layer consisted of soil 30 cm deep. The mean water depth in the pond was 0.8 m, and mean retention time was 9 days. The seed bed was also made within earth dikes, and PVC foil was used for the sealing of this system. Average retention time in the subsurface system was estimated at 3–4 days. The reed bed was filled with 0.8 m medium sand (grain diameters: $d_{10} = 0.3$ mm, $d_{60} = 0.8$ mm) . The mean capacity of this constructed wetland over the study period was 79±21 $m^3\ d^{-1}$, including 8 m^3 of septage from the holding and septic tanks per workday. The surface area of each of the constructed wetlands was 300 m^2, so the mean hydraulic load was 132 mm d^{-1}.

Both systems work simultaneously, which means that after trickling through the filter, wastewater is divided and pumped between these two systems. *Phragmites australis* are planted in the reed bed and reed pond. The reed pond (G2) and reed bed (G3) characterize surface and subsurface flow respectively. The goal of this study was to show efficiency of the purification of wastewater by a reed pond and a reed bed.

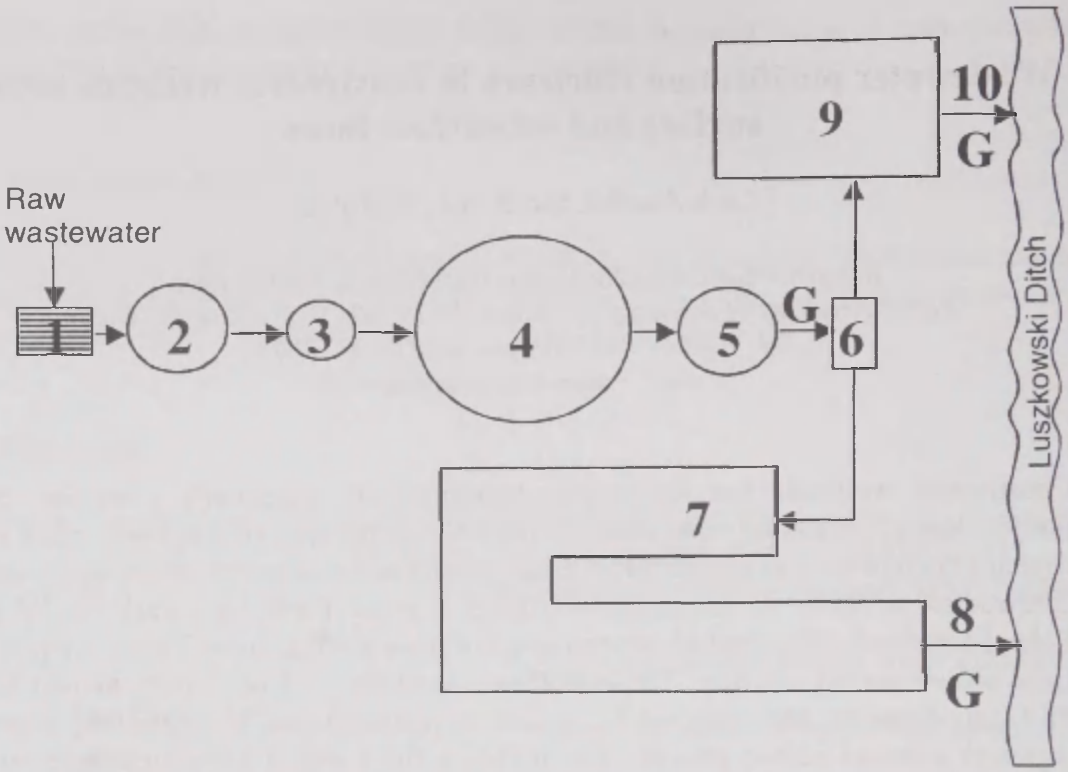

Figure 1. Flow scheme of the wastewater treatment system in Kopaszewo.1 – bar screen, 2 – Imhoff tank, 3 – first pumping station, 4 – trickling filter, 5 – secondary clarifier, 6 – second pumping station and water meters, 7 – reed pond, 8 and 10 – the outlets into the receiving ditch, 9 – reed bed.

The investigations were carried out from April 2000 to December 2001. Wastewater samples were taken twice monthly from three locations in the plant (spot G1, G2 and G3). Location G1 is situated behind the secondary clarifier and the other two locations at the final outlets of G2 and G3 into the receiving ditch. The wastewater was filtered twice using mean density filter paper, and pHs were measured. $N\text{-}NO_3^-$, $N\text{-}NH_4^+$ $N_{Kjeld.}$, N_{Total}, $P\text{-}PO_4^{-3}$, P_{Total} contents were determined. C-organic, C-inorganic concentrations were carried out using the carbon analyzer TOC 5050A-Shimadzu, Japan.

For the estimation of the efficiency of treatment, the factor of the purification of wastewater η was used. This factor was calculated using the following equation:

$$\eta = (1 - C_2 / C_1) \cdot 100\% \qquad (1)$$

where: C_1 – the concentration of the substance in the inlet (spot G1) and C_2 – the concentration of this substance in the final outlet (spot G2 and G3; mg l^{-1}; Table 1).

After planting of the reed in beds in 1998, studies of morphometry, biomass, density and content of nutrients in plants were carried out in the two next years, during the period of the plants' intensive growth. The amount of nutrients stored in plants on the unit of area was the result of biomass and the content of individual elements in tissues. 100 reed shoots from both beds were cut for investigations.

Table 1. Mean values of efficiencies of the treatment for G2 and G3 in (%). G2 – final outlets of reed pond G2 with surface flows, G3 – final outlets of reed bed with subsurface flows

Sampling site	N-NO_3^-	N-NH_4^+	N_{Kjeld}	N-org.	N-Total	P-PO_4^{-3}	P-org.	P-Total	TC	IC	TOC
					2000						
					Spring						
G2	93.9	0.0	0.0	0.0	0.0	0.0	–	–	–	–	–
G3	91.2	0.0	0.0	0.0	0.0	0.0	–	–	–	–	–
					Summer						
G2	84.9	0.0	6.5	13.5	16.6	0.0	–	–	–	–	–
G3	77.0	20.6	15.6	45.2	40.1	0.0	–	–	–	–	–
					Autumn						
G2	41.3	12.8	10.6	9.2	19.6	24.0	–	–	–	–	–
G3	53.9	31.8	35.8	39.3	41.1	13.2	–	–	–	–	–
					2001						
					Winter						
G2	63.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.9
G3	70.4	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	33.2
					Spring						
G2	90.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
G3	77.9	0.0	0.0	39.7	0.0	0.0	0.0	0.0	0.0	0.0	21.4
					Summer						
G2	95.4	0.0	0.0	0.0	0.0	0.0	22.6	6.4	0.0	0.0	0.0
G3	74.5	7.2	9.7	21.2	37.4	10.5	19.8	13.1	0.0	0.0	30.4
					Autumn						
G2	75.8	20.7	21.3	24.6	34.6	16.5	2.6	13.4	0.0	0.0	19.8
G3	28.9	52.0	52.9	59.3	46.9	19.7	0.0	8.7	20.4	11.4	42.5

A significant reduction of different forms of nitrogen, phosphorus and carbon was observed. It was noted that the reduction of nitrates during the entire study period depended on the season. In the reed pond (G2), the concentration of N-NO_3^-

decreased by 93.9% in spring 2000. A similar trend was evident in spring 2001, when a 90.1% decrease in concentration took place.

The lower efficiency of treatment (84.9%) in the reed pond (G2) was more notable in summer than in spring 2000. The concentration of $N\text{-}NO_3^-$ decreased in summer from 7.62 mg/l to 1.15 mg/l. The summer of 2001 was characterized by a greater decrease of $N\text{-}NO_3^-$ than in the summer of 2000, i.e. from 12.53 mg/l to 0.98 mg/l. The factor of efficiency for this elimination of $N\text{-}NO_3^-$ was equal to 95.5%. In autumn 2000, a lower rate of this process was noted. The concentration of $N\text{-}NO_3^-$ in the inflowing wastewater (location G1) was 41.3% higher than in summer. In the colder months, as a result of a decrease in the rate of chemical, biochemical and physico-chemical processes of importance for the operation of this system, a small decrease in $N\text{-}NO_3^-$, from 12.8 mg/l to 7.51 mg/l in reed pond G3 was also observed. The efficiency of the treatment of $N\text{-}NO_3^-$, (34.9%) in winter was lower than in autumn.

The highest value of the efficiency of ammonium treatment was observed in autumn 2000. For the reed pond (G2), η was equal to 12.8%, but in reed bed G3, η=31.8%. A low concentration of ammonium in G3 was observed in summer 2000, when η=20.6% and in 2001 η=7.2%. The low content of this compound during the said season may be explained as follows. In this season ammonium was absorbed by the plants and participated in the process of nitrification. In the winter, as a result of the coexistence of the decrease in the rate of biological processes, and also the decrease in the rate of nitrification, the $N\text{-}NH_4^+$ content in reed beds and reed ponds was high. In the winter, the decomposition of the rest of the plants supplied an input of nitrogen. The Increase of $N\text{-}NH_4$+ in the reed bed and in the reed pond was caused by the lower absorption of $N\text{-}NO_3^-$ by the plants, and also as a result of the reduction of $N\text{-}NO_3^-$ to $N\text{-}NH_4$+ by H_2S.

In the reed pond (G2) and in the reed bed (G3), a decrease of N_{Total} was observed in summer 2000 (η=16.6% and η=40.1%, respectively). In autumn the highest efficiency of treatment was noted in G3, with η=41.1%, and in G2 η=19.2%. In the winter, as result of the low rates of the biochemical processes in the reed bed and reed pond, the efficiency of the treatment was close to zero.

Two forms of phosphorus were also determined. Our investigation showed that the efficiency of the purification of forms of phosphorus was lower than for nitrogen compounds. In autumn 2000 the factor of efficiency of the treatment (η) of $P\text{-}PO_4^{-3}$ in G2 was equal to 24.0%, but was 13.2% in G3. In the winter, as a result of the low rate of biological processes, a decrease in the rate of purification of phosphates was observed, and the value of η in the reed pond was equal to 19.4%, but in G3 was smaller, i.e. η=6.2%. The removal of total phosphorus was observed in the summer 2001 in the reed bed and reed pond. In this season the factor of efficiency was equal to 6.1% in the reed pond (G2), and was 13.1% in

G3. During winter and spring 2001 the purification of phosphorus dramatically decreased, and η was zero.

Our results show that the removal of nitrogen, phosphorus and organic carbon from wastewater by a reed bed with subsurface flow (G3) was higher than by a reed pond with surface flow (G2). During the whole 2001 G3 removed 116.1 kg N, 18.4 kg P and 100.4 kg of total organic carbon, whereas G2 removed 101.9 kg N, 16.4 kg P and 26.6 kg of total organic carbon.

The potential rate of nutrient uptake by a plant is determined by its net productivity and the concentration of nutrients in the plant tissue. The mean density of the reed in reed pond G2 in 2001 was equal to 110 plants per m^2. This value was 14.5% higher than in 2000. In 2000 the content of nitrogen in the reed in G2 and G3 was equal to 30.3 g m^{-2} and 41.2 g m^{-2} respectively, but in 2001 was 31.9 g m^{-2} and 51.3 g m^{-2} respectively. In 2000 all reeds from G2 and G3 removed 302 kg N ha^{-1} yr^{-1} and 412 kg N ha^{-1} yr^{-1}, respectively. In 2001 a higher nitrogen content in the tissue of plants was observed than in 2000. In this year the reed in G2 removed 319 kg N ha^{-1} yr^{-1} and from 513 kg N ha^{-1} yr^{-1} from G3.
In 2001 the phosphorus content in the reeds of the reed pond (G2) was equal to 1.27 g m^{-2}, and the biomass of reed was equal to 979.0 g m^{-2}. In 2000 the phosphorus content in plants was 7.3% higher than in 2001, but the biomass was lower, i.e. 808.4 g m^{-2}. In reed bed G3 the phosphorus content in reeds in 2000 was equal to 2.41 g m^{-2}, and in 2001 was equal to 2.19 g m^{-2}. The plant biomass in 2000 was 1152 g m^{-2}. In 2001 a 21.3% increase in biomass was observed. In 2000 the reeds in the reed pond removed 13.7 kg P ha^{-1} $year^{-1}$and 24.1 kg P ha^{-1} yr^{-1} from G3, while in 2001 reeds removed 12.7 kg P ha^{-1} yr^{-1} from G2 and 21.9 kg N ha^{-1} yr^{-1} from G3.

The decomposition processes in the wetlands also contribute to nitrogenous oxygen demand and biochemical oxygen demand. In the reed bed (G3), the efficiency of treatment of organic carbon (TOC) in winter 2001 was 33.2%, in spring 21.4% and in summer 30.4%. In reed pond G2, the TOC content decreased in winter 2001 η=14.9%. It can be also said that mineral carbon was not removed.

On the basic of the results presented, the efficiency of the purification of the wastewater in reed bed G3 with subsurface flow was higher than in reed pond G2 with surface flows. The reed bed with subsurface flow revealed a higher efficiency of treatment during the whole vegetation season for nitrates, ammonium, N-total, P-total and phosphate than the reed pond with surface flow. The relationship was observed among the efficiency of the purification of nutrients from the wastewater and the density and biomass of the reed, and also the content of the nutrients in the reed. Higher contents of nitrogen than phosphorus in the reeds of the reed bed and reed pond contrasted with the low contents of various forms of nitrogen and high amounts of phosphorus in wastewater. This may indicate that reed played a greater role in the removal of nitrogen from wastewater than did phosphorus.

Laboratory study on trapping efficiency in wetland ponds

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Introduction and objectives

The objectives of this study have been to develop a better understanding of the flow and sedimentation patterns under controlled conditions in the laboratory. Specifically, the influence of vegetation on the flow structure as well as the deposition of sediments has been addressed.

Model set-up

From the hydraulic key data of northern constructed wetlands (Norway, Finland, Estonia, Sweden) compiled in a questionnaire (Schmid, 2001), it became obvious that the flow conditions in most wetland ponds are laminar or near-laminar. To consider also viscous influences on the current due to these very low Reynolds numbers, it is necessary to use a 1:1 model scale.

The physical model, the geometry of which was chosen on basis of the aforementioned questionnaire (Schmid, 2001), consists of two almost 40 m long and 1.5 m wide straight, rectangular canals. Thus, the whole model length is about 80 m. Plant stems cover the first length of the model (Figure 1). The two straight canals are connected by a 180° curve. The layout was inspired by the pilot project Hovi in Southern Finland (Koskiaho, 2003).

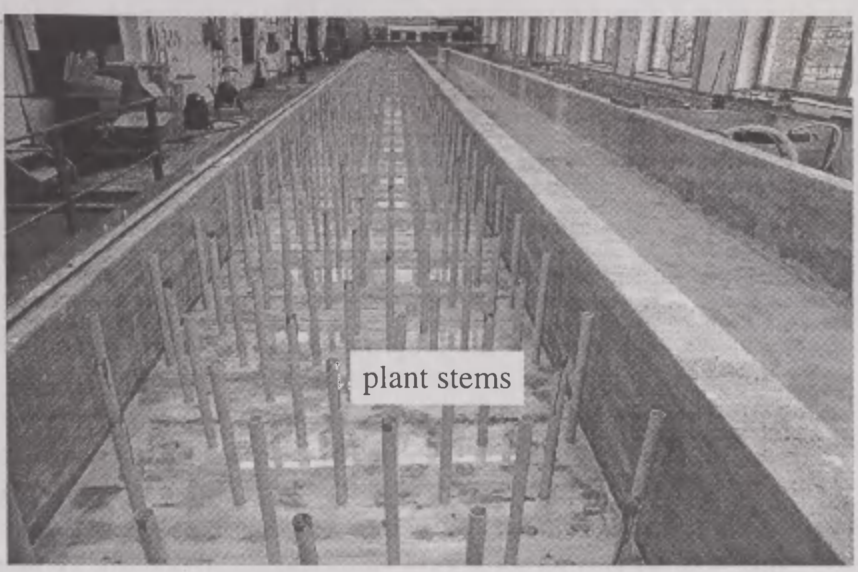

Figure 1. Physical model with artificial vegetation.

Definition of governing parameters

To analyse the main influence on the deposition process and the particle motion in the fluid in wetlands one has to distinguish primarily between sedimentologic and hydraulic influences. On the one hand, it is important to know the physical properties as well as the concentration of the suspended solids TSS_{in}. Thus, kaolin was taken as the modelling substance with ρ_S=2600 kg m^{-3} and a mean particle diameter between 7.4 and 8.6 μm as kaolin is a well know and uniform material with clear chemical and physical properties. On the other hand, the movement of each particle is a combined movement of both the settlement of the particle and the fluid motion. The fluid motion can be described by the hydraulic parameters flow velocity v, flow depth H as well as Reynolds number Re characterising a laminar or turbulent flow situation. In addition, the flow velocity is also influenced by the existence of plants in the wetland, which may possibly lead to different deposition processes of the suspended solids.

Summarising all influences, the governing parameters, which were varied in the physical model in different experimental runs are: v, H, Re, TSS_{in} and stem distance x_P (see Table1). A questionnaire concerning these main parameters gave hints for the main boundary conditions for the hydraulic, sedimentologic as well as plant parameters according to what was found in natural and constructed wetlands.

Table 1. Derivation of the model parameters according to the field parameters

	Wetland data (Schmid, 2001; Koskiaho, 2001)	Physical model
Flow depth H [m]	0.2 to 4.0	0.2 and 0.5
Reynolds number Re [–]	13 to 983 (Berg-Kinn/Norway Re = 64103)	1000 and 2000
TSS_{in} [mg l^{-1}]	8.4 to 4400	200 and 2000
Stem diameter d_P [cm]	0.5 to 5	3.2
Stem distance x_P [cm]	3 to 30	14 and 28 (=49 and 13 stems m^{-2})

Experimental programme and collected data

The parameter variation focused on the changes and dependencies in the deposition process due to different boundary conditions, i.e. parameter combinations. Thus, three main experimental series were conducted (see also Table 2):

a) Series I – variation of hydraulic parameters (H = 0.20 / 0.50 m and Re = 1000 / 2000),
b) Series II – variation of hydraulic and sedimentologic parameters (H = 0.20 / 0.50 m, Re = 1000 / 2000 and TSS_{in} = 200 / 2000 mg l^{-1}),
c) Series III – variation of hydraulic, sedimentologic and plant parameters (H = 0.20 / 0.50 m, Re = 1000, TSS_{in} = 200 / 2000 mg l^{-1} and x_P = 14 / 28 cm).

The collected data are:

a) Distribution of flow velocities,
b) Qualitative information on the vertical distribution of TSS concentration in several cross sections,
c) The TSS concentration [mg l^{-1}] of the water body at the model inlet and outlet and, in addition, in one cross section (x = 34.15 m) at different times during each experimental run,
d) The mass per unit area [g m^{-2}] of the deposited material in longitudinal direction after the end of each experimental run.

This paper focuses on c). The first step was to calculate the trapping efficiency E_{TSS} for each experimental run from

$$E_{TSS} = 1 - \frac{TSS_{out}\,[\mathrm{mg/l}]}{TSS_{in}\,[\mathrm{mg/l}]} \tag{1}$$

In a second step, the trapping efficiencies of different experimental runs were compared to each other to analyse them for each individual parameter influence.

Results and discussion

Preliminary results on the trapping efficiency of the used modelling substance kaolin indicate that the bulk trapping efficiency is quite high and ranges from 54% to almost 100% depending on the boundary conditions. For each experiment, one to two replicas were taken and the mean standard deviation σ ranges from 1 to 10%. To analyse the influence of each governing parameter on the trapping efficiency, the ratios of the trapping efficiencies were taken for experimental runs with variation of one parameter while the others were kept constant. Consequently, the ratios comparing the following parameters

a) high and low Reynolds numbers, i.e. high and low discharge per unit width,
b) high and low input concentrations,
c) the situation with and without plant stems,
d) high and low plant density

were calculated. Figure 2 shows the results for c) as an example. It indicates that, for flow through vegetation, the trapping efficiency seems to depend on both the hydraulic flow condition and the input concentration of the flow. But these influences, however, are rather complex and, thus, the interpretation just allows a qualitative statement at this time.

Table 2. Examples of experimental runs.

Run	Q [l/s]	H [m]	v [mm s^{-1}]	Re [-]	TSS_{in} [mg l^{-1}]	x_P [m]	E_{TSS} [%] (outlet)
R1	1.6	0.2	5.3	1000	200	---	90.5
R3	1.6	0.5	2.1	1000	200	---	75.9
R5	1.6	0.2	5.3	1000	2000	---	90.6
R7	1.6	0.5	2.1	1000	2000	---	84.8
R9	1.6	0.2	5.3	1000	200	0.14	80.3
R10	1.6	0.5	2.1	1000	200	0.14	86.3
R11	1.6	0.2	5.3	1000	2000	0.14	92.6
R12	1.6	0.5	2.1	1000	2000	0.14	97.1

The comparison was done for low TSS input concentration TSS_{in} and small and large flow depths (R9/R1 and R10/R3, see Table 2), *i.e.* high and low flow velocities, respectively. While the trapping efficiency deteriorates due to the existence of plant stems in the first case, it improves in the second. In case of R11/R5 and R12/R7 for high TSS input concentration and high and low flow velocities this tendency is much weaker but also shows. From these results the conclusion is drawn that, first, the trapping efficiency, generally, is higher for high input concentrations TSS_{in} than for low TSS_{in} which could also be derived from corresponding analyses. These results could be explained by flocculation processes due to higher TSS input concentrations which leads to an increased mean particle size and, thus, to an increased settling velocity (Lau, 1993). Secondly, the influence of plants on the trapping efficiency depends on the general hydraulic conditions, and plants do not necessarily improve the trapping efficiency. Due to the constant water level for the flow situation with and without plant stems the mean flow velocity is higher for flow through plants, because the plant stems reduce the cross sectional area. This leads to a notably (R9/R1) or slightly (R11/R5) reduced trapping efficiency for the low flow depth (H=0.2 m) and high flow velocities. In case of high flow depth (H=0.5 m) and low flow velocities the trapping efficiency is not reduced but slightly improved due to plant stems. In spite of an increased mean flow velocity the trapping efficiency is not decreased, which might be due to an altered vortex structure of the flow. However, two different hydraulic situations were analysed and the trapping efficiency turned out to behave completely different. Further analyses are expected to define the situation more clearly.

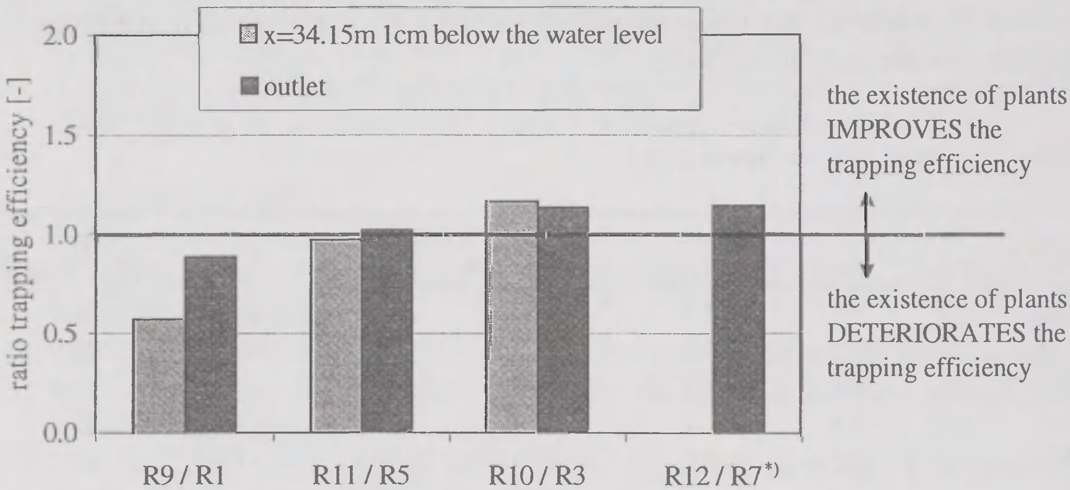

Figure 2. Comparison of the trapping efficiency with and without plants (experimental runs see Table 2), *) no data available for run R7 at x = 34.15 m.

Conclusions

Generally, preliminary results indicate that the trapping efficiency is higher for low flow velocities and high TSS input concentrations. For flow through vegetation, the trapping efficiency tends to depend on hydraulic flow condition and is the higher the lower the mean flow velocity is. Thus, the present analyses reveal that the existence of plants does not necessarily improve the trapping efficiency in all cases.

Acknowledgement

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A polyphasic analysis of the microbial community structure of a planted soil filter for the treating of domestic wastewater

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Introduction

The main processes used for purifying wastewater in constructed wetlands are mediated by microorganisms. Most investigations of microbial communities in constructed wetlands rely on direct or indirect methods of enumeration or measurement of microbial biomass and activity.

The aim of this study was to analyze microbial communities from Kodijärve subsurface flow constructed wetland using a polyphasic approach applying both culture-based and molecular methods.

Material and methods

The Kodijärve horizontal flow planted soil filter consists of two beds (each 25*6.25*1 m) filled with coarse sand. The beds are called a dry bed and a wet bed due to the different water levels therein.

The soil samples for the microbiological analyses were collected around individual water sampling wells at two depths, 0–10 cm ("upper layer") and 50–60 cm ("lower layer") with a soil core drill in October 2002 from the dry bed. Twenty sub-samples were pooled from the composite sample and were stored at –20°C until analysis.

The number of aerobic heterotrophic bacteria was determined on R2A (Difco) agar plates using the drop-plate method. The number of ammonium oxidizers was determined using the most probable number method on microplates. Microbial DNA was extracted from soil samples using the UltraClean Mega Soil DNA kit (Mo Bio Laboratories, Inc.). Bacterial community structure was assessed using two primer pairs, 318f-GC/535r and 968f-GC/1401r respectively. The community structure of ammonium oxiders was determined using two primer pairs. One primer pair CTO189f-GC andCTO654r amplifies a fragment of the 16S rDNA gene from Proteobacteria β-subgroup ammonium oxidizing bacteria. A second primer pair AmoA-1F (forward) and AmoA-2R-TC targets the ammonium monooxygenase gene from a β-subgroup ammonium oxididizing bacteria. Fragments from archaeal 16 rDNA were amplified with the nested PCR approach. A denaturating gradient gel electrophoresis (DGGE) system DCode (Bio Rad, Inc.) was used to separate the amplified gene fragments. The banding pattern of the obtained gels was analyzed using Principal Coordinate Analysis and a multivariate randomization test.

Results

The comparison of the number of heterotrophic bacteria between inlet and outlet pipe as well as between two sampling depths showed no significant differences. In addition, the variation of abundance of ammonia-oxidizing bacteria showed no clear spatial pattern.

To compare microbial communities within constructed wetland, molecular methods were used. The Denaturing Gradient Gel Electrophoresis (DGGE) method based on universal bacterial primers distinguished microbial communities depending on depth and inlet or outlet. The difference between sampling depths may be related to the vertical gradient of chemical parameters such as redox potential and the oxygen level in the filter. In general, the diversity of the bacterial community was higher in the upper layer than in the deeper horizon, where the microbial community was characterized by less than ten dominant species. At the same time, the bacterial community of the deeper horizon exhibited greater heterogeneity than the upper layer. The higher complexity of bacterial community in the filter's upper layer may be supported by plant growth, presumably through rhizodeposition.

The same method was used to analyze communities of autotrophic ammonia oxidizers and archaea. The community diversity of ammonia oxidizers depended on depth, while archaeal communities were distinguished depending on inlet or outlet. The distribution of ammonium monooxygenase genes in the filter bed was quite similar to the spatial structure of the ammonia oxidizers' community.

Archaeal community structure near the inlet pipe was vertically more homogenous than at the outlet.

The reasons for microbial community variability are not clear due to the absence of the soil's chemical data. We can only speculate that changes in the quantity and quality of organic matter as well as in the concentration and spatial distribution of electron acceptors within wetlands are one probable explanation for the spatial variation of microbial community. Another important factor that may affect microbial community structure in constructed wetlands is above-and below ground plant production as well as plant diversity. It will be necessary to analyze both microbial and chemical data from the studied wetland together to validate these speculations.

Cold climate phosphorus uptake by submerged aquatic weeds in a treatment basin

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Abstract

In late autumn, biomass phosphorus (P) of submersed aquatic weeds, mainly slender waterweed (*Elodea nutalli*, L.) and sago pondweed (*Potamogeton pectinatus*, L.) was measured in shallow zones of a sewage treatment basin. Together these areas represented 27% of the total area of the basin. The greenshoots contained 0.02 kg P and the coarser vegetation parts 0.04 kg P. In total this was equivalent to 2.5% of the daily phosphorus load from sewage water. After exposure to a pulse of P-32 ($18*10^9$ Bq), the greenshoots close to the site of injection had higher average β-activity (250 disintegrations per minute and milligram dry weight (DPM mg dw)) than the coarser vegetative parts of the submerged plants (120 DPM mg dw). However, since the latter represented more biomass, nearly as much P-32 was taken up through processes associated with the coarse vegetation than those associated with the greenshoots. In two shallow central zones the former was equal to 0.009% of added radioactivity, while 0.011% was found in the greenshoots in the same zones of the basin. Close to the inlet a fast assimilation of the biomass P was demonstrated. The role of submerged weeds in phosphorus turnover in sewage treatment basins under winter conditions is discussed.

An assessment of the hydraulic properties and performance of a subsurface flow constructed wetland, Nowa Słupia, Poland

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The sub-surface flow constructed wetland in Nowa Slupia is an example of the most common type of constructed wetland in Poland. The wetland was constructed in 1995 in order to treat municipal wastewater. It is located in the Holy Cross Mountains region (21°05' E – 50°52' N) at 250 m a.s.l. with an annual mean temperature of 8°C and annual precipitation of 700 mm. The wetland consists of three parallel gravel beds (78m x 24m x 1.2m each) overgrown with common reed (*Phragmites australis*). A sedimentation pond and an aeration tank provide preliminary treatment. A polishing pond (volume of 750 m^3) is used as the final stage of treatment. The main aim of this research was to evaluate the wetland's performance in summer and late autumn conditions after nine years of operation, using combined hydraulic, physicochemical and biological observations.

Three tracer tests were performed in July 2001 (KBr), November 2001 (KBr) and June 2002 (KBr and tritium) in order to investigate the wetland's hydraulic properties. The tracer tests were accompanied by measurements of the basic physicochemical characteristics of the influent and effluent. In addition, during the second and third tests nutrient contents in the influent and effluent as well as fluxes of CO_2 and CH_4 released from the wetland into the atmosphere were determined. The following qualities of the wastewater were measured: temperature, pH, electric conductivity, dissolved oxygen, TSS, BOD_5, COD, TOC, TKN, N-NH_4, N-NO_3, TP and P-PO_4. All analyses were performed over a five-day period after tracer injections, so that observations of nutrient cycling could be related to the actual hydraulic characteristics of the wetland. The density and biomass of reed specimens were measured monthly from March to December 2002. The phosphorus and nitrogen contents of plants were determined once, at the peak of the growing season.

Figure 1 presents the results of the tracer test performed in June 2002. Discharge measurements revealed that one of the beds received significantly more hydraulic loading than the other two. Moreover, patches of water on the surfaces of all beds, as well as the short-circuiting of wastewater flow through the surfaces of gravel beds were observed. These hydraulic phenomena are unfavourable from the viewpoint of wetland performance, as the influent is distributed unevenly between the beds and within each bed. Uneven distribution of wastewaters between beds is clearly reflected in breakthrough curves presented on Figure 1, as most of the tracer was discharged through bed 1. The quantitative hydraulic characteristics of the wetland (water residence times, active volumes, flow velocities, dispersive characteristics) were inferred from tracer residence time distributions (RTD) using mathematical modelling. It must be noted that wetland charactersistics derived from bromide and tritium RTDs agree within 10%, and recoveries of both tracers were even closer to one another. The similar behaviour of both tracers shows that bromide can be regarded as a conservative tracer in the environment of sub-surface flow wetlands. Figure 2 compares the measured and modelled RTDs of the tritium tracer for bed 2. The total flow was mathematically deconvoluted into three components that are clearly visible in the RTD plot. Three flow components were also identified in the RTDs obtained for the other two beds. Physically, these flow components could be related to preferential flow pathways and patches of stagnant water. The flow velocities varied from 3 m d^{-1} to 45 m d^{-1}, whereas dispersivities varied from 3 to 20 m.

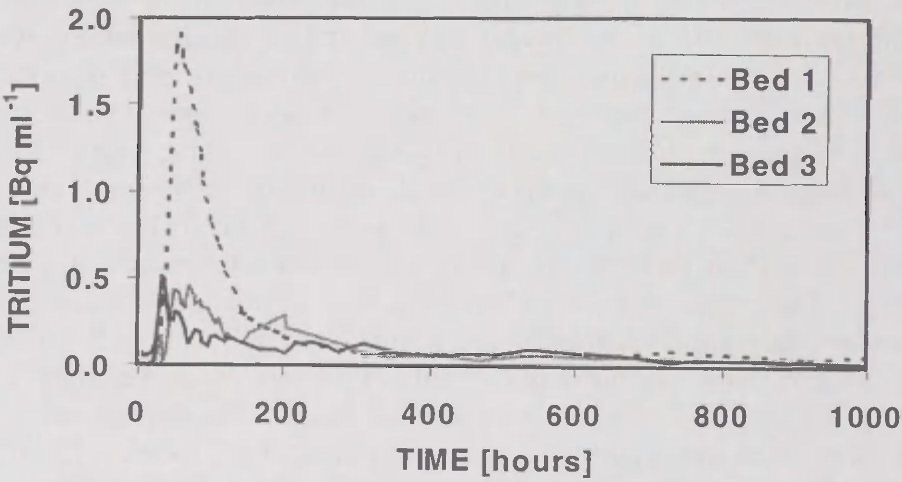

Figure 1. Breakthrough curves obtained after instantaneous injection of 25 mCi of tritium.

Table 1 compares the efficiencies of removal of selected pollutants for all wetland beds, in late autumn and summer conditions. The removal of nutrients was generally poor, with release of N and P in some cases. Only TSS, BOD_5 and COD were removed efficiently (removal between 40% to 80%). Removal efficiencies for organic carbon, nitrogen and phosphorus were higher in late autumn than in summer.

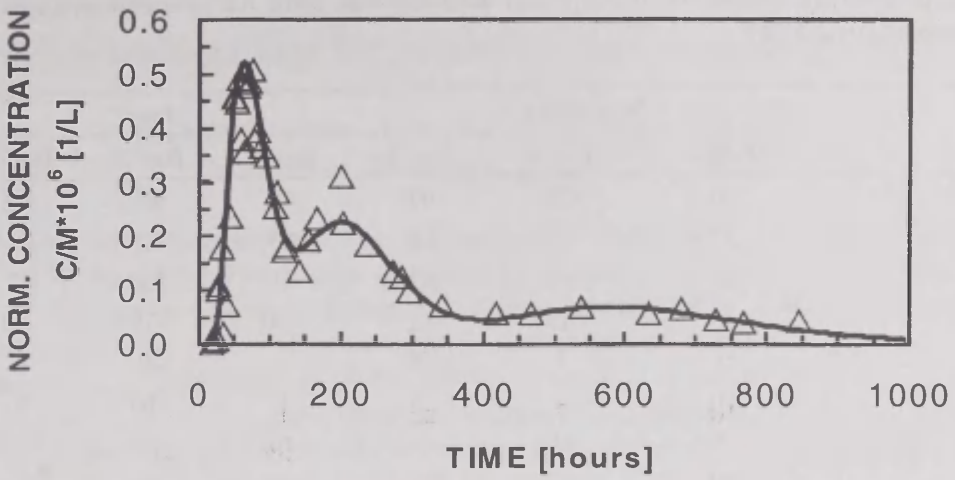

Figure 2. Residence time distribution of tritium tracer in bed 2.

Bed 3, which received the lowest hydraulic loading, removed nutrients more effectively than the other two beds. The removal of $N\text{-}NO_3$ appeared to be very poor, but low concentrations of $N\text{-}NO_3$ could not be determined with a high rate of precision. Poor removal of ammonia and total nitrogen results from insufficient oxygen supply, which limits the oxygenation of ammonia and organic nitrogen. The low removal efficiency of total phosphorus may be related to the reduced sorbing capacity of the clogged medium and to internal recycling of this element. Effluent limits (according to Polish legislation) were exceeded only in bed 2 in November for TP, TN (TKN + $N\text{-}NO_3$) and COD. It must, however, be noted that the wetland receives only municipal wastewater with moderate nutrient content. The quality of the effluent is finally improved in the polishing pond.

Fluxes of CO_2 and CH_4 into the atmosphere showed significant variability between beds and within each bed. There was a large decrease in CO_2 and CH_4 fluxes along beds, from inlets to outlets. CO_2 fluxes showed diurnal fluctuations with maxima in the daytime. This diurnal pattern of CO_2 emissions can be explained by changes in the availability of oxygen in the beds, which were in turn related to the photosynthetic activity of the plants. The details of gas flux observations are presented elsewhere in this volume.

Inter- and intra-bed differences were also reflected in the seasonal and spatial patterns of reed density, biomass and nutrient content, all of which were lowest in bed 1. Plant distribution and biomass density partly reflected the availability of wastewater to plants. The details of plant observations are presented elsewhere in this volume.

Table 1. Removal efficiencies (%) in November 2001 and June 2002. All values are averages of 8 results obtained over 5 days.

	November			June		
	Bed 1	Bed 2	Bed 3	Bed 1	Bed 2	Bed 3
TSS	40	58	67	49	60	82
BOD_5	72	74	88	63	62	71
COD	61	43	57	41	59	67
TOC	33	45	45	–10	–35	–21
TKN	9	7	15	1	–9	4
N-NH4	10	7	22	1	–10	5
N-NO3	–5	–91	–9	–66	–64	–117
TP	20	28	14	–18	–8	11
P-PO4	–7	–15	–14	–37	–22	10

Three identical beds in the constructed wetland in Nowa Słupia receive different hydraulic loadings and reveal observable differences in their hydraulic properties, vegetation characteristics and performance.

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Scale methodology for nutrient export in stream networks

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The future European water policy calls for methods by which we can estimate effects of different pollutant pressure on the export load due to transport in surface waters from catchments. Special focus of this study is to predict phosphorus transport in the stream networks and, furthermore, to take into account the marked heterogeneity that characterises many stream properties. The study is based on data from Morsa catchment in Norway close to Oslo, a small and basically agricultural catchment.

A model framework that can be generalised to conditions other than those under which it was developed need to be based on a physically plausible representation of the governing exchange processes as well as a statistical representation of the stream network characteristics. An essential process to the solute transport in streams is the flow-induced exchange between the stream channels and the hyporheic zone, in which reactive solutes like phosphorus are adsorbed to particulate material and can be retained for significant periods of time. This study applied a hydrodynamic description of the hyporheic exchange and combination of statistical principles to represent the network geometry and the variation of stream properties.

Evaluation of the geomorphological instantaneous unit hydrograph (GIUH) of Morsa catchment indicated that a fractal representation of the stream network prevails. Several stream tracer tests and field surveying gave a quantitative picture of both spatial tendentious variation (with position in the catchment) as well as more erratic variation.

The proposed model framework convolutes unit solutions with account taken to the physical scales of the hyporheic exchange, the fractal dimensions of the stream network and the spatial heterogeneity of transport properties. Comparison between the model behaviour and data of P-transport in Morsa catchment suggest an acceptable agreement.

Removal of heavy metals in a horizontal subsurface flow constructed wetland

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Constructed wetlands (CWs) with horizontal sub-surface flow, designed for the treatment of municipal sewage, have been monitored extensively with respect to removal of organics (BOD_5, COD), suspended solids, nutrients (N, P) and microbial pollution. However, the information on the removal and distribution of heavy metals in these systems are very limited. Heavy metals do not constitute a major problem in municipal or domestic wastewaters but the question of heavy metal concentrations could be important when plants or sediments need to be disposed. The concentrations of heavy metals in municipal sewage are usually low but metals tend to accumulate in sediments as well as in the plant biomass so the importance of heavy metal analyses increases over the operational period. The purpose of this study was to evaluate removal and retention of Al and 11 heavy metals: As, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sn and Zn, in constructed wetland at Morina near Prague during the third year of operation.

CWs at Morina (Figure 1) has been in operation since 2000. It is designed for secondary treatment of sewage from 770 PE but in 2000 only about 250 people were connected. The total area of vegetated beds 3520 m^2 is divided into four beds, each of 880 m^2. After pretreatment (screens, Imhoff tank) the wastewater is directed to a distribution box and then to two parallel beds. At the outlet from the first beds the water is collected and distributed to the second series of parallel beds. The designed mean flow is 98 $m^3\ d^{-1}$ but the measured average flow was 146 $m^3\ d^{-1}$ in 2002 due to the storm water penetration into sewer system under construction. The resulting hydraulic loading rate was 4.1 cm d^{-1}. Substrate is crushed rock (fraction 4–8 mm) and the average bed depth is 0.6 m. *Phragmites australis* (common reed) and *Phalaris arundinacea* (reed canarygrass) are planted in stripes perpendicular to wastewater flow.

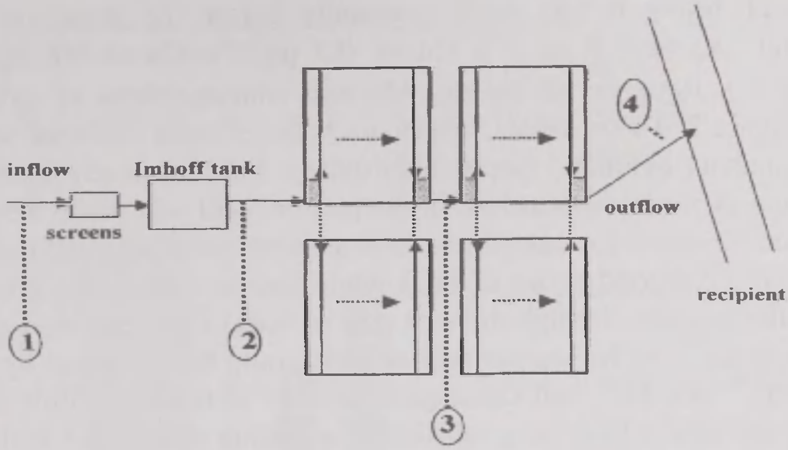

Figure 1. Constructed wetland at Morina. Numbers indicate sampling points. Not to scale.

Water samples were taken on monthly basis during 2002 at four locations (Figure 1): raw wastewater, after mechanical pretreatment, after first pair of beds and final outflow. In addition to organics, suspended solids, nutrients, Al and 11 heavy metals also SO_4^{2-}, Na, K, Ca, Mg, O_2 and pH were measured. Plant biomass was sampled at the time of peak biomass in September and separated into stems, leaves, flowers, roots and rhizomes.

In Table 1, removal of organics, suspended solids and nutrients is presented. The overall treatment efficiency is excellent for organics and suspended solids, removal of nutrients is lower but exhibits typical values for CWs with horizontal sub-surface flow. pH values are quite stable averaging 7.76 and 7.69 in inflow and outflow, respectively. Average dissolved oxygen concentrations were 7.1 mg l^{-1}, 3.3 mg l^{-1}, 0.1 mg l^{-1} and 1.6 mg l^{-1} at sampling points 1 to 4, respectively.

Table 1. Treatment performance of constructed wetland at Morina in 2002. Numbers in mg l^{-1} represent average values. For sampling points see Figure 1.

Parameter	Sampling point				Removal (%)
	1	2	3	4	
BOD_5	64	38.6	11.9	3.9	93.9
COD	167	97	26.2	17.4	89.6
Susp. Solids	155	30.6	3.6	2.5	98.4
Tot. P	3.66	2.91	2.65	2.17	40.7
NH_4^+-N	12.9	15.5	13.4	10.8	16.3
NO_3^- -N	4.3	2.1	0	0	100
NO_2^- - N	0.6	0.4	0	0	100
Org. N	6.2	4.9	1.7	1.2	83.9
N_{kj}	19.1	20.4	15.1	12.0	37.2
Tot. N	24.0	22.5	15.1	12.0	50.0

Four out of 11 heavy metals were constantly below the detection limit in raw wastewater: As (< 2.5 µg l^{-1}), Cd (< 0.2 µg l^{-1}), Co (< 0.8 µg l^{-1}) and Sn (< 3.0 µg l^{-1}). Removal of Fe and Mn and concentrations of sulphate are presented in Figure 2. These metals, which are believed to be removed via similar removal mechanisms, exhibited surprisingly quite a different removal pattern. The highest retention of iron was observed in the pretreatment unit while there was no such a retention observed for manganese. Iron concentration increased after water passage through the second series of beds while this increase has been observed for Mn also after passage through the first pair of beds. The steep increase of Mn and Fe concentrations in the second pair of beds could be explained by bacterial reduction of Mn^{4+} and Fe^{3+} and consequent release of reduced forms Mn^{2+} and Fe^{2+}. The data in Table 1 indicate a substantial reduction of organics in the second pair of beds, i.e. between sampling points 3 and 4. The data also indicate that nitrate is absent and therefore, manganic Mn and then ferric Fe are used by microorganisms as the terminal electron acceptors during respiration. Sulphate was reduced primarily in the first pair of beds while no reduction occurred in the second pair of beds (Figure 2).

In Figure 3, retention of copper, chromium, lead and nickel is presented. Inflow concentrations were quite low and for Cr, Cu and Pb the concentrations were below the detection limit at sampling points 2 (Cr < 0.5 µg l^{-1}) or 3 (Cu < 2.0 µg l^{-1}, Pb, < 2.0 µg l^{-1}). Nickel was removed in pretreatment units and the concentrations went slightly up, probably due to oxidation-reduction conditions in the bed. Removal of zinc exhibited gradual decrease along the treatment path – 109, 73, 10.5 and 6.4 µg l^{-1} at sampling points 1 to 4, respectively. Also retention of Al exhibited the same pattern: 1831, 551, 169 and 329 µg l^{-1} at sampling points 1 to 4, respectively.

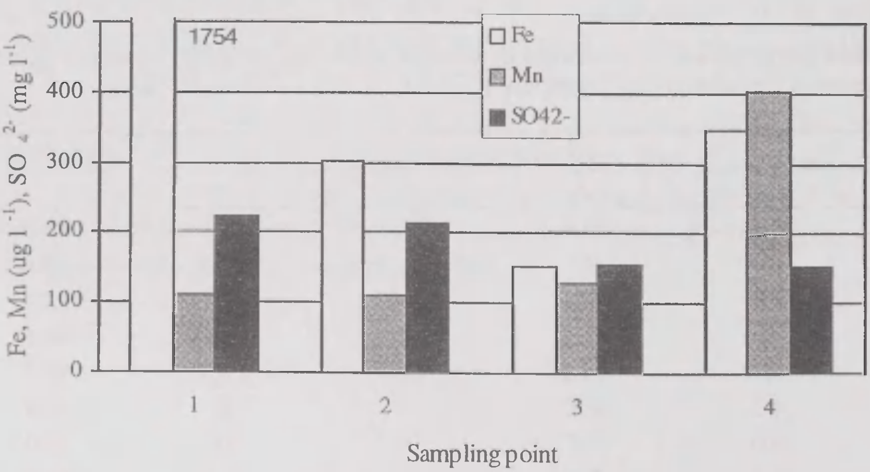

Figure 2. Removal of iron, manganese and sulphate in constructed wetland at Morina. For sampling points see Figure 1.

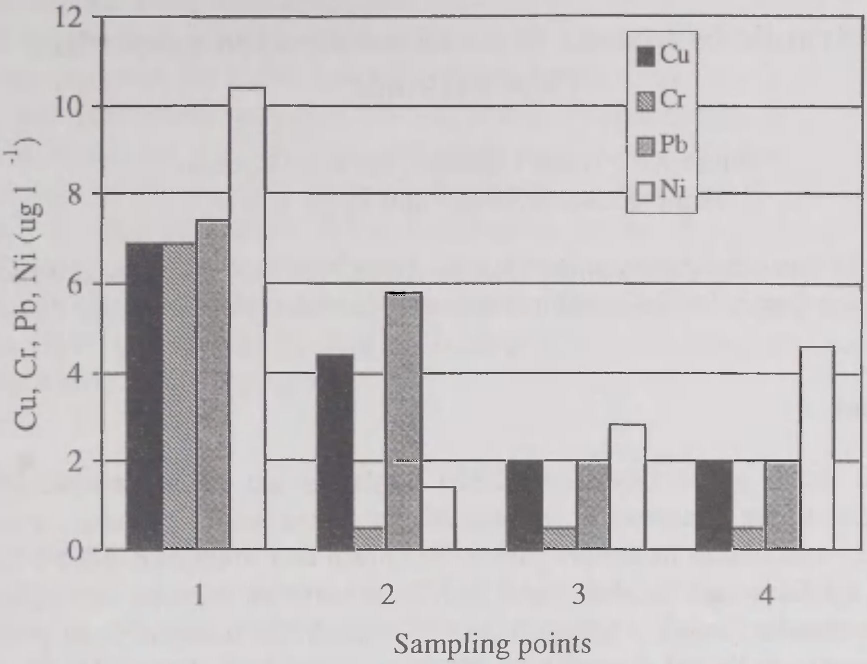

Figure 3. Removal of iron, manganese and sulphate in constructed wetland at Morina. For sampling points see Figure 1.

The analysis of plant material revealed that the highest concentrations of all metals were found in roots while in stems and leaves the concentrations were much lower. The data indicated that concentrations of all studies metals in aboveground plant tissues of *Phalaris arundinacea* and *Phragmites australis* from a constructed wetland at Morina do not show any excessive values and therefore, they do not form any environmental threat after three years of operation.

Acknowledgements

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Hydraulic behaviour of small constructed subsurface flow wetlands

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Introduction

Constructed small subsurface-flow (SSF) wetlands are an alternative in cold climate regions for wastewater treatment for households in rural areas not connected to wastewater treatment plants (Mæhlum and Stålnacke, 1999). Special porous bed medium such as shell-sand with large sorption capacity are applied for such SSF wetlands. Earlier studies show that a sandfilter treatment can provide a sufficient water quality of the effluent discharge from households (Mæhlum and Stålnacke, 1999). To design new treatment wetlands a dynamic model of the governing flow and chemical reaction processes is useful because it allows a number of scenarios to be tested such as critical hydraulic loading, process disturbances and temperature variation.

Owing to the presence of plant roots and other introduced anomalies, we can expect a large variability of hydraulic conductivities of the bed medium in the vertical and horizontal directions. This variability is not only in space but may change with time due to the geomorphological evolution caused by hydraulic loadings, rain and frost.

This study presents a model approach that combines hydraulic conductivity measurements and modelling effects of hydraulic conductivity on flow transport to characterise flow residence time PDF in SSF.

The anomalies in hydraulic conductivity imply that the flow in the wetlands will be dominated by the flow along preferable trajectories. This type of hydrodynamics can be characterised by the residence time probability density function (PDF) defined as $g(\tau)$ of inert water parcels travelling through the wetlands; an approach often referred to as the travel time approach (Dagan, 1989; Rodriguez-Iturbe and Rinaldo, 1997). In this approach we decomposes the transport problem into a multi- dimensional flow problem and one-dimensional transport problem. The predicted concentration of interest is a convolution of chemical/biological reaction model with the residence time PDF. A similar approach is recommended by Kadlec and Knight (1996) in modelling of wetlands.

The groundwater flow in porous medium has been well studied over a long history of research (Darcy, 1856; Freeze and Cherry, 1979; Bear, 1987). In this study we represent the hydrodynamics of groundwater by a non-stationary form of the Dupuit-Forcheimer equation. The hydraulic conductivities in a small wetland with shell-sand as bed medium, are measured in a laboratory set-up and the geostatistics of the conductivities is evaluated. Further, a laboratory tracer test with KI is used as a basis for a verification of the theoretical simulation of hydrodynamics of the wetland based. Numerical experiments were performed to derive relationships, in a more general sense, between residence time PDF and various flow loading and hydraulic conductivities reflecting the events such as rain, snowmelt and clogging etc.

Theory and method

Travel time approach

The proposed design model for SSF treatment wetlands is based on the travel time approach to describe solutes transport processes. The transport problem is decomposed into a multi-dimensional flow problem, in which the trajectory paths of inert water parcels are determined, and a one-dimensional problem in which the mass-transfer between the parcels and shell-sand material is determined (Dagan, 1989; Rodriguez-Iturbe and Rinaldo, 1997).

$$< f(t,\tau) >= \int_0^\infty f(t,\tau) g(\tau) d\tau \qquad (1)$$

in which g(τ) is the travel time probability density function (PDF) for a large number of inert water parcels arriving at a certain control section for the whole distribution of trajectories in SSF wetland, τ is the residence time of an inert water parcel travelling along one of the trajectory paths, and f(t, τ) is the residence time PDF of solute mass in the water parcel, where $f(t,\tau) = c(t,\tau)/M_0$, M_0 is the total mass of the solute inserted into the SSF wetland [kg], c is the concentration of solute per unit volume of water [kg m^{-3}] and t is the time [s].

Method to determine the hydraulic residence time PDF, g(τ)

A solute transport equation is used to describe the inert water parcels transport in SSF wetland:

$$\frac{\partial(\theta c)}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial c}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(\theta u_i c) + \sum R_R \qquad (2)$$

where c is the dissolved concentration of species [kg m^{-3}]; θ is the porosity of the subsurface medium, dimensionless; t is time [s]; x_i is the distance along the respective Cartesian coordinate axis [m]; D_{ij} is the hydrodynamic dispersion coefficient tensor [m^2 s^{-1}]; u_i is the seepage or linear pore water velocity [m s^{-1}]; c_s

is the concentration of the source or sink flux for species [kg m^{-3}]; R_R is the chemical reaction term [kg m^{-3} s^{-1}]. Since we use inert solute species as a tracer to obtain the residence time PDF of flow the last term in (2) is omitted.
The transport equation is related to the flow equation through the Darcy's Law:

$$u_i = -\frac{K_i}{\theta}\frac{\partial h}{\partial x_i} \tag{3}$$

where K_i is a principal component of the hydraulic conductivity tensor [m s^{-1}]; h is hydraulic head [m].
The hydraulic head is obtained from the solution of the three-dimensional groundwater flow equation:

$$S_s\frac{\partial h}{\partial t} = \frac{\partial}{\partial x_i}\left(K_i\frac{\partial h}{\partial x}\right) + L + R \tag{4}$$

where S_s is the specific storage [m^{-1}], L is a leakage term based on the head gradient between layers [m^3 m^{-3} s^{-1}] and R is the leakage from the domain in the bottom layer and precipitation in the top layer [m^3 m^{-3} s^{-1}].

A commercial software Visual Modflow was used to solve the above equations. Visual Modflow was developed by Waterloo Hydrogeologic Inc. based on MODFLOW, MODPATH and MT3DMS (McDonald and Harbaugh, 1988; Harbaugh and McDonald, 1996).

Experiments

A laboratory wetland system with shell-sand as bed medium was constructed in nearly prototype-scale used as study object. The system shown in Figure 1 consists of three sections where (a) and (b) represent sections of coarse and fine sand respectively. The wastewater was pumped into the system at a rate of three liters every hour. The walls of the system were impermeable which yield no-flux boundaries. Tracer experiments were performed using non-reactive tracer potassium iodide to derive the hydrodynamics in the system. Hydraulic conductivity of the wetland system was measured in horizontal, traversal and vertical direction with known separation distances.

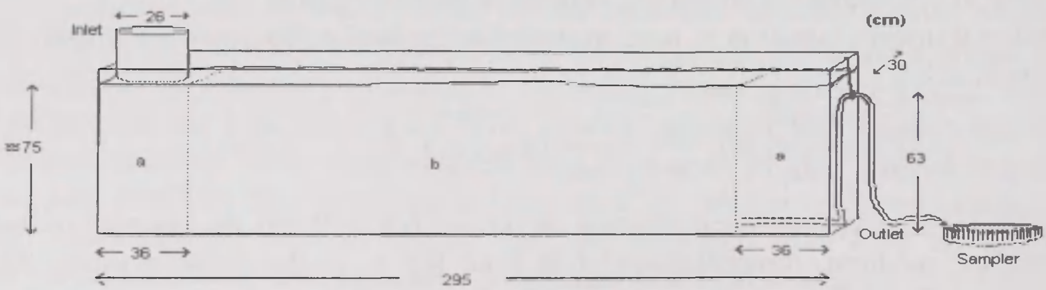

Figure 1. Schematic view of studied sub-surface flow wetland system.

Results

The measured hydraulic conductivity data set was analysed by means of semi-variograms, which gives a basis for conditional generation of stochastic conductivity field. The hydraulic conductivity field is generated by a three-dimensional sequential simulation algorithm, sgsim, given in GSLIB (Deutsch and Jounel, 1992). The conditional generated conductivity field is then imported into Visual Modflow to perform the simulations of the tracer experiment.

The simulated result is in good agreement with the measured data (Figure 2). In the simulation all the parameters are measured data applicable to the experimental set-up except the parameter of diffusivity, which is set as 0.025. This value is in the range of reported in literature (de Marsily, 1986). Only a single realisation of the stochastic hydraulic conductivity field was performed, since it was found that the average result in terms of flow and solute concentration was fairly constant.

Numerical experiments are performed by varying the flow (discharge) and hydraulic conductivity fields. When the conductivity field is unchanged, the peak value of the simulated flow residence time PDF increases with increasing discharge. Contrary, when discharge is kept constant, the peak value of the simulated flow residence time PDF decreases and the PDF became more skewed with increasing variability of the hydraulic conductivity field. Finally, to transfer the understanding of the hydraulic behaviour in small SSF wetlands into practical applications, a relationship is established between the dispersion coefficient, reflecting the hydraulic residence time PDF, and the Peclet number, reflecting the flow load, as well as variability of conductivity field.

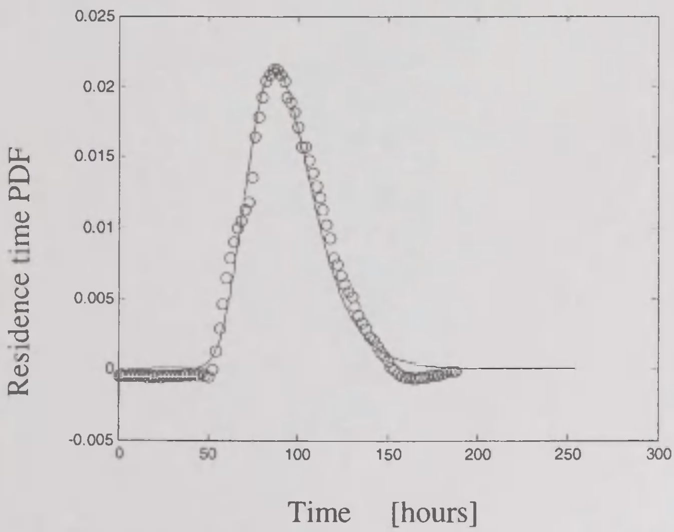

Figure 2. Measured and simulated flow residence time PDF of potassium iodide tracer experiment. The circles denote the measured data and the solid line denotes the simulated result.

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POSTER SESSION

The immobilisation of bacterial phosphorus in constructed wetlands

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Introduction

Purifying sewage of phosphorus (P) using biological means (immobilisation into bacteria) is a good alternative to several chemical means. Normal biological P immobilisation is the amount of P needed to maintain the normal growth and activity of bacteria (1–3% of dry matter; Bitton, 1994; Kunst and Mudrack, 1988). In the recent paper, we concentrated on enhanced biological phosphorus removal (EBPR). This process is very complex and difficult, it contains several metabolic pathways and mechanisms, and there is also a relation with many species of bacteria (Bitton, 1994; Kunst and Mudrack, 1988; Stante *et al.*, 1997; Ahn *et al.*, 2002; Mino *et al.*, 1998). Despite over twenty –years of research, this process has not been completely solved. It would be especially necessary to investigate the biochemistry of this process in order to understand enhanced biological phosphorus removal.

The principle of EBPR is the following: in the aerobic cycle, the energy from degrading polyphosphates is consumed, and at the same time organic matter is converted to poly-hydroxy alkanoates (PHA); in the anaerobic phase the PHA is oxidized and polyphosphates are synthesized. The difference between the aerobic and anaerobic phases gives us the amount of P removed. This principle is mostly used in conventional water treatment plants, but can probably be observed in some reciprocating wetland systems (Behrends *et al.*, 2000).

Earlier research based on a Kodijärve horizontal subsurface flow (HSSF) constructed wetland (CW) showed that the average efficiency of P removal from sewage in the Kodijärve HSSF system was 83%. From 1997–2002 the removal efficiency tended to decrease. Most of the P was adsorbed in the soil (88.1%), and bacteria were responsible for the removal of 4.4% of P (Mander *et al.*, 2003).

The main aim of this study was to test the EBPR effect in CWs using the reciprocating regime in the cells of an experimental filter system.

Materials and methods

The Kodijärve horizontal flow wetland was constructed in 1996 (Mander *et al.*, 2001). From 1997–2002 total-P from the inlet and outlet was measured (63 samples). In October 2001, 18 samples of soil were taken (two depths: 0–10cm and 20–30 cm) and P immobilised into microbial biomass was measured using the fumigation-extraction technique (Schinner *et al.*, 1996); microbial N was also measured. Microbial C was measured using the substrate-induced respiration method (Schinner *et al.*, 1996).

The normal ratio of C: N: P in bacteria is 50:14:3 (Badon, 2003). Using this knowledge, we calculated relative biomasses on the basis of C, N and P content. The final biomass value was the average of these three values. Then we were able to calculate the percentage of P in bacterial biomass.

An experimental filter system (Figure 1) was constructed on the territory of the Põltsamaa sewage treatment plant (Mander *et al.*, 2001). It was made of 4 cells (2 reciprocating cells where aerobic and anaerobic stadiums vary; 1 vertical and 1 horizontal cell; the size of all of the cells was 2.0 x 0.75 x 0.4 m) in order to compare the differences between different water regimes and to study the effect of a reciprocating system. All of the cells are filled with LECA (80% 4–10mm fraction, 10% 2–4mm and 10–20mm fractions) and contain a sand tube to inoculate the whole system with bacteria. Sand was taken from the Kodijärve HSSF CW. Raw sewage was taken from the inlet to Põltsamaa sewage treatment plant. After passing the filter cells, the water was channelled back to the treatment plant.

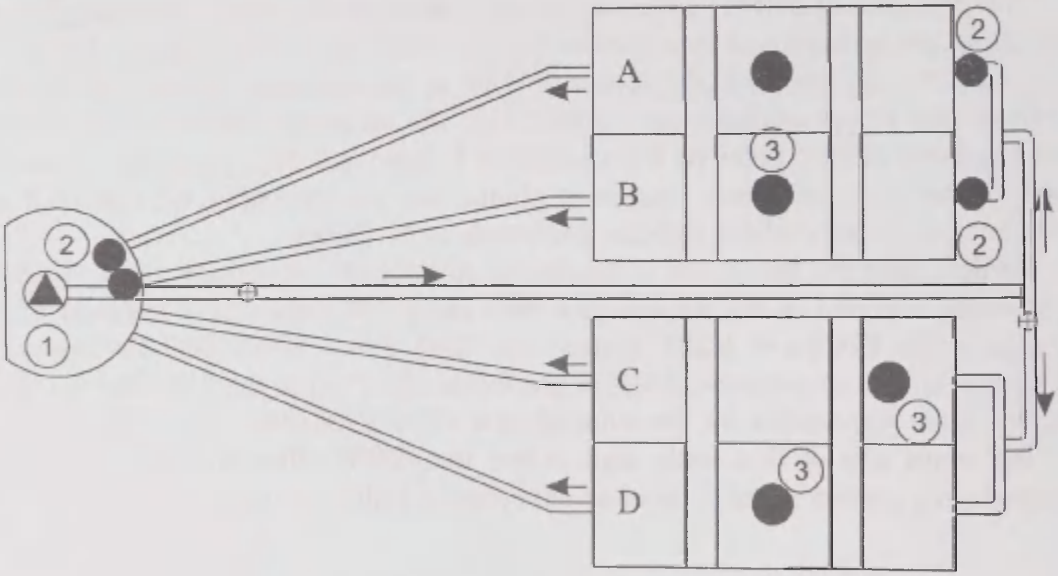

Figure 1. The experimental filter system in Põltsamaa. A and B – reciprocating cells; C – vertical flow cell; D horizontal flow cell; 1 – pump (inlet); 2 – magnetic valves; 3 – sand tubes.

After 2 weeks of work, water samples were taken three times from the inlet and outlet of every cell; in the lab, the content of total-P in water was measured.

The soil samples were taken from the sand tubes before commencement of the experiment and after 2 weeks of work. These samples were analysed together. P immobilised into microbial biomass was measured using the fumigation-extraction technique; microbial C was measured using the substrate-induced respiration method (Schinner *et al.*, 1996).

Results and discussion

Phosphorus immobilisation into bacteria in Kodijärve

We found that the average P concentration in bacteria in the upper layer of the Kodijärve horizontal filter was 2.7 % of dry matter. Studies show that P content in aerobic systems may vary in a wide range, reaching 4.2 % of dry matter (Kunst and Mudrack, 1988, cit. Stall and Sherrard, 1976). Average P content in the deeper layer was 5 %. This can be explained by changes in the water level, which might initiate the EBPR effect.

Total-P removal from sewage water

Based on the Kodijärve water analyses, we found that the average total-P concentration from the inlet was 15 mg l^{-1}; from the outlet 2.5 mg l^{-1}; so the purification was 83%. Average water flow was 1.8 l min^{-1}.

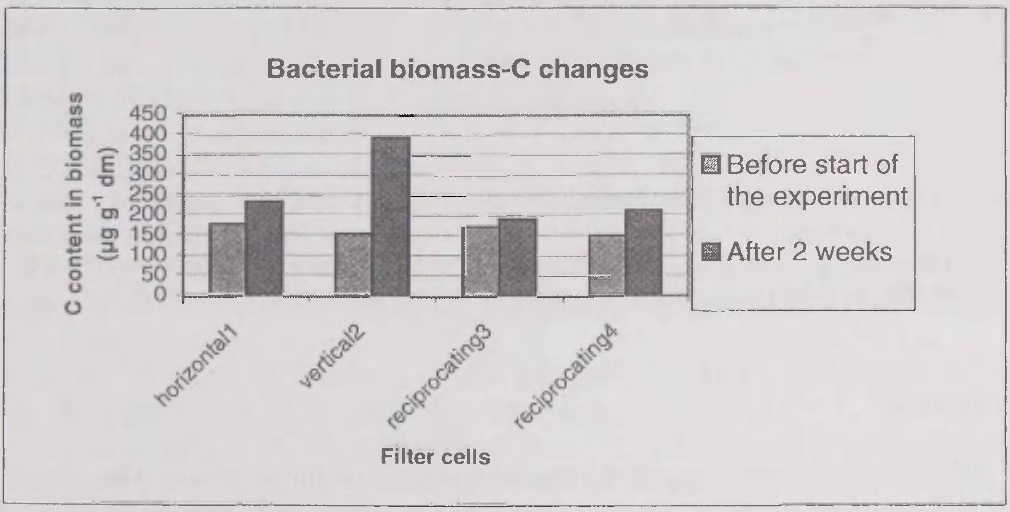

Figure 2. Bacterial biomass changes in the experimental filter system.

Biomass-C and real biomass are very closely related, so we can see that in every cell the biomass has increased. This might be caused by better nutrient supply, differences in environment (pH, temperature) and/or water regime. An especially high increase in vertical flow cell (155.7%) could be caused by better oxygen supply compared with other cells.

Total-P concentration from the inlet in the experimental filter system was 17 mg l^{-1} and the water flow was approximately 1 min^{-1} per cell. The average removal of total-P was 52.9%. It is interesting to compare the Kodijärve horizontal filter and the experimental filter system, because the water amounts are similar. The Kodijärve horizontal filter is 1389 times larger in volume than one cell of the Põltsamaa filter system, but total-P purification capability is 83% as opposed to 52.9%.

Bacterial biomass changes in the experimental filter system

The bacterial biomass-C was measured from soil samples before the commencement of the experiment and after 2 weeks of work. The changes are shown in Figure 2.

We monitored the changes in P content in bacteria, using P:C ratio before the start of the experiment and after 2 weeks of work. The results are shown in Figure 3.

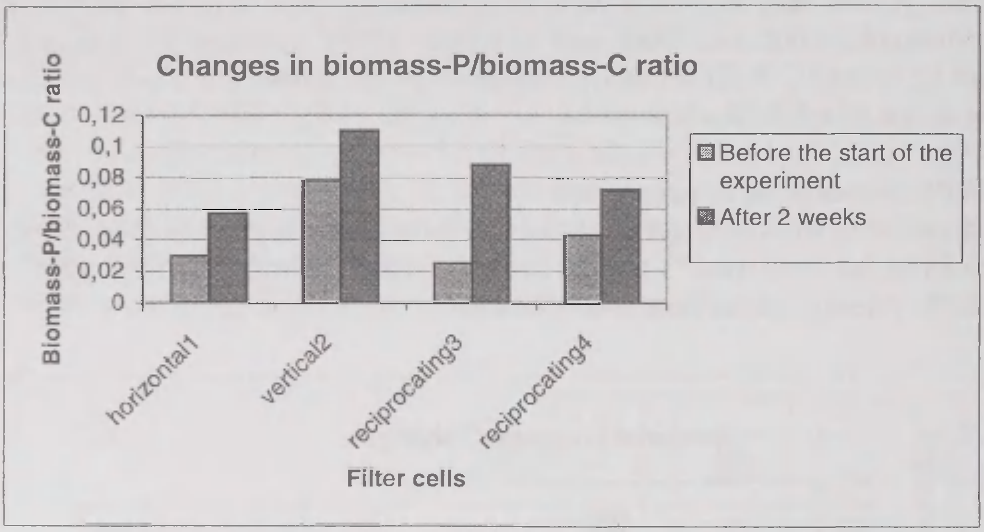

Figure 3. Changes in biomass-P/biomass-C ratio in the experimental filter system.

As can be seen, in every cell the P content in bacteria increased. The greatest increase was in the third reciprocating cell (248.1%), although the increase in biomass was smallest (only 10.5%). We can deduce that the third reciprocating cell offers the best conditions for P immobilisation.

Conclusions

The EBPR is an interesting alternative to several chemical means. The results of the Kodijärve soil analyses suggested that this process might happen if there were changes in the oxygen (water) regime. It is interesting to note that the purification efficiencies in Kodijärve and Põltsamaa are quite similar, despite there being great differences in scale.

The experimental filter system in Põltsamaa showed that microbial biomass growth and phosphorus immobilisation are affected by different water regimes, but this 2-week work cycle is probably too short to draw final conclusions.

It will be necessary to perform further studies of these processes and to improve the filter system so that it may be used as a model for real systems.

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Chemical and physical composition of sediments and filter-media from constructed wetlands related to re-use and waste legislation

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Summary

Twenty samples of sediments and filter-media from 17 different constructed wetlands were sampled during the summer 2001. The physical and chemical composition of the CW-materials varied considerably. In general, the level of organic material, nutrients and contaminants were much lower than in sewage sludge and various composts. The composition of CW-materials was similar to nutrient rich soils.

The contaminant limit values given in EC and national legislations for sewage sludge and composts do not seem to restrict the re-use possibilities of CW-materials.

Introduction

Depending on the wastewater source and type of filter-material and sediments, materials from constructed wetlands (CWs) will contain various amounts of nutrients, suspended solids and inorganic and organic contaminants. The content of phosphorous and nitrogen, as well as other macro- and micronutrients and organic matter, make materials and sediments promising as fertilisers and soil improvers. The content of contaminants in some of these materials may, however, reduce the recycling options. Great variation in type of wastewater, filter-materials, processes and design of CW, makes the CW-materials an inhomogeneous group of materials.

In order to make predictions on what to do with dredged sediments or excavated filter media, thorough characterisation of chemical and physical properties of the material and investigations on biological effects of the materials in soils have to be carried out. In this work the chemical and physical characteristics of 21 CW-materials from 17 different wetlands are presented and the legislation connected to various re-use options and disposal alternatives are discussed.

Methods

Sampling of sediments and filter materials

A total of 20 samples from 17 different wetlands in Norway, Finland, Poland and Estonia were collected during the period June-September 2001. It was emphasised to collect samples from the inlet of the CWs to ensure that the sediments or filter-materials were as saturated as possible with respect to nutrients and inorganic and organic contaminants. A composite sample (5–10 kg dm) was made of 15–20 subsamples taken from the inlet area of the wetland.

Chemical and physical analysis

Prior to chemical and physical analysis the CW-materials were dried at 40°C. The total amount of Cd, Pb, Cu, Ni, Zn, Cr, As, P, S, Na, Ca, Mg, K, Mn, Mo, Al, Fe were determined after Aqua Regia-extraction.

Cd, Pb, and As were quantified using graphite furnace-AAS (SIMAA 6000), while ICAP-AES (Thermo Jarrell Ash ICAP 6) was used for the other elements. Mercury was determined after extraction with concentrated HNO_3 and quantified using cold vapour-AAS (1100B AAS). The ammonium lactate (AL) soluble elements were determined after extraction with 0,1M ammoniumlactate + 0,4M acetic acid. The content of P, Na, K, Ca and Mg were determined using ICAP-AES (Thermo Jarrell Ash ICAP 6). Total organic carbon (TOC) and total nitrogen (TotN) were determined using a CHN Elemental Analyzer. NH_4-N and $NO_{3,2}$-N were determined after extraction with 2M KCl, filtration and spectrophotometric determination (TRAACS 2000). Oxalate extractable Fe and Al was determined after extraction with 0,1M oxalate buffer and Fe and Al were quantified using ICAP-AES (Thermo Jarrell Ash ICAP 6). The content of polycyclic aromatic hydrocarbons (PAH), phatalic esters and nonylephenols (NP) and nonylephenole-toxilates (NPEO) were determined after extraction with dichloromethane using GC/MS for quantification. Linear alkylbenzenesulphonates (LAS) were determined after extraction with alkaline methanol solution using LC/MS-SIM for quantification.

Table 1. Overview of wetlands sampled for the investigation of re-use options of CW-materials.

Wetland	Country	Year of construction	Type of media	Classification*	Waste source	Area (m^2)	Flow ($m^3\ d^{-1}$)
Kompsa	Finland	1987	Peat	OGF	Peat mining	24000	873
Lakeus	Finland	1996	Sediment	FSW1	Municipal	44000	3976
Hovi	Finland	1998	Sediment	FSW1	Agriculture	6000	78
Tveter	Norway	1992	Leca (0–4mm)	GWS	Municipal	96	1,0
Spillhaug	Norway	1998	Gravel/sand	FSW2	Landfill	1000	150
Skjønnhaug	Norway	1999	Sediment	FSW1	Municipal	4000	150
Ski	Norway	1999	Shale-sand	GWS	Municipal	0,9	0,075
Bogstad	Norway	1999	Leca (2–4mm)	GWS	Municipal	300	6,0
Kodijärve	Estonia	1996	Gravel/sand	GWS	Municipal	375	8,6
Aarike	Estonia	1995	Gravel/sand	OGF	Municipal	210	5,0
Põltsamaa I	Estonia	1997	Sediment	FSW2	Municipal	12000	1500
Põltsamaa II	Estonia	1997	"	"	"	–	–
Põltsamaa III	Estonia	1997	"	"	"	–	–
Tänassilma	Estonia	1948	Sediment	FSW2	Municipal	92000	6850
Nowa Słupia	Poland	1995	Gravel/sand	GWS	Municipal	6480	325
Mniow	Poland	1993	Sediment	FSW1	Municipal	2600	150
Esval	Norway	1993	Sediment	FSW1	Landfill		
Haugstein I	Norway	1991	Fe-sand/gravel	GWS	Municipal	100	1
Haugstein II	Norway	1991	Leca (2–4mm)	GWS	Municipal	–	–
Vassum	Norway	2000	Sediment	FSW1	Road runoff	50	–

*OGF-Combined overflow and groundwater flow; GWS-Groundwater flow in porous media; FSW1-Free water surface system; FSW2- Free water surface system with vegetation

Results

The sampled CW-materials show great variation in chemical and physical properties. There is e.g. a distinct difference in texture between sediments from the FSW-systems (eg. Hovi and Põltsamaa), which are dominated by silt and clay, and the filter-materials from GWS-wetland (eg. Tveter and Kodijärve), which contain more sandy materials. The content of TOC in the CW-materials (range 0.1–14%) are comparable with the range generally found in soils and much lower than the level found in organic fertilisers like sewage sludge and composts (TOC about 20%). The levels of phosphorous and nitrogen in the CW-material are only 5–10% of what is generally found in sewage sludge and composts. The mean content of total (TotP) and ammoniumlactate (AL) extractable phosphorous (P-AL) are, however, somewhat higher than is found in most agricultural soils. The fraction of P-AL to TotP varies considerably between CW-materials (Table 2).

The content of heavy metals is highest at Tänassilma (established in 1948, treating municipal runoff). The material from this wetland has relatively high concentrations of Cr, Hg and Pb. Also the sediment samples from Vassum (treating road and tunnel runoff), Skjønnhaug and Esval (treating landfill runoff) have above average levels of some heavy metals. The levels of heavy metals are, however, generally lower than those found in e.g. sewage sludge. The mean concentration of various heavy metals is somewhat higher in sediments (FSW-systems) than in filter-materials (GSW-systems).

The content of organic contaminants is also low in the CW-materials and concentrations above what may be considered as natural background values have been found only in samples from Põltsamaa, Tänassilma and Vassum (Table 2). LAS, which is the major surfactant in cleaning and cleansing products, was not detected in any of the CW-materials. The occurrence of pathogens in the sampled CW-materials has not been determined.

Legislation

For the re-use of CW-materials there are not specific useable detailed EC or national legislations. On one side the CW-materials can be classified as waste that should be disposed in a landfill. In these cases the EC landfill directive (directive 99/31/EC) or other national legislations subjected to waste management, waste characterisation and pollution control, will apply. Article 3 in the landfill directive excludes sludges resulting from dredging operations, which will be added to soils for the purpose of fertilisation or improvement. The low concentrations of contaminants that have been found in the CW-materials do not entail that these materials should not be considered as waste. In the management of used CW-materials legislations made for sewage sludge and composts both in the EC (sewage sludge directive 86/278/EC or coming directive for biological treatment of biodegradable waste (working document DG ENV.E.3) or nationally, seems the most appropriate to apply. To make permanent conclusions concerning the re-use of CW-materials, more assessments of risks, the premises of the directives and national legislation as well as the usability of the present legislation are needed.

Table 2. Chemical and physical characteristics of saturated filter-materials and sediments from different constructed wetlands. Unit: mg kg^{-1} dm (samples dried at 40°C).

Wetland	Sand	Silt	Clay	pH	TOC	TotN	NH_4-N	NO_3-N	TotP	P-AL	Cd	Cr	Cu	Hg	*Ni*	Pb	Zn	Sum Phtalic ester#	Sum *NP*+ NPEO	Sum PAH¤
Kompsa				5.3	14	6800	1.7	26.9	466	32.1	0.07	34.5	8.3	0.029	16.3	5.1	41.3	0.17	<0.2	0.05
Lakeus	90.5	7.0	2.4	6.5	1.1	3200	38.4	196	6740	733	0.04	19.3	43	0.039	11.7	7.6	43.1	0.53	0.98	0.02
Hovi	0.6	24.8	74.6	6.7	1.4	1900	15.5	24.9	630	132	0.14	83.8	37.8	0.058	40.0	11.3	132	0.26	<0.2	0.05
Tveter	91.4	6.7	1.9	8.8	0.2	<500	<0.6	42.9	497	229	0.05	39.9	19.3	<0.015	28.1	6.4	20.1	0.22	<0.2	0.01
Spillhaug	72.8	9.9	17.4	6.9	0.4	<500	0.43	3.2	407	11.5	0.045	20.2	13.7	<0.015	12.6	6.8	37.4	0.36	<0.2	0.02
Skjønnhaug	10.2	62.5	27.3	7.2	0.5	800	1.8	27.6	842	57.5	0.16	76.3	29.4	0.034	54.8	12.1	71.1			
Ski	100			8.3	0.9	<500	41.4	1.0	557	220	0.16	4.1	3.5	<0.015	1.5	<4.0	5.0			
Bogstad	75.0	19.8	5.2	9.5	0.6	<500	<0.6	14.3	498	120	0.086	17.7	19.3	0.029	14.4	6.4	46.3			
Kodijärve	88.7	9.0	2.4	8.2	<0.1	<500	0.33	11.5	329	68	0.053	2.9	3.6	<0.015	1.7	<4.0	13.8	0.05	<0.2	0.01
Aarike	91.3	6.4	2.3	7.9	0.2	<500	1.5	16.8	319	60.5	0.039	4.5	4.5	<0.015	3.1	<4.0	19.2			
Põltsamaa I	53	36.3	10.7	7.3	2.4	2900	5.9	171	1270	353	0.16	14.7	23.5	0.1	8.2	11.9	123	2.36	<0.2	0.41
Põltsamaa II	30.2	56.5	13.2	7.2	6.8	7500	9.4	143	1820	410	0.32	23.3	47.9	0.2	12.7	16.1	237			
Põltsamaa III	49.4	41.4	9.2	7.4	2.9	3400	1.4	31.1	1440	363	0.17	14.1	10.7	0.058	6.8	10.4	56.2			
Tänassilma	78.7	14.6	6.6	7.8	3.6	3400	0.83	31.1	1320	276	0.24	328	38.3	1.3	8.8	67.5	212	1.04	<0.2	5.02
Nowa Slupia	78.0	14.2	7.9	7.3		1700	0.73	2.7	784	153	0.66	50.1	25.9	0.15	11.7	36.6	276			
Mniow	66.2	27.2	6.6	4.9	0.2	<500	55.7	10.9	220	79.5	0.053	9	3.8	<0.015	5.0	<4.0	22.7	0.23	<0.2	0.01
Esval	7.7	61.8	30.7	7.5	3.3	3400	4.1	71.4	795	32.6	0.26	35.9	25.4	0.044	31.5	12.8	160			
Haugstein I Sand	90.2	6.7	3.1	5.9	0.5	<500	0.38	27.1	1200	306	0.058	23.7	15.1	0.019	16.1	6.0	44.9			
Haugstein II Leca	88.6	8.6	2.8	8.4	0.1	<500	<0.6	4.3	294	58.5	0.031	19.5	19.9	<0.015	21.4	<4.0	23.6			
Vassum	15.9	64.2	19.8	7.8	7.1	1100	0.66	<0.4	875	110	0.43	49.8	73.1	0.044	37.4	30	627	82	8.2	5.26
Mean	*62.0*	*26.5*	*13.6*	*7.3*	*2.6*	*1918*	*10.6*	*45.1*	*1065*	*190*	*0.16*	*43.6*	*23.3*	*0.108*	*17.2*	*12.9*	*111*	*8.72*	*4.6*	*1.09*
Minimum	*0.6*	*6.4*	*1.9*	*4.9*	*0.1*	*<500*	*0.33*	*1.0*	*220*	*11.5*	*0.031*	*2.90*	*3.5*	*<0.015*	*1.5*	*<4*	*5.00*	*0.054*	*0.98*	*0.01*
Maximum	*100*	*64.2*	*74.6*	*9.5*	*14*	*7500*	*55.7*	*196*	*6740*	*733*	*0.66*	*328*	*73.1*	*1.3*	*54.8*	*67.5*	*627*	*82*	*8.2*	*5.26*

Sum Di-(2ethylhexyle)phtalate (DEHP), Butylbenzyl-phtalate (BBP) and Di-n-octylphtalate (DOP). ¤ Sum of Naphtalene, Acenaphtylene, Acenaftene, Phenantrene, Fluorene, Antracene, Fluorantene, Phyrene, Benzo(a)antracen, Benzo(b+k)fluoranthene, Benzo(a)pyrene, Benzo(ghi)perylen, Indeno(1,2,3-c,d)pyrene

Conclusions

- The chemical and physical composition of filter-samples and sediments from CWs are dependent on the intrinsic properties of the materials, main process in the wetland (e.g. sedimentation or infiltration), wastewater source and loading.
- The contaminant level in the CW-materials is much lower than those generally found in sewage sludge and even in composts. When it comes to the "total" concentrations of contaminants in the CW-materials, the application of EC or national (e.g. Finland, Norway and Poland) sewage sludge or compost legislations will in most cases not restrict the re-use possibilities.
- The "total" content of contaminants and nutrients is not sufficient for exploring the beneficial use or detrimental effects of CW-materials. Leaching properties, as well as effects on soil bacteria, plants and soil-living organisms should be performed.

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Development of small municipal wastewater treatment plants in rural areas of Podlaskie province in Poland

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From 1998–2002, research was performed on the development of wastewater treatment plants in eastern Poland. The project entitled *Water, Wastewater and Waste in Small Localities of Podlaskie Province* was financed by the Podlaskie Province Fund for Environmental Protection.

Of nearly one hundred wastewater treatment plants under investigation, the majority used sludge-activated systems (flow or sequence systems), and only a few operated with bed filters as a biological treatment. The sequencing batch reactor (SBR) system proved to be the most popular in the modernization and building of completely new plants in the 1990s. Some improvements in the classic SBR plant were observed – there were many plants with constant inlet and periodic outlet. These were usually called semi-periodic systems. The observed problems with the high efficiency operation of S.B.R. systems were strictly connected with the high amount and wide range of nitrogen and phosphorus in raw wastes. This problem is typical in small towns and villages located in rural areas, where public sewer systems are underdeveloped. Wastewater utilization from individual septic tanks seems to be a very important problem. The characteristic of wastewater from such tanks is completely unpredictable and differs from wastewater from the public sewer system.

Most of the systems designed and built between the 1960s and 1980s utilized sludge-activated chambers with surface aeration. They operated in conditions where oxygen was present, so there were no possibilities for high biological efficiency in the removal of nitrogen and phosphorus. New regulations concerning wastewater treatment have forced great improvements. Existing aerobic sludge activated systems were rebuilt into systems with anoxic and anaerobic areas. We can assume that the problems of treating large quantities of wastewater have been solved. Nowadays we have to solve problems with proper treatment of wastewater from small communities. We still have some rotary biological contactors – which are still working, but not very well. The implementation of the British experience in treating wastewater with rotary biological contactors may yield better results, with constructed wetlands as the third stage of biological treatment. It has been proven that hybrid systems of constructed wetlands can improve outlet quality.

This solution permits us to avoid building sludge-activated systems for small amounts of municipal waste.

VF constructed wetlands as a second stage of treatment can solve the problem of wastewater treatment in areas where it is not possible to build a sewerage system. Such a solution seems to be the most favourable from the ecological and economic point of view.

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Wastewater Gardens® in the Carpathian Mountains – the promotion of decentralized wastewater treatment systems

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Untreated or insufficiently cleaned wastewater is a danger for water resources and a hazard to human health. Streams, rivers, lakes, coastal waters and groundwater are polluted. This affects human health by worsening the quality of drinking water and recreation sites and exposing people to pathogens that cause disease. Environmental problems from polluted water can cause damage for whole ecosystems as well as for individual plant and animal species.

In Poland, wastewater treatment is a very serious problem. According to statistical data, only 53.1% of the population is serviced by wastewater treatment plants.

On one hand, water is supplied to 91.5% of the population in cities and 30% of the population in rural areas. On the other hand, only 83% and 9.9% of these populations respectively are serviced by wastewater treatment plants.

The worst situation is in the rural, sparsely built-up areas. Small, isolated communities have difficulties in building and maintaining highly technical wastewater treatment systems. Very often, traditional treatment plants are not maintained because of financial problems, or the treatment plants are not operated professionally. As a consequence, wastewater remains untreated or is cleaned insufficiently. These problems become especially important in areas with important natural resources such as conservation areas, national parks and Biosphere Reserves. One example of such an area lies in the southeastern, mountainous part of Poland – Bieszczady, which is part of the Polish Carpathian Mountains. The Bieszczady Mts. are famous for their picturesque landscapes with extraordinary alpine meadows ("poloniny"), low population density (only 5–25 per sq. km), unique flora and fauna with many threatened species and natural and primeval beech forests inhabited by large predators like brown bears, wolves and lynxes. There are also herbivorous animals such as the legendary Carpathian deer, European bison, wolf, lynx and beaver. The Bieszczady Mts. are localized close to Poland's border with Slovakia and Ukraine. Bieszczady National Park, set up in 1973, protects the most valuable parts of the mountains. It is the third largest among Polish national parks, with a total area of 29,000 ha. A significant part of the Polish Bieszczady Mts. create "The East Carpathians" Biosphere Reserve – the only tri-lateral Biosphere Reserve in the World.

The wilderness of the area is uniform in its "underdevelopment" from the socio-economic point of view. Concerning water management, the main problems in this region are as follows:

- poor quality of water resources: flowing water, open reservoirs, groundwater and drinking water resources,
- lack of well maintained and functioning wastewater treatment plants,
- leaking or overflowing septic tanks,
- lack of money for investments in water supply and wastewater treatment systems,
- sparsely inhabited sites in difficult mountain areas,
- threat to the environment and unique nature posed by uncontrolled wastewater discharge,
- limitation to the sustainable development of the region through environmentally-friendly tourism and agriculture,
- great fluctuations in the amount and composition of wastewater,
- low ecological awareness in local communities.

The Carpathian Heritage Society has been realizing the program, the main aim of which is to increase the quality of water resources in the Polish Carpathians by constructing individual wastewater treatment systems and conducting an educational and promotional campaign at the same time. The Wastewater Gardens® technology is being promoted in the programme.

The Wastewater Gardens® technology was developed by Dr. Mark Nelson, working in collaboration with the Planetary Coral Reef Foundation (PCRF) (U.S.) and eminent systems ecologist Prof. H. T. Odum of the Center for Wetlands at the University of Florida. This innovative approach to wastewater treatment using man-made wetlands, employing high biodiversity and subsurface flow has been extensively tested and successfully applied in the United States and Europe, as well as in a number of tropical countries over the past decade. The Wastewater Garden is an utterly ecological design and raises constructed (artificial) wetlands to a complete system.

The Wastewater Gardens® system works by the gravity flow of wastewater into a septic tank and then into a specially engineered subsurface, horizontal flow, water-tight wetland cell filled with gravel and planted with a wide variety of plant species. Discharge of the treated wastewater to a final leachfield or water pond constitutes the final stage of treatment.

Features distinguishing Wastewater Garden from commonly used constructed wetland are: diverse plant composition adapted to local conditions and users' preferences, the special construction and filling of the cell, the special construction of an inlet and outlet pipe system.

Apart from its high aesthetic value, plant diversity also benefits wastewater treatment.

In Poland the Wastewater Gardens® technology is implemented by the Carpathian Heritage Society and Natural Systems Inc. under the name Gardens for Clear Water (Ogrody dla Czystej Wody®). In cooperation with the Botanical Garden of the Jagiellonian University in Krakow, Poland, the eco-technology was adapted to the climatic conditions of Eastern Europe. Among other things, a list of suitable plant species was created, that can be used in Eastern European and high altitude Wastewater Gardens®. The list includes 66 local decorative species, and 36 exotic species. A Polish trademark for Wastewater Gardens®, which illustrates how the system works, has been designed. Within the framework of the project, a website, http://www.carpathians.pl/gardens/index.html, was established in English and Polish, including extensive information about the technology and its possible use by investors.

The First Wastewater Garden® in Poland, which was built in June 2002, treats wastewater from the Research Station of the Jagiellonian University in Krempna in the Magurski National Park in Southern Poland. A beautiful Wastewater Garden featuring a variety of flowering species dominates the 24 m^2 of surface area in an orchard adjoining the septic tanks that perform preliminary treatment.

The second Wastewater Garden system was constructed for the community of Lutowiska in June 2003. It is a pilot, demonstration project. The system treats part of the total amount of sewage produced in 300 households located in the commune. They are treated in an existing conventional biological wastewater treatment plant, which is not fully effective. The wastewater garden has been located in the vicinity of this treatment plant.

It has been designed to treat about 0.7 m^3 of sewage, which is the average amount produced by a family of 4–5 people.

Both systems have been built in the framework of the project “Wastewater Gardens in the Carpathian Mountains” run by the Carpathian Heritage Society and funded by the Sendzimir Foundation, Austria, and the World Wide Fund for Nature International.

In addition to practical and tangible activities such as building the wastewater treatments system, a project seeking to increase knowledge about constructed wetlands and build confidence in such wastewater treatment solutions is being realized. This project is run in cooperation with the Institute of Environmental Sciences of the Jagiellonian University in Krakow, which has been granted the status of European Centre of Excellence (project IBAES Nr EVK2-CT-2002-80009). The project is one of the eight work packages, and is called “Promotion of constructed wetlands”. The main objectives of the project are:

a) To raise awareness of constructed wetlands as an alternative to industrial wastewater treatment among decision-makers and the general public;

b) To share knowledge and experiences in constructed wetlands between scientists from Poland and other CC and EU countries;
c) To utilise scientific knowledge on constructed wetlands in field projects in rural areas of southern Poland.

Based on the feedback information that has been received so far, it can be stated that public interest concerning the application of constructed wetlands is increasing as a result of the accomplished projects' tasks.

Removal of nitrogen and phosphorus in small municipal wastewater treatment plants using a sludge-activated system and constructed wetlands in rural areas of Poland

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In the 1990s a rapid development of wastewater and sewage treatment systems took place in Poland. These were designed for intensive biological and chemical treatment for the removal of carbon, nitrogen and phosphorus compounds. The research, which was organized in the Water, Wastewater and Sewage Department of the Technical University of Bialystok, focused on the evaluation of small municipal wastewater treatment plants operating in rural areas in Podlaskie province. The daily flow capacity of the analysed systems varied from 50 to 1000 m^3. The research was strictly connected with the project *Research in the Circulation of Biogenic Compounds in Municipal Waste Water Treatment Plants*" financed by the State Committee for Scientific Research (KBN). The goal of the project was to determine the nitrogen and phosphorus circulation in municipal wastewater treatment plants (WWTP). The possibility of controlling and balancing these elements in wastewater and sewage treatment was investigated. Other physicochemical parameters of wastewater and sludge were examined.

A wide range in the level of biogenic compounds was observed in raw wastewater. The quantity of nitrogen varied from 22 to 180 g N m^{-3}, while phosphorus varied from 8 to 60 g P m^{-3}. These results were completed during the period from 1998 to 2002 – when fifteen small municipal WWTPs located in the rural area of Podlaskie province were under investigation.

As measured by the control equipment, the quantity of nitrogen in the outlet varied from 8 to 50 g N m^{-3}, while phosphorus fluctuated from 1.5 to 12 g P m^{-3}. The range of nitrogen in sewage sludge after treatment was 1 to 64 g N kg^{-1} of dry mass, while phosphorus was 0.4–26 g P kg^{-1} of dry mass.

In a few systems, problems were observed with meeting Polish standards of biogenic compounds in outlet, especially during winter.

In our opinion we can provide a better quality of outlet from biological treatment, by using well-known constructed wetland (CW) systems as the third step of WWTP. Taking into account the nearly 100 wastewater plants operating in Podlaskie province, we could not find any positive examples of the full-scale application of CW. Some pilot facilities were constructed near schools and recreational centres, but the results are not yet known.

Wetland systems can be an alternative to traditional sludge-activated or bed filter systems. It has been proven that constructed wetlands have great potential for removing nitrogen and phosphorus – especially hybrid systems. One example of such systems is a wetland system located in Sobiechy. After mechanical treatment, the sewage is pumped into a biological treatment unit consisting of horizontal flow (HF) CW and vertical flow (VF) CW beds. The sewage outflow from the VFCW bed is recirculated. Due to improper operation, however, the average concentration of total nitrogen in the effluent is equal to 32.2 mg l^{-1} and exceeds the permissible level (30 mg l^{-1}).

The agricultural non-point source pollution model as a tool for planning the urban landscape with respect to catchment nutrient export

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Eutrophication of sea and lake ecosystems is basically being caused by high discharges of nutrients from terrestrial basins. With the spatial morphology of urban emission sources most likely affecting the landscape capacity of nutrient retention, a plausible question is whether there exists a generally optimal arrangement of urban infrastructure. In order to seek an answer to this question, the spatially distributed Agricultural Non-point Source Pollution Model (AGNPS) is applied to the densely monitored Morsa catchment of South-Eastern Norway. Under realistic conditions, the catchment distribution of nutrient retention is simulated and compared with the actual distribution of emission sources. By applying a cost/benefit optimisation algorithm to structural urban planning, the optimal location of emission sources is determined for future exploitation.

Applying a distributed model to a real world watershed requires large quantities of data. The Morsa catchment was chosen because of the extensive GIS (geographical information system) databases that have been accumulated in previous research projects. They contain highly resolved data on soil types, topography and land-use, together with time series on (*i.e.*) hydrology, meteorology, and water quality. The data generally holds high enough quality to make (parts of) the Morsa catchment an ideal site for the study performed.

AGNPS is a model that simulates nutrient, pesticide and sediment transport through watersheds. It is developed in an extensive collaboration between the US Agricultural Research Service and the US Natural Resources Conservation Service, and includes state-of-the-art technology as well as the features necessary for continuous watershed simulation on a daily basis.

Input requirements for the AGNPS model are:

GIS data	Meteorological data
Digital elevation model (DEM)	Daily precipitation
Soils	Temperature (minimum and maximum)
Landuse	Dew point temperature
	Sky cover
	Wind speed (for certain applications)

The AGNPS model is adapted to the actual watershed in a number of steps, whereof many utilise separate programmes written in FORTRAN or similar programming languages. The programs are controlled via a GIS (ArcView) interface supplied with the AGNPS model, but may also be run in manual mode. Each program creates an input file for the AGNPS Input Editor, where parameters describing the spatial characteristics of input data are calculated, edited, and saved for the final simulation. The model is capable of handling approximately 400 parameters, although the number in actual use varies with the application.

The AGNPS may be used to map the ratio with which individual landscape elements contribute to the total catchment emission of nutrients. It enables for landscape planners to identify areas that allow exploitation under simultaneous optimisation of the landscape capacity to retain urban pollution. By manipulating model parameters, different scenarios may be explored and compared. As an example, the optimal morphology of landscape wetlands may be explored with respect to the total catchment export of nitrogen. Although generally difficult to provide with sufficient amounts of data, the favourable characteristics of the AGNPS model makes it an ideal tool for the landscape planner to utilize in the minimisation of urban environmental impact.

The experience of wastewater treatment using constructed wetlands with horizontal subsurface flow in Lithuania

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Abstract

Natural wastewater treatment methods have become increasingly popular in Europe. One such method is constructed wetlands (CW). In Lithuania, CW of horizontal subsurface flow are more common. This wastewater treatment method is particularly suitable for the treatment of wastewater in small rural settlements. Currently, a certain proportion of former aeration wastewater treatment facilities are being reconstructed into CW, as their operation becomes economically inefficient due to small amounts of wastewater. In Lithuania CW are used for the treatment of domestic as well as industrial wastewater (*i.e.* wastewater from meat, vegetable processing industries, *etc.*). Constructed wetlands might be used for the treatment of wastewater of different chemical composition. The average wastewater treatment efficiency according to BOD_7 and suspended solids is 75–85% and 55–80% respectively. The efficiency of nutrient removal is variable, as it depends much on the extent of wastewater pollution and the physico-chemical characteristics of sand. He calculated wastewater treatment efficiency according to total N and total P is 20–50% and 15–60% respectively. In order to reach normative wastewater treatment efficiency according to BOD_7 – 28.8 $mgO_2\ l^{-1}$, (BOD_5 – 25 $mgO_2\ l^{-1}$) determined for small-scale wastewater treatment facilities in Lithuania, the average organic load of filters is suggested to be up to 3.7 g $BOD_7\ m^{-2}\ d^{-1}$.

Introduction

In Lithuania about 1.2 mln of the population live in small townships and villages. In rural areas and small townships the water management situation is poor from technical, economic and planning points of view. In rural areas there are about 680 wastewater treatment facilities, in most of which aeration wastewater treatment technology is used. The operation of such equipment requires the consumption of large amounts of energy. In most cases, typical projects were used. At the moment, the designed output of those projects is 30–50%. Along with changes in water consumption regime and the prices of energetic resources, such systems became economically inefficient, and a large number of them stopped functioning.

Natural wastewater treatment methods are becoming more and more popular in Europe. These methods include biological ponds, constructed wetlands, etc. Natural wastewater treatment methods always require less energy consumption, although they occupy larger areas. In Lithuania the question of area is not a problem, especially in rural areas. Moreover, having estimated their functioning characteristics, such wastewater treatment methods gain an advantage. Experimental wastewater treatment facilities containing CW have been constructed in Lithuania since 1994. The first full-scale constructed wetland, designed by G. Geller and A. Lenz, was built in 1995. It was a reed bed with filters of subsurface horizontal flow. Detailed investigations concerning the efficiency of wastewater treatment were carried out in those facilities. As the study results showed, such wastewater treatment facilities are applicable under Lithuania's climatic conditions.

Subsurface horizontal flow wetlands have the primary benefit that water is not exposed during the treatment process, minimizing energy losses through evaporation and convection. This makes wetlands more suitable for winter applications (Wallace *et al*, 2001). The studies performed in Norway showed (Mæhlum, 1999) that CWs can function smoothly under cold climatic conditions.

At present 20 such wastewater treatment facilities have been constructed in Lithuania. The output of the facilities is 5 to 600 m^3/d. Most of them are former rural wastewater treatment facilities reconstructed as CWs. There are, however, some new wastewater treatment facilities with higher outputs constructed for the treatment of wastewater from meat processing companies, milk collection points, canning factories and other enterprises.

Study objects

Two former aeration wastewater treatment facilities reconstructed into CWs were selected for the investigation. In Pagiriai village the reconstruction work was finished in 1995. The existing well-type reservoirs were reconstructed into septic tanks, and sand-reed filters were arranged. The former aero-canal was changed into two subsurface flow ponds that are used for the pre-treatment of excess water during wet periods. The main characteristics are as follows: designed output – 300 PE (population equivalents), payload volume of septic tanks – 86 m^3, reed bed filters – 4 units, surface area of 1 filter – 360 m^2, area of wastewater treatment facilities – 1.1 ha.

In 1998 the existing septic tank in Plinkaigalis village was supplemented with a sand-reed filter and two ponds. Here wastewater flows from the septic tank into the filter, and then flows via ponds and finally is released into the lake. As the study results show, during rainfall periods the amount of wastewater increases up to tenfold due to jumbled pipelines. This results in diluted wastewater flow into ponds, as the filter is incapable of treating such an amount of wastewater. The main characteristics are as follows: area of wastewater treatment facilities – 0.7 ha, designed output – 20 m^3 d^{-1}, payload volume of a septic-tank – 76 m^3, filter area 25x10 m – 250 m^2, payload volume of first pond – 660 m^3, payload volume of second pond – 450 m^3.

Two wastewater treatment facilities containing CWs for the treatment of industrial wastewater were selected for the analysis of wastewater treatment efficiency. These are domestic wastewater treatment facilities arranged in the "Lifosa" fertiliser plant and meat processing wastewater treatment facilities arranged in the "Nematekas" company. The technological scheme of wastewater treatment facilities in "Lifosa" includes a septic tank and CW. The main characteristics are: year of construction – 2000, designed output – 800 population equivalents (PE), payload volume of septic-tank 500 m^3, reed bed filters – 3 units (constructed 2 filters), filter surface area – F-1 – 1800 m^2, F-2 – 1980 m^2, designed output of 1 filter – 200 m^3 d^{-1}, area of wastewater treatment facilities – 1.1 ha.

The technological scheme of the wastewater treatment facilities in the "Nematekas" meat processing company includes primary physico-chemical wastewater treatment, a sedimentation process in two ponds and wastewater treatment in the CW. The main characteristics: year of construction – 2001, payload volume of two ponds for settling – 4300 m^3, designed output- 100 m^3 d^{-1}, reed- bed filters – 2 units, surface area of 1 filter – 940 m^2.

Results and discussion

All objects under investigation contain filters of similar construction. After primary treatment, wastewater is distributed into filters via distribution wells. In the filters, distribution pipes are arranged in breakstone prisms. Here water is distributed evenly throughout the whole filter. Then wastewater is filtered horizontally through semi-coarse sand with a filtration coefficient of 5–8 m d^{-1}. The filtration distance is 4.5–5.5 m.

The further analysis of wastewater treatment efficiency is performed mainly in constructed wetlands in all objects. Primary wastewater characteristics in each object as well as average wastewater treatment efficiency indices are given in Table 1.

The wastewater pollution of the first three objects is relatively low due to its dilution resulting from jumbled wastewater pipelines. In "Nematekas", wastewater pollution characteristics after primary treatment are similar to those of domestic wastewater. The obtained study data show rather favourable coefficients of wastewater treatment efficiency in terms of BOD and suspended solids. Nutrient removal efficiency is variable, as it depends on the extent of wastewater pollution. P-removal was greatly influenced by the physico-chemical characterristics of sand. In Lithuania no N and P removal levels are determined for small-scale wastewater treatment facilities up to 2000 PE. Wastewater treatment requirements according to BOD_5 – 25 $mgO_2\ l^{-1}$ (BOD_7 – 25x1.15 = 28.8), according to suspended solids – 30 mg l^{-1}. As the obtained study data shows, CWs are equally suitable for the treatment of wastewater with different extents of pollution.

Table 1. Wastewater treatment efficiency in constructed wetlands.

Index	Wastewater treatment facilities	Before treatment	Inflow in reed bed filters	Outflow from reed bed filters	Treatment efficiency in reed bed filters, %
BOD_7, $mgO_2\ l^{-1}$	Pagiriai	110.9±94.6	52.6±36.5	12.0±8.0	77.2
	Plinkaigalis	230.3±108.2	111.5±32.3	23.0±11.5	79.4
	Lifosa	90.0±65.0	51.2±16.2	7.8±3.0	84.8
	Nematekas	825.2±292.9	185.6±166.2	29.2±12.4	84.3
Suspended solids, mg l-1	Pagiriai	103.4±91.8	49.5±38.1	22.6±20.0	54.3
	Plinkaigalis	266.0±122.3	74.0±36.1	17.1±4.7	76.9
	Lifosa	81.2±39.6	30.6±12.2	12.2±4.1	60.1
	Nematekas	597.7±140.9	211.9±110.6	43.0±23.9	79.7
N total, mg l-1	Pagiriai	26.7±11.3	25.9±19.1	13.5±5.7	47.9
	Plinkaigalis	65.3±21.3	51.7±14.7	33.6±4.3	35.0
	Lifosa	11.3±7.8	9.4±3.7	7.4±2.8	21.3
	Nematekas	107.3±29.1	63.7±18.4	37.6±14.3	41.0
P total, mg l-1	Pagiriai	3.3±2.4	2.9±2.5	1.12±0.84	61.4
	Plinkaigalis	10.8±4.2	9.0±2.6	6.7±1.3	25.5
	Lifosa	12.0±5.3	11.2±4.2	9.6±5.5	14.3
	Nematekas	41.1±22.4	15.8±7.7	8.8±5.7	44.3

The correlation analysis of the study data has been performed in order to determine wastewater treatment efficiency according to BOD_7 with respect to filters' organic load. The obtained dependence is shown in Figure 1. Considering the given dependence, the organic load of filters should be 3.7 g $BOD_7\ m^{-2}\ d^{-1}$ in order to reach the normative treatment level according to BOD_7 – 28.8 $mgO_2\ l^{-1}$.

As the results of studies performed in other countries show, with a similar load the treatment efficiency exceeds 80%. According to Vymazal (2001), in 59 wastewater treatment facilities in the Czech Republic the average CW load according to BOD_5 is 3.52 g m^{-2} d^{-1}. Wastewater treatment efficiency was determined to be 88.2% when the pollution of wastewater flowing into the filters is 92 mgO_2 l^{-1} and the pollution of outflowing wastewater is 10.9 mgO_2 l^{-1}. Mander (Mander *et al.*, 1997) has determined that with filters loading 3.8 g BOD_7 m^{-2} d^{-1} the average removal efficiency is 82%.

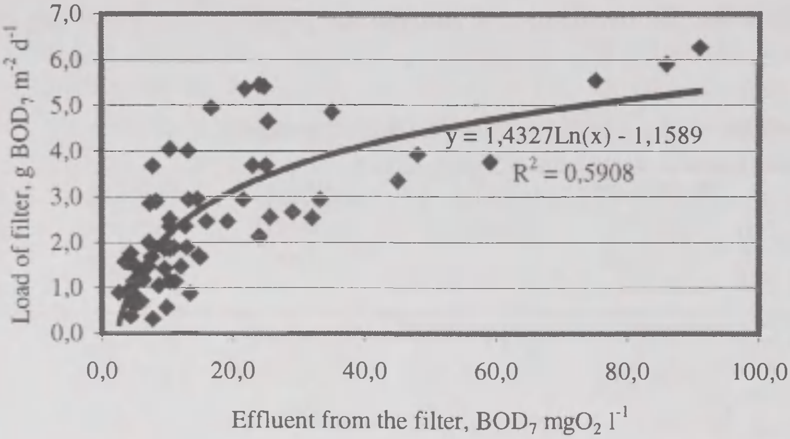

Figure 1. Wastewater treatment level in constructed wetlands depending on the load according to BOD_7.

Principles of the reconstruction of rural wastewater treatment facilities into constructed wetlands

In both rural areas and urban regions of Lithuania, the amount of consumed drinking water is decreasing. This is much influenced by the price of water and the extents of the arrangement of water meters. According to the data selected by the company operating wastewater treatment facilities in Kedainiai district, in 1999 drinking water consumption was 85 l d^{-1} per person, whereas in 2002 it decreased to 51 l d^{-1} per person. According to the data for the year 2002, the amounts of wastewater treated in 10 rural wastewater treatment facilities fluctuates from 4 to 12 m^3 d^{-1} when their designed output is 50–100 m^3/d. Only in two wastewater treatment facilities did the amounts of treated wastewater make up about 40 m^3 d^{-1}.

The amount of wastewater flowing into treatment facilities is variable due to the disorderly nature of the sewerage network. For example, during a spring flood period, wastewater discharges in two villages were observed to be 200 m^3 d^{-1}, although here the amounts of wastewater should not exceed 10 m^3 d^{-1}. Small amounts of wastewater cause inefficient functioning of aeration wastewater treatment facilities. The cost price of wastewater treatment in certain functioning treatment facilities is about 3 Euro m^{-3}, while the inhabitants pay 0.6 Euro m^{-3} for wastewater treatment. As a rule, these are small wastewater treatment facilities with an output of only 10–50 m^3 d^{-1}. Such treatment facilities are capable of functioning only with state support. Wastewater treatment rates are regulated by state resolutions. As the calculation results show, in order to cover electrical energy expenses, the output of the existing wastewater treatment facilities should be about 25–30 m^3 of wastewater a day. Figure 2 presents the relationship between expenditures on electrical energy and the amount of treated wastewater. The following example clearly depicts the inefficiency of small-scale aeration-based wastewater treatment facilities in rural areas.

Two principal schemes of aeration systems are prevalent in rural territories: 1) aero-tank – aerated pond – non-aerated pond; 2) aero-canal – aerated pond – non-aerated pond. Variants without any of the ponds (usually non-aerated one) are also possible. At present most such systems function without aeration equipment (i.e., wastewater flows through the existing equipment only). Such systems are called systems of open biological ponds. Currently wastewater amounts are significantly smaller, although this does not determine the normative wastewater treatment process.

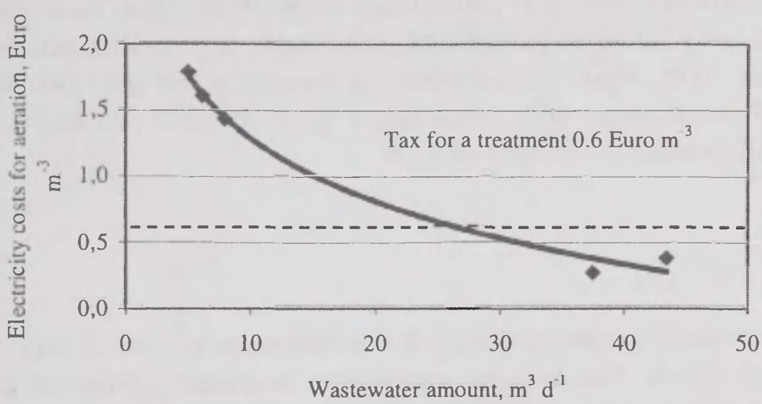

Figure 2. Expenditures on electrical energy consumption in small rural wastewater treatment facilities.

In selecting the technological scheme for the reconstruction of wastewater treatment facilities in rural areas, actual wastewater amounts and their fluctuation dynamics, as well as pollution indices, must be taken into consideration. As the study performed shows, normative calculating results should not be relied upon to too great an extent. Often the actual amount of wastewater fluctuates from 2–20 times, and its pollution index changes in the range of 300 to 10 mgO_2 l^{-1} according to BOD_7. In the Nociunai and Vainikai wastewater treatment facilities the amount of treated wastewater does not reach 10 m^3 d^{-1}. During the snow thaw period in spring, the facilities received 250 and 300 m^3 d^{-1} of wastewater respectively. Similar results were obtained in most of other rural wastewater treatment facilities. This implies that sewerage lines are disordered or turned into mixed lines, i.e. they collect surface water as well.

Under such conditions, it is economically inexpedient to arrange filters for the whole predictable amount of wastewater. The output of constructed wetlands is calculated according to pollution and hydraulic loads. The efficiency according to pollution load is calculated considering the amount of wastewater and its degree of pollution during the dry period of the year. Then calculations are performed to check the hydraulic conductivity of wastewater treatment facilities according to the calculated area. If filters are incapable to let through the wastewater discharges flowing during the wet period, biological ponds are designed for the treatment of the excess amount of wastewater. Such a scheme is called the principal scheme (or base scheme). Having estimated the parameters of existing equipment and wastewater characteristics, different solutions are possible. This particularly depends on the parameters of existing ponds. If the aero-canal is large enough, a certain part of it may be used as a settler, while in the other part a sand-reed filter may be arranged.

Sand-reed filters are susceptible to clogging processes, and therefore efficient settling in the primary link is of particularly great importance. Existing aero-tanks or aero-canals may be used as settlers. Aero-tanks are re-arranged into a three-chamber septic tank, while aero-canals are reconstructed into two open ponds. Reconstructed aero-tanks and aero-canals give 40–50% treatment efficiency according to suspended solids and BOD_7.

Conclusions

Constructed wetlands might be used for the treatment of wastewater of different chemical composition. The average wastewater treatment efficiency according to BOD_7 is 75–85%, and according to suspended solids it is 55–80%. Nutrient removal efficiency is variable, as it depends to a great extent on wastewater pollution level and physico-chemical characteristics of sand. The obtained wastewater treatment efficiency according to total N and total P is 20–50% and 15–60% respectively.

In order to reach the normative wastewater treatment level according to BOD_7 – 28.8 (BOD_5–25) mgO_2 l^{-1}, the average organic load of filters is recommended to be 3.7 g BOD_7 m^{-2} d^{-1}.

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Socio-economic and ecological considerations in natural wastewater treatment and utilization for rural areas, using the example of the Wielichowo district (in western Poland, watershed of Obra River)

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The inhabitants of rural areas in western Poland encounter many problems, such as a low soil fertility, deteriorating ground water conditions (lowering of the level of ground water), production is unprofitable in such conditions, resulting in low incomes, making it impossible for farmers to devote money to investments that would improve this situation. Simultaneously, the problem of wastewater treatment and utilization was neglected in these areas – most of the properties have reservoirs for sewage, which are often leaky. In this situation, sewage goes into the ground, which has a negative influence on the ground water and to a small extent on the surface water.

Because almost all of the localities in the Wielichowo district have water supplies, the quantity of pollution that escapes from leaky reservoirs is significant and has a negative influence on ground water.

At present, Polish districts are obliged to order the wastewater economy as a result of introducing the rural ecological policy. In addition to their huge investment and exploitation costs, traditional methods for wastewater treatment using a gravity sewerage system, have other faults: they lower water retention in the area occupied by the sewerage system and also can lead to the deterioration of sanitary and oxygen conditions in the river, below the point of purified sewage dropping

On the basis of research that has been performed, calculations, local visions and conversations with workers of the Wielichowo District Office and with inhabitants of this district, an analysis of the eventual cost of different variants of the wastewater economy was performed. **The first variant** concerned traditional methods of wastewater economy, meaning construction of one conventional wastewater treatment plant and the gravity sewerage system on the area of the whole district. This solution proved to be the most expensive in terms of both construction and exploitation. Additionally, such a solution brings a number of faults mentioned above, and also provokes protests among inhabitants, who are afraid of large costs (connected with construction of the sewerage system's attachments, and with the high price of every cubic metre of purified sewage). The high cost of sewerage system construction in this variant is caused by the fact that every village is within a distance of a few kilometres, and the number of inhabitants is relatively small. Construction or modernization of conventional wastewater treatment is a single cost that is too great for a rural district.

Construction of a few smaller conventional wastewater treatment plants would not lead to a decrease in cost, because the money that could have been saved on kilometres of sewerage system would be spent on construction of a few separate technological objects. Additionally, because of the social aspects, such a variant for the solution of wastewater management is undesirable because a conventional wastewater treatment plant is not an object that is attractive enough to encourage people to live and stay in its close vicinity.

In **the second variant** an analysis was made of the possibility of the construction of a small natural vegetation-pond sewage treatment plant (the Polish system) for every property in the district. Polish legislation states that in localities with populations under 2000 inhabitants it is permitted to refuse sewerage system construction and build individual wastewater treatments. If the property is larger than 1000 square metres, such treatment is recommended. In the case of western Poland's rural districts, almost every property is larger than 1000 square metres. Construction of such small, natural vegetation-pond sewage treatment plants would be the cheapest conception on the level of realization and one of the cheapest in terms of exploitation. But this solution requires a certain amount of interest and superintendence from each inhabitant: checking the correctness of a pump's operation, mowing the plants on a filter once or twice a year, sporadic cleaning of the pond's surface. From the ecological point of view, however, this solution would be the best for the environment, because natural vegetation-pond sewage treatment plants (the Polish system) assure nitrification and denitrification at the same, and often even at a higher grade than conventional wastewater treatment. Purified wastewater that is stored and still being purified in the pond (Figure 1) raises local water retention and improves biodiversity in biologically poor areas thanks to the introduction of aquatic plants which bring in insect and amphibian species typical of lakes and marshes. These ponds can be restocked with fish, which causes an increase in the attractiveness of wastewater treatment, a way to retrain a decorative element of the garden. It is recommended to use water from the pond to water decorative greenery, which leads to the uptake of the remaining small amount of biogenic substances by aquatic plants, and additionally, the obtaining of effective yields without the need to use fertilizers, which finally results in a double economic profit. The surplus of purified wastewater, which arises in the wintertime, is piped away into the ground through the absorptive ditch or pond, thereby supplying the ground water. This variant of wastewater utilization is already partly being realized in neighbouring districts.

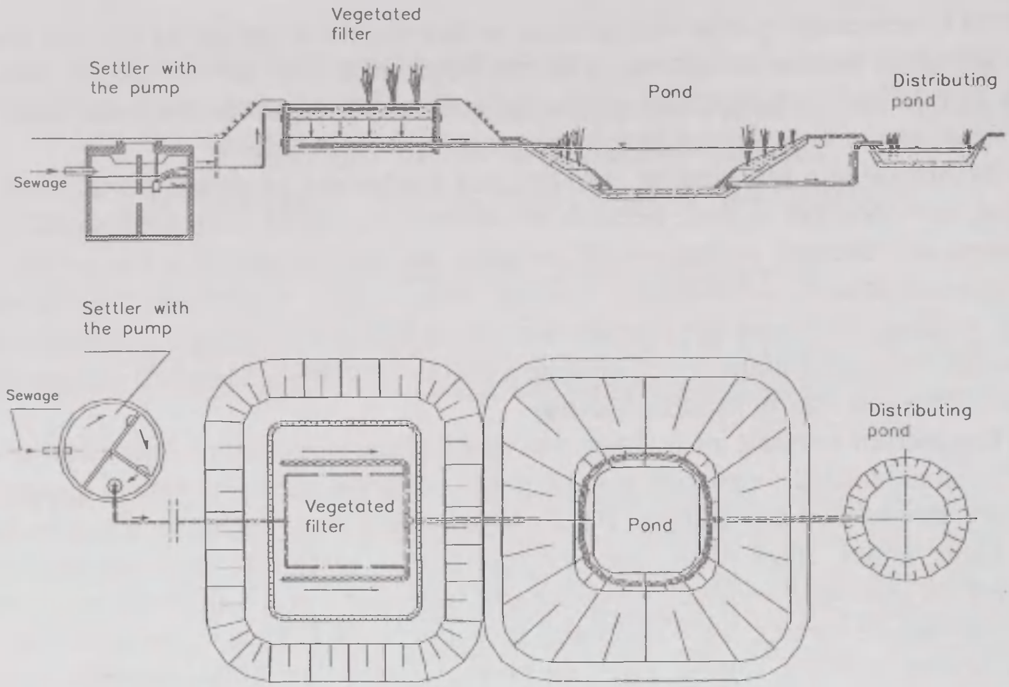

Figure 1. Natural vegetation-pond sewage treatment plant (the Polish system).

The way to avoid the faults mentioned in the first variant and at the same time retain the advantages of the second variant is to construct a group of local natural vegetation-pond sewage treatment plants (the Polish system) in a quantity of one or two for each village (**the third variant**). This project had the best opinions among the inhabitants of this district, because there is no need to connect a few localities into one sewerage system (there only has to be a gravity sewerage system in every village). The construction of such a wastewater treatment is cheaper in terms of realization and exploitation than the construction of conventional treatment plants, the result is an attractive object with plants and a big fishpond. It is easy to find a place for this investment in or near each village. The huge advantage of this proposal that is appreciated by the local authorities is the fact that this solution can be realized gradually, for example: at first in only one village, and later it can be continued in other villages, etc. This allows one to avoid the construction of oversized conventional wastewater treatment plants (at huge cost), which, during the coming years, would not be fully used because of the lack of money required to build a gravity and pressure sewerage system.

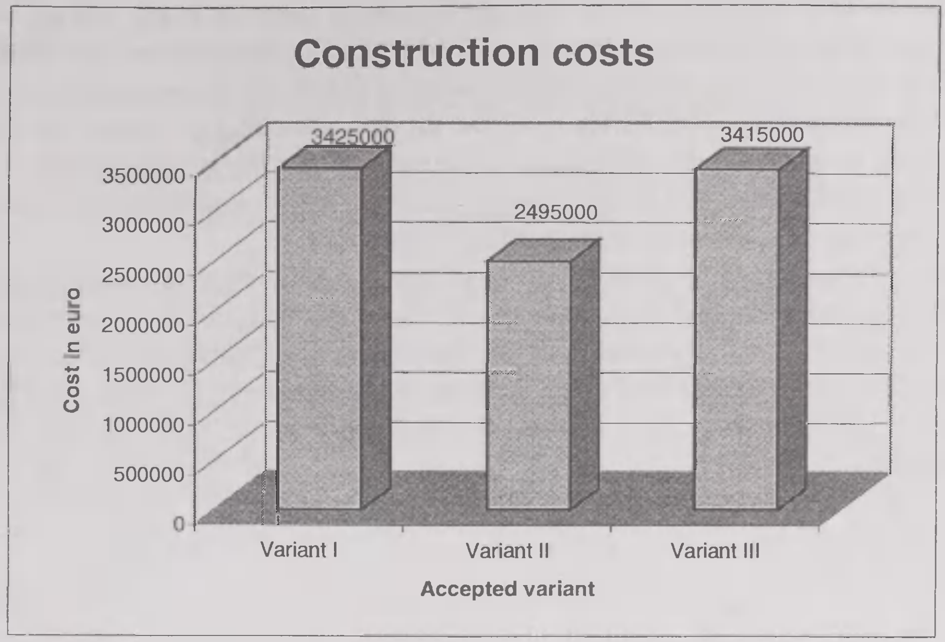

Figure 2. Construction costs of particular variants for wastewater treatment and utilization in Wielichowo district.

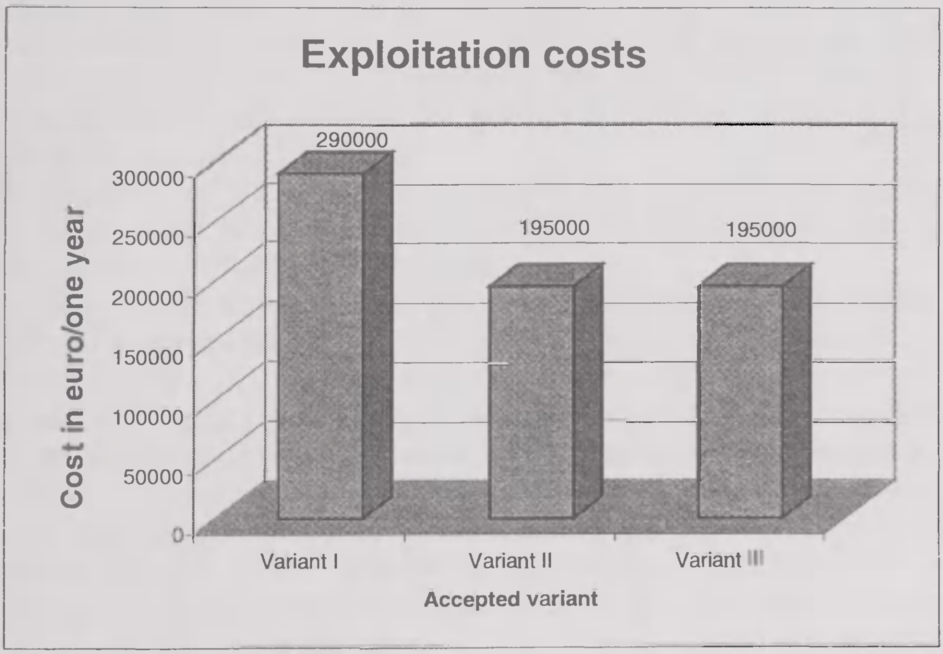

Figure 3. Exploitation costs of particular variants for wastewater treatment and utilization in Wielichowo district.

A local, natural vegetation-pond sewage treatment plant in every village would have increased the ecological awareness of its inhabitants. The possibility of watching the natural processes which occur in nature on the example of such a wastewater treatment model helps to give an understanding of nature, its way of responding to unfavourable changes in its surroundings, the ways in which human can easily change their negative influence thereon, and in a short time observe the many advantages that would arise from this behaviour.

The comparison of the construction and exploitation of particular variants for wastewater treatment and utilization in Wielichowo district that is presented in Figure 2 and Figure 3 proves that the construction of natural vegetation-pond sewage treatment plants (the Polish system) is economically justified, and thus it would be advantageous not only for the environment but also for the human population.

Hydraulic load and sedimentation of suspended solids in a constructed wetland treating secondary wastewater

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Abstract

Constructed wetland of the Lakeus Central Treatment Plant in Kempele, Finland is a Free Water Surface (FWS) wetland (64°53'82"N, 25°27'76"E) situated in the mid-boreal region. The treatment plant combines chemical and biological wastewater treatment methods and treats municipal wastewaters, where the FWS acts as a post-treatment unit. The FWS is constructed on natural reed field with clay embankments and water level controlling dam in the middle. The area of the wetland is 44 000 m^2 and its estimated volume is 9 300 m^3. In recent years the mean discharge has been 3 768 $m^3\ d^{-1}$.

The objectives of this work are to model flow velocities and patterns in different parts of the wetland as well as to clarify the sedimentation of total suspended solids (TSS) with different grain sizes.

The flow velocity and the retention times of wastewater were measured with KBr-tracer experiments in different discharge situations (Mikkonen, 2003). The geometry of the FWS was determined by a field survey of it dimensions (x- and y-coordinates) and depths (z- coordinates). Based on this information and discharge data, a two dimensional depth-averaged finite element numerical hydrodynamic model (RMA2) was used to calculate flow patterns. The output of RMA2 was used as an input for simulations of water break through curves (BTCs) made with a constituent transport model (RMA4) as described by Koskiaho (2003). Grain size distribution of suspended matter was analysed with developed filtering system. Wastewater from input and output of the FWS was first sieved through 0,074 and 0,062 mm mesh and then filtered through 12 and 0,45 μm filters. In some situations it was necessary to use compressed air (up to 10 bar) to get enough water through filters (Hallikainen, 2003).

The wetland system retained TSS effectively (>50%) in winter, spring and autumn. In summer results were less effective, varying from 6 to 27%. TSS concentrations in inflow varied from 1 to 68 mg l^{-1} and in outflow from 1 to 108 mg l^{-1}. Long-term trend from period of 1.1.1997 to 31.8.2002 shoved that the FWS slightly increases its TSS retention capability. Grain size distribution showed that the FWS captures all sizes of TSS. Although purification results in large grains (74–12 μm) were better than in small grains (0.45–12 μm).

FWS as a purification system of TSS as well as model calculations combined with TSS concentration and sedimentation data will be discussed. Also practical questions of design and management of FWSs will be discussed.

Acknowledgements

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The influence of episodic meteorological events on nitrogen, phosphorus and microelements in a floodplain ecosystem

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Introduction

In the boreal zone, weather conditions play an essential role in nutrient flows in rural landscapes. Interannual and seasonal variations in nutrient losses are very high in the boreal zone, where weather variability is high.

Discrete sampling data do not describe nutrient losses in small catchments (Vagstad *et al.*, 1997) with sufficient accuracy. The random discrete sampling strategy normally used for monitoring purposes underestimates P loss in certain cases by more than 50%, because most of the P loss occurs during storm events (Grant *et al.*, 1996). For predictive models it is very important to follow certain critical periods when biological and chemical processes switch from one type to another.

Winter and summer are used because they are the most stable seasons with respect to nutrient loss and changes in processes affecting nutrient release. On the contrary, spring and autumn are associated with generally high nutrient losses due to peak flow and to a large degree also determine the character of physicochemical and biological processes in the summer and winter respectively. The following factors have the highest priority in weather-induced changes in nutrient fluxes: a) duration of frozen surface, b) snowpack peak water, c) precipitation pattern over warm period, d) duration and continuity of certain weather, e) occurrence of night frost events.

Material and methods

Study area

The Sipe River catchment area (8.9 km^2) is situated in South Estonia, 10 km south-east of Tartu. The area studied in detail lies on the upper course of the Sipe River in a small branch of the primeval valley of the Porijõgi River. The bottom of the valley is occupied by a floodplain meadow. Stream discharge consists of base flow originating from groundwater and outflow from the lake adjacent to the study area. During long-lasting droughts it may happen that there is no water discharge from the lake.

The clay substrate underlying the floodplain wetland results in a shallow perched water-table, poorly drained and highly organic soils, and reduced inputs of regional groundwater. The depth of organic sediments in the floodplain area is up to 6 m, although in most of the area these remain between 2 and 4 m in depth. The deepest layers also contain gyttja.

The dominant vegetation patterns in the floodplain meadow are made up of *Carex spp.*, *Filipendula ulmaria*, gramineous plants, *Geranium palustris.*, *Anthriscus sylvestris*, *Urtica dioica* and *Salix spp.*

Meteorological measurements and water sampling

To follow certain critical periods when biological and chemical processes are switched from one type to another or hydrological and meteorological characteristics are rapidly changing, the process-oriented sampling method is used instead of traditional random sampling in the detailed study area in the Sipe River floodplain.

Water sampling wells on the Sipe River floodplain constitute an almost regular grid. The sampling wells are located at a distance of 100–200 m from one another along the river and at 15–90 m across the river. The exact location of the sampling wells was determined on the basis of features such as vegetation pattern, topography, soil properties and hydrological conditions. Three sampling weirs on the river are established at the inflow to the study area, in the middle section of the floodplain and at the outflow from the study area.

Observations made at 3-hour intervals are used to determine short-term changes in physicochemical parameters in surface and groundwater and their response to meteorological phenomena in its different phases. Detailed observations take place according to changing weather conditions, on average once every 1–2 weeks, more frequently in the spring and autumn when weather variability is higher than that of the summer or winter. During detailed observations, water samples are taken for chemical analysis (TIN, total-N, total-P, NO_3^-, NO_2^-, NH_4^+, PO_4^{3-}, SO_4^{2-}, Fe, Ca, Al, K, Mg), and the following water parameters are measured: water level, temperature, O_2, pH, ORP, electric conductivity and salinity.

Meteorological measurements were carried out using a GroWeather automatic weather station located in the middle of the detailed study area in the floodplain meadow. Meteorological parameters were registered at 1-hour intervals.

Results and discussion

Studies in a small Sipe River watershed with low relief show that the water-table level is strongly influenced by seasonal fluctuations in the level of regional groundwater, ephemeral runoff initiated by spring snow melt and episodic precipitation. Special attention must be paid to the seasonal variation of the water -table level, as this factor is highly determinant of general hydraulic flow parameters and thereby nutrient losses to the river. In contrast to other terrestrial ecosystems, the sink of phosphorus in floodplains is largely regulated by geochemical processes (sorption, precipitation), while nitrogen is controlled mainly by biological processes (accumulation of organic matter, immobilisation, denitrification). Any meteorological event that affects the physicochemical properties of shallow groundwater in the floodplain (ORP, pH, dissolved oxygen) will give rise to changes in nutrients fluxes.

Highly fluctuating ORP conditions are favourable to intensive denitrification processes. In anaerobic groundwater conditions decomposition processes result in high concentrations of ammonium (3.5–6.6 mg l^{-1}) as opposed to nitrate (0.005–1.5 mg l^{-1}). Under aerobic conditions, usually after episodic precipitation events or long-lasting dry periods and later in an oxygen-depleted environment, nitrifying bacteria oxidize ammonium to nitrate. If nitrate is transported to regions with a low concentration of dissolved oxygen it can be denitrified, or in the event of a sufficient presence of dissolved organic carbon, reduced back to ammonium. Lower floodplain strata usually contain processes of reduction, while upper strata contain oxidation, and zones of nitrification and denitrification are also separated in space and time. Therefore floodplain ecosystems are very efficient in nitrogen removal.

During periods of very low ORP and low pH value, significantly higher Al and Fe runoff is observed in the Sipe floodplain. ORP values about two times lower than the mean annual characteristic for a particular vegetation pattern lead to an increase in Al concentration 2–4 times above the annual mean (up to 0.89 mg/l). Under the same conditions, Fe concentrations were about twice as high (3.38 mg l^{-1}) as the annual mean in the same location. Reduced iron and manganese ions may play an important role during their oxidation in phosphate removal from water as the co-precipitation of phosphorus occurs.

In spring, when the groundwater level is close to the surface and both soil matrix and macropore transmissivity are exceeded, overland flow occurs. This leads to high nutrient losses but relatively low variation in stream water chemistry, where the concentrations remain 4–5 mg l^{-1} for TIN, 0.03–0.06 mg l^{-1} for total P and 2–10 mg l^{-1} for SO_4.

In dry summers when precipitation is less than 150 mm in May–September and the water-table level is more than 50 cm below ground level, the spatial variation in nutrient concentration is high. Compared to the interannual mean value, in dry summers the concentrations of TIN, NO_2, K and total-P are lower, while those of Mg, Ca, NH_4, Fe and Al have a tendency to increase. In wet summers when precipitation exceeds 500 mm in the period from May to September and the water-table level is only 0–35 cm below ground level, the nutrient concentration is spatially also relatively uniform (1–2 mg l^{-1} NH_4, 2–4 mg l^{-1} TIN, 0.04–0.3 mg l^{-1}

PO_4), although nutrient losses are high due to intensive subsurface flow and high water discharge. However, nitrogen removal is expected to be intensive due to the denitrification process in wetlands and river valleys.

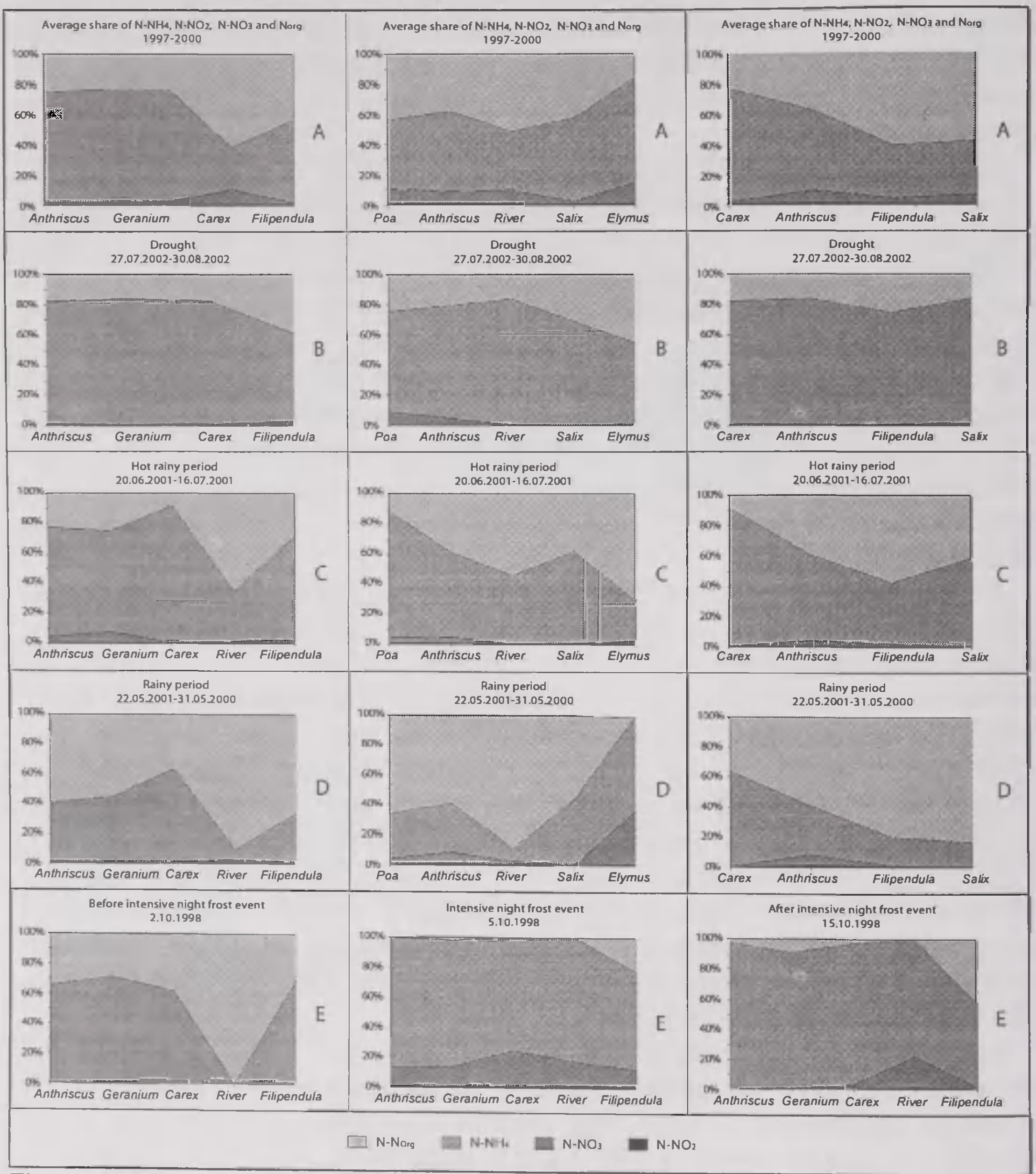

Figure 1. Response of nitrogen species in subsurface water to certain weather events, by vegetation type.

Compared to the interannual mean value (Figure 1, A), the concentrations of total-N, NO_2, K and total P are lower in dry summers (Figure 1, B), while those of Mg, Ca, NH_4, Fe and Al have a tendency to increase. Decreased water discharge keeps nutrient losses at a low level.

Table 1. Magnesium, potassium, nitrogen and phosphorus release during night frost episode in various plant communities.

Date	min t° °C	Mg ($mg\ l^{-1}$)			K ($mg\ l^{-1}$)			NO_3-N ($mg\ l^{-1}$)			Total-P ($mg\ l^{-1}$)		
		Carex	Filipendula	Carex	Carex	Filipendula	Salix	Filipendula	Salix	Salix	Carex	Filipendula	Salix
02.10.98	–0.6	23.4	2.9	3.0	12.1	1.2	1.82	0.005	0.005	0.005	0.03	0.06	0.37
05.10.98	–8.4	23.8	3.7	4.3	14.3	1.8	3.10	1.5	1.0	1.1	0.06	0.08	0.49
15.10.98	+6.2	23.0	3.7	4.4	16.2	1.2	2.14	0.2	1.1	0.03	0.04	0.07	0.46

A dry summer followed by intensive rainfall results in a steep increase in nutrient losses and SO_4 concentration (from 8 to 50 $mg\ l^{-1}$), total P (from 0.03 to 0.6 $mg\ l^{-1}$) and NH_4 (from 1.8 to 2.9 $mg\ l^{-1}$), NO_2 (from 0.005 to 0.1 mg/l) (Figure 1, C). A tendency towards increasing concentration is also shown by NO_3, total N, PO_4 and Ca. Mg and Al show no clear trend, and the concentration of Fe decreases slightly, but falling concentrations after rainfall are clearly characteristic of K (from 15 to 5 $mg\ l^{-1}$). Changes in the concentrations of Mg and K are mainly due to dilution. Changes in different nitrogen forms during rainy periods (Figure 1, D) indicate highly intensified denitrification, which benefits (especially after long drought) infiltrating precipitation and changes in O_2, ORP and humidity. Denitrification fluxes remain high even in late autumn. A very important factor affecting nutrient fluxes is the soil's freeze-thaw cycle. Short-term freeze-thaw cycles in autumn promote NO_2 and N_2O fluxes (Figure 1, E), especially the former. The freeze-thaw cycle increases denitrification and nutrient release in the whole floodplain, but the magnitude of release varies from one vegetation communities to another. The highest response is given by the *Carex* community (increase of NO_3 from 0.005 to 1.49 $mg\ l^{-1}$ and total P from 0.028 to 0.059 $mg\ l^{-1}$), where due to a high water table level, ice formation probably disintegrates aggregates more effectively. A short high NO_3 flux was also characteristic of the *Salix* forest (NO_3 from 0.005 to 1.09 $mg\ l^{-1}$), while the response in the *Filipendula* community was similar in extent, but with a longer time lag. The strongest pulse is triggered by the first strong night frost episode or frost period, whereas the following freeze-thaw cycles induce gradually weaker fluxes, until the surface freezes in winter and the next strong pulse takes place in spring. Total P has been reported to be less influenced by frost, but it appears that total P release is more complex and is also heavily dependent on other parameters (water level, ORP, O_2).

The release of cations exhibits different behaviour during the freeze–thaw cycle than nitrogen or phosphorus (Table 1). In peatland areas the accumulation in biomass is important for Ca, storage on ion exchange sites in peat is most important for Mg, and green plant tissues are the dominant site for K storage. Therefore different communities in the floodplain respond differently to cation release under a night frost event. Mg is not a limiting factor in the *Carex* community, and therefore a frost event did not yield any significant response, while in *Filipendula* (from 2.92 to 3.72 mg l^{-1}) and *Salix* communities (from 2.98 to 4.41 mg l^{-1}) an increase in Mg set in after a frost event. Despite the high biomass in *Filipendula* communities, the potassium concentration did not increase after a frost event, as most nutrients had already been allocated in September. Communities with a longer vegetation period, such as *Carex* and *Salix* (K from 1.82 to 3.10 mg l^{-1}) were more strongly influenced by night frost.

Conclusions

The following factors have the highest priority in weather-induced changes in nutrient fluxes in the Sipe River floodplain: a) duration of frozen surface, b) snowpack peak water, c) precipitation pattern over warm period, d) duration and continuity of certain weather, e) occurrence of night frost events and soil freeze-thaw cycles. The temporarily frozen surface has an effect by limiting gas exchange and leading to oxygen depletion, which is associated with a low ORP value. Snowpack melt water and the duration of certain weather conditions uniform processes intensity in floodplain. Snowmelt entering unfrozen soil significantly alters water chemistry and soil processes due to rapidly increasing O_2 concentration (from 0.0–0.1 to 6.0 mg l^{-1}). Long-term (>1.5–2 weeks) homogeneous weather events may change the intensity of or alter soil processes and result in or stop surface and subsurface flow with high nutrient runoff. Autumn rainfalls should be treated as a special case, as they continuously and gradually increase dissolved O_2 concentration and enable vertical solution exchange between soil horizons. When night frost events begin, the importance of autumn rainfalls is enhanced as the high water table intensifies the denitrification process but also the increase of Ca, Mg, K, N and P losses released during the frost-thaw cycle.

Acknowledgements

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Nitrogen and phosphorus assimilation and biomass production by *Scirpus sylvaticus* and *Phragmites australis* in a horizontal subsurface flow constructed wetland

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Introduction

Plants are an important part of constructed wetlands for wastewater treatment. They have several important functions that will improve purification efficiency and prolong the working life of constructed wetlands (Vymazal *et al.*, 1998). One important aspect is plant uptake; plants absorb nutrients that are assimilated in the tissues of the plants; these nutrients can be removed by harvesting or may be bound in stable compounds. However, part of the nutrients taken up by plants can be leached out of the system. Several plant species support various functions to a different extent and have a large variation in productivity and the ability to assimilate nutrients. The production of different plants used in constructed wetlands can vary from 700 to 11,000 g m^{-2} yr^{-1}, uptake of nitrogen from 12.5 to 585 g m^{-2} yr^{-1} and uptake of phosphorus from 1.8 to 112.5 g m^{-2} yr^{-1} (Kadlec and Knight, 1996). Maximum values, however, are achieved by tropical plants such as *Eichhornia crasipes*, *Pistia stratiotes*, etc. In subsurface flow constructed wetlands in colder climates, the most common plant used is *Phragmites australis*, because it has a well-developed root system (Hoffmann, 1997; Geller, 1997; Vymazal, 2002). Kadlec and Knight (1996) give a production rate for *Phragmites australis* of 1000 to 6000 g m^{-2} yr^{-1}, nitrogen uptake rate of 22.5 g m^{-2} yr^{-1}, phosphorus uptake rate of 3.5 g m^{-2} yr^{-1}. In Europe, the production of reed in constructed wetlands has been found to be 1200 g m^{-2} yr^{-1} (Obarska-Pempkowiak, 1997) to 4000 g m^{-2} yr^{-1} (Hoffmann, 1997). Assimilation of N and P was found to be 16 g m^{-2} yr^{-1} and 1.6 g m^{-2} yr^{-1} respectively (Geller, 1997) to 65.2 g m^{-2} yr^{-1} and 5.6 g m^{-2} yr^{-1} respectively (Obarska-Pempkowiak, 1997).

In the total budget of nutrient removal in a constructed wetland, however, the assimilation of nutrients in plants plays a minor role. It is usually less than 10% of the nitrogen and in most cases less than 5% of the phosphorus removed in a constructed wetland (Brix, 1994; Geller, 1997; Vymazal, 2002).

We studied production and nitrogen and phosphorus assimilation in horizontal subsurface flow constructed wetlands by reed (*Phragmites australis*) and by wood club-rush (*Scirpus sylvaticus*).

Materials and methods

Site Description

The Kodijärve planted sand filter is situated in southern Estonia, 20 km south of Tartu. The system was constructed in October 1996 by the Centre for Ecological Engineering Tartu (CEET) to purify wastewater from a hospital for about 40 persons. The wastewater enters a two-chamber septic tank and flows into a horizontal subsurface flow planted soil-filter that is a replica of the similar system in Germerswang, Germany (Geller *et al.*, 1991). The wetland consists of two separated beds (chambers), each measuring 25×6.25×1 m, which are filled with coarse sand. The wastewater enters the wetland through a splitter chamber that divides the water equally between the two beds, where it seeps through the inlet pipes into the sand. The beds have a somewhat different soil texture, which leads to a different moisture content in the two separate beds. The bed with finer filter material has more surplus moisture conditions and shall hereafter be referred to as the wet bed. The bed with coarser material and drier conditions is referred to as the dry bed.

In May 1997, aquatic macrophytes were planted in the system. The dry bed was planted with 360 young *Typha latifolia* (cattail) plants, and the wet bed with 360 young *Iris pseudacorus* (yellow iris) plants and 75 young *Phragmites australis* (reed) plants. During the following years, the plant cover did develop spontaneously, and by the year 2001 it was altered to a remarkable degree. The dry bed was dominated mainly by *Scirpus sylvaticus* (wood club-rush), *Urtica dioica* (nettle), *Epilobium hirsutum* (hairy willow-herb) and a few individuals of other species, but no cattail was found. In the wet bed, about 60% of the bed was dominated by wood club-rush; the rest of the bed was populated by patches of reed and some single individuals of other species.

Phytomass Sampling and Analysis

The biomass samples were collected from 5 plots of both beds in late July 2001 and 2002, during the maximum flowering time of the dominant plant species (see Milner and Hughes, 1968). Sampling plots were chosen in typical areas of the community. In the dry bed all plots were dominated by wood club-rush, while in the wet bed three plots were dominated by wood club-rush and two plots by reed. The above-ground biomass and the litter from the same year were collected from quadrates measuring 0.5x0.5 m. Below-ground root biomass was collected from soil cores taken using an auger (diameter 118 mm) in 10 cm layers, to a depth of

50 cm from each sampling plot. Roots were washed clean of soil, and the dry weight of the dried roots and above-ground biomass was measured and N and P content was analysed in the Plant Biochemistry laboratory at the Estonian Agricultural University. The ash content of roots was about 20%, which indicated that about 10% of the ash was contaminated by mineral particles, which is a satisfactory result for below-ground biomass.

Results and discussion

There was a considerable difference in production rate between the years 2001 and 2002, when the average productivity of plants in the whole constructed wetland was 620 and 3071 g m^{-2}, respectively. This was caused by great differences in weather conditions, while the year 2002 was much warmer than 2001. In conditions where there is a surplus of water, productivity is substantially affected by temperature. The impact of weather conditions was checked by calculating the value of the relation of root biomass divided by the whole plant's biomass, which did not differ considerably, not more than 0.2 units in all measuring points over two years. This small variation shows the importance of growing factors over a two-year period. Therefore, in 2002 there was significantly higher above and below ground biomass at all sampling points (Figure 1).

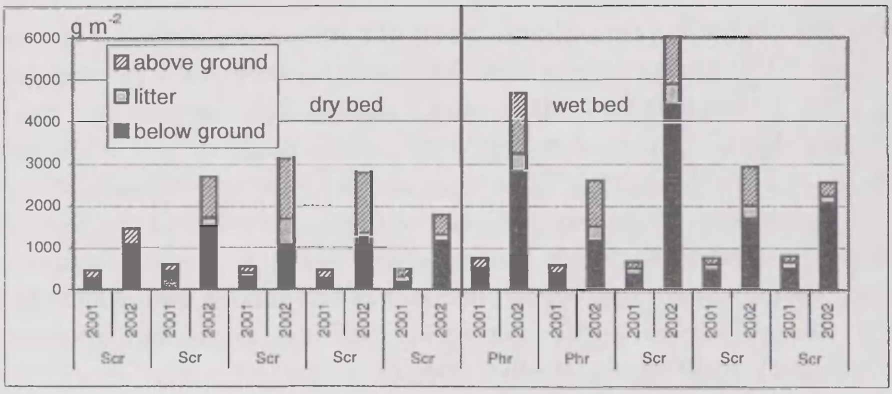

Figure 1. Above- and below-ground biomass and litter mass in Kodijärve horizontal subsurface flow constructed wetland by *Scirpus sylvaticus* (*Sci*) and *Phragmites australis* (*Phr*) in 2001 and 2002.

There was also a significant difference in biomass between the dry and wet beds. The average dry weight biomass of 5 plots in 2001 was 513 g m^{-2} in the dry bed and 729 g m^{-2} in the wet bed. In 2002 these values were 2520 and 3765 respectively.

The higher biomass values in wet conditions were reached mainly due to higher below-ground biomasses, which showed significant differences between the two beds. Likewise, the wet bed had a higher litter mass than the dry bed.

The nutrient assimilation in plants depends on the production rate required to equal the biomass value. Therefore, variations in biomasses will also appear in values for nutrient assimilation. The average nitrogen assimilation by plants was 16.6 g m^{-2} in the wet and 10.7 g m^{-2} in the dry bed in 2001, and 51.7 and 83.5 g m^{-2} in 2002 respectively (Figure 2).

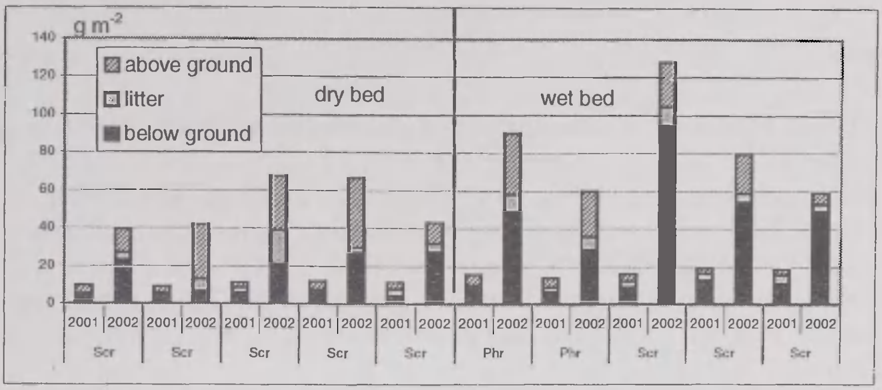

Figure 2. Assimilation of nitrogen in above- and below-ground biomass and in litter mass in Kodijärve horizontal subsurface flow constructed wetland by *Scirpus sylvaticus* (*Sci*) and *Phragmites australis* (*Phr*) in 2001 and 2002.

Roots assimilated 9.8 g m^{-2} in the wet and 5.1 g m^{-2} in the dry bed during 2001, and in the next year these values were 55.2 and 21.2 respectively. Total assimilation of nitrogen by wood club-rush in the wet bed reached 69.3 g m^{-2} in 2002, which was much higher than that estimated by Kadlec and Knight (1996) for the bulrush family (*Scirpus*) – 12.5 g m^{-2} yr^{-1}.

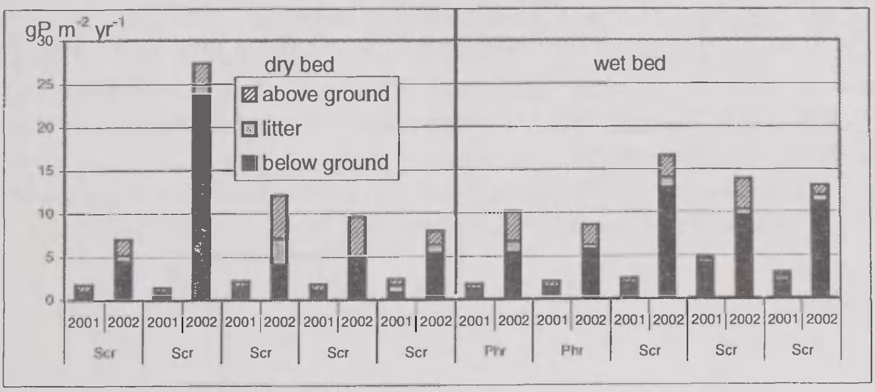

Figure 3. Assimilation of phosphorus in above and below-ground biomass and in litter mass in Kodijärve horizontal subsurface flow constructed wetland by *Scirpus sylvaticus* (*Sci*) and *Phragmites australis* (*Phr*) in 2001 and 2002.

Phosphorus assimilation was higher in the wet bed during 2001– 2.8 g m^{-2}, in the dry bed this value was 1.9 g m^{-2}. In 2002, the phosphorus assimilation was higher in the dry bed – 12.8 g m^{-2}, in the wet bed it was 12.43 g m^{-2} (Figure 3). As in the case of nitrogen, phosphorus assimilation by wood club-rush in the wet bed in 2002 (14.5 g m^{-2} yr^{-1}) was higher than that estimated by Kadlec and Knight (1996) for bulrushes, i.e. 1.8 g m^{-2} yr^{-1}. These results show the effectiveness of wood club-rush in assimilating nutrients that can be considered possible plant species for subsurface flow constructed wetlands.

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Phosphorus retention as affected by phosphorus load in an agricultural wetland

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Introduction

Diffuse agricultural runoff waters are the most important non-point sources of nutrients to rivers, lakes and seas provoking their eutrophication. Constructed wetlands (CWs) offer a promising tool for the treatment of agricultural runoff, which is difficult and expensive to treat with conventional, massive wastewater purification techniques (Reddy *et al.*, 1999). The main objective of the CWs established in agricultural areas is to reduce the loading of suspended solids, phosphorus (P), and nitrogen (N). Particularly, the removal of P is of importance, since P is generally the major limiting nutrient for algal growth in freshwater ecosystems. We studied the efficiency of P removal in the agricultural constructed wetland of Hovi (the Hovi CW) in Finland under various P loading and oxygen conditions.

Materials and methods

The Hovi wetland (60°25´N, 24°22´E, Finland) was constructed on the arable land in 1998 to study the nutrient removal from agricultural runoff water. The 12-ha catchment of the wetland consists entirely of clayey arable land. In the construction of the wetland, the topsoil (0–30 cm) was removed to avoid the release of P accumulated in the plough layer as a result of fertilization history. A deeper region (deep water area, depth 1–2 m) was excavated at the inlet of the

CW to enhance sedimentation. After the deep water area water flows through shallow water area (depth of 0–0,5 m) growing cattail (*Typha latifolia*), club-rush (*Scirpus sylvaticus*), common waterplantain (*Alisma plantago-aquatica*), reed canary grass (*Phalaris arundinacea*), meadowsweet (*Filipendula ulmaria*), yellow flag (*Iris pseudacorus*) and compact rush (*Juncus conglomeratus*).

Reduction of P in the Hovi CW *in situ* was studied during 11 months in 1999–2001 The daily mean inflows and outflows in the CW were derived from the curves drawn by continuously recording water-stage gauges at control weirs (with pre-known stage-discharge relationships) installed at the inlet and outlet. Water samples for P analyses were taken flow-proportionally from the inflow and outflow. A total of 89 pairs of samples from the inlet and outlet were analyzed for dissolved reactive P (DRP) using the molybdate blue method (Murphy and Riley, 1962). Daily DRP loading to and from the CW was calculated by multiplying daily concentration (concentrations for the days between the sampling occasions were obtained by linear interpolation) by the daily mean flow of the corresponding day. Daily P reductions were obtained by subtracting output P loading from input P loading. The monthly average DRP fluxes (mg P m^{-2} d^{-1}) and retentions (%) were calculated from the daily values, and were compared to monthly averages of input DRP concentrations.

Retention of P with various oxygen conditions and DRP loadings were studied in the laboratory in 2001. Intact wetland sediments were incubated in a laboratory microcosm (at 15 °C) under a continuous water flow (Liikanen *et al.*, 2002). The sediments were taken in June 2001 from the deep water area (water depth 1.5 m) and in July 2001 from the shallow water area, littoral sediments growing cattail (water depth 0–10 cm). The deep water sediments without plants were incubated in the dark first under oxic and then under anoxic water according to Liikanen *et al.* (2002). The littoral sediments with living plants were incubated under artificial lighting under oxic water only, since the littoral sediments are not exposed to anoxic water *in situ*. The sediments were incubated with artificial runoff water having 16, 47 and 155 µg PO_4^{3-}-P l^{-1} corresponding to the minimum, mean and maximum PO_4^{3-}-P concentrations the CW had received in 1999–2000. The flux of DRP (mg m^{-2} d^{-1}) and reduction (%) were determined from concentration differences between in- and out-flowing water using flow rates and sediment surface area. DRP was analyzed according to Finnish standard SFS 3025 (SFS Standardization, 1986).

Results

The Hovi CW was efficient in retaining P (Figure 1A-D). The *in situ* flux of P from overlying water to sediment varied from 0.03 to 2.6 mg P m^{-2} d^{-1}. The wetland was able to retain more P when the inflowing water DRP concentration increased (Figure 1A). However, the relative retention *i.e.* reduction-% decreased as DRP concentration increased (Figure 1B). In April 2000 the CW did not retain any DRP, this record makes an exception when DRP retention is compared to the inflowing water DRP concentration (76 mg P l^{-1}). The total DRP reduction in the Hovi CW in 1999–2001 was 49%.

In the laboratory studies the sediment retained DRP from overlying water in all conditions (Figure 1C-D). As expected, with increasing DRP concentration in inflowing water the flux of DRP from water to sediments increased (Figure 1C) but the reduction-% of DRP decreased (Figure 1D). Fluxes of DRP to the sediment observed in the laboratory sediment were higher, from 0.15 to 10.5 mg P m^{-2} d^{-1}, than measured for the whole CW *in situ*. However, the relative DRP reductions measured in microcosm were similar to those measured *in situ*.

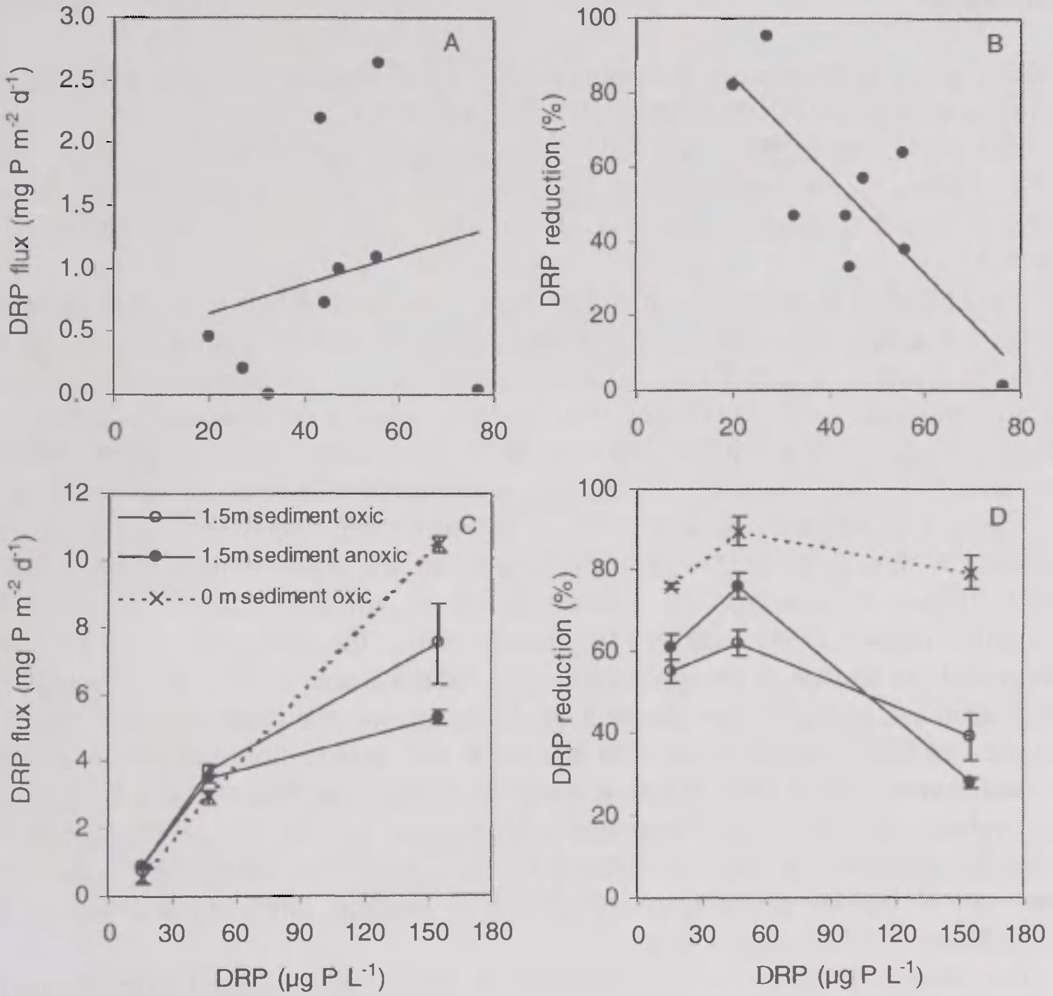

Figure 1. Reduction of DRP in the Hovi CW with various DRP concentrations. A) flux of DRP and B) relative DRP reduction in wetland *in situ*, and C) flux of DRP and D) relative DRP reduction in intact sediment cores in laboratory.

The DRP reduction-% was at highest when inflowing water DRP concentration was 47 μg P l^{-1}, the mean DRP concentration of the inflowing water in the Hovi CW.

The sediments incubated under dissimilar conditions did not significantly differ in their P retention when P concentrations in the inflowing water were low (Figure 1C-D). At the highest P concentration (155 μg P l^{-1}), however, the oxic littoral sediment retained P more effectively than did the deep water sediment. The anoxic deep water sediment had the lowest retention ability. The DRP reduction- % was higher in the littoral sediment than in the deep water sediment (Figure 1D).

Discussion

Both *in situ* and laboratory measurements showed that the CW was efficient in retaining P. The DRP reduction of 49% for the Hovi CW *in situ* is similar to the average reduction of 58% for 83 various wetlands reviewed by Reddy *et al.* (1999). When negligible retention for DRP was detected in April 2000, some P was probably released to dilute snowmelt water from newly sedimented matter rich in P.

The soil of Hovi wetland, which had been exposed to P rich agricultural runoff waters for rather short time, had excellent capacity to retain P even with small P loads. Previously it has been reported that at low P concentrations wetlands release rater than retain P (Reddy *et al.,* 1999). However, the highest amounts of DRP were retained when the input concentration of P was at highest. When inorganic P is added at concentrations, greater than those present in the porewater of sediment or wetland soil P is retained by oxides and hydroxides of Fe and Al and by Ca (Reddy *et al.*, 1999). The bottom of the wetland was mineral soil, which, before the construction, had plenty of Al- and Fe-oxides of very low P saturation degree (Räty, 2000). In Finnsih soils, the retention of P by Ca-compounds is shown to be immaterial (*e.g.* Hartikainen, 1979). With low DRP concentrations, probably the retention by Al oxides was the most crucial, since the retention of DRP was efficient with both oxic and anoxic flow conditions. If the Fe oxides would have been the main sorption component, there would have been some release of P or lower P retention with the anoxic flow, where P retained by Fe oxides dissolves as Fe^{3+} is reduced to Fe^{2+}. However, when DRP load was high, also Fe oxides probably contributed to P binding, since the retention of P was reduced with the anoxic flow.

The relative P reduction i.e. reduction-% decreased with the highest P load, indicating that the soil sorption capacity was over-loaded. Also others have found that relative retention decreases with high P loads (Pant *et al.*, 2001; Sakadevan and Bavor, 1999). The highest DRP concentration tested occurs rarely *in situ*, with the mean DRP concentration of 47 μg P l^{-1} the relative retention was at highest.

The Hovi CW constructed on the former agricultural soil was efficient in retaining P from agricultural runoff waters. The removal of P-rich topsoil, generation of deep sedimentation pond, and tortuous shallow flow path were the essential factors behind the good purification efficiency. Mineral soils offering plenty of long-term sorption sites for inorganic P seem to provide a good basis for constructed wetlands.

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Typha populations in wastewater treatment wetlands in Estonia: Biomass production, retention of nutrients and heavy metals

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Introduction

Plants are an important part of constructed wetlands for wastewater treatment. Macrophytes absorb nutrients that are assimilated in the tissues of the plants, provide a surface for microorganisms, prevent soil clogging and insulate the surface against frost during winter, control erosion, transport oxygen to the littoral from the atmosphere, as well as additional ecological roles like creating habitats for birds and mammals and increasing the aesthetic value of the site (Kadlec and Knight, 1996; Brix, 1997; Bachand and Horne, 1998). Wetlands are considered among the most productive ecosystems in the world. The production of different plants used in constructed wetlands (CW) can vary from 700 to 11 000 g m^{-2}, uptake of nitrogen (N) from 12.5 to 585 g m^{-2} and phosphorus (P) from 1.8 to 112.5 g m^{-2} (Kadlec and Knight, 1996). The potential to recover biomass and assimilate N compounds and P in plant tissues are among current criteria for macrohpyte species selection. Another advantage of macrophytes in constructed wetlands is the potential usefulness of their biomass (Ennabili *et al.*, 1998).

The most commonly used macrophytes in CWs are cattails (*Typha latifolia* and *Typha angustifolia*), club-rush (*Schoenoplectus lacustris*) and common reed (*Phragmites australis*); Kadlec and Knight, 1996; Bachand and Horne, 1998). *T. latifolia* is the only species of cattail usually found in relatively undisturbed habitats throughout the entire world. The tolerance of *T. latifolia* to high concentrations of lead, zinc, copper, and nickel has been demonstrated (Taylor and Crowder, 1984). This species has most commonly been employed in secondary wastewater treatment schemes (Gopal and Sharma, 1980).

The aim of this paper is to evaluate and compare cattail biomass production and nitrogen, phosphorus, carbon and heavy-metal (Cd, Cu, Pb, Zn) assimilation rates in 3 treatment wetland systems in Estonia.

Materials and methods

Site description. Biomass production and nutrient and heavy metal assimilation by cattail were studied in three wetlands: in the subsurface flow semi-natural wetland in Tänassilma (58°22' W 25°31' N) and in two free water surface CWs in Põltsamaa (58°38' W 25°58' N) and Häädemeeste (58°5' W 24°29'N).

The Tänassilma semi-natural wetland is located in a primeval valley 0.5 km from the town of Viljandi (~22,000 inhabitants). Since 1948, large amounts of untreated municipal wastewater from town have been discharged into the wetland. About 69 ha is covered with dense cattail stands (Nõges and Järvet, 2002). The Põltsamaa CW is a cascade of 4 serpentine ponds with aquatic macrophytes located in the flood plain of the Põltsamaa River. This system is designed for the secondary treatment of wastewater from the conventional treatment plant. It treats wastewater from the town of Põltsamaa (~5000 inhabitants) and was constructed by the Tartu Centre of Ecological Engineering in 1997 (Mander *et al.*, 2001a). The Häädemeeste CW was built by the Tartu Centre of Ecological Engineering in 1999. The system consists of a conventional treatment plant, five infiltration ponds and a cattail basin. The wetland treats the municipal water for the settlement of Häädemeeste. The primary purpose of the wetland is the removal of N and P (Mauring, 2002).

Phytomass sampling and analysis

Biomass assessments were carried out just after maximal growth of macrophytes, at the end of August and the beginning of September in 2002. Cattail was divided into four fractions: roots-rhizomes (dead and living parts together), shoots with leaves, spadixes(spikes) and litter. The aboveground biomass was harvested at the ground level. The belowground roots-rhizome samples were collected using an auger (Ø108.6 mm). The weight of cattail fractions were measured on 15 1-m^{-2} plots in Tänassilma, 15 plots in Põltsamaa and 10 plots in Häädemeeste. The nutrient (N, P) and carbon (C) content of each fraction was analysed from 9 samples in Tänassilma and Põltsamaa, and 10 samples in Häädemeeste. The cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn) content was measured in each fraction from 4 samples in Tänassilma and Häädemeeste and from 6 in Põltsamaa. In order to analyse heavy metal content from the cattail spadixes, all spadix samples were mineralized in a microwave oven with HNO_3. All chemical analyses were performed at the laboratories of the Tartu Environmental Research Ltd.

Statistical analysis

The statistical analysis was carried out using the Excel and STATISTICA 6.0 (StatSoft Inc.) programs. The normality of the variables was verified using the Kolmogorov-Smirnov, Lilliefors' and Shapiro-Wilk's tests. Variables were normally distributed. Level of significance α=0.05 was accepted in all cases.

Results and discussion

Biomass

In Tänassilma the highest average total cattail phytomass was 6.9 kg DW m^{-2}. In Põltsamaa and Häädemeeste this value was 2.7 and 3.1 kg m^{-2} respectively.

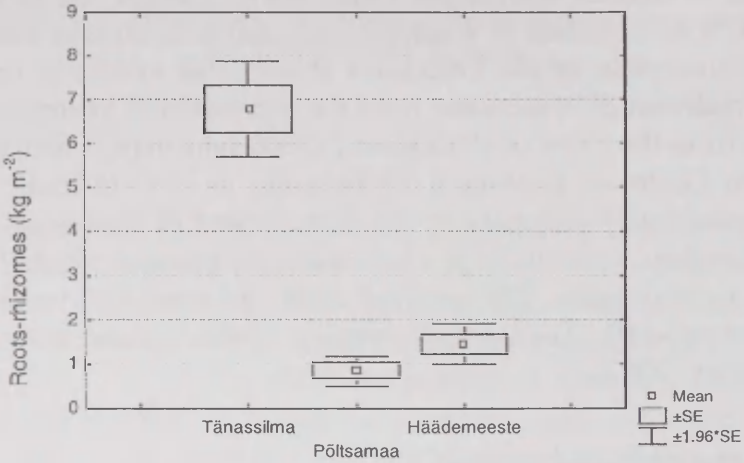

Figure 1. Cattail root and rhizome phytomass (kg m^{-2}) in Tänassilma seminatural wetland and Põltsamaa and Häädemeeste constructed wetlands.

The estimated biomass of shoots was 1.52 kg m^{-2} in Tänassilma, 0.99 kg m^{-2} in Põltsamaa, and 0.65 kg m^{-2} in Häädemeeste. In the Häädemeeste CW, 0.21 kg m^{-2} was made by spadixes, which was about 0.1 kg m^{-2} more than in Tänassilma and Põltsamaa. Likewise, the litter mass was greater in Häädemeeste – 0.76 kg m^{-2}, being 0.45 and 0.71 kg m^{-2} correspondingly in Tänassilma and Põltsamaa. The average total aboveground biomass production of the whole plant was not significantly different in three wetland systems, although there were significantly more roots-rhizomes in Tänassilma than in Põltsamaa and Häädemeeste (Figure 1).

We found a significant positive correlation between the fractions of aboveground biomass, whereas the belowground phytomass values did not correlate with the aboveground ones.

The aboveground biomass values in the studied systems are smaller than reported by Ennabili *et al.* (1998) – 3.5 kg m^{-2}, but exceed those found in Germany (1.3–1.45 kg m^{-2}; Wild *et al.*, 2002). The roots and rhizomes biomass values in Põltsamaa and Häädemeeste CWs were similar to those recorded by Ennabili *et al.* (1998) and Romero *et al.* (1999), varying from 0.7 to 1.6 kg m^{-2}.

There was significantly greater roots-rhizomes phytomass in Tänassilma than that in Põltsamaa and Häädemeeste. The Tänassilma semi-natural wetland is much older and part of roots-rhizomes has been accumulated in sediment. Due to the anaerobic condition in sediment, this material decomposes more slowly than in free water surface CWs.

The litter, spadixes and shoots biomass values indicated that in free water surface CWs, plants grow and die more quickly than in subsurface wetlands. Therefore, in Tänassilma we found significantly less spadixes and litter than in Põltsamaa and Häädemeeste.

Nutrients

The average N and P contents in shoots were greater in Põltsamaa – 2.1% N and 0.3% P. The results in Tänassilma and Häädemeeste were 1.8% N, 0.28% P and 1.6% N, 0.22% P respectively (Figure 2).

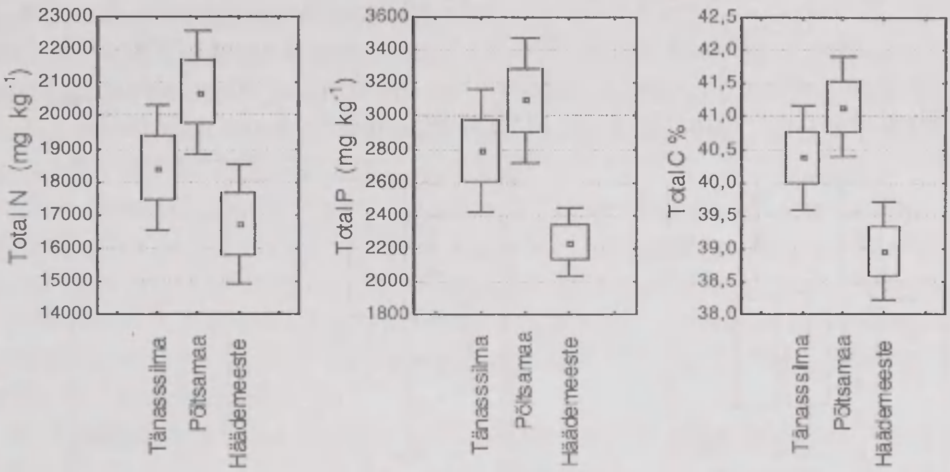

Figure 2. Assimilation of N, P and C in cattail shoots in Tänassilma semi-natural wetland, Põltsamaa and Häädemeeste CWs.

In Tänassilma the N, P and C content in roots-rhizomes was 2.2%, 0.23% and 42.3% respectively, whereas in Põltsamaa and Häädemeeste the corresponding values were 2.1% N, 0.39% P and 40.1% C, and 1.5% N, 0.24% P and 33.8% C, respectively. N and P concentrations were slightly lower in litter than in shoots, being 1.2% N and 0.24% P in Tänassilma, 1.3% N and 0.2% P in Põltsamaa, and 0.9% N and 0.15% P, respectively in Häädemeeste. The C content in litter and shoots did not differ significantly.

In spadixes the highest average N and P content – 2.3% N and 0.48% P – was found in Põltsamaa CW. In Tänassilma and Häädemeeste the spadixes contained less N and P: 2.0% N and 0.41% P, and 1.7% N and 0.37% P respectively. The greatest C assimilation was observed in Häädemeeste (44.2%), however, this did not significantly differfrom values found in Tänassilma and in Põltsamaa: 43.5% and 43.1% C, respectively.

The N and P contents in the belowground portion were negatively correlated in the Tänassilma samples, whereas a positive correlation was found between N and P phytomass in the belowground biomass from Põltsamaa and Häädemeeste. In shoots and litter there were correlations between N and P assimilations in Tänassilma and in Põltsamaa.

We could not find any correlation between the biomass and N, P and C content in plant tissues. This might mean that nutrient content in cattail fractions do not depend on aboveground or belowground biomass production.

The nutrient contents in cattail tissues were similar to that estimated by Kadlec and Knight (1996), Ennabili *et al.* (1998) and Hume *et al.* (2001). In these reports, shoots and roots and rhizomes contained 2.0–2.5% N (Kadlec and Knight, 1996; Ennabili *et al.*, 1998), 0.25% P (Kadlec, Knight, 1996; Ennabili *et al.*, 1998) and 40–44% C (Hume *et al.*, 2001). Only a half of these amounts were found to be contained in litter – 1.2% N and 0.12% P (Kadlec and Knight, 1996). We found that N and P are stored in reserve organs after the fruiting stage, while C storage has not been observed. Ennabili *et al.* (1998) reported the same conclusion.

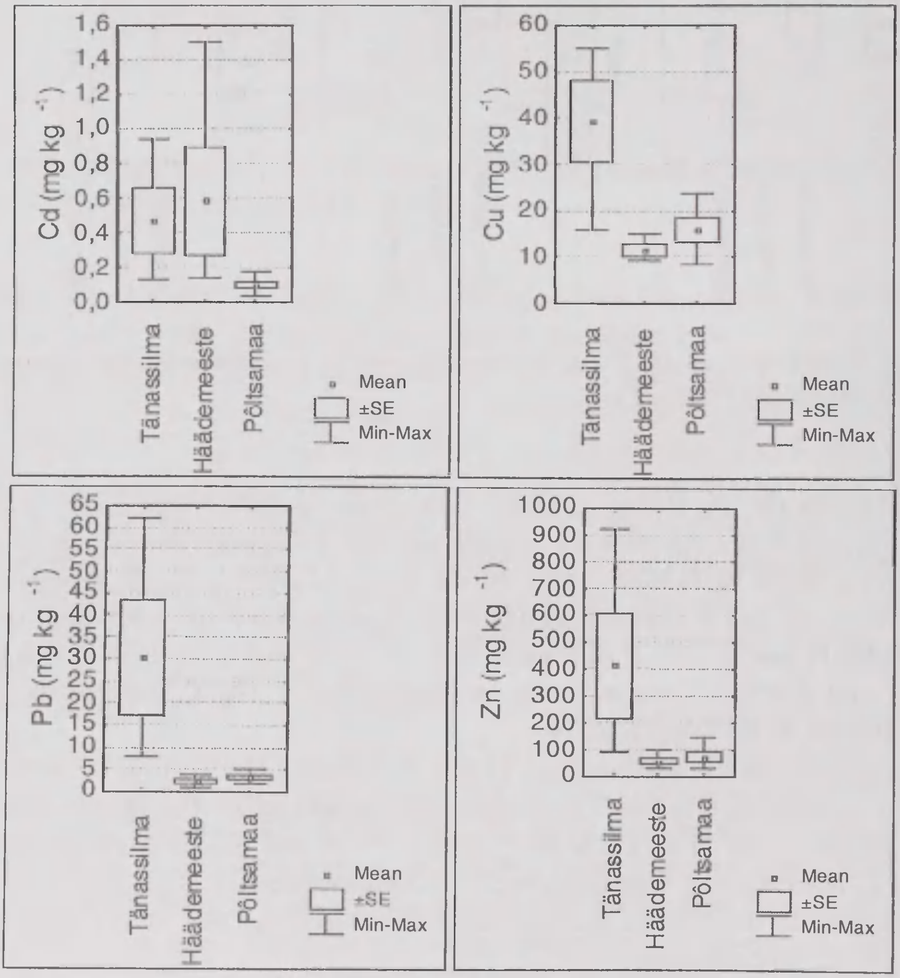

Figure 3. Assimilation of Cd, Cu, Pb and Zn in cattail roots-rhizomes in Tänassilma semi-natural wetland and Põltsamaa and Häädemeeste constructed wetlands.

Heavy metals

The Cd concentrations in all samples (shoots, spadixes, litter) varied from <001 mg/kg to <002 mg/kg. In Põltsamaa and Häädemeeste CWs the average content of Cu in shoots was 3.6 mg kg^{-1}, but in Tänassilma it was significantly lower – 2.3 mg kg^{-1}. The average Pb content was the same in almost all shoot samples, namely <0.1 mg kg^{-1}. The highest average content of Zn in shoots was 15,4 mg kg^{-1} in Põltsamaa, whereas in Häädemeeste and Tänassilma it was 14.3 mg kg^{-1} and 13.9 mg kg^{-1}, respectively.

Values of Cd, Cu, Pb and Zn concentrations in the roots and rhizomes of *T. latifolia* are shown in Figure 3. In Tänassilma research found a significantly higher average contents of Cu (39.3 mg kg^{-1}), Pb (30.4 mg kg^{-1}), and Zn (412.3 mg kg^{-1}) than those in Häädemeeste or Põltsamaa: Cu – 11.6 and 15.9, Pb – 2.3 and 3.3, and Zn – 57.5 and 73.2 mg kg^{-1}, respectively. Likewise, the highest Cd content was found in roots and rhizomes. In Häädemeeste, it was 0.59, in Tänassilma 0.47, and in Põltsamaa 0.1 mg Cd kg^{-1}.

The lowest heavy metals contents were found in litter. The average concentration of Zn varied from 12.2 mg kg^{-1} in Häädemeeste to 12.6 mg kg^{-1} in Tänassilma and 13.3 mg kg^{-1} in Põltsamaa. In Tänassilma, the Cu and Pb content in litter were 1.4 mg Cu kg^{-1} and 0.17 mg Pb kg^{-1}, whereas in Põltsamaa and in Häädemeeste the relevant values were 2.3 and 1.7 mg Cu kg^{-1}, and 0.2 and 0.18 mg Pb kg^{-1} respectively.

In Häädemeeste, the highest average Zn and Cu concentrations were found in spadixes: 23.8 and 14.8 mg kg^{1} correspondingly. At the same time, the lowest average Pb and Cd contents were 0.10 mg kg^{-1} and <0.02 mg kg^{-1}, respectively. The average Zn content was 21.8 mg/kg in Tänassilma and 19.6 mg kg^{-1} in Põltsamaa, whereas Pb concentrations varied from 0.17 to 2.0 mg kg^{-1}. The average content of Cu in spadixes was 9.0 mg kg^{-1}in Põltsamaa and 5.4 mg kg^{-1} in Tänassilma.

The highest accumulation of heavy metals occurred in the roots and rhizomes of *T. latifolia*. Spadixes also showed a significantly higher accumulation capacity than shoots. Thus the harvesting of spadixes for building material (Mander *et al.*, 2001b) can help to remove some accumulated heavy metal content. The highest measured heavy-metals contents were observed in the Tänassilma samples. This can be explained by its location and long-term loading with municipal wastewater, which is mixed with diffuse runoff from roads and streets.

Conclusions

The total biomass of *T. latifolia* in the Tänassilma semi-natural wetland was higher than that in the Põltsamaa and Häädemeeste CWs.

In free water surface wetlands plants grow and die more quickly than in subsurface wetlands. Therefore, significantly less spadixes and litter were found in Tänassilma than in Põltsamaa and Häädemeeste. The phytomass values did not display a correlation with N, P and C contents in plant tissues.

The N and P contents in plant fractions demonstrated that these nutrients were stored in reserve organs after the fruiting stage, while carbon storage had not been observed. In Põltsamaa, the N, P and C content in all plant fractions was higher than in other test areas.

Whereas heavy metal concentrations in cattail are generally low, zinc was found at relatively high levels. Apart from Cd, the heavy metal concentration in plant tissues from Tänassilma was significantly higher than at other sites.

Regular harvesting of above-ground biomass, especially spadixes for building material or energy biomass production, can help remove a significant portion of nutrients and heavy metals.

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The use of overland flow in the peat production areas of Vapo OY in Finland

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Vapo Oy Energia is the leading supplier of local biofuels in the Baltic region and the most important producer of peat in Finland, Sweden and Estonia. In addition to the fuel peat and fuel wood, Vapo Oy produces heat, bioelectricity and wind power. In 2002, the fuel peat production in Finland covered 42 000 hectars. The total production of peat was 26.1 million cubic meters, of which 21.2 million cubic meters were milled fuel peat, 2.2 million cubic meters were stock peat, and 2.7 million cubic meters were other kinds of peat.

In its environmental policy, Vapo Oy has commited itself to continual improvement of the control of environmental matters. According to this principle, Vapo Oy has been active in developing and implementing new and effective solutions for water protection in the peat production areas.

A general description on the overland flow method

Overland flow is one method which can be used for the cleaning of the drainage waters of peat production. It is mainly used during the frost free season, but this method can also be used all year round. The method is commonly used, and experiences of using it have been gathered from the oldest fields during 15 years. Finnish authorities have approved overland flow as the best available technique (BAT).

In overland flow, the drying waters of a peat production area are grouted into the surface layer of a bog in the natural state. The vegetation of the surface layer acts as a mechanical filter into which the solid matter and sludge are caught. Due to chemical and biological processes, the soluble nutrients stay in the peat layers. Water flows down to the depth of about 0.5 meter. The operation of an overland flow field can be compared to the normal flow of water in the natural bogs and to its cleaning in the ground.

Description of operation

The water is usually led to the overland flow field via a settling or a pumping pond. For the sake of functionality, even spreading of the water over the entire field is essential. The water is distributed to the field by means of a distribution ditch or pipings. The water-absorbent area can be limited by ditching or embankments, if necessary. Crosscut flows, if any, must be prevented.

In case the natural differences in altitude are not sufficient, a pumping station must be built to lift the water to the distribution ditch. A pump will also even out the volume of water coming to the field by damming off some water in the ditch systems of the production area and in the pumping pond.

After the overland flow, the water can be collected into a collection ditch, which can be, for instance, an old forest ditch. A collection ditch is not needed in case the volume or quality of the water coming from the field does not require monitoring.

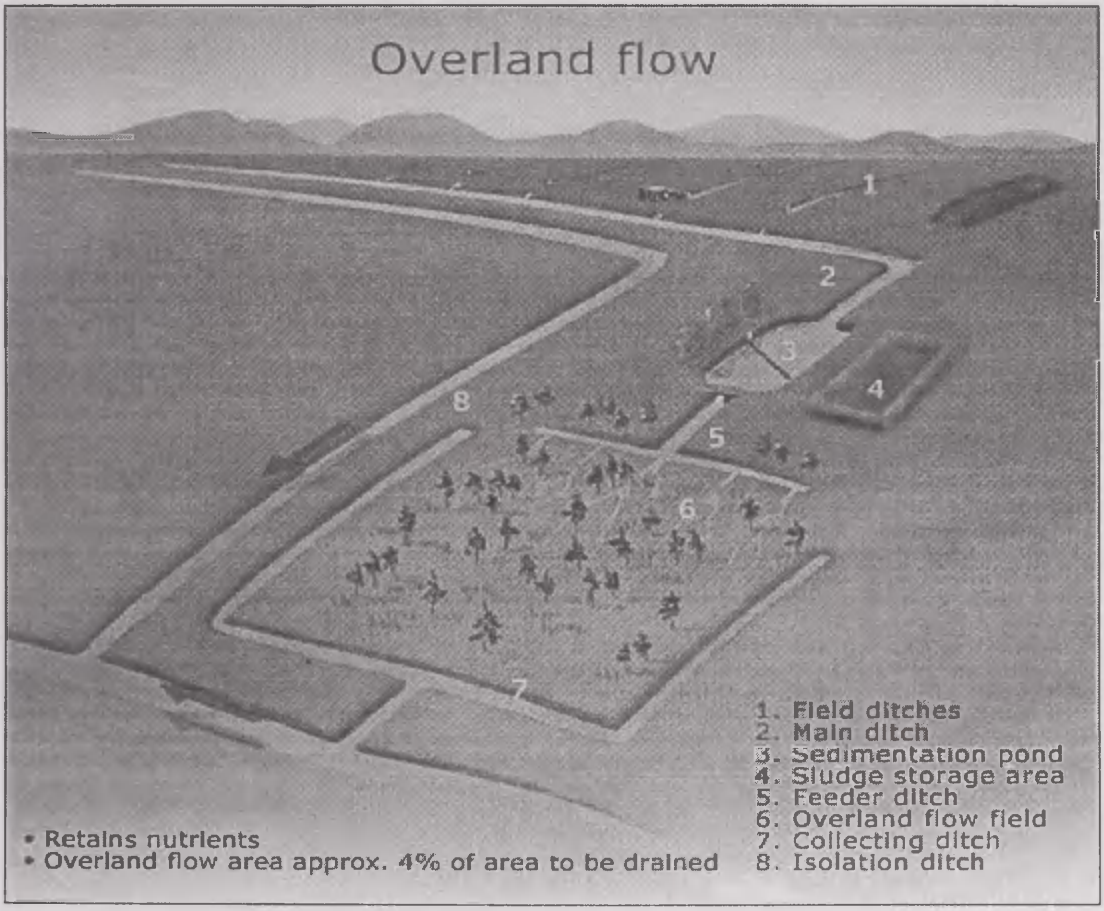

Figure 1. Operating Principle of the Overland Flow System

The overland flow fields are usually only in use during the frost free season, but they can also function all year round, especially if the water can be led to the field without pumping.

Founding and dimensioning

The overland flow field is founded in a bog area as natural as possible, but functional fields have also been built in old forest drainage areas.
The overland flow field is dimensioned bearing in mind the following target values:

- The size of the field is at least 3.8% of the drainage area.
- The length in relation to the width of the field 1:2.
- Recommended inclination 0.2–1%.
- The inclination must be the same over the entire field.
- The minimum peat thickness 0.5 meter.
- The peat layer must be uniform in structure, a little decomposed sphagnum peat or carex peat (H1-H3).
- The hydraulic load must not exceed 340 m^3 ha^{-1} d^{-1}.

The dimensioning criteria are not always met, but the field must be designed according to the properties of the area available. The use of the method is limited by the availability and purchasing of suitable bog areas, especially in old production areas.

Cleaning efficiency

A functional overland flow field reduces the solid matter load during the frost free period by 55–92% in the average, the total nitrogen load by 49%, the ammonium nitrogen load by 79%, the nitrate nitrogen load by 41%, the inorganic nitrogen load by 63%, the total phosphorus load by 46%, phosphate phosphorus load by 51%, and the iron load by 30% (Ihme, 1994).

The building costs of an overland flow field based on gravitation are about 100 € ha^{-1} per production hectare, and the operation and maintenance costs about 5 € ha^{-1} per annum.

The building costs of an overland flow field operated by a pumping station are about 430 € ha^{-1} per production hectare, and the operation and maintenance costs about 23 € ha^{-1} per annum.

The subcontracting costs of an overland flow field are about 10 € production $hectare^{-1}$. Standing wood in the area may increase the price.

The overland flow fields of Vapo OY

The drained water of 14 200 hectars, i.e. 25% of the Vapo Oy production areas, are treated by means of overland flow. 148 fields in total have been built. The average field is 7.8 ha in size. The fields function well. Table 1 gives a summary on the results of summer time load monitoring of the Vapo Oy fields in the Northern Finland in 1992–2002.

Table 1. Average water quality and specific load of Vapo Oy overland flow fields in 1992–2002, in the Northern Finland. 54 monitoring points and 1362 samples. As comparison, the water quality results of the Primrose target, Kompsasuo bog.

	Water quality			Specific load (gross)	
	Average 1992–2002	Kompsasuo 1992–2002	unit	Average 1992–2002	unit
Suspended solids (SS)	5.8	2,4	mg l^{-1}	40.0	g ha^{-1} d^{-1}
Chemical oxygen demand (COD_{Mn})	40	21	mg l^{-1}	300	g ha^{-1} d^{-1}
Total phosphorus (P_{tot})	54	38	ug l^{-1}	0.39	g ha^{-1} d^{-1}
Phosphate phosphorus (PO_4-P)	18	20	ug l^{-1}	0.13	g ha^{-1} d^{-1}
Total nitrogen (N_{tot})	1368	720	ug l^{-1}	11.6	g ha^{-1} d^{-1}
Ammonium nitrogen (NH_4-N)	205	16	ug l^{-1}	2.13	g ha^{-1} d^{-1}
Iron (Fe)	3175	1854	ug l^{-1}	23.2	g ha^{-1} d^{-1}
Mean run off (Mq)		7,3	l s^{-1} km²	10.8	l s^{-1} km²

In order to measure the water quantity, Vapo Oy uses a pressure sensor and datalogger connected to the measuring weir. In 2002, 63 measuring devices were in use in the overland flow fields, and 146 measuring devices in total.

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The use of reed and cattail produced in constructed wetlands as building material

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Reed from wetlands is already a well-known building material. It is most widely used for roofs, but insulation plates made from reed are also highly valued in ecologically-oriented construction. Likewise, cattail (*Typha latifolia*) biomass can be easily used in various construction work. The leaf mass of cattail shows a high porosity and elasticity of the arenchyma tissue. At the same time, there is a uniform distribution of blast fibres. Thus, the leaves have a high stability and show excellent insulation qualities. Furthermore, the leaf tissue has a high content of polyphenols, so the dry raw material shows a high resistance to decay.

Table 1. Relation between lightweight clay material weight and thermal conductivity.

Weight; kg m^{-3}	Thermal conductivity; W m^{-1} K^{-1}
300	0.1
400	0.12
600	0.17
800	0.25
1000	0.35
1200	0.47

Figure 1. *Typha* leaves and stems in constructed wetland in wintertime before harvesting (left). Finished insulation blocks (right).

Cattail chips mixed with clay are used in the production of healthy and cost-efficient building blocks. The material is light and has good thermal insulation properties. The lighter the material, the better its insulation properties (Table 1). Wood chips are commonly mixed with clay pulp and dried. Clay content in the material is around 30%.

Our experience shows that using cattail chips instead of wood gives the material much better properties. It has less weight (as cat-tail plant tissue is very porous) and very good physical properties.

The basic advantage of this technology is that the resulting house is considered very healthy because of the very stabile humidity in rooms throughout the year. Another positive aspect is that the material has a very low energy content. No fossil energy is used for drying. Primary energy content in local clay material is around 5–30 kWh m^{-3}, whereas the same number for concrete material is 3000 kWh m^{-3}.

Figure 2. An experimental house insulated with clay blocks in Estonia (year of construction 2001). Wood covers the house on the outside.

One hectare of constructed wetland can annually produce enough raw material (15–30 t) for the insulation of 1–2 houses.

The aboveground biomass will be harvested in winter when the leaf mass has a minimum water content. The harvesting technique has been adopted from a common reed harvester.

Another interesting feature of cattail is its extremely lightweight fibre, which is positioned around the seeds. Approximately 1cm-long hairs are porous and have good insulative properties. One hectare of constructed wetland produces ca 2 t of this material.

Figure 3. Constructed wetland with cattails in autumn. The seeds are ready for harvesting (left). Fine fibre material after initial treatment (right).

Experiments show that the fibre is excellent material for clay plaster reinforcement. Natural clay is a highly valued material in finishing, but in practice it is difficult to attain high-quality surfaces (as water evaporates it causes cracks). The fibre is an ideal material to avoid this problem. Approximately 2–5% (weight) of "wool" is added to the clay-sand mixture. Ready-made dry mixtures of reasonable quality can be produced and sold in the market.

Our experience shows that 1 hectare of wetland can annually produce fibre material for 4–6 houses.

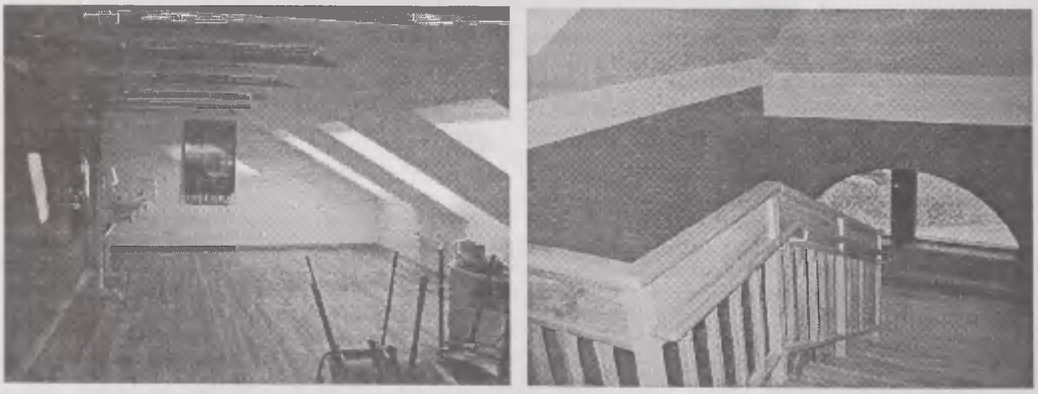

Figure 4. Rooms finished with *Typha*-fibre reinforced clay plaster.

This material has shown an ability to compete in the market. *Typha*-fibre is unique in giving the material high quality. Thus the cultivation of the plant in CWs can be a relevant aspect of wetland management.

Figures 1–4 show various qualities and uses of cattail as a building material.

Application of the static chamber method for gas emission measurement from a constructed wetland

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Both carbon dioxide and methane are considered to be gases that play a key role in promoting the greenhouse effect. These gases are commonly produced by constructed wetlands. Carbon dioxide release from the wetland bed occurs due to the respiration and fermentation processes in aerobic and anaerobic zones of wetland. Methane emission by methanogenesis is observed widely over the whole wetland, indicating reactions taking place in the unaerated zone. Investigation of the rate of emission of gases can be useful in the quantification of carbon budget and the estimation of biochemical reactions occurring in the wetland bed.

There are numerous methods for gas emission quantification, although the application of a static chamber is the most useful. Our chambers were prepared in the workshop in the shape of cut cones made of stainless steel with two vents on the cover to permit the flashing of the chamber with other gases (e.g. Helium). Chambers are sealed with the wetland bed with a water gasket and have a volume of 50 l and an area of 0.25 m^2, and can detect flowrates in the range of 0.1 μmol $m^{-2} h^{-1}$ with satisfying accuracy.

Samples were collected every 10 minutes in gas-tight syringes (50 ml volume) and measured within one hour using the gas chromatography technique. We were using portable GC (SRI-Tech) equipped with FID and TCD detectors. Reproducibility of CH_4 analysis was estimated at a level of 0.1%, and CO_2 analysis was performed to an even better precision (0.05%).

All of the analyses were calibrated against the NOAA mole fraction scale via several secondary standard gas mixtures. The results of the single set of five samples were analyzed using linear or exponential regression, which in turn give an estimation of flowrate and its uncertainty. The statistical uncertainty of the emission fluxes were usually below the limit of a few percent. In some cases, however, this increased to 20%. In later experiments, the sampler operator was standing far away from the chamber, thus helping to avoid mechanic disturbance of the wetland bed.

The constructed wetland in Nowa Slupia was chosen as the location for the study. The wetland was built in 1995 in order to treat municipal wastewater. It is located in the Holy Cross Mountains region (21°05' E - 50°52' N) at 250m a. s. l.

with an annual mean temperature of 8°C and an annual precipitation of 700 mm. The wetland consists of three parallel gravel beds (78m x 24m x 1.2m each) overgrown with common reed (*Phragmites australis*). Gas flux measurement was exercised at all three beds independently. It revealed large differences between the beds (by two orders of magnitude). Within the single reed bed, the emission of gases can also vary significantly. A strong gradient of gas production from the entrance of the wastewater to the reed bed toward its end was observed. This is clearly due to nutrient distribution inside the bed.

The results of the two samples performed in autumn and during the summer season are compared in Figure 1 below.

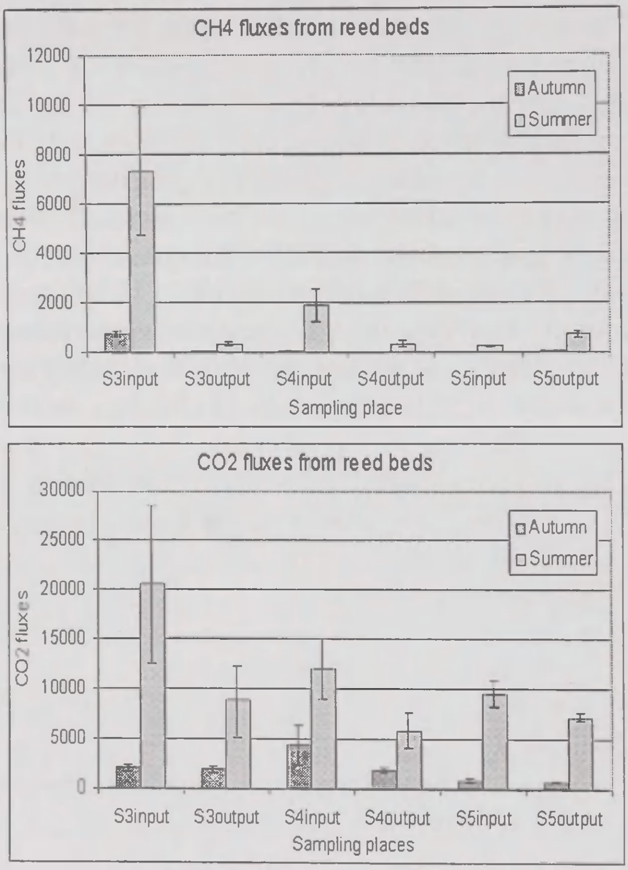

Figure 1. Comparison of gas fluxes from reed beds between autumn (darker bars) and summer (lighter bars) seasons and between the input and output regions of the wastewater to reed bed.

As the duration of the measurement taking covered three days, it was possible to distinguish a slight diurnal cycle of carbon dioxide production inside the bed. During the night-time, flux of this gas decreased to almost zero value and reached maximum values during the late afternoon hours. This diurnal pattern of CO_2

emissions can be explained by changes in the availability of oxygen in the beds, which are in turn related to the photosynthetic activity of plants.

Methane flux can reach extremely high values (up to 4 mmol m^{-2} h^{-1} in the case of our wetland), which is typical of natural wetlands in low latitudes.

Direct vertical profiles of gas concentration in atmospheric air at three elevations were also performed several times. These revealed that a constructed wetland absorbs rather than emits carbon dioxide during the day, according to the assimilation process. CO_2 concentration inside the reed decreases by 100 ppm compared to the surrounding ambient atmosphere.

Acknowledgements

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Aeration effects and the application of the k–C* model in the Kodijärve subsurface flow constructed wetland

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Introduction

The Kodijärve constructed wetland (CW) is composed of a vertical subsurface flow wetland (VSSF, built in summer 2002), a horizontal subsurface flow wetland (HSSF; September 1996) and a phosphorus (P) removal unit (March 2002). The system was designed to purify the wastewater from a hospital for about 40 people. The HSSF system consists of two beds with a total area of 312.5 m^2 (Mander *et al.*, 2001). Before entering the wetland system, the wastewater flows trough a two-chamber septic tank.

Material and methods

As a source for analysis of oxygen demand and aeration capacity, and for compiling an area-based first-order model, the input-output water quality was determined in water samples taken in 63 events from autumn 1997 to spring 2003 (Mander *et al.*, 2003). After the VSSF system was built, seven sampling events were undertaken from October 2002 to April 2003.

Oxygen demand and aeration capacity

The oxygen demand in the wastewater inlet and the system's aeration capacity were analyzed. The analysis showed that in the study period the average oxygen demand of the wastewater entering Kodijärve HSSF wetland was 1.19 kg O_2 d^{-1} (3.2 g O_2 m^{-2} d^{-1}) and the average aeration capacity was 65% of the oxygen demand (0.79 kg O_2 d^{-1}; 2.1 g O_2 m^{-2} d^{-1}). The HSSF system mostly had problems with NH_4-N and total nitrogen (N) removal. The conclusion was that the Kodijärve HSSF system does not have sufficient aeration capacity for effective BOD and nitrogen removal. In order to improve aeration, a VSSF system was designed and constructed in 2002.

Application of the k–C model in the Kodijärve HSSF wetland*

The removal of BOD_5 and total N in Kodijärve was described using an area-based first-order model (Kadlec and Knight, 1996)

$$\ln[(C_o - C^*)/(C_i - C^*)] = - k/q, \quad (1)$$

for total P and NH_4-N the equation is:

$$\ln(C_o/C_i) = - k/q, \quad (2)$$

where: k is the area-based, first-order rate constant (m yr^{-1}), q is the hydraulic loading rate (m yr^{-1}), C_o is the effluent concentration (g m^{-3}), C_i is the influent concentration (g m^{-3}), and C^* is the irreducible background wetland concentration.

Based on the published data, the C^* values of 1 mg l^{-1} for BOD_5 and 1.5 mg l^{-1} for total P were chosen (Kadlec, 2000). It is known that wetlands have very low natural total P and NH_4-N background concentrations, and therefore equation 2 was used for the evaluation of these parameters.

After calculating k values for all four parameters, the dependence of k on hydraulic loadings (cm d^{-1}) and mass loading rates (g m^{-2} d^{-1}) was investigated. In addition, a comparison was undertaken of mean observed concentration profiles (transect data from March-April 2000) with predicted profiles, derived from the k–C* model using mean observed input concentrations and calculated mean k values..

Results and discussion

Oxygen demand and aeration capacity

Aeration capacity depends predominantly on the efficiency of BOD and NH_4-N removal. The oxygen balance in Kodijärve CW has significantly improved due to the VSSF system. From 1997–2002, the average BOD_7 purification efficiency and NH_4-N removal efficiency of the HSSF wetland were 85.9% (±13.7%) and 49.7% (±24.7%) respectively. After construction of the VSSF system, the combined average BOD_7 purification efficiency of the VSSF and HSSF components was 96.7% (96.7% of BOD_7 removed), the BOD_7 purification efficiency of the VSSF system alone was 61%.

Unfortunately, the VSSF system has not yet remarkably raised nitrification efficiency. This is evaluated on the basis of NH_4-N removal efficiency, which was 49.7% before and 55.3% after construction of the VSSF. The NH_4-N purification efficiency of the VSSF system alone was only 6.7%.

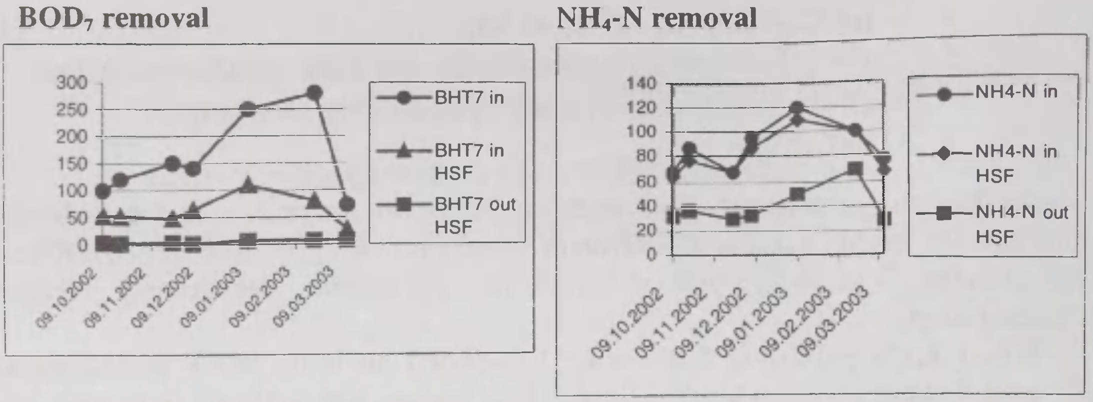

Figure 1. BOD_7 and NH_4-N removal in Kodijärve CW.

Vymazal *et al.* (1998) summarize that nitrification is influenced by temperature, pH, inorganic C source, and concentrations of ammonium N and dissolved oxygen. The minimum temperatures for growth of nitrifying bacteria are 4–5°C. Since nitrifying bacteria are also very sensitive organisms and extremely susceptible to a wide range of inhibitors (Vymazal *et al*, 1998; Laber *et al.*, 2003), there are two main reasons, why nitrification may not be so effective:

- sensitive nitrifiers are inhibited by various household chemicals (such as disinfectants for toilets);
- the extremely cold winter of 2002/2003 inhibited the development of nitrifying bacteria communities

Application of k–C* model in Kodijärve HSSF wetland

Table 1 shows that mean *k* values in Kodijärve are much lower than reported in the literature. The HSSF system in Kodijärve has been working under its design conditions: mean observed inflow is 3.2 m^3 d^{-1}, but the designed inflow is 10 m^3 d^{-1}. Therefore the observed mean hydraulic loading is also much smaller (0.68 cm d^{-1}) than reported in the literature.

Table 1. Mean area-based, first-order rate constant k (m yr^{-1}) in Kodijärve, and values from Kadlec and Knight (1996).

	Mean *k* values in Kodijärve	Literature-reported *k* values
BHT_5	5.80	180.00
Total P	4.90	11.70
Total N	1.84	15.10
NH_4-N	1.72	7.61

The *k* values for all measured parameters in Kodijärve are dependent on hydraulic loading. The correlation coefficient was always >0.9, except for NH_4-N (0.83). Kadlec (2000) demonstrated that k is quite dependent on hydraulic loading, and thus it was expected that *k* values are low.

In the Kodijärve HSSF system, the mass loading rates (g m^{-2} d^{-1}) decreased from year to year (Mander *et al.*, 2003). At the same time, a significant positive correlation ($p<0.05$) was found between the *k* values of all parameters and their mass loading rates. This also explains the declining trend of *k* values. The correlation coefficient for total P and total N was 0.92 and 0.87 respectively. The dependence between mass loading rates and the *k* values of BOD_5 and NH_4-N was not as strong (correlation coefficient 0.77 and 0.58 respectively). This means that BOD_5 and NH_4-N removal processes in Kodijärve are influenced more by endosystemic conditions (i.e. oxygen requirement) than by mass loading rates. It also demonstrates that the mechanistic input-loading correlation analysis can cause misleading interpretations (see Mander and Mauring, 1997)

Table 2. Application of the k–C* model in the HSSF part of the Kodijärve CW

Parameter	R^2 in right bed	R^2 in left bed
BOD_5	0.82	0.88
Total P	0.93	0.94
Total N	0.79	0.90
NH_4-N	0.77	0.82

R^2 – coefficient of determination

The HSSF part of the Kodijärve CW is unable to supply enough oxygen for the incoming wastewater, satisfying only 65% of the requirement. This causes insufficient mineralisation of organic nitrogen, and thus it may be reasoned why *k* values of total N and NH_4-N are lower than reported in the literature (Table 2). Likewise, this is valid for the *k* value of total P, because in anaerobic conditions ferric iron(III)phosphorous oxide releases back to water in forms of Fe^{2+} and PO_4^{3-}.

Mean observed concentration profiles were compared with predicted profiles to evaluate k–C* model fitting in the Kodijärve HSSF system, although transect data already (March-April 2000) supported the first-order kinetics of pollutant removal in the system. These results are presented in Table 2, which indicates that the area-based first-order model relatively effectively describes pollutant removal processes in the system.

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Microbial characteristics and nitrogen transformation in a planted soil filter treating domestic wastewater

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Introduction

Water purification in constructed wetlands is mediated by a multitude of biogeochemical processes. The microbial consortia have the key role in nutrient transformation and retention in the wetlands. Thus the investigating of the microbial characteristics of the wetlands is crucial for a deeper understanding of the processes of ecotechnological water purification and the sophisticated dimensioning of individual systems.

Subsurface-flow soil filters have no free water surface and turbulent water mixing. The lack of water mixing allows persistent O_2 gradients and nutrient gradients caused by microbial as well as physical and chemical processes. These gradients are localized in the direction of the O_2 diffusion into soil and wastewater seeping through the wetland, which makes them easily measurable, in contrast to open-water wetlands. Sand as a filter material is a homogenous environment that lowers site-specific random fluctuations in the measurable microbiological characteristics caused by non-biological factors. We assumed that these conditions can give good spatial correlations of water detention time and resulting site-specific water quality indicators in soil filter with microbiological characteristics of wetland soil.

The purpose of this study was to estimate microbial characteristics and the activity of nutrient transformation processes in the constructed wetland. We were also interested in the relationships between measured microbiological parameters and water quality indicators, physicochemical factors, and gas emissions from the wetland.

Material and methods

Site description

The Kodijärve planted sand filter was constructed in October 1996 by the Tartu Centre for Ecological Engineering in order to purify wastewater from a boarding house for about 40 persons. The constructed wetland is located 120 m downhill of the house on the shore of the lake called Väike Kodijärv (3.5 ha). Before entering the system, wastewater flows through a two-chamber septic tank. The soil filter consists of two beds (each 25x6.25x1 m) filled with coarse sand. The beds are called a dry bed and a wet bed due to the different water levels therein. Wastewater enters the filter over two Thompson weirs and seeps through the inlet pipes into the sand of the beds. Two outlet pipes between the two beds collect purified water to the outlet well. The standpipe in the outlet well allows the regulation of the beds' water levels. Both beds are isolated from the soil with PVC foil. 18 water-sampling wells, 9 in either bed, are distributed evenly throughout the sand filter for spatial water sampling for analyses. In 2001 the dry bed was dominated by wood club-rush (*Scirpus silvaticus*) and the wet bed had wood club-rush and patches of reed (*Phragmites australis*).

Field work

The soil samples for microbiological analyses were collected at the same time as measurements of other characteristics of the wetland, data of which are presented by Mander *et al.* (2003).

Soil samples for microbiological and chemical analyses and water samples from water sampling wells were collected on 4 October 2001. Sampling for the analysis of N_2O, CH_4, CO_2 and N_2 emissions with the "closed chamber" method and He-O method was done a day later (Mander *et al.*, 2003). Soil samples for microbiological analyses were collected randomly around the individual water sampling wells (well nos.: 1, 3, 4, 6, 7, 9 in dry bed and 10, 12, 13, 15, 16, 18 in wet bed) of both beds at two depths: 0–10 cm ("upper layer") and 20–30 cm ("lower layer") with a soil core drill.

Microbiological methods

Soil samples for microbiological analysis (except for Biolog microplates and potential nitrification measurements, which were stored at 4°C) were stored in plastic boxes at room temperature. Biolog-plates were used to differentiate the microbial consortia of individual soil samples based on the intensity of average well colour development of 3 replica of 31 wells caused by the growth of microbes during incubation. Soil respiration (basal respiration) was determined with a method called soil respiration by titration, microbial biomass was determined with a method of substrate-induced respiration (SIR) and microbial nitrogen or immobilized nitrogen was measured using the fumigation-extraction technique coupled with ninhydrin-reactive nitrogen (NH_2-nitrogen) detection. In addition, potential nitrification was measured from collected samples (Schinner *et al.*, 1996).

Data analysis

All microbiological parameters were log-transformed prior to analysis. Statistica software was used for the statistical analyses. All of the microbiological characteristics were compared between two beds with a t-test. The respective data from two layers and two rows were analysed with a paired t-test, and data between three transects were analysed with ANOVA in either bed separately. Spearman rank correlation coefficients were determined between the microbiological data, water quality data and the data of gaseous emissions from the wetland. Biolog microplate data were analysed additionally using principal component analysis and the Shannon index of diversity. The structures of micro-bial consortia in both beds were compared using a multivariate randomization test.

Results

The measured microbiological characteristics did not, except for the Biolog microplate data, differ between the two beds. At the same time, we found significant differences within individual beds. We conclude that the variability of conditions that affect the measured characteristics is greater on a small scale than between the average values of the beds. The different spatial pattern of the measured microbiological characteristics in both beds indicates spatial differences in soil conditions. The Biolog microplate analysis showed that the microbial activity of the wet bed was significantly higher than in the dry bed, and significantly higher in the upper layer than in the lower layer in the dry bed. The Microbial community of the wet bed was in a more active state, most likely due to the better availability of nutrients caused by the higher water table. This conclusion is supported by the fact that nitrogen immobilization and BOD_7 concentrations were positively correlated ($r=0.85$, $p<0.05$) in the wet bed, and were not in the dry bed. The higher activity in the upper layer (significant within the dry bed) was caused by the better availability of nutrients from plant remains.

We conclude that part of the nutrient availability caused by the nutrients from plant remains was not distinctive in the wet bed, where the majority of the nutrients came directly from wastewater. We also found that the structure of microbial community was different between the two beds ($p<0.01$). There was a higher variability in the structure of microbial community in the dry bed than in the wet bed. The metabolic diversity of microbial consortia was higher in the wet bed. Greater structural differences in the dry bed are caused by the bigger difference between conditions in individual sites.

The total amount of immobilized nitrogen for the entire wetland was 5.983 kg, i.e. 8.53% of the total amount of nitrogen in the wetland soil (69.8 kg). The total amount of biomass-carbon for the entire wetland was 46.2 kg, 3.95% of the total amount of soil carbon in the wetland soil (1167 kg). We found that nitrogen immobilization was higher in the upper layers of the wetland ($p<0.01$ in the dry bed and $p<0.05$ in the wet bed) and negatively correlated with water table height ($r = -0.82$, $p<0.05$). The rapid growth of soil N to immobilized N ratios, when moving from the topsoil to deeper layers, occurred in the first 30 cm. This suggests that nitrogen immobilization in deeper layers of Kodijärve HSSF was limited by the availability of oxygen for carbon mineralization.

Nitrogen immobilization was positively correlated with soil N ($r=0.73$ in the dry bed, $r=0.61$ in the wet bed; $p<0.05$), soil C ($r=0.67$ in the dry bed, $r=0.74$ in the wet bed; $p<0.05$) and BOD_7 concentrations in the water sampling wells of the wet bed ($r=0.85$, $p<0.05$). The comparison of Kodijärve HSSF with the other HSSF (Nguyen, 2000) shows that nitrogen immobilization (8.5% of total soil nitrogen in Kodijärve and 0.67–2.14% in the other HSSF) is higher in the Kodijärve HSSF. We presume that this difference is caused by the higher availability of mineralisable carbon (soil C to N ratio was 16.5 in Kodijärve and 9.0–12.0 in the other HSSF) in Kodijärve HSSF.

We found a strong positive correlation between biomass carbon and N_2 emissions from the wetland in the wet bed ($r=0.92$, $p<0.05$). Biomass-C was correlated with CO_2 emissions from the wetland in the dry bed ($r=0.60$, $p<0.05$). These differences between beds could be explained by differences between water table height, which supports anaerobic processes in the wet bed and aerobic processes in the dry bed.

Soil respiration, based on measured basal respiration, was 433.5 and 431.7 kg ha^{-1} d^{-1} in the dry bed and wet bed respectively. The actual CO_2 emissions from the wetland (measured using the closed chamber method) were 173.7 and 131.8 kg ha^{-1} day^{-1} in the dry bed and wet bed respectively. Soil respiration was negatively correlated with CH_4 emissions in both beds ($r=-0.81$, $p<0.05$ (dry bed), $r=-0.64$, $p<0.05$ (wet bed)). The negative correlation between respiration and CH_4 is due to the different needs for aeration of respective processes.

Potential nitrification was positively correlated with soil C ($r=0.83$, $p<0.05$), soil N ($r=0.76$, $p<0.05$), soil P ($r=0.69$, $p<0.05$) and immobilized N ($r=0.63$, $p<0.05$), and negatively correlated with water table height ($r=-0.81$, $p<0.05$). The positive correlation with soil C shows that measured potential nitrification could not primarily be attributed to autotrophic nitrification. Also, the findings of other investigations support the hypothesis that heterotrophic nitrification is prevalent in this wetland (Mander *et al.*, 2001).

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Lemnaceae in wastewater treatment – case study, Mniow, Poland

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Introduction

Different techniques using hydrophytes for wastewater treatment are recommended, one of which is a method based on floating plants. Throughout the world, twelve species of pleustonic plants have been tested for use in constructed wetlands (Gutenspergen *et. al.*, 1989). Of these, the water hyacinth is ideal, as it can increase at a phenomenal rate, i.e. 20–40 tons of its fresh weight could be harvested per hectare per day (Wolverton and McDonald, 1975). Use of *Eichhornia crassipes* is limited to warm climates. Like the water hyacinth, *Lemnaceae* float on water and absorb nutrients directly from it, and thus they application for wastewater treatment is promising. *Lemnaceae* have a wider range of distribution, have succeeded in colonizing nearly all climates and regions of the world except areas with cold summer climate. The main limiting factor for the whole family of *Lemnaceae* is at least 50 days with a mean temperature above 10°C (Landolt, 1982). Five species of *Lemnaceae* are native to Europe: *Spirodela polyrrhiza*, *Lemna trisulca*, *L. gibba, L. minor* and *Wolffia arrhiza*, and three taxa are reported to have been introduced: *S. punctata*, *L. aequinoctialis* and *L. minuscule* (Landolt, 1986). In Poland *L. minor* is the most common.

At present fourty "Lemna Systems" are operating in Poland. The designers of this system assume that plants treat wastewater directly through nutrient uptake and accumulation in their tissue, and indirectly by creating environments for nitrifying and denitrifying bacteria by the stratification of oxygen in the water column. Additionally, *Lemnaceae* can eliminate smelt, algae by shading, and mosquitoes by forming a physical barrier for their larvae (Poole, 1996). Among Polish scientists and managers, there is no consensus whether this system is useful or not in cold climates. The efficiency of this system depends on a whole biocoenosis (not only a plant community) developing in a pond with *Lemnaceae* (Ozimek, 1996). Thus the questions were – what do *Lemnaceae* species dominate in "Lemna System" wastewater treatment plant, what are the dynamics of their density and biomass in the course of year, what animals can live in this system and what role can they play?

Study site and methods

The case studies were done in the "Lemna System" built according to the specifications of Lemna Corporation. The system was built in 1993 in order to treat municipal wastewater. The designed average diurnal flow of this system is 150 m^3 d^{-1}. The system consists of two ponds in series, first an aerated pond and a second covered with a floating plant of the family *Lemnaceae*. The aerated pond is 0.2 ha in area and is 3m deep. The area of duckweed pond is 0.26ha, it is 2.4m deep and there are floating plastic barriers used for control of the duckweed.

Investigations were carried out from May to November in 1997 and from December 2001 to May 2003. Temperature, conductivity, dissolved oxygen, pH, BOD_5, COD, Total Kjeldahl nitrogen, oxygen concentration, ammonia nitrogen, nitrate nitrogen and phosphorus phosphate, area overgrown by floating macrophytes, species composition, density and biomass of plants were measured on a monthly basis, and a qualitative analysis of the taxa composition of animals were carried out in May and August 2002.

Results

Physical and chemical parameters

In the pond with *Lemnaceae*, the surface water temperature ranged from 0 to 23°C in 1997 and from 0 to 22°C in 2002, reaching the highest value in August 1997 and in July 2002. Differences between surface and near bottom temperature were ca 3°C during all months. Selected parameters are presented in Table 1 (1997) and Table 2 (2002)

Table 1. Selected physical and chemical parameters in 1997.

	pH	Conductivity [$\mu S\ cm^{-1}$]	N-NH4 [$mg\ dm^{-3}$]
	Inlet to aerated pond		
May 1997	7.07	1042	5.39
August 1997	7.11	1042	4.71
November 1997	7.11	1042	4.50
	Inlet to Lemna pond		
May 1997	7.18	1011	3.17
August 1997	7.18	1016	2.99
November 1997	7.11	1016	2.72
	Outlet from Lemna pond		
May 1997	7.00	995	3.32
August 1997	7.21	1016	2.94
November 1997	7.11	1047	2.68

Table 2. Selected physical and chemical parameters in 2002.

	pH	Conductivity	N-NH4	TKN	P-PO4	TP	SS	COD	TOC	BOD_5
		µS cm^{-1}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}	mg dm^{-3}
Inlet to aerated pond										
January 2002	7.45	540	1.06	3.5	2.17	4.44	39.40	210.00	10.30	36.68
March 2002	7.56	660	12.90	38.2	8.69	9.36	228.75	335.00	18.25	199.04
April 2002	7.65	840	19.43	58.2	8.42	12.92	160.00	495.00	17.85	86.36
June 2002	7.21	743	10.25	30.8	9.23	14.30	83.50	295.00	17.50	51.32
July 2002	7.48	713	14.48	43.5	8.87	14.77	106.27	166.67	24.47	82.77
September 2002	7.51	740	13.07	39.2	6.79	12.36	36.75	110.00	21.50	30.00
December 2002	7.55	567	11.66	38.1	6.19	12.78	36.75	135.00	24.15	37.07
Inlet to Lemna pond										
January 2002	7.48	772	14.99	44.2	8.42	10.00	23.50	150.00	16.00	15.18
March 2002	7.85	610	10.07	30.5	8.15	9.18	13.00	50.00	14.10	2.42
April 2002	8.17	720	14.31	42.6	8.42	12.92	28.75	205.00	15.30	5.48
June 2002	7.57	719	7.59	23.0	3.53	6.42	37.50	120.00	19.55	24.52
July 2002	7.53	662	12.72	38.1	8.51	14.55	19.50	40.00	23.33	32.00
September 2002	7.72	675	13.78	41.2	7.61	11.12	70.00	150.00	23.00	61.20
December 2002	7.99	597	13.07	39.8	4.82	10.14	28.50	115.00	21.25	54.19
Outlet from Lemna pond										
January 2002	7.49	820	15.54	47.0	7.88	10.00	14.50	165.00	16.00	11.78
March 2002	8.13	580	7.95	23.1	6.79	9.19	7.75	35.00	14.25	0.22
April 2002	8.36	670	11.66	34.7	6.52	10.00	24.75	180.00	14.50	1.30
June 2002	7.52	765	10.78	31.9	5.97	9.65	17.25	100.00	18.40	4.92
July 2002	7.54	551	10.32	30.8	4.89	7.63	22.67	30.00	20.03	6.24
September 2002	7.48	562	12.54	37.5	5.70	9.58	19.75	40.00	16.00	5.30
December 2002	7.62	575	12.72	39.5	6.45	13.45	20.75	50.00	20.05	0.87

In 1997 a roughly twofold reduction in ammonia nitrogen concentrations and only a slight reduction in phosphorus phosphate concentrations in wastewater were found compared to wastewater in the inlet and outlet of the aerated pond. No further reduction of ammonia nitrogen and phosphorus phosphate occurred in the pond with *Lemnaceae*. In the warm months of 2002, SS, COD and BOD_5 were removed efficiently. Removal of nutrients was generally poor in both aerated and *Lemna* ponds. Theconcentration N-NH4 and P-PO4 at the outlet from *Lemna* pond increased threefold from 1997 and 2002. The oxygen concentration in surface water under *Lemnaceae* mats was very low (about 1mg dm^{-3}), and was ten times lower than in surface water under thick mats of *Lemnaceae*.

Plants

Two species of *Lemnaceae* – *Lemna minor* and *Spirodela polyrrhiza* formed the plant community in the "Lemna system" pond in Mniow. L. Minor was dominant, i.e. its contribution to total biomass was ca. 95% during the whole investigated period. The area covered by plants changes during the growing season. In May 1997 only 30–40% of water surface was overgrown by plants. In the middle of June about 0.5t fresh weight of *Lemnaceae* was introduced into the system from natural stands. After that, duckweeds occupied ca. 75–99% of the water surface. Dry weight per m^2 was highest in June (122±22 g m^{-2}). In 2002 the area covered by plants varied from 1% in April to 90% in July. The highest dry weight per m^2 was in August (305±82 g m^{-2}).

Animals

A relatively high diversity of fauna was observed in the pond that had been planted with Lemnaceae. Of verterbrates, *Rana ridibunda Pallas* was very common. In the introductory qualitative analysis performed in August 1997, 17 species of invertebrates and undetermined to species of *Ostracoda* and *Cyclopoida* were recorded (Koperski, unpublished data). Zooplankton and fauna dwelling with macrophytes were the most rich and abundant. Bottom fauna was very pure because of anoxic conditions. Among invertebrate species, *Cataclysta lemnata* is most closely associated with *Lemnaceae* by consumption of their tissue and case-building activities using the fronds of duck weeds.

In May and June, when only 30–40% of the pond's surface was covered by plants, an abundant occurrence of *Culex sp*. larvae was noted. They were not recorded in the rest of the vegetation season.

Discussion

The density and biomass of *Lemna minor* was quite high in the "Lemna System" wastewater treatment plant in Mniow, except for the beginning of the growing season. In spring, the decreased density and biomass of plants, as well as limited pond area covered by plants indicated that only small part of *Lemna* population did survive winter conditions. The growth rate and reproductive rate of *L. minor* were very low because of the low temperature. The optimal temperature for *Lemna minor* growth ranges from 20 to 25°C. (Landolt, 1986). It is necessary to introduce plants into "Lemna System" ponds in spring. After the introduction of *Lemnaceae* from natural waters, their biomass increased rapidly. Biomass and population density were stable for the rest of the season. Biomass was in dynamic equilibrium. The growth rate attended logistic phase affected by limiting factors, e.g., area and light.

The maximum density and biomass of *Lemna minor* in Mniow was ten times greater than reported for natural eutrophic waters in a cold climate (Kobuszewska, 1973), but similar to the value found by Ozimek (Ozimek, 1983), for ponds supplied by post-sewage water, and lower than that reported for natural waters in a warm climate (Landolt, 1982). The role of plants in nutrient cycling is proportional to their biomass. *Spirodela polyrrhiza* did not play a significant role due to its low contribution to total plant biomass.

Why was the relatively high biomass effect of *Lemna minor* on wastewater treatment nevertheless non-significant? At any time during the season, the *Lemna minor* population was comprised of fronds of various physiological ages, as a result of continual vegetative reproduction from the budding of new daughter fronds. A frond lives 10–15 days on average. When the biomass of *Lemna minor* fitted to environmental capacity, nutrients began to recirculate between old and young fronds directly from mother frond to daughter frond or via water (releasing from mother frond to water and taking up from water by daughter frond). A rich community of invertebrates was found in the pond containing *Lemnaceae*. Many animals, especially immature insects, that eat live *Lemna minor* or use its fronds as material to build a cocoon, cause extensive damage. For example, *Cataclysta lemnata* can eat 0.1 g fresh weight per individual per day. (Koperski and Tulczyński, personal communication). Its last larval stages are the most voracious and cause most damage to macrophytes by consumption and cocoon-building activities. Losses of nutrients from plant tissue are even more rapid, since these plants have a more finely dissected structure. During the vegetation season, invertebrates can consume from 2.5 to 20.3% of total plant biomass (Sand-Jensen and Madsen, 1989)

In 1997 0.1t of dry weight was harvested, which means that with the plants, 0.0015t of phosphorus and 0.006t of nitrogen was removed. This represented only 0.2% of the nitrogen and 0.4% of the phosphorus that annually flowed into the

"Lemna System" wastewater treatment plant in Mniów. Plants did not play a significant role in wastewater treatment in this temperate climate. Other roles such as oxygen stratification, smelt and mosquitoe reduction were confirmed.

Acknowledgements

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Nitrogen and phosphorus removal in the reed bed of a horizontal subsurface flow constructed wetland

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Introduction

Constructed wetlands used for wastewater treatment provide a relatively simple and inexpensive solution for the treatment of wastewater for small communities and industries as well as storm sewage and agricultural runoff. In the Czech Republic, the majority of the constructed wetland systems are horizontal sub-surface flow systems and are designed for the secondary treatment of domestic or municipal wastewater. Functioning constructed wetlands are effective, especially in removing organic pollution and suspended solids. Treatment efficiency is high in terms of BOD_5 (88.0% for vegetated beds) and suspended solids (84.3% for vegetated beds). The removal of nutrients is lower for vegetated beds and averages 51.0 and 41.6% for total phosphorus and total nitrogen, respectively (Vymazal, 1998; Vymazal, 2002).. Some constructed wetlands differ in the efficiencies of nitrogen and phosphorus removal, although the basic parameters of these systems are very similar. The factors and the processes affecting nitrogen and phosphorus removal efficiencies have not yet been adequately explained.

Our study focused on the fate of nitrogen and phosphorus in the bed of a newly-constructed wetland. The aim was to determine the forms and concentrations of nitrogen and phosphorus in the inflow-outflow transect and at two depths in the vegetated bed. The results are shown in context, with redox conditions measured *in situ*.

Study site

The study was performed at a constructed wetland with horizontal sub-surface flow in Slavošovice (near České Budějovice and Třeboň), South Bohemia, Czech Republic. It commenced operations in August 2001. Pre-treatment mechanisms consist of a storm overflow, a horizontal sand trap and an Imhoff septic tank. Two vegetation beds are planted with common reed *(Phragmites australis)* in a gravel (1.0–2.0 cm) substrate. Each bed has dimensions of 17 m in length by 22 m in width, with a mean bed depth of 0.9 m. The area of each bed is 374 m^2. There is a

1% slope in each bed. The number of PE is 150 with an area of 5 m^2 per PE. Flow rate at the inflow at one bed fluctuates from 5 to 60 l per minute, which corresponds to a hydraulic retention time of 18 and 1.5 days respectively. The water level in the vegetated bed was kept at 5 cm under the gravel surface.

Methods

All data were collected from July to November 2002. Redox potential and temperature were measured continuously using field computers. Water was sampled at four dates (July 23, August 28, October 2 and November 20). Platinum electrodes and argent-chloride reference electrodes were used for measuring redox potential and Pt100 sensors for measuring temperature. The electrodes were placed in perforated tubes that were positioned at depths of 20 and 60 cm and were also used for water sampling.

Water was sampled from perforated tubes (110 mm in diameter, 5 mm holes), which were installed vertically in a bed, at depths of 20 and 60 cm. The tubes were placed in one gravel bed in three transects oriented from the inflow to the outflow, at distances of 0, 1, 2, 3, 4, 5, 7, 10, 13 and 16 m from the inflow. Water was sampled from two transects and two depths using a syringe and pipe. BOD_5 was measured using the OxiTop system. Total nitrogen and total phosphorus were determined in the water samples using an alkaline persulphate oxidation method. Total N, NO_3-N, NO_2-N, NH_4-N, total P, PO_4-P were analyzed with a flow injection analyzer (FIA).

Results and discussion

The efficiency of organic pollution removal was 76.4%. Average BOD_5 was 21.6 $mg.l^{-1}$ at the inflow (maximum 45.8 $mg.l^{-1}$) and 5.1 $mg.l^{-1}$ at the outflow. The concentrations of total nitrogen and total phosphorus decreased from the inflow to the outflow at the average rate of 72.9% and by 72.0%, respectively. The efficiency of nitrogen removal was high, although the effect of plant roots on the water treatment was negligible because of the low development of the root system in the first vegetation season. In their experiments, Drizo *et al.* (1997) and Zhu and Sikora (1995) found that the presence of plants (common reed) increased nitrogen removal efficiency significantly as compared to an unplanted system. Therefore the efficiency of nitrogen removal may increase at the studied site in the near future if the loading rate, hydraulic retention time and other parameters do not change.

Concentrations of total phosphorus and total nitrogen decreased rapidly within the 5 m zone from the inflow and remained constant through the rest of the transect to the outflow (Figure 1, Figure 2). The high variability of total N and total P concentrations within the first 5 m of the bed was caused by the diluting effect in periods with strong rainfalls. Ammonium was the predominant form of nitrogen, and phosphates the predominant form of phosphorus, in the pore water through the whole transect.

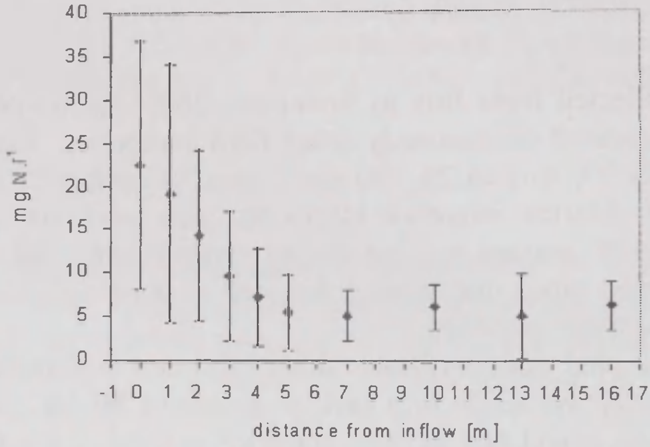

Figure 1. Concentration of total N in the pore water in the vegetated bed in a transect oriented from the inflow to the outflow. The means of four sampling periods from July to November 2002 are shown (mean ± S.D., n=4).

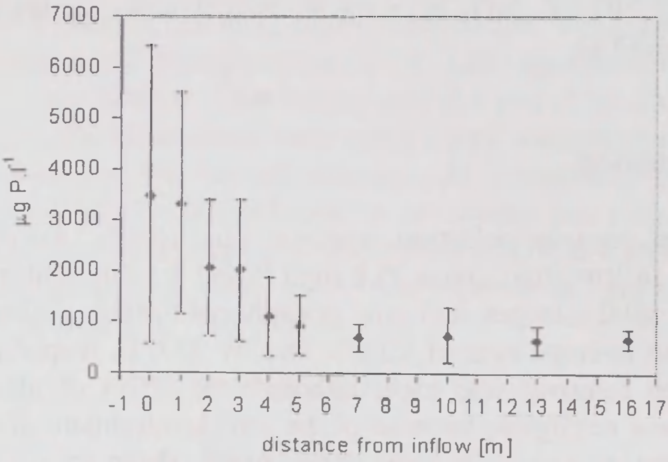

Figure 2. Concentration of total P in the pore water in the vegetated bed in a transect oriented from the inflow to the outflow. The means of four sampling periods from July to November 2002 are shown (mean ± S.D., n=4).

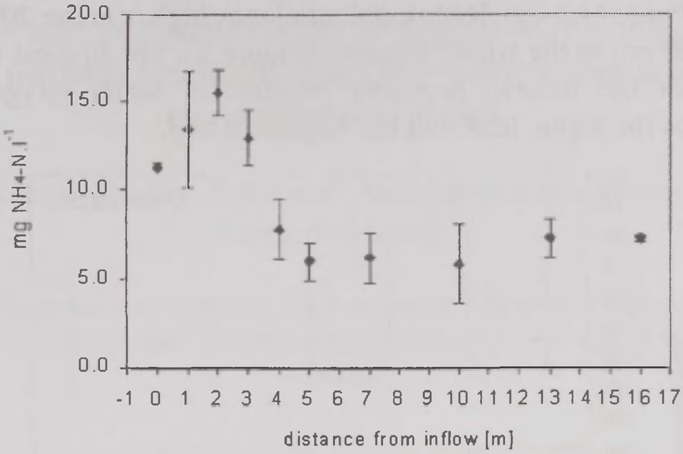

Figure 3. Concentration of NH_4-N in the pore water sampled on 2 October 2002 in the vegetated bed in a transect oriented from the inflow to the outflow (mean ± S.D., n=4).

Domestic and municipal wastewaters do not usually contain high concentrations of organic N, which is readily converted to ammonia (Kadlec and Knight, 1996). The concentration of NH_4-N decreased in the transect from the inflow to the outflow (Figure 3).

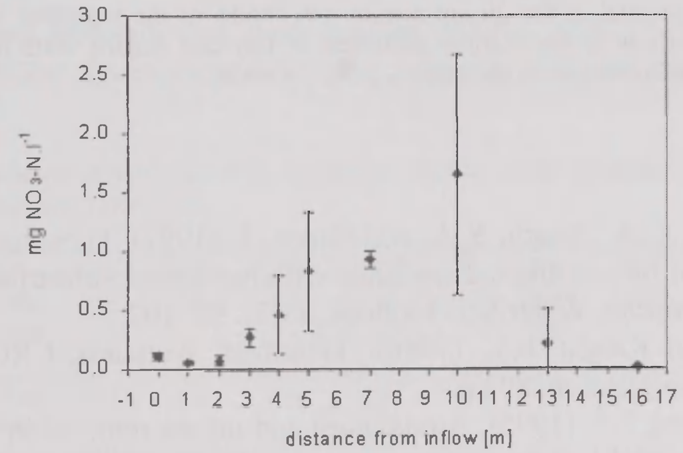

Figure 4. Concentration of NO_3-N in the pore water sampled on 2 October 2002 at the 20 cm depth of the vegetated bed in a transect oriented from the inflow to the outflow (mean ± S.D., n=2).

The concentration of PO_4-P also decreased from the inflow to the outflow. However, concentration of NO_3-N increased in the first 10 m zone from the inflow, then decreased again (Figure 4). NO_3-N concentration was higher at the depth of 20 cm than at 60 cm. Redox potential was higher at the 20 cm depth than at the depth of 60 cm in the whole transect (Figure 5). The highest redox potential was measured at the inflow, probably because of water oxygenation during transport between the septic tank and the vegetated bed.

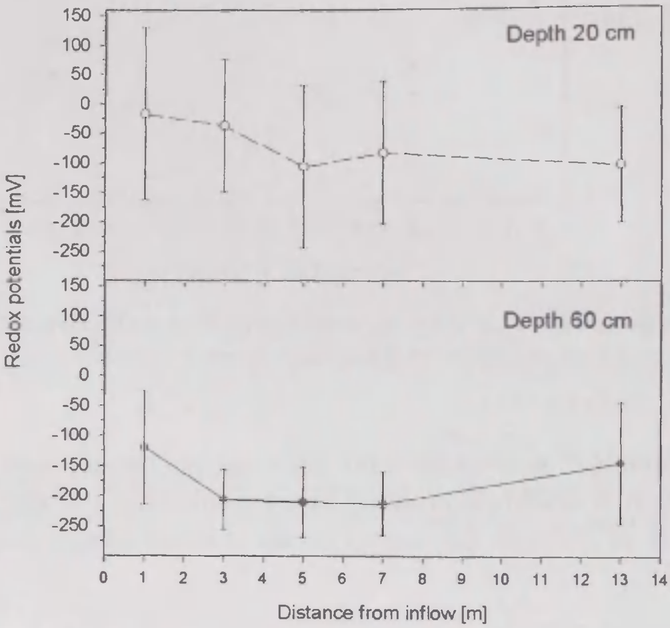

Figure 5. Redox potential at the 20 cm and 60 cm depths of the vegetated bed in a transect oriented from the inflow to the outflow measured in July and August 2002 (calculated to the standard hydrogen reference electrode, mean ± S.D., n = 4000)

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Phytomass and concentrations of phosphorous and nitrogen in three natural wetlands used for wastewater treatment in northern Finland

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Introduction

The goal of this study was to find out the role of the vegetation in phosphorous and nitrogen reductions in wetlands. Specially we were interested in plant uptake: The above- and below-ground plant biomass and the concentrations of P and N in phytomass.

Material and methods

Three constructed wetlands has been studied for the effects of vegetation on processes in wetland. Characteristics of these wetlands are presented in table 1.

Table 1. Characteristics of wetlands with vegetation studies. OGF (Overland and groundwater flow), FSW (Free surface water flow).

Wetland	Wet-land type	Waste water type	Area [m^2]	Depth [m]	Width [m]	Length [m]	V [m^3]	Dis-charge [Q]	Q/A [m/d]	Domi-nant vege-tation	Year of const-ruction
Kompsa-suo	OGF	Peat mining	24000	0,5	110	220	12000	873	0,0364	*Carex*	1987
Lakeus	FSW	Muni-cip.	44000	0–1,2	147	395	9300	3976	0,0940	*Phragm ites*	1996
Ruka	OGF	Muni-cip.	8200	0,3	94	87	2460	283	0,0345	*Carex*	1995

The study included the estimation of the projection coverages of the plant species in the wetlands and in their reference areas, harvesting the above-ground plant biomasses, collecting the below-ground parts (rhizomes and roots) and analyzing the total N and total P of the phytomass. The plant biomasses are harvested and analyzed by conventional methods (Sjörs, 1991).

Kompsasuo

Vegetation at the Kopsasuo wetland was sampled on 13th–14th August, 2002. Vegetation coverages were analysed from twelve 1 x 1m plots from the field and twelve plots from natural wetland just outside of the constructed part. From each plot the above ground biomass of vascular plants was collected from 0.5 x 0.5 m plot and the biomass of bryophytes was collected from the plots of 0.1 x 0.2 m. Below ground biomasses were collected by taking earth (peat) sample with the area of 0.1 x 0.1m and depth 0.2m from each plot.

Lakeus

Vegetation analyses at the Lakeus constructed wetland (CW) was done on 6 plots size 1 x 1m each on 7th–8th August, 2002. Also 3 control plots were analysed on adjacent natural wetland area as reference area. Above ground vegetation biomasses were collected from 0.4 x 0.4 m plots and below ground biomasses were collected from constructed wetland by using a sampler with area 0.0071 m^2 and depth 0.2 m. Below ground biomass from reference area was collected using the same method as in theKompsasuo.

Ruka

Field work in Ruka was done on 12th August, 2002. Nine vegetation plots from CW and 3 from adjacent natural wetland were analysed. Above ground biomass samples were collected from 0.2 x 0.2 m plots and below ground samples by using the same method as in the Lakeus CW.

Further processing of all collected material from the Kompsasuo, Lakeus and Ruka was done in laboratory. Below ground biomass samples were prepared manually and the living parts of roots and vegetation diameter larger than 1 mm were collected. All collected plant material (above and below ground) was dried at 40°C for approx. two days and weighted. N and P contents were analysed in the laboratory of North Ostrobothnia Regional Environment Centre.

Results

The above and below ground vegetation biomasses of the CWs in dry weight were generally higher than in their reference areas. However, there were no statistically significant differences. Only the below ground biomass of the reference area of Lakeus wetland in dry weight was higher in respect to the Lakeus CW (Mann-Whitney Test p=0,02) (Table 2). Also the amount of the plant material bound nitrogen and phosphorous in above and below ground seemed to be higher in all other CWs but in the Lakeus CW the below ground amounts of N and P were lower than in the reference area. However there were statistically significant differences only in the above ground amounts of N and P in the Ruka and Kompsasuo CWs in comparison with their reference areas.

Table 2. Mean biomasses and amounts of N and P in above and below ground plant material at the CWs of Kompsasuo, Lakeus, Ruka and Hovi and their reference areas. The pairs of constructed wetlands and reference areas with statistically significant difference (Mann-Whitney Test $p<0,05$) have been marked with (*) and frame.

Site	Above ground [drw g m^{-2}]	Below ground [drw g m^{-2}]	Above ground [N g m^{-2}]	Below ground [N g m^{-2}]	Above ground [P g m^{-2}]	Below ground [P g m^{-2}]
Kompsasuo CW	340,7	945,6	4,65*	6,44	0,49*	0,67
Kompsasuo ref.	323,0	844,1	2,67	4,75	0,24	0,38
Ruka CW	618,9	365,6	11,37*	4,16	0,83*	0,18
Ruka ref.	257,3	226,8	1,75	1,07	0,07	0,08
Lakeus CW	1507,2	260,3*	32,20*	3,16	1,82	0,17
Lakeus ref.	950,3	1645,7	10,82	5,38	0,55	0,52

The concentrations of nitrogen and phosphorous in biomass (per cents of dry weight) differed more clearly between the CWs and the reference areas. In all cases the concentrations of the reference areas were lower than concentrations of the CWs. Below ground concentaration of phosphorous did'nt show any statistically significant difference, neither did above ground P in Lakeus and below ground N in Kompsasuo (Table 3).

Table 3. The mean concentrations (per cents of dry weight) of N and P in above and below ground plant material at the constructed wetlands of Kompsasuo, Lakeus, Ruka and Hovi and reference areas (ref.). The pairs of constructed wetlands and reference areas with statistically significant difference (Mann-Whitney Test $p<0,05$) have been marked with (*) and frame.

	Above ground	Below ground	Above ground	Below ground
Site	N %	N %	P%	P%
Kompsasuo CW	1,37*	0,76	0,14*	0,06
Kompsasuo ref.	0,92	0,58	0,10	0,05
Ruka CW	2,02*	1,31*	0,14*	0,06
Ruka ref.	0,80	0,54	0,04	0,05
Lakeus CW	2,10*	1,82*	0,13	0,11
Lakeus ref.	1,14	0,34	0,06	0,04

Total amounts of biomass, N and P at the constructed wetlands of Kompsasuo, Ruka and Lakeus were calculated using the measured area of the wetland and the analysed data of N and P concentrations of the biomass of the wetlans (Table 4). The vegetated area of the Lakeus CW used in calculations is 35000 m^2.

Table 4. Total amounts of biomass, nitrogen and phosphorous at the Kompsasuo, Ruka and Lakeus constructted wetlands.

Site	Above ground [drw kg]	Below ground [drw kg]	Above ground [N kg]	Below ground [N kg]	Above ground [P kg]	Below ground [P kg]
Kompsasuo CW	8176,7	22694,0	111,6	154,6	11,7	16,1
Ruka CW	5075,1	2997,7	93,2	34,1	6,8	1,5
Lakeus CW	58189,2	10050,9	1243,3	121,9	70,3	6,7

Conclusions

There were no differences in above ground biomasses between the CW and the reference area at Kompsasuo, Ruka and Lakeus CWs. However the amounts of nitrogen and phosphorous (g m^{-2}) in above ground biomasses were clearly higher in the CWs than in the reference areas. Only the Lakeus CW did not show any statistically significant difference on the P amount. Also the concentrations of N and P were higher in the CWs except the P concentration in the Lakeus.

Patterns of differences in below ground biomasses and amounts of N and P and in the concentrations of N and P in were also quite clear. Generally there was not any difference in biomass or amounts of N and P or concentration of P between the CW and reference area. Only the concentration of nitrogen seemed to be higher in the constructed wetlands.

Concentrations of N and P at the Kompsasuo CW and its reference area were lower in this study than in the earlier study made 10 years ago (Huttunen *et. al.*, 1996).

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Investigations of wastewater produced on cattle-breeding farms and its treatment in constructed wetlands

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Abstract

The paper presents the results of investigations of wastewater formation and its treatment in CW with horizontal flow. The investigations were carried out on two cattle-breeding farms in the period 1995–2002.

On farmsteads containing cattle-sheds, the pollution of wastewater is 2–3 times higher than with domestic wastewater. The highest degree of wastewater pollution was observed on a dairy farm containing modern milking and milk refrigeration equipment.

The largest amount of slowly decomposing organic pollutants is retained during wastewater treatment in a septic tank (from 48.2 to 62.7% according to COD).

During wastewater filtration through a horizontal CW, the amount of retained organic pollutants (BOD_5 and COD) are 81.8 to 94.6%.

During wastewater filtration through CW, the efficiency of nutrient (N_{total} and P_{total}) treatment depends on the wastewater's initial pollution load: on a pig-breeding farm, where N_{total} concentration is 31.1 mg l^{-1}, CW retains 38.6% of nutrient load; on a dairy farm, on the other hand, where the N_{total} concentration is 101.0 mg l^{-1}, 61.4% of the nutrient load is retained in CW. On a pig-breeding farm where the P_{total} concentration is 9.6 mg l^{-1}, CW retains, on average, 91.7% of P; on a dairy farm, where the P_{total} concentration is 21.5 mg l^{-1}, 41.1% of P is retained in CW.

The analyzed wastewater treatment facilities (septic tank + CW) are distinct due to their buffering capabilities. They are effective enough, although due to limited P removal processes in a sustainable natural environment, additional P-removal means are to be used in wastewater treatment facilities arranged on dairy farms.

Introduction

Since 1990, a restructuring of the agricultural sector has taken place in Lithuania, resulting in the elimination of a large number of farms constructed in the Soviet period. Cattle-breeding production has moved to new private farms. In 1990 there were no private dairy farms or pig-breeding farms. At present there are 410 dairy farms containing 20 to 500 animal units and 170 pig-breeding farms with 50 to 5000 swine units.

In searching for new technological solutions, the Environment Ministry of Lithuania initiated a program of scientific research in 1994. This program was based on the treatment of wastewater in constructed wetlands (CW). The research was carried out in collaboration with scientists from Germany (G. Geller) and Switzerland (U. Schori and Ph. Wyss). This collaboration was of great importance and contributed much to the construction of the first wastewater treatment facilities of this kind on cattle-breeding farms in 1995.

Wastewater treatment facilities containing plant filters could be constructed using the local material workforce, and therefore their arrangement is not expensive. All geographical zones in Lithuania contain sufficient supplies of sand suitable for the construction of sand filters.

When calculating the constructional and technological parameters of experimental treatment facilities, primary attention was paid to primary wastewater treatment quality and particularly on the reduction of suspended solids (SS) load, as these are the main factors determining the durability of constructed wetlands. For the primary pre-treatment of wastewater, a three-chamber septic tank was selected, the design of which was chosen taking into consideration the industrial standards approved in Germany (DIN 4261). Investigations on the operating efficiency of constructed wetlands have been carried out for over two decades (Kadlec and Knight, 1996; Geller *et al.*, 1991; Reed *et al.*, 1984; Wyss, 1996; Burka, 1990).

The main objectives of the study included the analysis of the wastewater treatment process when wastewater was flowing via a three-chamber septic tank and CW.

Materials and methods

Pilot objects were arranged in an intensive karst region in northern Lithuania (Birzai district). The first object was arranged on a pig-breeding farm in 1995. The farm contains 40 sows. Treatment facilities collect wastewater produced by the farmer's family (3 members) and wastewater from a feeding kitchen. Wastewater is directed into a three-chamber septic tank with a storage capacity of 6.0 m^3. The ratio of septic chambers W_1:W_2:W_3 is 2:1:1. After pre-treatment in the septic tank, wastewater is directed into a CW with 50 m^2 storage capacity (L=10 m, B=5.0 m). The CW is arranged in a pit with flashing membrane. The depth of the sand layer is 0.8 m. In 1995 the surface of the filter was planted with reed (*Phragmites australis*).

The second study object was arranged on A. Visockas' dairy farm in 1999. The farm (with 100 milking cows) contains modern *Alfa-Laval* systems for milking and milk refrigeration. Wastewater from the dwelling (the family has 7 members) and wastewater from dairy equipment rinsing are canalized into a 17 m^3 three-chamber septic tank. After pre-treatment in a septic tank, wastewater flows into a CW with an area of 100 m^2 (L=B=10m). Wastewater enters the filter through distribution pipes arranged on both sides of the reed bed filter. The principal scheme of wastewater treatment facilities arranged on both farms is given in Figure 1.

Natural sand from a quarry was used in both study objects. The chemical composition of the sand is as follows: pH – 8.9–9.1, organic matter – 0.15–0.32%, Si – 43.7–45.1%, Fe – 0.71–0.99%, Ca – 0.43–0.63%, Mg – 0.11–0.18%, porosity – 34.40%, comparative mass – 2620–2680 kg m^{-3}, sand particles d_{10} – 0.27–0.33 mm, d_{60} – 0.68–0.77 mm ($CU=d_{60}/d_{10}$=2.33–2.52). The sand filtration coefficient is 18.5–20.4 m d^{-1}. The distance from the wastewater distribution pipe to the drainage pipe is 5.0 m.

The investigation of wastewater treatment process on a pig-breeding farm and a dairy farm were carried out in the periods 1996–2001 and 1999–2002 respectively. During the study period on both study objects, wastewater samples were taken for laboratory analysis every month at the following points in the wastewater treatment process: wastewater inflow into the septic tank, wastewater treated in chambers 1 and 2 of the septic tank, wastewater outflow from the septic tank and wastewater outflow from the CW. While sampling wastewater, the wastewater discharges were also measured.

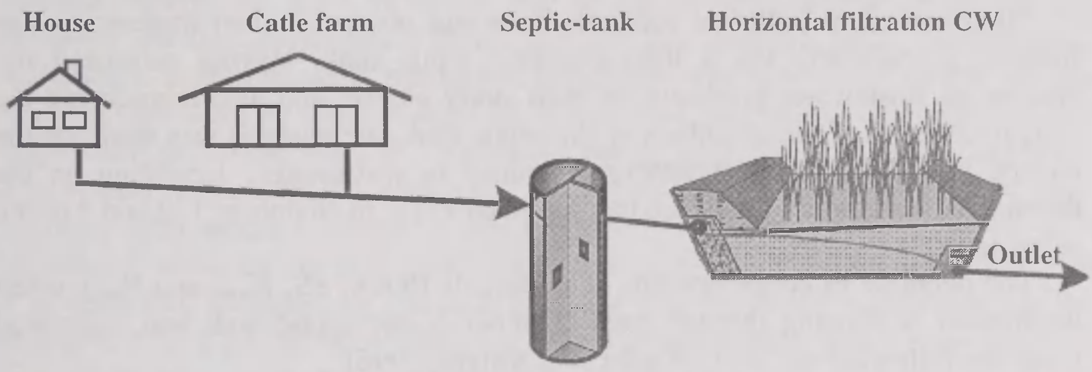

Figure 1. Technological scheme of the treatment process of wastewater produced on a cattle-breeding farm.

Results and discussion

On the pig-breeding farm the duration of wastewater flow into wastewater treatment facilities fluctuated from 1.75 to 2.67 h within a 24-hour period.

Maximum momentary wastewater discharges were measured during the premises' washing process ($0.61\ l\ s^{-1}$).

On A. Visockas' dairy farm the duration of wastewater flow into the treatment facilities fluctuated in the range of 2.51 to 3.28 h within a 24-hour period. Maximum momentary wastewater discharges were measured during the washing process of the milk refrigerator and other premises ($0.91\ l\ s^{-1}$). Wastewater non-uniformity coefficients of an hour varied from 7.38 to 15.48. Wastewater discharge in a 24-hour period (Q_d) measured in different study periods fluctuated from 0.49 to 0.94 $m^3\ d^{-1}$ and from 1.6 to 3.7 $m^3\ d^{-1}$ on a pig-breeding farm and dairy farm respectively.

The study results showed a relatively high wastewater pollution load that exceeded that of domestic wastewater 2–3 times. A similar wastewater pollution load has also been determined by other scientists (Kern, 1998; Biddlestone *et al.*, 1991).

The amounts of easily decomposed organic pollutants contained in wastewater on the two farms differed greatly: the average BOD_5 was 578.0 $mgO_2\ l^{-1}$ on the pig-breeding farm and 920.0 $mgO_2\ l^{-1}$ on the dairy farm. The percentage of easily decomposed organic pollutants (BOD_5/COD_{Cr}) was also different – 80.1 and 40.6% respectively. Comparatively high N and P concentrations were determined in wastewater produced on the dairy farm – 135.0 and 30.0, which are 1.8 and 2.1 times higher respectively than on the pig-breeding farm. Ammonia nitrogen $N\text{-}NH_4$ makes up the largest proportion of pollutants contained in wastewater produced on both farms (69.2 and 71.5% respectively).

Mineral phosphorus $P\text{-}PO_4$ makes up the largest amount of total phosphorus contained in wastewater on both farms (63.6 and 68.0% respectively).

The decrease in pollutant concentrations was observed when wastewater was flowing successively via a three-chamber septic tank. Having estimated the amount of wastewater produced on both study objects and also considering the storage capacity of the chambers of the septic tank, the analysis was made on the change in pollutant concentrations contained in wastewater, depending on the duration of wastewater residence in a septic tank, i.e. in chambers 1, 2 and 3 of the septic tank.

The decrease in concentrations of pollutants BOD_7, SS, N_{total} and P_{total}, when wastewater is flowing through each chamber of the septic tank was calculated using the following equation (Kadlec and Knight, 1996):

$$E=100(Q_{in}C_{in}-Q_{out}C_{out})/(Q_{in}C_{in}) \tag{1}$$

where Q_{in} and Q_{out} – wastewater inflow and outflow, m^3d^{-1}; C_{in} and C_{out} – pollutant concentration in wastewater inflow and outflow, $mg\ l^{-1}$.

Regressive analysis of the data showed rather reliable correlations – the square deviation R^2 fluctuated within the range of 0.79 to 0.90. Wastewater pre-treatment efficiency E% according to BOD_7, SS, N_{total} and P_{total} with respect to the duration of wastewater treatment process in a septic tank t_{sept} is expressed by the following empirical equations:

$$E_{BOD} = 0.487\ t^2_{sep}+9.35\ t_{sep}+3.01\ (R^2=0.90) \quad (2)$$
$$E_{SS}=0.69\ t^2_{sep}+13.77\ t_{sep}+0.70\ (R^2=0.89) \quad (3)$$
$$E_N=0.33\ t^2+6.53\ t_{sep}+3.28\ (R^2=0.79) \quad (4)$$
$$E_P=0.30\ t^2+5.98\ t_{sep}+1.40\ (R^2=0.88) \quad (5)$$

Having summarized the results of 7-year studies on a pig-breeding farm and 3-year studies on a dairy farm, it was determined that a septic tank is most capable in the retention of SS (64.5%, on average). The retention of organic matter (according to BOD_5) is 45.5 and 50.9%, the amounts of retained nutrients (N_{total} and P_{total}) are 27.7 and 25.2, and 19.1 and 26.8% respectively. Investigations of wastewater treatment efficiency in the CW were performed under the conditions of free wastewater filtration (*i.e.* without drainage affluent). Summarized data on the investigations carried out on the pig-breeding farm and dairy farm are given in Tables 1 and 2 respectively.

A septic tank is an important device when preparing wastewater for further biological treatment. It stabilizes the pH of wastewater and is capable of retaining large amounts of SS and slowly decomposing pollutants COD (64.9 and 48.2% from wastewater produced on pig-breeding farms respectively; 64.2 and 62.7% from wastewater on dairy farms accordingly). BOD_5 amounts are also reduced in a septic tank. The retention of BOD_5 here is 45.5 and 50.9% respectively.

During the wastewater filtration process via the CW, from 81.8 to 94.6% of organic pollutants (according to BOD_5 and COD) are retained. The efficiency of nutrient removal in the CW varied due to the different degree of wastewater pollution on the investigated farms. On the pig-breeding farm, where initial wastewater pollution with nitrogen compounds was low, only 38.6% of the nutrient load were retained in the CW, while on a dairy farm with higher initial wastewater pollution, the CW retained as much as 61.45% of the nutrient load. The results regarding the efficiency of P-removal are opposite: on the pig-breeding farm, where the initial P-load contained in wastewater is low, 91.7% of P is retained; while on the dairy farm, which has a high initial P-load, only 41.4% of P is removed. Considering the data obtained, the following conclusion can be drawn: CW is a perfect decomposer of nitrogen compounds, although its removal capabilities of phosphorus compounds are limited as natural sand contains small amounts of Fe+Ca+Mg (on the average 1.52%).

The results of calculations of CW load with phosphorus show that on the pig-breeding farm, when the average CW load is 0.22 g P $m^{-2}d^{-1}$, 1 m^2 of CW retains an average of 0.2 g $m^{-2}d^{-1}$ of phosphorus. On the dairy farm, where the average CW load is 0.43 g P $m^{-2}d^{-1}$, 1 m^2 of CW retains the same amount of phosphorus as 1 m^2 of CW on the pig-breeding farm – 0.18 g $m^{-2}d^{-1}$. The studies performed in

Estonia (Mander *et al.*, 2001) show similar results of wastewater treatment efficiency in CWs with horizontal filtration.

Table 1. Wastewater treatment efficiency on pig-breeding farm.

Investigated waste-water	pH	BOD_5	COD	N_{total}	$N\text{-}NH_4$	P_{total}	$P\text{-}PO_4$	SS
		mg L^{-1}						
Before treatment	7.4± 0.23	578± 101.3	722± 163.0	43± 8.2	20.5± 4.4	14± 3.1	8.9± 2.7	506± 147.4
After treatment in a septic tank	7.5± 0.19	315± 41.0	374± 60.2	31.1± 4.0	16.6± 3.7	9.6± 2.3	6.4± 1.6	177.4± 39.7
After treatment in CW	7.1± 0.08	17± 9.6	68.2± 17.1	19.1± 3.9	11.6± 5.2	0.8± 0.3	0.6± 0.2	29.5± 11.3
Treatment efficiency [%]:								
Septic tank		45.5	48.02	27.7	19.1	31.4	28.1	64.9
CW		94.6	81.8	38.6	30.1	91.7	90.1	85.4
Wastewater treatment facilities		97.1	90.5	55.6	43.5	94.3	93.3	94.2

Table 2. Wastewater treatment efficiency on dairy farm.

Investigated wastewater	pH	BOD_5	COD	N_{total}	$N\text{-}NH_4$	P_{total}	$P\text{-}PO_4$	SS
		mg L^{-1}						
1	2	3	4	5	6	7	8	9
Before treatment	6.5± 0.12	920± 215.0	2266± 377.0	135± 21.0	96.5± 21.4	30.0± 5.9	20.4± 4.2	480.0± 127.8
After treatment in a septic tank	7.0± 0.21	452.7± 102.0	846.0± 127.0	101.0± 14.0	70.6± 14.6	21.5± 4.7	16.9± 3.3	172.0± 68.0
After treatment in CW	7.26± 0.24	28.7± 17.8	109.0± 21.4	39.2± 8.2	28.7± 7.7	12.6± 2.1	10.6± 2.7	18.3± 9.4
Treatment efficiency, %:		rde						
Septic tank		breeding	62.7	25.2	26.2	28.4	17.2	64.2
CW		farm.	87.2	61.4	59.3	41.4	37.3	89.3
Wastewater treatment facilities		50.9	95.2	71.1	70.3	58.0	48.1	96.2
		93.6						
		96.9						

Conclusions

On farmsteads containing cattle-sheds, the pollution of wastewater is 2–3 times higher than domestic wastewater. The highest degree of wastewater pollution was observed on a dairy farm containing modern milking and milk refrigeration equipment. The largest amount of slowly decomposing organic pollutants is retained during wastewater treatment in a septic tank (from 48.2 to 62.7%, according to COD). During wastewater filtration through a horizontal CW, the amounts of retained organic pollutants (BOD_5 and COD) are 81.8 to 94.6%.

During wastewater filtration through the CW, the efficiency of nutrient (N_{total} and P_{total}) treatment depends on the initial pollution load of the wastewater: on a pig-breeding farm, where the N_{total} concentration is 31.1 mg l^{-1}, the CW retains 38.6% of the nutrient load; while on the dairy farm, where N_{total} concentration is 101.0 mg l^{-1}, 61.4% of the nutrient load is retained in the CW. On the pig-breeding farm, where P_{total} concentration is 9.6 mg l^{-1}, the CW retains an average of 91.7% of P; on the dairy farm, where P_{total} concentration is 21.5 mg l^{-1}, 41.1% of P is retained in CW. The analyzed wastewater treatment facilities (septic tank + CW) are distinct in terms of their buffering capabilities. They are relatively effective, although due to limited P removal processes in a sustainable natural environment, additional P-removal means are to be used in wastewater treatment facilities arranged on dairy farms.

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Preliminary studies of flow patterns in model subsurface flow constructed wetlands

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Introduction

Constructed wetlands (CWs) are ecological systems that combine physical, chemical and biological processes in an engineered and managed system. They are designed to take advantage of many of the same processes that occur in natural wetlands. CWs have proven their efficiency in treating wastewater and reducing pollutants like BOD_5, COD, SS, N, P and pathogens (Ayaz and Akca, 2001; Gomez Cerezo *et al.*, 2001; Drizo *et al.*, 1999; EPA, 2000; Kadlec and Knight, 1996; Kivaisi, 2001) in both warm and in cold climates when providing enough insulation and retention time (Jenssen and Krogstad, 2002; Jenssen *et al.*, 2002; Mæhlum and Jenssen, 2002).

Water movement is one important factor that influence the physical, chemical and biological treatment processes in CWs. Nevertheless, inadequate knowledge about water movement in the CWs has led to the over-sizing of these systems and variability in treatment performance (Fisher, 1990; King *et al.*, 1997)). Plug flow (ideal flow) has been presumed by many of the existing design guidelines (WPCF, 1990, Reed *et al.*, 1995), ignoring the existence of short-circuiting and dispersion. The ideal flow implies that the incoming water volumes have an equal residence time in the system and that no mixing between volumes occurs. The ideal flow has been proven incorrect by several investigations (Kadlec, 1994; Fisher, 1990; Urban, 1990); where intermediate degree of mixing and dispersion is shown to occur. In order to investigate physical and chemical properties as basis for modelling of processes in CWs a box experiment was designed. The box experiment provides possibilities of process study in a controlled environment. This paper discusses preliminary results of flow pattern in the boxes.

Methodology

Laboratory set-up

A box experiment using a special light weight aggregates (LWA) with a high phosphorus sorption capacity termed Filtralite-P^{TM} (Jenssen and Krogstad, 2000) is performed. The LWA has a grain size in the range 0–4 mm. The effective porosity of the LWA is 40% (Heistad, 2001). Particle density has a range of 600–800 kg m^{-3} with a corresponding dry bulk density of 300–550 kg m^{-3}, and a saturated hydraulic conductivity of 100 m d^{-1} (OPTIROC, 2001). Jenssen *et al.* (2002) suggest a value of 30 m d^{-1} for the saturated hydraulic conductivity in full scale-systems due to reduction in porosity caused by roots and potential biological and chemical clogging. In order to homogenize the media, one cubic meter of the LWA was mixed properly through a special splitter device with 4 equally sized compartments that received the LWA from a funnel placed on the top of the device. One forth of the mixed volume was remixed and then one forth of the remixed quantity has been mixed and run through smaller splitter device used for distributing animal food pellets into equal quantities. LWA was filled in a 21 transparent boxes with dimensions of 26 x 7 x 7 cm (1274 cm^3) for the packed material in each box.

Tracer studies

The preliminary tracer study is performed in one box that has been used for phosphorus (P) sorption studies (Adam, 2003). 1.7 ml of NaCl (0.02 mg ml^{-1}) was continuously added as droplets to the flow without changing the flow gradient in the box. The electrical conductivity in the outflow was measured using conductivity meter (RE 387 TX). The loading rate was 1.25 l d^{-1} and the P concentration was 15 ppm. The tracer test was followed by investigations of flow in a computer tomograph (CT-scanning), using potassium iodide as a tracer, in order to visualize the flow.

Results and discussion

In a parallel extraction experiment of the P from different location in the box, variations in P concentration along the box were obtained; some points had high values and other had low values over a short distance (Adam, 2003). This implies that the flow in the box was not uniform (plug) flow, which contradicts our assumption about the homogeneity of the media. Theoretical calculations assuming plug flow suggested a theoretical retention time of 10 hours at a loading rate of 1,25 l d^{-1}.

Retention time = Effective volume (L^3) available to flow / volume flow (L^3 T^{-1})

= Total Volume (L^3) * Porosity / volume flow (L3 T^{-1}
= 1274 cm^3 * 0.4 / 1250 cm^3 d^{-1}
= 1274 cm^3 * 0.4 / 52.08 cm^3 h^{-1}
= 9.78 hours ≈ 10 hours

The results from the preliminary tracer studies are shown in Fig 1. The peak of the curve (Figure 1a) indicating an early breakthrough at 44 minutes, followed by a relatively long tail. Dual porosity effect could be the main reason for the long. According to Gaussian density function (Figure 1b) 68% of the original point source is still exist within a distance of 2σ around the central value of 5,00E-1 as shown in Figure 1b, indicating an average retention time of 60 minutes comparing to the theoretical one that is 10 hours. These results suggest that the effective volume available to flow (effective porosity) is less than of the medias' total pore volume. Assuming a retention time of 1 hour the effective porosity becomes:

Retention time = Total Volume (L^3) * Effective porosity / volume flow (L^3 T^{-1})
1 hour = 1274 cm^3 * Effective porosity / 52.08 cm^3 h^{-1}
Effective porosity = 1 hour * 52.08 cm^3 h^{-1} / 1274 cm^3
Effective porosity = 0.04

If this picture is correct it may seriously affect the contact area between the free flow water face and the porous media.

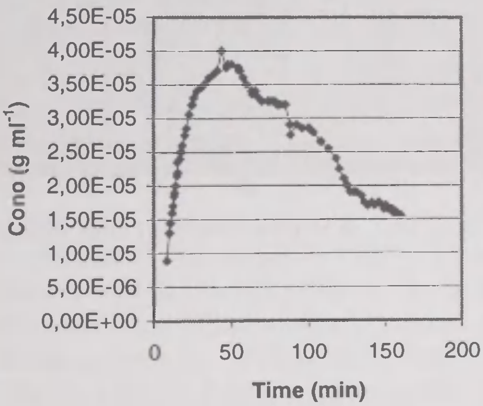

Figure 1a. Breakthrough curve for NaCl.

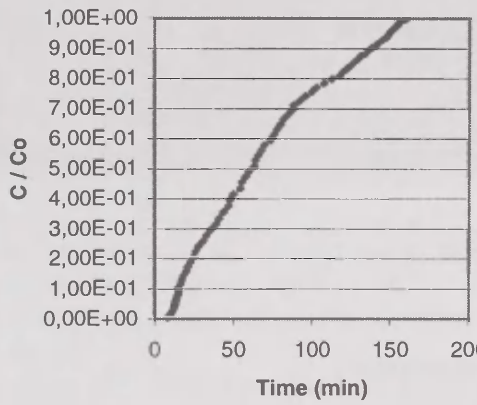

Figure 1b. Normalized cumulative breakthrough curve for NaCl.

From a total loaded mass of 0.034 gm as a NaCl tracer, only 10% of it had been recovered. This could be attributed to the dual porosity effect in the media or due to potential reaction between the tracer and Ca^{2+} and Mg^{2+} components of the media. This questions the use of NaCl as a conservative tracer for the tested media. In order to further elucidate the flow in the box, CT scanning was carried out using potassium iodide tracer. The images in Figure 2 shows an existence of density variation in the media, the darker areas being the lighter density areas. Macropores are also present as black spots. However, there were no indications that the macropores were continuous.

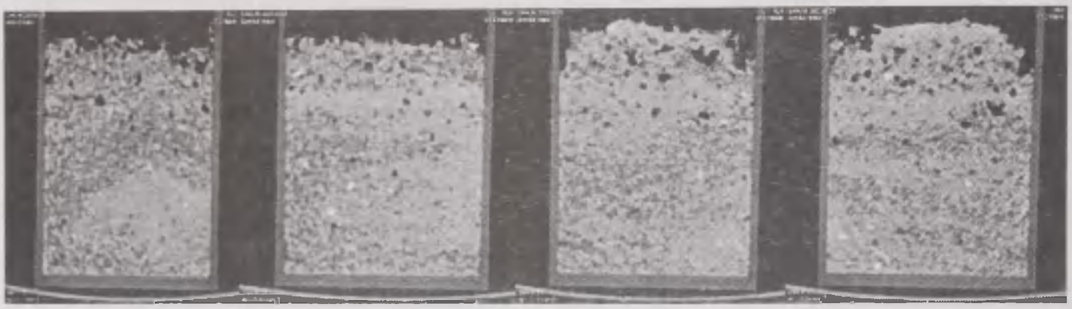

Figure 2a. C T scanning of the tested box (vertical cross section in flow direction) before adding the potassium iodide.

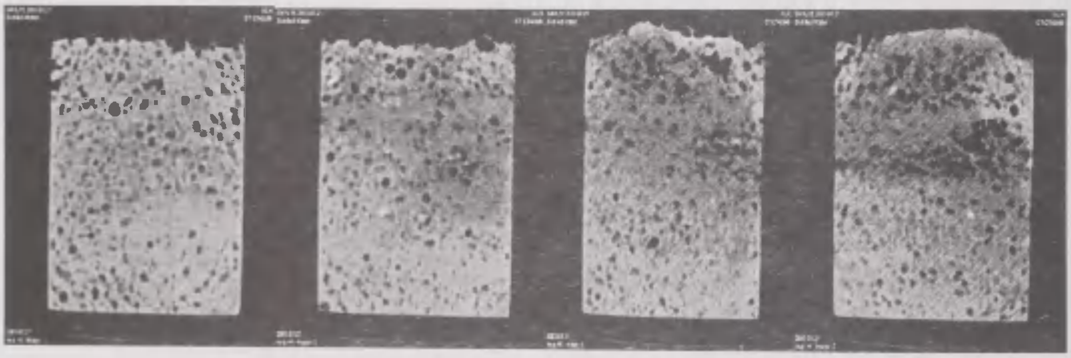

Figure 2b. C T scanning of the tested box (vertical cross section in flow direction) after adding the potassium iodide.

Conclusion

Homogeneity is not easily achievable with such media even in a lab scale experiment. This inhomogeneity can promote different preferential flow mechanisms. Tracer test alone does not fully reveal the flow patterns in the box experiment. Preliminary studies indicate that CT-scanning of media can help elucidate the flow patterns. However further studies using different tracers and more replicates are necessary to produce more conclusive results regarding the hydraulic behaviours of the system.

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Experience in artificial wetland construction for wastewater purification in the Arctic latitudes (Murmansk region)

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Abstract

Empirical data and the calculation of required insulation show that a cold climate does not restrict the use of constructed wetlands for wastewater purification if the proper design considerations are followed. Cold climates, however, require larger and deeper systems than those found in warm climates (Maehlum and Jenssen, 2003).

A pilot artificial wetland was constructed in Northern Russia (Murmansk region) in the village of Shonguy during the period November 2001 – June 2002. This is the first experience of artificial wetland construction under Arctic conditions.

The total area of the wetland is 2700 m^2, and it includes an area with vegetation and a sedimentation pond. The trees in the vegetation unit include various indigenous willow species (*Salicaceae*), shrubs of *Rubus idaeus L.* and some helophytes, such as sedges (*Cyperaceae*). In June 2002 the artificial wetland was put into active operation.

The functioning of the newly constructed artificial wetland was initially irregular, and apparently the biological parameters had not yet stabilised. Later, however, in December 2002, it appeared that the biological processes had more or

less stabilised. Even at an air temperature of minus 32°C the system worked adequately as far as the breakdown of organic compounds was concerned. The efficiency of the artificial wetland was highest in the case of suspended substances, and generally it was more than 81%.

Introduction

The treatment principle of the bioplato technology is to combine anaerobic and aerobic treatment phases in the filtration load, when the liquid slowly flows along the site spindle.

The purpose of our project is to test artificial wetland technology for wastewater purification, which has proven to be efficient in cold climates, but for the first time under Arctic conditions.

An artificial wetland was constructed in the village of Shonguy, not far from Murmansk. The bioplato is based on experiences obtained in the framework of a four-year INCO/Copernicus project in the Ukraine and Estonia (Mander *et al.*, 1999; Stolberg *et al.*, 1999), as well as on comparable studies in Canada, Finland and Sweden (Mander and Jenssen, 2003). The purifying capacities of this pilot system, which has for the first time been studied in an Arctic climate, will be monitored by analysing the inflows and outflows.

The planning of the artificial wetland has been the result of detailed exchanges of ideas between Murmansk Vodokanal on the one hand and SYKE, the University of Tartu and KSAME on the other hand. Construction of the artificial wetland in the village of Shonguy took place during the period November 2001 – June 2002 and was carried out by the company "Spetsstroymekhanizatsia".

The village of Shonguy where the bioplato is located lies 30 km south of Murmansk, at $68^0 45'$ north latitude and $33^0 10'$ east longitude. The total area of the wetland is 2700 m^2. The climatic conditions of the area are characterized by the following parameters: the duration of sunshine is 1289 hours a year, and snow cover varies from 3–21 cm. The duration of snow cover is 198 days; stable snow cover is present on average from 1 November until 7 May. The average amount of precipitations is 488 mm a year. During the spring and summer season, the amount of precipitation is double (322 mm) compared to during the autumn and winter period (166 mm). Such a seasonal imbalance of precipitation influences the water regime of the bioplato units, considering slight variations in relative air humidity (from 74–85% during the autumn and winter period and up to 69–78% during the spring and summer period). The efficiency of the functioning of the bioplato units located in an open site depends greatly on the area's temperature regime during the winter period. The average daytime temperature during the period from 20 November to 20 March is –8 ^{0}C. The temperature drops below 0^0C during the first decade in November. During the 60 days from 20 December to 18

February, the average daytime temperature is -10^0C. During this period the freezing of water intended for additional treatment is very possible, with ice covering the first and second units. However, frequent thaws typical of this period make it possible to keep the bioplato in functioning regime. The continuity of polar day during the period from 22 May to 22 July, as well as the significant length of sunshine up to the beginning of autumn and the supplementation of the heating of the additional treatment units by the upcoming treated wastewater make it possible to conclude that the climatic conditions of the area are quite favourable for the exploitation of an infiltration and surface bioplato.

Description of work

All of the initial groundwork was performed in Shonguy during the winter season in 2001. Subsequently, the natural vegetation (willows) around the existing pond was cut down. After this initial phase, groundwork was commenced in and around the existing pond, the borders of the future filtration unit were planned, as well as the preparation of the waterproof foundation. Concrete pipes with holes for wastewater supply were put in position and covered by a layer of gravel, and subsequently the vegetation unit was delimited, while retaining as much as possible of the natural vegetation. Trees had to be cut down in the vegetation unit; however, stubs and roots were partially kept in the beds. For wastewater drainage two iron-and-concrete rings were dug in the willow unit and covered by stones. This is to make it possible for the water to flow through a layer of stones. After the groundwork had been finished, the borders were strengthened with boards, and the dam between the wetland's filtration unit and the Kola River was strengthened with iron-and-concrete blocks and a concrete foundation.

In the early spring (before the leaf buds opened) (beginning of May – end of May), willow trees and bushes were planted, and grass was sown in the willow unit. Subsequently, fertile soil was delivered to the beds. The trees in the vegetation unit consist of various willow species (*Salicaceae*) of indigenous flora and *Rubus idaeus L.* – the most stable bush against industrial gases. In addition, non-woody plants were also planted. At the beginning of July, planting was carried out on the borders of the artificial wetland. Helophytes such as sedges (*Cyperaceae*) were planted in the vegetation unit at the beginning of July and the end of August 2002.

Results and discussion

Wastewater entering the treatment facilities in the village of Shonguy is characterized by a high BOD, nitrogen of ammonia salts, oil products, phosphates and 'synthetic surface active substances' (SSAS). Over a period of one year the average technical purification before entering the artificial wetland was 62.5% for suspended substances, 88.3% for BOD, 34.3% for nitrogen (ammonia salts), 8% for oil products and 69% for SSAS. This implies that after passing the purification plant, the chemical composition of the wastewater does not yet meet the environmental standards to permit it to be released into the river.

Wastewater from the village of Shonguy is, before entering the artificial wetland, first purified in a nearby conventional wastewater plant. The purification process in this plant is not very effective and is not up to standard. Before construction of the artificial wetland, this partly purified wastewater was released directly into the Kola River. Currently this wastewater, after treatment in the plant, is first led through the vegetation unit via a sedimentation well, and subsequently through the sedimentation unit, which consists of gravel, before it is discharged into the Kola River. Leaves and the stems of high plants and periphyton are a biological filter that possesses such abilities as photosynthetic-producing, demineralisation, detoxication, and phytoremediation (Einor, 1990). Macrophyte roots regulate accumulative-destruction processes in the block's bottom deposits, where infiltration occurs through ground saturated with macrophyte roots and microorganisms, and filtrating filling up.

During the first months, the functioning of the newly constructed artificial wetland was irregular, but later, in December 2002, it appeared that the biological processes had more or less stabilised. Even at an air temperature of minus 32°C, however, the system worked adequately as far as the breakdown of organic compounds was concerned.

The efficiency of the artificial wetland was highest for suspended substances, and was generally more than 81%. The system was also more effective at purifying oil products (more than 80%), synthetic surface-active substances (more than 70%). The effect on easy-oxidizing organics (BOD) was very irregular, and the decrease in BOD varied from 50% to 80%. When the efficiency of the willow unit and the filtration unit are compared, the efficiency of the willow unit varied from 13% to 74% during the year, and the gravel unit fluctuated from 32% to 90%. The gravel unit proved more stable than the willow unit, as it is better protected against the influence of external factors such as weather conditions. It may, however, be expected that the instability of the willow unit will decrease after the willow trees have become more appropriately settled down.

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The long-term phosphorus retention capacity of a horizontal subsurface flow constructed wetland

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Introduction

In contrast to other pollutants, phosphorus (P) has no relevant gaseous phase in its biochemical cycle, and thus the P sorption capacity is the main factor limiting the life expectancy of a sub-surface flow (SSF) constructed wetland (CW). When in contact with the wetland filter material, P may be retained via adsorption and precipitation by oxides of iron, aluminium and calcium carbonates. Both mechanisms are controlled by properties of the substrate (Fe- Al- and Ca-minerals, porosity) and physicochemical environment (pH, Eh, dissolved ions). Hydraulic parameters (loading rate, retention time) also affect P removal.

In this paper the dynamics of the long–term phosphorus purification efficiency of the horizontal subsurface flow (HSSF) planted sand filter in Kodijärve, Estonia, is analysed.

Materials and methods

Site description

The Kodijärve horizontal subsurface flow (HSSF) constructed wetland (in south Estonia, constructed in October 1996) purifies wastewater from a hospital for about 40 persons. From the septic tank, the wastewater flows into the vertical subsurface flow (VSSF) CW (constructed in Summer 2002). Wastewater passes the wetland, and from the outlet the water flows into the phosphorus sedimentation filter bed (that is filled with calcareous fly-ash, established in Summer 2002). From the filter bed outflow it reaches through a channel to the natural reed stands on the lakeshore. The HSSF wetland system consists of two beds (chambers), each measuring 25×6.25×1 m, filled with coarse iron-rich sand. (The left bed, with finer filter material, has more surplus moisture conditions. The right bed has coarser material and drier conditions).

Water samples

Water samples were generally taken once a month from inflow and outflow, 77 times from 1997–2002, and 24 times from the outflow of both beds (2000–2002) for further analyses for PO_4-P and total P (according to APHA, 1989) in the lab of South Estonian Environmental Research Ltd.

Retention and removal calculations

Phosphorus input and output values for the Kodijärve CW were calculated by multiplying the P concentration and water discharge, averaged over the period between the sampling events and divided by the area.

The input of P into the system is defined as a P load (in g m^{-2} d^{-1}). The mass removal rate in g m^{-2} d^{-1} (Kadlec and Knight 1996) was calculated as follows:

$$R= \sum (Q_{in}C_{in}-Q_{out}C_{out})/A \qquad (1)$$

where $(Q_{in}C_{in}-Q_{out}C_{out})$ is the daily retention, A is the wetland area, Q_{in} and Q_{out} = inflow and outflow values (m^3d^{-1}), and C_{in} and C_{out} = corresponding concentration values (mg L^{-1}).

Removal efficiency E (%) (Percent Mass Removal; Kadlec and Knight, 1996) of P in treatment wetland was estimated as:

$$E = 100*(Q_{in}C_{in}-Q_{out}C_{out})/(Q_{in}C_{in}) \qquad (2)$$

In addition, in every October since 1997, complex soil samples have been taken from 3 different depths (0–10, 30–40 and 50–60 cm) around the piezometers (Figure 1) to analyse for Kjeldahl N, lactate soluble P and organic C (ignition loss) in the Laboratory of Plant Biochemistry of the Estonian Agricultural University (LPB-EAU.

Results and discussion

Average P inlet and outlet concentrations in the Kodijärve CW system were 14.6 ± 5 mg P l^{-1} and 3.2 ± 1,8 mg P l^{-1} (Table 1). Average purification efficiency for the whole system was 78.4% and the mass removal rate for the same period was 0.14 g m^{-2} d^{-1}.

Table 1. Average purification efficiency and inflow-outflow parameters of Kodijärve HSSF (Average ± Standard Deviation).

Parameter	1997	1998	1999	2000	2001	2002	Average
Q ($l\ min^{-1}$)	3.5 ± 1.9	3.1 ± 1.1	1.7 ± 1.5	0.7 ± 0.6	1.25 ± 0.9	4.39 ± 3.3	2.3 ± 2.2
Removal Efficiency, E (%)	95 ± 3	76 ± 21	84 ± 12	64 ± 26	83 ± 9	77 ± 20	79 ± 19
Average inflow ($mg\ l^{-1}$)	16.7	11.1	13.8	15.3	16.2	13.5	14.6 ± 4.6
Average outflow ($mg\ l^{-1}$)	1.1	2.3	2.2	4.4	3.0	4.2	3.2 ± 1.8
Removal rate ($g\ m^{-2}d^{-1}$)	0.3 ± 0.3	0.14 ± 0.1	0.09 ± 0.08	0.04 ± 0.04	0.07 ± 0.06	0.22 ± 0.21	0.14 ± 0.16

There are some differences in the purification efficiencies of the right and left beds. During the investigation period, the average P removal efficiency in the right bed was slightly higher (81%), than in the left bed (70%). This could be the result of the anaerobic conditions in the left bed. The average mass removal rate between the beds did not vary significantly (0.13 $g\ m^{-2}d^{-1}$ for the left bed and 0.14 $g\ m^{-2}d^{-1}$ for the right one).

In 1997–2002 the horizontal flow sand-plant filter had an average loading of 557 $kg\ ha^{-1}yr^{-1}$ for phosphorus. The average inlet concentrations of P did not vary significantly, but the fluctuating flow rates and thus the loading of P in the last year may be the reason why the purification efficiency in the filter bed has been mutable (Table 1; Figure 2). Likewise, Mander and Mauring (1997) have found a positive correlation between nutrient loading and purification efficiency.

Although the outlet concentrations in Kodijärve sand planted filter system showed a slowly increasing trend from 1997–2002 (Figure 1), the purification efficiency of phosphorus has been quite good (63–95% respectively).

According to the results, it is clear that purification efficiency did not decrease during the cold season. The most critical period for purification efficiency was found in spring and early summer (Mander *et al.* 2003a).

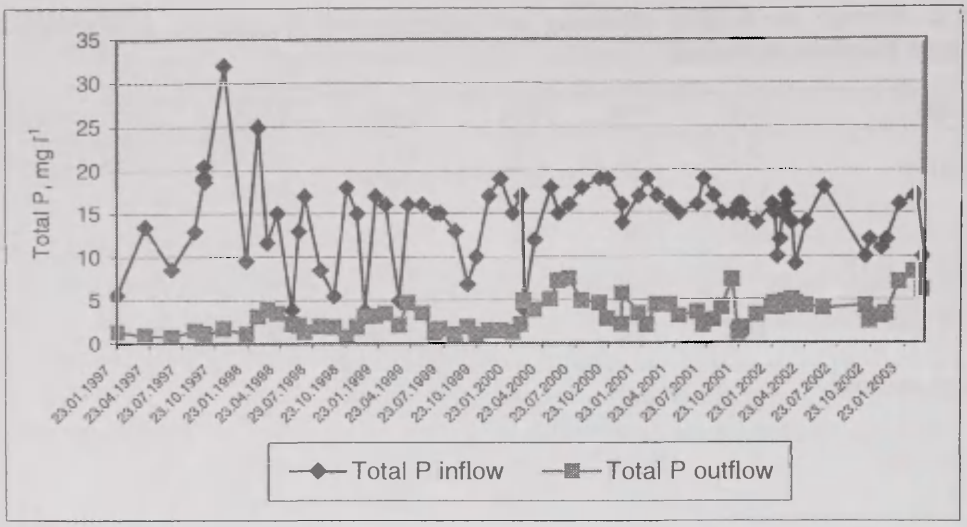

Figure 1. Inlet and outlet concentrations of total P in the planted sand filter in Kodijärve from 1997–2002.

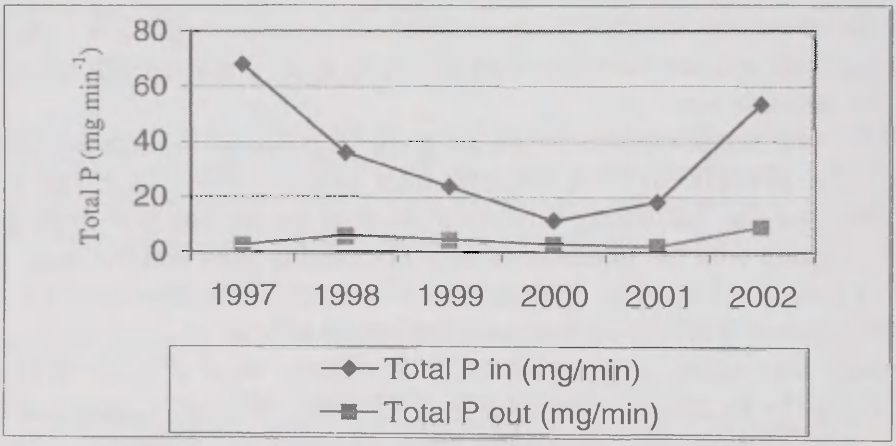

Figure 2. Inlet and outlet load of P in the Kodijärve HSSF CW.

Average annual P removal from the system was 49.5 g m^{-2} yr^{-1} respectively. This is a satisfactory value for phosphorus (Kadlec and Knight, 1996).

Phosphorus in soil

The annual P removal rate (adsorbed in soil as lactate-soluble P) has been decreasing (Mander *et al.*, 2003a). One of the reasons for this may be the saturation of filter sand with phosphorus (Kadlec and Knight 1996). The increasing outwash of Fe (Mander *et al.*, 2003a) may also be a sign of decreasing adsorption capacity.

After five years' operation, the cumulative phosphorus retention in Kodijärve HSSF CW was 52.8 kg, of which the majority (88.1%) was adsorbed in soil. Assimilation by plants (6.1%) and microbial immobilization (4.4%) mainly supported the removal process (Mander *et al.*, 2003b).

Thus the filter bed for phosphorus removal at the outflow, filled with calcareous fly ash from the oil-shale burning process, was established in Summer 2002. The preliminary results show good performance and improved water quality.

Conclusions

The phosphorus adsorption capacity of the filter media is the main factor limiting the life expectancy of sub-surface flow SSF CW.

Decreasing annual P adsorption shows that planted soil filters like the Kodijärve HSSF CW can be saturated with P during 5–6 years of operation.

To maintain the P removal in the Kodijärve system, a P sedimentation filter was built to the outflow ditch. Preliminary results show improved performance.

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Neural network based modelling of constructed wetlands: Prediction of nitrogen and phosphorus concentration

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The artificial neural networks (ANN) have been applied to solve a wide variety of environmental problems including atmospheric chemistry, river system modelling and groundwater systems. In the PRIMROSE project the use of ANN modelling technique was applied for constructed wetlands. The efficiency of wastewater treatment is closely connected to nitrogen and phosphorus removal capability. Thus the modelling was focused on the prediction of phosphorus and nitrogen concentration at the output from the wetland in two aspects. The first was to identify the physical and chemical parameters, which are controlling the abovementioned concentrations, and the second aspect was to calculate the residence time distribution function of nitrogen and phosphorus in the wetland. For the simulations a feed forward neural network was used. As the activation function the sigmoid function was chosen. In all cases the network geometry was similar (Figure1).

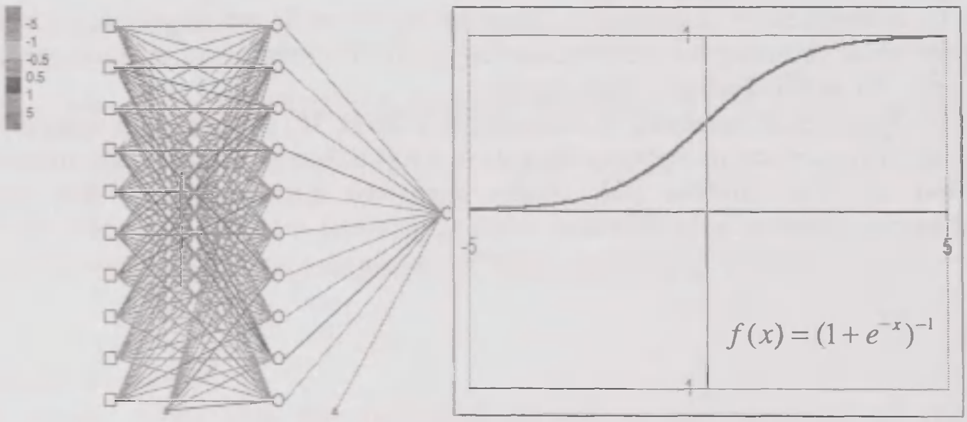

Figure 1. Neural network geometry with an example of activation function.

It consisted of 3 layers. In the first one, the number of neurons depends on number of input parameters as described later. The number of neurons in the second (hidden) layer had been optimised during the test runs to obtain the best generalization capability of the network. The output layer consists of one neuron which signal represented the output nitrogen or phosphorus concentration from the wetland. In the first aspect of modelling (see above) different physical and chemical parameters measured at the input to the wetland was put as an input to the network. The number of parameters was determining the number of neurons in the input layer and was between 4 and 12. In the case of second aspect modelling the number of neurons in the input layer was determined by the length of time-dependent input vector of nitrogen or phosphorus concentration. Results of modelling using data from two objects are presented. The first is Kodijärve wetland (Estonia) and the second is Kompsasuo (Finland).

In case of nitrogen modelling the NH_4 concentration at the input generally had a higher contribution to the output than NO_2 or NO_3 suggesting that insufficient aeration of the wastewater in the wetlands occurs (Figure 2).

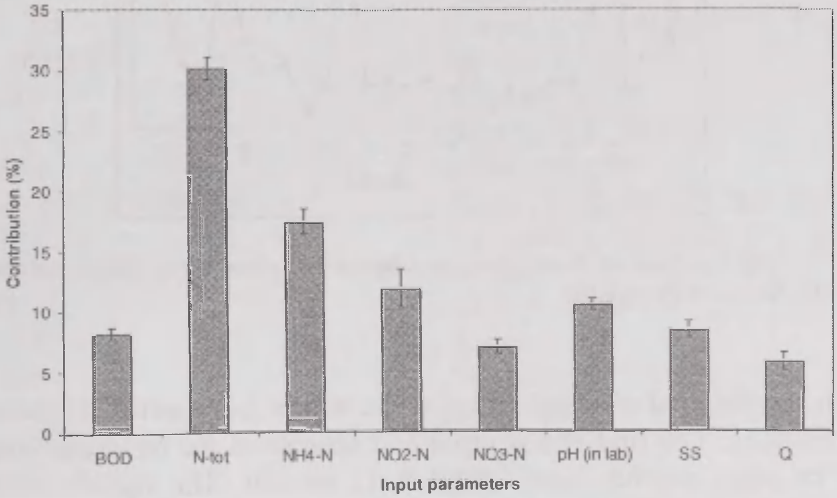

Figure 2. Contribution of input water parameters to total nitrogen concentration at the output from the Kodijärve wetland.

In case of phosphorus no significant differences in input signals was obtained. The main removal mechanism of phosphorus is adsorption and probably this process depends stronger on conditions present in the wetland soil than used input parameters. The calculations of the residence time distribution function (RTD) are similar to the previous. The difference is in the input vector, which represent a time dependence of concentration at the input for assumed period (12 and 24 months) and the associated output value occurred at the end of assumed time. An example of RTD function of nitrogen and phosphorus for 24 months in Kodijärve wetland is presented on Figure 3.

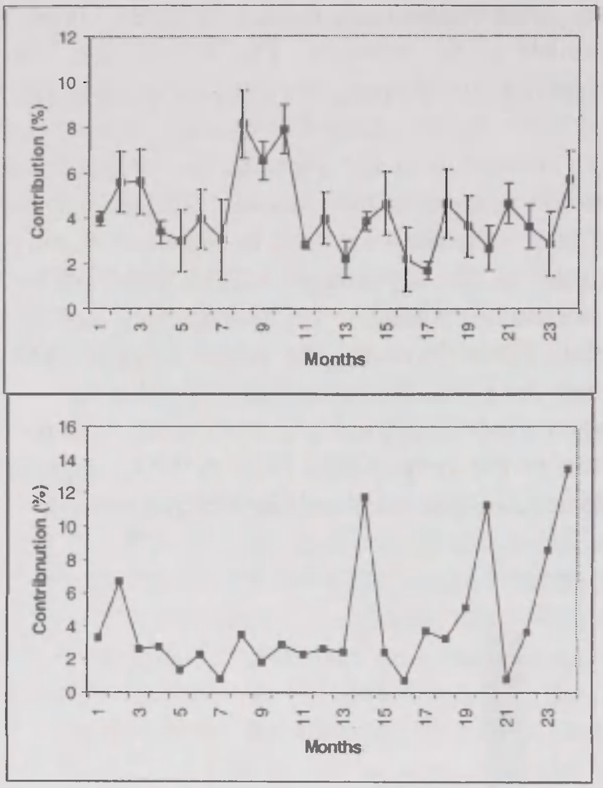

Figure 3. The RTD function for total nitrogen (above) and phosphorus (below) in the Kodijärve wetland for the period of 24 months.

As one can see the total nitrogen signal at the output from wetland is composed by two contributions. The first one is rapid and represents the residence time ca. 2–3 months. The other one has value about 9–11 months. The signals older than 12 months are negligible (the error bars comparable with values itself).

In case of phosphorus, the calculations show that very old components have a significant contribution to output. The possible conclusion is that the residence time of the phosphorus in the wetland is longer than 12 months.

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