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**Particulate phosphorus accumulation and net retention in
constructed wetlands receiving agricultural runoff**

Critical analysis of factors affecting retention estimates

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Front cover: A constructed wetland in Skilleby, Sweden.

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Till pappa,
vad stolt du skulle ha varit!

Abstract

Eutrophication is one of the more serious current environmental problems, causing algal blooms and anoxic bottoms. In fresh and brackish water, phosphorus (P) is often the most limiting nutrient, and various mitigating strategies are used to reduce the load of P to sensitive recipients. In the agricultural sector, this includes both on-field measures (*e.g.* managing P inputs) and measures at the field edge (*e.g.* buffer zones and constructed wetlands). Previous evaluations of constructed wetlands (CWs) in Sweden have indicated a variable and relatively low P retention. However, the uncertainties in the estimates are large, and related to an incomplete knowledge about both retention processes and factors determining the P load from agricultural land. Hence, the overall aim of this thesis was to investigate possible reasons for the variation in wetland P retention estimates, and to assess the P retention in wetlands located in agricultural areas where losses are expected to be high.

When comparing seven CWs located downstream small catchments with predominantly arable land, the particle and P net accumulation varied considerably (13-108 t particles ha⁻¹ yr⁻¹ and 11-175 kg P ha⁻¹ yr⁻¹, respectively). Catchment factors that were correlated with accumulation of particles and P in the CWs were the slope of the arable land, the P content of the top soil, the animal density (expressed as livestock units per arable land) and the percentage clay in the topsoils.

In four of the wetlands, resuspension was studied using sediment traps and plates. The results showed that up to 87% of the settled material was resuspended, and indicated that erosion of the wetland sides and bottom probably contributed a substantial part of the trapped particles.

In order to critically evaluate existing retention data from earlier investigations, the temporal dynamics of P concentrations and P retention in seven CWs were evaluated. The relationships between water flow and concentration (from grab sampling) varied, and depended on the season (warm or cold period of the year), water flow (high or low) and the inlet type (drainage pipe or open ditch). In CWs that received water through an open ditch, flow-concentration relationships were negative during low flow periods but positive during high flow periods. These differences in flow-concentration relationship have implications for water sampling, since P loads can be both over- and underestimated with grab or automatic sampling guided by clock-time. Also composite automatic sampling, regulated from the water flow at the outlet, can lead to errors in transport calculations. This may have an effect

on estimates of P retention in both past, present and future investigations of constructed wetlands.

Finally, a synoptic sampling approach with ten sampling points was used in an agriculturally dominated catchment area (160 km²) to identify differences in nutrient transport dynamics and areas with the highest losses. Spatial differences in P concentrations were strongly correlated with some of the catchment factors, for instance with soil type, and particle concentrations were weakly correlated to agricultural practices associated with bare soils during winter. This supports the practice to focus P mitigation measures – such as constructed wetlands – to erosion sensitive areas.

Populärvetenskaplig sammanfattning

Övergödning är ett allvarligt miljöproblem, som bland annat orsakar omfattande blomningar av alger och blågrönbakterier. I söt- och brackvatten är fosfor ofta det mest begränsande näringsämnet för dessa organismer, varför en minskning av fosfortillförseln är nödvändig för att nå förbättringar. I Sverige beräknas jordbruket bidra med 44% av fosforbelastningen till Egentliga Östersjön, och olika åtgärder för att minska fosforförlusterna från jordbruksmark tillämpas runtom i Sverige.

Våtmarker anläggs ofta för att fånga näringsämnena och partiklar från jordbruksmark innan de läcker ut i vattendrag och slutligen i Östersjön. Tidigare utvärderingar av anlagda våtmarker i Sverige har visat på en varierande och relativt låg fastläggning av fosfor. Osäkerheten kring dessa utvärderingar är dock ganska hög, och bottnar i kunskapsluckor både vad gäller processer för fastläggning och transport av fosfor från mindre jordbruksområden. I denna avhandling utreds därför hur anlagda våtmarker fungerar som fällor för jordpartiklar och partikelbunden fosfor i områden med höga fosforförluster.

Sju anlagda våtmarker i jordbruksområden med mycket lerjordar studerades, och mängden fosfor och partiklar som fastlades på botten varierade mycket mellan olika våtmarker (13-108 ton partiklar/ha/yr och 11-175 kg fosfor/ha/yr). De faktorer i området uppströms som var kopplade till fosforfastläggning var lutningen i området, markens lerhalt och innehåll av växttillgänglig fosfor samt områdets djurtäthet.

Resultat från fyra våtmarker visade på en hög resuspension (partiklar från botten virvlas tillbaka upp i vattnet), men en del av de uppvirvlade partiklarna kom troligtvis från erosion från våtmarkernas sidor och inte från det material som fastlades på botten. Man såg även indikationer på resuspension från vattenprover tagna i utloppet av en annan våtmark. Där var partikelbunden fosfor klart dominerande, vilket kan ha varit en konsekvens av resuspension från botten.

Variationerna av fosforkoncentrationer vid in- och utlopp i sju anlagda våtmarker studerades, för att kritiskt kunna granska tidigare retentionsskattningar. Det var stora variationer i sambanden mellan vattenflöde och fosforkoncentrationer mellan de olika våtmarkerna. De faktorer som påverkade sambandet mellan flöde och koncentration var 1) om det var varm eller kall årstid (d v s sommar eller vinter), 2) om det var högt eller lågt vattenflöde, samt 3) om inflödet bestod av ett dräneringsrör eller ett öppet dike/åfåra. I våtmarker med öppet dike som inflöde var flödeskoncentrationssambandet av totalfosfor negativt vid låga flöden och positivt vid

höga flöden. De olika sambanden visar hur viktig sättet att provta vatten är, då inkommande mängd fosfor både kan över- och underskattas om man inte är medveten om sådana variationer. Vid automatisk provtagning styrd av flödesmätningen sker detta ofta i utloppet, men eftersom vattnet har en viss uppehållstid i våtmarken (speciellt sommartid) kan retentionsberäkningen påverkas av att all provtagningen styrs av flödet i utloppet.

För att identifiera vilka områden som bidrar med mest näring och partiklar valdes ett stort område dominerat av jordbruksmark ut – för att undersöka hur man skulle kunna bedöma *var* anlagda våtmarker skulle kunna göra mest nytta. Området delades in i tio mindre områden, och vattenprover samlades in från diken och åfåror. Resultaten visade på stora skillnader i fosfordynamik mellan olika delområden med olika jordtyper, trots att de hade liknande markanvändning. Det fanns ofta en koppling mellan höga fosforkoncentrationer i vattnet och en hög andel lerjordar i området. För partiklar fanns det en tendens till samband mellan höga koncentrationer och hög andel vinterbar mark. En anlagd våtmark skulle antagligen ha högst effekt om den placerades nedströms områden som är känsliga för erosion – områden med hög andel lerjordar eller med hög andel vinterbar mark.

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Abstract

Populärvetenskaplig sammanfattning

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List of papers

This thesis is based on the work contained in the following papers. They are referred to in the text by their Roman numerals.

- I. Johannesson, K.M., K.S. Tonderski, P.M. Ehde and S.E.B. Weisner (2015). **Temporal phosphorus dynamics affecting retention estimates in agricultural constructed wetlands.** *Submitted to Ecological Engineering.*
- II. Johannesson, K.M., J.L. Andersson and K.S. Tonderski (2011). **Efficiency of a constructed wetland for retention of sediment-associated phosphorus.** *Hydrobiologia* 674: 179–190.
- III. Johannesson, K.M., P. Kynkäänniemi, B. Ulén, S.E.B. Weisner and K.S. Tonderski (2015). **Phosphorus and particle retention in constructed wetlands – a catchment comparison.** *Ecological Engineering* (doi:10.1016/j.ecoleng.2014.08.014).
- IV. Kynkäänniemi, P., K.M. Johannesson, B. Ulén and K.S. Tonderski (2015). **Assessment of particle deposition and accumulation in newly constructed wetlands receiving agricultural runoff.** *Submitted to Hydrological Processes.*
- V. Johannesson, K.M., G. Lindström, A. Heeb, A. Milver, R. Rönnberg, K.S. Tonderski and L. Andersson (2015). **Can spatial and temporal nutrient concentration variability be captured by catchment agro-geographical characteristics and water quality modelling?** *Manuscript.*

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My contribution

My contribution to the papers included in this thesis was as follows:

- I. Did all the data and statistical analyses as well as some laboratory work. Participated in data interpretation and did most of the writing, with assistance from the co-authors.
- II. Planned the sampling strategy in cooperation with co-authors, and did all the field sampling and analyses of the sediment samples. Did most of the data analyses, including calculation of retention using data from the water monitoring program as well as the statistical analyses. Did most of the data interpretation and paper writing with assistance from the co-authors.
- III. Planned the experimental work and setup with the co-authors. Performed all the field work and most of the laboratory analyses together with the second author. Did all statistical analyses, part of the results interpretation and most of the writing.
- IV. Planned the experimental work and setup with the co-authors. Performed all the field work and most of the lab work jointly with the first author. Participated in data interpretation and in writing of the paper.
- V. Participated in the planning of the monitoring with the other co-authors. Participated in some of the water sampling campaigns, did some of the lab analyses. Did all the statistical analyses of the results from both the monitoring and the modeling, but had no part in setting up the model, or model calibration and validation. Was responsible for the data interpretation discussions and did most of the writing, in collaboration with the co-authors.

1 Introduction

1.1 Phosphorus in agriculture

Eutrophication is an environmental problem in Swedish lakes and the Baltic Sea, potentially causing algal blooms, anoxia and dead bottoms (Carstensen *et al.*, 2014). In freshwater and brackish systems, phosphorus (P) is often the most limiting nutrient (Kalff, 2002). Since the mid-19th century, P load to the Baltic Sea have increased eightfold (Larsson *et al.*, 1985). The cause of eutrophication is an excess of nutrients due for example to legacy P in soil, in combination with an increasing population and more concentrated animal hosting. The population in Sweden has risen from approximately four million in 1860 to almost ten million today. With a growing population, there was a need to increase agricultural productivity, and this is reflected in the increased use of fertilizers. In the 1930s, mineral fertilizer use was less than 5 kg P ha⁻¹, but in the 70s it had increased to 20 kg P ha⁻¹ (Löfgren *et al.*, 1999). However, the use of mineral P fertilizers has decreased since the 1980's (Official Statistics Sweden), and most P fertilizers are now in the form of manure. According to recent estimates, in the south and south east of Sweden P loads from agriculture to the southern Baltic Sea was 156 ton P yr⁻¹; this represents 44% of the total P load from Sweden (Stolte *et al.*, 2009). Hence, there is need to reduce the P load from agriculture. This goal constitutes a part of the Swedish Environmental Objective “No eutrophication”, and is also included in the Baltic Sea Action Plan (EPA, 2014; HELCOM, 2014). Measures to control and decrease P losses from agricultural land can be implemented at different levels. At the first level, a more efficient recovery and reuse of organic P sources – both animal manure and human wastes (Cordell *et al.*, 2009; Schoumans *et al.*, 2015) would mean a decreased import and use of mineral P. At the second level, measures to decrease the field losses of P can be implemented, for example by incorporating manure and mineral P fertilizers in the soil to improve the contact with soil particles and increase the soil adsorption of added P (Djordjic *et al.*, 2002). Also, measures can be taken to level out the water infiltration into the soil and strengthen the soil aggregates in clayey top soils (*i.e.* reducing the P losses from the fields) through structure liming (Ulén & Etana, 2014) or by other soil structural improvements.

However, even though various on-farm and on-field measures can reduce the losses, there will still be some export of P from agricultural fields. The average P loss from agricultural land in Sweden is 0.4 kg ha⁻¹ yr⁻¹ (Ulén *et al.*, 2007). However, the variation is large, and from single fields representing an entire farm with clayey soil and general high soil-P status the mean P loss was 1.5 kg ha⁻¹ yr⁻¹ (Stenberg *et al.*, 2012).

At the third level, the P load to sensitive recipients can be reduced by retaining P that has been lost from the fields, *i.e.* increasing the retention of P in the agricultural landscape. There are several different management practices to achieve this, including buffer strips between fields and ditches (Uusi-Kämpää & Jauhiainen, 2010), two stage ditches (Powell *et al.*, 2007) and constructed wetlands (Braskerud *et al.*, 2005).

1.2 Transport of phosphorus from arable land

To be able to implement the best management option to reduce P transport from arable land, information on the sources of P is needed. In other words, knowledge regarding *from where* P is lost, and *when*. Particles and P can be transported from agricultural fields in several ways, through surface runoff, via (fast) preferential flows in macropores (*e.g.* Ulén *et al.*, 2013), as (slow) leaching through the soils and further via subsurface drains. In Scandinavia, silt and clay soils (where most of the agricultural land is situated) have a relatively high risk of erosion of particles and particulate P (*e.g.* Ulén & Jacobsson, 2005). In soils with small particle sizes, the infiltration capacity of the soil is very low, which means that soil particles cannot infiltrate the underlying soil but is transported as surface runoff. Clay soils are often drained in Sweden, and a large proportion of the transported particles and P is reaching open ditches and other surface water via subsurface runoff in tile drains.

Pionke *et al.* (2000) investigated critical source areas in the Chesapeake Basin in the USA, and identified that only a small portion of the land area (11% of the total catchment area) was responsible for the main export of the fraction of P that was considered bioavailable, hence of particular importance for eutrophication of the Chesapeake Bay. Identifying these critical areas, or ‘hot spots’, is an important part of any program to reduce the transport of P from arable land.

1.2.1 Catchment factors affecting particle and phosphorus transport

As previously mentioned, not all agricultural soils generate P to receiving waters – the potential for P losses depend on several catchment factors. For instance, Ekholm *et al.* (2000) showed that in an agricultural catchment in Finland, the two most important factors determining P losses from the soils were **land use** (in their study defined as the proportion of agricultural fields compared to forests) and the **topography** of the landscape.

Several studies from Sweden and Scandinavia have shown that in runoff from agricultural fields in clay and silt soil dominated areas, P is transported predominantly as particulate P (PP; defined as particles with diameters larger than 0.2 μm (Ulén, 2004; Uusitalo *et al.*, 2000, 2003) or 0.4 μm (Koskiaho *et al.*, 2003)). PP usually consists

of soil particles and organic matter that are eroded from arable land during irrigation, precipitation or snowmelt, either as surface runoff or through the drainage system (Maynard *et al.*, 2009). Clay particles can contain 12 times the amount of PP found associated with sand particles (Pacini & Gächter, 1999). A substantial part of this P is potentially bioavailable; a study of sediments in constructed wetlands confirmed this, as the clay content correlated well with the potentially bioavailable P (Maynard *et al.*, 2009). A Finnish study, on the other hand, showed that the PP had low bioavailability and only about 5 to 10% of the P bound as PP was directly bioavailable (Uusitalo *et al.*, 2000). Hence, knowledge of the **soil type** in a catchment area is important for assessing the magnitude of particle and P transport, as well as the potential bioavailability of the P.

The most critical areas contributing particles and P according to the study by Pionke *et al.* (2000) were located where hydrologically active areas overlap areas with high **soil P content** and areas with high erosion risk. Similarly, in a recent investigation of particle and P losses from five Swedish agricultural fields with artificial drains, the transport from a slope-length estimate was the most important factor affecting water flow and sediment losses (Solís *et al.*, 2014).

1.2.2 Dynamics of P transport

In agricultural landscapes, P transport is very variable and episodic in nature. It often occurs during storm events or snowmelt, when P concentrations in agricultural streams can change dramatically over shorter time than an hour (Bieroza *et al.*, 2014). In clay soil fields the existence of macropores and preferential flow pathways is highly critical for fast P transport, as mentioned above. In streams P dynamic is also quick, demonstrated for instance, by Kronvang *et al.* (2003) in two streams in Denmark. In their study, the highest P concentrations were found during rainfall in autumn, when much of the PP retained in the stream channel during low flow periods before was washed away due to resuspension. Similarly, Pionke *et al.* (2000) showed that 90% of the P export from an agricultural watershed in the USA occurred during storm flow, which represented 10% of the time. Ekstrand *et al.* (2010) observed that in a >800 km² large catchment, 76% of the annual stream TP transport was discharged during 18 days in December and January. Such sudden events will have a significant impact on P transport from arable lands. In contrast, summer runoff is commonly very low and sometimes drops to zero (this could also happen during the winter). During low-flow periods the water in ditches can become stagnant which alters the biogeochemical properties of the sediment, and bioavailable P can possibly be released from the stream sediment, contributing to commonly observed high concentrations in low flow periods in small streams (*e.g.* Sharpley *et al.*, 2007).

Since P losses are very episodic, mitigation measures to reduce P transport should be focusing on capturing the P that is lost during these few events. In order to design and dimension various abatement strategies (such as buffer strips, two-stage ditches or wetlands) there is a need to investigate the dynamics of P concentrations from agricultural land, and assess *e.g.* the magnitude of the concentration changes that can occur during high flow, and whether or not there are flow-concentration relationships and seasonal variations in P concentration dynamics in Swedish catchments.

1.3 Constructed wetlands as traps for P

The construction of wetlands has been advocated as one measure to reduce the export of P from catchments dominated by agriculture. In Sweden, farmers can receive subsidies for constructing wetlands on productive land. One of the Environmental Objectives in Sweden is to construct or restore 12 000 ha wetlands in agricultural areas and up until 2014 almost 7 600 ha have been constructed or restored (Fig. 1). One of the purposes of these wetlands is to reduce the load of nutrients to the Baltic Sea; information about the potential for P retention is therefore important. Many studies have shown that constructed wetlands can function as sinks for P from non-point sources, but the retention efficiency is highly variable, and occasional releases of P have also been observed (Braskerud *et al.*, 2005, Jordan *et al.*, 2003, Kovacic *et al.*, 2006).

Figure 1. Total amount of constructed or restored wetlands in Sweden for the period 2000-2014. The environmental objective is 12 000 ha. Data obtained from miljomal.se (for 2000-2011) and the Swedish Board of Agriculture (for 2012-2014).

The hydraulic load is an important factor for the P retention, because it will affect both the total load of particles and P to a wetland, and the water velocity which in turn affects the settling rate of the particles (Carleton *et al.*, 2001). The hydraulic load and P load are closely linked to the ratio between the wetland area and the catchment area ($A_w:A_c$). There is no clear ‘rule of thumb’ regarding the optimum $A_w:A_c$ ratio. First, the runoff varies considerably between different climatic regions. For instance, in the south of Sweden, the annual runoff on the west coast can be more than twice that of the dryer east coast (600-700 mm vs. 200-300 mm). Second, the sizing of the wetland depends on its purpose – if the objective is to achieve clean water in the outlet (*i.e.* a high relative retention, expressed as percentage of load) a wetland should be large in relation to its catchment area. If the objective is to achieve a high area-specific retention (expressed as kg of P removed per hectare wetland area and year), a wetland can be quite small relative to its catchment areas (0.05-0.38 %, Braskerud *et al.*, 2005; Maynard *et al.*, 2009). In agricultural landscapes, fertile land area is preferable used for agriculture production, and the willingness to construct wetlands is usually low.

1.3.1 Catchment factors affecting P retention

Previous studies on wetlands constructed on arable land have shown a clear relationship between P load and P retention, where high retention was found in wetlands receiving high loads (Braskerud *et al.*, 2005). The load of P, *i.e.* the amount of P that is lost from the catchment and enters the wetland, is determined by various catchment characteristics (factors marked with bold letters in 1.2.1). Since we know some of the catchment factors that usually affect P transport from agricultural land, and we know that P retention in wetlands is positively related to the P load, there is reason to believe that it should be possible to identify ‘hot spot’ areas by using knowledge of the catchment area itself. In other words, using available geographical information (including land use, topography, soil type and P content in the soils) to identify areas where P losses are high, and where constructed wetlands would thus best be located. However, little is known in Sweden about the potential for P retention in wetlands situated in areas where P losses are high. Since P transport differs between different areas, P retention in wetlands should also differ between catchments with various characteristics.

1.3.2 Mechanisms for P retention in constructed wetlands

P enters a wetland in inorganic and organic form, and in particulate and dissolved reactive form (PP and DRP respectively). DRP is considered bioavailable, whereas organic and particulate P forms must generally undergo transformations to inorganic

forms to become bioavailable (Reddy *et al.*, 1999). Wetland P retention can be defined as the result of a number of physical and biogeochemical processes leading to removal of P from the water column and storage in a more or less non-bioavailable form in the sediment. PP is retained by sedimentation as the water velocity drops when water enters a wetland and particles can settle on the bottom. DRP is retained by both chemical and biological processes. It can be sorbed to particles or form chemical precipitates with metal cations. Uptake of DRP by biota is also an important retention process (Fig. 2). However, all the processes described above are reversible. For instance, particles that have settled on the bottom could be resuspended due to high flows or bioturbation by fish, birds and invertebrates (Adámek and Maršálek, 2013). DRP can be released from the chemical bonds should the chemical status (such as redox potential or pH) of the wetland change (Fig. 2, vertical dashed line). Furthermore, most of the DRP assimilated by biota is released back into the water column after the death of the organism, as decomposition proceeds. According to Richardson (1985), 35-75% of plant P is rapidly released again. The balance between the internal processes sedimentation and resuspension, adsorption and desorption, and biological uptake and decomposition will determine whether the wetland is a sink or a source of P. The desired effect of constructed wetlands is long-term retention of P, *i.e.* the risk for leakage back to the water column should be minimized.

Since a significant part of the P transported from agricultural soils is presumably attached to particles, the main retention mechanism for wetlands constructed in these areas is probably sedimentation of particulate P. Sedimentation rates depend on the size and shape of the particles, but also on the water velocity and the wetland depth. According to Stoke's law, it will take the coarsest clay particles (with a diameter of 2 μm) approximately 88 h to sink 1 m in fresh water (15°C, Sheldrick & Wang, 1993). Therefore, in order to have clay particles sink to the bottom, the residence time in a wetland needs to be quite long (indicating that the hydraulic load should be low). In a study of several constructed wetlands in Norway receiving high hydraulic loads, Braskerud (2003) observed that fine clay particles entering the wetlands had sedimentation velocities similar to coarse clay or silt, which was explained by a high degree of soil particle aggregation (Sveistrup *et al.*, 2008). On the other hand, in the drain flow from a clay soil in Sweden, a majority of the particles were colloids with a theoretical settling velocity of 0.08 cm day^{-1} (Ulén, 2004). In a catchment with such colloidal soils, the sedimentation in wetlands might be a lot lower than in wetlands located in catchments with a high proportion of aggregates, as those studied by Braskerud (2003). Because circumstances differ between Sweden and Norway regarding precipitation, topography, erosion and proportion of clay soils (Ulén, 2004; Ulén *et al.*, 2007), the results from Braskerud (2003) might not be applicable to

Swedish conditions, and there is a need to evaluate the function of Swedish constructed wetlands receiving high loads, and to investigate their efficiency as traps for P and particles.

Figure 2. Processes for dissolved reactive phosphorus (DRP) and particulate phosphorus (PP) retention and release in constructed wetlands. This thesis mainly focuses on PP processes and retention. Illustration: Karin Johannesson.

In addition to the difficulties in predicting clay particle sedimentation, resuspension of particles on the wetland bottom could lead to recycling of P, and it could consequently be re-used biologically (Reynolds & Davies, 2001). Hence, the dynamics of sedimentation and resuspension of clay particles in constructed wetlands need to be further investigated.

A challenge when constructing wetlands for P retention is that we have an incomplete understanding of how factors such as variable water flows, wetland design and location in the landscape affect P retention. Some of the difficulties lie in the gaps in knowledge regarding P behavior in catchment and wetlands, *i.e.* in which form P is transported, but also how P retention in wetlands change over seasons.

1.3.3 Methods for estimating P retention in constructed wetlands

Some of the variations in P retention in the studies cited above can probably be explained by differences in sampling technique and estimates of water and P input-output balances. Performing a transport study in the wetland scale is difficult due to the problem of obtaining accurate data on both water flow and the very variable inflow and outflow P concentrations. For example, in many cases, water flow is only

measured at one point (either the inlet or the outlet), which leads to a risk for under- or overestimations of the water entering and leaving the wetland (Reddy *et al.*, 1999). This can be quite important when calculating P transport, especially in low-loaded constructed wetlands. For example, Kovacic *et al.* (2006) showed that in two low-loaded wetlands (with hydraulic loads of 11 and 16 m yr⁻¹, respectively) the outlet only represented 64 and 68% of the total water volume leaving the wetlands. The rest of the water was lost either by evaporation or seepage. Hence, if mass transport calculations in such low-loaded wetlands are based solely on water flow measurements at the outflow, nutrient budgets will be inaccurate. Similar estimation errors can occur in wetlands that receive higher loads. Since wetlands buffer the water flow, the outflow is usually less variable than the inflow, and if mass transport calculations are based solely on outflow measurements, the errors can be quite substantial.

In addition to difficulties in obtaining accurate water balances, the water sampling itself (whether grab or automatic sampling) can result in samples that do not accurately reflect the fast fluctuations of particle and P concentrations. The simplest water sampling strategy is grab sampling, *i.e.* manual sampling at in- and outlet at specific occasions. Automatic time proportional sampling includes a water sampler that collects water at both inlet and outlet at specific time intervals, in combination with continuous flow measurements. In comparison, automatic flow proportional sampling includes a water sampler that collects water in relation to the water flow (controlled by flow measurements only at the in- or outlet, or both in- and outlet), and not at specific times, and probably captures the concentration dynamic rather well. However, the time lag in flow variations between inlet and outlet is not accounted for if the flow is only measured at one point with flow proportional sampling. In addition, if concentrations are correlated with flow variations, this can introduce systematic errors in the estimates of mass in- and outflows. These errors in concentration data can lead to both over- and underestimation of P transport, and it is particularly difficult to capture the event-based movement of particles (Jarvie *et al.* 2002). For example, Kronvang & Bruhn (1996) showed that in two Danish streams, TP transport was nearly always underestimated, especially for PP.

In conclusion, P retention estimations are difficult to perform in wetlands constructed in agricultural catchments, and more knowledge on how monitoring programs could be adapted to the dynamic nature of water flow and concentrations of P and particles is needed.

2 Objectives

The overall aims of this thesis were i) to investigate possible reasons for observed variation in wetland P retention estimates, and ii) to assess the potential P retention in wetlands located in agricultural areas in Sweden where losses are expected to be high.

Specific objectives for each paper were:

- I. To evaluate the temporal dynamics of inflow P concentration and P retention in seven constructed wetlands, in order to critically evaluate existing retention data from earlier investigations. Further, to improve the understanding of factors affecting P retention in wetlands in agricultural areas.
- II. To estimate annual and monthly P retention in a low-loaded constructed wetland, and to investigate the dominating processes retaining inflowing P. In addition, to investigate how well estimates of P retention based on inflow-outflow measurements compared with the amount of P accumulated in the sediment.
- III. To quantify the annual particle and particulate P accumulation in constructed wetlands in agricultural catchments with various soil types. In addition, to investigate if there were relationships between catchment and wetland characteristics and particle and P accumulation, to improve the basis for selecting appropriate sites for constructed wetlands.
- IV. To analyse the relationship between seasonal sediment deposition and estimated load of particles to constructed wetlands, and to evaluate the importance of resuspension and internal erosion for annual sediment accumulation.
- V. To analyze spatial and temporal variability in nutrient and particle losses in a large agricultural catchment, and to investigate if some catchment characteristics were correlated with high nutrient and particle concentrations. In addition, to compare two methods for identifying areas with high nutrient losses (synoptic sampling *vs.* HYPE model).

3 Presentation of the study sites and the main methods

Figure 3. Location and design of the 13 wetlands included in Papers I-IV. White arrows represent inlets and outlets, striped areas represent shallow parts with dense vegetation; thick black lines in Gen and Nyb represent drainage pipes connecting the two ponds. The black line in Ste represents an embankment that diverges the flow, and the dotted area in L.B. is an embankment separating the two ponds. The wetlands are not according to scale.

3.1 Study sites

The constructed wetlands investigated in Papers I-IV were all situated in agricultural catchments in the south of Sweden (Fig. 3). The oldest wetland was L.B. (Paper I), constructed in 1991, and the most recent was Nyb (Paper IV) that was constructed in 2011.

The agricultural catchment monitored in Paper V was situated in the county of Östergötland in Sweden (Fig. 4). The catchment area was 160 km² and the dominating land use was agriculture (88%). Water covered 0.1% of the area, and consisted mainly of small rivers, ditches and irrigation ponds. There were no major industries in the area. The bedrock was mainly granite but sandstone and limestone also covered the area. The soil type was rather diverse, but was dominated by moraine, sand and clay.

Figure 4. The catchment in Paper V was situated in the south east of Sweden. The large catchment was divided into ten smaller sub-catchments (10-99) in order to monitor nutrient loss from different source areas. Water ways (ditches and streams) are marked in white in the figure, and the black points represent stations for water sampling.

3.2 Water flow measurements

Water flow was measured in ten of the thirteen investigated constructed wetlands (Table 1). In most cases, water flow was only measured at the outlet, but for Ber,

Nyb, and Ste water flow was measured both at the inlet and the outlet. In Ski, only the inlet was equipped with flow measuring gauges. Hydraulic loads, expressed as m yr^{-1} , were calculated for each wetland by dividing the cumulative annual inflow (for four of the wetlands) or outflow (for six of the wetlands) (m^3) by the area of the wetland (m^2).

Table 1. The methods for measuring water flow in the ten constructed wetlands with water quality monitoring used for this thesis (listed in alphabetical order).

Wetland	Water flow measurement	Sampling period (years)	Paper
Ber	Inlet: V-notch and water stage recorder Outlet: V-notch and water stage recorder	3.5	IV
Böl	Inlet: - Outlet: Area velocity flow meter in outflow pipe	2	I
Ede	Inlet: - Outlet: V-notch and pressure flow meter	1	I
Gen	Inlet: - Outlet: V-notch and mechanical water stage recorder	5	I
L.B.	Inlet: - Outlet: V-notch and pressure flow meter	3	I
Nyb	Inlet: V-notch and water stage recorder Outlet: V-notch and water stage recorder	2	IV
Råb	Inlet: - Outlet: Mechanical water stage recorder	9	I
Ski	Inlet: V-notch and water stage recorder Outlet: -	3	IV
Slo	Inlet: - Outlet: Mechanical water stage recorder	6	I
Ste	Inlet: Area velocity flow meter in inflow pipe Outlet: V-notch and pressure flow meter	4	I and II

In the large agricultural catchment in Paper V there was a permanent station for discharge monitoring at one of the sampling points (80 in Fig. 4). In addition, a pressure gauge was installed at a second location (in the center of the town Skänninge, not visible in Fig. 4). Estimates of river flow were made after establishing a rating curve. Daily river discharge records from these two stations were used for HYPE model calibration and validation, which was done by Milver (2014). The modeled flow was then used to estimate the flow in each upstream sub-catchment sampling station using area-weighting (Fig. 4).

3.3 Water sampling and analyses

In Paper I, the data used was results from analyses of grab samples that were taken regularly (once a week) in L.B., Bøl and Ede. In Ste, Gen, Råb and Slo, grab samples were taken less regularly. Daily grab samples had been collected for shorter periods in three of the wetlands (L.B., Gen and Ste), in order to investigate the movement of P with a higher resolution. Those periods were often with high flow; grab samples were collected once or twice a day and the water samples were then analyzed for P concentrations (Table 2).

In Paper II, results from analyses of grab samples were combined with those from automatic composite flow-proportional sampling at the inlet and outlet of the wetland. The latter was done using combined flow meters and water samplers where the composite sample consisted of several sub-samples along the water-flow peak representing a certain volume of water (for details, see Paper II). This resulted in an extensive set of in- and outflow nutrient concentrations and water flow data, which made it possible to accurately calculate the long-term P retention based on load and outflow measurements.

In Paper IV, data from flow-proportional composite samples taken at the inlet, and collected every fortnight for three of the wetlands (Ber, Nyb and Ski), were used to estimate the load of particles to the wetlands. The water samples had been analyzed for suspended solid concentrations (Table 2). The automatic samplers were controlled by the water flow in the inlet to all those wetlands.

Table 2. Chemical analyses of water samples from the various papers (I, II, IV and V). TP=total phosphorus, DRP=dissolved reactive phosphorus (i.e. bioavailable phosphorus), SUSP=suspended material (i.e. particles).

Paper	Analyses	Comment
I	TP, DRP	DRP analyzed on filtered samples (0.45 µm)
II	TP, DRP	DRP analyzed on filtered samples (0.45 µm), and only on samples collected during three high-flow periods
IV	SUSP	Filter size 0.2 µm
V	TP, DRP, SUSP	DRP analyzed on filtered samples (0.45 µm)

The agricultural catchment in Paper V was monitored with synoptic grab samples at ten sampling points (Fig. 4). Samples were collected during periods that represented various flow regimes, resulting in a total of 18 occasions. During snowmelt, an automatic water sampler (ISCO) sampling every third hour were placed at a strategic

location in the catchment to capture the movements of nutrients and particles during high flow.

3.4 Sediment sampling

3.4.1 Sediment core samples

In Paper II, sediment samples were collected from five different sections of the wetland (for details, see Paper II) using a core sampler (diameter 7 cm, Fig. 5). The total P content as well as the different fractions of P in the sediment were analyzed. For each core sample, the thickness of the accumulated sediment was also measured. The total volume of the accumulated sediment was estimated by multiplying the area of each section with the mean sediment thickness in that section.

3.4.2 Annual accumulation

In Paper III, sediment plates were used to investigate the annual accumulation of sediment in the seven wetlands. The plates served as reference bottoms and were exposed to natural processes such as sedimentation, bioturbation and resuspension caused by high flow periods. The sediment plates were sampled once per year, and the results interpreted to represent the annual *net* sedimentation. The plates were made of plastic-coated plywood and were 40×40 cm or 25×25 cm (Fig. 5). After approximately one year, the plates were lifted and the average thickness of the accumulated sediment was measured. After that, sediment was collected from the most undisturbed part (usually close to the center of the plate), and the samples were stored in plastic containers. Visible plant parts were removed from the sediment samples, which were dried and analyzed for total phosphorus content (see Paper III for details).

Figure 5. Three different methods used for collecting sediment from the eight wetlands in Papers II, III and IV. Sediment plates were used in Papers III and IV, traps were used in Paper IV, and core samplers were used in Paper II. Illustration: Karin Johannesson.

3.4.3 Deposition of particles

In order to estimate sediment deposition, sediment traps (not exposed to resuspension) were placed in four wetlands to compare sediment deposition and accumulation. Plastic cylinders (11 cm high and 7.5 cm in diameter) were placed adjacent to the sediment plates at the bottom of the wetlands (Fig. 5). The traps were dug down into the sediment, with the edge approximately 2 cm above the sediment surface. The intention was that the walls would prevent the sediment captured in the traps from being resuspended and transported away. The traps would thereby act as collectors for suspended solids by lowering water turbulence inside the cylinders. The traps were sampled three times during one year in three different seasons: autumn, winter (including snowmelt) and spring-summer, to enable comparison of the different seasons.

Table 3. Catchment characteristics collected for the areas in Papers I, III and V, and the source of the data.

	Land use	Soil type	Soil P content	Catchment slope	Livestock units	Rural sewage discharge
Paper I	Data from County Administration Boards and old reports	Expressed as % clay, data from the Swedish Board of Agriculture and soil surveys	Data from the Swedish Board of Agriculture and farmers' soil surveys			
Paper III	Data from the Swedish Board of Agriculture	Expressed as % clay, data from the Swedish Board of Agriculture and farmers' soil surveys	Data from the Swedish Board of Agriculture and farmers' soil surveys	Data from the Swedish Land Survey	Data from field surveys, contact with landowners and information from municipalities	Data from field surveys and the Swedish Land Survey
Paper V	Data from the Swedish Board of Agriculture and contact with landowners and advisors	Data from the Swedish Board of Agriculture		Data from the Swedish Land Survey	Data from the Swedish Board of Agriculture	Data from municipalities

3.5 Catchment characteristics

In Papers I, III and V geographical information was collected from a combination of databases, municipalities, field surveys and contact with landowners (Table 3). The information included land use data (the proportion of arable land, and in Paper V, the proportion of different crops), soil type, topography (in Papers III and V) and P content in the soils.

3.6 Modeled loads of water and phosphorus

In Paper III, loads of both water and P were modeled, since in three of the wetlands (Eks, Lin and Wig), there were no measurement data on water flow or P concentrations. Hydraulic loads were estimated using runoff data from large catchment areas provided by the Swedish Meteorological and Hydrological Institute (SMHI). Those data are modeled by the catchment model S-HYPE calibrated for the entire Sweden (Strömqvist *et al.*, 2012). Runoff data for the larger catchment area (light grey area in Fig. 6) for the sampling period (Aug 2010-Aug 2012) was downloaded from SMHI's Water Web (SMHI, 2014) and recalculated to represent the runoff volume of the wetland catchments (dark grey area in Fig. 6) based on area-weighting. The modeled hydraulic load (m day^{-1}) was estimated by dividing the inflow from the small catchment by the wetland surface area.

The load of P to the wetlands in Paper III was estimated using the ICE-CREAM model (Larsson *et al.*, 2007) which generated average P concentrations for the different catchment areas by tailoring the model with in-data for the actual catchment areas themselves (Table 2 in Paper III).

A different version of the HYPE model was used for modeling nutrient concentrations from the sub-catchments investigated in Paper V. The model was setup and calibrated using detailed local agro-geographical data on the catchment, and is described in detail in Milver (2014).

Figure 6. Hydraulic loads to the wetlands were estimated using runoff data from larger catchment areas from the Swedish Meteorological and Hydrological Institute's Water Web.

4 Main results and discussion

4.1 Water flow and P concentration dynamics affecting P retention estimates

4.1.1 Previous estimates of P retention

Some of the variations in P retention found in the literature can probably be explained by differences in sampling technique and estimates of water and P budgets. There has been a wide variety of sampling strategies in previous studies of constructed wetlands (Table 4), ranging from grab samples to flow-proportional water sampling, and water flow measurements at both the in- and outlet, and only at the outlet.

Table 4. Methods for water flow measurements and water sampling in previous investigations of nutrient retention in constructed wetlands, as well as from earlier studies of the seven wetlands from Paper I.

Country		Flow measurement	Water sampling method	Reference
Norway	Six wetlands	Outlet	Flow-proportional	Braskerud, 2002
Finland	Wetland 1	Inlet and outlet	Time-proportional	Koskiaho <i>et al.</i> , 2003
	Wetland 2	Outlet	Time-proportional	Koskiaho <i>et al.</i> , 2003
	Wetland 3	Outlet	Grab samples	Koskiaho <i>et al.</i> , 2003
USA	One wetland	Inlet and outlet	Flow-proportional	Jordan <i>et al.</i> , 2003
Switzerland	One wetland	Inlet and outlet	Time-proportional at inlet, grab samples at outlet	Reinhardt <i>et al.</i> , 2005
USA	Two wetlands	Inlet and outlet	Flow-proportional	Kovacic <i>et al.</i> , 2006
USA	Two wetlands	Inlet and outlet	Grab samples	Maynard <i>et al.</i> , 2009
New Zealand	Three wetlands	Inlet and outlet	Flow-proportional	Tanner & Sukias, 2011
Sweden	One wetland	Inlet and outlet	Flow-proportional	Kynkäänniemi <i>et al.</i> , 2013
Earlier investigations of the wetlands from Paper I				
Sweden	Three wetlands	Outlet	Time-proportional	Wedding <i>et al.</i> , 2003
Sweden	One wetland	Inlet and outlet	Flow-proportional	Eriksson <i>et al.</i> , 2009
Sweden	Three wetlands	Outlet	Flow-proportional	Johannesson, 2011

The seven wetlands in Paper I have been investigated in previous evaluations of P retention in Swedish constructed wetlands. Three of the wetlands in the south of Sweden (Slo, Råb and Gen) were monitored using flow measurements in the outlets (Table 4) and time-proportional sampling at the inlet and outlet (Wedding, 2003). The three wetlands in the west of Sweden (L.B., Böl and Ede) were monitored using flow measurements in the outlets, and flow-proportional sampling at the inlet and outlet (controlled by the flow in the outlet). In the seventh wetland (Ste, from both Papers I and II), water flow was measured at both the in- and outlet, and flow-proportional sampling was carried out. Results from the previous investigations (Johannesson, 2011; Eriksson *et al.*, 2009; Wedding, 2003) have shown variable P retentions for all seven wetlands (from 3 kg ha⁻¹yr⁻¹ in Ste to 50 kg ha⁻¹ yr⁻¹ in Slo).

4.1.2 Flow-concentration relationships

A relationship was found between water flow and inflow TP concentration in all seven wetlands in Paper I, but the strength and type of the relationship varied between wetlands. For three wetlands with a drainage pipe inflow the relationship between water flow and inflow TP concentration was positive, both during low and high flow as well as during cold and warm periods (Table 4 in Paper I). In the other four wetlands with an open ditch or a stream inlet, the relationship between inflow TP concentration and water flow varied with the flow conditions. During low flow periods, the relationships were negative in all four wetlands, while during high flows they were positive (Fig. 7). The positive relationship between water flow and inflow TP concentrations during high flow in Paper I has implications for sampling strategy. For example, if time-proportional sampling is carried out, the estimates of in- and outflow amounts of P could be quite incorrect because of the co-variance of water flow and concentration. Grab samples will also introduce considerable errors during periods of rapid flow changes, as the concentrations will also change rapidly, but not at the same time in the inlet and outlet, discussed further below, both because of the possible flow dampening function of the wetland and because of the complex hydraulic flow patterns.

Previous investigations of P retention in the three wetlands in the south of Sweden in Paper I (Slo, Råb and Gen) were based on time-proportional sampling (Wedding, 2003). Since flow-concentration relationships were evident in all wetlands (Table 4 in Paper I), the earlier P retention estimates should be interpreted with caution. However, in those three wetlands the correlations in the data sets were strongest during low flow periods when the overall transport to and from the wetland is relatively small, so the potential errors may not have affected the overall estimates of annual P retention. However, for the wetlands with larger data sets in Paper I,

correlations between water flow and P concentration were observed during high flow periods as well.

Figure 7. The relationship between log water flow and the log inflow TP concentrations in two of the wetlands investigated in Paper I. One received inflow water through a drainage pipe, and one received water through an open ditch. Data from Paper I.

This underlines the importance of using flow-proportional sampling to capture the movements of P in both in- and outflow and arrive at more accurate retention estimates. However, if the automatic water sampler at the inlet is controlled only by the measured flow at the outlet, there is a risk for missing peaks of PP in particular at the inlet. This was obvious in the Ste wetland, where water flow was measured at both in- and outlet. There were periods of high PP concentrations at the inflow that coincided with high inflow (Fig. 8).

Figure 8. Particulate phosphorus (PP) concentrations at in- and outlet in one of the investigated wetlands in Paper I and II. The water flow at both inlet and outlet is also shown in the figure.

Inflow PP concentrations increased from 123 to 413 $\mu\text{g l}^{-1}$ from one day to the next, and the inflow increased twelve times in the same time period. However, there was no concurrent flow peak at the outlet (Fig. 8), which indicated that *if* the automatic sampling had been controlled only by the outflow, the peak in the inflow PP concentration would have been underestimated. Rapid changes in nutrient concentrations were also observed in a larger catchment scale during a snowmelt water flow peak (Paper V). TP concentration increased from 28 $\mu\text{g l}^{-1}$ to 220 $\mu\text{g l}^{-1}$ in 24 hours (Fig. 7 in Paper V), and the PP fraction represented most of that increase. Similarly, the particle concentration increased almost 14 times. These rapid changes are probably easily missed if water sampling is not flow-proportional and controlled by flow measurement at both inlet and outlet.

4.1.3 Phosphorus retention variations and calculation examples

In several of the wetlands investigated in Paper I large variations were observed in P retention between sampling days, when calculated based on grab samples only. For example, P retention varied from $-737 \text{ mg m}^{-2} \text{ day}^{-1}$ to $1180 \text{ mg m}^{-2} \text{ day}^{-1}$ over five days in one of the wetlands. Moreover, on single occasions very high P retention was observed in some of the wetlands. In Böl, two single occasions (with P retentions of 3480 and 1810 $\text{mg m}^{-2} \text{ day}^{-1}$, respectively) probably represented the main part of the P retention over a year (Fig. 5 in Paper I). Similar single occasions with high P retention occurred in several of the other wetlands as well.

There is a great risk of not detecting the concentration peak, either in the inlet or the outlet (or both) with grab sampling, because of the drastic changes occurring in P concentrations especially at high-flow events (as seen from Paper V). This in turn makes it difficult to correctly measure the annual P retention in wetlands through water sampling in in- and outlet. Missing flow or concentration peaks may occur in grab sampling and time-proportional automatic sampling, and to a much lesser degree in composite flow-proportional sampling if it is not perfectly controlled by flow measurements in both in- and outflow. Previous investigations of annual P retention (Johannesson, 2011; Eriksson *et al.*, 2009) in four of the wetlands in Paper I (L.B., Böl, Ede and Ste) were based on flow-proportional sampling, but only in Ste was inflow sampling controlled by the inlet flow. To illustrate the differences, a calculation example for Ste was made including 1) only outflow for calculating mass balances and 2) using both inflow and outflow for calculating mass balances. The results showed that when both in- and outflow were used in the calculations, the resulting P retention was more than doubled (82 mg m^{-2} in 14 days) compared to when only the outflow values were used (38 mg m^{-2} in 14 days). This simple calculation example highlights a problem in estimations of the P load to the wetlands

when water flow is measured only at the outlet. There is therefore reason to believe that previous estimates of the P loads to the three wetlands with flow-proportional sampling controlled only by the outflow may be underestimated – and hence, also the P retention.

A comparison of P retention estimates was made for one of the seven wetlands in Paper I (L.B.). The example included five different ways of accounting for the residence time, and showed that the lowest monthly P retention was estimated when the residence time was disregarded (Fig. 6 in Paper I). The highest P retention estimate was obtained when using the theoretical residence time (T) for each day, but the P retention was also relatively high when $\frac{1}{4}T$ and $\frac{1}{2}T$ was used. Earlier tracer studies in small constructed wetlands have shown that concentration peaks often reach the outflow at approximately $\frac{1}{4}T$ and $\frac{1}{2}T$ (Bodin *et al.*, 2012), so the estimated P retentions in L.B. wetland at these residence times probably reflect the ‘true’ P retention best. They were 1.3 and 1.6 larger than when the residence time was disregarded. The results further illustrated the difficulty in accurately estimating P retention in constructed wetlands, and points to the importance of considering potential errors when estimating P retention.

In conclusion, these results indicate that some of the previous investigations of constructed wetlands should be interpreted cautiously, especially if water sampling was performed either by grab sampling or time-proportional sampling, or if the flow-proportional sampling was only controlled only by outflow measurements. Hence, future evaluations of constructed wetlands should include composite flow-proportional sampling, where inflow water sampling is controlled by the water flow in the inlet, and the outlet sampling is controlled by the outflow.

4.2 Constructed wetlands for particulate P and particle accumulation

4.2.1 Annual PP and particle accumulation

In the low-loaded constructed wetland investigated in Paper II, P retention was relatively low ($3 \text{ kg ha}^{-1} \text{ yr}^{-1}$, Fig. 9), and there were months when there was a net release of P from the wetland (Fig. 4 in Paper II), usually during winter – either during high flow periods or when ice covered the wetland. In the seven constructed wetlands investigated in Paper III, P accumulation varied from 11 to $175 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Fig. 9, grey bars). Particle accumulation varied from $13 \text{ t ha}^{-1} \text{ yr}^{-1}$ in Wig to $108 \text{ t ha}^{-1} \text{ yr}^{-1}$ in Gen. Hence, all seven wetlands functioned as traps for both P and particles, and the annual accumulations were in concordance with other wetland studies using water quality and quality data. For example, Maynard *et al.* (2009) observed P retentions of 82 to $173 \text{ kg ha}^{-1} \text{ yr}^{-1}$ in two constructed wetlands in the USA, and in a

recent study, P retention in a sedimentation basin investigated by Beutel *et al.* (2014) was 200 kg ha⁻¹ yr⁻¹. In an investigation of constructed wetlands in the UK, Ockenden *et al.* (2014) observed P retentions of 36-85 kg ha⁻¹ wetland area yr⁻¹ in three small (20-110 m²) wetlands situated in catchments with clayey soils, and 196-520 kg ha⁻¹ yr⁻¹ in three small wetlands (50-200 m²) where silty loam soils were dominating in the catchments.

Figure 9. Annual phosphorus accumulation in the low-loaded wetland from Paper II (estimations from inflow-outflow data), and the seven wetlands from Paper III (estimations from sedimentation plates). For wetlands with multiple sampling years, the standard deviation is shown as error bars.

In the four constructed wetlands investigated in Paper IV, particle accumulation varied from 1 to 23 kg m⁻² yr⁻¹ (Table 1 in Paper IV). The lowest particle accumulation was in Wig, where the P accumulation was also the lowest. In a study of ten small constructed wetlands in the UK, Ockenden *et al.* (2012) investigated sediment accumulation. The annual sedimentation rate was 2 to 5 kg m⁻² yr⁻¹ in catchments dominated by clay soils, 5 to 110 kg m⁻² yr⁻¹ in catchments with silty soils, and up to 600 kg m⁻² yr⁻¹ in catchments with sandy soils. The catchments in Paper IV have relatively high clay content, and should therefore be compared to the sedimentation rates in clay soil areas in the study cited above. In a study of gross and net

sedimentation in two constructed wetlands in the USA, Mitch *et al.* (2014) observed gross sedimentation rates of 30 to 90 kg m⁻² yr⁻¹, while the net sedimentation was lower, 4 to 6 kg m⁻² yr⁻¹. In Paper IV, gross sedimentation was generally lower than the results from Mitsch *et al.* (2014) for three of the wetlands; Ber (30 kg m⁻² yr⁻¹), Ski (10 kg m⁻² yr⁻¹) and Wig (4 kg m⁻² yr⁻¹), while in the fourth wetland, Nyb, it was several times higher than the results from the USA (190 kg m⁻² yr⁻¹).

4.2.2 Internal processes

Three periods with intensive grab sampling over 14 to 20 consecutive days in the wetland investigated in Paper II revealed that the outflow water was dominated by PP – on average, 78% of the outflow P was particulate P (Fig. 10). PP dominated the outflow regardless of water flow (Fig. 2 in Paper II), and during high flow resuspension could be one of the processes providing PP to the outflow water. Results from Paper IV showed that resuspension of the deposited material was high, varying from 77% in Wig to 87% in Nyb. Hence, only 13 to 23% of the deposited sediment “stayed” as accumulated sediment. In the study by Mitsch *et al.* (2014), the corresponding figure for sediment accumulation was 6 to 20% of the deposited particles. In contrast, Braskerud (2001) measured annual resuspension in parts of wetlands that were rather densely vegetated, and observed a steady decline in resuspension over time – from approximately 40% resuspension to almost zero only five years after construction. Three of the wetlands investigated in Paper IV were relatively newly constructed, and the results from Braskerud indicate that as the wetlands age and more vegetation establishes, resuspension may decrease.

Figure 10. The proportion of particulate phosphorus (PP) for in- and outflow water during one period with intensive grab sampling in the wetland investigated in Paper II.

However, inflow-outflow calculations from the wetland in Paper II concluded that almost 80% of the P load to the wetland settled in the proximity of the inlet (Table 3 in Paper II). Hence, most of the PP leaving the wetland must have originated from the wetland itself. Since PP was the dominating form in the outlet also during low flow, other internal processes than resuspension probably caused the export when water flow was low. For example, bioturbation by invertebrates and birds (Adámek & Maršálek, 2013) and high production of green algae (*i.e.* particulate organic P) could have contributed to the export of PP during the summer months. Similar patterns with a high proportion of PP in the outflow during the warm period was also seen in some of the wetlands investigated in Paper I (see L.B., Fig. 2 in Paper I).

The annual P retention in the low-loaded wetland in Paper II was relatively low (Fig. 9), despite an apparently efficient removal of incoming P, as discussed above. At times there was even a net release of P from the wetland. One source of the P leaving the wetland could be the dense stands of vegetation (dominated by *Typha latifolia* L.) that covered approximately half of the wetland area. Such emergent macrophytes can obtain their nutrient requirement from the sediments, and act as ‘P pumps’ transferring P from the sediment to the above-ground tissues (Granéli & Solander, 1988). A calculation example in Paper II showed that, theoretically, an amount equivalent to the entire load to the wetland could have been incorporated into the plant community. Since a large proportion of plant P is released during decomposition, this internal cycling of P could have affected the overall P retention in the wetland in Paper II.

4.2.3 Factors affecting P retention in constructed wetlands

The wetlands investigated in this thesis were all situated in different catchment areas, and received different loads of water either through a drainage pipe or an open ditch.

Hydraulic load

The hydraulic load to the wetlands was determined by the size of the wetland in relation to the catchment area. In Paper III, the wetlands receiving the highest modeled hydraulic load (according to section 3.6) was Böl and Wig (650 and 400 m yr⁻¹, respectively). Generally, there was a positive relationship between P accumulation and hydraulic load (Fig. 11), up to a certain point, in which high hydraulic load instead generated lower P accumulation. This was also observed in Paper IV, where both deposition and accumulation of particles were positively correlated to the hydraulic load in all wetlands except the one with the highest hydraulic load (Table 2 in Paper IV). We hypothesize that there is a ‘breakpoint’ for

the hydraulic load, which would cause low particle and P accumulation due to too high water flow.

Figure 11. The relationship between average phosphorus accumulation and the hydraulic load in one wetland from Paper II (estimations from inflow-outflow data) and seven wetlands from Paper III (P accumulation estimations from sedimentation plates). The dashed line represents a linear relationship for the six wetlands with relatively low hydraulic load (<200 m yr⁻¹).

Location

The inlet structure is determined by the location in the catchment, because it is dependent on whether or not the catchment area is sub-drained or if open ditches are present. As discussed above, the inlet type proved to affect the flow-concentration relationships in the seven wetlands (Paper I). In three wetlands that received water from a drainage pipe, the water flow-concentration relationship was positive irrespective of flow conditions. In subsurface tile drains, nutrients can rapidly be transported from agricultural land. For example, Gentry *et al.* (2007) observed an increase of especially PP in tiles during high flow events. They suggested that the PP in the tiles was associated with transport of fine clay particles entering the drainage systems through macropores. One explanation for the positive relationship between water flow and TP concentration observed in wetland inlets could therefore be that PP has accumulated in the drainage pipes, and was then mobilized during increased water flow.

On the other hand, in the four wetlands in Paper I that received water via an open ditch or a stream, the flow-concentration relationship was only positive during high flow periods. During low flow periods, the relationship was the opposite, with high TP concentrations occurring at low water flows. The high concentrations of DRP during low water flow periods may be related to anoxic sediments upstream. In small streams in southern Sweden, periods with no flow commonly occur, both during summer and winter. In such stagnant water, anoxia can develop due to decomposition of organic material, and iron-bound P can be released from the stream sediments and transported to the wetlands. A second, and more probable explanation for high P concentrations during low flow can be that inflow water was affected by rural wastewater, which increases in relative importance when water flow is low (Withers *et al.*, 2011; Ulén *et al.*, 2015). Wastewater from rural households often have a high proportion of DRP, and in one of the wetlands there were occasions in the summer months (Jun-Aug) when 100% of the inflow P was DRP, indicating an effect of rural wastewater.

In the studied wetlands, particle deposition was affected by the flow variability in one of the wetlands that was constructed in a ditch (Paper IV). No such correlation was found for two of the other wetlands, with drainage pipe inlets. A more variable flow was positively related to particle deposition in traps, indicating that more particles were eroded and transported during periods of rapid flow variations. Large flow variations mean a higher frequency of events with high water velocities which could cause more erosion, in a similar way as in small streams (Veihe *et al.* 2011). For example, Kronvang *et al.* (1997) and Laubel *et al.* (2003) showed that in a Danish agricultural stream, 40 to 80 % of the particle transport was generated from erosion of the stream bank and bottom. Kronvang *et al.* (1997) estimated that approximately 20 kg of particles eroded from each meter length of stream per year.

The importance of the inlet type for the wetland P retention was also observed in an ongoing investigation of constructed wetlands as traps for P in Sweden (Weisner *et al.*, manuscript). In that study, a linear regression model for estimating P retention was developed, using methodology similar to that in Paper III (see below). Data from the same seven wetlands as in Paper III were used, and one additional wetland was included for the analyses. The resulting P retention model included, among other factors, the inlet type as a dummy variable (ditch=1, drainage pipe=0) in the multiple regression equation, and P and particle retention was generally higher in wetlands with an open ditch as inlet. However, from a construction point of view, excavating soil in a ditch with running water proved difficult, as discussed in Paper IV. For that

reason, opening and redirecting the flow from a drainage pipe into a wetland could be a simpler construction solution.

Catchment characteristics

Some of the investigated catchment factors were related to accumulation of both particles and P in wetlands (Paper III). The factors with the highest correlation coefficients with P and particle retention were average slope, P content in catchment soils, livestock density, and clay content in the catchment soils (Table 5 in Paper III).

Catchment slope is an important factor influencing the risk for soil erosion. For example, Ekholm *et al.* (2000) found that concentrations of particles and total P in runoff increased when the slope of the fields increased, and Ulén *et al.* (2010) stated that the risks for soil particle and particulate P losses depend on both the slope of the catchments and the soil texture. The positive relationship between catchment slope and particle and P accumulation indicates faster and possibly more channelized water flow areas with high slopes, and more of the transported particles (and the P attached to them) are captured in the wetlands.

Figure 12. Sediment phosphorus (TP) concentration for the seven wetlands investigated in Paper III in relation to the P content of the soils in the catchments. Modified from Paper III.

Generally in Paper III, catchments with high P content in the top soils (120-150 mg P kg⁻¹ soil) had the highest P content in the settled sediment (Fig. 12). The P content in the wetland sediments was variable, but in some wetlands very high. For instance, in the Gen wetland, the P content in some of the samples was over 5000 mg kg⁻¹

which is six times higher than the highest classification of P content in Swedish soils (800 mg P kg⁻¹ soil, Bergström *et al.*, 2008). In the other wetlands, P content varied from approximately 750 to 2200 mg kg⁻¹, which is also high in relation to the average P content in Swedish agricultural soils.

Livestock density was positively related to retention, which was probably an effect of higher P content in the soils due to manure addition with high livestock density. Paper III demonstrates a positive correlation with the livestock density of arable land and the soil P content in the catchments ($r=0.72$). Livestock density varied from zero to 0.97 LU ha⁻¹ in the catchments of the seven wetlands in Paper III. In the most animal-dense counties in Sweden the average density is 0.9 LU ha⁻¹ while in Sweden as a whole the average is 0.43 LU ha⁻¹ (official Swedish statistics). The present catchments may therefore reflect the typical variance.

The relationship between the soil types in the catchments and the particle retention in wetlands is complex. On the one hand, silt and clay soils in Scandinavia generally have a high risk for erosion and P losses (Ulén & Jakobsson, 2005) which would increase the load to a wetland – and consequently also the area specific accumulation of particles and P. On the other hand, dissolved clay particles settle slowly and, hence, a short retention time in wetlands with high hydraulic loads might counteract particle accumulation. In the wetlands with high retention studied by Braskerud (2002), it was shown that the clay particles in the inflow water were aggregated and rather settled as silt particles (Sveistrup *et al.*, 2008). In the wetlands studied in this thesis, a positive relationship between catchment topsoil clay content and clay content in the accumulated sediment was observed, and the clay content was higher in the sediments than in the soils (Fig. 4 in Paper III). This suggests that a larger proportion of fine particles was lost from the catchments and transported to the wetlands. However, there was a negative correlation between particle and P accumulation in the wetlands and the clay content of the topsoil in the catchments. This suggests a slower settling of particles lost from the two catchments with the highest clay content, *i.e.* Ski and Wig. This agrees with the findings of Ulén (2004) that tile drain runoff from agricultural land with clay contain high proportions of colloids and very fine particles (with settling rates of less than 1 cm day⁻¹) that would not settle in wetlands with hydraulic loads similar to some of those in Paper III.

The subsequent multiple linear regression analysis (log P and particle retention versus the catchment factors, and including hydraulic load class – high or low) showed that for P, the best model included P content in the soils and the hydraulic load class. This model explained approximately 80% of the variation (Table 6 in Paper III).

Similarly, for particle retention, the best model included P content in the soils and the hydraulic load class, with a R^2 value of 0.72.

The identified catchment factors can be useful for selecting locations suitable for P trapping wetlands in Sweden. The best location of future wetlands is therefore not necessarily on the edge of the fields – where farmers and landowner would prefer them, and it may become necessary with increased compensations to the farmers in the future in order to reach the Environmental Objective. However, it is important to note that the results are based on a low number of wetlands ($n=7$), and that further analyses including more constructed wetlands would be desirable to refine the model.

4.3 Identifying ‘hot spot’ catchments

In another attempt to try to find locations that would be suitable for constructing wetlands in Sweden, the variation in nutrient concentrations from the large agricultural catchment depicted in Fig. 4 was analyzed in order to identify areas with high losses of nutrients and particles – ‘hot spot’ areas.

4.3.1 Spatial variations in P and particle transport

The average flow weighted concentrations differed between the different sub-catchments investigated in Paper V (Fig. 13). From monitoring data, TP and DRP concentrations were highest from area 60 and 30. An explanation for the high concentrations of P from at least area 60 could be a high proportion of clay soils in those areas. Several studies from Sweden have shown that clay soils generally have high losses of P. For example, long-term P losses from agricultural catchments with clay soils varied from 0.3-0.8 kg ha⁻¹ yr⁻¹ but was 0.09-0.3 kg ha⁻¹ yr⁻¹ from catchments with more permeable soils (Ulén *et al.*, 2007). The main transport of P from clay soils is usually in particulate form (*e.g.* Uusitalo *et al.*, 2000, 2003), and the PP concentrations from area 30 and 60 were also the highest. However, the particle concentrations (SUSP) were highest from area 70. This sampling point is a connectivity point, *i.e.* a point where water from six other sub-catchments join into one ditch (Fig. 4). One possible explanation for the high particle concentration in this sampling point could be in-stream erosion (induced by turbulence) when the many different inflows join. The more turbulent water could possible keep particles in suspension (even during periods of low water flow) causing high particle concentrations at all sampling occasions regardless of the water flow. The second highest particle concentration was at sampling point 90, which is also a connectivity point. The results from the concentration modeling were of a similar magnitude as the monitoring data for TP and DRP although the HYPE model did not capture the variation between the sampling points (Fig. 13). Furthermore, for PP, the modeled

concentrations were significantly lower than the monitored concentrations. Moreover, the sub-catchments with the highest concentrations of nutrients differed from the monitoring data for all nutrients (Fig. 13).

Figure 13. Monitored (grey bars) and modeled (white bars) average flow weighted concentrations of total phosphorus (TP), dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and suspended solids (SUSP, only monitored concentration) from the ten sampling points representing different upstream sub-catchment areas in Paper V.

4.3.2 Catchment factors affecting P concentrations

In Paper V, some of the characteristics of the ten different sub-catchments were related to the flow-weighted average concentrations of P (TP, DRP and PP) and particles. For P, the soil type in the sub-catchments was important (Table 3 in Paper V), and there was a positive relationship between a high proportion clay soils and high losses of TP, DRP and PP. Furthermore, there was a relationship between the size of the sub-catchment and concentrations of TP and DRP, and smaller areas had higher concentrations. PP transport was affected by the proportion of autumn-sown crops (mainly winter wheat and some winter rape), and areas with a low proportion had high PP concentrations. Particle transport was affected by the proportion of fields with vegetables and spring-sown crops – and the higher the proportion of either crop, the higher the particle transport. The relationship between nutrient losses

and different crops could be explained at least to some extent. Fields with autumn-sown crops means that the fields are covered with vegetation during winter, thus protecting the fields from surface erosion. For instance, Ockenden *et al.* (2014) observed the largest transport of particles from fields when rainfall coincided with periods with bare soil or poor crop cover. In Paper V, both PP and particle concentrations were related to the proportion of autumn-sown crops (for particles indirectly as the positive relationship with spring-sown crop was statistically significant).

The regression equations for each investigated nutrient (Table 3 in Paper V) were used to generate 'modeled' nutrient concentrations, and gave a relatively good fit (Fig. 14). The best fit was for DRP ($R^2=0.97$, $p=0.000$) and TP ($R^2=0.72$, $p=0.001$), while the fit was lower for PP and particle concentrations ($R^2=0.50$, $p=0.013$ and $R^2=0.37$, $p=0.04$, respectively), but still fairly accurate.

Figure 14. Comparison of observed flow weighted nutrient concentrations and concentrations derived from multiple regression models for total phosphorus (TP), dissolved reactive phosphorus (DRP), particulate phosphorus (PP) and suspended solids (SUSP). The solid lines represent 1:1 relationships. Data from Paper V.

In Paper I, one of the ways of expressing the P loads to the seven wetlands was as the flow-weighted average TP concentrations based on grab samples collected during one to nine years, depending on the site. In this paper, the catchment factors included in the analyses were the proportion of agricultural land, average P content in the top soils of the catchments, and the average clay content in the top soils. The only catchment factor related to flow-weighted inflow TP concentrations was the clay content, and catchments with high clay content generated high TP concentrations.

Hence, both the studies presented in Papers I and V identified clay soils as the most important contributor to high losses of P from catchments, and thus one of the most important factors to consider when identifying locations where the load of particulate P to a wetland would be high. However, there was a negative relationship between clay content in the catchment and particle and P accumulation in wetlands from Paper III. This suggests that it is difficult to capture the finest clay particles (and the associated P) in constructed wetlands, and especially if the hydraulic load is high (as discussed above). Therefore, it is important to consider the clay content in the catchments when designing and dimensioning future wetlands in order to have them large enough for capturing fine clay particles.

4.4 Summary of method evaluation

4.4.1 P accumulation estimates

As discussed in 1.3.3 and 4.1, it is important to be aware of the various assumptions and limitations connected to water sampling, whether it is at the in- and outlet of a constructed wetland (Papers I, II and IV) or in a stream (Paper V). This thesis also includes an alternative method for estimating particle and P retention, by investigating the annual sediment accumulation on the bottom (Paper III). This method also has some limitations, for example, the sediment can be overestimated if the bottom and sides of the wetland are prone to erosion (Mitsch *et al.*, 2014). This was evident in the most recently constructed wetland included in this thesis, Nyb, where we observed internal erosion and sediment movement within the wetland, as discussed in Paper IV. This internal movement of particles probably led to an overestimation of accumulated material on the sedimentation plates. Internal erosion from the sides of a newly constructed wetland was also observed in one of the wetlands investigated by Ockenden *et al.* (2012). In that study, the authors observed that the sediment loss from the sides of the wetland decreased after the first year, due to vegetation establishment that stabilized the sides of the wetland. It is also possible that similar development of vegetation in Nyb could reduce internal erosion in the coming years.

In summary, estimating P accumulation in wetlands by using sedimentation plates could be a useful method, especially because of the relatively low cost in comparison with flow-proportional sampling of water quality. In a study by Kynkäänniemi *et al.* (manuscript), the sediment accumulation method was compared with mass balance calculations and for some wetlands the two methods did agree. However, it is important to be aware of the potential risk of overestimating particle and P accumulation in constructed wetlands where erosion of the sides and bottom could contribute to the total build-up of particles on the sedimentation plates. This is likely a larger problem during the first couple of years after construction before the plant cover has established and the sides and bottom become stabilized.

4.4.2 Water quality modeling

In Paper III, regional hydro-chemical models were used to estimate the load of both water and P to the wetlands. When comparing the modeled loads with monitoring data that was available for some of the wetlands, there were large differences in both hydraulic and P loads. For example, in Bøl the P load based on water flow and water quality measurements was $182 \text{ kg ha}^{-1}\text{yr}^{-1}$ (Paper I), which is more than eight times lower than the P load estimated using the regional hydro-chemical model. Similarly, in Gen the P load based on monitoring was almost three times lower (Paper I) than what the regional model estimated. The reasons are large discrepancies in *both* the modeled hydraulic load and the modeled P concentrations. In contrast, Olli *et al.* (2009) measured water flow in the ditch upstream of the Wig wetland between 1997 and 2000, and the results corresponded well with the mean hydraulic load calculated using modeled runoff values for this wetland. This is also true for Ber and Ski, where water sampling and flow measurements during 2010–2012 resulted in measured hydraulic loads of similar magnitudes to that modeled (Paper IV). On the other hand, the modeled P concentrations were approximately 1.3 times higher than that measured in Ber and Ski (Kynkäänniemi *et al.*, 2013, Kynkäänniemi *et al.*, manuscript). The discrepancy between monitoring and modeled loads could be due to uncertainties arising when downscaling regional models and input data to local sites. In those cases, the hydraulic and P load estimates were based on models adapted to simulate larger river basins (3000–20 000 ha) or leachate regions (up to 500 000 ha), while the catchment areas of the respective wetlands varied from 22 to 267 ha.

In the larger catchment study, the L-HYPE model calibrated with the catchment data generated TP and DRP concentrations of a similar magnitude to the monitoring data, although the model did not capture the variation between the ten different sub-catchment areas. The modeled PP concentrations, however, were significantly lower

than the monitored concentrations, and the L-HYPE model results did not identify the areas with the highest nutrient contributions, as indicated by monitoring data.

The results from both Paper III and Paper V showed that it is difficult to use regional hydro-chemical models to estimate nutrient transports from small catchments. The scale is an important factor to bear in mind when discussing mitigation strategies in large agricultural catchments. The exact location of the mitigation measure – whether it is a constructed wetland, buffer zone or a two stage ditch – needs to be determined in a very small scale (*i.e.* in the vicinity of individual fields), but in order to identify *where* the measures would be most efficient, whole catchments (*i.e.* larger scale) have to be evaluated with respect to sub-catchments contributing most to nutrient losses (Paper V).

5 Conclusions

- Particle accumulation varied from 13 to 108 t ha⁻¹yr⁻¹ and PP accumulation varied from 11 to 175 kg ha⁻¹yr⁻¹ in seven wetlands constructed in agricultural catchments with varying clay content. In a more low-loaded wetland, P retention was relatively low (3 kg ha⁻¹yr⁻¹).
- A substantial amount of the deposited material in four wetlands was resuspended. Only 13-23% of the deposited particles stayed as accumulated sediment.
- Particulate P dominated both inflow and outflow in one wetland. Almost 80% of the incoming P settled close to the inlet, and the outflow PP was therefore probably generated from within the wetland itself, for example due to resuspension.
- In seven constructed wetlands there were significant relationships between water flow and inflow TP concentrations. This has implications for sampling strategy, and highlights the importance of composite flow-proportional sampling when monitoring P retention in wetlands. Future evaluations of constructed wetlands should include flow-proportional sampling, where inflow water sampling is controlled by the water flow in the inlet, and the outlet sampling is controlled by the outflow.
- High P losses (both particulate and dissolved P) were related to a high proportion of clay soils in a large agricultural catchment. Also, the proportion of winter sown crops in the catchment was related to PP and particle loss.
- Catchment characteristics positively related to P accumulation in wetlands included the average slope, P content of the top soils and livestock density; in contrast, the clay content in the catchment soils was negatively related to P accumulation in the wetlands. This suggests that it is difficult to capture the P associated with fine clay particles in wetlands. In particular, the results suggested that there may be hydraulic load breakpoint, when accumulation of particles and P will be very low. Hence, it is important to consider the clay content in the catchments when designing and dimensioning future wetlands in order to have them large enough for capturing fine clay particles.

- A hydro-chemical model did not capture the large variations between areas obtained with synoptic sampling, suggesting that the value of such models might still be limited when assessing variability on the sub-catchment scale.

- In Sweden, approximately 4400 ha of additional wetlands need to be constructed or restored to reach the Environmental Objective. If the future wetlands were to function as the wetlands investigated in this thesis, the total P accumulation could be between 13 and 770 ton P per year. Future wetlands should be located where P losses are high (*i.e.* in areas with high proportion clay soils, high soil P content, high animal density and high average slope) and dimensioned so the hydraulic load does not exceed 200 m yr⁻¹ in order to capture the fine clay particles. Optimum location and dimensioning would potentially generate the highest local retention effect (770 tons yr⁻¹). However, the effect on the Baltic Sea would be less due to retention in water courses and lakes on the way towards the sea.

Broader perspective and future outlook

In the two papers that investigated the sediment of the wetlands (Papers II and III), P content in the sediment was variable, and in some wetlands very high, as discussed in 4.2.3. In addition, there was significant accumulation of particles in almost all the investigated wetlands. For example, annual accumulation of particles varied from 10 ton ha⁻¹ in Wig to 230 ton ha⁻¹ in Nyb, and in the inlet area in Ste wetland (Paper II) sediment accumulation was 220 ton ha⁻¹yr⁻¹. This sometimes rapid sedimentation accretion requires management, *i.e.* removal of sediment, in order for the wetland to continue to function as traps for particles and P. The Swedish Board of Agriculture recommends that wetland sediment should be removed from so called P-wetlands every fifth year (Börling, 2010).

The question then arises, what to do with the “harvested” wetland sediment? Since the P content is usually relatively high, one recommendation from the Swedish Board of Agriculture is that the sediment should be recycled back to agricultural fields. This could be a soil structure improvement as well as a form of P fertilizer. However, little is known about the potential of P rich wetland sediment as fertilizers in Sweden, and little research has been done on recycling wetland sediment back to agricultural land. However, some studies on the potential for using sediments from aquaculture systems in South East Asia have been conducted, and have been reviewed by Rahman *et al.* (2004). Fish pond sediments are richer in nutrients and organic matter (OM), than sediments from agricultural wetlands. Using fish pond sediments on agricultural land can increase the quality of the soil as high organic matter content could increase the water holding capacity of the soil. On heavy clay soils, application of wetland sediment enriched with high concentrations of OM can potentially improve the quality of the soil by contributing to soil aggregate formation and increasing porosity. The authors further stated that application of fish pond sediment can increase the nutrient levels of the soil. For example, in a study where sediments from Tilapia fingerling production ponds were mixed with compost in a pot study, 67% of the P taken up by plants originated from the sediment (Rahman *et al.*, 2004).

Closing the P loop

In a wider perspective, Tilman *et al.* (2002) predict that by 2050 the global population will be 50% larger than today, and the global demand for grain will have doubled. Hence, there is a need to increase agricultural productivity throughout the world but without causing further detrimental effects on the environment. As mentioned in the introduction, agriculture contributes high amounts of P to aquatic ecosystems. Should past agricultural practices continue to be used in order to double food production, the rates of P loading may triple (Tilman *et al.*, 2002). The International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD) stated in their report from 2009 that if the present industrial agricultural system continue its trajectory, huge problems regarding both human and environmental health will arise (IAASTD, 2009). The report also stated that business as usual is not an option, highlighting the need for more sustainable agricultural methods in the future.

Mineral P used for fertilizer production is a non-renewable resource, since P cycles over geological time scales (Childers *et al.*, 2011). Globally, 80% of the mineral P extracted from rock reserves is used for fertilizer production (Stewart *et al.*, 2005), but only about 25% of the fertilizer P used in agriculture is recycled back to the fields (Childers *et al.*, 2011). Much of the excess P is lost from the fields by erosion and leaching, causing eutrophication problems in recipients downstream.

The main sustainability problem related to the question of P in agriculture is that, globally, the present use of P fertilizers is not sustainable. Since mineral P used for fertilizer production is a finite resource, the reserves will not last. Furthermore, there is no known chemical or technological substitute for P, either in natural ecosystems or in agro-ecosystems (Childers *et al.*, 2011), making the depletion of natural reserves even more severe than ‘peak oil’. A recent study showed that the available P reserves will last for 50 to 100 years with an expected global peak in P production around 2030 (Cordell *et al.*, 2009). The expected time point for ‘peak P’ differs between researchers, but all are in agreement that peak P is inevitable, which means that the availability, affordability and quality of mineral P fertilizer will limit food production in the future.

In order to be able to feed the growing world population, agriculture needs to be intensified. Increasing the area of land in agriculture is not an option due to the large environmental costs (Garnett *et al.*, 2013). The term ‘sustainable intensification’ refers to the approach of increasing food production from existing farmland in ways that preserves the environment, while not undermining the capacity to continue

producing food in the future. In their paper from 2011, Tilman *et al.* conclude that, globally:

A trajectory that adapts and transfers technologies to under-yielding nations, enhances their soil fertility, employs more efficient nutrient use worldwide, and minimizes land clearing provides a promising path to more environmentally sustainable agricultural intensification and more equitable global food supplies.

Here, “employs more efficient nutrient use” could include for instance the effective cycling of P back to the agricultural fields, and recycling wetland sediment rich in P (as several of the wetlands in Paper III) could be one option.

Apart from the necessary changes in agricultural practices, towards general sustainable intensification, there is a need to find alternatives to mineral P fertilizers in the future. Mineral P is a non-renewable resource, and the global distribution of mineral P is very uneven, with approximately 80% of the global P supply allocated to five countries (United States, China, Morocco, Russia and Tunisia; Stewart *et al.*, 2005). This uneven distribution has already begun to produce sociopolitical complexities (Childers *et al.*, 2011), which will probably worsen as we approach peak P, should no alternatives be available. Using P-rich sediments as *one* alternative to mineral P fertilizer could be one way of reducing the future unstable sociopolitical state in the world.

In addition, the quality of the mineral P from mines is decreasing, and will probably continue to do so in the future as reserves of high-quality phosphate rock are being depleted (Stewart *et al.*, 2005). The poorer quality mineral requires more energy and chemical products (*e.g.* sulphuric acid) to purify to a fertilizer of acceptable quality. This means that future mineral P fertilizer will probably be more expensive than today, and the value of other alternative sources of P (from wetland sediments, for instance) will increase. In a recent investigation of sedimentation in constructed wetlands in UK, Ockenden *et al.* (2014) analyzed the Olsen-P (*i.e.* P available for crops) in some sediment samples. The concentration of Olsen-P was similar in the sediment as in the soils in the catchment area, which led the authors to conclude that wetland sediment probably was better for soil remediation than fertilizer, at least if the current fertilizer application practice continues. But since ‘business as usual’ cannot continue, as stated above, there is reason to believe that current practices may change in the future, and the use of P rich sediment could be a viable fertilizer option.

However, as stated by Withers *et al.* (2015), research on P fertilizer substitutes (manures and other bioresources) and their value compared to mineral fertilizer, is surprisingly lacking.

Research needs

Since only a few studies have been performed, and most of those on sediment from fish ponds, the potential for using wetland sediment as fertilizers remains unexplored. Several questions need to be answered, including:

- How is the P bound to the particles in the sediment, *i.e.* how large proportion can be considered to be bioavailable?
- What is the potential crop yield when using sediment as fertilizer?
- Can the crop take up the sediment P or is it unavailable for plant absorption?
- How will sediment from different wetlands differ in fertilizer value? Can the differences be explained by variations in soil type and P content of the soils in the catchments?

Final remark

I believe there is potential for using P-rich sediments as fertilizer in future agriculture. The P content in wetland sediment is evidently high, but further research is needed in order to investigate the availability of the sediment P for crop uptake. In addition, technical innovations for effective harvesting and spreading of the sediment are needed.

Another important function of constructed wetlands (apart from providing P-rich sediment to agriculture) is the overall ecosystem service of water purification. Not only P but also nitrogen (N) can effectively be removed in constructed wetlands, which further enhance their importance as natural filters in the landscape. Another important ecosystem service is biodiversity, because wetlands provide many different ecological niches for plants, invertebrates, birds, amphibians and mammals. In their paper from 2010, Raudsepp-Hearne *et al.* concluded that the greater the multifunctionality of the landscapes the better they are at regulating ecosystem services (such as soil P retention, carbon sequestration and soil organic matter). Hence, an agricultural setting where constructed wetlands are a natural part of the landscape will probably increase the overall ecosystem functionality in addition to water purification and future recycling of P fertilizer.

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