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Ecological Engineering 19 (2002) 41–61

ECOLOGICAL ENGINEERING

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Factors affecting phosphorus retention in small constructed wetlands treating agricultural non-point source pollution

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Received 2 October 2001; received in revised form 22 February 2002; accepted 27 February 2002

Abstract

Four surface flow constructed wetlands (CWs) have been intensively investigated for phosphorus retention, from 3 to 7 years in the cold temperate climate of Norway. The aim of this study was to identify factors that affect phosphorus retention from non-point sources. The wetlands were located in first order streams, with surface areas of 0.06–0.4% of the watershed (CW-area 350–900 m^2). Volume proportional composite samples were taken from inlet and outlet, and sedimentation plates were used in selected areas. The average retention of total phosphorus for the individual CWs was 21–44% of input, despite the high hydraulic load (mean load was 0.7–1.8 m per day). This equals a retention of 26–71 g phosphorus m^{-2} surface area per year. A first-order model was fitted to the data giving an average removal constant, k, of 214 m per year. However, the constant increased with increasing hydraulic load due to the simultaneous increase particle settling velocity. Hence, retention increased in spite of increasing hydraulic loads. Moreover, linear multiple regression models showed that retention was influenced by several external variables, e.g. input of phosphorus, season, phosphorus content on suspended solids and phosphorus settling velocity. The results suggest that the first-order model is less suitable to estimate phosphorus retention in similar gravity fed wetlands. The best of the proposed statistical prediction models, reproduced observed data from two independent test-CWs with a deviation of 0.1%. The investigation shows that small wetlands are a useful supplement to best management practice on arable fields. However, the present study focuses on the necessity to investigate how pollutants enter wetlands. Such knowledge can then be used to suggest improvements of wetland layout. © 2002 Elsevier Science B.V. All rights reserved.

Keywords: Aggregates; Arable fields; Cold climate; Dissolved phosphorus; Prediction model; Sedimentation; Selective erosion; Stormwater; Surface-flow wetlands; Total phosphorus

1. Introduction

1.1. Particle and phosphorus retention

When initiatives such as best management practice (BMP) on arable fields are insufficient or fail, constructed wetlands (CWs) could be the 'last

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PII: S0925-8574(02)00014-9

resort' to mitigate the effect of agricultural pollution reaching vulnerable recipients. Phosphorus (P) is given special attention, because it is often the limiting nutrient for algal growth in freshwaters (e.g. Berge et al., 1997; Correll, 1999). However, in order to fit into the typical small-scale Norwegian agriculture, the CWs have to be small. Hence, clay particle retention will be low because of high hydraulic loads (Chen, 1975; Novotny and Chesters, 1981). Since phosphorus is often closely adsorbed to clay particles (Kronvang, 1992), phosphorus retention in small wetlands may also be low.

However, the analysis of sediments from several small Norwegian wetlands showed that the clay content in CW-sediments approximated the clay content in the topsoil, and mean annual retention of soil particles was 57–71% (Braskerud, 1998). Sedimentation of aggregates explained the high clay content in CW-sediments (Braskerud, 2001a). As a consequence of the positive results, Norwegian authorities encourage farmers to build wetlands by contributing 70% of the establishment costs. The high clay particle retention did, however, question the predictability and use of models that include constants for settling velocity or removal rates.

The first-order area model is widely used, and is related to the previously mentioned sedimentation models for suspended solids. According to Kadlec and Knight, (1996), the first-order area model can be used to describe phosphorus removal in wetlands.

$$C_{out} = (C_{in} - C^*)\exp(-kq^{-1}) + C^* \quad \text{Mod1}$$

where k is the removal rate constant (m per year), q is hydraulic loading (m per year), C_{in} and C_{out} concentration of phosphorus in inlet and outlet (mg l^{-1}). The background value (C^*) is often deleted (Kadlec and Knight, 1996), which simplifies the equation to $C_{out} = C_{in}\exp(-kq^{-1})$. Hydraulic load ($q$) is runoff ($Q$) divided by CW-surface area (A). The main assumptions for using Mod1 are that the constant k does not depend on variables q and C_{in}, and that plug flow is a reasonable approximation of the hydraulic conditions in the wetland (Kadlec, 2000). There is, however, a growing awareness that the removal rate constant k is not a real constant, but changes with the hydraulic load and sometimes with the inlet concentration of pollutants. The common presumption of plug flow is also not fulfilled (Kadlec, 2000; Persson et al., 1999). This could lead to serious implications for process modeling.

1.2. The aim of this study

This paper aims at analyzing: (i) the interaction between wetland retention efficiency and phosphorus loss from arable watersheds, and (ii) to what degree erosion processes in the watershed play a similar role in the retention of phosphorus, as they do for soil particles. Specific attention is directed to the effect of hydraulic load on phosphorus sedimentation. The influence of hydraulic load on the first-order model rate constant is evaluated, and important factors affecting retention are analyzed through simple and multiple linear regression analysis. Finally, alternative models for the prediction of phosphorus retention are presented and evaluated.

2. Methods

This investigation differs from other investigations with regard to one or several of the following factors: (1) Wetlands have high hydraulic loads; (2) climate is cold temperate; (3) water flows into the wetlands naturally (i.e. not pumped); (4) sampling of water quality is flow proportional; (5) phosphorus loss is from arable land; (6) time-series are relatively long.

2.1. Site description

2.1.1. Watersheds

Six watersheds in central and southern Norway were studied; Berg (A), Finsrud (B), Kinn (C), Storlaus (D), Flatabekken (F) and Grautholen (G). Most of the arable land in Watersheds A–D is used for barley and oat production, even though Watershed A includes some dairy farming. Site F is located near Trondheim in central Norway, and is characterized by extensive agricultural production, since the runoff feeds the city's drinking water reservoir. The opposite is true for Watershed G, located close to Stavanger in southwestern Norway. This part of the country has the highest density of dairy cattle (two animal units ha^{-1} arable land).

Arable topsoil samples were taken from all watersheds. Silty clay loams dominated arable fields in Watersheds A, B and C, while D also included soils with coarser texture. Silty moraine dominated F and G (Table 1). The highest phosphorus content in topsoils was observed in Watersheds F and G, due to the relatively high manure application rates. Manure from former fox farms in Watershed F still contributes to the high phosphorus values in the soils. In Watershed C, more than 50% of the arable land has been artificially leveled. Generally, leveled fields have lower aggregate stability and are more prone to erosion than corresponding non-leveled fields (Lundekvam and Skøien, 1998). The soil particle content in streams with runoff from cereal production was usually higher than from grass covered watersheds (Table 1). The pH level in the topsoil varied from 4.9 to 6.9 (Braskerud, 2002a). Non-arable land primarily consisted of forests on shallow soil.

Five of the watersheds, viz. A–F, are located in areas with a cold temperate climate, with 5 months of frost per year. In Watershed G, the average temperature is just above the freezing point even in mid winter, due to the influence of the North Sea. Annual precipitation varied from 750 mm in Site A to 1400 mm in Site G. Additional characteristics of the studied watersheds are given in Table 1.

2.1.2. Wetland design and characteristics

The CWs, located in the first-order streams, were made by expanding the stream banks. Low dams were placed between CW-components where creation of a water surface area was wanted (Fig. 1a). Streams in Watersheds C and G were tile-drained, and the CWs were placed at the outlet of the tile-drains.

Each CW had a composite water sampler at the inlet and the outlet. CW-G, however, had one additional site, which provided data from two CWs, G1 and G2. Wetlands B and D were used for model validation (see end of the paper 3.4.3). Sampling was done as in the other wetlands until 1997. Due to poor frost protection, the winter season could not be included. CWs B and D were situated close to CWs A and C, respectively.

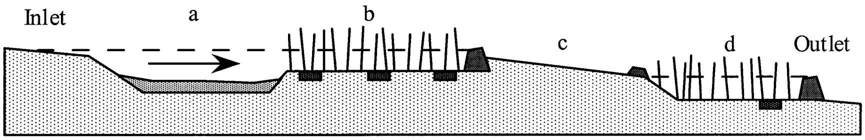
CWs A, C and F were rectangular, with L:W ratios from 12:1 to 5:1. CW-G, however, was divided into several cells with L:W ratios from 1.2:1 to 2.4:1. The water surface area (A) increased with increased runoff. This was accounted for in CWs A–F. Wetland G components were divided by several permeable dams which made a correction difficult (Table 2).

Aquatic submerged and emerged vegetation was either planted or naturally introduced. As

Table 1
Watershed characteristics (watershed area, average concentration of suspended solids in stream (SS) per season, percentage of field under cultivation, major agric. production), and content of clay, total phosphorus (TP) and plant available phosphorus (P-AL) in the plow layers

Site	Watershed (km^2)	SS-stream ($mg\ l^{-1}$)	Arable land (%)	Agriculture production	Clay in soil (%)	TP in soil ($g\ kg^{-1}$)	P-AL (% of TP)
A	1.48	20–520	17	Cereals/milk	28	0.7–1.0	4–15
B	0.86	19–570[b]	11	Cereals	28	0.8–1.0	7–15
C	0.50	39–559[a]	27	Cereals	20–33	0.6–0.9	6–14
D	0.9	76–420[b]	28	Cereals	18–29	0.6–0.8	5–11
F	1.03	10–153	14	Beef/horse	6	1.2–3.2	6–63
G	0.22	26–170	99	milk	6	0.6–1.2	3–33

Figures in intervals are observed min and max. B and D are model validation test sites.
[a] Extreme episode excluded (see text).
[b] Winter seasons not measured.



(A)

(B)

Fig. 1. Components of the constructed wetlands (A): (a) sedimentation basin, (b) wetland filter, (c) overflow zone and (d) outlet basin. At low water flow, water depths were 1, 0.2–0.8, 0 and 0.2–0.8 m in components a, b, c and d, respectively. Four sedimentation plates were pressed into the sediment in component b and d. (B): The V-notch in the wetland outlet measures runoff (CW-B, photo B.C. Braskerud).

time passed, every component in the wetlands had some vegetation (Fig. 1A). Vegetation cover increased from 50 to 100% throughout the experiment, with exception of CW-F, which increased from 20 to 42% (details are presented in Braskerud, 2001b, 2002a).

2.2. Sampling procedure

2.2.1. Water flow measurements

A 120° V-notch weir was installed in the dam outlet of every wetland (Fig. 1B). A Campbell CR10-logger with a pressure probe recorded the discharge. The logger also monitored precipitation, water and air temperature, and controlled a water flow proportional sampling system in the inlet and outlet.

2.2.2. Flow proportional composite sampling

The logger sent signals to a timer unit (e.g. one signal per 40 m^3 runoff), which started the composite sampler. A centrifugal drain pump transported a 120-ml *sub-sample* to a *sample container*. On average, 11 sub-samples were collected daily (variation 0.4–230), and stored in the sample container. The sample container was a 27-l plastic bucket with lid. The sample container was placed in a dark, isolated soil cellar. Heating cables kept the water pump and sampling tubes frost-free throughout the winter (not for CWs B and D). The flow proportional composite sampling procedure is detailed in Braskerud et al. (2000).

A *1-l sample* was taken from the sample container, usually at 9–12 day intervals. Two *mini-samples* were taken from the 1 l sample in the field: (i) 100 ml for total phosphorus (TP) analysis, and (ii) 50 ml after filtration through a 0.45 μm cellulose acetate filter (P-free) for dissolved phosphorus (DP) analysis. As a preservative, 4 M H_2SO_4 was added to the mini-samples to a final concentration of 1%.

2.2.3. Sedimentation plates

Three sedimentation plates were placed in the vegetation filter (b) in CWs B, C, D and G1 (Fig. 1A). CW A had four plates, CW-F had five plates. Only one plate was placed in the outlet basin (d) of CW-G2. Plates were made of $25 \times 25 \times 0.9$-cm plastic-coated plywood and pinned to the wetland bottom. Plates were collected each year in June, during a period of no- or low water flow, by wading. Replacement sedimentation plates were placed in undisturbed locations adjacent to previously sampled sites. Plates could not be placed in water deeper than an arm's length if collecting and replacement was to be done under full control. Plates were placed in the longitudinal center of the CWs. An evaluation of soil particle sedimentation on plates showed that the plates correctly estimated sedimentation rates (Braskerud et al., 2000).

Table 2
Constructed wetland size (actual and as percentage of the watershed) and the order and type of components according to Fig. 1a

CW	Established	Size		CW-components
		m^2	% of watershed	
A	1990	900	0.06	a b $_\wedge$
C	1990	345	0.07	a b $_\wedge$
F	1994	870	0.08	a $_\wedge$ c b $_\wedge$ b $_\wedge$ c d $_\wedge$
G1	1993	460	0.21	a $_\wedge$ b $_\wedge$ b $_\wedge$ b $_\wedge$
G2	1993	840	0.38	c $_\wedge$ c $_\wedge$ c $_\wedge$ d $_\wedge$ c $_\wedge$ c $_\wedge$ c $_\wedge$ d $_\wedge$
B	1990	630	0.07	a b $_\wedge$
D	1990	265	0.03	a b $_\wedge$

$_\wedge$ low dams. B and D are test wetlands.

CW-sediment used for TP analyses were taken from the plates.

2.3. Analyses

2.3.1. Chemical analyses

Total-P in topsoil and sediment was determined by using Aqua Regia digestion. An estimate of plant available P (P-AL) in the topsoil was determined by using an ammonium acetate solution (0.1 M ammonium lactate and 0.4 M acetic acid, pH 3.75). In water, TP was determined after digestion with potassium peroxo disulphate solution. Dissolved-P (DP) was determined on 0.45 μm filtered samples. Particulate phosphorus (PP) was determined by subtracting DP from TP. For measuring suspended solids, 500–800 ml from each *composite sample* was filtered through a dried and preweighed Watman GF/A glassfiber filter. The filter was dried at 105 °C for 24 h and the amount of suspended solids on the filter was weighed. Soil texture in agricultural fields was measured by the pipette method (Elonen, 1971). Soil was treated with H_2O_2 and HCl to remove organic matter and oxides, respectively. Dispersion agent was sodium-pyrophosphate.

2.3.2. Statistical methods and approach

The statistical methods applied were simple linear regression and multiple linear regression, in addition to t-tests, ANOVA, and the Wilcoxon/Kruskal–Wallis test. The term '*statistically significant*' is used when statistical tests gave $P < 0.05$. Retention was calculated as the difference between input and output of P-mass transport and presented as changes in percent, or as specific retention. Changes in percent were always converted to a [−100 to 100%] interval to mitigate large negative values, which otherwise would act as outliers. If value IN > OUT, retention was $(100 - \mathrm{OUT} \times 100 \times \mathrm{IN}^{-1})$%. However, if OUT > IN, retention was $-(100 - \mathrm{IN} \times 100 \times \mathrm{OUT}^{-1})$%.

Stepwise mixed selection was conducted among the following explanatory variables: specific load and concentration for TP, suspended solids and total nitrogen, TP fraction of suspended solids, hydraulic load, runoff, first-order removal constant, pH, conductivity, year, CW-age, season, site, and water and air temperature. Residuals were always approximately normally distributed. All calculations and statistical analyses were made on seasonal data from each wetland.

Variance inflation factors (VIFs) were used to test multicollinearity among the explanatory variables in the regression analyses (see e.g. Montgomery and Peck, 1992). Standardized regression coefficients were calculated in order to compare the magnitude of the explanatory variables (see e.g. Montgomery and Peck, 1992).

2.4. Methodological errors

2.4.1. Possible errors in composite sampling

The composite sampling system was not exactly volume proportional, because the centrifugal drain pump stayed submerged. Under high runoff situations, the water table rises inside the tube, which transports sub-samples to the sample container, increasing the sub-sample volume. The deviation was studied for suspended solids under a low flow and a high flow episode in CW-A (Braskerud, 2001a). Flow peaks were 0.4 and 5 $l\ s^{-1}\ ha^{-1}$, and included first-flush flows and hysteresis effects. Individual samples were taken at 3-h intervals for 3 days with ISCO-samplers in the inlet and outlet. Compared with the exact sub-sample volume, the suspended solid content in the composite sampler increased by up to 3 and 29% for the low flow and high flow episodes, respectively. Hence, sub-samples are systematically over-represented under storm runoff situations. This probably affected the values for mass flow entering the wetlands. However, the relative retention was probably not overestimated, because the error was similar in the wetland inlet and outlet. In fact, the relative retention was slightly underestimated in the experiment, and was probably underestimated generally, because the sampling procedure was only determined by the runoff at the outlet. A delay in sampling-intensity as runoff increases underestimates the high concentrations in the inlet due to the first flush. In the same way, the low particle concentration was over-sampled through the recession part of the hydrograph. The deviation due to the composite sampling system was, however, small compared with the rapid changes in concentrations in small streams. The variations from one season to the next could exceed an order of magnitude (e.g. Fig. 2). Many pond and wetland investigations are based on pumped inlet water and/or bi-weekly spot samples, and the dynamics of the watershed are thus not considered.

2.4.2. Changes of TP and dissolved-P

2.4.2.1. Sample container. Due to the composite sampling strategy, the dissolved phosphorus contents in the samples may change due to sorption/desorption to particles and microbiological activity. This effect was studied by taking a 3-l spot sample at low water flow in mid-winter from each of the streams entering into CWs A and C, in addition to a third stream (Braskerud, unpublished data). Mini-samples from the spot samples were immediately preserved as described above, and analyzed the same day. The initial TP-concentrations were 0.023, 0.029 and 0.043 mg l^{-1}, respectively. The dissolved-P content was 48, 55 and 63% of TP. The 3-l spot samples were stored in the plastic bottle at 4–10 °C for 15 days before the phosphorus-analysis was repeated. After that period, the dissolved-P fraction of TP had decreased to 31, 46 and 50% for streams A, C and the third, respectively.

2.4.2.2. Sample preservation. Braskerud (1995) showed that the dissolved-P concentration decreased by 63% in the sample container for samples in the cold part of the year (spring and autumn), and increased by 63% for summer samples. This could be an effect of clogging of the 0.45 μm filter (Lambert et al., 1992). Since the inlet sample contains 50–70% more particles than the outlet sample, the inlet filter is more clogged, the filter diameter decreases, and the particles entering the mini-sample are fewer and

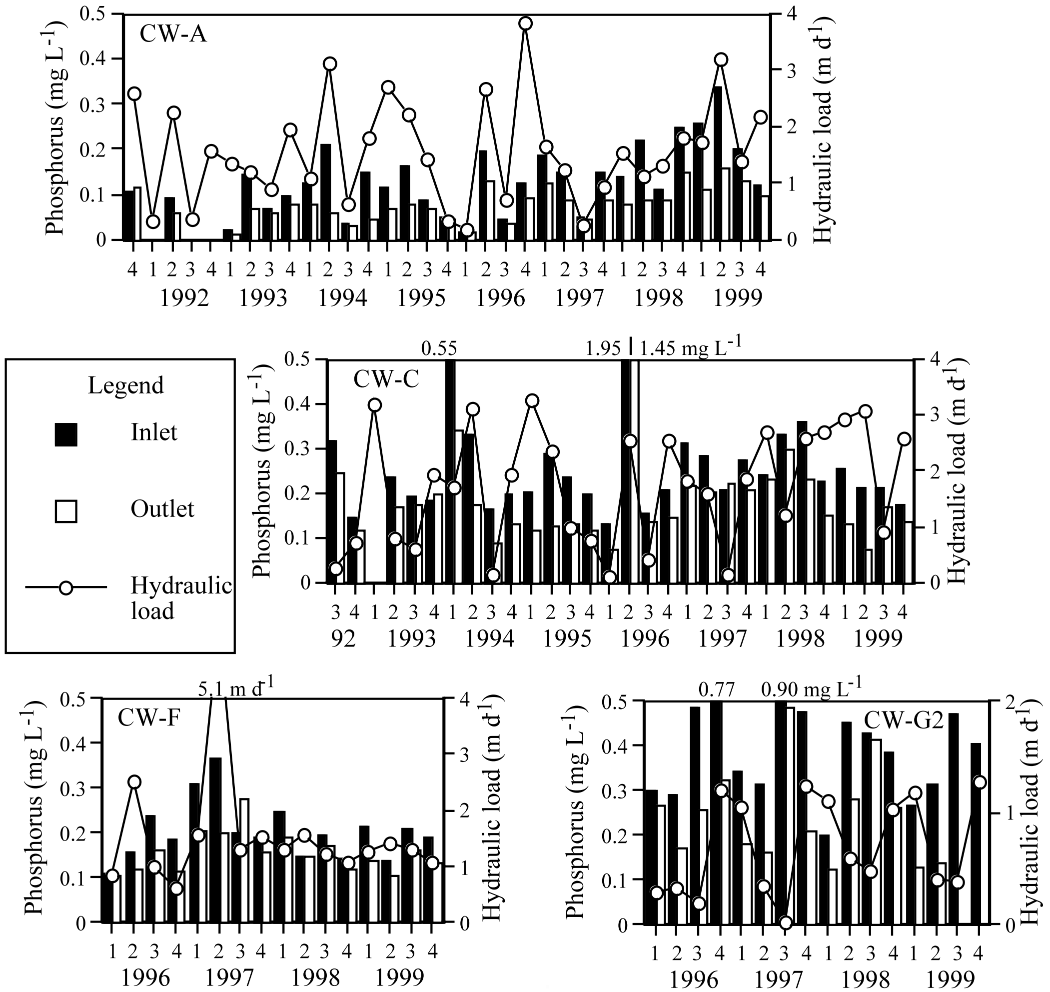


Fig. 2. The average seasonal changes in total phosphorus concentration and hydraulic load rates of Constructed Wetlands A, C, F and G2. Season 1: December February–2: March May–3: June–August, and 4: September–November. Some seasons in the first years of the investigation are missing due to a breakdown of sampling systems. Hydraulic load (q) is average water flow rate (m^3 per day) divided by CW-surface area (m^2) during the season.

smaller. This could result in lower dissolved-P concentration in the inlet samples, because sulfuric acid tends to release phosphorus from the particles (Turtola, 1989). The suspended solid concentration in streams is often higher in the spring and autumn than in the summer (e.g. Braskerud et al. 2000; Braskerud, 2001b). Hence, decreased dissolved-P concentrations in the spring and autumn could be an effect of fewer particles in the sample. However, higher soil particle content in the inlet composite sampler complicates the measurement of dissolved-P even further.

Table 3
Correlation matrix (r) for the input variables in models 2–7 (Table 5), and estimates of the VIFs

	TP_{Cin}	TP_{spL}	SS_{Cin}	SS_{spL}	q	Q	w'	TP/SS	Sum.	VIF
TP_{Cin}	1									1.0–1.9
TP_{spL}	0.65	1								2.2–4.7
SS_{Cin}	0.30	0.58	1							1.9–2.6
SS_{spL}	0.21	0.69	0.89	1						3.2
Q	0.03	0.69	0.47	0.7	1					1.0–2.3
Q	0.63	0.64	0.05	0.14	0.34	1				2.4–2.6
w'	0.16	0.62	−0.53	0.76	0.73	0.33	1			2.2–2.6
TP/SS	0.32	−0.13	−0.53	−0.45	−0.36	0.19	−0.37	1		1.2–2.2
Summer	−0.08	−0.3	−0.18	−0.24	−0.36	−0.32	−0.35	0.09	1	1.2

Suffixes: C_{in}, concentration in the CW-inlet; spL, specific load.

Phosphorus sorption to soil particles may decrease the dissolved-P content, even though desorption also is possible.

Consequently, the dissolved-P contents in this paper must be regarded as not precise (i.e. underestimated). The dissolved-P results presented only give an indication of the P-fractions for TP. The changes in TP were within the deviation of sampling and chemical analysis, and is generally regarded as more robust (e.g. Lambert et al., 1992; Turtola, 1989).

2.4.3. Multicollinearity

A linear multiple regression model assumes independent variables. Apparently, the SS and TP-load are highly related (Table 3). The same holds true for the hydraulic load (q) and the settling velocity (w'). As a rule of thumb, Montgomery and Peck, (1992) claim that VIFs *larger* than 5–10 imply problems with multicollinearity between input variables, that can lead to models with poor prediction. None of the models presented had VIF-values larger than five.

2.5. Exclusion of data

The measurements started in the autumn of 1991 for CW-A, in spring of 1992 for CW-C, and in winter of 1996 for CWs F and G, and include the autumn of 1999 for CWs A–F. Despite earlier measurements in CWs A and C, only data from 1993 and onwards were used, due to several breakdowns of the sampling equipment. In CW-G, data from 1999 were excluded for the same reason. In addition, samples from the following periods were excluded from statistical analysis in this paper:

(i) Summer 1997 (season 3) in CW-G1 and G2, since the 'outlier' analysis indicated errors. In one important episode, the sample containers overflowed several times. This could have led to increased sample concentration (Braskerud, 1995).

(ii) Spring 1997 (season 2) in CW-F: In several episodes the input pump remained too close to the stream bottom, resulting in a possible over-representation of the bottom transport.

(iii) An intense rainfall event in the spring of 1996 resulted in massive erosion at site C: Within 24 h, 6.1-kg phosphorus ha^{-1} arable land was lost, and the flow peak was 1370 l s^{-1} km^{-2}. The median phosphorus loss from arable land in this watershed is 4.6 kg ha^{-1} per year. This *extreme episode* is detailed in Braskerud et al. (2000).

The excluded events in (i), (ii) and (iii) are shown in Fig. 2. In total, 90 seasons (i.e. observations) are included in the analyses of total phosphorus. However, 15 seasons were not analyzed for dissolved-P, three in CW-F and six in CWs G1 and G2.

3. Results and discussion

3.1. Loss and retention

3.1.1. Phosphorus flux through the wetlands

Phosphorus was exported from all watersheds in all seasons of the year (Fig. 2). Phosphorus concentration in Watershed A was lowest in the summer (season 3) and highest in the spring (season 2). In Watershed G, however, phosphorus content was lowest in winter (season 1) and highest in spring. The differences were statistically significant. For the other watersheds, more irregular seasonal patterns were noted. However, when hydraulic load was considered, a much more distinct seasonal pattern was revealed; with a 50% lower load in the summer season compared with the load in autumn, winter and spring. This was due to a statistically significant lower runoff in summer (approx. 60 mm) than in the other seasons (ca. 120-mm per season). The phosphorus concentration in streams was generally higher from watersheds with a high percentage of cultivated land.

Phosphorus losses from Watershed C showed a larger variability and were more episodic than in the other watersheds. This may be a consequence of more erodible soil due to land leveling and soil cultivation practices (e.g. the *extreme episode* in spring 1996 Section 2.5, Fig. 2).

The phosphorus concentration in the streams (i.e. CW-inlets) showed a higher variability than the concentration in the CW outlets (Fig. 2). Moreover, the input concentration in almost all seasons and CWs was higher than the outlet concentration. The high retention rate was notable despite the high hydraulic load.

In this investigation the mean relative annual retention for the individual wetlands varied from 21 to 44% (Table 4). This is higher than previously reported in the Nordic countries. For example, Wedding (2000) investigated three shallow ponds in southern Sweden (depth ca. 0.5–2 m). Time-integrated composite samplers measured the phosphorus flux continuously in the inlet and outlet for a period of 2–7 years. The average hydraulic loads were 77, 160 and 657 m per year and the relative retention was 26, 10 and 16%, respectively. In other small, shallow Swedish and Finnish ponds, the average annual retention was up to 20% (Uusi-Kämppä et al., 2000). The examined Swedish and Finnish ponds are situated in a climate similar to the investigated wetlands, even though precipitation is less than in Norway. Three of 48 US wetlands surveyed by Carleton et al. (2001), fit three of the six items pointed out as specific for this investigation (cold to temperate climate, gravity fed inflow, and watersheds dominated by agriculture). Annual total-P retention varied from 27 to 40%. The hydraulic loads were

Table 4
Average phosphorus retention in the investigated constructed wetlands (CWs): hydraulic load (q), and total phosphorus (TP) in the inlet and the outlet

CW	Data (years)	♀ (m per year)	TP (mg l^{-1})	TP out (mg l^{-1})	k (m per year)	TP load (g m^{-2} per year)	TP retention	
							%	g m^{-2} per year
A	7	595	0.17a	0.10	316	97	41b,a	40
C	7	661	0.25a	0.17	255	178	32b,a	57
F	4	588	0.22a,a	0.17	152	124	21b,a	26
G1	3c	445	0.43a	0.27	207	191	37b,a,a	71
G2	3c	241	0.43a	0.24	140	106	44b,a	47

k is the first-order area model removal constant according to Mod1 (see text).
a, Different letters indicate a statistically significant difference.

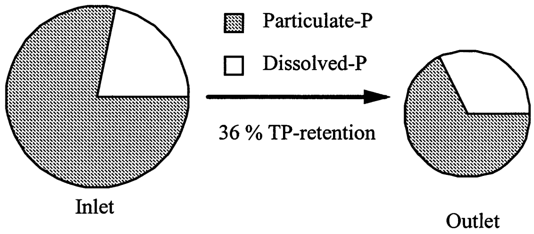


Fig. 3. The share of particulate-P and dissolved-P of the total phosphorus (TP) concentration in the inlet and outlet of the four tested wetlands ($n = 75$).

not always specified, but were probably less than 6 m per year. Hence, wetlands in this study performed similar to, or better than, those described in other studies, even though the hydraulic load was several times higher. Better TP-retention performance of wetlands compared with ponds could be due to difference (1) in design and/or (2) in watersheds.

CWs are shallower and have more vegetation than ponds (Uusi-Kämppä et al., 2000). Water detention time and particle settling distance increase with depth. Hazen (1904) was probably the first to state that increased depth did not significantly influence retention through sedimentation. However, resuspension of sediments in shallow ponds may occur under storm runoff situations. Braskerud (2001b) showed that resuspension became negligible as the vegetation cover in CWs A, B, C, and D increased. Hence, CWs should be shallow to allow rapid particle settling and to enable macrophyte growth. The effect of watershed is discussed later (Section 3.1.2 and Section 3.3.4).

3.1.2. Phosphorus fractions

As an average for the entire investigation, particulate-P ranged from 51 to 88% of TP, for Watersheds $F \ll G < A < C$, respectively, or 78% as an average for all CWs (Fig. 3). The difference in particulate-P between CWs reflects the difference in plant available phosphorus in the topsoil (P-AL in Table 1), with higher availability in soils of watershed F.

The median TP content of soil particles (SS) in the streams ranged from 0.11 to 0.64% for Watersheds $A \leqslant C \ll G \leqslant F$, respectively. Differences between the streams reflected the TP content in the arable soil (Table 1). However, the TP: SS ratio in stream water was often higher than the ratio phosphorus to particles in the topsoil. This indicates selective erosion and retention processes in the watersheds, and thus losses of phosphorus rich soils.

Approximately 45% of the particulate-P was retained (Fig. 3), compared with only 5% for dissolved-P. Dissolved-P retention was not statistically significant, and was probably underestimated due to methodological problems (confer Section 2.4.2).

The content of dissolved-P in the Swedish and Finnish streams is probably higher than in the Norwegian streams, because erosion in Sweden and Finland is less. This could lead to lower P-retention in ponds compared with the CWs. For example, Wedding (2000) reported of phosphate contents of 64–86% of TP, as an average for the three Swedish ponds. The retention of phosphate varied from 12 to 31% in the individual ponds, as an average of several years. On the other hand, Ydstebø et al. (2000) reported of approximately 90% dissolved-P retention in a Norwegian CW, which was located in the same county as CW-G. This study differs from the other Nordic investigations in two ways, the TP inlet concentration was very high (average 1.53 mg l^{-1}) and the ratio surface area to watershed area was large (1%).

Since particulate-P retention exceeded dissolved-P retention in Wetlands A–G, the dissolved-P fraction increased relatively when water flowed through a small wetland (Fig. 3). This indicates that sedimentation was a far more important retention process than uptake by algae and macrophytes.

3.1.3. Phosphorus in the wetland sediment

The average TP content in the sediments was the same in CWs A and C. For Wetlands F, G1, and G2 the phosphorus content in the sediments was higher (Fig. 4). Generally, the phosphorus content in wetland sediments was higher than in the surrounding topsoil. In Watershed F, however, the phosphorus content in the topsoil was higher and the variability larger than in the other watersheds. This was due to the former fox farms and the scattered arable fields in this watershed. Hence, the fields with the highest phosphorus content in the topsoil may not have contributed significantly to the wetland sediment. As a result, it is not possible to exclude F from the general trend mentioned earlier.

Even though the phosphorus contents in the wetland sediments were high, phosphorus contents on suspended solids in the stream were even higher (0.11–0.64%). Hence, the most phosphorus rich fraction was not retained, probably because soil particles or aggregates were too small for sedimentation.

3.2. The first-order model and retention

3.2.1. Problems with the k value

The first-order model (Mod1) can estimate the actual k value, since the phosphorus concentrations in the inlet and outlet were known, in addition to the hydraulic load. Results of annual average k values are presented in Table 4. The background value (C^*) was set to zero, which is a common assumption, though not always true (Kadlec, 2000). On average, the k value in this study was estimated to 214 m per year. As a

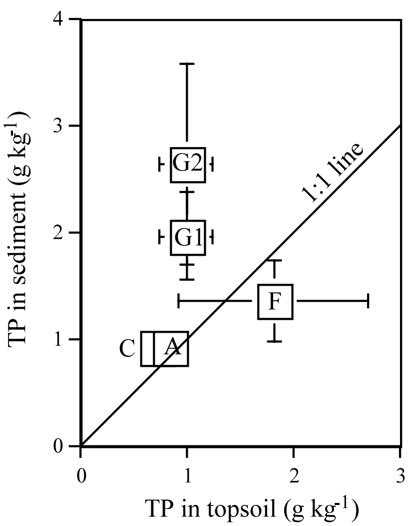


Fig. 4. Relationship between phosphorus (TP) content in the topsoil and in wetland filter sediments of CWs A, C, F, G1, and G2 ($\pm$S.D.). The 1:1 line indicates equal TP content in soil and sediment.

comparison, Kadlec et al. (2000) estimated the k values for non-forested wetlands with a central tendency of $k \approx 10$ m per year. Carleton et al. (2001) found that k values from US storm water treatment wetlands were 11.3 ± 17.6 m per year, and concluded that k was almost identical to k values found in wastewater treatment wetlands. The high k values in the Norwegian CWs are probably due to the high hydraulic loads, often exceeding other references 20 times (Kadlec and Knight, 1996). This is further supported by studies of shallow ponds in Sweden. In ponds with 'low' hydraulic loads (0.2–0.4 m per day), Wedding (2000) found k values of 17 and 23 m per year, compared with 113 m per year for the pond with the 'high' hydraulic load (1.8 m per day).

According to the first-order model (Mod1), the phosphorus outlet concentration increases with increasing hydraulic load. Total-P retention is thus expected to decrease as the hydraulic load increases. The performance of Mod1 is illustrated in Fig. 5, which shows how Mod1 estimate phosphorus retention of observed input TP-concentrations and hydraulic load. The k value used was 214 m per year, which was the average, observed k value (Table 4). When real data from this study were considered, there was a slightly positive correlation between TP-retention and the hydraulic load (Fig. 5). However, there were differences between the wetlands: On one side, CWs A and G1 clearly had an increase in retention with increased hydraulic load (r^2 was 0.24 and 0.33, respectively; $P < 0.01$). In contrast, CW-F showed no clear response to increased hydraulic load at all ($r^2 = 0.001$ and $P < 0.78$). Wetlands C and G2 fell in between these two extremes (r^2 was 0.09 and 0.19, respectively; $P < 0.18$).

Fig. 6 shows that seasonal k values often increased with increased hydraulic load (q). Values for k often increased with increased hydraulic load in CW-F too. The increase was, however, not significant ($P < 0.19$), probably due to the high content of dissolved-P in this stream (49%). A multiple linear regression analysis with q and the phosphorus concentration in the inlet (C_{TPin}) as explanation variables was performed Eq. (1). The standardized variable coefficients were:

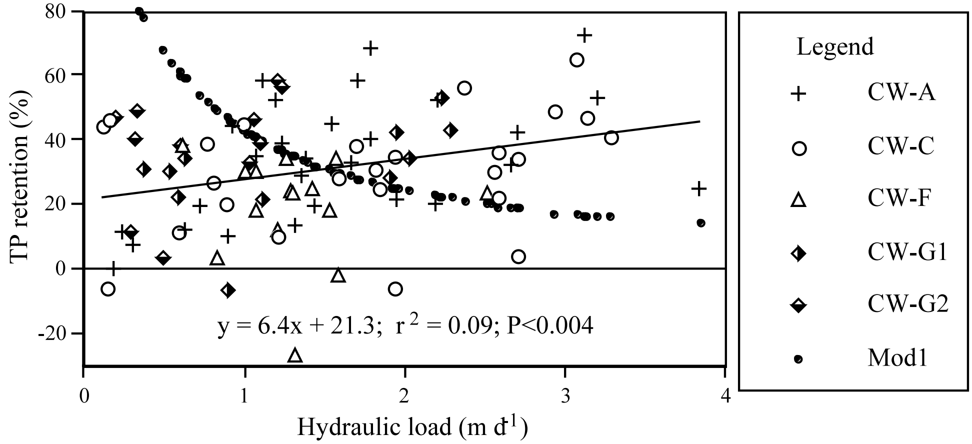


Fig. 5. Observed relationship between P-retention (%) and hydraulic load for the constructed wetlands (CWs), and according to the first-order area model (Mod1) with $k = 214$. Negative retention was net loss of phosphorus from the CWs.

$$k\text{-value} = 0.72q + 0.14C_{\text{TPin}}, \quad r^2 = 0.55(n = 90) \tag{1}$$

In addition to a clear hydraulic load effect, the results showed that k was positively correlated and almost statistically significant ($P <$ 0.001), ($P < 0.062$) with the inlet phosphorus concentration. The findings of Kadlec (2000) and other scientists that k is not a true constant, but changes depending on the hydraulic load, is thus supported by this investigation.

3.2.2. The constant k and particle settling velocity

For retention of particles, the first-order removal constant, k, equals the settling velocity, w (Kadlec and Knight, 1996). Since particulate-P dominates total phosphorus in this study (Fig. 3), it would be reasonable to use k as an estimate for w, further denoted as w'. Moustafa et al. (1996), Carleton et al. (2001) suggested a similar approach.

It has earlier been shown that the settling velocity for clay particles in CWs A and C increases with the hydraulic load (Braskerud, 2001a). Average k-values are presented in Table 4. If $k = w' \approx w$, then the average phosphorus settling velocities were approximately the same as for fine silt particles (2–6 μm), according to Stokes' Law.

3.3. Factors affecting the phosphorus retention

3.3.1. Statistical models

Statistical models can be an alternative to the first-order model (Mod1). In this study, factors affecting phosphorus retention in small wetlands were examined. Retention is often described as a change in the CW-outlet concentration, as specific retention or as relative retention. The three response variables are used in an analysis of factors affecting retention. Moreover, first regressor variables are selected manually (models 2, 4 and 6) to investigate the function of Mod1 and to make possible candidates for operative prediction models. Secondly, models 3, 5 and 7 are formed after carrying out a stepwise selection of the variables presented in Section 2.3.2. Hence, the statistical program selected the regressors.

On the basis of the findings, five prediction models are evaluated. Note that regression coefficients in Table 5 are standardized, or dimensionless (Montgomery and Peck, 1992), to indicate the importance of the individual variable (coefficients usually vary within the range −1 to 1). Non-standardized coefficients are used in connection with the presentation of the prediction models (Section 3.4.1).

Note that negative coefficients lower the output-concentration in Mod 2 and 3, and thus have a positive effect on retention (e.g., an increased w'

value increases retention). The value for VIF in Mod 2 was 1.0. The other models usually had VIFs below 2.6. Specific TP load in Mod4, however, had a VIF value of 4.7 (see Table 3).

3.3.2. Selected variables

The first statistical model (Mod 2) uses the same input variables as the first-order model (Mod1); with flow weighted concentrations of TP at the inlet and the hydraulic load as explanatory variables (Table 5). The TP-concentration in the wetland outlet (C_{out}) increased with increasing input (C_{in}) of phosphorus, as for Mod1. The C_{out} decreased, however, as the hydraulic load (q) increased. This was the opposite of what could be expected. When the effect of hydraulic load on the relative phosphorus retention was investigated, the result was even more remarkable (Fig. 5). Retention increased as the hydraulic load increased. The result was statistically significant, even though the correlation was weak, and falsifies Mod1 as a prediction model for the investigated wetlands. The same phenomenon was also observed for the retention of soil particles (Braskerud et al., 2000). The result was explained as a combination of selective erosion and transportation processes in the watershed:

(i) Soil erosion and concentration of suspended solids increases with runoff.

(ii) Streams carry more and coarser sediments and aggregates as runoff increases.

The stream water transports the eroded material to the wetlands, where the combination of factors (i) and (ii) increases sedimentation, despite lower detention time. As a conclusion, erosion in the watershed influences the retention of phosphorus since a major part is adsorbed to soil particles (Fig. 3). This may also explain the unexpected result of Mod2. More precisely, the hydraulic load (q) in Mod2 may be an expression of the transport of phosphorus, rather than q as such. This was supported by a regression model (not shown) in which q was exchanged with runoff (Q). With this model, the TP-concentration in the CW-outlet decreased with increasing Q. Hence, the TP-concentration in the CW-outlet seems to depend on processes in the watershed.

Model 4 (specific retention, Table 5) includes the effect of q and Q. The results show that retention decreased as q increased, while increased Q led to increased retention. This result is in accordance with what could be expected.

For relative retention (model 6), q was found to be insignificant, while both increased Q and input of soil particles increased retention. Similar results were obtained when the concentration variable was replaced with specific load. However, the coefficient of determination was very low, indicat-

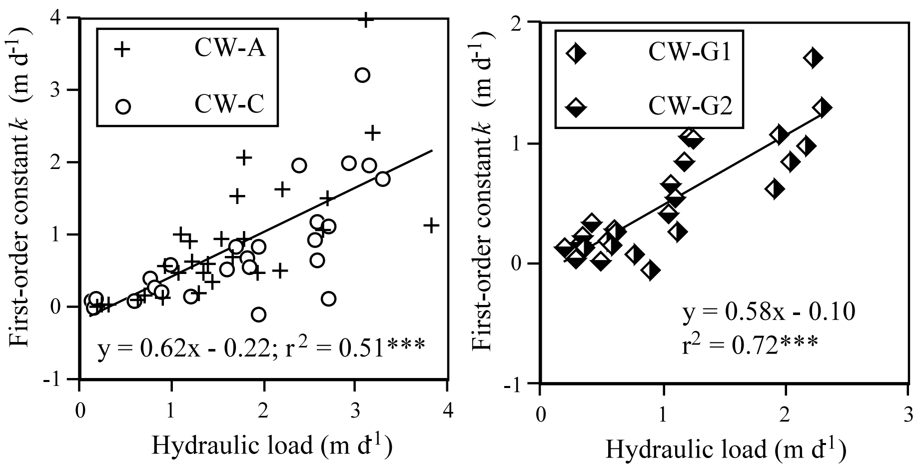


Fig. 6. Scatter plot of the relationship between the first-order removal constant k and hydraulic load in Wetlands A and C (left graph), and G1 and G2 (right graph).

Table 5
Multivariate regression models with standardized coefficients. Dependent variables were total phosphorus (TP) outlet concentration, specific retention and relative retention

Model	y	x_i							
		TP	SS	q	Q	w'	% TP of SS	Summer	V^2
2	TP_{Cout}	0.85_{Cin} ***		−0.13*					0.73
3	TP_{Cout}	0.81_{Cin}***	0.15_{Cin}*	0.27***	NS	−0.48***	0.22***	−0.10	0.87
4	TP_{spR}	0.73_{spL}***	0.32_{spL}***	−0.19**	0.14*				0.85
5	TP_{spR}	0.85_{spL}***	NS	−0.41***	NS	0.54***	−0.05*		0.96
6	$TP_{\%}$	-0.07_{Cin}NS	0.48_{Cin}***	−0.04NS	0.34**				0.29
7	$TP_{\%}$	0.13^{*}_{Cin}	NS	−0.51***	NS	1.02***	NS	0.19	0.70

Regressors (x_i) are TP, suspended solids (SS), hydraulic load (q), specific runoff (Q), the settling velocity of TP (w'), ratio TP of SS, and summer season (June–August). All regressors are seasonal ($n = 90$). Suffixes: $C_{in/out}$, concentration in the CW-inlet/outlet; spR, specific retention; $_{spL}$, specific load; and %, relative retention. Specific load is $C_{in}\ Q\ A^{-1}$ (mass per m^{-2}). Statistically significant regressor: *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$; NS, non-significant. Summer is a dummy variable (0 or 1).

ing that other variables are needed to explain retention better.

3.3.3. Stepwise selected variables

When w' was included as an explanatory variable, the variable runoff (Q) became statistically insignificant (Mod3, 5, and 7). However, r^2 increased considerably. In these models, hydraulic load (q) negatively affected TP-retention and the factors *TP:SS ratio* and *summer season* became statistically significant. More precisely, retention decreased with increasing phosphorus to particle ratio, probably because more of the phosphorus was dissolved. Summer affected retention positively, even though TP-retention was lower in this part of the year. Low retention was probably due to lower w' in the summer compared with the rest of the year (0.2 and 0.8 m per day, respectively; $P < 0.001$). However, an increase in aggregate stability and size in summer can probably compensate some of the lower settling velocity. Aggregates are largest and most stable during the summer (Lundekvam and Skøien, 1998; Stone and Walling, 1996). The same is valid for flocs (or aggregates) in rivers (Droppo and Ongley, 1994).

This corresponds to the analysis of the relative clay particle retention in CWs A and C (Braskerud, 2001a): as the hydraulic load increased, the retention of clay particles decreased. However, as runoff increased the input of clay aggregates with higher settling velocities also increased. The net effect was increased clay particle retention. A similar process presumably also occurs for phosphorus. Moreover, the factor *summer* affected clay particle retention positively (Braskerud, 2001a). The other explanatory variables listed in Section 2.3.2 were not statistically significant, and thus not included in the models.

3.3.4. The relationship between watershed factors, CW-size and retention

Obviously, the input of pollutants affects retention. Generally, there is a significant correlation between C_{in} and C_{out}, and between specific load in and out (e.g., $r^2 = 0.71$ and 0.82 in this study, respectively). However, this correlation 'overshadows' the effect of other variables. Relative retention mitigates this effect (Mod7), and probably highlights the best explanatory variables. In this case, *TP-concentration* had some influence, but it was far from dominant. Moreover, the *specific phosphorus load* was not significant. Hence, *settling velocity*, *hydraulic load* and *summer* are put forward as key explanatory variables. Flocculation processes inside the CWs could be important. However, Braskerud (2001b) was not able to detect any effect of clay flocculation in CWs A, B, C, and D. Wetlands located in low order streams probably receive soil particles as aggregates previously formed by soil processes.

Hence, this study suggests that erosion and transportation processes in the watershed may be of strong significance for P-retention. This means that the factors which influence soil erodibility are of particular importance, e.g. field slope and length, soil texture and aggregate stability, preferential flow through the soil matrix, situation under snow melt, precipitation intensity after soil tillage, and type of crop (see e.g. Lundekvam and Skøien, 1998; Øygarden, 2000). The influence of those factors has not been specifically studied in this investigation.

The ratio wetland-area: watershed-area was often used as an explanation for retention performance (e.g. Uusi-Kämppä et al., 2000; Carleton et al., 2001). Retention is assumed to increase as the relative wetland surface area increases, due to a reduction in hydraulic load. With regard to hydraulic load, the Norwegian wetlands are small both in area and relative to the watershed area. One rule of thumb specifies that stormwater wetland size should be 1–5% of the contributing watershed (Kadlec et al., 2000). This is far from the case in the investigated wetlands (Table 2), and also for the ponds and wetlands constructed in Finland and Sweden, which are usually less than 0.3% (e.g. Uusi-Kämppä et al., 2000; Wedding, 2000). However, even US wetlands are often small. Carleton et al. (2001) presented data from 48 stormwater treatment wetlands in the US. The wetland to watershed ratios for the US wetlands were usually less than 0.1%.

The retention data from the US wetlands show that TP-retention is highly scattered for wetland to watershed ratios under 0.1% (range −55 to 82%). A similar response was observed for CWs A, C and F, which are quite similar with respect to the wetland to watershed ratios (0.06–0.08%). For these three wetlands, TP-retention *decreased* as the relative surface area increased (Tables 2 and 4). Carleton et al. (2001) mentioned several factors responsible for the substantial variations, from factors in the wetlands, via the climate to factors in the watershed. However, in their survey the influence of the various factors could often not be distinguished. In the present study, sampling methods and even wetland form are comparable. It is therefore likely that the difference in retention between the CWs was due to factors within the watershed.

The effect of the watershed on the wetland retention performance sees retention from a new viewpoint. Can we plan or manage our rural and urban watersheds in a way that stimulates retention in CWs, e.g. by increasing the soil aggregate (or floc) size and stability? Often wetland or pond literature focuses on design (e.g. Kadlec and Knight, 1996; Braskerud, 2002b). The present study focuses on the necessity to investigate *how* pollutants enter wetlands in order to be able to suggest layout improvements. A closer co-operation with scientists investigating the loss of pollutants from watersheds is highly welcome.

3.4. Prediction models

3.4.1. Operative models

An operative retention model should predict retention effectively. However, this requires that it is possible to obtain reliable data to the input variables. Runoff and loss of particles and phosphorus are relatively easy data to obtain. The variables in models 2, 4 and 6 were selected to fulfill the last requirement. However, the TP-load had a high VIF in Mod4 (4.7), and the r^2 is not very high for Mod6. As a result, Mod2 is our first candidate:

$$C_{\mathrm{TPout}} = 0.048 + 0.55C_{\mathrm{TPin}} - 0.014q \quad \text{(Mod2)}$$

where q is hydraulic loading rate (m per day), and C_{TPin} and C_{TPout} are concentration of TP in the inlet and outlet (mg l^{-1}). Models developed after carrying out a stepwise regression had high r^2, and low VIF values (models 3, 5, and 7). However, estimating the phosphorus settling velocity, w' is not a straightforward process. Eq. (1) estimates w', but probably reduces the predictability of the models, because regression lines may be site specific. As long as there are no reliable methods for field observations or measurements that could estimate w', the prediction version of Eq. (1) is the only alternative:

$$w'\text{-value} = -0.39 + 0.60q + 0.70C_{\mathrm{TPin}} \quad (2)$$

where q and C_{TPin} are the same as Mod2. Even though they are not very operative at the moment, prediction models 3, 5 and 7 are presented in Table 6.

To omit possible multicollinearity problems between variables, a new simple linear regression model is suggested (Mod8, Fig. 7). Results showed that the r^2-value was 0.82, compared with 0.87 in Mod4. Runoff or q is included in the model indirectly, since TP-load increases with runoff or q (Table 3). This model is the easiest to use, since data on phosphorus loss from watersheds are often available or can be estimated. In addition, the wetland surface area is needed.

3.4.2. Input values

Caution should be taken when using the models, and the background data presented under 'site descriptions' should be taken into consideration. In addition, input data were average *seasonal* (3 month) and in the following intervals (min–max):

- TP-concentrations (mg l^{-1}), 0.02–0.77.
- Specific TP-load (mg m^{-2} per day), 4–1700, recommended 30–800.
- Specific SS-load (g m^{-2} per day), 4–1668.
- k or w' values (m per day): 0–3.9.
- Hydraulic loads, q (m per day), 0.1–3.8.
- Runoff, Q (mm per season), 2–544.
- Specific TP-retention (mg m^{-2} per day), −98 to 563.
- Relative retention (%), −27 to 72.

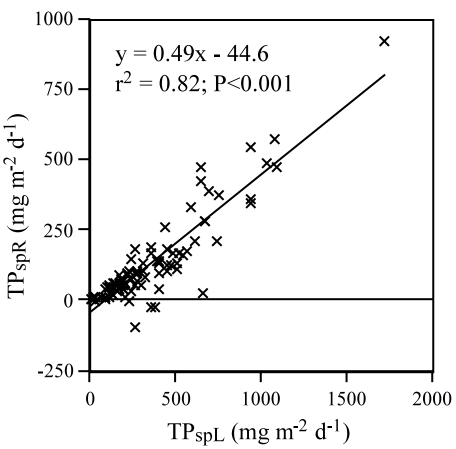


Fig. 7. Mod8 has specific TP-load as the input variable (abbreviations, see Table 6). The model is estimated of ninety seasons from Wetlands A, C, F, G1 and G2.

3.4.3. Validation of regression models

3.4.3.1. Validation dataset. The first-order model (Mod1) and the statistical Models 2, 3, 5, 7 and 8 were compared with data from CWs B and D not included in the model building. A total of 22 seasons, 12 from CW-B and 10 from CW-D, are presented in Fig. 8.

Mod2 usually overestimated the outlet concentration in CW-B, while the opposite occurred in CW-D. The outlet concentration in the summer season was usually estimated correctly, while spring was overestimated and autumn underestimated. A similar result was achieved for Mod3, however, with less deviation. Opposite results were observed for Mod5 and Mod8. Hence, those models tend to overestimate the retention in the smallest wetland D, and in the autumn. The annual average hydraulic loads in CWs B and D were 438 and 1241 m per year, respectively. Hence, regarding hydraulic load, CW-D differs from the model wetlands (confer Table 4). On the other hand, the validation supports the insignificant effect of hydraulic load on the retention in small wetlands with high hydraulic loads.

For specific loads of less than 30 mg TP m^{-2} per day, Mod8 usually estimated a net loss of phosphorus from the CWs, even though retention was observed. Hence, Mod8 is probably not linear for low specific loads. A similar error was observed for Mod5. However, the deviation was not as large as for Mod8. The relative retention (Mod7) was modeled well, without any serious deviation. Mod1, however, usually underestimated retention in both wetlands.

In autumn 1996, CW-D had input values larger than the model wetlands (Section 3.4.2). The hydraulic load was 5.1 m per day, the input concentration 0.8 mg l^{-1}, and the specific load 4050 mg TP m^{-2} per day. Models 2, 5 and 8 did not predict autumn 1996 well (Fig. 8). The other models, however, predicted this season reasonable well. This autumn was excluded from the evaluation of the models (Table 7).

Observed values from CWs B and D were also compared with the estimated data according to the first-order area model (Mod1) and models 2, 3, 5, 7, and 8 in Table 7. The predictability of the

Table 6
Multivariate regression models with estimated regression coefficients for models 3, 5 and 7

Model	y	x_i								r^2
		Intercept	TP_{in}		SS_{in} (mg l^{-1})	Q (m per day)	w' (m per day)	TP/SS (%)	Summer (yes/no)	
			mg l^{-1}	mg m^{-2} per day						
3	TP_{Cout}	0.003	0.52		0.00013	0.029	−0.061	0.06	−0.01/0	0.87
5	TPspr	8.1		0.457		−77.1	120.2	−23.9		0.96
7	$TP_{\%}$	23.2	16.6			−10.1	26.0		4.1/0	0.70

Dependent variables were total phosphorus (TP) outlet concentration, specific retention and relative retention. Regressors (x_i) are TP, suspended solids (SS), hydraulic load (q), the settling velocity of TP (w'), TP/SS ratio, and summer season (June–August). Suffixes: $C_{in/out}$, concentration in the CW-inlet/outlet; spR, specific retention; $_{spL}$, specific load, and %, relative retention. Specific load is $C_{in}QA^{-1}$ (mass per m^{-2}).

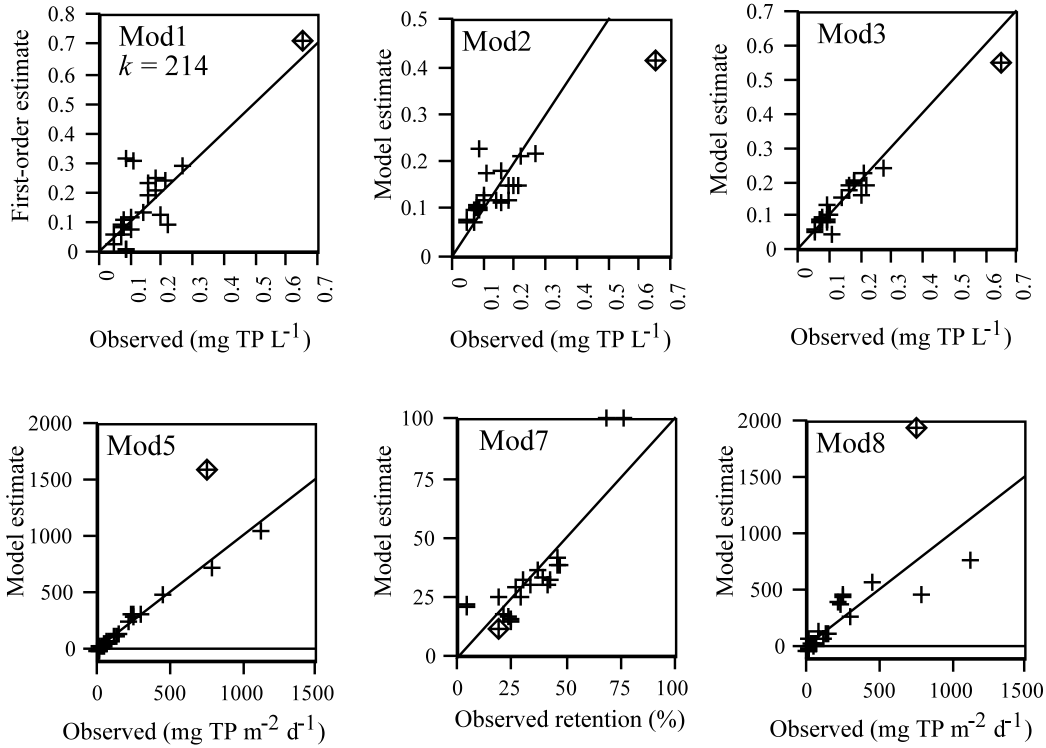


Fig. 8. Evaluation of the prediction of Models 1, 2, 3, 5, 7 and 8 using input data from CWs B and D. The 1:1 ratio line indicates the perfect fit. Excluded season marked by square (see text).

models was compared with each observed season for the validation dataset (CWs B and D) and the model dataset (CWs A, C, F and G). The relative deviation is presented as average or absolute figures. The average deviation indicates the precision of the model. Absolute deviation, however, shows the average deviation in absolute figures, indicating the possible average deviation for a single season. Mod8 had a very high absolute deviation for the validation dataset. This was due to seven seasons with average specific loads higher than 800 mg TP m^{-2} per day. If the seven seasons were excluded, the absolute deviation would decrease to 33%.

3.4.3.2. Model dataset. The statistical models usually predicted the average retention well (0.1–2.6% deviation). There was, however, a tendency of underestimating the retention in CWs A and G2, while the opposite often occurred in CWs C, F and G1. Hence, the models tend to level out the difference between 'high' and 'low' retention wetlands, just like in the validation dataset. In a similar way, low specific TP-loads tend to be estimated as negative retention in models 5 and 8. Mod1 did not predict retention well. The main reason can be seen in Fig. 5, where the modeled TP_{Cout} is used to estimate relative retention.

3.4.3.3. The extreme episode. The TP_{Cout} in CW-C in spring 1996 was 1.45 mg l^{-1}. Mod1, Mod2 and 3 estimated the concentration to be 1.6, 1.1, and 1.4 mg l^{-1}, respectively. The modeled results were relatively close to the observed value. The specific retention was 1270 mg m^{-2} per day. Mod5 and 8 arrived at estimates of 2170 and 2390 mg m^{-2} per day. Retention was 26%, but Mod7 estimated it at 49%. Hence, models 5, 7 and 8 overestimate retention under runoff situations when CWs are overloaded.

3.4.4. First-order model or statistical model for prediction

Generally, the validation data support the statistical models. At present, Mod2 and 8 are the only operative models presented in this study, since the phosphorus settling velocity (w') is hard to estimate. However, with the growing awareness of the environmental function of aggregates and flocs, future research may provide reliable data for w' estimates.

Caution should be taken for seasons that include extreme episodes or have input figures outside the model-dataset range (Section 3.4.2). The first-order model underestimates the TP_{Cout} for low hydraulic loads, and overestimates the TP_{Cout} for high q. This model does not take erosion and transportation processes in the watershed into account, hence, retention always decreases with increasing hydraulic load. Mod1 is also very dependent of the k value used. For example, Carleton et al. (2001) found an average k-value (11.3 m per day) for US storm water wetlands. If this value was used in Mod1 on the validation dataset, the average deviation would have been −56% (confer Table 7). With this in mind, the prediction performance of Mod2 and 8 seems reasonable, if outlier values are omitted.

4. Conclusions

- Four small surface-flow CWs, treating non-point source pollution from arable land, were studied. These reduced the phosphorus content in streams significantly, despite high hydraulic loads.
- Wetland sediments contained similar amounts or often more phosphorus than the cultivated topsoil in the watershed.
- Sedimentation of particle bound phosphorus was the main retention process. In addition, hydraulic load, summer season, and phosphorus content in topsoil influenced retention. Most of the particulate phosphorus entered the wetlands on particles and aggregates, which settled as fine silt.

Table 7
Comparison of observed and *estimated* TP_{Cout} and TP-retention according to Mod1, Mod2, 3, 5, 7 and 8

	Validation dataset [a]	Model dataset
TP_{Cout} observed (mg l^{-1})	0.129	0.162
TP_{Cout} Mod1 (first-order $k = 214$)	0.145	0.146
TP_{Cout} Mod2	0.132	0.161
TP_{Cout} Mod3	0.131	0.166
TP_{spR} observed (mg m^{-2} per day)	206	130
TP_{spR} Mod5	206	130
TP_{spR} Mod8	201	130
$TP_{\%}$ observed (%)	33.5	30.5
$TP_{\%}$ Mod7	34.6	30.5

	Validation dataset		Model dataset	
Relative deviation (%)	Average	Absolute	Average	Absolute
Deviation observed-Mod1	−12.7	43.7	10.0	37.4
Deviation observed-Mod2	−2.4	27.5	0.3	20.6
Deviation observed-Mod3	−2.1	14.0	−2.6	14.6
Deviation observed-Mod5	0.1	22.3	0.1	16.5
Deviation observed-Mod8	5.1	94.0	−0.6	37.5
Deviation observed-Mod7	−1.1	9.4	−0.1	28.0

Deviation in % of average and absolute difference between observed and modeled results. Negative average deviation occurs when model values are higher than observed values. C_{out}, concentration in the CW-outlet, spR, specific retention; and %, relative retention.
[a] Autumn 1996 in CW-D had higher input values than the model dataset, and is excluded.

- The positive effect of increased phosphorus-settling velocity as the hydraulic load increased compensated for the negative effect of shorter detention time. As a result, a net retention was observed for increased hydraulic load.
- The first-order area model did not take processes that led to phosphorus loss in the watershed into account. Hence, the first-order area model constant, k, increased with increased hydraulic load. The first-order area model is thus not suitable for predicting retention in small wetlands receiving water from watersheds with some erosion.
- Several of the suggested statistical models explained and predicted TP-retention well. The models, which included phosphorus settling, gave the best fit. However, due to lack of input data, these models are seldom operative. The two operative models suggested predict total phosphorus retention well compared with the first-order area model. Three additional models can be used if data is provided on how the settling velocity of aggregates and flocs changes with the runoff.

Acknowledgements

The Norwegian Ministry of Agriculture and the Research Council of Norway are acknowledged for their financial support. I would also like to thank B., K. and J. Hoel, E. Halvorsrud, J. Krangsaas, T. Kverneland, R. Dreier Olsen, G. Tveiti, Ø. Johansen, O. Gimmingsrud and B. Dalby for their practical assistance, and B. Kløve, T. Mæhlum, P. Stålnacke, and the anonymous referees for their constructive criticism of the manuscript.

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