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

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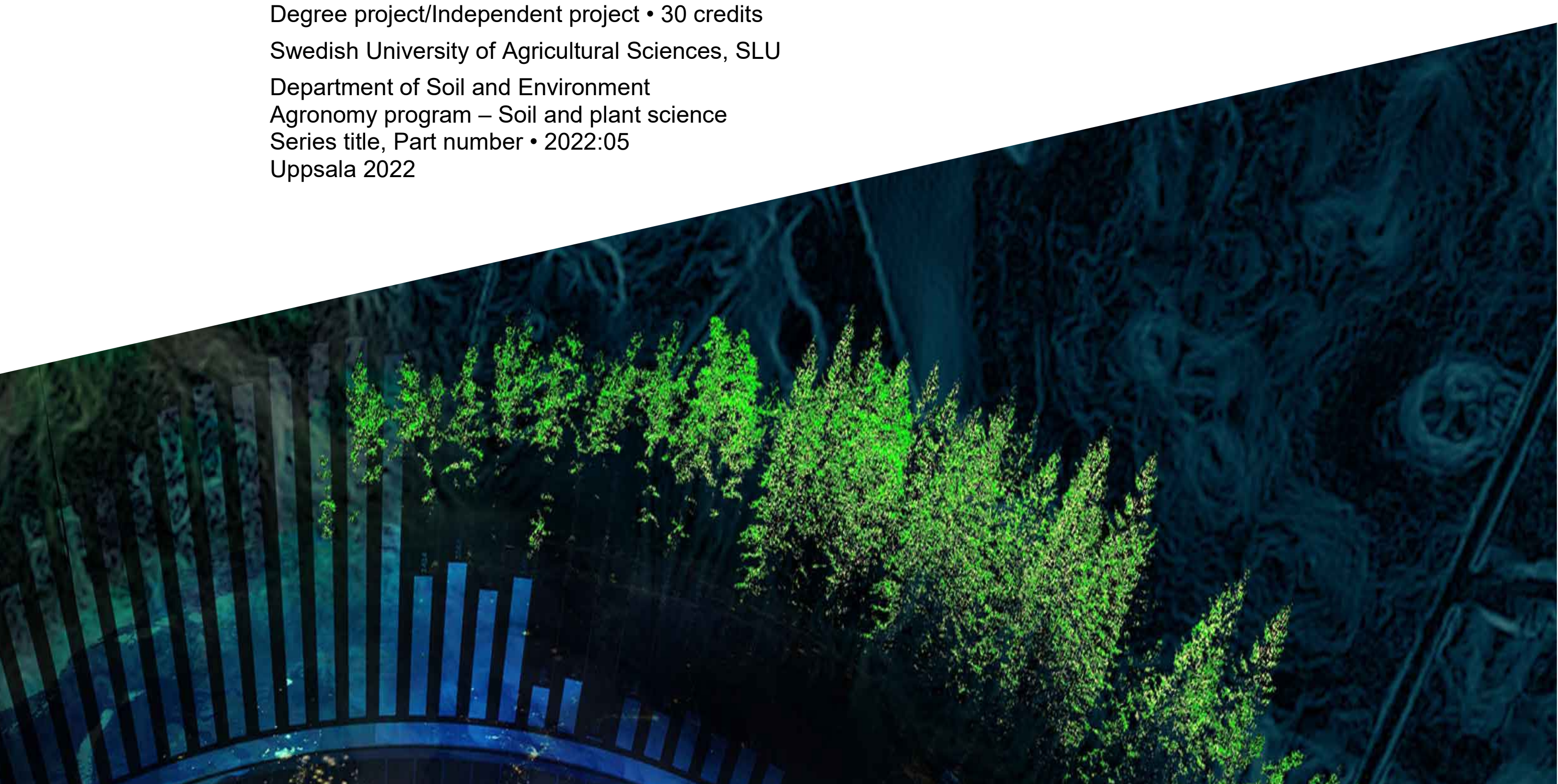

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Abstract

Human activities have increased nutrient losses to agricultural ditches and streams, including phosphorus. Phosphorus leakage leads to a worsening of water quality in downstream ecosystems, including lakes and seas. In addition, higher phosphorus concentrations lead to eutrophication, which negatively affects aquatic systems. Many land and water management solutions to reduce nutrient losses have been proposed. One of them is a two-stage ditch (SD) built from a traditional trapezoidal ditch (TD) by its widening and introducing floodplains. During high flow conditions, the floodplains will be flooded, which will, in turn, reduce the water velocity and reduce losses of phosphate downstream. The slower water velocity makes it possible for the sediment to settle and adsorb phosphate, potentially creating a sink for phosphate dissolved in stream water and improving water quality downstream.

In this study, SD's capacity for retention of phosphate was tested. The study was conducted in four SDs as part of a project *Two-stage ditches in Sweden.* Sediment samples were collected upstream/downstream floodplains and channels. The studied SDs are located in Sörmland, Östergötland and Skåne. The sediment samples were collected between February to April 2022. The collected sediment samples were analysed in a laboratory to determine the equilibrium phosphate concentration at net zero sorption (EPC_0) to measure the sediments buffering capacity to phosphate. The EPC_0 was correlated with the phosphate concentrations in the stream water to determine if the sediment adsorb ore desorb phosphate. The EPC_0 was later analysed to find conjunction with the stream water pH and DO (oxygen levels) and clay content referenced in the literature (Smolders et al. 2017; Bergström et al. 2015; Trentman 2020; Palmer-Felgate et al. 2011). Statistical analysis was made to find differences in EPC_0, retention capacity, and SRP concentration within the SD. The analysis was made between sites, stream location, and between the terraces and channel. The result pointed the EPC_0 value varies between the sites but with similar patterns, and the study show fluxes of phosphate in both ways from all SDs among the samples. The highest source of phosphate came from SD7 upstream with a release of 144 PO4-P ug/L/gDW by the channel in March 2022 and the highest retention by 82 PO4-P ug/L / gDW was observed in SD3 upstream in February 2022 on the terrace. SD2 showed the best result with more retention among the sample in both the channel and terraces and SD7 showed more phosphate release among the samples on channels and terraces. The statistical analysis showed no differences within the sites nor in SRP concentration within the SD. This could indicate drainage water attached to the SD leaching SRP. There was no conjunction that point pH, DO, and clay content influenced EPC_0 during this investigation.

Keywords: Phosphorus, Two-stage ditch, Equilibrium phosphate concentration, Phosphorus retention, Eutrophication, Soluble reactive phosphorus

List of content

List of Tables

List of Figures

Abbreviations

SD	Two-stage ditch
TD	Traditional trapezoidal ditch
P	Phosphorus
TP	Total phosphorus
SRP	Soluble reactive phosphorus
PP	Particle bound phosphorus
SS	Suspendered sediment
DW	Dry weight
FW	Fresh weight
EPC_0	Equilibrium phosphate concentration at net zero sorption
N	Nitrogen

1. Introduction

1.1 Background

Human activities have increased the negative impact on the environment by overexploiting natural resources and pollution. Changes in land use, excess use of mineral fertilizers, and livestock manures harm the water quality (Bieroza et al. 2019). An increased population rate, industrial processes, and the use of fertilizers are leading causes of eutrophication (de Jonge et al. 2002; Bieroza et al. 2019). The increased land management by agricultural activities has increased losses of phosphorus (P) and nitrogen (N) through leaching and erosion to many aquatic environments, including the Baltic Sea (Granstedt et al. 2008; Andersen et al. 2017). Higher concentrations of nutrients have affected the aquatic ecosystems by increased algal blooms (Andersen et al. 2017; Uusitalo et al. 2001; Malone & Newton 2020) Increased coverage of hypoxic zones within the Baltic Sea has negative consequences for many species (Murray et al. 2019), and P is the limiting nutrient for eutrophication in the Baltic Sea. It is estimated that 37 000 t/yr of the total P (TP) reaches the Baltic Sea from agricultural land use. It is the primary source contributing 40% of the total anthropogenic sources (Bergström et al. 2015). Biological, chemical, and physical properties like pH, soil structure, soil type, redox state, and mineralization strongly influence on P losses to watercourses (Trentman 2020; Simpson et al. 2021; King et al. 2015). Two-stage ditches (SDs) are designed to slow down the water flow and reduce nutrient and sediment losses to downstream ecosystems, but most of the studies were carried out in the US, and this is the first Swedish evaluation of SDs. The sediment and vegetation can retain the particle-bound phosphorus (PP) and soluble reactive phosphorus (SRP) on the floodplains and channel. Here, we evaluate if floodplains and channel of SDs can promote the sorption of SRP to the metal oxides attached to sediment particles. The objective of this study was to measure the adsorption capacity of SRP to the sediments by measuring equilibrium phosphate concentration at net zero sorption (EPC_0) (Simpson et al. 2021). Sediments can act as a sink or source of P to the water column depending on the EPC_0 value due to the stream water phosphate concentration. The sampling was conducted once a month, February to April, in ten SDs in Sweden and covered a wide range of geographical and land use properties.

1.3 Aim of the study

The study investigated the SDs' phosphate sorption capacity in the sediment of the floodplains and channels. The main object was to determine the equilibrium phosphate concentration at net zero sorption (EPC_0) and how it develops over time. Performance in reducing or increasing phosphate concentrations was evaluated by measuring both upstream and downstream of the SDs. Furthermore, it was tested if pH, dissolved oxygen (DO), and particle sizes in the sediment and stream water explained any patterns observed in the EPC_0 data. Specific questions included:

1. How does EPC_0 change over time (from one month to another) and space (between SDs and within SD between floodplains and channel) in SDs and are these patterns statistically significant?

2. Are SDs sinks or sources of SRP?

3. Can observed patterns in EPC_0 be explained by water quality, pH, DO, and particle sizes?

2. Literature review

2.1 Artificial drainage in Sweden

Tile drainage reduces surface water flows in agricultural fields, decreases water stress, and increases crop resilience to pests and diseases (Mattsson et al. 2018; Wesström et al. 2017). As a result, tile drainage creates a more stable growing environment for the crops. Tile draining also provides increased control through earlier planting seasons, less compaction, increased trafficability, and a wider crop choice, increasing the farmers' long-term benefits (King et al. 2015). The first drainage in Sweden was introduced in the Middle Ages, and the technique came from the continent with traveling monks (Larsson et al. 2013). In the 1500s, when Gustav Vasa was the king, there were appointments to maintain the drainage water flows (Larsson et al. 2013). The first clear directive of drainage came in 1889 and described the rights and obligations to drainage by leasing other people's land. This increased crop production to feed a growing population and handle the emigration problem in the 1800s. As a result, there was an expansion of drainage in Swedish soils in the late 1800s and early 1900s. During the 1800s, the Swedish cropland increased four times to supply the growing population. It was because of lowering the groundwater levels by drainage in farmland that was otherwise too wet to cultivate. From the beginning to the middle of the 1800s, the drainage included mainly open ditches and channels. In the middle of the 1800s, Sweden started to use subsurface tube pipes. Tube pipes in tile were more labor-intensive and expensive but removed the maintenance burden. In 1879 a new law was established, making it possible to start local drainage compounds and establish large-scale system drainage (Wesström et al. 2017). Between the 1800s to 1960s, the arable land increased further from 1.5 to 3.7 million hectares. After the 2nd World War, agriculture started to be more specialized and mechanized, with higher use of mineral fertilizers and better structure among the farmers, increasing cropping intensity and production (Wesström et al. 2017). Today almost 50% of the arable land in Sweden is systematically tile drained (Bergström et al. 2015).

2.2 Legislation and mitigation of water pollution

There are several programs and legislation aimed at reducing eutrophication and water pollution. The European Commission implemented the Nitrates Directive in 1991 to reduce nitrate pollution from agriculture. The Water Framework Directive was introduced in 2001 to improve European waters' chemical and ecological status. More locally, HELCOM (Helsinki Commission) works for a healthier Baltic Sea environment in different programs (Granstedt et al. 2008). The mission is to reduce the excessive levels of N and P from agricultural land use, which leads to eutrophication. The goal was to reduce the nitrogen losses by half ere agreed in 1986 at the North Sea conference in Paris and was first introduced in 1987 to 1995. Achievement of the goal failed, but the improvement has been seen. In 2014 the Baltic Sea was estimated to receive 114 600 tN and 3340 tP from anthropogenic sources in Sweden (Jordbruksverket 2020). Anthropogenic N (42% gross load) and P (35% gross load) come from agriculture activities which include both livestock and fertilizer usage (Jordbruksverket 2020). The period from 1950 to 1980 showed an increase in pollution rate from 36 to 80 kg N/ ha year and for P an increase from 7 to 17 kg P/ ha year but was later reduced to 8 kg P/ ha year in 1990 (Granstedt et al. 2008). Nitrates Directive, WFD, and HELCOM programs have been introduced to achieve better water quality. Common Agricultural Policy (CAP) is the common agricultural policy in the EU and is implemented to give a better environment throw financial implications and take measures by the farmers. CAP finances several mitigation measures and programs like WFD to reduce nitrate and phosphorus eutrophication (Bång et al. 2018). Several mitigation measures have been implemented in Sweden to reduce nutrient and sediment losses to the watercourses. Some implemented measures include cover crops, liming, precisions farming, buffer zones, lime filter drainage, and wetlands (Jordbruksverket 2013; Bång et al. 2018). In addition, in Sweden, there is not either allowed to spread more than 22 kg/ha P manure on the fields and not fertilize on frozen, flooded ore snow covered fields (Ulén & Jackobsson 2005). Many of these measures are implemented with some help from Greppa Näringen, a free counseling service that works to reduce the agricultural impact on eutrophication by farmers (Jordbruksverket 2013).

2.3 Two-stage ditches

Introducing the first two-stage ditch (SD) was made in Wood Country, Ohio, in 2002. A scientist designed the SDs to reduce problems in trapezoidal ditches (TD), such as erosion and nutrient losses (Ranjan & Witter 2020). The design was suggested to have better ditch stability, reduced bank erosion, support, and flooding of adjacent fields (Ranjan & Witter 2020; Larsson 2016). The first SD in Sweden was constructed in 2012 outside Nyköping in Sörmland by Anders Herlitz on Åkra farm (SMHI 2014). Since Anders Herlitz built the first SD in Sweden, several more have been made to reduce nutrient transport to the Baltic Sea. The shape of an SD can vary due to soil type (Jordbruksverket 2016). The slopes are constructed with different angles to prevent erosion on both sides of the ditch. The second step down to the ditch is the terrace which works as a floodplain. The first stage is the small channel between the terraces filled during baseflow (figure 1). During storm flows, the water levels will increase and fill the floodplains (figure 1). The design aims to slow down the water flow and reduce sediment losses from the bottom of the ditch (Jordbruksverket 2016). The vegetation on the terraces and slopes helps reduce erosion during storm flows which helps to prevent suspended P from leaching downstream and enables soluble P to be taken up by the vegetation and benefit denitrification (Hodaj et al. 2017; Jordbruksverket 2016). The SD design has a higher capacity to remove water from the landscape, reduce flooding on the nearby fields, and reduce nutrient losses of P and N (Jordbruksverket 2013; Nilsson & Johnasson 2015). In addition, the moist floodplains can create new habitats and increase biodiversity. An SD is more expensive than a traditional trapezoidal ditch to construct the floodplains, giving arable land losses. The losses will be 0,5 to 1 ha/km of land by converting a TD to an SD (Jordbruksverket 2016). Nevertheless, if the land were vegetative buffer zones adjacent to the ditch, no losses would be necessary (Christopher et al. 2017). However, buffer zones, unlike SDs do not capture nutrient losses from tile drains. An SDs increased water flow capacity could also reduce floods downstream with a 25-100% higher flow capacity (Ranjan & Witter 2020). The cost for an SD varies in terms of land losses used to produce crops and construction costs. Data from 2014 points out that the cost varies from soil class 10 in Skåne costs 360 000 kr/ha to 220 000 kr/ha from the soil with class 6. The cost of transforming a TD to an SD depends on the depth width and amount of shaft masses needed to be removed. Jordbruksverket point that the coast would be around 250-550 kr/m with a SD with 2m depth (Nilsson & Johnasson 2015).

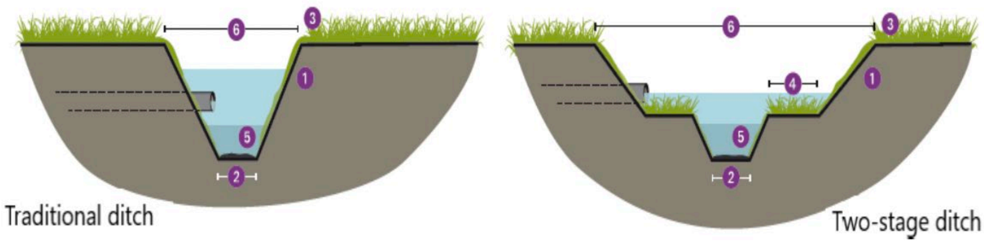

Figure 1. Cross section of a traditional trapezoidal (TD) dich to the left and a Two-stage ditch (SD) to the right. 1, Side banks 2, bottom bed 3, bank top 4, terraces 5, base flow 6, high flow (Jordbruksverket 2016).

2.4 Phosphorus losses in agricultural soils

There are mainly three ways for the nutrients from soils to enter watercourses: nutrient leaching, erosion of nutrients bound to particles, and surface runoff of soluble nutrients. Erosion is relatively not a big problem in Sweden compared with other countries (Eriksson et al. 2011). The erosion that appears is a problem for different water environments. Especially nutrient-bound particles like phosphors attach to clay minerals transported to watercourses. Most of the P is leakage as phosphates bound to other particles. The rest (20-40%) is leaking as soluble phosphates (Eriksson et al. 2011). The 90% of the P losses come from the 10% arable land in 1% of the time, which points to the control measures being vital in both space and time (Eriksson et al. 2011). The P losses from a Swedish agricultural field vary between 0,03-1,5 kg/ha/year (Börling 2010).

2.4.1 Surface runoff and transport

Surface runoff of soluble nutrients is a problem in frozen soils or steep topography. It is especially soil erosion by surface water runoff, contributing to suspendered sediment (SS) and particle-bound phosphorus (PP) losses (King et al. 2015; Djodjic & Markensten 2019). This is a problem with soils within soils with clay and silt with particle size (0,02-0,0002 mm), which is sensitive to erosion (Börling 2010; Eriksson et al. 2011). Surface runoff occurs when the water intensity by precipitations is higher than the macropores, soil cracks, and fissures permeability. (King et al. 2015; Simard et al. 2000; Djodjic et al. 1999) Most of the P losses are associated with surface runoff because of the adsorbing of P to fine sediment and erosion from sensitive soils. The subsurface pathway can be strengthened by artificial drainage and reduce surface runoffs in agricultural fields (Simard et al. 2000). Tile drainage water has less TP than surface pathways after being filtered by the matrix in the subsoil (Simard et al. 2000). Therefore, placing the tile pipes deeper gives P a higher chance of adsorbing by the matrix in the subsoil (King et al. 2015).

2.4.2 Nutrient leaching by transport and soil texture

Nutrient leaching of soluble nutrients arises when water flows through the soil profile. The tile drainage later transports the water, including dissolved nutrients, from the surface to groundwater and finally to watercourses. Leaching will reduce nutrients in the soil profile and reduce base saturation degree (Eriksson et al. 2011). The subsurface pathways involve water transported through the matrix or preferential flow in the subsoil (Djodjic et al. 1999). The matrix pathway is the slow flow in the subsoil and therefore does not contribute to significant P losses and plays a vital role in filtering P in the water. SRP adsorbs to organic matter and clay minerals as metal oxides and reduces P leaching to the drainage water. Finer minerals have a higher absorbing area. This filtering process is regulated by the proportion of matrix and preferential flow (Simard et al. 2000). The preferential flow is the fastest flow through the subsoil, decreasing the possibility of P being absorbed into organic matter and clay minerals in high flow. Subsurface water P-pathways by preferential flow are influenced by two categories of processes, abiotic and biotic. Abiotic processes influence cracks, fissures, and biotics as bio-pores interference by root and earthworms (King et al. 2015). Abiotic processes are formed by the natural desiccation process and are linked to finer textured clay soils. Biotic processes form bio pores by root channels, and earthworms borrows have higher SRP losses than the soil of disturbed macropores (Djodjic et al. 1999).
However, this flow increases the leaching of P to the environment. This flow with disturbed macropores is more common in soils with bad structures with more surface runoff ore courser texture with less absorbing affinity (King et al. 2015; Djodjic et al. 1999). Abiotic and biotic processes have an essential role in P movement from the surface to the drainage. P transport follows the water pathways through the subsurface to the tile drainage by macropores and cracks, the most common pathway by preferential pathways (Simard et al. 2000). The high

infiltration rate has a high-water infiltration capacity from the surface to the tile drainage, increasing losses of P in storm flows (Simard et al. 2000; Djodjic et al. 1999). Soil characteristics with fine texture soils had higher losses of P in the drainage waters. Clay loam soils had higher losses of P in drainage waters than sandy loam and found that the PP losses were 80% in the clay soil and 20% in the sandy soil. In addition, a fine texture with large macropores and cracks, has 70 times higher risk of P-losses to tile waters of clay soils than sandy soil (King et al. 2015). Greater losses from fine texture are attributed to preferential flow despite the higher adsorbing capacity from finer minerals in the fine texture soil. The preferential flow in fine texture has an important role in subsurface P losses (King et al. 2015).

2.5 pH

Physical, chemical biological characteristics regulate the P leaching by mineral composition, structure, redox state, and pH (Eriksson et al. 2011; Djodjic et al. 1999). pH has an essential role in P chemical processes in the soil and sediment. pH is strongly related to P state and form and is the measure of hydrogen ions in the soil solution affecting the acidity (Eriksson et al. 2011). In pH 7.2, there is an equilibrium between $H_2PO_4^-$ and HPO_4^{2-} in the soil solution (Eriksson et al. 2011). The soil's charge balances negative anions and positive cations, neutralizing the soil. These ions will be attached to the particle surfaces on various bound strengths. The replaceable ions are placed in the water solution and placed around the particle instead of directly attached to the particles as surface ions. A complex in chemistry is a composite ion with the same compound of molecules when the complex is formed. A complex cannot change oxidations number because the ions will be reacting to each other and create new molecules.

2.5.1 High pH

At higher pH, $H_2PO_4^-$ has less affinity to absorb minerals and organic material. The acids anions like $H_2PO_4^-$ have a less absorbing affinity to higher pH because they $H_2PO_4^-$ easily form complex H^+, Fe^{3+}, and Al^{3+} (Eriksson et al. 2011). P acid is not a strong acid as nitric acid and has that solid adsorbing affinity to form surface complex compared with nitrate ion, anion to nitric acid and has low affinity to form a surface complex to H^+, Fe^{3+}, and Al^{3+} (Eriksson et al. 2011). At higher pH, ligand sorption to anions reduces after the particle surface positive charge decreases. Higher pH will increase the negative phosphate charge, and the surface solution particles are less positive, affecting the absorbing potential of phosphate to particles (Eriksson et al. 2011). Phosphate binds to Ca^{2+} in higher pH and gets very insoluble in soils with pH eight or higher. Al, Fe bounds to OH groups in higher pH after a more negative charge surface on the metal oxide. Liming of CaO ore $Ca(OH)_2$ are used to give a better structure in clay soils. Liming makes P more available for vegetation uptake and is most soluble in pH 6-6,5 (Eriksson et al. 2011).

2.5.2 Low pH

Soils with low pH have high phosphate adsorbing capacity, decreasing phosphate leaching. The phosphate binds mainly to humus, Fe, Al oxides as a complex inner, which gives a strong bond at low pH. They are directly bound to the particle surface with no bridge to water but share an oxygen ion with the metal oxide. This process by a phosphate replacing an OH_2 group is called ligand change (Eriksson et al. 2011). In pH lower than 5.5, the phosphate cerate hard soluble precipitates with Al^{3+} and are common in the solution in acid soils. Al^{3+} gets more soluble in low pH. and reacts with phosphate and creates hard soluble precipitates (Eriksson et al. 2011).

2.6 Redox change

Redox conditions and P adsorption capacity in the subsoil affect the P concentrations in preferential and matrix water flow (King et al. 2015). A high-water table changes the redox conditions in the subsoil and increases the SRP concentrations in the filled pore space. Change in redox conditions gives a higher concentration of SRP in tile drainage waters. Reducing conditions in the drainage pipes increases SRP compared with oxidizing conditions for free drainage (King et al. 2015). In oxic conditions, iron is soluble as Fe^{3+} and has a high affinity to phosphate, making the phosphate less soluble. In anoxic conditions, iron goes from Fe^{3+} to Fe^{2+} and phosphate releases both in the soil and sediment (Smolders et al. 2017; Trentman 2020).

2.7 Biological processes

Biological processes regulate the release and uptake of phosphate in the soil solution. A high amount of P can be stored as organic material and need to mineralize to be available as phosphate, and 50% of the total organic material is organic P (Eriksson et al. 2011). The microbes mineralize a few percent of the organic P to phosphate every year. In addition, some phosphates are being taken up by the microorganism during the immobilization process (Eriksson et al. 2011). Vegetation can take up much phosphate in the floodplains and fields. P is most soluble in pH 6-7 and forms no hard soluble precipitate with Al, Fe Ca, which is able for vegetation uptake and leaching. This increases the risk of water transport of soluble P to the tile drainage (Eriksson et al. 2011; Trentman 2020) .

2.8 Soil structure

Soil structure is the composite of soil particles of minerals, humus, and the void space called pores (Rabot et al. 2018; Eriksson et al. 2011). The structure is influenced by the soil texture, mineral composition, humus content and biological activity has a big influence on structure. The colloids are fine clay and humus particles and are often charged and attached to Fe, and Al-oxides (Eriksson et al.

2011). Humus is important to have a stable structure and secondary precipitate of oxide and hydroxides such as Al, Fe, and Mn (Eriksson et al. 2011). If a high colloid content around the particles is large enough, the aggregate's structure will be formed as a secondary structure in different sizes. This is because the fine minerals are easier to form aggregate structure, and Ca^{2+} ions (liming) make it easier to make the collides charge easier to attach. In soils with aggregates like clay soils, the water infiltration is more complex in different pore sizes. Large pores or cracks called macropores affect the permeability of the soil. Both abiotic processes like freezing and biotic processes from worms and roots positively influence soil structure and P form and concentration. Soil stricture influences water retention, infiltration, gas exchange, soil organic matter, nutrient dynamics, root penetration, and erosion susceptibility (Rabot et al. 2018; Eriksson et al. 2011). Soils also have a damaged structure by compaction, less water infiltration, gas exchange, increasing water runoff, and soil erosion (Romero-Ruiz et al. 2018). Soil compaction has a negative influence on P by increased surface runoffs. For the last 40 years, Swedish soils have had the worst permeability in clay soils due to heavy machinery that destroys the soil structure and results in higher losses of P by less infiltration capacity and more surface runoff (Wesström et al. 2017). Applying Ca^{2+} ions by structural liming, organic material, and secondary precipitate of oxides and hydroxides (Al, Fe, Mn) improves the stability of the aggregates. (Bergström et al. 2015; Eriksson et al. 2011). As a result, the aggregates get less sensitive to degradation by rain and snow melting, leading to losses of PP and SS to tile drainage or surface runoff (Etana & Rydberg 2006; Eriksson et al. 2011; Ulén & Jackobsson 2005) . Structural liming reduces this process by the clay minerals attaching to each other and reducing PP losses (Ulén & Jackobsson 2005). This also increases the infiltration capacity with small cracks and better P filtering by sorption from the soil (Blomqvist & Berglund 2015).

2.9 Cropping system influence on P leaching

The cropping system has a different impact on P losses. Conservation tillage is a promoted way of healthy soil with minimal soil disturbance and prevising crop residues. This implication reduces soil erosion of SS and PP. No-till is also set to have more subsurface of P by preferential flow throw cracks, fissures, and macropores. The P-transport in the sub-flow is greater in conservation tillage than conservation tillage (King et al. 2015; Renwick et al. 2018). The preferential flow, together with P application on the surface, is one of the main reasons for the higher losses in no-till, and the result shows 11 times higher P losses of SRP than plowing and specially by implication of no-till for a longer time (King et al. 2015; Renwick et al. 2018). This gives more P concentration on the surface layer after the soil gets fertilized witch increases the SRP concentration in surface runoff by the P gets dissolved at the surface soil layer rather than incorporated with the soil by tillage (Renwick et al. 2018). Soils with fine texture and various temperatures by drying and freezing form cracks. Several studies point out that tillage has a low impact on P-losses by subsurface pathways and cropping systems can reduce P losses but not the P transport in the subsoils. (King et al. 2015). Heavy machinery and tillage tend to destroy the soil structure with worse aggregate stability and dispersion of clay minerals in heavy rains. This leads to losses of SRP but mostly of PP (Etana & Rydberg 2006). The study by Etana & Rydberg (2006) shows total phosphorus (TP) in drainage waters with systems with plowing compared with no-till and with some more SRP with plowing. The plow pan is the soil under the topsoil caused by the tractor wheel. Heavy machinery causes smaller aggregates and smaller pores in the subsoil than the topsoil, which gives worse structure, and the water infiltration decreases from the soil layer (Eriksson et al. 2011). This leads to higher losses of PP and SS when the structure with aggregates gets depolymerized (Etana & Rydberg 2006). Different fertilizers affect leaching to tile drainage, especially organic P, which tends to be transported deeper in the subsoil and has higher concentrations than inorganic fertilizers. That is because that organic manure sorbs less strong the inorganic fertilizers on their way down in the subsoil, which gives a higher P concentration in drainage waters (King et al. 2015). The amount of fertilizing during the season and the precipitation after P fertilization determine the amount of P lost to drainage water. Preferential flow from the surface inlets has more considerable risks of P losses in high water intensity (King et al. 2015).

2.10 Phosphorus in sediments

SDs reduce the P concentrations downstream in baseflow and stormflow (Hodaj et al. 2017). Most of the P that comes from the water column will likely be stored within the sediment and soil by sedimentation of PP and chemical precipitation, sorption of SRP, and deposition of suspended organic matter (Palmer-Felgate et al. 2011). The construction makes it possible for the flooded floodplains to slow down the water velocity during high flows and favor the sedimentation of SRP and PP (Hodaj et al. 2017). Binding nutrients to sediments can reduce eutrophication. The SRP binds to metal oxides and retains P. It is dependent on the design of floodplains, management, and composition to bind $H_2PO_4^-$ by sorption by the surface of Fe and Al oxides in clay sediment (Bergström et al. 2015; Trentman 2020; Palmer-Felgate et al. 2011). The binding of $H_2PO_4^-$ occurs in the sediment by the floodplains or terraces on the SDs. Humic soil organic material forms stable dissolved complexes with Fe^{2+} and Fe^{3+} and inhibits precipitation. The organic material acts as a coupling ion for P sorption despite changing P sorption (Trentman 2020; Palmer-Felgate et al. 2011). The sediment has the potential to leach $H_2PO_4^-$ through physical, geochemical, and microbial processes. Benthic bacteria can also concert the fluxes of $H_2PO_4^-$ from the sediment to the water column by mineralization (Smolders et al. 2017; Palmer-Felgate et al. 2011). PP can be released from the sediment by hydraulic resuspension, and $H_2PO_4^-$ can be released back into the water in a change of redox status during anaerobic conditions (Palmer-Felgate et al. 2011). The water levels are generally lower in the summer, making the floodplains more exposed to aerobic conditions. During the summer periods with baseflow, the floodplains get exposed to oxidation in the sediment, leading to less P sorption affinity. This condition increases the labile and reductant-soluble forms of bound P in the sediment (Kindervater & Steinman 2019). After a long dry period and after the first storm events in the autumn, the floodplains could be releasing the mobile bound P to the water column (Kindervater & Steinman 2019).

2.11 EPC_0 and sediment conditions influences on EPC_0

The most common method for measuring the sediment's P buffering capacity is equilibrium phosphate concentration at net zero sorption (EPC_0). EPC_0 measures if SRP is desorbed or adsorbed by the sediment from the surrounding solution in laboratory conditions (Simpson et al. 2021). In this term, the sediment can act as a strong buffer and reduce the $H_2PO_4^-$ levels in the stream (Simpson et al. 2021). P binds mainly to Fe, Al, Ca, Mg, where Ca has the hardest bound to P (Kindervater & Steinman 2019). Therefore, the $H_2PO_4^-$ will bind to Fe^{3+} and Al^{3+} oxidases in the sediment, and a higher binding potential gives a higher buffering capacity which gives EPC_0 a lower value. When the SRP concentrations in the water column are more significant than the EPC_0 in the sediment, the sediment will absorb the SRP. This process can be reversed by changing conditions (Simpson et al. 2021). Factors like pH and finer sediment containing more clay minerals with more metal oxides have the potential to adsorb P, which gives EPC_0 a lower value.

Simpson et al. (2021) conducted a meta-analysis of phosphorus buffering in stream sediments, and the pH varied from 4.8 and 8.6, which points out that $H2PO_4^-$ was the dominant form in the sediment. The high adsorbing affinity to metal oxides in low pH gives a lower EPC_0. $H_2PO_4^-$ does not necessarily have to be a high adsorbing potential to the sediment if there is a low contact between the metal oxides and phosphate in high streams. At high pH, the phosphate can be released into the water column after precipitation with Ca minerals in hard waters (Simpson et al. 2021). The sediment minerals have a unique role by giving a lower EPC_0 value. Finer minerals increase the sorption, and lower contact between the water column and the sorption site decreases the EPC_0. The finer sediment has more clay minerals and more metal oxides, making finer sediment have a better buffering capacity than coarser sediment (Simpson et al. 2021). The buffering of $H_2PO_4^-$ - depends on the physical and chemical characteristics. However, the bottom of the sediment is constantly changing by surrounding land use, and other disturbances like wildfire stream impoundments affect the sediment composition and EPC_0. Storm events can scour the sediment, and new material can be replaced, affecting the EPC_0. The EPC_0 in the sediment is not constant and changes over time (Simpson et al. 2021).

2.12 Biotic processes in the sediment

P will also to be stored in organic form by biotic processes. $H_2PO_4^-$ can be stored in biomass from the floodplains, storing high contents of P in biomass. Algae and microbes will mineralize back $H_2PO_4^-$ from organic form to inorganic form into the water column (Simpson et al. 2021; Trentman 2020). These processes vary a lot during the season (Trentman 2020). During the winter, the flooding period is more extended than in the summer. In the summer, the flooding on the terraces is more frequent but for shorter periods. The lower vegetation rate and lower flow period give almost double nutrient loads downstream (Hodaj et al. 2017). Vegetation has a vital role in $H_2PO_4^-$ uptake to store in organic material (Lewandowski & Nützmann 2010).

3. Methods and site description

3.1 Site description

Four SDs were selected in Sweden for this study. The study is a part of the research project *Two-stage ditches in Sweden* financed by Formas, Oscar, and Lili Lamm Foundation and the Swedish Agency for Marine and Water Management. Two of the ditches are located near Nyköping in Sörmland and Norrköping in Östergötland, while the other two are in Skåne near Trelleborg (Table 1).

SD2 Åkra was constructed in 2012 near Nyköping and had terraces on both sides. The ditch is 730 m long.

SD3 Hestad is located near Norrköping, and it was constructed in 2014 with mixed terraces from two sides to a single side and a total length of 1500 m.

The two SDs in Skåne are part of the same river Tullstorpsån. Both SDs were constructed in 2013, and upstream SD7 St Markie has mixed terraces (single but of different heights) with a total length of 1960 m, and downstream SD8 Källstorp has two-sided terraces with a total length of 1770m.

Table 1. The four different SD`s in Sweden with different constructions of terraces and length. Data from Lukas Hallberg.

Site ID	Municipality	Construction year	Terraces	Terraces length (m)	SD length (m)
SD2	Nyköping	2012	Two-sided	660	730
SD3	Norrköping	2014	Mixed	1350	1500
SD7	Trelleborg	2013	Mixed	790	1960
SD8	Trelleborg	2013	Two-sided	890	1770

All studied SDs are in the agricultural landscape, and especially SD3,7,8 have a high proportion of agricultural land use (Table 2). SD2 has the lowest proportion of agricultural land use, with only 7% of the total landscape dedicated to crop production. SD7 has the highest proportion of wetland, with 7% of the total landscape. SD3 has the highest share of ley production. SD7 and 8 have the highest share of agricultural land use as the rest of Skåne.

SD2 and 3 have the highest share of clay soils which constitutes the characteristic of soil type in the middle part of south Sweden (Table 2). The most southern part of Sweden has more distribution of all particle sizes and is characteristic of till soils. SD 7 and 8 have a high proportion of sand in the soils.

Table 2. The FAO class in the catchment area with distribution of the mixed soil texture and agricultural land use. The agricultural land use is the summery of crop production, ley, and wetland. Data from Lukas Hallberg.

Site ID	FAO Class	Clay %	Silt %	Sand %	Agricultural land use (%)	Crop production (%)	Ley (%)	Wetland (%)
SD2	Silty clay loam	34	48	18	27	7	13	7
SD3	Clay loam	40	40	20	70	48	22	0
SD7	Loam	23	38	39	81	76	5	0
SD8	Loam	19	35	46	81	69	10	2

3.2 Methods

Data collection

Stream water, soil, and sediment samples from SDs were taken in the spring of 2022. One field trip was made every month to samples. In addition, the water quality data since 2020 was used to correlate EPC_0 results with P-concentrations and the fractions.

3.2.1 Sediment and water sampling

During the fieldwork, samples were taken, as shown in Figure 2. The samples were collected with a spade on the terraces and the channel. Sometimes the soil was frozen, and a sledgehammer was used to break up the topsoil. 3 cm^3 of sediment on the terraces were taken as ten subsamples every 2 m. In the channel, the sampling was collected with five subsamples with 6 cm^3 every 4 m. The upper layer of the sediment was collected to get a proper amount of deposit that reflects the bottom of the channel. Every subsample was put in a plastic bag as one composite sample. All the plastic bags were marked as shown in Figure 3. In the laboratory, the samples were mixed, and large roots were removed and put in a freezer to reduce phosphorus concentration changes due to temperature. Stream water samples were taken in a plastic bottle both up and downstream. In total 16 sediment and 8 water samples were collected during the field trips.

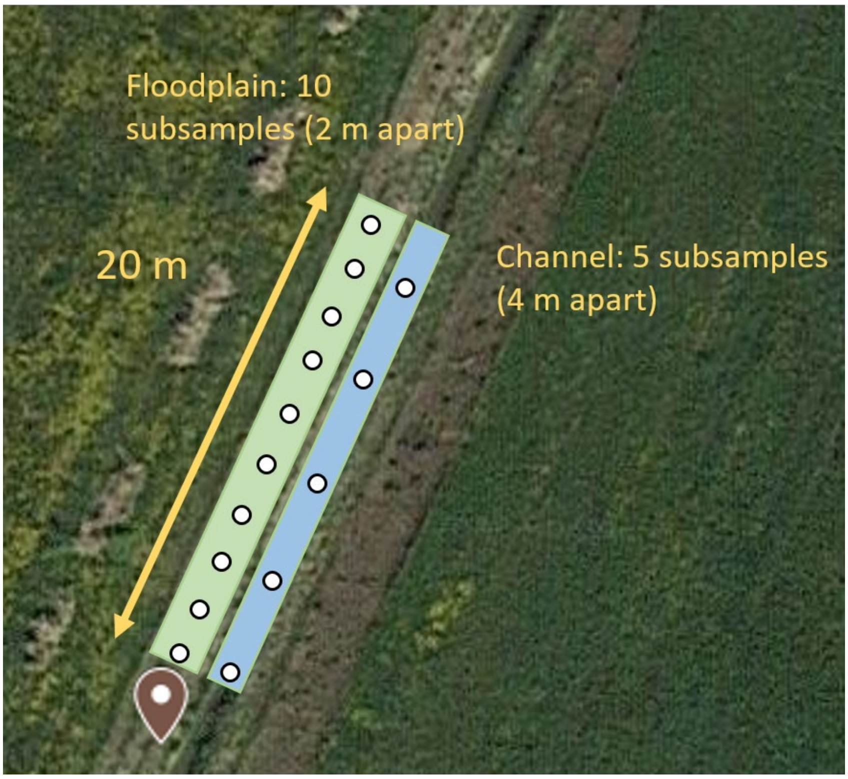

Figure 2. Fields sampling method with sediment collection on the terraces and channel.

Figure 3. SD2 Åkra showing field sampling method, downstream and upstream.
The samples were marked after Two-stage ditch (SD3,4,7,8), terraces (T), channel (C) and stream water (S), site up (1) and downstream (2).

3.2.3 Laboratory methods

To measure sediment P buffering capacity, the most common method was implemented for measuring the sediment's P buffering capacity. Equilibrium phosphate concentration at net zero sorption (EPC_0) measures the sediment's absorption and desorption capacity of SRP to the surrounding water solution. Also, water samples were collected to see various phosphorus forms and concentrations due to the sediment buffering capacity. This method is similarly used by Trentman (2020) to calculate EPC_0, fractions and P concentrations by a spectrometer (Appendix 1).

3.2.4 Measuring P-standard concentrations

A P-buffer solution was made to calculate the added P concentration before incubating to calculate the sediment buffering capacity. First, the P-standard was made by mixing KH_2PO_4 with MQ water and later diluted down to 2000 PO4-P ug L^{-1}. Then, a spectrometer was used to measure the absorbance for P- concentrations (0, 100, 200, 400, 800 ug L-1) (Figure 4).

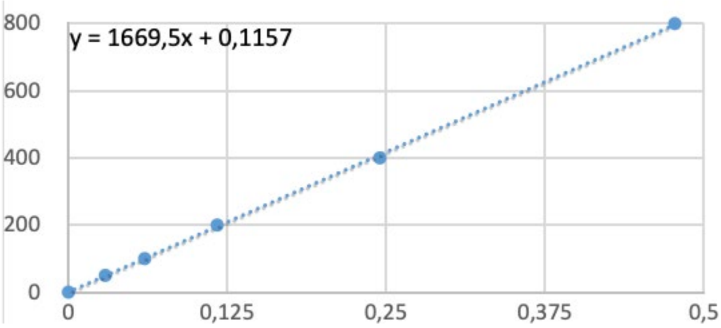

Figure 4. Plotted standard curve with (Y= P conc 0-800 ug L^{-1}, X= abs) the equation (Y= slope + interference) needed to calculate the P-standard concentration.

The reaction with the reagent solution mixed with the P-standard solution makes the samples blue. The connection between higher P-concentration (bluer color) and higher absorbance is used to calculate the P-standard concentrations. The absorbance was later plotted in Excel to make a standard curve. The standard curve shows slope and intercept, making it possible to convert the absorbance to actual P concentrations. The actual P-concentration was calculated after the formula:

$Abs \times slope + intercept = Actual\ P$ conc (ug L^{-1}).

$\frac{Actual\ P}{Added\ MQ}$ = Undiluted concentration (ug L^{-1}).

To make sure KH_2PO_4 and MQ give 2000 PO4-P ug L^{-1} concentration, a calculation was made by dividing the added MQ-water by the undiluted concentration. The average concentration from the undiluted P reveals the P-standard concentration. The P-standard solution shall have a concentration near 2000 PO4-P ug L^{-1}.

3.2.4 Sediment and water analyses of phosphate

Before incubation

The homogenized sediment was taken out of the freezer. First, the spike (P-standard) was added to a 60 ml tube in five-step concentrations (0, 100, 250, 500, 2000 PO4-P ug L^{-1}). Later 5g of sediment were added to the tubes, and then the tubes were filled with steam water up to 40 ml. The incubation started after the tubes were put in the shaker for 24 h at 150 rpm. In addition, the spike and stream water were also added into tubes without the sediment for further analysis of P concentrations without the sediment. The tubes were placed in the fridge during the incubation period. The incubation means interacting SRP with the sediment and measuring how much of the added P in the sediment can be adsorbed by the metal oxides.

After incubation

The next day the tubes were centrifuged at 3000 rpm for 15 minutes. After that, some muddy samples were centrifuged for 15 minutes more to separate the sediment from the supernatant and facilitate the filtering step. Finally, 5 ml of the supernatant were filtered into a 10 ml tube. In total, 93 samples were filtered, plus the spike and stream water samples. The samples were put in the fridge until the next day. To analyze the phosphate concentrations, a spectrometer was used. First, a new P-standard was analyzed to convert the absorbance to P-concentration by the same methods as for the P-standard. Then, the filtered supernatant was pipetted to another 10 ml tube mixed with MQ water. 2 ml of reagent was added. After 10 minutes, the analysis started by pipetting from the tube to the cuvette by adding MQ-water first to blank the spectrometer. Higher absorbance capacity gives less SRP left after the incubation, and less blue color with the reagent gives less absorbance and lower EPC_0.

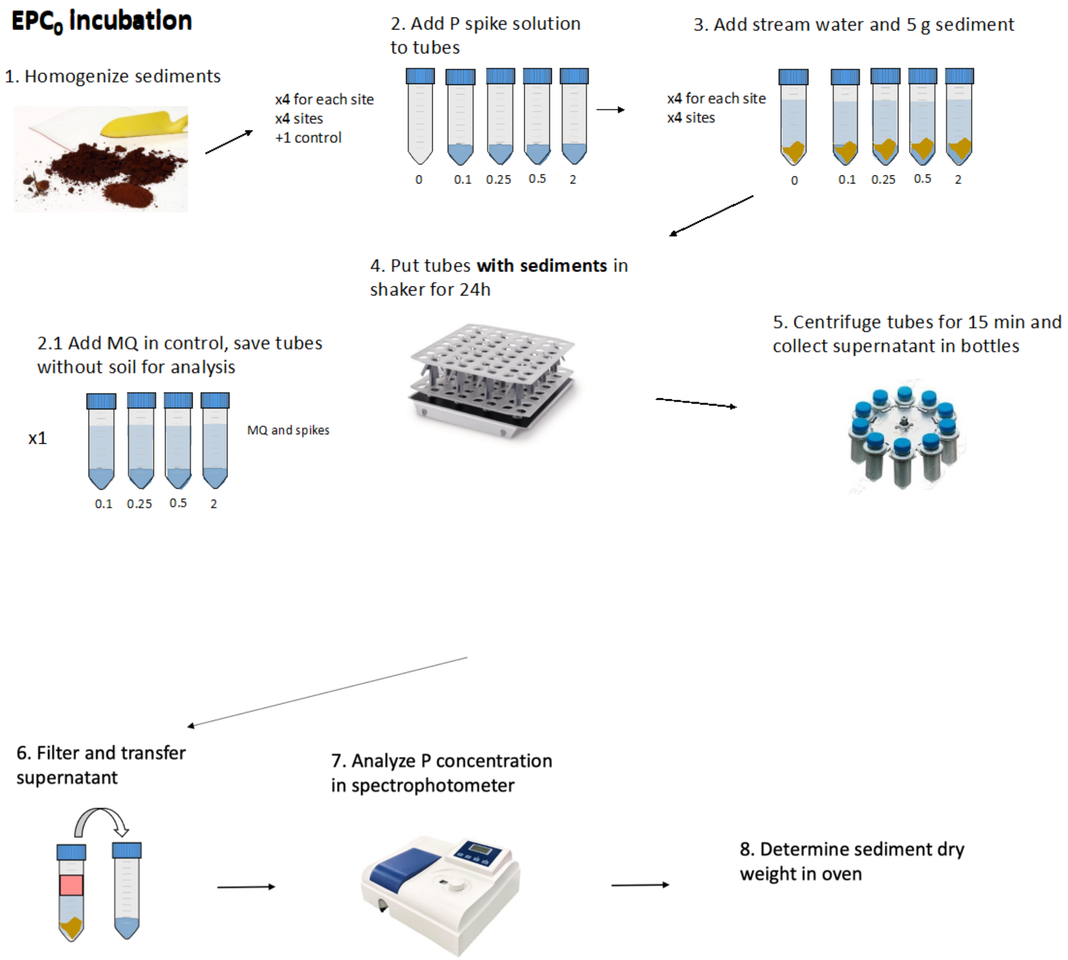

Figure 5. Number of steps to analyze the phosphorus concentration before and after the incubation.

3.2.5 Dry weight

All 16 sediment samples were weighted to measure the dry weight. The aluminum form with sediment measured the fresh weight (FW). Then the aluminum forms were put in the oven (105°) for 24h. The next day the aluminum forms were weighed again to calculate the dry weight (DW) and corrected for the aluminum forms initial weights.

3.2.6 EPC_0 calculation and convection of units

The absorbance from all 93 samples was converted from nm to ug L^{-1} by using the same methods as section 3,2,3. The total added P (ug L^{-1}) is the additional P from the spike (ug L^{-1}) and stream water (ug L^{-1}) before the incubation without the sediment. The equilibrium P is the phosphorus after incubation after being absorbed or desorbed from sediment (ug L^{-1}). The change of P (ug L^{-1}) is the difference in phosphorus concentrations before and after the incubation. To calculate EPC_0, units need to be converted to (ug/L /g DW) for total added P and change in P. All conversions of units and calculations are made by the formula:

1. $\frac{DW\ (g)}{FW\ (g)} = weight\ quota$

2. $Weight\ quota \times\ FW\ before\ incubation\ (g) =$
$DW\ before\ incubation\ (g)$

3. $\frac{Total\ added\ P\ (\text{ug L} - 1)}{DW\ (g)} Total\ added\ P\ (\text{ug/L /g DW})$

4. $\frac{Change\ P(\text{ug L} - 1)}{DW\ (g)} = Change\ P\ (\text{ug/L /g DW})$

The converted units to ug/L /g DW are then plotted in Figure 6, and every point is the change P by the added P. Intercept on the x or y-axis is the EPC_0. When the change of P is 0, there is no sorption or desorption of P, which gives the EPC_0. The EPC_0 was later correlated with the P concentration in the stream water. If the EPC_0 is lower than the stream water concentrations, the sediment is a sink of P. Some of the spots gave poor correlation, which gives a different trendline and high EPC_0. To give a better correlation, some of the points were deleted to give a better reflection of EPC_0.

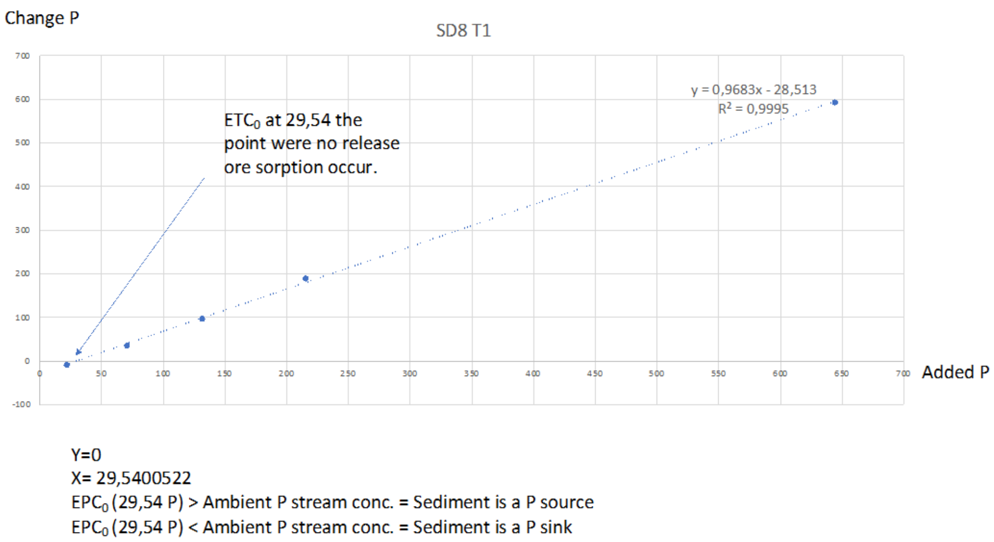

Figure 6. The method to calculate EPC_0 by a diagram on one of the terraces in SD8.

The calculated EPC_0 from a different site and stream location was later correlated with pH, DO (oxygen levels), and clay content to find correlations and strengthen the literature. This data was analyzed by the SLU soil and environment department and later evaluated in Excel.

3.2.7 Phosphors fractionation

The investigation of the sediment's fractionation was used by sequential chemical extraction method in a parallel MSc project. This method is also known as phosphorus fractionation to analyse the fractions distribution of phosphorus. The method is based on a protocol from the Department of Aquatic Sciences and Assessment at SLU (Appendix 1). This compendium is based on Psenner & Pucsko (1988) and Hupfer et al. (1995, 2009), who have developed this method. This method gradually removes bound P based on adsorption affinity from the loosely bound P first and later the bound P. This stepwise method by dividing different P forms in order of the P adsorbing capacity makes it possible to divide the P in order of redox sensitivity, Mn/Fe bound P, Al-hydroxides, non-reducible P, organic P Ca bound P and refectory P. All these fractions give the TP from each sediment sample. NaOH and HCl were added in several steps to dissolve the different P bounds and analyse the fraction concentration. The analysis was made by adding a P-standard solution in concentration from 0-800 ug P L^{-1} for later analyse the fractions by a spectrometer and calculating the concentration from nm to ug P L^{-1} by plotting the absorbance in a P-standard curve and converting the equation from the curve to concentration by the slope and intercept. The P concentrations were later converted to P kg DW^{-1} by recalculating the sediment's dry weight to the correct units.

3.2.8 Visual and statistical analysis

All diagrams were made in Excel to present the evaluated data and correlations.
All the data between the site, stream location, and channel vs terraces were analysed using the T-test and Anova test in Excel. This was made to find correlations and significant differences between the sites and mainly between the stream and the terraces and channel to reflect the SDs function to reduce SRP downstream. Excel was also used to calculate EPC_0 effectively and to keep all data. The EPC_0 was later analysed to find if pH, DO, and clay content had any relationship with EPC_0.

4. Results

The distribution of P forms and concentrations in stream water is shown in Figure 7 from February to May between 2020-2022. The highest concentrations of TP were in SD3 with 640 ug L^{-1} and the lowest in SD7 with 10.9 ug L^{-1}. The PP concentration varied between 2 and 360 ug L^{-1}. The SRP concentration varied between 2 and 121 ug L^{-1} during the period.

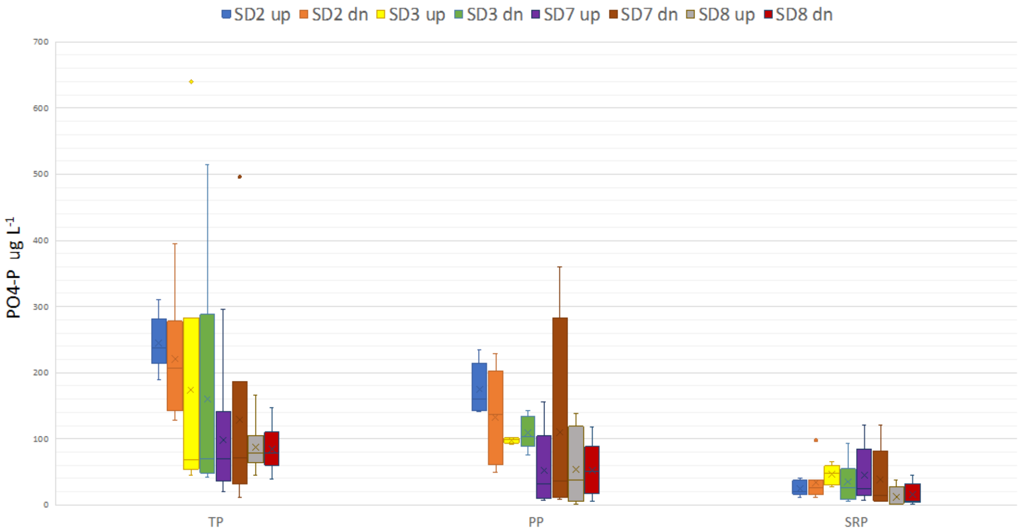

Figure 7: Boxplot of total phosphorus (TP), particle bound phosphorus (PP), soluble reactive phosphorus (SRP) (PO4-P ug/L) in SD2,3,7,8 downstream (dn) and upstream (up) February to may between 2020-2022. Analyzed data from SLU.

The stream water SRP concentrations varied from 2 to 66 PO4-P ug L^{-1} during the investigation period from February to April 2022 (Figure 8). There was a significant difference between the SD's SRP concentration but not within the SD between upstream and downstream (Table 3). February had the highest average concentration of SRP and April the lowest in the stream water during the investigation period.

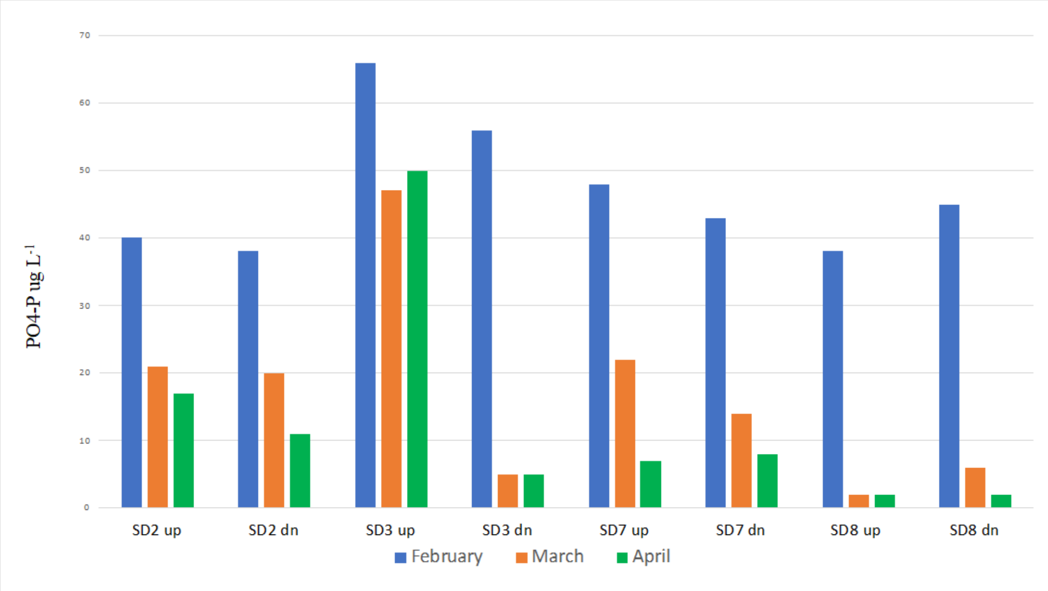

Figure 8. Stream water SRP concentration (PO4-P ug/L) from February to April. Analyzed by the SLU, soil and water department and collected by the farmers up and downstream.

The sediment EPC_0 varied from 0.49 PO4-P ug/L / g DW in February in SD3 to 166 PO4-P ug/L / g DW in March in SD7 (Figure 9). The highest average EPC_0 during the test period was in March in all the channel sites upstream with 76.4 PO4-P ug/L / g DW, and the lowest was on all terrace sites downstream in April 2021 with 1.2 PO4-P ug/L / g DW. A lower EPC_0 gave the larger buffering capacity due to the SRP concentration in the stream water. Anova points out a significant difference between the sites, but a T-test shows no statistical deference between the stream location within the SDs. There was no statistical difference in time and between the channel and terraces (Table 3). Every site shows similar pattern throw time, which implies that EPC_0 varies during the test period but not that much.

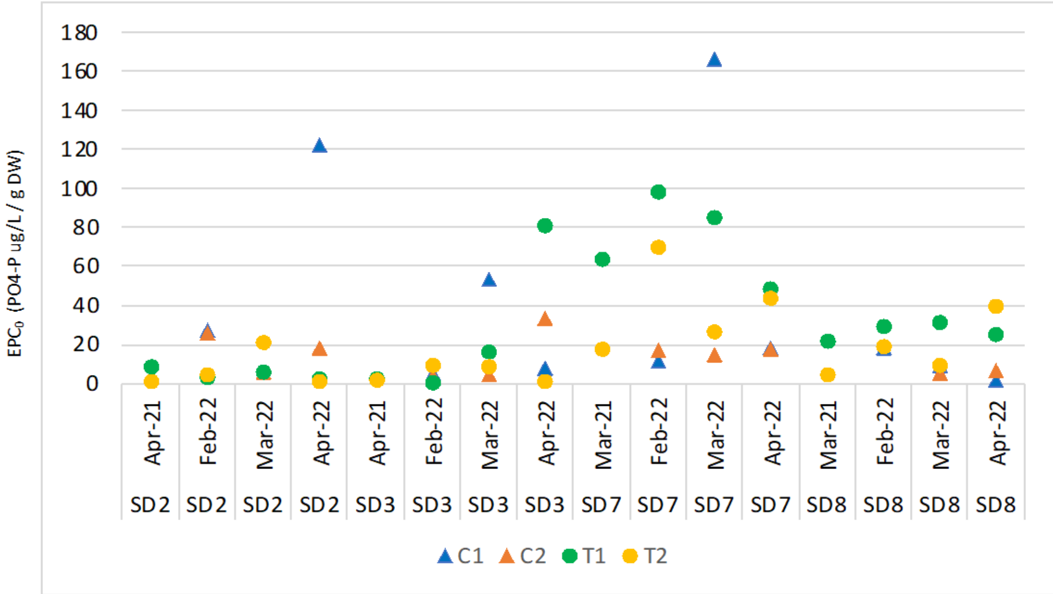

Figure 9. Channel (C), terraces (T), upstream (1), downstream (2). *Calculated EPC_0 (PO4-P ug/L / g DW). Data from March and April 2021 from Emilien Casali research.*

The calculated EPC_0 in Figure 9 was correlated with SLU stream water SRP concentrations data (Figure 8) to calculate the retention capacity from the sediment during spring 2021 from Emilien Casalis data and 2022 (Figure 10). SD2 had more samples that had retention of SRP. SD3 also showed an advantage in retention among the samples. SD7 had the most samples released of SRP. SD8 also had more samples with the release of SRP rather than retention. The highest release of phosphates was in the channel upstream of SD7, with the released 144 PO4-P ug/L/gDW in March 2022. The highest phosphorus retention was in February 2022 in SD3 upstream terraces, with an uptake of 82 PO4-P ug/L / gDW. SD3 had the highest average retention capacity on the terraces and channels, which could depend on the high clay content in the sediment. The sediments collected in February had the highest average retention capacity during the test period. The retention capacity varied between the SDs and between the time series. An Anova test confirmed a significant difference between the months. A T-test showed no statistical differences between streams retention capacity and between channel and terraces (Table 3).

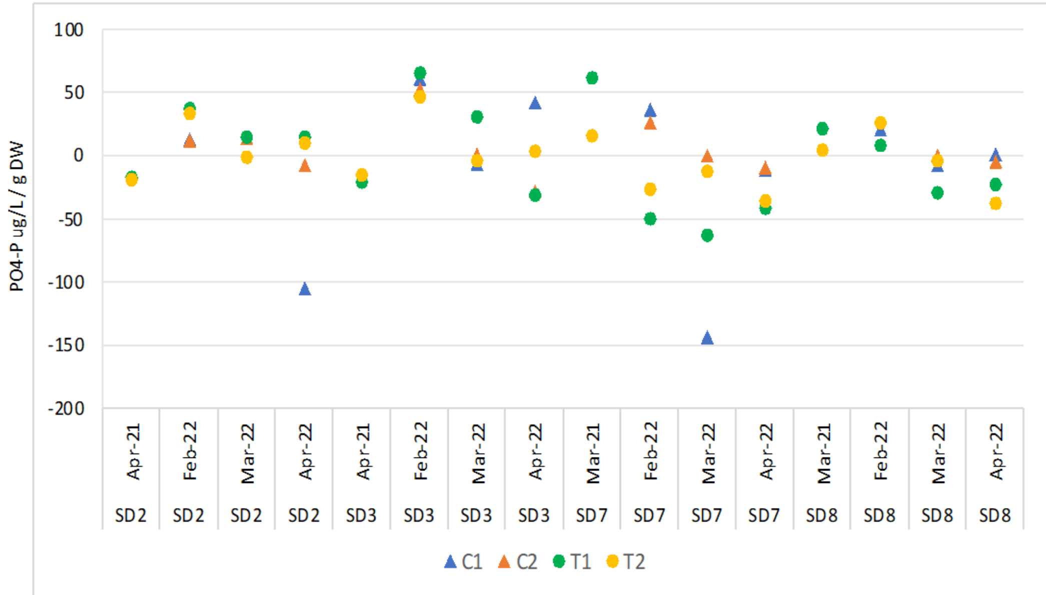

Figure 10. Channel (C), terraces (T), upstream (1), downstream (2). *Retention capacity (PO4-P ug/L / g DW) correlated EPC_0 with SRP concentration in the stream water from the SLU. Positive retention means uptake of SRP. Negative retention means release of SRP from the sediment. Data from March and April 2021 from Emilien Casali research.*

The correlation in Figure 11 shows SRP concentrations versus EPC_0 and how EPC_0 changes with higher SRP concentration to be a source or sink of phosphate to the stream water. If the differences between EPC_0 and SRP are more significant than zero, there is a net P exchange. The black 1:1 line shows where the zero P fluxes appear from the sediment. When the plots do not follow the line, there are P fluxes either by retaining ore release SRP. This figure points out that there are P fluxes in both ways. SD2 and 3 have more retention than release, and SD7 and 8 have the opposite fluxes (Figure 11). SD7 was the ditch with the most phosphate release in both channels and terraces. In SD8, there were equal fluxes in both ways for the terraces, but some more were released from the channel. SD2 had more retention among the test samples and especially on the terraces. SD3 had equal fluxes on booth ways on the terraces but much more retention among the samples on the channels. The SDs had the most average retention in February compared with April, which had the most average phosphate release. March had samples fluxes in both ways, but some samples had a high phosphate release.

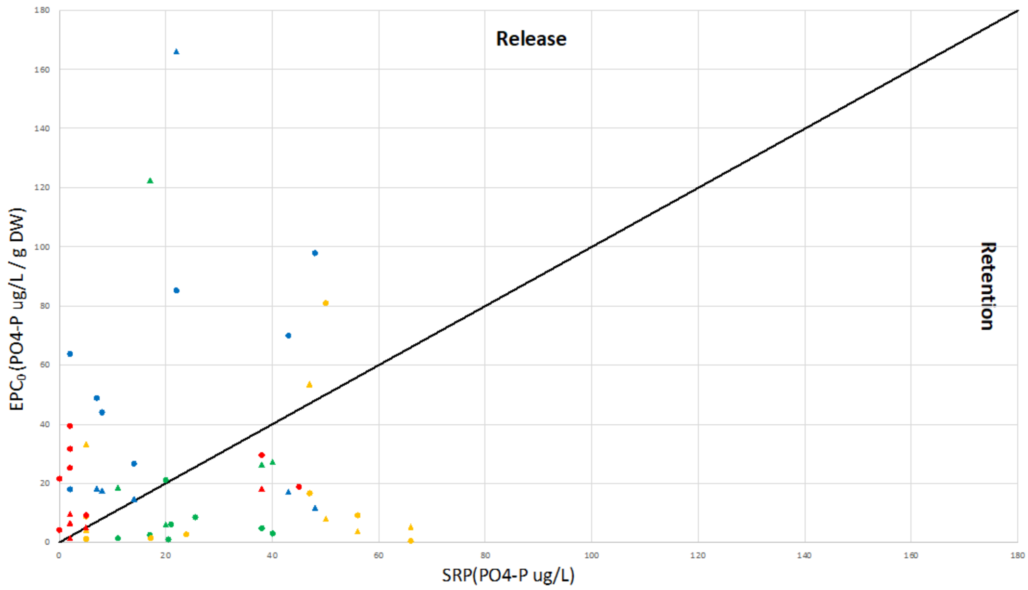

Figure 11. Scatted plots with correlation between EPC_0 in terraces (circle) and channel (triangle) and SRP in the stream water. SD2 (green), SD3 (yellow), SD7 (blue), SD8 (red). SRP data from the from SLU. The black line shows the 1:1 line which appear zero P fluxes from the sediment.

A statical analysis was made to find correlations between upstream and downstream EPC_0, retention capacity, and specially SRP concentrations. There were no significant differences between the stream location among the variables, and a T-test made the analysis. This indicates that there is not any decrease in SRP concentration on the stream water transport downstream among any of the sites. An Anova test pointed out a significant difference for SRP and EPC_0 between the samples between the SD sites, which indicates that the SDs perform differently. There was a significant difference between the retention capacity between every month, which points that the retention is different from month to month. The EPC_0 was not significant differences in time, which points that the EPC_0 does not change that much over this test period, but there are not either equal. There were no significant differences between the channel and terraces on EPC_0 and by retaining phosphorus from the stream water (Table 3).

Table 3. Statistical analyzes by T-test and Anova. SD2-SD8 done by T-test between up and downstream. Total differences between all sites and stream location. Time analyzes differences between February to Mars. A total analyzes between all terrasses and channels.

Variables	EPC_0	Retention	SRP
SD2	0,333	0,587	0,425
SD3	0,260	0,479	0,985
SD7	0,164	0,217	0,889
SD8	0,679	0,925	0,566
Total	0,037	0,074	0,026
Time	0,844	0,005	
Channel, Therraces	0,930	0,883	

The literature refers to oxygen levels (DO), pH, and particle sizes to influence the EPC_0 (Smolders et al. 2017; Bergström et al. 2015; Trentman 2020; Palmer-Felgate et al. 2011). The DO, pH and clay content data were correlated with EPC_0. However, no good correlations were observed to affect EPC_0 on these variables (figure 12-14).

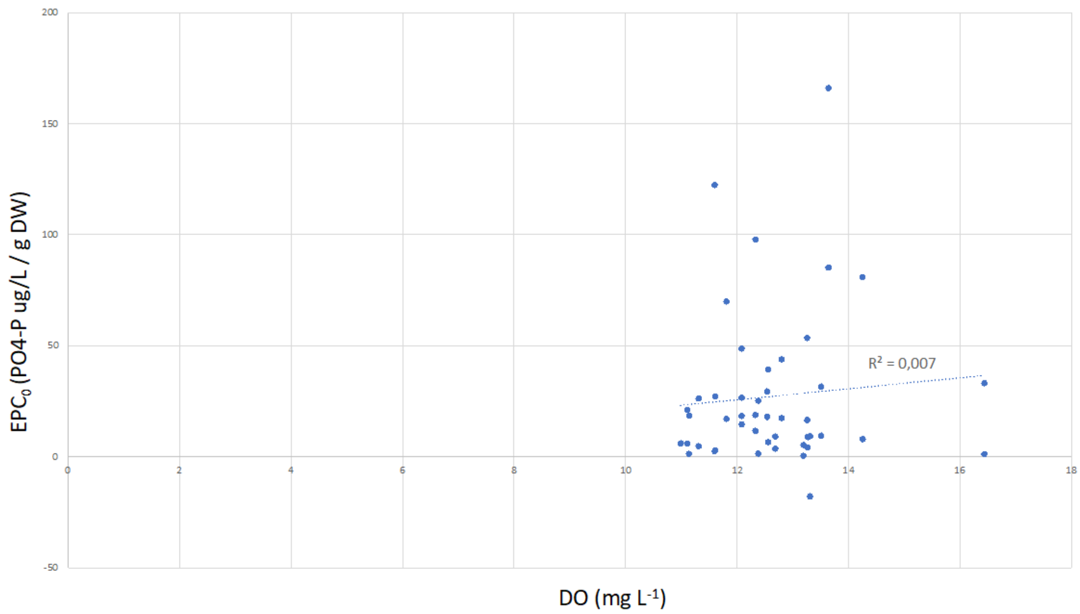

Figure 12. Conjunction in R2 between EPC_0 and PH.

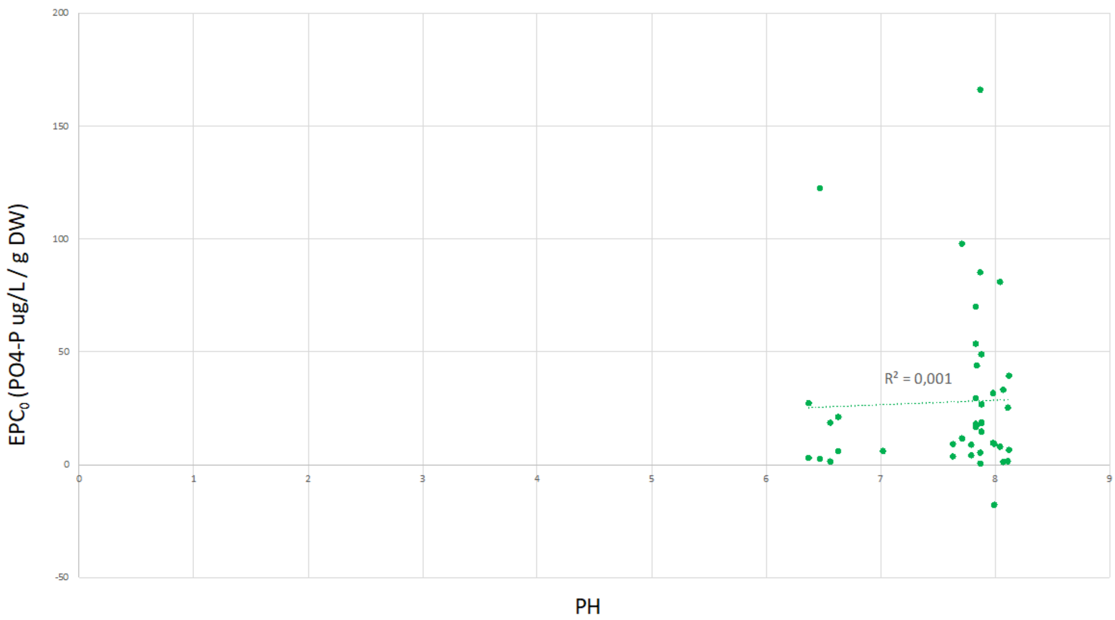

Figure 13Conjunction in R2 between EPC₀ and PH

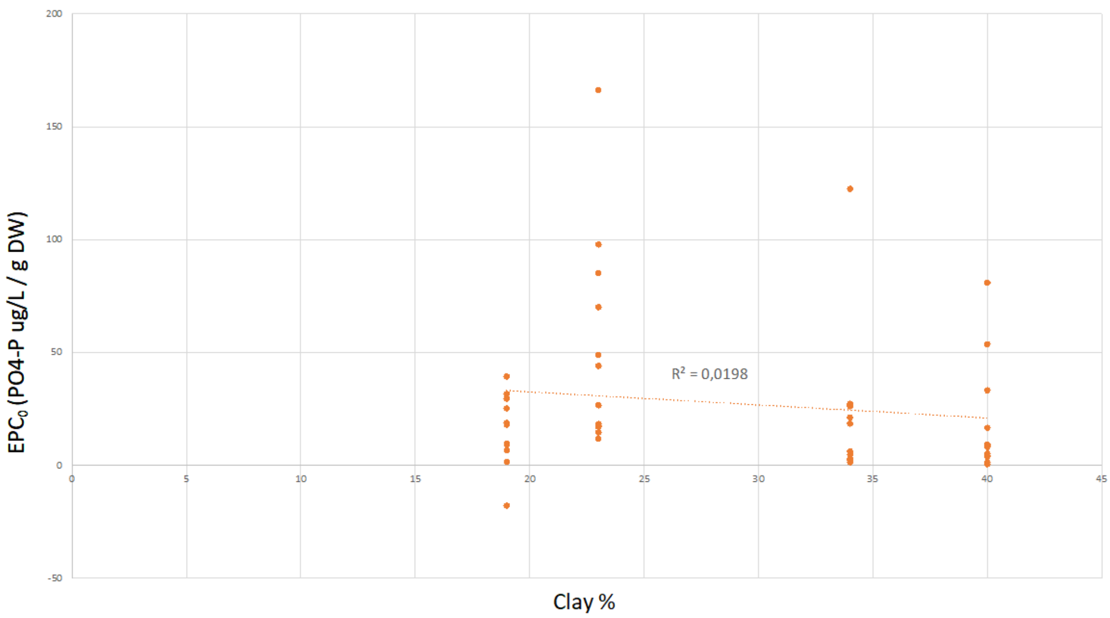

Figure 14. Conjunction in R2 between EPC_0 and clay content.

A cross-reference of data was made with Emma Ryding, who has been investigating the sediments fractions of P. By comparing the EPC_0 with the different fractions of Mn/Fe bound P, Al-hydroxides, non-reducible P, organic P Ca bound P. Refectory P correlations were plotted in Figure 15. No significant correlations were observed between the different P fractions and EPC_0. What was found is that there is some correlation with the MQ-P, which is the most loosely bound phosphorus among the fractions, which indicates that the MQ-P increases with higher EPC_0 in the sediment

(Figure 16). This indicates that in higher EPC_0 conditions, the sediment is statured with P, and more loosely bound P will therefore be stored in the sediment.

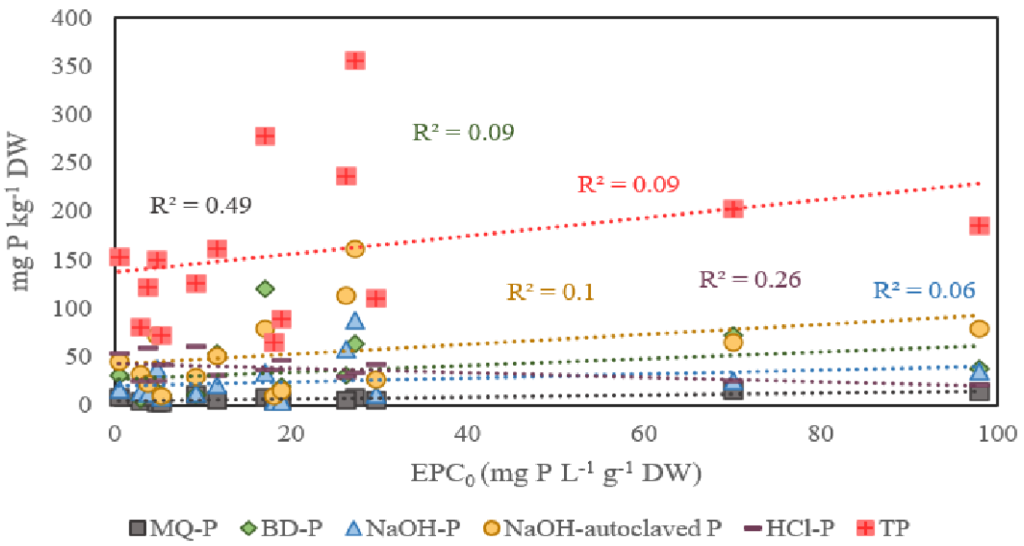

Figure 15. Correlation between the P fractions and EPC_0.

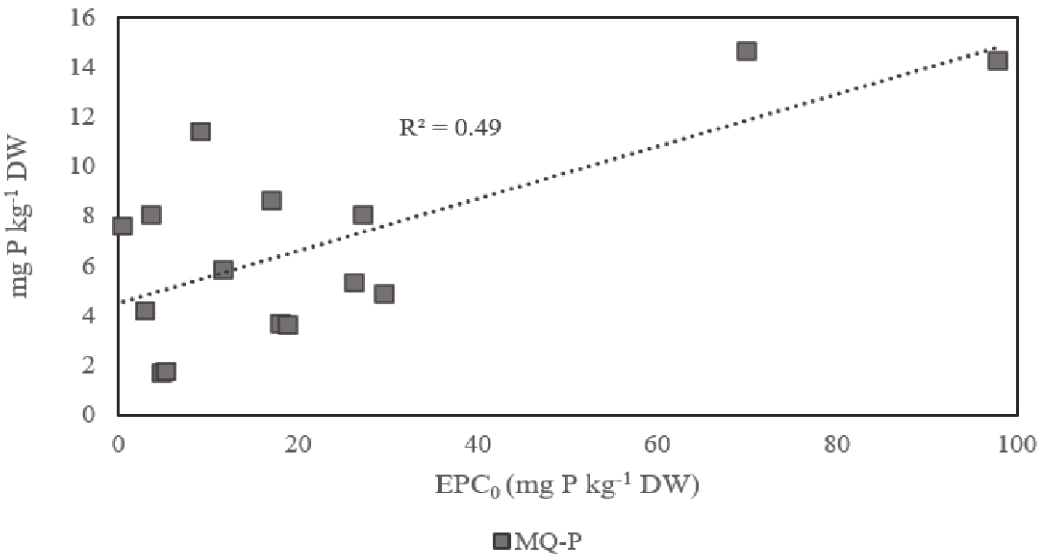

Figure 16. Correlation between MQ-P witch is the loosely bound fraction of P and EPC_0.

5. Discussion

5.1 Stream water phosphate concentrations

Phosphorus losses to watercourses lead to eutrophication. Several mitigation measures have been proposed to reduce phosphorus losses, and SD is one of them. The stream water phosphate concentrations were measured to evaluate the SD potential for reducing P losses. The stream waters SRP concertation was measured in two ways. The first was by the same method as analyzing EPC_0 and the second by water samples sent by the responsible for the SD for later be analyzed by SLU for a paradelle PhD project. The SLU data were used before the own laboratory data because the SRP analysis from the SLU is more precise. Therefore, later this data was used to calculate the retention capacity among the samples. SD3 had the highest SRP concentration during the test period, most likely due to the high clay concentrations. Also, SD2 showed the same pattern and especially for March and April. SD7 had the highest P concentration in the stream water since 2020 and could be explained by the varied topography with many hills and a wetland close to the SD, which is probably leaching SRP. In February, there was a significantly higher concentration of SRP. Most likely due to the higher streamflow during that month. SD8 had the lowest SRP concentration during the test period and lowest clay content, strengthening the theory that clay contents influence SRP concentrations in the stream water. No significant differences were observed along the stream from all the ditches. Additionally, the conditions could be different between the stream sites with finer minerals upstream and with the higher stream downstream, including higher SRP concentration in the water. The result has not considered the nearby agricultural fields and their SRP leakage by drainage and surface runoff. The fields' cropping system, crop history, fertilizing, and other chemicals, physical, and biological processes have not been considered. One of the biggest reasons that SRP does not decrease downstream is probably the contribution from drainage from the nearby fields, which increases the SRP concentrations in the stream water. Fewer samples showed the capacity to reduce SRP and instead be a source of P, especially in SD7 and SD8, which could explain the increase of SRP downstream.

5.2 Sediment's role in phosphate losses

EPC_0 is the most common method to measure if the sediments are source ore sink of phosphate. The EPC_0 and SRP concentration in the stream water varied during the test period, but EPC_0 showed no significant differences during the time series within the SDs. The retention capacity varied a lot and shifted from being a sink to being a source within the same sampling location. The similar pattern of EPC_0 but various P concentrations in the stream water indicate the deciding factor for P fluxes. Chemical and physical processes affect sediment sorption and desorption capacity. Simpson (2021) shows a similar result in their meta-analyze with data from 45 studies with 942 parried observations, and 83% showed either significant retention or release of phosphate and the rest were equilibrium. The study pointed out a net average of more retention among the observation. The literature refers to DO, pH, and particle sizes to influence EPC_0 (Smolders et al. 2017; Bergström et al. 2015; Trentman 2020; Palmer-Felgate et al. 2011). No good correlations could be found between EPC_0, DO pH, and clay content. Why this variables do not influence the sorption capacity from the sediment is unclear, but one reason could be that there is no contact between the stream water and sediment. In the high streamflow, the colloids with Fe and Al oxides will not be able to adsorb the phosphate. The stream water was analyzed for pH and DO during the project to find correlations with the sediment EPC_0. Stream water could influence the sediment's pH and DO concentrations. But not as much as the pH and DO concentrations in the sediment, which would be more relevant to analysis. The sediments have probably not the same pH and DO as the stream water. The sediment influences P fluxes, forms, and concentration in the sediment and therefore affect the EPC_0. The correlation would be better with EPC_0 if the sediment's pH and DO were analyzed rather than the stream water. SD2 and 3 had more retention among the samples than released, and SD7 and 8 showed more release of phosphate among the samples witch, indicating that clay content ore other factors could have some influence on sorption of phosphate despite the resulting point something else. The cross-reference pointed out that by higher EPC_0, there are more loosely bound phosphates in the sediments, indicating that the sediments' adsorbing places are full. Not either any conjunction was made with the Fe and Al fractions which was a surprise because the amount of metal oxides affects the sediments buffering capacity. No correlations were found for EPC_0 and retention capacity between the stream location and terraces vs channel. No statistical differences have been made between upstream and downstream concentrations in SRP concentration, questioning the SDs performance. The samples showed fluxes in both ways on the terraces and channels, which is expected that the sediment will be able to retain and release SRP from the different sites. To install SDs, the utility of reducing SRP downstream needs to be considered due to the installation cost and losses of farmland. Further investigations of EPC_0 are needed to decide on overlying performance.

5.3 Method limitations

The P-concentration for the incubation (0, 100, 250, 500, 2000 PO4-P PO4-P ug L^{-1}) measures the sediment fluxes. Concentrations with 250, 500, and 2000 are incredibly high P-concentrations, and 2000 PO4-P ug L^{-1} in stream water does not appear often. Instead, the concentrations should be between 0-200 PO4-P ug L^{-1} to reflect real P-concentrations in the stream water. To measure EPC_0, there is essential to have a suitable absorbance to give the correct correlations between total added P and change in P. Some of the measured absorbances did not follow the trendline to give good correlations between the added P and change in P. Therefore, some measurement points were removed to give better trendline. Sometimes the whole sample was deleted to give a realistic EPC_0. For example, SD2 C1 in March had an EPC_0 value on 380 PO4-P ug/L / g DW, which is almost four times higher than the second-highest value. Why some sediment has a high EPC_0 value can have several explanations. Nevertheless, a bad filtering process could explain the high EPC_0 values. Another thing to mention with the method is the incubation with a shaker. During the incubation, all the sediment can react with the phosphorus from the spike and stream water, which does not reflect the conditions and only reflects the EPC_0 in laboratory conditions. There is also an assumption that the 5g of sediment before the incubation can represent the whole test site, and therefore, the homogenization of the sediment is essential. Also, the samples in the plastic bag contain different amounts of water, and some samples contain much more water than sediments. Sometimes, the taken samples contain much water, especially the case for the samples from the channel. The samples from the terrace contain less water, and especially in March and April no water on the floodplains. The sediment to the incubation contained more water and roots from the channel and contained fewer colloids with Fe and Al oxides, which could adsorb less phosphate than a sample from the terraces containing more colloids in the wight of 5g. Especially in the channel, there was challenging to get the same amount of sediment from every sample site with a spade. There was also challenging to know how deep the samples were, so some could come from deeper sediment and some more shallow. In the future investigation, another tool, then a spade, could be used to better control collecting 6 cm^3 of sediment on the right deep. However, this process could be complicated, mainly in high water flow.

5.4 Two stage ditches roles in future climate.

Climate change will increase the annual precipitation in some parts of Sweden. The water intensity by cloudburst will increase the requirements of drainage intensity. The higher temperature will affect the structure of clay soils by less ground frost, which gives less water infiltration capacity. Also, heavy rain can damage the structure of fine particle soils, affecting water permeability (Mattsson et al. 2018). The intense water periods lead to more significant erosion problems, and soil erosion is a problem for eutrophication (Ansari et al. 2011). The higher temperature will boost the hydrological cycle, increasing the total precipitation by 10-20% (Larsson et al. 2013). The simulations point out that the total precipitation per year will increase by 100-400 mm in different parts of Sweden in the year 2100. The snow depth will decrease by 70% in the south parts of Sweden and along the coast in the north. Some areas of Sweden will have fewer top flows. Other areas will have higher top flows by 10-30% in different parts of Götaland and some parts of Svealand. (Larsson et al. 2013). Drainage intensity will be more important to maintain good crop production. A future scenario with higher water intensity leads to higher nutrient losses than we have today due to the higher temperature. There need to be more implemented measures to prevent eutrophication in the future. Today, there are several measures to prevent nutrients from the agricultural fields from reaching the watercourses, but not that many measures for the phosphate in the watercourses to reach the sea. SDs could be an implication of decreasing the environmental impact on aquatic systems in a future change in the climate by sedimentation and sorption. A rapport by Nilsson & Johansson (2015) says that the higher water intensity leads to more flooding and more nutrient losses. Drainage is essential to reduce surface waters from cropland and reduce yield losses. Also, this report taking into the accent of increase in flooding in the future climate in Skåne. To handle crop losses by flooding, drainage needs to be improved and prevent nutrients from reaching the sea. The higher water intensity increases the demand to take the water from the landscape, from drainage to ditches. SDs do not only reduce the nutrient losses downstream but give 29% higher volume water capacity compared with a TD. This makes the SD manage the top flows better and faster than the TD. An implication of an SD would decrease the flooding from the fields by 62% with a 10 cm flooding and reduce the cost losses (Nilsson & Johnasson 2015). Installation of SDs costs money and reduces land that could be used to grow crops due to the broader ditch, so a future analysis of the SDs environmental benefits needs to be higher than the cost. This investigation of SDs shows mixed results, and the SDs need to be continually analysed. A fair analysis would be to analyse more SDs during the whole season and preferably some years to get a full eluviation of Swedish SDs sorption capacity.

Conclusions

The investigation of the SDs sorption capacity from the sediment has been showing different fluxes. The sediment analyses from the sites show fluxes of being a sink and source of phosphate can vary in time and space. The SDs in the southeast part of Sweden had more retention of P among the analysed samples in the channel and terraces. The SDs in the south part of Skåne had more P release among the samples in the channel and terraces. There were no significant differences in EPC_0, within stream location and between terraces and channels but between the sites. There were no differences in SRP concentration in stream water within the SD but between the sites and between the months. There was not either any differences in retention capacity by the sediment along stream location and terraces towards the channel but there were differences between the sites and between the mounts. No conjunction was found that pH, DO, and clay content should influence EPC_0. A cross-reference with a parallel MSc works with P fractions from the same SDs, showing no statistical conjunction between EPC_0 and the P fractions. But with some similarity with higher loosely bound P in the sediment with higher EPC_0. Further investigation is needed by investigate more SDs and over a longer period to give a better support of the SDs sorption of P performance in Swedish conditions.

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6. Popular scientific summary

Increased use of mineral fertilizers and livestock manure gives higher and more increased crop production. This has led to increased losses of nutrients to nearby watercourses, which transport the nutrient nitrogen and phosphorus to the big oceans and lakes, which causes increased algae blooms. This process called eutrophication has a negative impact on aquatic systems with dead bottoms. One measure to reduce the nutrient transport to seas and lakes is to resign the ditch that transports away from the water from the landscape to a two-stage ditch (SD). Compared with a Traditional trapezoidal ditch (TD), the SD has two terraces on the side of the channel, which act as a floodplain in high flow (figure 1). This will lead to a slower water flow in the ditch, reducing nutrient losses attached to the sediment. Instead, the phosphorus in the form of phosphate in the stream water can be adsorbed to the sediment and act as a sink instead of being transported downstream, increasing water quality. This process can be reversed by biological, physical, and chemical processes, which could make the sediment from retaining phosphate to releasing phosphate in changing conditions to the stream water and instead be a source of phosphate.

This study has been analysed four SDs sorption capacity from the sediment by calculating the EPC_0, which is the sediments equilibrium phosphate concentration at net zero sorption. EPC_0 is the common value to measure the sediment buffering capacity to phosphate by absorbing ore desorbing of phosphate. This value will later be correlated with the soluble particle-bound phosphate (SRP) in stream water to measure if the sediment is a source ore sink of phosphate. If the sediment EPC_0 value is higher than the stream water, the sediment act as a source of phosphate and reverses if the stream water concentration is higher than the sediment EPC_0. A further analysis was made to find a conjunction between pH DO (oxygen levels) in the stream water and clay content, which is claimed by the literature EPC_0 (Smolders et al. 2017; Bergström et al. 2015; Trentman 2020; Palmer-Felgate et al. 2011) to influence the EPC_0. The sediment and water samples were collected one time every month between February to April 2022 by filed execution and later analysed in a laboratory. The sediment and water samples were collected at the SDs beginning and at the end on the channel and terraces. By doing a statistical analysis between the SD site, stream location, and cannel towards terraces, a statistical difference reveals the SDs performance in retaining phosphate. The result pointed out that the EPC_0 varies between the sites but shows similar patterns within the sites. When the EPC_0 were correlated with SRP, the fluxes were in both ways, with the highest release with 144 PO4-P ug/L/gDW in SD7 upstream on the channel in March 2022 and the highest retention of 82 PO4-P ug/L / gDW in SD3 upstream on the terraces in February 2022. The investigation pointed that the SDs in southeast of Sweden with more clay soil had more retention than the loam soils in Skåne. The statistical analysis showed no differences in variables. Not either a conjunction was made between EPC_0 with pH DO (oxygen levels) and clay content.

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Linus Holgersson, Uppsala 2022

Appendix 1

Analysis: Phosphorus fractionation
Department of Aquatic Sciences and Assessment

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Sequential fractionation of phosphorus in sediment

To assess phosphorus in sediment a sequential chemical extraction can be used, commonly called phosphorus (P) fractionation. Different pools, with varying solubility and reactivity, that P is bound to is examined. This method description is developed from Psenner & Pucsko (1988) and Hupfer *et al.* (1995, 2009).

The pools are arbitrarily defined as in figure 1.

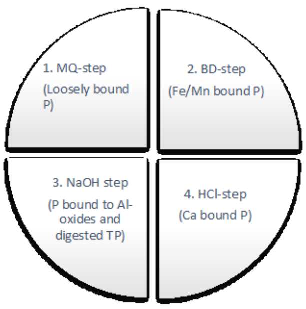

Figure 1. Explored pools in method.

Quantification range: 0-800 µg/l

Equipment
Scale:
Spectrophotometer:
Autoclave:
Centrifuge:

Appendix 1: Chemical and solution preparation
Appendix 2: Phosphorus extraction with spectrophotometer
Appendix 3: Dry weights and loss on ignition
Appendix 4: Methodological background and references
Appendix 5: Adaptations of the method
Appendix 6: Quality assurance

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Figure 17. Lab manual for the fractionation and P concentration from SLU.