



The role of bed sediments for fractionation of phosphorus in remediated ditches in Sweden

Sediments betydelse för fosforfraktionering i åtgärdade svenska diken

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Abstract

Sweden's environmental quality objectives "A varied agricultural landscape" and "Zero eutrophication", aims to ensure food production while reducing negative impact from fertilizers on the environment. The input ceilings of nutrient concentrations in the Baltic Sea are still critical. A great part of the nutrient pressures on lakes and seas that can be seen today are historically caused by agriculture and have turned into legacy P (phosphorus). However, there are several management practices to improve the nutrient use efficiency in agriculture, e.g., improved drainage, buffer zones, wetlands, or two-stage ditches. A two-stage ditch is a ditch design with terraces, that aims to improve water quality and increase nutrient retention within the landscape. The terraces should even out high and low flow to reduce flooding and stabilize the ditch to reduce erosion.

According to the latest estimations, the N (nitrogen) pressures in the Baltic Sea have slightly improved, but not the P pressures. Yet, the goals of nutrient concentrations and inputs in the Baltic Sea are still not achieved. P has two main pathways, either leaching in dissolved form, or absorbed to soil particles or incorporated in organic material. The P concentrations absorbed to particles are substantially higher than the P concentrations in dissolved form. There are also different types of bonds and bond strengths between P and soil particles.

The aim of this study was to investigate P dynamics in sediments of two-stage ditches, focusing on how hydrology, water chemistry and catchment properties affects P stocks.

In the study, sediments from 10 two-stage ditches in Sweden were sampled to analyse the P fractions: loosely bound P, Fe-P, Al-P, organic bound P and Ca-P, with a chemical extraction method. The P fractions and TP (total phosphorus) were then tested to see if any changes in concentration within and between the sampled two-stage ditches could be found. The P fractions and TP concentrations were also investigated with clay content, pH, DO (dissolved oxygen), BFI (baseflow index) and EPC_0 to see if there were correlations that could explain observed patterns.

The results showed no significant changes for any P fractions or TP concentrations within same two-stage ditch or between the two-stage ditches. There were no significant correlations between the P fractions and TP and the characteristics of the catchment nor chemical and hydrological factors. However, patterns show that HCl-P (Ca-P), BD-P (Fe-P) and TP concentrations seem to be higher within the sediments of the channel rather than in the sediments of the terraces. HCl-P also seems to reside to a higher extent downstream within the sediments of the channels.

Higher P concentrations residing in the channels could indicate that two-stage ditches as a nutrient mitigation measure works as a P sink. Although there are risks of having large amount of P stored in sediments if any extreme weather events would occur and predicted future climate change is suggesting more frequent but intense weather events. Also, larger concentrations of P within stream sediments indicates that further research of P in sediments are needed to fully understand the P dynamics to tackle eutrophication problems and further evaluate two-stage ditches as a nutrient mitigation measure.

There are possible explanations to the varied results. E.g., features of the landscape, mineralogy, past or current agricultural practices, location of drainage pipes and dams, or legacy P could have affected the results.

Keywords: Phosphorus fractionation, agricultural stream, sediment, two-stage ditch, eutrophication

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Abbreviations

SD	Two-stage ditch
TD	Traditional trapezoidal ditch
P	Phosphorus
TP	Total phosphorus
DO	Dissolved oxygen
BFI	Baseflow index
TDP	Total dissolved phosphorus
SRP	Soluble reactive phosphorus
EPC ₀	Net zero phosphorus sorption

1. Introduction

According to Sweden's environmental quality objectives "A varied agricultural landscape" and "Zero eutrophication", the value of agricultural land should be protected to ensure food production and biodiversity while reducing negative impact from fertilizers on human health or the environment (Naturvårdsverket 2021). To enable food production in Sweden, approximately half of the agricultural land needs to be drained to create a favourable soil environment for growing crops. The ditches and small streams connected to the drained fields create valuable habitats as well as act as transport of nutrients and sediment that contribute to eutrophication of lakes and the Baltic Sea (Jordbruksverket 2018; Jordbruksverket 2016). Agriculture is a large contributor to the eutrophication issues in the Baltic Sea that we see today, and the recovery time is long. Efforts to improve the status in the Baltic Sea have been made and N (nitrogen) levels have slightly decreased, but the P (phosphorus) levels are still critically high (Havs- och vattenmyndigheten 2019). Calculations of estimated nutrient load and input ceiling are made every sixth year (Naturvårdsverket 2022) and from the latest follow-up of nutrient load in the Baltic Sea in 2017, the N load from Sweden to the Baltic Sea was 37 096 tonnes, which is 24% too high from the input ceiling. The estimated P input from Sweden was 691 tonnes, which is 132% over the input ceiling (Helcom 2020). This dilemma of ensuring food production while protecting the environment requires sustainable solutions (Ashley et al. 2011; Withers et al. 2015). There are strategies to use nutrients in agriculture in a more efficient way. For example, at a farm level, managing the type, amount, timing, and placement of fertilizer so that it matches the crop needs can both minimize the nutrient waste as well as have productivity and economic benefits for the farmer (Johnston & Bruulsema 2014). In addition to changing the fertilizer inputs, further crop management strategies like crop rotation, tillage practices (Schoumans et al. 2014) and management of drainage water, before it enters streams like constructed wetlands, controlled drainage, drainage filters and buffer zones can also have a positive effect on reducing nutrient and sediment losses to streams (Carstensen et al. 2020). Another approach to reduce pollution is managing the design of agricultural streams and ditches. Traditional ditches require maintenance that can be reduced with implementing a stream design that is more consistent with natural streams. Ditches or streams with constructed terraces, called

two-stage ditches, can provide an alternative to traditional ditches and lead to water quality improvements (D'Ambrosio et al. 2015; Mahl et al. 2015).

As an effort to advance the understanding of the role of two-stage ditches in improving water quality, a Formas project (Hallberg & Bieroza 2021) that evaluates the use of two-stage ditches in agricultural catchments in Sweden is currently ongoing at the Department of Soil and Environment at Swedish University of Agricultural Sciences (SLU).

The aim of this study is to investigate phosphorus dynamics in sediments of two-stage ditches, focusing on how hydrology, water chemistry and catchment properties affects P stocks. This study will contribute to increasing the knowledge and understanding of phosphorus losses from agricultural streams and the processes behind these losses.

1.1 Two-stage ditches

The original idea and purpose of a two-stage ditch is to minimize nutrient losses from a catchment by evening out high and low flows. That reduces risk of flooding and erosion with a terrace design (Figure 1) that increases water retention in the landscape. With this design, the maintenance of a two-stage ditch would occur less frequent compared to a traditional ditch. That in turn would favour the terrace vegetation and biota to establish and catch nutrients and sediments as well as stabilize the ditch (D'Ambrosio et al. 2015; Hodaj et al. 2017; Jordbruksverket 2016). In addition to water quality and stream stability improvements, there are cost benefits with reduced maintenance (D'Ambrosio et al 2015). Although there are many potential advantages to gain from constructing a two-stage ditch, the prospects of constructing a successful two-stage ditch highly depends on the properties and suitability of the catchment area (D'Ambrosio et al. 2015; Jordbruksverket 2016).

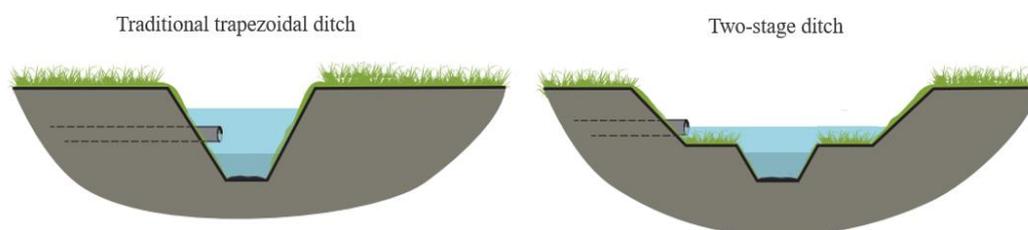


Figure 1. Traditional trapezoidal ditch and two-stage ditch illustrated in cross sectional view (Jordbruksverket 2016).

1.2 Phosphorus pathways in ditches and streams

Despite implementation of the EU water framework directive that aims to improve ecological and chemical status in surface and groundwaters, water quality in many rivers and lakes remains poor, largely due to diffuse pollution sources from land to water systems (Schoumans et al. 2014). Moreover, even if successfully reducing P inputs in a catchment, results of decreased P levels in streams and lakes may not immediately show. Often the impact of mitigation measures is poor due to legacy P that has accumulated for decades in soils and sediments from past agricultural management and sewage treatment systems. The legacy P residence time is not certain, though the legacy P can be released for many years, creating time lags for implemented mitigation measures leading to insufficient results in the short-term (Lannergård et al. 2020; McCrackin et al. 2018). P enters streams mainly via two pathways: leaching in dissolved form through the soil profile or erosion bound to particles or incorporated in organic material (Schoumans et al. 2014). The P concentrations leaching in dissolved form through the subsurface are usually quite small (Tesoriero et al. 2009). Clay and silty soils have a larger risk of eroding and thus for P losses. Some areas in agricultural landscapes can act as nutrient sources and some can act as sinks, this means that a catchment or a larger area as whole can have a net buffering capacity (Schoumans et al. 2014).

1.3 Forms of phosphorus in ditches and streams

The forms of P in the soil are strongly regulated by pH, mineral composition of the soil, amount of organic material, redox state, and biological activity (Eriksson et al. 2011; Schoumans et al. 2014). The fractions that P can include, in order of bond strength, are loosely bound P or water soluble P, Fe-bound P, organic-bound P, Al-bound P and Ca-bound P (Condron & Newman 2011; Eriksson et al. 2011; Kindervater & Steinman 2019; Lannergård et al. 2020). Most soils have a large capacity for binding P, and the fraction that is loosely bound to soil particles or in soil solution that is directly accessible for plants is usually quite small (Eriksson et al. 2011). The Fe-P concentration is larger than the loosely bound fraction and somewhat accessible when changes in redox state, bioactivity and pH occurs (Eriksson et al. 2011; Sandström et al. 2021). P bound to organic material can constitute for a large part of the TP in the soil and therefore make up an important P source (Eriksson et al. 2011; Kindervater & Steinman 2019). Organic bound P can be viewed as difficult to access, but upon mineralization this P fraction can also become bioavailable (Eriksson et al. 2011). The availability of Al-P is pH regulated but considered stable within stream pH ranges (Sandström et al. 2021). The Ca-bound P can be very difficult access and depends on the Ca level in the soil (Eriksson et al. 2011). The sum of the earlier mentioned P fractions approximately

adds up to TP concentration in each sample (Lannergård et al. 2020; Psenner & Pucsko 1988). See Figure 2 for example.

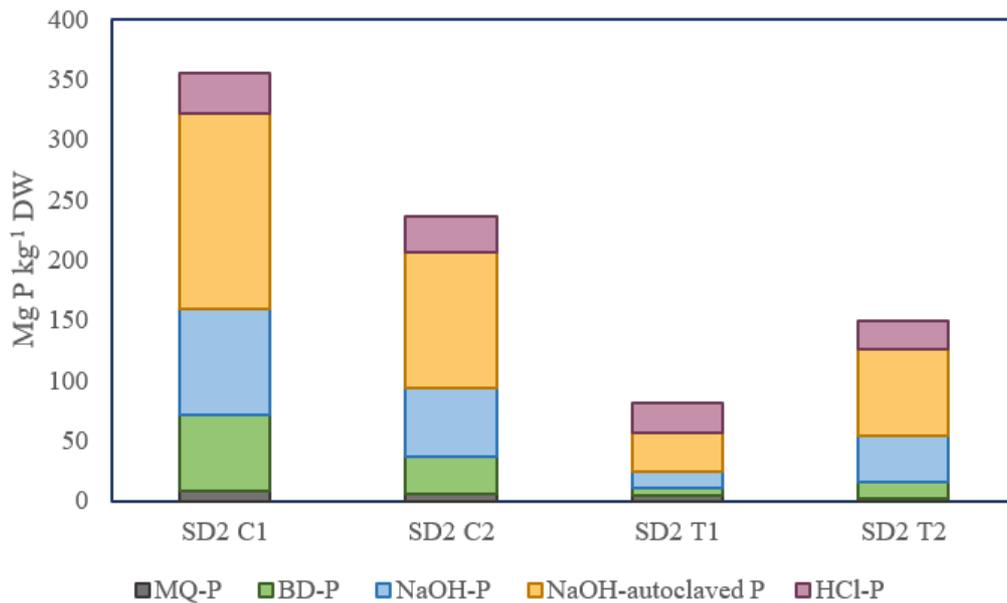


Figure 2. Example of how the P fractions contribute to TP at site SD2 in channel (C) and terrace (T) at upstream (1) and downstream (2). The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) add up to TP for each sampled site.

1.3.1 Phosphorus fractionation

The forms of P present in sediments and water are governed by a number of chemical and biological properties and processes. It is therefore of great importance to characterize the forms of P in soils and sediments to enable detailed investigation of P transformations and understanding what factors that regulates the forms of P (Condrón & Newman 2011; Lannergård et al. 2020). Additionally, P speciation is of the essence when estimating potential losses of P in streams to successfully reduce eutrophication (Lannergård et al. 2020). To determine forms of P in soil and sediment samples, separation, and identification of different forms of P is required. A common chemical extraction implementation for P speciation is sequential extraction that selectively and step by step removes different forms of P with specific reagents. There are numerous fractionation approaches developed that include determination of different P fractions or combinations of them (Condrón & Newman 2011). There are also several other techniques, e. g. XANES and NMR, that are spectroscopic techniques for P speciation (Kizewski et al. 2011).

Using, P fractionation, Kindervater & Steinman (2019) showed that sediment samples from two-stage ditches and traditional ditches in the midwestern US with loamy soils showed no significant changes in P fractions spatially between channels

and terraces. The same study also showed no significant changes in P fractions over time between spring and autumn. The study suggested that P retention is impacted by biochemical and physical stream characteristics and age of the constructed two-stage ditch. In a study by Sandström et al. (2021), sampled soil and sediment from Mideast Sweden implied that P fractions differ among ditches and streams depending on geology, clay content, P sources and flow conditions. There can be seasonal changes for the P fractions, due to changes in flow, temperature, and redox state. Rattan et al. (2019) investigated the TP, TDP (total dissolved phosphorus) and SRP (soluble reactive phosphorus) in grab water samples and from agricultural streams in midwestern Canada and US. The results showed significant changes in those P fractions over seasons and years, mainly due to flow changes that are connected to seasonal weather events. Hodaj et al. 2017 also used grab water samples in midwestern US. TP decreased in two-stage ditches, but the study suggests that it is highly dependent of weather events and flow.

1.3.2 Phosphorus sorption

To successfully reduce eutrophication and gain improved understanding of what regulates potential P losses from sediment to stream water, the P sorption and desorption to sediment particles is of importance. Sediments do in most cases have the potential to absorb P to particles or desorb P into stream water. This depends on the characteristics of the stream and the sediment, pH, and exchangeable P concentrations. Calculating EPC_0 is a common method for measuring P sorption and desorption of sediments. EPC_0 is the sediment equilibrium P concentration at net zero P sorption. If the stream water P concentration is larger than EPC_0 , the sediments act as a sink (P adsorption to sediments) and if the stream water P concentration is smaller than EPC_0 , the sediments act as a source (P desorption to sediments) (Agudelo et al. 2011; Simpson et al. 2021).

1.4 Aims and study objectives

The aim of this study is to investigate phosphorus dynamics in sediments of two-stage ditches, focusing on how hydrology, water chemistry and catchment properties affects P stocks. To further understand phosphorus dynamics in two-stage ditches, the study objectives were to collect sediment samples from upstream and downstream channels and terraces in 10 two stage-ditches. The sediment samples were analysed for the following P fractions: loosely bound P, Fe-P, Al-P, organic P and Ca-P using a sequential fractionation of P in sediment based on the method developed by Psenner & Pucsko (1988) and Hupfer et al. (1995, 2009). Further, correlations of different P fractions and variables describing biogeochemical properties and processes, such as clay content, pH, DO (dissolved

oxygen), and BFI (baseflow index) for all 10 sampled SD sites was investigated. Relationships of the P fractions and EPC_0 were also examined in four of the SD sites, using data from another MSc projects by Linus Holgersson (2022, in review). Following research questions were investigated:

1. Are there any significant changes in P concentrations and fractions along selected two-stage ditches?
2. Can any of the measured variables related to hydrological and biogeochemical processes explain observed patterns in P fractions?

2. Material and methods

The ten studied SDs (two-stage ditches) are located in Mideast and Southern Sweden according to Figure 2. The SDs were constructed between 2012 and 2019 and have been monitored by SLU since 2020. P fractionation data used in this project were obtained from channels and terraces in all ten SD sites on one occasion in February 2022. The pH and DO data were sampled from the channels at the same occasion as the sediment samples were taken. BFI was calculated from flow data between spring 2020 and spring 2021, each SD has one assigned BFI for upstream and downstream respectively. BFI is an index, in the range of 0-1, of base flow relative to total flow volume, measured for a specific period, which can be used as an indicator for measuring relative surface runoff or baseflow volume. High BFI means more groundwater flow and low BFI means more runoff flow (Tesoriero et al. 2009; Gustard et al. 1992). The sorption data were obtained from a parallel MSc project on three occasions on February, March, and April in 2022 from SD2, SD3, SD7 and SD8. The clay content data was obtained from the Swedish arable soil map (Söderström & Piiki 2016).

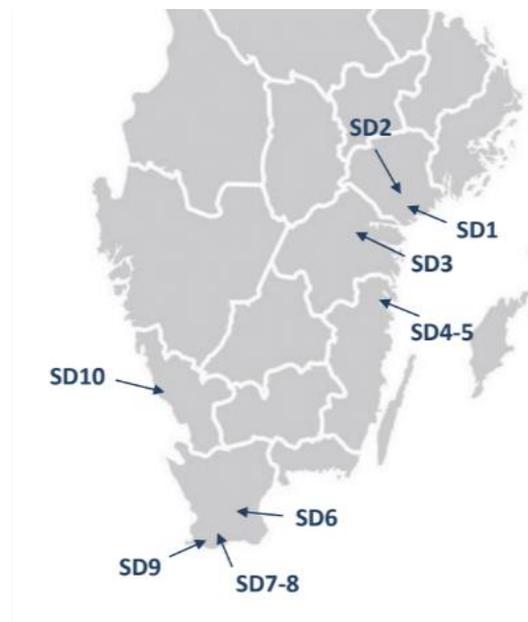


Figure 3. Locations of the studied SD sites (reproduced from Lukas Hallberg).

2.1 Site description

SD1, SD2, SD3, SD4 and SD5 are located in Mideast Sweden in areas dominated by agricultural and forest land use. They were constructed between 2012-2019 and have a high clay content of $\geq 40\%$. SD6, SD7, SD8, SD9 and SD10 are located in Southern Sweden with lighter and more sandy soils. They also have a larger agricultural land use proportion in their catchments than the SDs in Mideast Sweden. The length of the SDs varies from 320 to 1960 meters, and so does the catchment area, 7.11 km² – 42.41 km² (Table 1).

Table 1. Summary of the catchment and SD descriptions. Site name, length of the SD, catchment area of the SD, clay content in the SD and agricultural land use. The agricultural land use includes arable land and pasture land (data from Lukas Hallberg et al., in review)

Site	Construction year	SD length (m)	Catchment area km ²	Clay content %	Agricultural land use %
SD1	2013	340	9.73	48	16
SD2	2012	730	7.91	40	27
SD3	2014	1500	7.11	48	60
SD4	2019	320	8.12	41	35
SD5	2012	780	16.32	40	38
SD 6	2016	400	13.09	16	84
SD 7	2013	1960	10.84	17	81
SD 8	2013	1770	42.41	15	81
SD 9	2019	630	31.02	12	86
SD 10	2014	1760	16.38	4	58

2.2 Data collection

Sediment samples were collected from all SDs in February 2022. The sediment samples were taken to determine the phosphorus fractionation status for each ditch and compare it within the ditches as well as between the ditches. Only one set of samples was taken from each SD since the results of Kindervater & Steinman

(2019) and Sandström et al. (2021) imply that the P fractionation in streams does not significantly change over seasons, but it is under the influence of the long term-variables such as geology, clay content, P sources and flow conditions within the SD.

2.2.1 Sampling

Four samples at each SD were collected, one sample from the terrace and one from the channel both upstream and downstream of the SD. For each terrace, 10 samples of 3 cm³ with 5 meters between were mixed in a sampling bag to create composite terrace samples. As for the channel samples, 5 samples of 5 cm³ were taken in the middle of the stream with 10 meters apart to create composite channel samples. Because of high flow downstream at SD8 at the time of sampling, no channel sample could be taken. At SD10, the terrace upstream had collapsed so no terrace sample could be taken either. The number of samples collected adds up to a total of 38 samples from the ten SDs that were used for phosphorus fractionation. The sediment samples were named after each ditch SD1-10 with an indication where the sample was taken, channel (C) or terrace (T), and depending on whether the sample was taken from upstream (1) or downstream (2). E.g., SD2T2 is sediment sample from the downstream terrace in SD2 and SD7C1 is sediment sample from the upstream channel in SD7.

2.3 Sequential phosphorus fractionation

The sediment samples were analysed in sequential chemical extraction method commonly known as phosphorus fractionation, where the approximate fractions of phosphorus in the sediment samples can be obtained. An established protocol from the Department of Aquatic Sciences and Assessment at the SLU was followed for this procedure. The protocol is based on the method developed from Psenner & Pucsko (1988) and Hupfer et al. (1995, 2009). The procedure starts with removing loosely bound P and stepwise remove more strongly bound P in the sediment samples. According to the protocol, the phosphorus pools that can be obtained in the method, in order from most loosely bound to strongest bound, are loosely bound P, redox sensitive Fe/Mn-bound P, Al-hydroxides bound P and non-reducible P, organic P, Ca-bound P and refractory P. See Table 2 for corresponding fractionation steps that gives an indication of the approximate P speciation.

The accuracy of the P fractions obtained from sequential chemical extractions has received some criticism. When comparing sequential chemical extractions with reference spectroscopy methods like e.g., XANES, the P fraction concentrations can show different results. Chemical extraction methods can overestimate some P pools due to redistribution of P during extractions steps, or not completely dissolve

the expected P fraction and leave residual P that is extracted during the next extraction step, giving misleading results (Gu et al. 2020). However, many techniques for P speciation have disadvantages and a chemical extraction method can still provide a somewhat picture of the P fraction status (Gu & Margenot 2021).

Table 2. Sequential fractionation step, its corresponding phosphorus fractionation and relative bond strength to the five fractions (Eriksson et al. 2011; Psenner & Pucsko 1988).

Fractionation step	Phosphorus fraction	Bond strength
MQ-P	Loosely bound	Loose
BD-P	Fe/Mn-bound	Intermediate/Strong
NaOH-P	Al-bound	Strong
NaOH-autoclaved P	Organic material bound	Bioactivity dependent/strong
HCl-P	Ca-bound	Very strong

2.3.1 Removing loosely bound P, MQ-step

The sediment sample bags were well mixed before starting the fractionation. Between 100 and 200 mg of wet sediment from each of the 38 samples were added to 15 ml centrifuge tubes with a spoon. 10 ml of MQ-water was added to the centrifuge tubes and shaken to make sure all the sediment particles ended up in solution and settled for two hours. The centrifuge tubes were then put in the centrifuge for 10 minutes at 3000 rpm. 5 ml of the supernatant from each centrifuge tube was pipetted to scintillation vials. The remaining supernatant in the centrifuge tubes was discarded without disturbing the sediment in them. 5 ml of MQ water was added to the scintillation vials to make 10 ml in total. 3 scintillation vials with 10 ml MQ water were used as blanks. All the scintillation vials were covered with parafilm and put in a fridge at 4°C overnight.

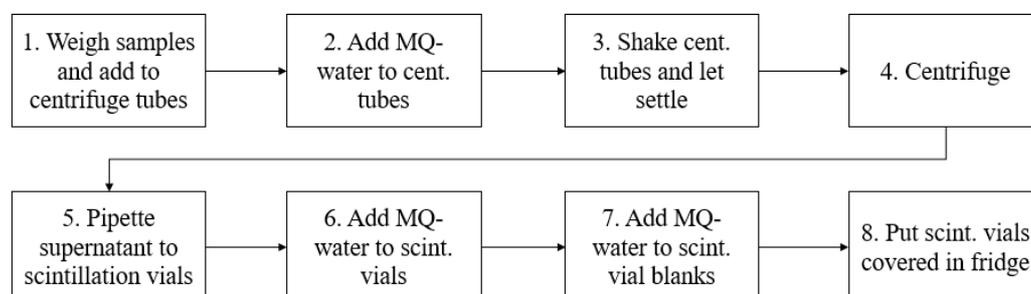


Figure 4. Summarized scheme of the MQ-step in the sequential phosphorus fraction procedure.

2.3.2 Removing redox sensitive Fe/Mn bound P, BD-step

10 ml of BD-solution ($\text{Na}_2\text{S}_2\text{O}_4/\text{NaHCO}_3$) was added to the centrifuge tubes, mixed with a spatula to make sure all sediment particles ended up in the solution and settled for 1 hour. The centrifuge tubes were centrifuged for 10 minutes at 3000 rpm and 2 ml of the supernatant from each tube was put in new scintillation vials and the remaining supernatant was discarded. 2 ml of BD-solution was put in 3 scintillation vials to be used as blanks. All the scintillation vials were put uncovered in a fridge at 4°C overnight to oxidize.

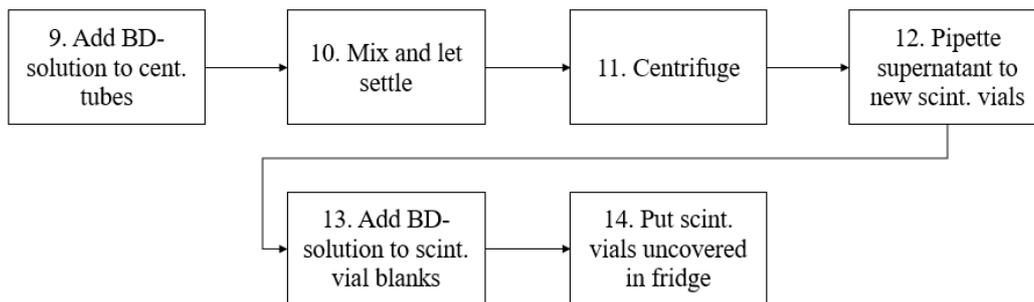


Figure 5. Summarized scheme of the BD-step in the sequential phosphorus fraction procedure.

2.3.3 Bound to Al-hydroxides and organic bound P, NaOH-step

10 ml of 0.1 M NaOH was added to the centrifuge tubes and mixed with a spatula and left for 16 hours. The centrifuge tubes were centrifuged for 10 minutes at 3000 rpm and 2 ml of the supernatant was added to new scintillation vials and digestion tubes respectively, the remaining supernatant in the centrifuge tubes was discarded. 2 ml of NaOH 0.1 M was added to 3 scintillation vials and 3 digestion tubes to be used as blanks. 0.5 ml of 0.5 M HCl was added to the scintillation vials and digestion tubes respectively, including the blank ones. 3 ml of $\text{K}_2\text{S}_2\text{O}_8$ was added to the digestion tubes, including the blanks. MQ-water was added, 7.5 ml to the scintillation vials and 4.5 ml to the digestion tubes so the total volume in all the vials and tubes add up to 10 ml. The digestion tubes were concealed with caps and put in autoclave for 30 min at 120°C. The scintillation vials were covered with parafilm and put in a fridge at 4°C and the digestion tubes were put in room temperature overnight.

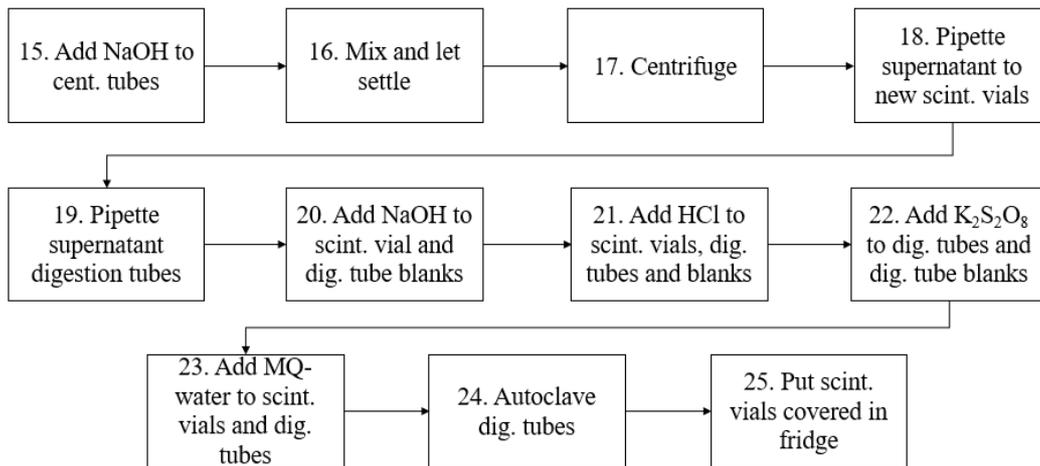


Figure 6. Summarized scheme of the NaOH step in the phosphorus fractionation procedure.

2.3.4 Ca bound and refractory P, HCl-step

10 ml of 0.5 M HCl was added to the centrifuge tubes and mixed with a spatula to make sure all particles end up in the solution and let the samples sit for 16 hours in room temperature. P-standard solution was poured into a beaker to bring it to room temperature. The HCl scintillation vials were centrifuged for 10 minutes at 3000 rpm and 2 ml of the supernatant was pipetted to scintillation vials. 2 ml 0.5 M HCl was pipetted to 3 scintillation vials to be used as blanks. 0.5 ml 2 M NaOH was pipetted to scintillation vials. 7.5 ml MQ water was pipetted to all the scintillation vials to make a total volume of 10 ml, including the blanks. 8 ml of MQ water was also added to the BD-scintillation vials to make a total volume of 10 ml.

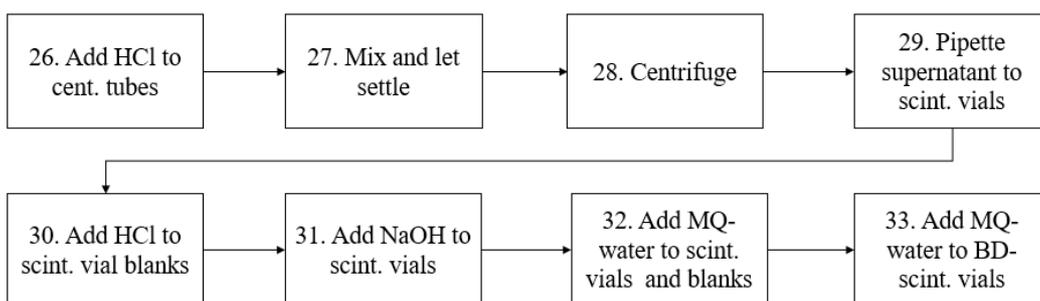


Figure 7. Summarized scheme of the HCl-step in the phosphorus fractionation procedure.

2.3.5 Measuring sediment phosphorus concentrations with a spectrophotometer

The MQ-samples (scintillation vials), the BD-samples (scintillation vials), NaOH (scintillation vials), NaOH (digestion tubes) and HCl-samples (scintillation vials) were analysed using the spectrophotometer to measure absorbance and obtain the P fraction concentrations. A P-standard was made and diluted in 6 scintillation vials with varying P concentrations between 0-800 $\mu\text{g P L}^{-1}$ to calibrate the spectrophotometer. The calibration curve showed a R^2 value of 1.00 and straight-line equation $P=0.72Abs-1.11$. The straight-line equation was used to convert the result fractionation values from nm to $\mu\text{g P L}^{-1}$ (Figure 8).

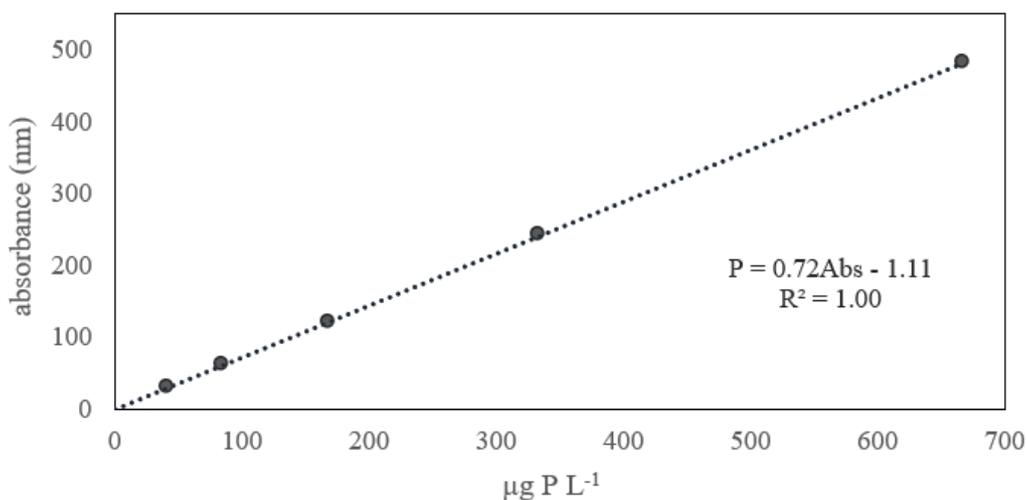


Figure 8. Calibration curve used for converting the absorbance values from nm to P concentration $\mu\text{g P L}^{-1}$.

2.3.6 Dry weights and conversion of units

The P fraction units were then converted from $\mu\text{g P L}^{-1}$ to mg P kg DW^{-1} . The reason for converting the units of the P concentration in the samples is that previous studies within the subject from Lannergård et al. (2020) and Kindervater & Steinman (2019) presented their results in mg P kg DW^{-1} .

New samples of 80 ml from respective sediment bags were weighed to determine the fresh weight for each of the 38 samples. The samples were then individually weighed in aluminum containers to obtain the fresh weight and after that heated in an oven at 105°C for 24 hours, to determine the dry weight. By dividing the dry weight with the fresh weight, a quota of them, a weight index, was obtained and then used to calculate the dry weight in the fractionation samples. See calculation below.

$$\frac{\text{Dry weight}}{\text{Fresh weight}} = \text{Weight index}$$

$$\text{Weight index} \times \text{Sample fresh weight} = \text{Sample dry weight}$$

When the dry weight of the fractionation samples was obtained, it could be used to convert the P concentration in $\mu\text{g P L}^{-1}$ of solution to mg P kg DW^{-1} of soil in the fractionation samples. The samples contained 10 ml of P in solution. See calculation below.

$$P \text{ concentration } (\mu\text{g L}^{-1}) \times 0.01 \text{ (L)} = P \text{ mass in sample } (\mu\text{g})$$

$$\frac{P \text{ mass in sample } (\mu\text{g})}{\text{Sample dry weight (g)}} = \frac{P \text{ mass in sample (mg)}}{\text{Sample dry weight (kg)}} = \text{mg P kg}^{-1} \text{ dry weight}$$

2.3.7 Phosphorus sorption and desorption, EPC_0

The method used for measuring sediments P buffering capacity was similar to Trentman et al. (2020), where the equilibrium phosphate concentration at net zero sorption (EPC_0) was calculated. Sediment and water samples were collected for this analysis in SD2, SD3, SD7 and SD8 at three occasions, one in February, one in March, and one in April 2022. The EPC_0 analysis was done by another MSc project by Linus Holgersson.

A P-standard solution and stream water were added to sediment samples. They were incubated, centrifuged, and filtered to measure P concentration absorbance (nm) in a spectrophotometer. The P-standard was also added to stream water samples to measure P concentration with the spectrophotometer. A calibration curve like Figure 8, was obtained by diluting the P-standard with varying P concentrations between 0-2000 $\mu\text{g P L}^{-1}$, to convert the P concentration absorbance values (nm) to $\mu\text{g P L}^{-1}$. Dry weight of the sediment samples was obtained like described in 2.3.6. To calculate EPC_0 , the total added P (P-standard and stream water before incubation) is plotted against the change in P (difference between P concentrations before and after incubation when P has either absorbed or desorbed from the sediment). The units were converted from $\mu\text{g P L}^{-1}$ to $\mu\text{g P L}^{-1} \text{ g}^{-1} \text{ DW}$ to relate to the added sediment before incubation. The intercept in the straight-line diagram is the EPC_0 , if EPC_0 is lower than the stream concentration, the sediment is a sink and if EPC_0 is higher than the stream concentration the sediment is a source.

2.3.8 Visual and statistical analysis in Excel

When the raw data was converted from nm to mg P kg⁻¹ DW, the 38 sampling sites were presented to give an overview of the P fraction status in every two-stage ditch. Statistical analysis of P fraction concentrations and TP concentrations between upstream and downstream, and channels and terraces in the two-stage ditches were done using a t-test with a 95% confidence interval. The P fraction status in the two-stage ditches were also put into correlation with numerous factors; clay content, pH, DO (dissolved oxygen), BFI (baseflow index) and EPC₀ to investigate if any significant patterns in P fraction dynamics could be found.

3. Results

The general distribution of the P fractions in all the 10 SDs can be seen in Figure 9. Lowest contribution to TP in the sediments and least varied fraction is the MQ-P fraction, which varied from 0-11 mg P kg⁻¹ DW. The BD-P and NaOH-P fractions had a similar amounts of mg P kg⁻¹ DW, ranging between 0-120 mg P kg⁻¹ DW for BD-P and 0-88 mg P kg⁻¹ DW for NaOH-P. The NaOH-autoclaved P fraction was both the largest fraction and had the largest variation in distribution between 4 – 160 mg P kg⁻¹ DW. HCl-P varied from 14 – 60 g P kg⁻¹ DW (Figure 9).

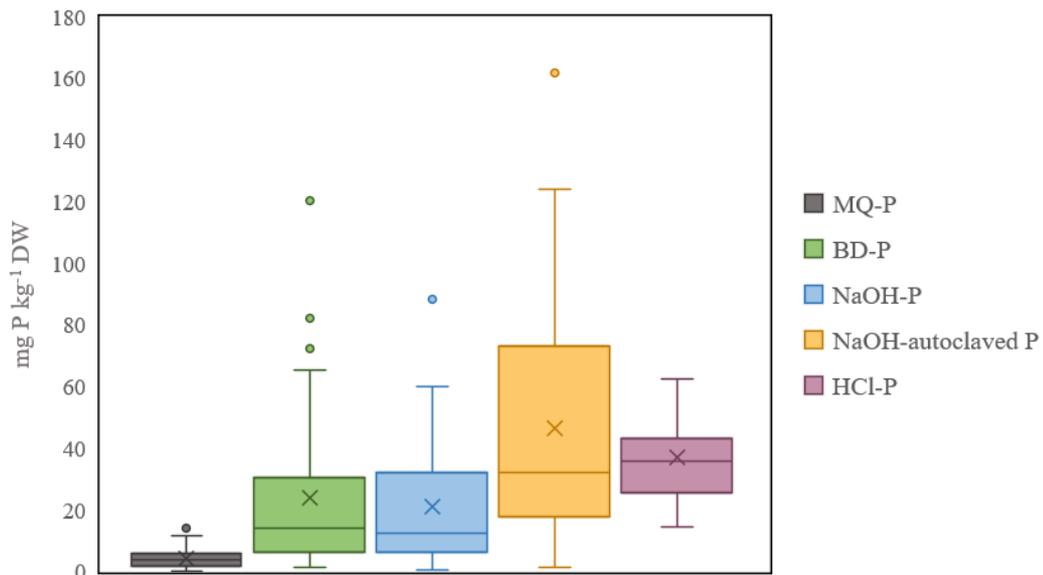


Figure 9. Distribution of P fractions in mg P kg⁻¹ DW from terraces and channels, upstream and downstream in SD1-SD10. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) are shown.

In Figure 10 the variation of the fractions can be seen in %. When comparing it to Figure 9, the fractions NaOH-autoclaved P and HCl-P have a changed distribution. The relative contribution in % from NaOH-autoclaved varied between 0% and 60%. For HCl-P the relative fraction varied between 10–90% of TP (Figure 10).

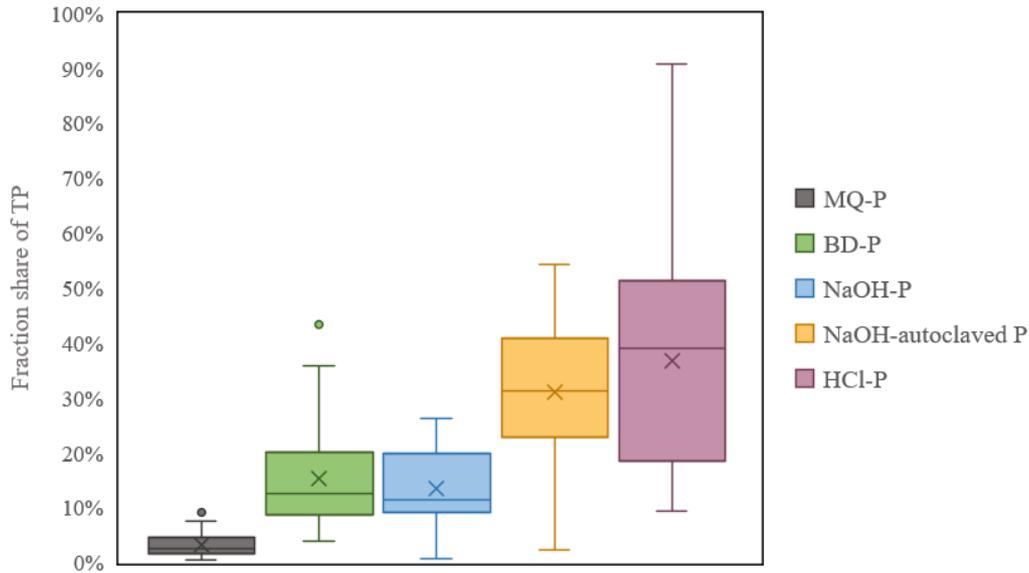


Figure 10. Distribution of P fractions in % from terraces and channels, upstream and downstream in SD1-SD10. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) are shown.

The P fractions and the TP varied between SDs, but between channels and terraces, no clear pattern was found (Figure 11). The TP values varied remarkably between some of the sites e.g., SD1 and SD9. The highest TP values ($>200 \text{ mg P kg}^{-1} \text{ DW}$) were found in the sediment of the channels, both before and after the SDs. The contribution of loosely bound P fraction was low in all the SDs (Figure 11).

SD1 had similar stable levels of TP and fractions of P before and after the SD in the channel and on the terrace. The NaOH-autoclaved P fraction was the largest both in channels and terraces. TP varied from $150\text{-}250 \text{ mg P kg}^{-1} \text{ DW}$ (Figure 11).

SD2 had variation within the SD. SD2C1 had the largest amount of TP of all the sampled sites, $350 \text{ mg P kg}^{-1} \text{ DW}$ and SDC2 had reduced TP to $250 \text{ mg P kg}^{-1} \text{ DW}$. The terraces in SD2 had a lower TP than the channels, ranging between $50\text{-}100 \text{ mg P kg}^{-1} \text{ DW}$. The NaOH-autoclaved P fraction was the largest fractions in all sampled sites in SD2, $40\text{-}50\%$ of TP.

SD3 and SD4 had a lower and similar amount of TP compared to SD1 and SD2, ranging from $80\text{-}150 \text{ mg P kg}^{-1} \text{ DW}$, they also had a larger fraction of HCl-P, and smaller amount of NaOH-autoclaved P.

SD5 had varying amounts of TP in the sampling sites. A large reduction of TP from $275 \text{ mg P kg}^{-1} \text{ DW}$ in SD5C1 to $100 \text{ mg P kg}^{-1} \text{ DW}$ in SD5C2. The NaOH-autoclaved P, NaOH-P and BD-P fractions were reduced while the HCl-P increased. There was a reduction of TP in SD5T1 to SD5T2 as well, from $150 \text{ mg P kg}^{-1} \text{ DW}$ to $100 \text{ mg P kg}^{-1} \text{ DW}$. The distribution of the fractions was similar with a slight decrease for HCl-P from SD5T1 to SD5T2.

SD6 had a noticeable TP increase in SD6C1 from 125 mg P kg⁻¹ DW to 275 mg P kg⁻¹ DW in SD6C2. The terraces were very similar upstream and downstream.

SD7 had a similar pattern of change as SD6, increase in TP from upstream to downstream in the channels and close to unchanged on the terraces. There was a noticeable increase in the BD-P fraction from SD7C1 to SD7C2, where it is nearly 50% of the total fractionation share.

SD 8 had, compared to other studied sites, a lower amount of TP at all sampling sites and high fractionation shares of HCl-P. No comparison could be made between upstream and downstream in the channel and the P fraction distribution on the terrace is stable both upstream and downstream.

SD9 and SD10 had similar patterns in all sampled sites and had the lowest amount of TP and highest HCl-P fraction compared to the rest of the SDs, especially for the channels in SD9C2 (Figure 11).

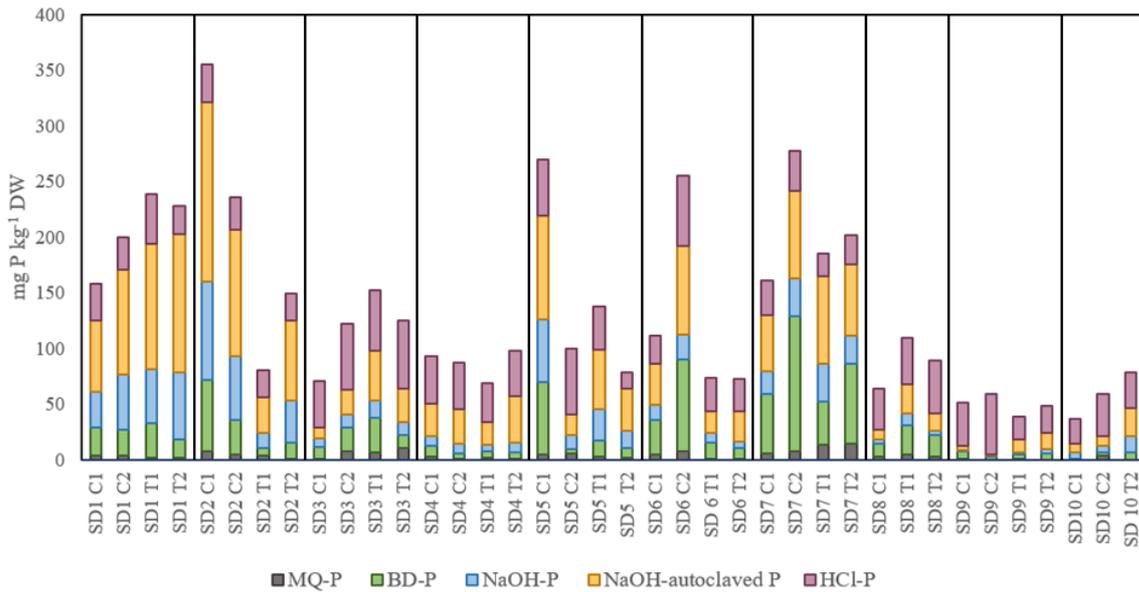


Figure 11. P fractions and TP from all sampled sites in mg P kg DW⁻¹. The lines separate the SDs. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) add up to the TP for each sampled site.

3.1 Net change of phosphorus fractions between upstream and downstream, channels and terraces

The P fractions in all 10 SD sites were compared from upstream to downstream with a t-test. The comparison of the sediment in the channels showed no significant changes between upstream and downstream for any of the P fractions or TP. However, the HCl-P fraction showed a trend of increasing downstream in the

channel, but this was not significant (p-value of 0.08). The MQ-P fraction also showed the same trend, increasing downstream of the channel (p-value of 0.23). The BD-P fraction distribution had more varied values downstream than upstream, especially in the fourth quartile. The NaOH-P fraction distribution values were very similar comparing upstream and downstream. The NaOH-autoclaved P fraction had more varied values upstream than downstream. The TP showed signs of increased concentration downstream compared to upstream, although the distribution of the values upstream in the fourth quartile was more varied than downstream (Figure 12).

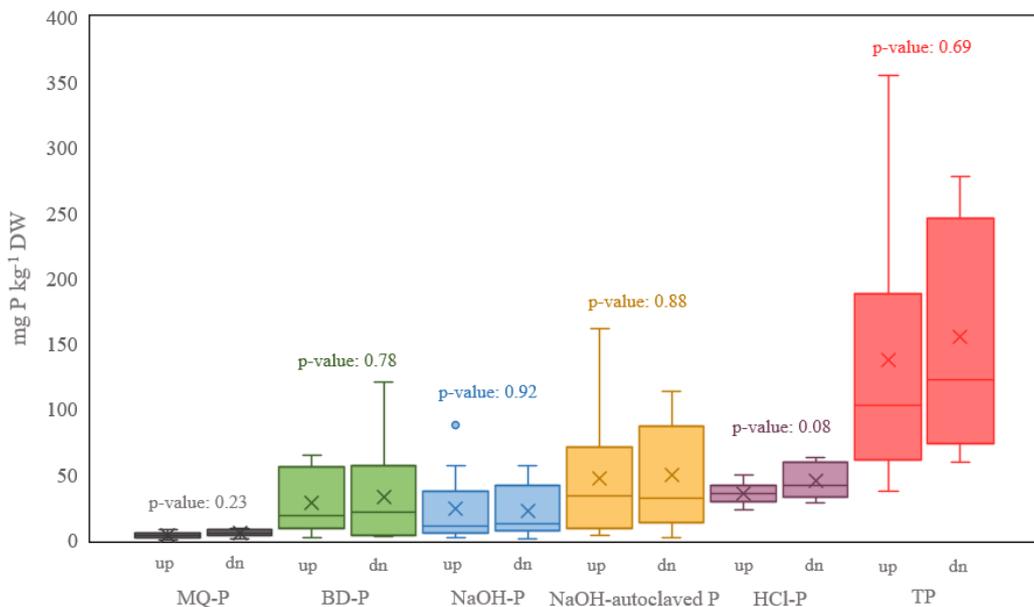


Figure 12. P fraction and TP box plots comparison between upstream (up) and downstream (dn) in the sediment of the channels for all the sampled SDs, including average value (the cross in the boxplot), median (the line in the box plot) and p-value. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP are shown.

The comparison of the sediment in the terraces showed no significant changes between upstream and downstream. They all had p-values of 0.72 or higher, and the distribution of the box plots upstream and downstream of the same P fraction were similar. The MQ-P and the HCl-P downstream had lower concentration values in the second and third quartile, but wider distribution of the concentration values than upstream. The BD-fraction had one extreme value in the downstream box plot that increases the average value, although the third and fourth quartile was smaller downstream than it was upstream. The NaOH-P and the NaOH-autoclaved P had very similar boxplots upstream and downstream (Figure 13).

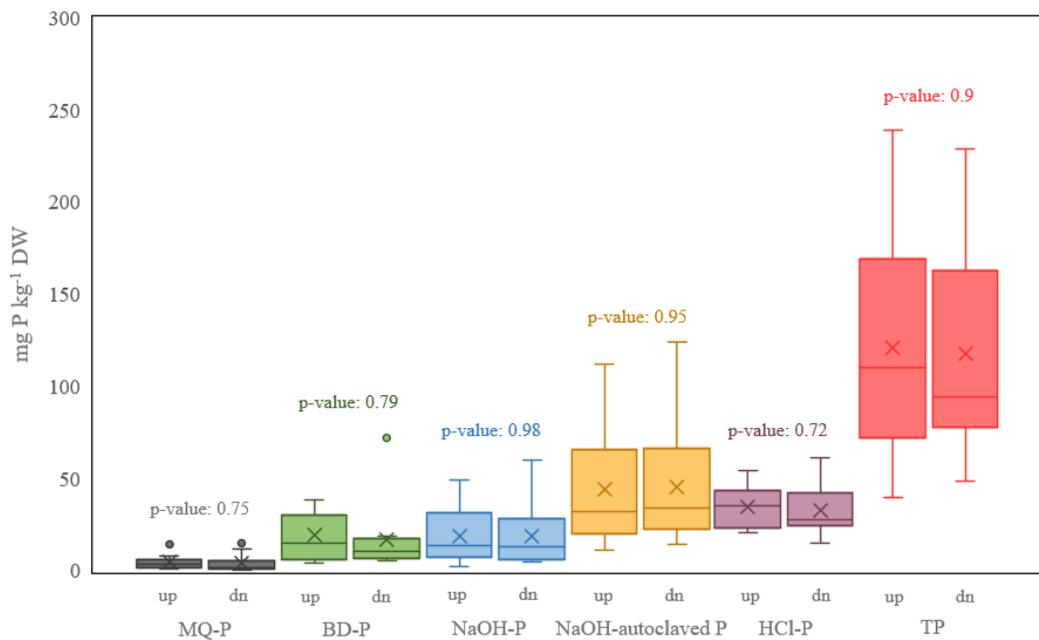


Figure 13. P fraction and TP box plots comparison between upstream (up) and downstream (dn) in the sediment of the terraces for all the sampled SDs, including average value (the cross in the boxplot), median (the line in the box plot) and p-value. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP are shown.

Looking at the relative changes of P fractions rather than changes in mass indicated by the box plots, no clear trends were found when comparing the upstream and downstream sites. TP in SD5 was the only parameter that showed a noticeable decrease in both channel and terrace fractions, -62% and -43% respectively. TP in the channel of SD6 showed the largest increase of +227 % when comparing upstream from downstream, and the terrace in the same SD showed a small decrease of TP, close to zero. Overall, the TP has increased to different degrees in 11 out of 20 sites, decreased in 7 out of 20 sites, and 2 sites could not be compared due to lack of samples (Table 3).

Table 3. Net change of P fractions and TP between upstream and downstream in sediments of channel and terrace for each SD, both in mg P kg⁻¹ DW and relative net change in %. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-a. c. P (organic bound P), HCl-P (Ca P) and TP are shown.

Site	MQ-P mg kg ⁻¹ DW	MQ-P %	BD-P mg P kg ⁻¹ DW	BD-P %	NaOH- P mg P kg ⁻¹ DW	NaOH- P %	NaOH- a.c. P mg P kg ⁻¹ DW	NaOH- a.c. P %	HCl- P mg kg ⁻¹ DW	HCl-P %	TP mg P kg ⁻¹ DW	TP %
SD1C	+0.8	+19.4	-3.5	-13.3	+18.4	+58.8	+30.5	+47.6	-4.4	-13.2	+41.7	+26.4
SD1T	-0.1	-2.3	-13.6	-45.1	+10.5	+21.2	+11.9	+10.6	-19.1	-42.9	-10.4	-4.4
SD2C	-2.7	-33.7	-32.9	-51.6	-31.1	-35.3	-48.5	-29.9	-4.2	-12.3	-119.3	-33.6
SD2T	-2.5	-59.7	+7.3	+112.0	+24.9	+184.5	+39.6	+123.7	-0.3	-1.3	+69.0	+85.1
SD3C	+6.3	+354.0	+10.8	+101.0	+4.0	+51.9	+12.3	+125.1	+17.5	+41.9	+50.7	+70.9
SD3T	+3.8	+50.4	-19.2	-63.5	-4.4	-27.2	-14.3	-32.4	+7.1	+13.1	-27.0	-17.8
SD4C	-1.6	-47.8	-4.9	-50.5	-0.6	-6.7	+2.4	+8.0	-1.4	-3.2	-6.0	-6.4
SD4T	-0.7	-33.3	-0.4	-7.3	+3.4	+55.7	+21.7	+109.5	+5.4	+15.2	+29.4	+42.4
SD5C	+0.9	+17.4	-60.9	-93.5	-44.3	-78.5	-74.6	-79.9	+9.7	+19.4	-169.2	-62.7
SD5T	-1.5	-42.1	-5.4	-37.0	-12.4	-44.1	-15.5	-29.4	-24.4	-62.6	-59.2	-42.9
SD6C	+3.5	+71.4	+50.4	+158.7	+9.7	+76.2	+42.0	+112.0	+37.3	+147.3	+142.9	+127.3
SD6T	-0.3	-26.2	-4.6	-31.6	-2.3	-26.4	+6.6	+33.5	-0.3	-0.9	-0.9	-1.2
SD7C	+2.7	+46.9	+67.2	+126.2	+13.4	+64.4	+27.9	+55.2	+5.4	+17.4	+116.6	+72.2
SD7T	+0.4	+3.0	+34.2	+89.7	-9.9	-28.8	-13.9	-17.6	+5.1	+24.7	+16.0	+8.6
SD8C	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SD8T	-1.3	-25.9	-7.8	-29.1	-5.7	-56.2	-10.8	-41.0	+5.2	+12.6	-20.3	-18.5
SD9C	-0.6	-50.7	-3.6	-55.2	-1.1	-72.4	-2.2	-60.1	+15.0	+38.5	+7.5	+14.4
SD9T	-0.7	-60.1	+1.6	+41.5	+2.2	+100.4	+3.4	+30.7	+2.6	+12.2	+9.1	+23.2
SD10C	+4.3	+2290.9	+1.5	+100.4	+0.2	+3.5	+1.0	+14.3	+15.5	+68.2	+22.6	+60.3
SD10T	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

When comparing the P fractions between channel and the terraces, no significant trends were found but there were some patterns that showed that there was more P bound to the sediment in the channels rather than on the terraces, more noticeable for the HCl-P fraction (p-value 0.09), BD-P fraction (p-value of 0.16) and TP (p-value of 0.3) (Figure 14).

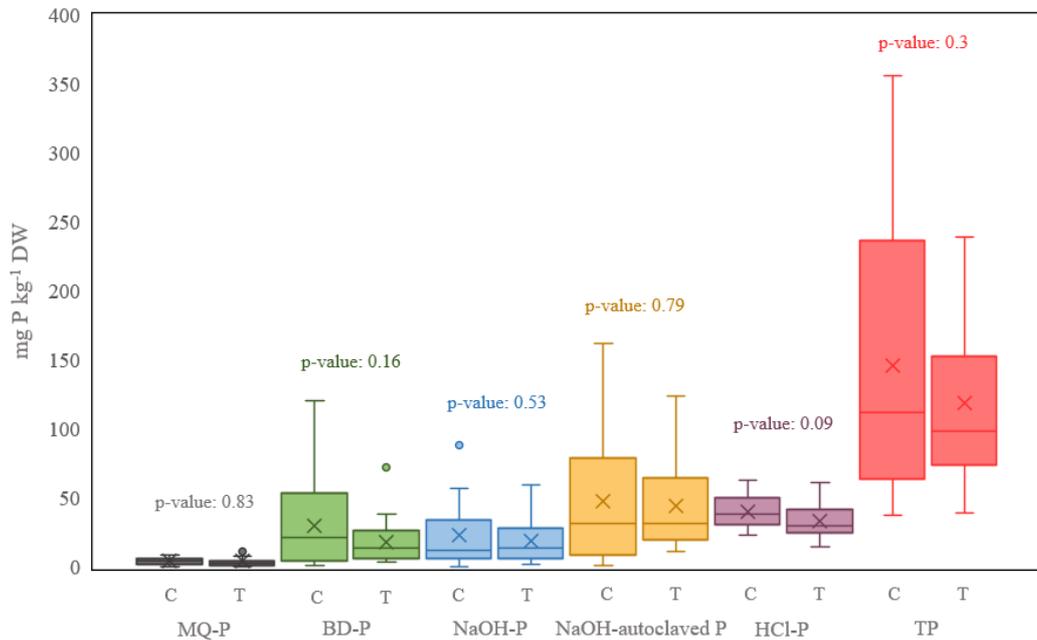


Figure 14. P fraction and TP box plots comparison of sediment between terraces (T) and channels (C) for all the sampled SDs, including average value (the cross in the boxplot), median (the line in the box plot) and p-value. The fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP are shown.

3.2 Phosphorus fractions and clay content

According to Eriksson et al. (2011) and Kindervater & Steinman (2019) the clay content affects the P fractions. The P fraction correlation results with clay content were different, dependent on regionality. When the values were put into correlation with all the sampling sites at the same time, there was no correlation. When separating the SD sites into two groups, sandy soils with <40% clay content (SD6-SD10) and clay soils with $\geq 40\%$ clay content (SD1-SD5), some patterns were shown.

There was poor correlation for the MQ-P fraction in SD1-5 with clay soil (R^2 value 0.14). SD6-10 with sandy soil showed a pattern of increasing MQ-P concentration with increasing clay content (R^2 value 0.41) (Figure 15).

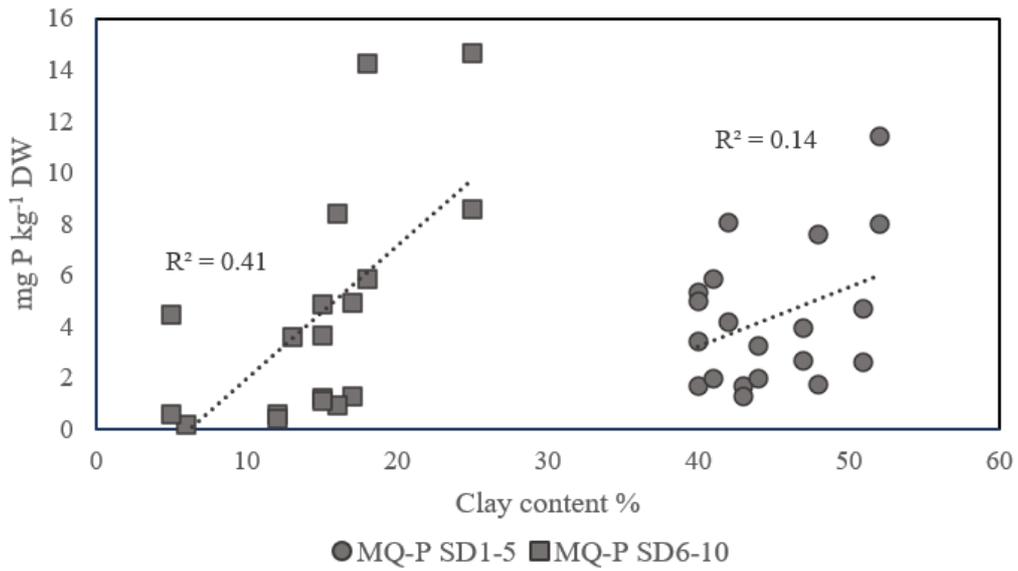


Figure 15. MQ-P (loosely bound P) fraction mg P kg⁻¹ DW plotted with corresponding SD1-5 and $\geq 40\%$ clay content (right), SD6-10 <40% clay content (left). R² value for respective group is shown.

The BD-P fraction in SD6-10 with sandy soil showed a pattern of increasing BD-P concentration with increasing clay content (R² 0.55). There was no correlation between BD-P fraction and clay content in SD1-5 with clay soils (Figure 16).

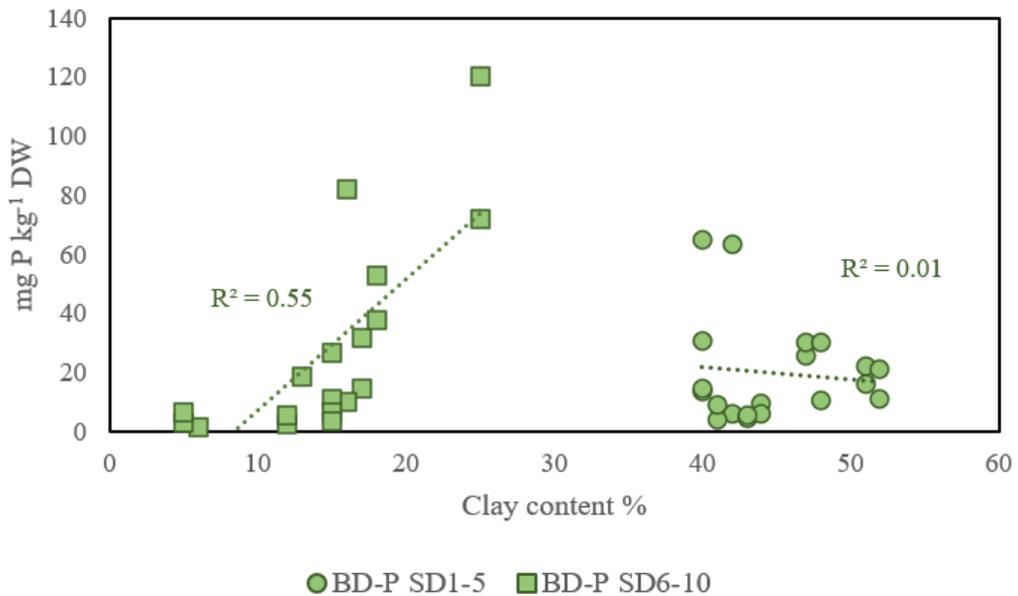


Figure 16. BD-P fraction (Fe P) mg P kg⁻¹ DW plotted with corresponding SD1-5 and $\geq 40\%$ clay content (right), SD6-10 <40% clay content (left). R² value for respective group is shown.

The NaOH-P fraction in SD6-10 with sandy soil showed slightly increasing pattern of NaOH-P concentration with increasing clay content. There was no correlation between NaOH-P fraction and clay content in SD1-5 with clay soil (Figure 17).

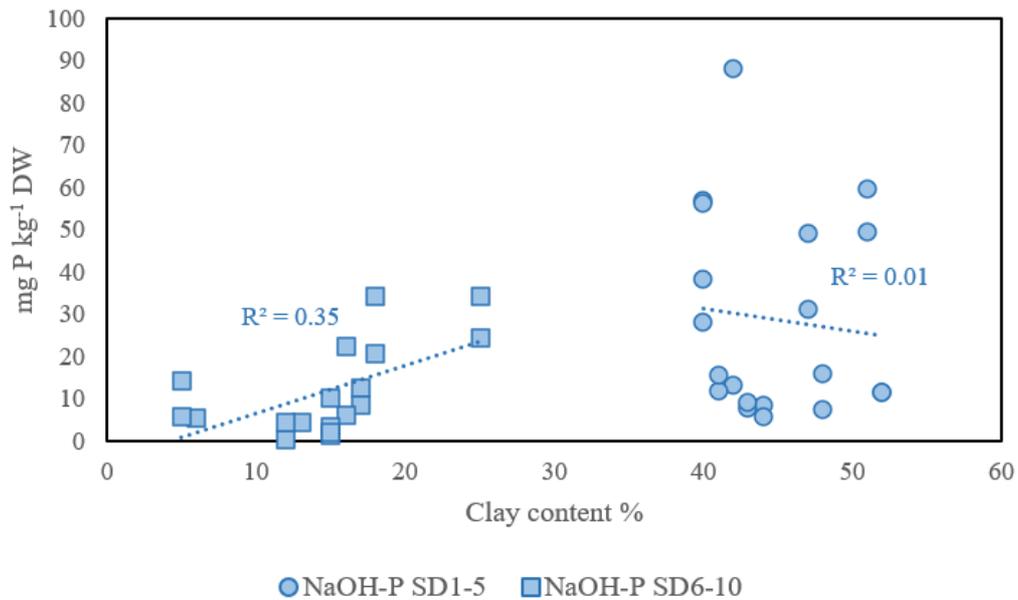


Figure 17. NaOH-P (Al P) fraction mg P kg⁻¹ DW plotted with corresponding SD1-5 and ≥40% clay content (right), SD6-10 <40% clay content (left). R2 value for respective group is shown.

No correlation was shown for the NaOH-autoclaved P fraction in SD1-5 with clay soil. The NaOH-P autoclaved fraction in SD6-10 with sandy soil showed an increasing pattern of NaOH-autoclaved P concentration with increasing clay content (Figure 18).

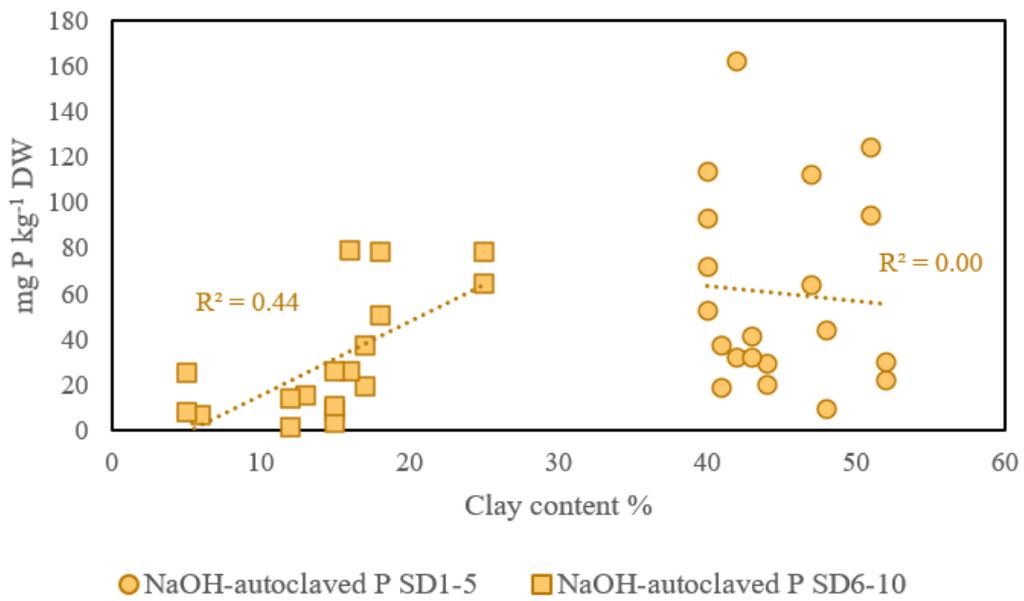


Figure 18. NaOH-autoclaved P (organic bound P) fraction mg P kg⁻¹ DW plotted with corresponding SD1-5 and ≥40% clay content (right), SD6-10 <40% clay content (left). R² value for respective group is shown.

No correlation was discernible for the HCl-P fraction, both in SD1-5 with clay soil and SD6-10 with sandy soil (Figure 19).

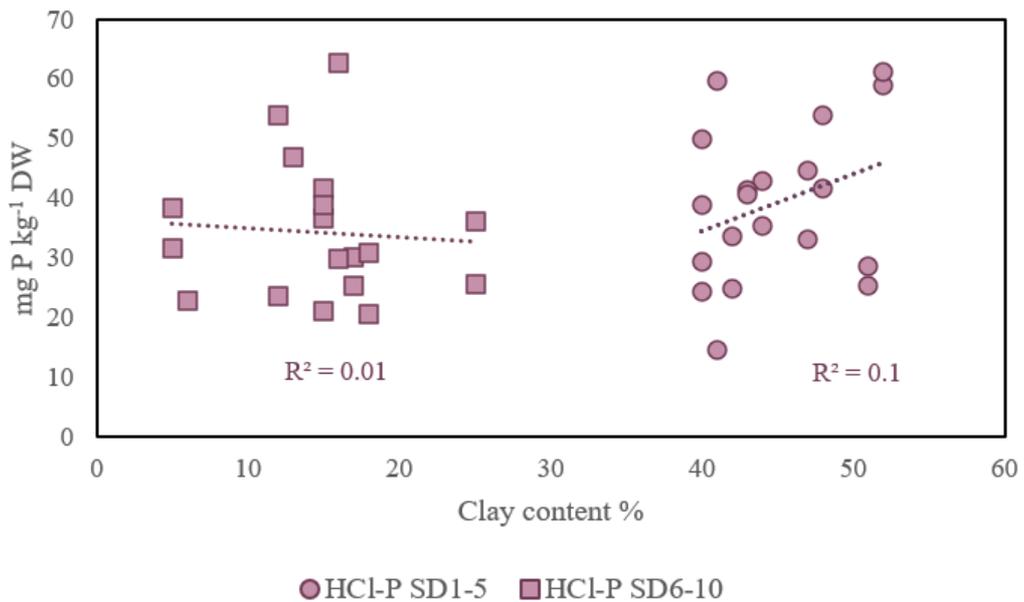


Figure 19. HCl-P (Ca P) fraction mg P kg⁻¹ DW plotted with corresponding SD1-5 and ≥40% clay content (right), SD6-10 <40% clay content (left). R² value for respective group is shown.

No correlation was shown for the TP in SD1-5 with clay soil. The TP in SD6-10 with sandy soil showed an increasing pattern of TP concentration with increasing clay content (Figure 20).

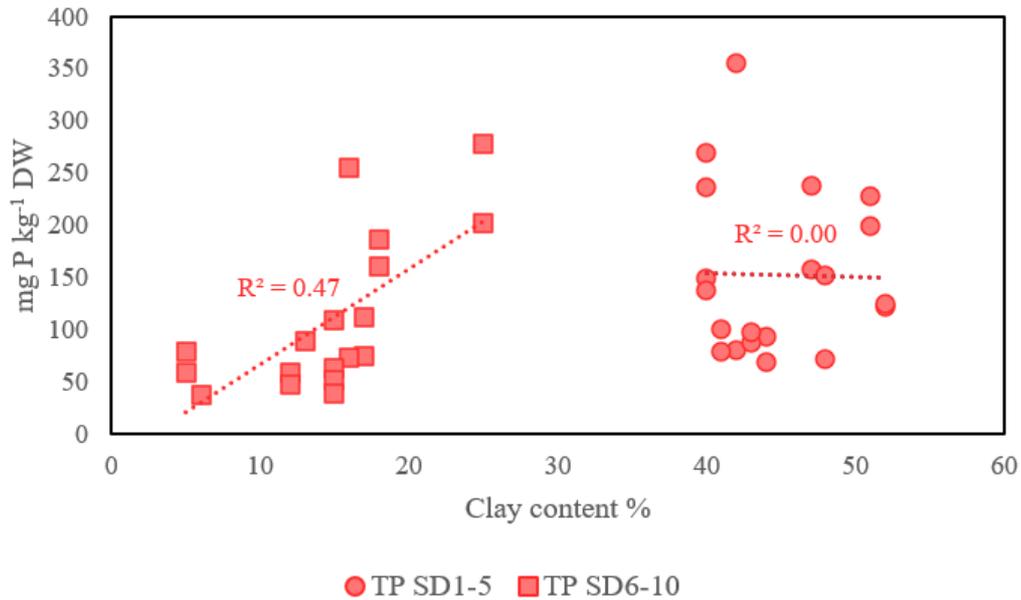


Figure 20. TP mg P kg⁻¹ DW plotted with corresponding SD1-5 and ≥40% clay content (right), SD6-10 <40% clay content (left). R² value for respective group is shown.

3.3 Phosphorus fractions and dissolved oxygen, pH and baseflow index

The P fractions were tested for correlations with pH, BFI (baseflow index) and DO (dissolved oxygen) to see if any relationships could be found. pH and DO are sampled from the channels at the same occasion as the sediment samples were taken. BFI is calculated from flow data between spring 2020 and spring 2021, each SD has one assigned BFI for upstream and downstream respectively.

Poor correlation was shown for all the P fractions and pH. Most of the pH values were distributed within the range of between 7-8 and most of them around 7.7 (Figure 21).

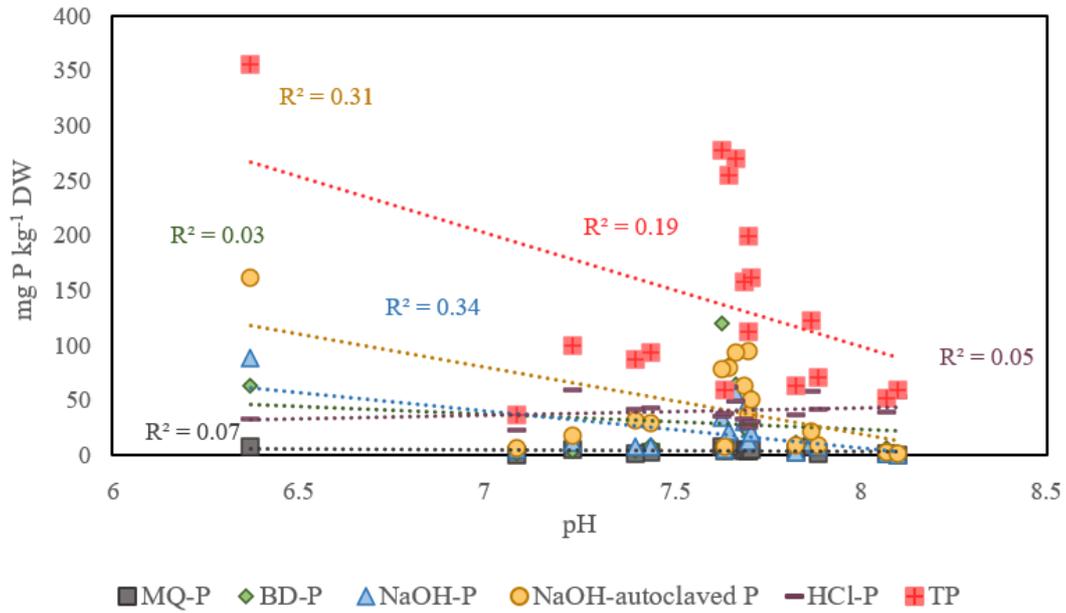


Figure 21. P fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP correlated with pH, including R² value.

No correlation was shown with the P fractions and the BFI. No R² value were higher than 0.06. The BFI varied from 0.55 to 0.95 and most of the BFI values were between 0.8 and 0.95 (Figure 22).

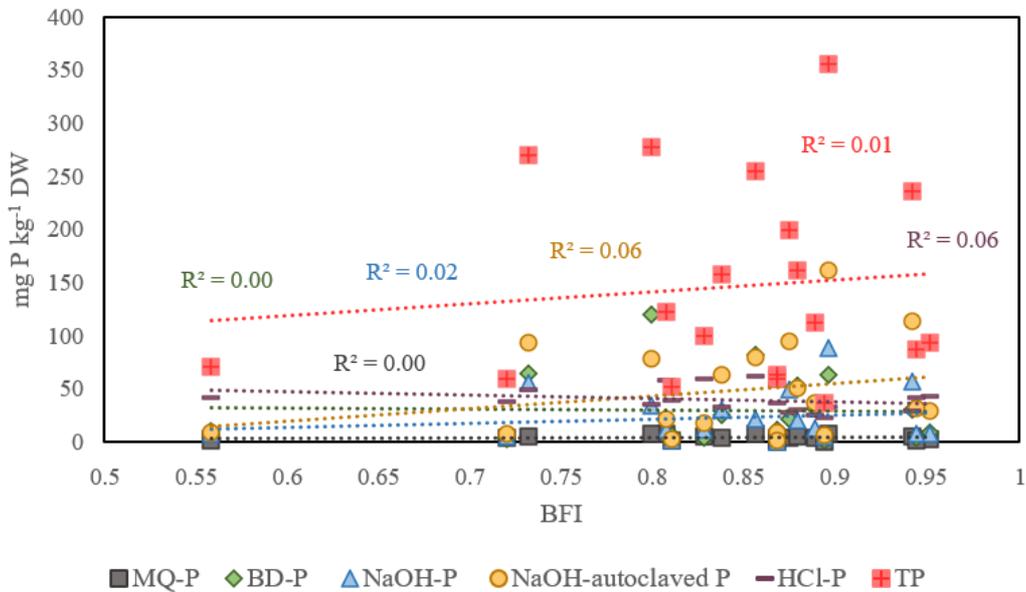


Figure 22. P fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP correlated with BFI (baseflow index), including R² value.

Poor correlations were shown for the P fractions and DO in the sampled sites. No R^2 values were higher than 0.21, which was the loosely bound MQ-P. The DO values varied between 11.3 and 13.7 (Figure 23).

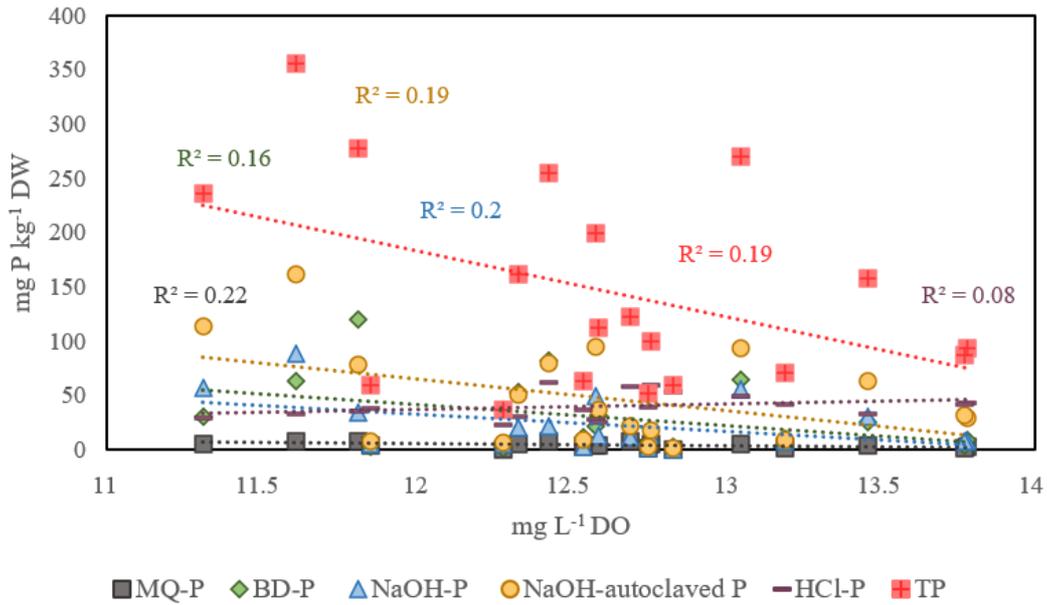


Figure 23. P fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP are correlated with DO (dissolved oxygen), including R^2 value.

3.4 Phosphorus fraction data and cross-referenced sorption data

Sorption and desorption data between sediments and water in SD2, SD3, SD7, SD8 were obtained to test the correlation with the P fractions from the same SDs. Only the sorption data from February was used since the P fraction data was from the same occasion.

Very poor correlation was shown for all the P fractions and EPC_0 except MQ-P that showed increasing EPC_0 with increasing MQ-P fraction. MQ-P varied from 1.7 to 11.4 $mg P kg^{-1}$ and EPC_0 from 0 to 100 $mg P L^{-1} g^{-1} DW$, although most EPC_0 values varied between 0 and 35 $mg P L^{-1} g^{-1} DW$ correlated with MQ-P (Figure 24 & Figure 25).

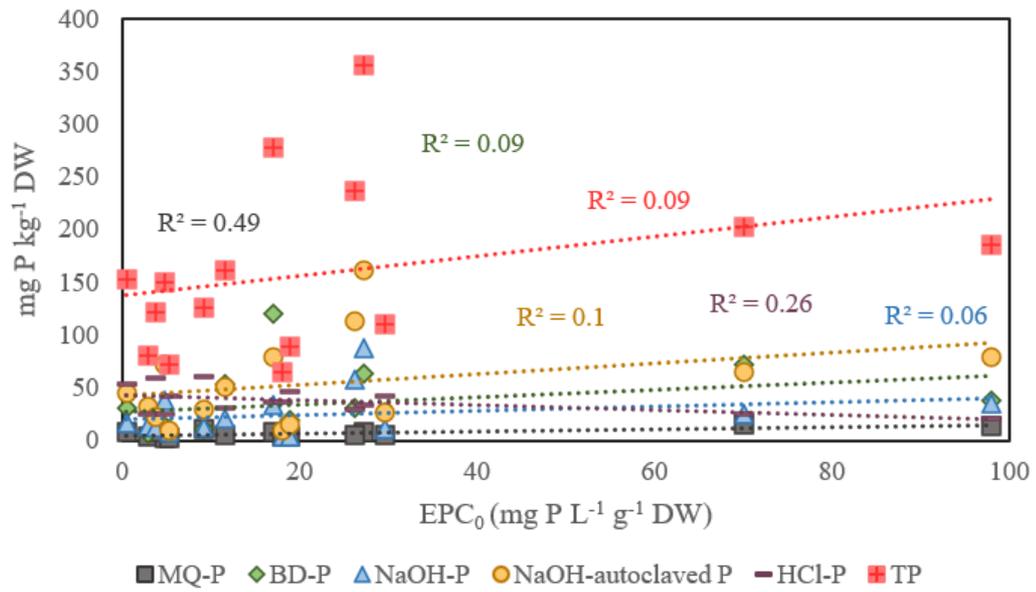


Figure 24. P fractions MQ-P (loosely bound P), BD-P (Fe-P), NaOH-P (Al-P), NaOH-autoclaved (organic bound P), HCl-P (Ca P) and TP correlated with EPC_0 at SD2, SD3, SD7, SD8, including R^2 value.

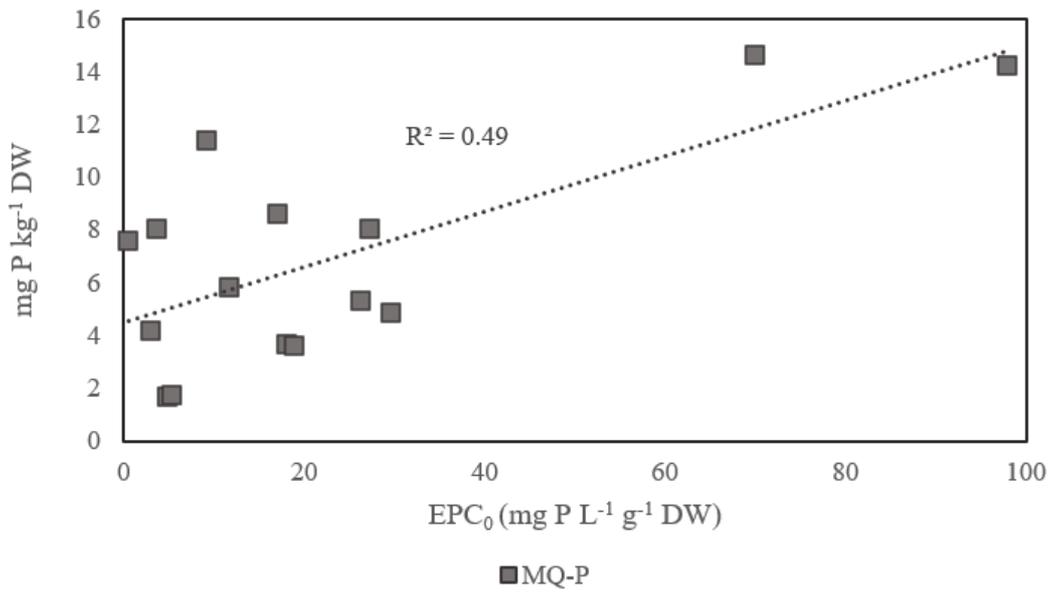


Figure 25. P fractions MQ-P (loosely bound P) correlated with EPC_0 at SD2, SD3, SD7, SD8, including R^2 value.

4. Discussion

No significant results for any of the P fractions or TP was found among the selected sites and no correlations with tested factor could explain the observed patterns. There were no significant changes within the two-stage ditches as well as no changes between the ditches. There were no clear correlations between any of the P fractions or TP with any of the tested factors. Nevertheless, there were some interesting patterns for some of the P fractions and tested factors.

The P fraction distribution patterns appear to align well with what previous studies of Eriksson et al. (2011), Kindervater & Steinman (2019) and Lannergård et al. (2020) showed. MQ-P is the smallest fraction in both mg P kg⁻¹ DW and fraction share of TP (%). NaOH-autoclaved P usually has a large variation in distribution (Figures 9 and 10), depending on organic material content in the samples (Eriksson et al. 2011). The high distribution or content of fractionation HCl-P (Figure 10) could be explained by the high HCl-P fractionation in SD8, SD9 and SD10 (Figure 11).

No significant P concentration changes for any P fraction were observed when comparing upstream and downstream samples (Figures 12 and 13). Although some patterns of increased HCl-P and MQ-P were shown from upstream to downstream in the channels. The reason for this is uncertain, possibly explained by features of the landscape e.g., local environment changes or mineralogy or higher SRP concentrations downstream. Gu et al. (2020) mentions that a chemical extraction method for determining P fractions can overestimate the amount of HCl-P due to redistribution of P fractions during the earlier fractionation extraction steps.

The general trend indicates that more of HCl-P, BD-P and TP resides in the sediments in the channels rather than in the terraces. This suggests that there is more P activity in the sediments of the channels than in the terraces (Figures 12, 13, 14).

There is no clear correlation between the extent of agricultural land use and TP levels in the SDs, which Lannergård et al. (2020) and Palmer-Felgate et al. (2009) suggest that there usually is. There are also high TP values in SD2C1, SD5C1, SD6C2, SD7C2 that stand out from the other sampled sites. This could be explained with various conditions in the catchment that have not been taken into consideration, i.e., any ponds or drainage pipe outlets that are close by where the samples have been taken that drastically changes the TP concentration in the sediments (Figure 11; Table 1).

Large P concentrations can be stored in streambed sediments (Lannergård et al. 2020; Kindervater & Steinman 2019). Patterns of the highest TP concentrations being held within the channel sediments are shown (Figure 11 and 14). This could mean that P bound to sediment particles accumulate in the stream bed in some of the two-stage ditches and act as a P sink, which could be considered to be one of the aims of a two-stage ditch (D'Ambrosio et al. 2015). However, Palmer-Felgate et al. (2009) emphasized the potential risks of having large amounts of P stored in streambed sediments. The sediment could release P or convert to other more or less bioavailable fractions when undergoing environmental changes e.g., redox status. The sediment could also travel downstream during weather events causing high flow and affect ecosystems further away from the initial P input, potentially causing eutrophication (Palmer-Felgate et al. 2009). With projected future climate change and more extreme weather events, fluctuating and higher intensity precipitation (Arheimer & Lindström 2015; Grusson et al. 2021), higher intensity flow and surface runoff from agricultural land could be expected (Grusson et al. 2021). This could affect stream turbidity and cause high P and sediment concentrations traveling downstream. Because of this, sediment as a phosphorus pathway could be of great importance when tackling eutrophication problems, now and in the future.

Reactions between soil particles and soil solution are affected by the particle size and the reactive mineral on the particle surfaces (Eriksson et al. 2011; Agudelo et al. 2010). No significant results were shown of that neither any P fraction nor TP increased with increasing clay content in the entire spectrum of clay contents analysed. Yet, there were some interesting patterns that indicates that the P fractions MQ-P, BD-P, NaOH-P, NaOH-autoclaved P and TP increase with increased clay content, between 5-25% clay content, and no correlation between any of the P fractions or TP and soils with clay content $\geq 40\%$ (Figure 15, 16, 17, 18 and 20). The reason for TP increasing with increased clay content could be that all P fractions except HCl-P increased, since the P fractions add up to TP. The soils with 5-25% clay content are SD6-SD10, which are located in southern Sweden and have a higher agricultural land use extent (Table 1), which could have a possible connection to the observed pattern. Although, there are only 19 samples in respective group, SD1-5 with clay soils and SD6-10 with sandy soils, and the same clay content is used for 4 samples at each SD. This means that there are actually only 5 samples per group, which in a statistical point of view might not give reliable results. According to Kindervater & Steinman (2019), the soils with highest Ca are usually sandy soils which can be seen in Figure 11, where variation patterns of TP are shown. In accordance with the statement of Kindervater & Steinman 2019, a larger fraction share of HCl-P is shown in SD8-10, which are the soils with highest sand content.

The concentration of soluble and plant available P has its optimum at pH 6-7. BD-P and NaOH-P absorption occurs at pH 3-5 and decreases rapidly at pH 7 or higher. At the same time, at higher pH, P tends to precipitate as rather insoluble HCl-P compounds (Eriksson et al. 2011) that are likely to become legacy P (Kindervater & Steinman 2019; Zhu et al. 2013). No trends were found between pH and any of the P fractions (Figure 21), although it could be interesting to investigate for any correlations if the pH distribution in the sampled SDs had been more varied and not distributed within the range of 7-8.

BFI is an index, in the range of 0-1, of base flow relative to total flow volume, which can be used as an indicator for measuring relative surface runoff or baseflow volume. High BFI means more groundwater flow and low BFI means more runoff flow (Gustard et al. 1992; Tesoriero et al. 2009). Nitrate concentrations of stream water can show correlations with BFI, whereas P concentrations usually do not, since P interacts with sediments to a wider extent than nitrate does. The P concentration that is in solution is usually very small and does not show any noticeable correlations with BFI either (Tesoriero et al. 2009). BFI did not show any patterns at all with any of the P fractions in this study, which was not unexpected since no literature or research papers suggested that any of the P fractions or TP would be affected by BFI (Figure 22).

The DO concentrations did not show any signs of affecting the dynamics of the P fractions. The DO values, varied between 11.3-14.8 mg P L⁻¹ (Figure 23). In a study by Zhu et al. (2013), sediments of a lake in China showed an increased dissolution of MQ-P and BD-P with decreased levels of DO in the water, the DO levels ranged from 0-10.5 mg L⁻¹ depending on season and depth of the lake. Rydin (2000) showed with anoxic release experiments in a Swedish lake that MQ-P and BD-P are strongly affected by redox status. It is possible that if DO levels in the sampled SDs had been lower and more widely distributed, a correlation could have been shown in Figure 23. Although, a lake is different from a stream in terms of flow conditions (Agudelo et al. 2010) and depth from surface to sediment. In other words, they might not be directly comparable.

The P fractions and EPC₀ showed an intermediate correlation, increasing EPC₀ with increasing MQ-P concentration (Figure 25), but not for any of the other P fractions (Figure 24). SRP (soluble reactive phosphorus) concentrations in the water column above EPC₀ suggest absorption to sediments, and SRP concentrations below EPC₀ suggest desorption (Agudelo et al. 2010; Simpson et al. 2021). This could indicate that with increased EPC₀, SRP absorbs to the MQ-P fraction. However, there were two EPC₀ values in the range of 70-100 mg P L⁻¹ g⁻¹ DW, that stood out from the rest, which were in the range of 0-30 mg P L⁻¹ g⁻¹ DW. The two values in the first mentioned range might make it look like there is more of a correlation than there actually is (Figure 25).

D'Ambrosio et al. (2015) suggested that two-stage ditches have potential to increase nutrient retention within the landscape. No significant results were found in this study to confirm that. Jordbruksverket (2016) clarifies the importance of taking properties of the catchment into consideration when constructing a two-stage ditch. The locations and properties of the 10 investigated two-stage ditches differed and there are factors that have not been taken into consideration in this study that could affect the P dynamics at the sampling sites. E.g., where the closest drainage pipes or dams are in the stream and the state of them, the current or past production and use of the land within the catchment, any other nutrient management strategies or legacy P.

P can be stored in sediments for many years, even decades, as legacy P. Estimations of how large these legacy pools are, and the time lags of P pathways are uncertain. Nevertheless, there is a great need for better understanding of how past and present agricultural practises (Lannergård et al. 2020; McCrackin et al. 2018) and regional variation (Withers et al. 2015) affect these P stores, and what P sources that contribute to both stable, and more mobile P legacy pools. This to further improve planning and deciding on management strategies for reducing P losses (Lannergård et al. 2020; McCrackin et al. 2018). It is uncertain how P fraction distribution and agricultural streams affect legacy P in water. There is a possibility that there is an improvement going on within catchments of this study, but due to legacy P and time lags, it might not show in this investigation.

5. Conclusions

No significant changes were found in P concentrations or fractions along the selected two-stage ditches. Although there were indications that larger concentrations of P reside in the channels compared to terraces. The HCl-P fractions seems to have larger concentrations in the channels than the terraces and accumulate to larger concentrations downstream in the channels. The concentrations of the P fractions MQ-P, BD-P, NaOH-P, NaOH-autoclaved P and TP appeared to increase with increased clay content, between 5-25% clay content.

None of the measured variables related to hydrological and biochemical processes significantly explained the observed patterns in P fractions. However, there are other factors that have not been investigated that could explain these results, e.g., location of dams, drainage outlets, mineralogy, agricultural management practices or legacy P.

This study implicates that more attention to streambed sediments is needed when tackling P and eutrophication issues. More research is needed to fully understand and evaluate the properties of P in stream sediments and two-stage ditches as a mitigation measure.

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Popular science summary

Sveriges miljömål ”Ingen övergödning” och ”Säkrad livsmedelsproduktion” syftar till att övergödningen i svenska vattendrag och Östersjön ska reduceras kraftigt samt att bibehålla eller öka en hållbar livsmedelsförsörjning. Det har de senaste åren skett förbättringar, kvävenivån har minskat något men fosfornivån i Östersjön är fortfarande kritisk och näringstaken överskrids fortfarande. Det finns flera möjliga åtaganden för att minska näringsläckage från jordbruk, bland annat bevuxna kantzoner, förbättrad dränering, dammar och tvåstegsdiken.

Denna studie syftar till att undersöka koncentrationen av olika fosforfraktioner och totalfosfor i tvåstegsdiken, om fraktionernas andelar förändras inom och mellan tvåstegsdikena samt jämföra fördelningen av fosforfraktioner med dikeskaraktär, hydrologiska och kemiska processer som kan vara påverkande faktorer. Sedimentprover från 10 tvåstegsdiken i mellersta och södra Sverige samlades och analyserades för bestämning av koncentration av fosforfraktioner och totalfosfor.

Resultaten visade inga signifikanta skillnader inom och mellan diken, varken för fosforfraktioner eller totalfosfor. Inga signifikanta resultat visades heller för korrelation av dikeskaraktär, hydrologiska och kemiska faktorerna. Det fanns dock tecken på att fosforkoncentrationerna var högre i sedimenten nedströms i diken än på dikenas terrasser.

Det finns flera möjliga förklaringar till de ojämna resultaten. Det har inte tagits hänsyn till faktorer såsom mineralogi, placering av dräneringsutlopp samt nuvarande eller tidigare förvaltning och drift av jordbruksmarken runt om tvåstegsdikena. I och med att fosfor binds in i markpartiklar så kan fosfor dröja kvar i systemet och bli långvariga fosforkällor som byggs upp i sediment, vilket under en lång tid framåt kan släppa ut fosfor i vattendrag. Sediment kan därför vara avgörande för att hantera övergödning av fosfor och för att uppnå hållbara lösningar på problemet.

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