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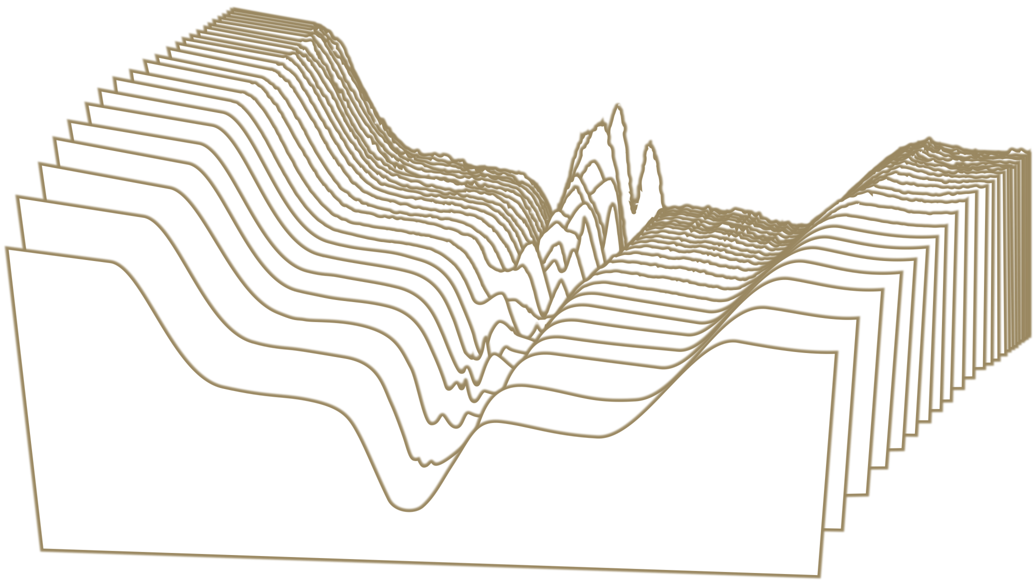
Floodplain remediation in agricultural streams

- Improved process understanding for reduced eutrophication

Lukas Hallberg



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Floodplain remediation in agricultural streams

Abstract

Anthropogenically modified headwater streams are key pathways for high nutrient and sediment loads from agriculture, sustaining eutrophication and poor water quality in downstream aquatic ecosystems. To address this challenge, stream mitigation strategies are required that complement in-field measures, to intercept internal pollutant loading in streams. In this thesis, the capacity for instream nutrient and sediment mitigation by floodplain remediation was evaluated. Ten catchments in Sweden with streams subjected to floodplain remediation were studied, covering differences in soil texture distributions, land uses and floodplain designs. Reductions in particulate and total phosphorus losses demonstrated that floodplains on both sides of the channel reduce erosion and improve downstream water quality, compared to both streams with floodplains on one side and trapezoid-shaped reference streams. Denitrification in floodplain sediments accounted for 33 % of total denitrification in remediated streams, controlled by floodplain inundation and stream nitrate concentration. However, at higher inundation frequencies, there was a concurrent increase in denitrification and soluble reactive phosphorus release from floodplain sediments, revealing a trade-off in solute nutrient mitigation. The overall impact on water quality by remediated streams was variable, with greater potential for sediment and associated phosphorus reductions compared to nitrogen. This was explained by high pollutant pressures and limitations in floodplain designs. To realize water quality improvements through floodplain remediation in agricultural streams, site-specific dominant pollutant forms (nitrogen, phosphorus and sediments) need to be accounted for by appropriate floodplain designs. Yet, as floodplains primarily target instream mobilization and transport, their efficacy can be obscured by excessive pollutant inputs from agricultural soils.

Keywords: Stream remediation, constructed floodplains, two-stage ditch, denitrification, nitrogen, phosphorus, stream metabolism, agricultural stream, eutrophication

Tvåstegsdiken i jordbruksvattendrag

Sammanfattning

Antropogent modifierade jordbruksvattendrag är framträdande spridningsvägar för höga närings- och sedimentbelastningar från jordbruksmark, vilket vidmakthåller övergödning och undermålig vattenkvalitet i akvatiska ekosystem. För att möta denna utmaning är åtgärdsstrategier utöver fältåtgärder nödvändiga, med kapacitet att fånga upp vattendrags interna utsläppskällor. I denna avhandling utvärderades kapaciteten för att begränsa närings- och sedimenttransport med tvåstegsdike som vattendragsåtgärd. Tio avrinningsområden i Sverige med anlagda tvåstegsdiken studerades. Dessa områden skiljer sig med avseende på jordmånens texturfördelning, markanvändning samt svämplansutformning. Minskade förluster av partikulär- och totalfosfor demonstrerade att tvåstegsdiken med svämplan på båda sidor om vattenfåran minskade erosion och förbättrade vattenkvaliteten nedströms, jämfört med svämplan på en sida och trapezoidformade referensvattendrag. Denitrifikation i svämplanssediment stod för 33 % av total denitrifikation i åtgärdade vattendrag, styrt av översvämning av svämplan och nitratkoncentration i vattendrag. Vid högre översvämningsfrekvens ökade dock både denitrifikation och desorption av löst reaktivt fosfor från svämplanssediment, vilket innebär en avvägning mellan kväve- och fosforreduktion. Den överlag varierande påverkan av vattenkvalitet längs tvåstegsdiken, med högre potential för fosforreduktion jämfört med kväve, förklarades av underliggande höga utsläppsbelastingar och begränsningar av svämplansutformningen. För att uppnå förbättrad vattenkvalitet genom svämplansåtgärder i jordbruksvattendrag behövs platsspecifika, dominerande former av näringsämnen (kväve, fosfor och sediment) tas i beaktande med hjälp av lämplig svämplansutformning. Eftersom svämplan främst syftar till att begränsa mobilisering och transport inom vattendrag kan deras effekt därför döljas av förhöjda närings- och sedimentutsläpp från jordbruksmark.

Nyckelord: Vattenåtgärder, tvåstegsdike, denitrifikation, kväve, fosfor, vattendragsmetabolism, jordbruksvattendrag, övergödning

Preface

At the age of ten, me and a group of friends at a summer camp spent most of our awake time following a little stream that ran through the surrounding forest. Each day, we returned to the stream and went a little bit further upstream, sticking to our clear mission to hunt down the spring of the stream (and the extent of our liberties). As it turned out, we never discovered its source but ended up in an agricultural ditch, much to our disappointment. To this day, this memory has stayed with me and I still feel the same excitement of exploring and tracking down streams. Although in a more formalized fashion, I am as intrigued by the enigma of a stream, ever-changing and shape-shifting along its extent. Yet, owing to this thesis work, the previous disappointment of agricultural ditches has been replaced by a sustained fascination. By now, I wish I could tell the ten-year old me that these at first sight unremarkable, boring ditches are just as exciting as forest streams, indirectly influencing the possibilities to swim, drink and sustain life of humans and animals.

Dedication

Till morfar

”Algerna blommar det är otjänligt vatten
dyker och hoppas få tillbaka på skatten”

- Rockbotten, David Ritschard

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Hallberg, L., Djodjic, F., Bieroza, M. (2023). Phosphorus supply and floodplain design govern phosphorus reduction capacity in remediated agricultural streams. *Hydrology and Earth System Sciences*, 28(2). doi: 10.5194/hess-28-341-2024

II. Hallberg, L., Hallin, S., Bieroza, M. (2022). Catchment controls of denitrification and nitrous oxide production rates in headwater remediated agricultural streams. *Science of The Total Environment*, 838, 156513. doi: 10.1016/j.scitotenv.2022.156513

III. Hallberg, L., Hallin, S., Djodjic, F., Bieroza, M. (2024). Trade-offs between nitrogen and phosphorus mitigation with floodplain remediation in agricultural streams. (in review)

IV. Hallberg, L., Bernal, S., Bieroza, M. Flow-driven oscillations in supply and demand ratios of nitrate and carbon in an agriculturally impacted stream. (manuscript)

Papers I-II are reproduced with the permission of the publishers.

The contribution by Lukas Hallberg to the papers included in this thesis was as follows:

I. Planned the study in collaboration with co-authors. Main responsibility for field data collection (from 2020), lab work and data analysis. Wrote the manuscript with support from co-authors.

II. Planned and designed the study in collaboration with co-authors. Main responsibility for field data collection, lab work and data analysis. Wrote the manuscript with support from co-authors.

III. Planned and designed the study in collaboration with co-authors. Main responsibility for field data collection (from 2020), lab work and data analysis. Wrote the manuscript with support from co-authors.

IV. Planned and designed the study in collaboration with co-authors. Main responsibility for field data collection. Performed the modelling and data analysis. Wrote the manuscript with support from co-authors.

Additional papers

During the thesis work, Lukas Hallberg made contributions to the following papers, not included in the thesis:

Bieroza, M., Acharya, S., Benisch, J., Ter Borg, R. N., **Hallberg, L.**, Negri, C., Pruitt, A., Pucher, M., Saavedra, F., Staniszewska, K., van't Veen, S, G, M., Vincent, A., Winter, C., Basu, N. B., Jarvie, H.P., & Kirchner, J. W. (2023). Advances in Catchment Science, Hydrochemistry, and Aquatic Ecology Enabled by High-Frequency Water Quality Measurements. *Environmental Science & Technology*, 57(12), 4701-4719. doi: 10.1021/acs.est.2c07798

Bieroza, M., **Hallberg, L.**, Livsey, J., Prischl L, A., & Wynants, M. (2024). Recognizing Agricultural Headwaters as Critical Ecosystems. *Environmental Science & Technology*, 58(11), 4852-4858. doi: 10.1021/acs.est.3c10165

Abbreviations

C	Carbon
CS	Control stream
DIN	Dissolved inorganic nitrogen
DOC	Dissolved organic carbon
DOM	Dissolved organic matter
DS	Downstream
EPC_0	Equilibrium phosphorus concentration
ER	Ecosystem respiration
Fe	Iron
GPP	Gross primary production
K_{600}	Oxygen gas transfer velocity
MS	Midstream
N	Nitrogen
N_2	Nitrogen gas
NEP	Net ecosystem productivity
NH_4^+	Ammonium
N_2O	Nitrous oxide gas
NO_3^-	Nitrate
O_2	Oxygen gas

P	Phosphorus
PCA	Principal component analysis
PEP	Phosphorus exchange potential
PP	Particulate phosphorus
SRP	Soluble reactive phosphorus
SS	Suspended solids
TP	Total phosphorus
US	Upstream

1. Introduction

By the turn of a tap, we consume and expose ourselves to water that has travelled a far and long journey, water that carries the fingerprint of the landscape it has encountered on its way. Albeit a resource almost considered to be free and ever-abundant in some areas, clean freshwater is not generated *ex nihilo* but is the product of its interaction with its surrounding environment, including soils, climate and biota. As human activities continue to alter ecosystems to satisfy the increasing demands for food, energy and other natural resources, pressures on water resources have also risen dramatically in response (Elser & Bennett, 2011; Fowler et al., 2013). Artificial draining and intensified fertilization of cropland have together increased crop production far beyond pre-industrial levels and secured food production to a growing population (Blann et al., 2009). However, this has come with an environmental cost – the resulting loss of waterbodies, nutrient enrichment and erosion have accelerated degradation of water quality and habitats, from the smallest creeks to coastal seas (Dahl, 2000; Skaggs et al., 1994).

The two macronutrients nitrogen (N) and phosphorus (P) are essential building blocks of all biological processes and are integral as fertilizer inputs to agricultural cropping systems for the production of food and energy. Transfer of these nutrients from agricultural soils to freshwaters has detrimental consequences for functioning and integrity of aquatic ecosystems by alleviating the nutrient limitation of primary producers, like algae and other autotrophs (Conley et al., 2009). In many freshwater ecosystems, P is often the primary limiting nutrient for autotrophs but co-limitation with N has also been shown to occur seasonally, motivating efforts to mitigate N and P simultaneously (Glibert et al., 2014; Jarvie et al., 2018b; Shatwell & Köhler, 2019). Elevated nutrient concentrations in waters, i.e., eutrophication, have cascading effects on algal growth and oxygen depletion. The highly modified hydrology of agricultural catchments has further increased mobilization and erosion of sediments, driving P transport (Withers & Jarvie, 2008) and negatively affecting stream biota (Jones et al., 2012). Together, increased nutrient and sediment exports from agricultural catchments have impaired aquatic ecosystem functions and services across the freshwater continuum (Smith et al., 1999). Poor water quality is further propagated into societal impacts, manifested in restricted cultural and recreational use of waters as well as higher treatment costs for production of drinking water (Dodds et al., 2009).

To date, eutrophication is a widespread phenomenon occurring on a global scale and its causes can be traced back to the modernization and intensification of agriculture during the Green Revolution[1] (Foley et al., 2005). During the last 70 years, sustained anthropogenic inputs of N and P have been recognized as a key disruptor of global biogeochemical cycles in aquatic environments (Galloway et al., 2003; Haygarth et al., 2014) and in response, generated important scientific advances aiding in successful eutrophication management (Basu et al., 2022; Bol et al., 2018; Sharpley et al., 2013). Initial successes in mitigating pollution from easily identifiable industrial and urban point sources have put diffuse pollution from agricultural soils at the forefront as the primary source of eutrophication (Bijay-Singh & Craswell, 2021). Despite intensified efforts and implementation of regulatory water quality targets such as the EU Water Framework Directive, attempts to mitigate diffuse pollution have often failed to meet expectations and poor water quality persists around the world (Jarvie et al., 2013; Lintern et al., 2020; Meals et al., 2010). Diffuse pollution originates mainly from agricultural activities and exerts strong negative effects on water quality in both inland surface waterbodies and marine environments (Hansson et al., 2019). In the three Swedish water districts (North Baltic Sea, Skagerrak and Kattegat, South Baltic Sea) wherein the majority of the riverine systems in agricultural land is located, only 19 % of the surface water bodies reach the target of good ecological status (Water Authorities, 2023). To successfully improve the water quality, it is necessary not only to consider past and current levels of nutrient inputs but also to plan

[1] The Green Revolution signifies a period of technological modernization of agriculture, beginning in 1950s, wherein advances in crop breeding and the introduction of mineral fertilizers and pesticides led to unprecedented increases in food production.

for future increases in nutrient delivery. The combined effect of a warmer climate with more frequent occurrences of droughts and flood events, together with mobilization of the legacy of nutrients from historic fertilization are predicted to accelerate nutrient and sediment delivery from land to water (Basu et al., 2022; Ockenden et al., 2017).

Current water management in agricultural landscapes is up against significant challenges. The delicate task at hand is to strike a balance between demands for food production and the provision of water purification in the landscape, without jeopardizing current yield levels (Ellis et al., 2019). This task, being the central motivation behind this thesis, is addressed by exploring the potential for reducing the delivery of nutrients and sediments in Swedish agricultural streams and ditches by floodplain remediation. This is a stream-integrated measure, also known as a two-stage or compound channel (hereafter referred to as floodplain remediation). The targeting of both field and tile drainage runoff together with instream sources of pollution distinguishes floodplain remediation from other measures such as wetland construction and cover crop cultivation. The efficacy of floodplain remediation as a mitigation measure is synthesized in this thesis using hydrological and biogeochemical approaches, focusing on fluvial stability and particulate P mobilization (**Paper I**), N removal (**Paper II**), dissolved P cycling and its interaction with N removal (**Paper III**) and linkages between hydrochemical and stream biology processing (**Paper IV**).

2. Aim and objectives

The overarching aim of this thesis was to investigate the hydro- and biogeochemical functions of floodplain remediation in Swedish agricultural headwater streams and to evaluate its effectiveness in mitigating eutrophication and instream erosion. Responses to floodplain remediation were studied across catchments differing in hydrology, soil type and land use. Each remediated stream was compared to an upstream trapezoid-shaped channel, a typical design for modified agricultural streams and ditches in the Swedish landscape. The study was undertaken in ten diverse catchments across two contrasting regions in Sweden to determine the capacity for N, P and sediment removal, and to evaluate hydrological, geomorphological and biogeochemical processes behind observed water quality responses (Figure 1). The thesis was guided by the following research questions:

I. Can floodplain remediation promote sediment deposition and reduce sediment and particulate P mobilization from stream channels and banks, compared to trapezoidal channels? (**Paper I**)

II. Does floodplain remediation support higher N removal rates, compared to trapezoidal channels? (**Paper II**)

III. Is simultaneous biogeochemical removal of dissolved N and P achievable with floodplain remediation? (**Paper III**)

IV. What are the seasonal patterns in stream metabolism in a remediated stream and are they linked to hydrochemical responses? (**Paper IV**)

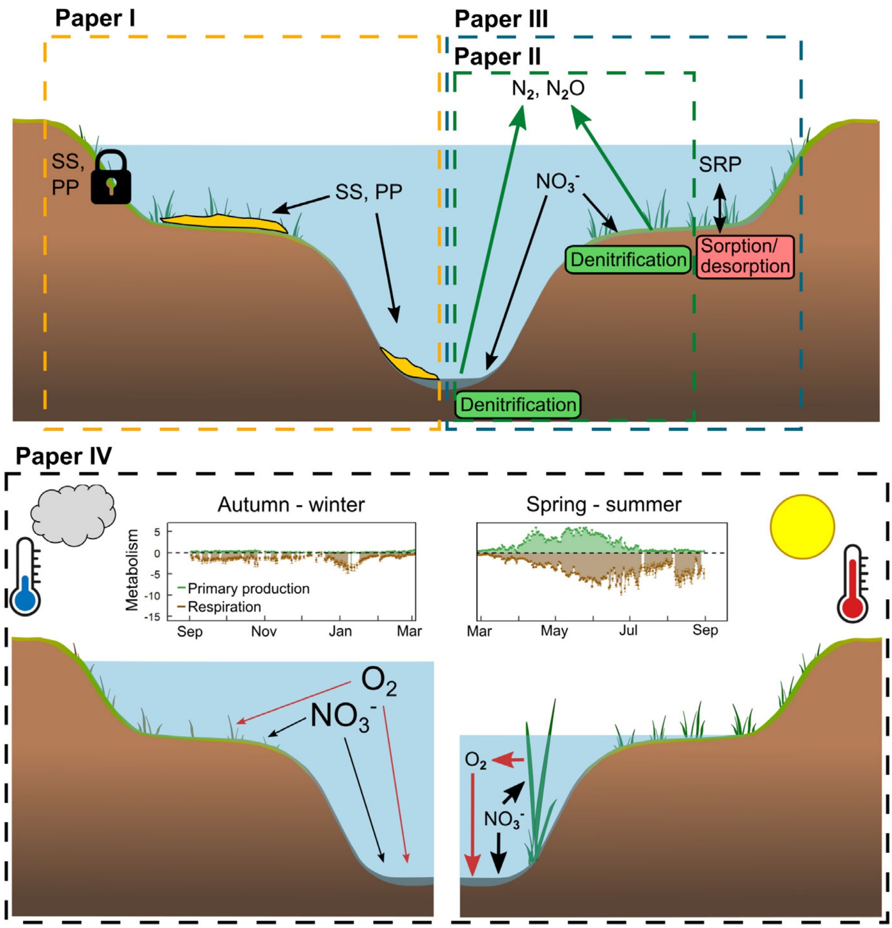


Figure 1. Conceptual profile of a stream subjected to floodplain remediation and the key processes studied in papers I-IV, including the interlinkages between the papers. N_2 = nitrogen gas, N_2O = nitrous oxide, NO_3^- = nitrate, O_2 = dissolved oxygen, PP = particulate phosphorus, SRP = soluble reactive phosphorus, SS = suspended solids.

3. Background

3.1 Nutrient and sediment mitigation in agricultural catchments

3.1.1 The diffuse pollution transfer continuum

To improve water quality and alleviate environmental impacts of eutrophication and erosion, it is necessary to identify the major contributing sources of N, P and sediments as well as their pathways in the landscape across space and time. Inputs of these pollutants to catchments can be broadly divided into either point or diffuse sources. Point sources are discrete effluent points that can have a disproportional negative effect on their immediate environment by discharging high nutrient loads directly into sensitive ecosystems (Jenny et al., 2016; Kiedrzyńska et al., 2014). These include domestic sewage pipes, wastewater treatment plants and farmyards but due to their spatially constrained coverage, targeted measures have successfully reduced much of their negative impact on aquatic ecosystems (Gustafsson et al., 2012; Neal et al., 2010). However, diffuse, non-point sources have proven to be far more challenging to identify and mitigate, and are therefore the focal point of contemporary water management (Jarvie et al., 2013). The origin of diffuse pollution is often strongly linked to agricultural land use due to excess application of nutrients for crop cultivation combined with soil management leading to erosion (European Environmental Agency, 2021). Atmospheric N deposition from fossil fuel combustion also contributes to diffuse pollution, but technological improvements have reduced its share to the overall terrestrial pool of reactive N (Vet et al., 2014).

Diffuse nutrient sources are scattered over vast areas of agricultural land and are mobilized and delivered to streams by seasonal hydrological events that vary in frequency and intensity (Mellander et al., 2018). The spatial heterogeneity of catchment topography, soil morphology and land use give rise to significant time lags in delivery from sources to streams. This can delay the detection of water quality improvements from measures and fail expectations of immediate success (Meals et al., 2010). The fate of nutrients and sediments in agricultural catchments can be conceptualized as a transfer continuum, wherein source, mobilization, delivery and impact represent specific processes in the landscape that regulate water quality (Haygarth et al., 2005). The transfer continuum allows for connecting disparate, localized pollution pathways and guides a systematic catchment-scale approach to nutrient and sediment mitigation. This framework can assist the planning of measures from farms to waterbodies, e.g., improved nutrient management, cover crops, controlled drainage, wetland construction and stream remediation. Further, cost-efficient mitigation planning relies on the identification of locations that contribute disproportionally to nutrient and sediment transfer in the catchment, known as critical source areas (Pionke et al., 2000).

To date, numerous mitigation efforts have been focused on in-field and edge-of-field measures, reflected by available funding and economic incentives of farmers. However, there are rising concerns that mobilization of legacy nutrient sources in soil, groundwater, stream bed and banks contribute with a significantly higher proportion of total exports than previously suggested (Margenot et al., 2023; Sharpley et al., 2013; Van Meter et al., 2018). Legacy sources represent a potential missing link and partial explanation of the mixed success of measures intercepting field runoff (Bieroza et al., 2021; Bol et al., 2018; Lintern et al., 2020), which typically do not target nutrients and sediments already present in the stream corridor. It is therefore imperative to venture beyond field measures and develop our understanding of the potential for stream remediation to reduce anthropogenic pressures on aquatic ecosystems.

3.1.2 Headwaters - biogeochemical hotspots

Stream networks draining agricultural landscapes are dominated by lower order headwaters that represent the first point of contact between terrestrial and aquatic ecosystems. As such, they are the gateways and transport pathways for most diffuse pollutants originating from agricultural land (Abbott et al., 2018). However, headwater streams are not merely passive conduits of solutes and particles. Their extensive terrestrial connectivity can support high rates of biogeochemical processing, rendering them vital for reducing nutrient and sediment inputs and safeguarding water quality in downstream ecosystems (Bieroza et al., 2024). By physical, chemical and biological means, headwaters have the capacity to either permanently or temporarily remove constituents as well as transform their speciation and availability (Falk et al., 2023; Seybold & McGlynn, 2018).

The potential for instream processing, be it of sediments, nutrients or carbon (C), is strongly regulated by the hydrological regime. This is recognized in the Pulse-Shunt concept (Raymond et al., 2016) that accounts for the influence of storm flows and hydrological disturbances on stream networks' processing capacity. It proposes that episodic events are important modifiers that give rise to large inputs to headwaters (pulse), combined with rapid downstream transport due to increased flow (shunt). In this sense, the concept incorporates the effect of hydrological variability, unaccounted for in previous models of stream network dynamics (e.g., the River Continuum Concept; Vannote et al., 1980). Building on the central tenet of hydrology as key control of instream processing, the role of uptake/removal dynamics (i.e., spiraling theory; Newbold et al., 1981) and its balance in relation to inputs is further developed in the River Saturation Concept (Wollheim et al., 2018). This general framework, using supply and demand ratio of water constituents, has allowed for prediction and partitioning of biogeochemical processing in stream networks. It further highlights the disproportionate influence of headwaters for determining the downstream water quality signature.

Stream network processing is not solely confined to stream channels but also extends into transitional riparian and hyporheic zones, characterized by intermittent hydrological connectivity, that closely link terrestrial and aquatic ecosystems. Despite their limited areal coverage, riparian zones have been recognized as important ecological interfaces that support both higher biological productivity and biodiversity in relation to surrounding terrestrial biomes (Singh et al., 2021). From the point of hydrochemistry, riparian interfaces can act as buffers that through increased water residence time and bioreactivity retain and transform diffuse pollution from both landscape and stream sources (Wohl, 2021). However, this capacity is strongly dependent on the extent and structure of riparian zones, which can vary greatly from narrow grass buffer strips to densely vegetated floodplains and wetlands. Despite the existing focus on the use of riparian zones as interfaces for nutrient mitigation, concerns have been raised around the lack of understanding of interlinkages between the cycling of multiple water constituents (e.g., C, N and P) in riparian zones (Stutter et al., 2023). This highlights the need for assessing synergies and trade-offs as well as the risk of pollution swapping (Stevens & Quinton, 2009) when targeting specific pollutants with riparian interventions.

3.1.3 Untapping the potential for nutrient and sediment retention and removal in agricultural streams

For centuries, the backbone of agricultural water management in rain-fed, temperate areas has consisted of efficient removal of excess water from cultivated fields. To achieve this, natural stream networks have been reconfigured and short-circuited in agricultural areas around the world, mainly by digging open ditches, straightening and channelizing existing streams (Simon & Rinaldi, 2006) and installing subsurface tile drainage systems (King et al., 2015). Channelized streams, a term describing trapezoidal-shaped and narrow channels, often run straight in the landscape and are designed to occupy minimal agricultural land. Unfortunately, channelized streams in agricultural catchments have also proven to be efficient conduits for nutrients and sediment transport, due to their short water residence times and limited riparian areas that restrict nutrient and sediment retention (Booman & Laterra, 2019). Although eutrophication has grown into a longstanding issue and generated a suite of actions against water pollution, many channelized streams have remained unaltered (Blann et al., 2009).

By capitalizing on the current understanding of controls of stream nutrient transformations (Wollheim et al., 2018), it is evident that many agricultural streams with their unique position in the landscape represent key interfaces with an untapped potential for nutrient and sediment retention and removal. These headwater streams are recipients and conduits of nutrient inputs and can thereby act as ecosystem control points. However, it has been proposed that the reluctance to modify existing channels by farmers stems from the risk of jeopardizing drainage function, crop yields and farm income (Rhoads et al., 1999). In addition, measures that infringe on agricultural land will also result in higher opportunity costs. Any channel modification is therefore constrained to balance nutrient retention capacity with maintained drainage function and minimal land put out of production. A measure that has been proposed to meet these demands is the construction of lateral floodplains along existing stream channels (Figure 2). Floodplain remediation originated as an effort to reduce channel maintenance, by mimicking benches or floodplains formed in natural unmanaged streams that exhibits long-term fluvial stability and limited stream bed deposition (Landwehr & Rhoads, 2003; Powell et al., 2007). With floodplain remediation, this state is reached in a controlled way by excavating floodplains from bank slopes.

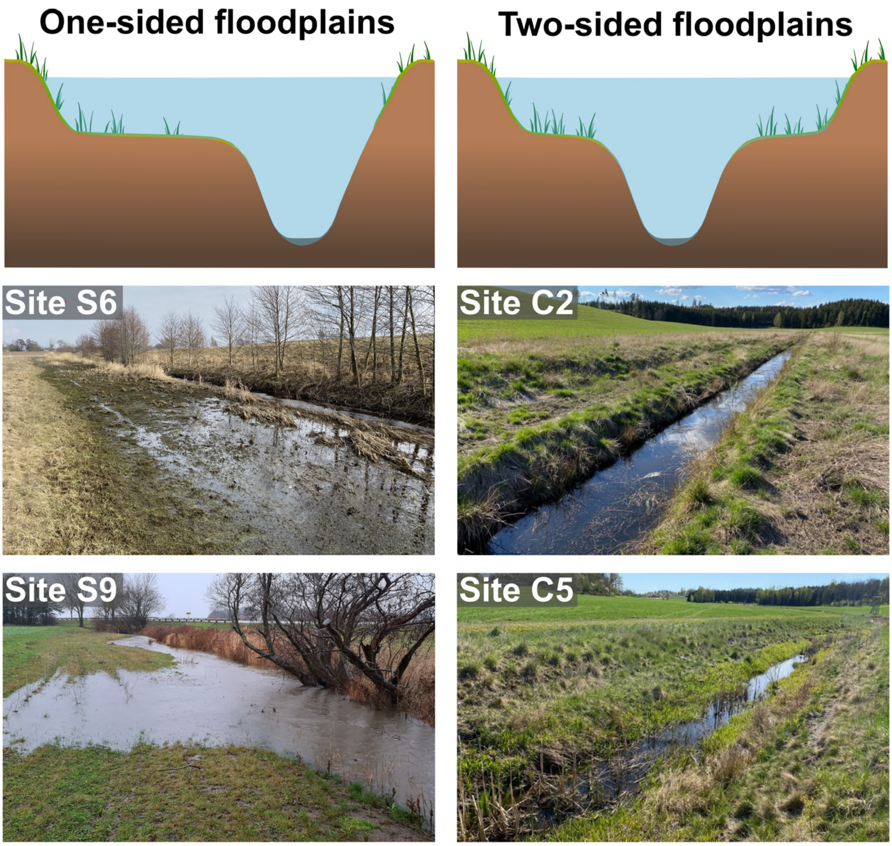


Figure 2. Cross-section profiles of floodplain remediation with one-sided and two-sided floodplain designs. Four floodplain remediated streams included in papers I-III with one-sided floodplains (sites S6 and S9) and two-sided floodplains (sites C2 and C5). Photographs by Lukas Hallberg. Adapted from paper III.

The purpose and aim of this riparian expansion have since extended to also leverage water and nutrient retention (Bukaveckas, 2007; Mahl et al., 2015; McMillan & Noe, 2017). However, to fully realize the potential for water quality improvement with floodplain remediation, it is necessary to further assess the influence of catchment variability as well as different floodplain designs on processes that govern nutrient and sediment export.

3.2 Fluvial stability and stream geomorphology

3.2.1 Channel evolution

Unmanaged natural streams are shaped by fluvial processes that give rise to lateral depositional zones (benches or floodplains) and an incised channel that is capable of transporting sediments during lower flows (Ward et al., 2003). Natural streams are thus reaching fluvial equilibrium over time, as fluvial processes balance out sediment supply, storage and transport. Upon channelization of a stream, an imbalance is often introduced between sediment supply and transport, resulting in an unstable system that either accumulates or erodes sediments. If left unmaintained, trapezoid-shaped channels can result in changes in stream power (i.e., transport capacity) in relation to sediment supply and consequently contribute with sediment loads to such a degree that they dominate the total catchment exports (Simon & Rinaldi, 2006). In underpowered, low gradient headwaters, channelization can also cause depositional responses that threaten hydraulic drainage capacity. In addition, changes in transport capacity also have cascading effects on upstream and downstream sediment transport dynamics (Brookes, 1987).

Gradual fluvial adjustments that occur after channelization are conceptualized in the channel evolution model as a sequence of development stages (Schumm et al., 1984; Simon, 1989). This model can be applied to assess the past, present and future impact from interventions on channel geomorphology. Posterior to channelization, changes in channel geomorphology often commence with channel bed incision and subsequent lateral instability with toe scour and bank failures, resulting in widening of the channel. In the wider channel, the capacity for fluvial transport of eroded bank material gradually decreases so that lower benches or floodplains are formed, establishing a fluvial equilibrium (Simon, 1989). The slow and uncontrolled establishment of fluvial equilibrium in channelized streams, occurring on decadal timescale (Landwehr & Rhoads, 2003), can be circumvented by constructing floodplains directly where they would form by natural fluvial processes (Powell et al., 2007). Such efforts have been shown to reduce bank erosion and equilibrate imbalances in sediment transport, with stable floodplains over decades (D'Ambrosio et al., 2015; Krider et al., 2017). Increased water conveyance capacities with the greater cross-section volume from floodplain excavation further reduce the risk of overflow onto adjacent fields. The transferability of these benefits to clay soils with erosion-prone banks (Peacher et al., 2018; Thorne & Tovey, 1981) remains unexplored but are of critical interest in Swedish agricultural catchments, dominated by fine textured soils. Likewise, fluvial stabilization with different floodplain designs (e.g., two-sided vs. one-sided floodplains) is yet to be tested.

3.2.2 Channel maintenance and management

Agricultural drainage interventions have predominantly relied on approaches that restrict rather than enhance water and pollution retention processes, with the underpinning notion that channelized streams require recurring maintenance to preserve their function (Dollinger et al., 2015). In channels, this is accomplished by recurring bed sediment removal and bank reprofiling. Therefore, the progression of channel evolution is routinely halted through interventions to restore the initial trapezoidal form (King et al., 2015). Bed sediment dredging, bank reprofiling and instream macrophyte cutting restore drainage capacity and remove nutrients associated to sediments and biomass. Although sediment dredging may reduce downstream nutrient exports, increased release of P has been observed immediately after dredging (Smith et al., 2006). It can have further detrimental impact on biodiversity and stream biota (Shaw et al., 2015) and represents a cost-inefficient practice for nutrient and sediment removal as sensitive streams may require clean out down to every five years (Posthumus et al., 2015). By introducing floodplain interfaces to channelized systems it has been proposed that the need for maintenance can be drastically lowered or even be redundant (D'Ambrosio et al., 2015; Krider et al., 2017; Västilä et al., 2021). This is supported by observations of a self-cleaning channel that redistributes deposited sediments to floodplains. However, the effect is likely to be highly site-specific and it is unclear if it is a generalizable process.

3.3 Nutrient cycling in streams

3.3.1 Nitrogen removal by denitrification

Inputs of bioavailable, inorganic N are high in agriculturally impacted streams and are often dominated by nitrate (NO_3^-), followed by lower proportions of ammonium (NH_4^+). This is explained by the greater water solubility, lower biological reactivity and lower affinity to soil minerals of NO_3^- compared to NH_4^+ (Follett & Delgado, 2002) as well as rapid microbial oxidation of NH_4^+ to NO_3^- (Norton & Ouyang, 2019). In streams, inorganic N is subjected to a complex of biological processes that either transform and transiently store N or permanently remove it as gaseous N products (Galloway et al., 2003). Apart from biotic assimilation, the main process is microbial denitrification which is the primary pathway for permanent N removal in streams. Denitrification is also closely coupled to oxidation of NH_4^+ to NO_3^- via microbial nitrification, representing a source of NO_3^- that often operates in tandem with denitrification. Another important N sink is the biotic assimilation of N to instream primary producers and microbes that seasonally retain N in biomass.

Denitrification is defined as the microbial reduction of NO_3^- to gaseous N under oxygen limited conditions, most often using organic C as electron donor. It is performed by a diverse group of bacteria and archaea that are facultative anaerobes as well as some fungi, sequentially reducing NO_3^- to nitrite, nitric oxide, nitrous oxide (N_2O) and finally dinitrogen gas (N_2). If the final step is terminated, either by environmental suppression of N_2O reducing enzymes or lack of genetic capacity for enzyme synthesis, N_2O is released (Philippot et al., 2007). Outgassing of N_2O and N_2 into the atmosphere enables the permanent removal of inorganic N, attenuating downstream eutrophication impacts. Denitrification in streams and rivers is estimated to remove up to 20 % of N inputs along reaches, which have implications for the global N cycle as cumulative stream network removal accounts for 13 % of all terrestrial N sources (Seitzinger et al., 2006). However, incomplete denitrification resulting in N_2O emissions has severe negative implications for the global climate since N_2O is a potent greenhouse gas with high warming potential (Forster et al., 2007) and also leads to the destruction of the stratospheric ozone layer (Ravishankara et al., 2009). Instream N_2O production is linked to elevated NO_3^- concentrations and agricultural streams have correspondingly been identified as hotspots for

N_2O production (Beaulieu et al., 2011; Yao et al., 2020). Any measure targeting stream denitrification should therefore also consider the effect on N_2O yields in relation to N_2 production to safeguard against environmental trade-offs.

Denitrification generally occurs in bed sediments, which in lotic systems are characterized by a sharp redox boundary at the sediment-water interface (Seitzinger et al., 2006). Due to oxygen diffusion from the water column, the uppermost sediment layer is aerobic and supports production of NO_3^- through the microbial process nitrification. When NO_3^- is supplied to anoxic sediment layers below the oxic interface, either through stream or groundwater flow, denitrification typically occurs. Beyond anoxia and NO_3^- supply, organic C quantity and composition are also regulating denitrification rates (Taylor & Townsend, 2010) and in some instances N_2O yields (Barnes et al., 2012). Sediment denitrification is spatially heterogeneous, occurring in microsites that are modified by water residence time and hydrology (Seitzinger et al., 2006). Denitrification hotspots can be further amplified by the presence of macrophyte roots that extend the redox boundary vertically and also release labile organic C through root exudates. In addition, biotic N assimilation has also been shown to be coupled with denitrification in time, wherein a release of inorganic N from biomass after growing season can be followed by increased denitrification rates (Jarvie et al., 2018a).

Denitrification dynamics in floodplain sediments differs from that of stream sediments because it is controlled by intermittent inundation, driven by stream water flooding, field runoff or tile drainage discharge. During these events, it has been demonstrated that constructed floodplains in agricultural streams increase N removal through denitrification (Mahl et al., 2015; Roley et al., 2012a; Speir et al., 2020) and produce N_2O yields comparable to that of stream sediments (Dee & Tank, 2020). However, it is currently unknown how N removal and dynamics in N_2O yields from floodplains translate to a wider range of catchments and underlying soil textures.

3.3.2 Physical and geochemical phosphorus cycling

Stream water P can be broadly divided into soluble reactive P (SRP) and particulate P[2] (PP), which together with P associated to colloidal particles form total P (TP; Chen & Arai, 2020). The SRP fraction is considered bioavailable to aquatic organisms, while only part of the PP fraction, which is associated to suspended solids (SS), is immediately available for assimilatory uptake (Muscarella et al., 2014). There is a considerable exchange between the SRP and PP fractions in stream water due to geochemical sorption/desorption of SRP to SS, which continually alter the proportion of P forms transported in streams. SS are either inorganic minerals or organic substances that originate from distal sources by soil erosion (Djodjic & Villa, 2015) or proximal sources such as bank erosion and bed sediment resuspension (Fox et al., 2016). The form of solids determines the affinity to P and consequently the bioavailability and reactivity of the associated P (Ballantine et al., 2009; Sandström et al., 2021). In particular, catchments dominated by clay soils have been identified as high contributors of PP due to the close association of P with fine clay minerals (Sandström et al., 2020). Hence, SS are important vectors of P and the key focus of numerous efforts to reduce P exports and its eutrophication impact (Bol et al., 2018). Besides the role in P transport, SS also have a direct physical impact that negatively affects stream biota (Davis et al., 2018).

Once mobilized and delivered from land to stream, P in both dissolved and suspended forms is repeatedly intercepted and remobilized by physical, chemical and biological processes. Over time, this results in the accumulation of large P stores in sediments across agricultural stream corridors, termed sediment legacy P (Fox et al., 2016; Margenot et al., 2023; Wohl, 2015). The shift of stream ecosystems from P sinks to sources and the generation of internal P loading have complicated eutrophication management and prompted the need for an improved understanding of internal P cycling in streams (Withers & Jarvie, 2008).

In channelized streams, the stream bed is the primary interface for P cycling as it receives P-enriched SS that settle from gravitational forces. Stream beds also provide transient storage of SRP through geochemical sorption/desorption exchange. Concerning biotic P cycling, various forms of primary producers and bacteria assimilate P and many of these organisms are

[2] Particulate P is traditionally defined as P associated with solids > 0.45 µm (Shand et al., 2000), which can be estimated as the difference between unfiltered and filtered total P with a 0.45 µm filter membrane.

abundant in bed sediments where high stocks in organic matter provide energy and nutrients (Battin et al., 2016). This gives rise to a tight coupling between biotic and abiotic P removal, as desorbed SRP can be assimilated and mineralized by biota, further regulating the timing and magnitude of P delivery (Simpson et al., 2020).

Phosphorus sedimentation

By the process of sedimentation, large proportions of P-enriched particles delivered to streams can be retained during receding flows that allow SS to settle on stream beds (Ballantine et al., 2009). Deposited sediments are temporary storages which can be reversed either by physical resuspension, bioturbation or via geochemical release (Wohl, 2015). Conditions for P sedimentation arise in streams with high P loading and limited stream power, wherein the latter is a function of water velocity, particle size and stream geomorphology (Worrall et al., 2020). Accordingly, agricultural headwater streams draining fine textured land with flat topography have been identified as hotspots for P accumulation (Ballantine et al., 2009). Sedimentation is actively promoted to intercept P when constructing various forms of flow-through wetlands (Djodjic et al., 2020) and stream impoundments (Littlejohn et al., 2014), which reduce hydraulic load (runoff per wetland/reach area) and stream power.

Phosphorus sorption to sediments

The primary control of SRP dynamics is through sorption/desorption to redox-sensitive iron oxides in stream sediments, active during low flows (House, 2003). Aerobic conditions in the upper sediment layer oxidize ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) which can readily bind SRP that diffuses into sediment pore water. If sediments switch to anaerobic conditions, due to hydrological disturbance or biotic oxygen depletion, this process is reversed and SRP is released. Clay minerals and aluminium oxides can also be important contributors to P sorption in fine-textured stream beds (Gérard, 2016), but the available binding sites are often dominated by amorphous Fe oxides with higher reactivity, forming labile Fe-P associations (Simpson et al., 2022). In calcareous streams draining landscapes with chalk and karst geologies, this is complicated by the co-precipitation of P to calcium carbonates that under neutral pH can surpass P removal rates by sorption (House, 2003). Following sustained SRP loading, sorption capacity diminishes as sediment binding sites gradually saturate with P (Withers & Jarvie, 2008). Hence, measurements of P sorption potential and distribution of P pools in sediments serve as a key indicator that can be used to inform both the potential for P storage as well as the risk of P transfer (Jan et al., 2013; Lannergård et al., 2020; Simpson et al., 2021).

Phosphorus cycling in floodplains

To address the issue of internal P loading from streams, it is critical to implement mitigation measures that target both particulate transport and geochemical cycling of P. Constructed floodplains have been proposed to accommodate these needs by intercepting and accumulating P-rich sediments (Davis et al., 2015; Mahl et al., 2015; McMillan & Noe, 2017) and propagating successive SRP retention through coupled sorption and biotic uptake (Trentman et al., 2020). It has been suggested that floodplains reduce downstream export of P-rich bed sediments by promoting lateral transfer of bed sediments to floodplains during inundation (D'Ambrosio et al., 2015; Krider et al., 2017). Prolonged storage of P in floodplains has also been predicted by the tight coupling between sediment P exchange and macrophyte assimilation of P (Trentman et al., 2020). Established floodplain vegetation is reported to yield higher organic matter content in floodplain sediments compared to stream sediments (McMillan & Noe, 2017; Speir et al., 2020), which can stabilize loosely-bound P (Kang et al., 2009). However, drastic shifts in redox conditions following intermittent inundation increase the risk of P release from floodplains (Preiner et al., 2020). It is therefore necessary to further explore potential benefits and risks for P storage and release with constructed floodplains and to link these processes to both N removal and effects on stream morphology.

3.4 Evaluating ecological functioning using stream metabolism

Despite instances of successful eutrophication mitigation, responses in ecological recovery are often mixed and lag behind nutrient reductions (Jarvie et al., 2013). This highlights the complexity in predicting and controlling eutrophication in systems impacted by long-term pollution, which needs to be further articulated to bridge the gap between policy expectations and achievable outcomes through water management (Harris, 2012). One approach to investigate controls of temporal mismatches between improvements in chemical and ecological functioning is the combined monitoring of nutrients and stream metabolism. By partitioning gross primary production (GPP) and ecosystem respiration (ER), stream metabolism is increasingly applied to study the ecological function and structure in aquatic ecosystems (Bernhardt et al., 2022) as well as the response to pollutants (Arroita et al., 2019). As such, stream metabolism is an integral ecosystem metric that contains information about shifts in trophic states, energetic balance and biotic processing capacity down to daily time steps (Dodds et al., 2009).

The improved accuracy of *in situ* sensors has enabled high-frequency monitoring of dissolved oxygen (O_2), among a suite of other water constituents, such as NO_3^- and dissolved organic carbon (DOC; Bieroza et al., 2023). Daily dynamics of O_2 concentrations, coupled with data on light availability, temperature and water depth, are used to estimate reach-scale stream metabolism (Odum, 1956). Bayesian modelling approaches allow for accurate estimation of stream metabolism rates by attributing variation to GPP, ER and gas exchange with atmosphere (Appling et al., 2018). This has resulted in recent proliferation in stream metabolism studies and the characterization of inter-regional metabolic regimes, controlled by climate and land use (Appling et al., 2022; Bernhardt et al., 2022). It has been shown that lotic systems in general are net C sources, as ER often surpass GPP, in contrast to terrestrial biomes (Battin et al., 2023).

In addition to ecological structure, stream metabolic rates can be linked to monitoring of water constituents (e.g., C, N, P and SS) to improve biological process understanding. Estimations of biotic organic C decomposition and inorganic N uptake derived from metabolic oxygen flux have provided key insights into C and N cycling in floodplains, partitioned between autotrophs and heterotrophs (Roley et al., 2014). However, there is a lack of understanding of the interactions between C, N and metabolism during storm events in floodplain systems. Likewise, the balance between C and N supply and biological demand remains unknown in streams with remediated floodplains.

4. Methodology

4.1 Study catchments

The ten catchments studied in **papers I-III** are located in central east ("Central East", C1-5) and south Sweden ("South", S6-10), representing small (< 45 km^2) headwater catchments draining low sloping or flat land (Figure 3; Table 1). The geographic distribution of these catchments is reflected in their broad range in soil texture and land use, wherein the Central East catchments (C1-5) are characterized by higher clay content and lower proportions of agricultural land use, compared to the South catchments (S6-10; Table 1). The agricultural land across all catchments is predominantly tile-drained, with cultivation of mainly winter and spring sown cereal crops and ley.

All ten catchments have been subjected to floodplain remediation along reaches adjacent to agricultural land, ranging from 320 to 1960 m in length. In each catchment, remediated reaches were paired with upstream trapezoid-shaped reaches of equivalent lengths to evaluate floodplain remediation according to a control-impact design, except for the remediated reach in site S7 which was not paired as it originates from a wetland. In **paper IV**, site S8 was selected based on its high impact of agricultural land, long floodplain reach and availability of long-term high-frequency data.

Floodplains have been constructed either on both sides of the inset channel (C1-3, and S8) or on one side with doubled width (C4, S6-7 and S9-10; Figure 2). Site C5 is the exception with both designs present along the reach. Floodplain designs vary also across sites in terms of elevation to channel bed and width.

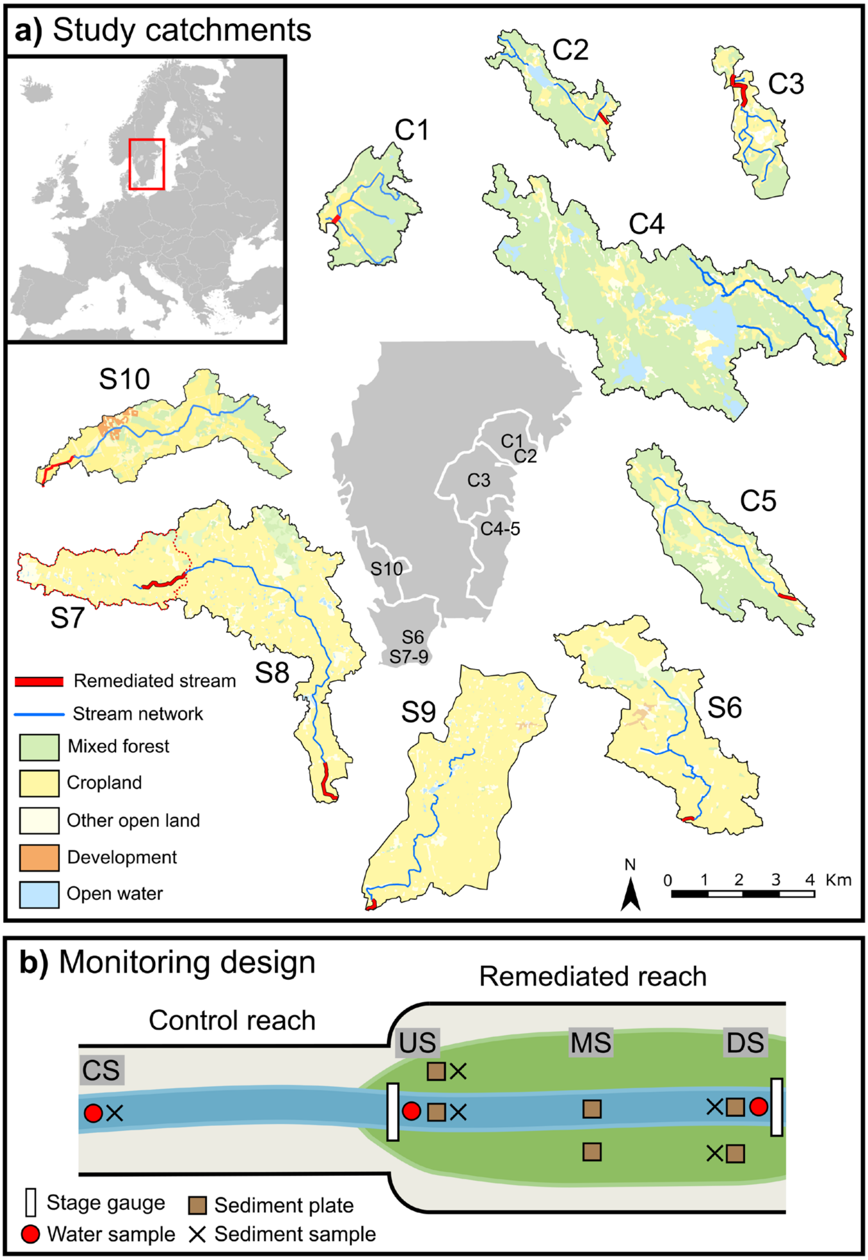


Figure 3. **a)** Locations of the ten study catchments in Central East (C1-C5) and South (S6-10) used in papers I-III, and catchment S8 used in paper IV. **b)** Schematic design of control-impact monitoring, with locations control stream (CS), upstream (US), midstream (MS) and downstream (DS) sampled between 2020 and 2023. Adapted from paper I.

Table 1. Characteristics of catchments and remediated reaches. Sites are ordered from north to south. Median discharge (Q_{50}) was calculated for April 2020 to December 2023, with the exception of site C3 where monitoring started in March 2018. Adapted from paper I.

Site	Area (km^2)	Agricultural land use (%)	Soil texture	Slope (-)	Q_{50} ($m^3\ s^{-1}$)	Year	Length (m)
Central East							
C1	9.8	15	Silty clay	0.0004	0.03	2012	340
C2	7.9	27	Silty clay loam	0.0004	0.02	2012	730
C3	6.6	70	Clay loam	0.003	0.01	2014	1500
C4	45.5	16	Clay loam	0.001	0.11	2019	320
C5	16.3	38	Clay loam	0.0014	0.06	2012	780
South							
S6	22.7	77	Loam	0.0017	0.08	2016	400
S7	10.8	81	Loam	0.0009	0.03	2013	1960
S8	42.4	81	Loam	0.0047	0.17	2013	1770
S9	31.0	86	Loam	0.0044	0.05	2019	630
S10	16.4	58	Sandy loam	0.0009	0.19	2014	1760

4.2 Hydrology and water chemistry monitoring

Combined monitoring of sub-daily hydrology and monthly water quality were performed in the ten study catchments between April 2020 and December 2022 (**Papers I-III**). In site S8, monitoring was continued until December 2023 (**Paper IV**). Due to channel reconfiguration by land managers, all sampling was terminated in site C5 in May 2022 and water quality sampling was terminated in the trapezoidal reach of site S9 in June 2021.

4.2.1 Hydrology and biophysical surveys

Continuous discharge (10 min intervals) was estimated in upstream (US) and downstream (DS) remediated reaches by establishing stage-discharge rating curves (**Papers I-IV**). Manual discharge measurements during five to eleven occasions were fitted to water stage measured with pressure sensors, using the power law:

$$Q = K(h + a)^p \quad (1)$$

where Q is discharge ($m^3 s^{-1}$), h is stage (m), a is stage at zero flow (m) and K and p are fitted parameters (Rantz, 1982). Continuous water stage time series were subsequently converted to discharge. Additional rating curves were calculated according to the velocity-area method (Herschy, 2014) for high discharges outside of measurement range to minimize overestimation.

To determine geomorphology of the remediated reach **(Papers I-IV)**, cross-section surveys with approximately 50 to 100 m interval were conducted using a GPS levelling device (E600 GNSS Receiver, E-survey; Ilao Åström, 2021). The derived floodplain elevation in relation to stream bed were subsequently combined with water stage measurements to estimate daily inundation frequency at US and DS locations of remediated reaches.

Vegetation cover percentage on floodplains was visually estimated in the locations US, MS and DS, using three square plots (1 m^2) at each location (Sakponou, not published). Vegetation was surveyed during summer 2021 and 2022 and classified into the four functional groups: Reeds, Grasses, Herbs and Bryophytes **(Papers I-IV)**.

4.2.2 Water sampling and analysis

Bottled water surface samples were collected monthly by local land managers upstream of trapezoidal reaches (control stream; CS) and at the locations US and DS of remediated reaches (**Papers I-IV**). Additional samples were collected during limited periods at midstream of remediated reaches (MS). Water samples were stored at 4 °C before analyzed at the accredited geochemical laboratory at SLU for NO_3^--N (ISO 15923-1:2013), NH_4^+-N (ISO 15923-1:2013), TP (SS-EN ISO 6878:2005), before and after 0.45 µm filtration, SRP (ISO 15923-1:2013), SS (SS-EN 872:2005) and DOC (SS-EN 1484). Particulate P (PP) was calculated as the difference between unfiltered and filtered TP. Water samples were further used to determine dissolved organic matter (DOM) properties optically using a spectrophotometer (Aqualog, Horiba). Fluorescence and absorbance indices (Huguet et al., 2009; McKnight et al., 2001; Ohno, 2002) indicating DOM origin and quality were calculated in Matlab (R2020a).

Stream water O_2 and pH were measured monthly to bimonthly, using optical and electrode-based sensors mounted on a handheld device (ProDSS, YSI).

4.2.3 High-frequency water quality monitoring and validation

A multiparameter sonde (EXO2, YSI), comprising sensors measuring NO_3^--N, fluorescent DOM, O_2, turbidity, specific conductance and temperature was deployed at DS of site S8 between July 2021 and December 2023 (**Paper IV**). Sensors measured their respective parameters at 15 min intervals and were periodically calibrated in laboratory or field. Due to sensor malfunction, no measurements were made between May and August 2022.

Measurements of fluorescent DOM (fluorescence emission intensity measured at excitation wavelength 365 nm) were converted to DOC by first correcting for temperature, turbidity and instrument-specific coefficients (Snyder et al., 2018), followed by calibration to measured DOC concentrations in water samples, using linear regression. Optical sensor measurements of NO_3^--N were corrected using default coefficients for turbidity and organic N within the EXO2 software (YSI, 2020). Corrected sensor NO_3^--N concentrations were then calibrated against NO_3^--N concentrations in water samples, using linear regression.

All high-frequency measurements were subjected to a two-step procedure for data validation: firstly, by removing biophysically unrealistic values and secondly, by removing outliers produced by sensor noise and biofouling. Outliers were detected by applying a 5-hour sliding window with Hampel filter and median absolute deviation, using the function *findOutliers* in the R package seismicRoll (Callahan et al., 2020).

4.3 Sediment sampling and analyses

4.3.1 Sediment sampling

Sampled sediments were distinguished into two different types: composite sediments and deposited sediments. Composite sediments refer to the upper 3-5 cm of intact surface sediments, representing the integration of short- and long-term sediment deposition. Deposited sediments denote sediments deposited on surfaces within six months. The timeline of sediment sampling in **papers I-III** is summarized in Figure 4.

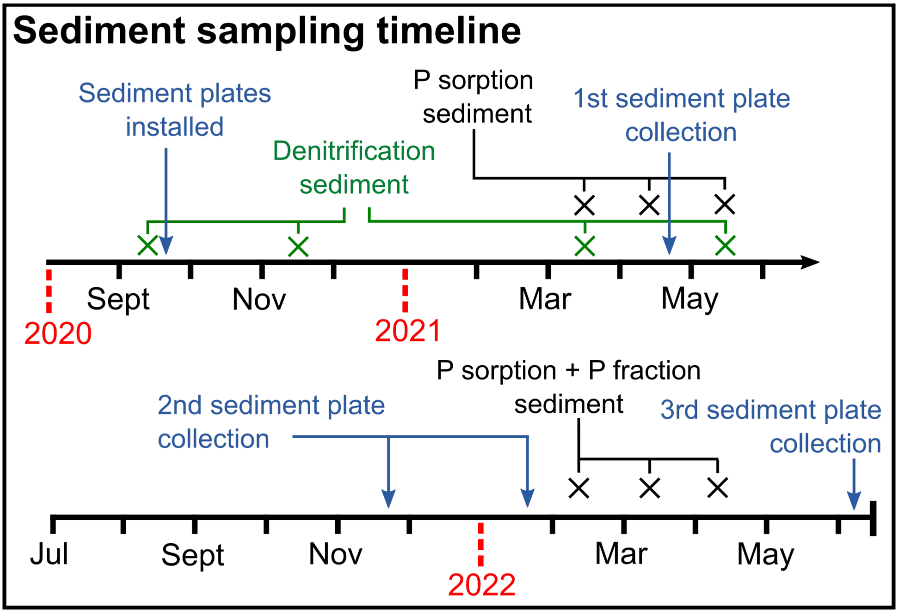


Figure 4. Schematic timeline of sediment sampling conducted in remediated reaches between September 2020 and June 2022. Adapted from papers I-III.

Sediment deposition rates were measured with 0.16 m^2 square wooden fiber plates, attached with a metal rod through its center to channel beds and floodplains at locations US, MS and DS (Figure 5; **Paper I**). Channel beds of sites S6 and S8-9 were not monitored, since their coarse beds could not be penetrated by the metal rods. Between September 2020 and June 2022, deposited sediments on plates were collected three times, approximately every six months (Figure 4). To minimize influence from plate edges, deposited sediments were collected from the inner 0.04 m^2. To collect intact sediment deposition from channel beds, a cylindrical frame was put on top of the plates before lifting it up from the channel to dewater sediments.

Composite sediments were sampled during three seasons to determine potential denitrification and P sorption (**Papers II-III**) and P fractions (**Papers I, III**). For denitrification measurements, channel sediments were collected at CS, US, MS and DS and floodplain sediments at US, MS and DS, during 2020 and 2021. Each sample consisted of 3 subsamples collected down to 3 cm depth (3 cm^3) using a trowel.

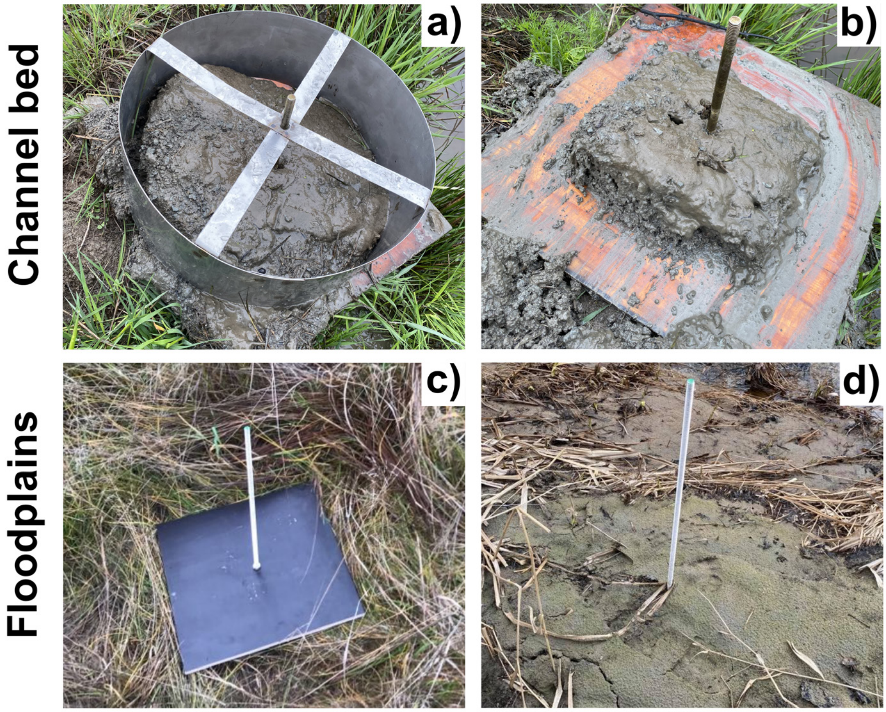


Figure 5. Plates for collection of deposited sediments installed on **a-b)** channel beds and **c-d)** floodplains. **a)** Cylindrical frame used for collecting channel sediments. **b)** Sampled plate area. Photos by Lukas Hallberg.

Sediments designated for P sorption experiments were sampled in 2021 from banks at CS, and floodplains at US, MS and DS, using 10 subsamples. In addition, sediments were sampled in 2022 from channels and floodplains at US and DS for determining P sorption and P fractionation, using 5 subsamples. Sediments for P analysis were sampled down to 5 cm (5 cm^3) using a trowel. All composite sediment samples were collected in airtight plastic bags, and after transport to laboratory stored at 4 °C until further analysis.

Fresh subsamples of all sediments were oven dried at 105 °C to determine dry matter. Composite samples for determining denitrification were subsequently analyzed for total C and N by dry combustion on a CN analyzer (Leco, TruMac). Deposited sediments were analyzed for TP with ICP-OES (Avio 200, PerkinElmer).

4.3.2 Denitrification and nitrous oxide production assays

Potential denitrification, hereafter mentioned as denitrification, was measured in composite sediments under anoxic conditions without substrate limitation, using the acetylene inhibition method (Pell et al., 1996; **Papers II-III**). This approach prevents conversion of N_2O to the end-product N_2 and relies on measuring the production of N_2O, thereby circumventing the issue of measuring N_2 under high ambient N_2 concentrations. In parallel, potential N_2O production rates were measured under the same conditions but without the addition of acetylene. Chloramphenicol was not added since this can inhibit not only *de novo* enzyme synthesis but also the activity of already existing enzymes (Pell et al., 1996). Potential N_2O and N_2 production rates were quantified in sediments collected during September 2020 and May 2021. Incubations were established following the procedure of Hellman et al. (2019), with substrate additions reaching final concentrations of 6 mg L^{-1} NO_3^--N and 7 mg L^{-1} C (glucose, acetate and succinate), corresponding to conditions where enzyme activity is rate-limiting. After incubation onset, gas was sampled at the interval 30, 75, 120, 150 and 180 min and gas concentration was subsequently measured with a gas chromatograph (Clarus-500, PerkinElmer). Denitrification/N_2O production were estimated by linear or quadratic regression of N_2O mass over time. The N_2O production (not inhibited by acetylene) during denitrification was calculated as the difference between denitrification rates of incubations with and without acetylene.

4.3.3 Phosphorus sorption and fractionation analyses

Phosphorus sorption isotherms

Sediment SRP sorption capacity was quantified as the equilibrium P concentration (EPC_0), i.e., the SRP concentration in overlying water at which no SRP uptake or release from sediments occur (Taylor & Kunishi, 1971). The EPC_0 was determined in composite sediments using P isotherms (**Paper III**; Holgersson, 2022), including four solutions with SRP target concentrations 0, 100, 250 and 500 µg PO_4^{3-}-P L^{-1}. Solutions were mixed with stream water from each site to account for the ambient ionic strength and final concentrations were therefore slightly higher than targets as stream water SRP averaged 28 ± 35 µg L^{-1}. Isotherms were prepared according to Trentman et al. (2020), using fresh sediments and liquid to solid ratio of 1:8 that were incubated in an end-over-end shaker at 30 rpm for 24 h. After measuring SRP concentrations colorimetrically in supernatants, target solutions and stream water, SRP sorption mass was determined as the difference between initial and final SRP concentrations at each target concentration, multiplied with the quotient of solution volume and sediment dry matter. Subsequently, EPC_0 was estimated by linear or non-linear regression of SRP sorption over initial SRP concentration, where x-intercept denote EPC_0 concentration.

To predict the direction of SRP exchange between water column and sediments, P exchange potential (PEP) was calculated as the difference of log10-transformed EPC_0 and SRP concentrations in stream water (Simpson et al., 2021).

Sequential phosphorus fractionation

To quantify the forms of P present in sediments (**Papers I and III**; Ryding, 2022), a sequential P fractionation was conducted according to Psenner & Puckso (1988) and Hupfer et al. (1995, 2009). This method was used to determine the operationally defined P fractions of water-soluble P, redox-sensitive P adsorbed to Fe and manganese, hydroxide-exchangeable P adsorbed mainly to aluminium P, P bound in organic compounds and calcium-bound P. The fractions water-soluble P and Fe-P are here considered as labile P. Sediment TP was calculated as the sum of all fractions since no residual non-reactive P was determined after fractionation.

4.4 High-frequency data analysis

4.4.1 Stream metabolism modelling

Daily stream metabolism rates were modelled between July 2021 and November 2023 in site S8 (**Paper IV**), using time series of O_2, stream surface light, water temperature and mean water depth with the one-station method (Odum, 1956):

$$\frac{dO_{2t}}{dt} = GPP_t + ER_t + K_{600t}D \tag{2}$$

where $\frac{dO_{2t}}{dt}$ is rate of O_2 change at time point t, K_{600} is gas transfer velocity and D is mean water depth. A Bayesian framework implemented in R package streamMetabolizer (Appling et al., 2018) was used to estimate daily rates of GPP, ER and K_{600}. To minimize equifinality in the three-parameter model, both observation and process errors were accounted for in the model. Discharge was further used to constrain estimation of K_{600}. Estimates of GPP, ER and K_{600} with biologically impossible values and poor fit between observed and modelled O_2 concentrations were removed from subsequent analysis (Figure 6a). The occurrence of equifinality was tested by regressing K_{600} and ER, wherein no correlation suggests absence of equifinality (Figure 6b).

The modelled stream met the criteria of homogenous upstream hydromorphology, three times the O_2 turnover length, required for accurate application of a one-station model (Reichert et al., 2009). The O_2 turnover length was estimated by dividing daily velocity with K_{600}. The influence of groundwater O_2 inputs was not accounted for but flow accumulation in the catchment was dominated by tile-drainage inputs, which minimize inputs of deep groundwater with low O_2 concentrations.

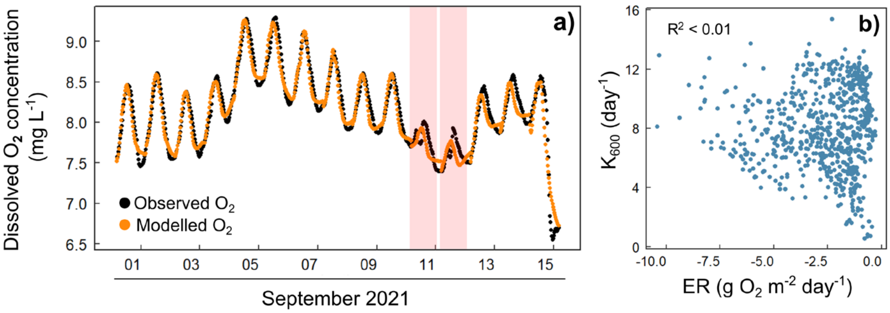


Figure 6. **a)** Modelled O_2 and measured O_2 during two weeks in September 2021. Red facets show two days removed from analysis due to poor fit ($R^2 < 0.6$). **b)** Relationship between gas transfer velocity (K_{600}) and ecosystem respiration (ER). The lack of correlation indicate absence of equifinality in metabolism model. Adapted from paper IV.

4.4.2 Supply-demand and storm event responses

To determine the capacity for biological processing of N and C inputs in site S8, the daily supply (g m^{-2} day^{-1}) were estimated as:

$$Supply = \frac{QC}{wL} \tag{3}$$

where C is NO_3^--N or DOC concentration (mg L^{-1}), w is channel width (m) and L is O_2 turnover length (m).

Biological demand (g m^{-2} day^{-1}) is here referring to NO_3^--N assimilation and DOC mineralization by stream autotrophs and aerobic heterotrophs.

Modelled GPP and ER O_2 fluxes were converted to autotrophic and heterotrophic NO_3^--N and DOC uptake based on Roley et al. (2014), using a photosynthetic quotient of 1.1, algal C:N ratio of 12 and bacterial C:N ratio of 5.

The response in GPP and ER rates and NO_3^--N, DOC and turbidity concentrations to discharge variability during storm events were quantified with the hysteresis index (Lloyd et al., 2015) and the response index (also referred to as flushing index by Butturini et al., 2008). The hysteresis index describes the hysteretic direction and magnitude of concentration/metabolic rate changes across an event while the response index quantifies the magnitude of stimulation/accretion or suppression/dilution from onset to peak of an event. Dynamics in GPP and ER rates were analyzed at daily intervals while NO_3^--N, DOC and turbidity concentrations were analyzed at 15 min intervals.

4.5 Data analyses

4.5.1 Hydrological data analyses

All statistical analyses were performed in R version 4.2.1 (RStudio Team, 2022). To determine the change in hydrological regime along remediated streams, base flow index (Gustard et al., 1992) and flashiness index (Baker et al., 2004) were calculated annually at locations US and DS, using R packages hydrostats (baseflow index; Bond et al., 2022) and ContDataQC (flashiness index; Leppo, 2023; **Paper I-III**).

Unit stream power (*ω)*, i.e., the water force exerted on bed sediments per surface area, was used to predict depositional/erosive responses in channels (**Paper I**), calculated as:

$$\omega = \frac{\rho g Q S}{w} \tag{4}$$

where ρ is density of water (kg m^{-3}), g is gravitational acceleration (m^2 s^{-1}), S is dimensionless channel slope and w is channel width (m), here measured as the top width of the channel.

In **paper IV**, storm events were identified by separating base flow and storm flow on the hydrograph at DS location in site S8. Base flow was estimated using a Lyne-Hollick filter, implemented in the R package hydrostats (Bond et al., 2022). Events were defined as an increase of discharge with > 100 % compared to base flow, reaching a peak > 0.2 $m^3 s^{-1}$. The end of events was defined as discharge on the falling limb receding < 100 % of base flow.

4.5.2 Identifying predictors of processes and water quality

To disentangle the multiple environmental controls of N, P and sediment processing as well as emergent responses in water quality, the multivariate datasets of **papers I-III** were explored using principal component analysis (PCA). By reducing dimensionality and maximizing variation, PCA can be used to explore relationships and covariation between multiple variables (Legendre & Legendre, 2012). To further determine the effect of predictors on response variables such as denitrification and P sorption, vectors of responses were fitted to PCA matrices to test their correlation to the sample distribution and predictors. This approach was used to identify important controls of processes but also clustering of sites.

After identifying specific controls, their isolated effect on responses were further tested using either linear regression and Pearson correlation coefficients (**Papers I-III**) or multiple linear regression (**Paper III**).

4.5.3 Exploring within and between site variability

In **papers I-III**, the variation in biogeochemical sediment processes between interfaces (channel and floodplains) and longitudinally (CS, US, MS and DS) were explored both within each site and by grouping all sites. The significance of variation differences was tested using t-test and ANOVA. In addition, Specific site groupings that shared similar controls, identified with PCA, were further tested to assess if certain families of sites (e.g., two-sided vs. one-sided floodplains, Central East vs. South regions) yielded different responses (**Papers I-II**).

4.5.4 Reach retention and cost efficiency analysis

To assess the effect of remediated reaches on water quality dynamics, compared to trapezoidal reaches, differences in N, P and SS concentrations along remediated reaches (US and DS) and trapezoidal reaches (CS and US) were regressed against concentration inputs for the entire monitoring period (**Paper I-III**). Differences in changes between the two types of reaches were only tested along comparable concentration inputs, using linear regression or ANCOVA.

Sample distributions of the concentration datasets were in general non-normal and left-skewed, which is a common property of water quality time series due to infrequency of high input events (Helsel, 1987). This was accounted for with non-parametric permutation of t-test and ANOVA, utilizing a statistical resampling approach that do not assume an underlying distribution but solely rely on the assumption of exchangeability, e.g., absence of temporal autocorrelation (Good, 2013).

To compare biogeochemical mass retention of floodplain remediation with alternative measures, annual loads of TP, NO_3^--N and SS were estimated at locations US and DS, using the flow-weighted mean concentration method (Elwan et al., 2018). The range of load changes and maximum cost efficiency of floodplain remediation were compared with constructed wetlands, vegetated buffer strips and integrated buffer zones, using literature values from Sweden and Northern Europe. Cost efficiency analysis of floodplain remediation was estimated as SEK kg^{-1} pollutant ha^{-1} yr^{-1} by dividing implementation costs with observed load reductions of sites with the greatest reductions. It therefore represents an estimate of the highest potential cost efficiency. Costs were distributed over 20 years and include implementation cost (1000 SEK m^{-1}; Jordbruksverket, 2024), opportunity costs for land put out of production and channel maintenance cost of 50 SEK m^{-1} (20-year intervals; Västilä et al., 2021). Opportunity costs are based on earlier reported leasing data, reflecting the period when the majority of the studied remediated streams were implemented (Collentine et al., 2015). The land cost of 3300 SEK ha^{-1} yr^{-1} was used in Central East Sweden and 7020 SEK ha^{-1} yr^{-1} in South Sweden.

5. Results and discussion

5.1 Hydrology and fluvial stability in remediated streams

The impact of floodplain remediation on both water quality and quantity primarily relies on the first order hydrological controls that connect floodplains to the channel. The following section addresses objective I by demonstrating that frequency of floodplain inundation but also designs of floodplains are key drivers for controlling water retention (**Papers I-III**) and fluvial stability (**Paper I**) in remediated streams.

5.1.1 Hydrological responses to floodplain inundation

Hydrological time series of the ten study sites are based on monitoring presented in **papers I-III**. The remediated streams were characterized by base flow-driven hydrology and comparable annual precipitation, except for higher precipitation in site S10 (Table 2). Overall, precipitation during the measurement period conformed with ten-year precipitation means.

At the onset of floodplain inundation, runoff in DS locations were decoupled from US and decreased in sites C3-5 and S7-8 (Figure A1). This suggests net water retention along remediated reaches, enabled by higher transient storage on floodplains and in the hyporheic zone (Bukaveckas, 2007). The responses were supported by lower values of flashiness index, suggesting reduced flow peaks at DS locations compared to US. The sites with increased water retention during high flows were characterized by > 700 m lengths (except site C4) and frequent floodplain inundation. By contrast, runoff increased substantially in sites S6 and S9 during floodplain inundation, which indicates that floodplain water storage was insufficient to counter the likely activation of lateral flow pathways along these reaches.

The analysis of hydrological responses to inundation was limited to remediated reaches and was not compared to changes in runoff along upstream trapezoidal reaches. However, with their lack of riparian water storage capacity, trapezoidal reaches are expected to show net increases in runoff generation. Future efforts in quantifying lateral inputs and extending the analysis to trapezoidal reaches could constrain uncertainties and quantitatively estimate transient storage increases in remediated streams.

Table 2. Characteristics of hydrology in upstream (US) and downstream (DS) locations of remediated reaches. Mean values of precipitation, runoff, annual base flow index (BFI) and flashiness index (RBI) and floodplain inundation frequency between April 2020 and December 2022. Ten-year precipitation means are shown in parentheses. Adapted from papers I-III.

Site	Location	Precipitation (mm)	Runoff (mm yr^{-1})	BFI	RBI	Inundation frequency (days yr^{-1})
C1	US	645 (607)	0.91 ± 1.01	0.89	0.21	53
	DS		0.54 ± 0.86	0.84	0.33	189
C2	US	635 (602)	0.44 ± 0.72	0.92	0.12	9
	DS		-	-	-	12
C3	US	621 (590)	0.15 ± 0.49	0.66	0.55	5
	DS		0.26 ± 0.45	0.81	0.33	62
C4	US	628 (597)	0.48 ± 0.62	0.84	0.26	168
	DS		0.48 ± 0.55	0.87	0.23	118
C5	US	629 (603)	0.62 ± 1.21	0.76	0.37	102
	DS		0.57 ± 0.81	0.85	0.23	24
S6	US	784 (740)	0.84 ± 1.10	0.93	0.12	77
	DS		0.90 ± 1.49	0.87	0.24	91
S7	US	734 (802)	0.61 ± 1.29	0.93	0.35	117
	DS		0.57 ± 0.99	0.87	0.27	209
S8	US	670 (725)	0.99 ± 1.68	0.87	0.19	130
	DS		0.84 ± 1.37	0.87	0.19	79
S9	US	654 (592)	0.49 ± 0.76	0.89	0.08	7
	DS		0.46 ± 1.01	0.88	0.11	2
S10	US	1027 (980)	0.98 ± 1.34	0.83	0.31	157
	DS		1.30 ± 1.21	0.88	0.21	-

5.1.2 Sediment deposition and particulate retention

Sediment deposition was five times higher on channel beds compared to floodplains (**Paper I**). Despite the dominance of channel sedimentation, deposition on floodplains surpassed that of observations from five Midwestern USA sites (0.5–13 mm sediment yr^{-1}; D'Ambrosio et al., 2015) and three Swedish sites (0.07 – 2.96 kg sediment yr^{-1}; Lacoursière & Vought, 2020).

Sediment deposition in channels was negatively correlated to unit stream power (Figure 7a) but was not influenced by SS load inputs ($p = 0.41$), suggesting that deposition was transport-limited rather than source-limited. On floodplains, deposition was primarily controlled by inundation frequency and vegetation cover (Figure 7b). During higher inundation frequencies (> 70 days yr^{-1}) there was further an independent effect of increased deposition with higher vegetation cover (adj. $R^2 = 0.06$, Vegetation $p = 0.04$, Inundation $p = 0.76$), indicating that vegetation trapped sediments on inundated floodplains.

Despite the response in sediment deposition on floodplains with increased inundation, any lateral transfer of sediment from channel beds to floodplains was insufficient to offset channel sedimentation in the seven sites where both interfaces were monitored (C1-5, S7 and S10). This indicates that fluvial imbalance persists in flat and fine-textured remediated streams and the previously reported principle of a self-cleaning channel without maintenance needs could not be confirmed (D'Ambrosio et al., 2015; Krider et al., 2017).

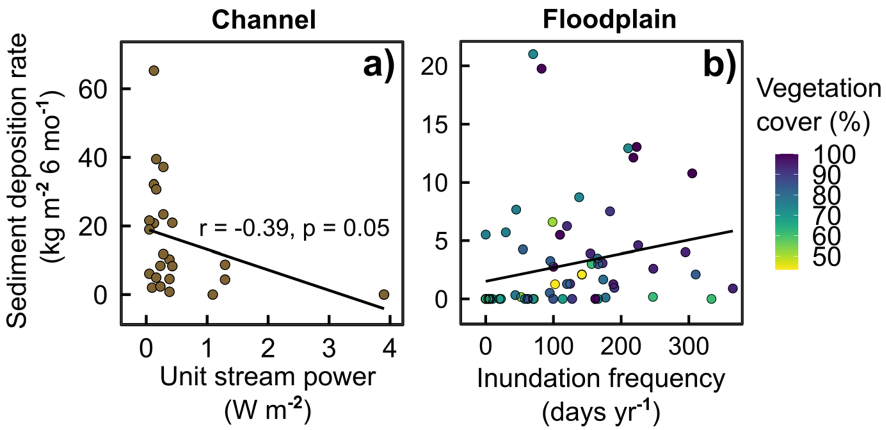


Figure 7. Linear regression between sediment deposition rates and primary controls on **a)** channel beds and **b)** floodplains. Adapted from Paper I.

Instead, it is possible that floodplains indirectly increase channel deposition by lowering the water velocity also in the channel, reflected in the low unit stream power.

Sediment deposition rates (mm yr^{-1}) correlated to TP content in deposited sediments ($r = 0.96$, $p < 0.01$) but there was no relationship between deposition rates on either channel beds or floodplains and reductions in TP, PP or SS concentrations and loads (**Paper I**). This could be explained by resuspension and erosion associated with inundation events, coinciding with deposition. It is further possible that floodplain areas were too small in the studied sites for sediment deposition to impact reach-scale SS and P reductions in stream water.

When grouping sites by floodplain design, reductions in TP and PP concentrations and loads were observed along two-sided but not one-sided remediated reaches. The same pattern was observed for PP loads, as higher PP load inputs in two-sided floodplains led to greater PP load reduction along two-sided but not one-sided floodplains. Correspondingly, SS loads were also lower along two-sided floodplains but there was no difference in SS concentration reductions between floodplain design. The difference in PP and SS dynamics could be related to their differences in particle sizes, as SS comprise the entire range of particles while PP is dominated by finer particles (Yao et al., 2016). Thus, it is possible that the reduction in erosion or mobilization was more pronounced for finer particles compared to coarser particles.

Reductions in PP concentrations along two-sided floodplains was likely an effect of reduced bank erosion, as lower water depths on floodplains compared to channels reduced shear stress on banks. Bank slopes were also lower in banks next to floodplains compared to banks next to channels, which may have further stabilized two-sided reaches. Together, these processes can explain the greater PP reductions in two-sided vs. one-sided designs, which is consistent with previous observations of one-sided designs being prone to erosion (D'Ambrosio et al., 2015). Moreover, the higher PP reduction with two-sided floodplains suggests that a substantial proportion of PP losses originated from proximate bank erosion or resuspension of deposited particles, rather than distal soil erosion. This finding supports the emerging concern that bank erosion may dominate P losses from agricultural catchments (Fox et al., 2016; Margenot et al., 2023) and warrants an intensified focus on instream mitigation beyond soil erosion protection.

5.2 Dissolved nutrient cycling in remediated streams

The link between instream nutrient cycling and emergent responses in water quality are critical for understanding the potential for stream remediation to reduce eutrophication. Addressing objective II-III, this section demonstrates the capacity of remediated streams for cycling and removal of N (**Paper II**) and P, as well as interlinkages between the two elements (**Paper III**).

5.2.1 Nitrogen removal by sediment denitrification

In the ten monitored remediated streams, denitrification rates (µg N g^{-1} DM h^{-1}) in floodplain sediments accounted for 33 % of the total denitrification capacity in the reaches (channel and floodplain sediments combined; **Paper II**). However, the floodplain and channel denitrification was highly variable, and in sites S7-8, floodplain denitrification occasionally exceeded that of the channels. Denitrification rates in channels did not differ between remediated and trapezoidal reaches (t-test, $p = 0.47$). Thus, there was no evidence of any indirect influences from floodplains on N removal in channels, which were mainly controlled by organic matter in sediments. Floodplain sediment denitrification was higher in the South sites (S6-10) compared to the Central East sites (C1-5; Figure 8a), due to higher NO_3^- delivery in the South catchments, dominated by higher agricultural land use proportions and coarse soil textures. Floodplain denitrification further increased with inundation frequency, in accordance with previous studies showing that inundation provides both anoxic conditions together with NO_3^- and organic C inputs (Mahl et al., 2015; McMillan & Noe, 2017; Roley et al., 2012b).

Sediment N_2O rates in floodplains were on average half of that in channels. However, the relative N_2O yields were > 50 % higher in floodplains, implying a greater risk for N_2O emissions with increased denitrification in floodplains compared to channels. The regional pattern in N_2O rates were opposite to that of denitrification, with the highest rates in Central East sites (Figure 8b). Higher N_2O rates and yields were associated with lower inundation frequencies (i.e., more fluctuating oxic conditions) as well as limited availability of NO_3^- and labile C in stream water that collectively suppressed complete denitrification.

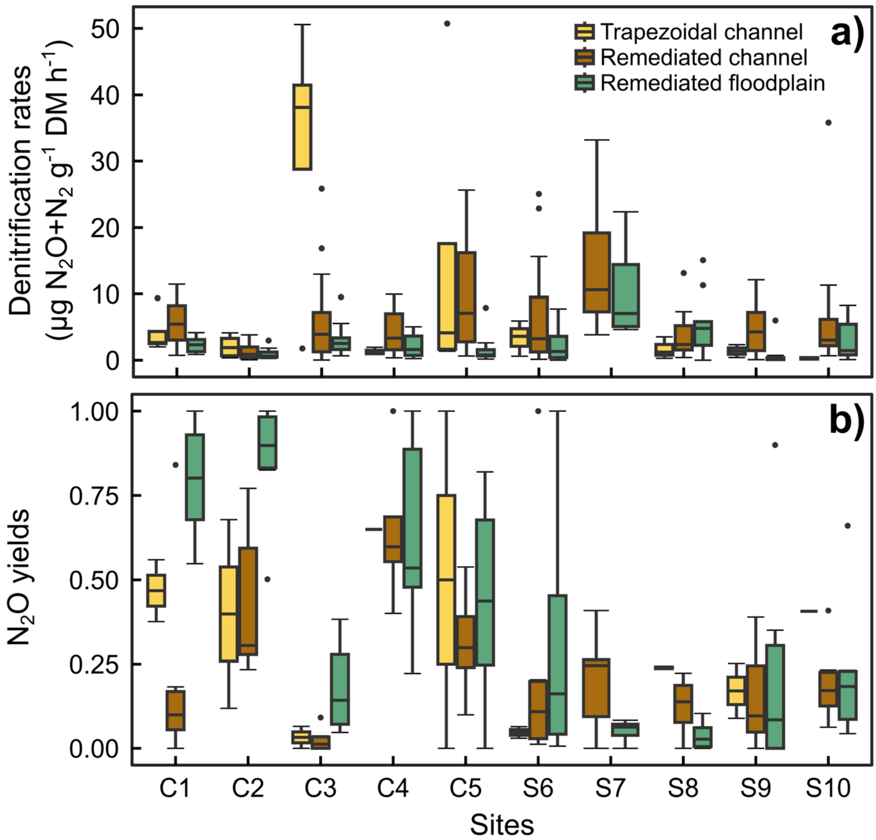


Figure 8. **a)** Potential denitrification rates and **b)** N_2O yields (N_2O:denitrification ratios) from sediments sampled in sites C1-5 and S6-10, between September 2020 and May 2021. Adapted from paper II.

5.2.2 Phosphorus exchange between stream water and sediments

The role of floodplains as sinks or sources of SRP were investigated in the ten remediated streams by estimating sediment SRP exchange using EPC_0 and PEP metrics (**Paper III**). Overall, there was no difference in SRP exchange between channel and floodplain sediments, neither for potential SRP sorption (EPC_0; t-test, $p = 0.14$) nor SRP exchange under site-specific stream water SRP concentrations (PEP; t-test, $p = 0.35$). Although the highest EPC_0 values (indicating P desorption) measured in floodplains were double that of channels, sediment SRP exchange with water column averaged net zero in both interfaces. By contrast, trapezoidal banks were in general sources of SRP, releasing SRP under background SRP concentrations in stream water. This is consistent with previous studies in agricultural streams (Ezzati et al., 2020; Kindervater & Steinman, 2019), explained by long-term accumulation of labile P in banks from exposure to agricultural field runoff. Thus, SRP exchange in floodplains was comparable to that in channel beds during inundation active periods and did not share the SRP desorption risk associated with trapezoidal banks.

Higher inundation frequencies together with higher two-year averages of stream water SRP concentration resulted in net SRP release from floodplain sediments, showing that hydrological connectivity and long-term SRP inputs determined the SRP sink/source behavior of floodplains. It has been suggested that floodplain excavation “resets” sediment P sorption capacity by increasing the exposure of available P binding sites when removing P saturated bank material (Trentman et al., 2020). However, sites (S7-8) showed high labile P and EPC_0 in floodplains, indicating that P saturation can occur within ten years after excavation. Further, organic matter in sediments can chelate and stabilize iron-oxide complexes with associated P (Kang et al., 2009) and have been linked to net SRP sorption in floodplain sediments (Trentman et al., 2020). However, there was no correlation between C content or the organic P fraction to SRP exchange in floodplain sediments in **paper III**, possibly confounded by differences in floodplain connectivity, SRP inputs and available sorption sites in sediments.

The response in stream water SRP due to SRP exchange in floodplain sediments was variable across sites; SRP desorption in floodplains was only linked to increased SRP concentrations in sites S8 and S10 but not in sites C4 and S6-7. Here, the monthly resolution in SRP monitoring restricted analysis of daily SRP dynamics, which have shown to follow a pattern of high sorption at inundation onset and subsequent slow desorption with prolonged inundation (Preiner et al., 2020). Overall, the observed patterns in SRP sorption/desorption from floodplain sediments demonstrate that remediated streams up to ten years of construction age have a limited impact on downstream SRP exports.

5.2.3 Trade-offs between nitrogen and phosphorus retention in floodplains

Trade-offs in nutrient processing

To determine the occurrence of synergies or trade-offs in N and P retention in floodplain sediments, process rates were compared between denitrification and SRP exchange (**Paper III**) as well as P deposition and SRP exchange (**Papers I and III**). A trade-off was identified between NO_3^- and SRP retention as denitrification rates correlated to EPC_0 in floodplain sediments. Denitrification and EPC_0 correlated both during simultaneous sampling in March to May in 2021 ($p = 0.02$) and when comparing entire study period averages ($p = 0.02$). The trade-off was governed by floodplain inundation frequency, where higher hydrological connectivity provided anoxic conditions that concomitantly increased denitrification rates and SRP release from sediments. This finding critically links the two processes to floodplain hydrological conditions, which have previously mainly been studied in isolation (denitrification: Forshay & Stanley, 2005; Roley et al., 2012b; SRP exchange: Surridge et al., 2012; Trentman et al., 2020). However, McMillan & Noe (2017) observed that both denitrification and P mineralization from organic matter can increase during floodplain inundation.

Labile P content correlated to sediment P deposition content in channel sediments, suggesting that high P deposition in channels increased the risk for SRP release to the water column. However, due to limited EPC_0 samples in channels, it was not possible to confidently test the link between channel SRP desorption and sedimentation. In floodplain sediments, deposited P did not correlate to either labile P or EPC_0, indicating that lower floodplain sedimentation, compared to channel, did not influence SRP release.

Water quality responses

Across all nine sites with paired remediated and trapezoidal reaches (C1-5, S6 and S8-10), reductions in stream water TP, PP and SRP concentrations in remediated reaches were greater than in trapezoidal reaches, when compared against concentration inputs to reaches (Figure 9a-c; **Papers I and III**). Reduction patterns similar to that of TP was also observed for SS concentrations. The greater reduction of PP compared to TP concentrations in remediated reaches was likely explained by increases in SRP, particularly during floodplain inundation. However, SRP concentrations overall did not increase in remediated reaches compared to trapezoidal reaches. This further demonstrates that PP reductions were not a result of pollution swapping between PP and SRP.

By contrast, reductions in dissolved organic nitrogen (DIN) concentrations were greater in trapezoidal reaches compared to remediated reaches (Figure 9d; **Paper II**). Despite the additional denitrification activity in floodplain sediments, this was not sufficient to affect overall NO_3^- reductions, compared to trapezoidal reaches (Figure 9e). This is consistent with previous studies also reporting a lack of substantial NO_3^- reductions along remediated streams (Davis et al., 2015; Mahl et al., 2015; Roley et al., 2012a). However, the monthly sampling strategy limited the detection of episodic NO_3^- removal peaks during onset of floodplain inundation, possibly resulting in underestimated NO_3^- removal in remediated streams (Dee & Tank, 2020; Roley et al., 2012b). Even so, the results in **Paper II** suggest that floodplain denitrification is, with current designs, restricted by NO_3^- residence times and reactive area. The resulting NO_3^- removal is therefore insufficient to produce observable reductions in the often high NO_3^- concentrations in agricultural streams. Instead, lower reductions in NH_4^+ concentrations along remediated streams, compared to trapezoidal streams, resulted in overall lower DIN reductions (Figure 9d, f). This was possibly explained by higher N mineralization rates and diffusion from floodplain sediments (McMillan & Noe, 2017; Weigelhofer et al., 2013). In addition, sustained anoxia in floodplain sediments may also have promoted dissimilatory nitrate reduction to ammonium (DNRA), and thus converted NO_3^- to NH_4^+ (Aalto et al., 2021).

It is important to note that the coupled comparison of concentration changes of the nine sites represents the collective effect of remediated streams with different designs, land use and soil texture. For TP and PP, concentration reductions were higher in sites with two-sided floodplains (C1-3 and S8) compared to one-sided floodplains (C4, S6-7 and S9-10). Likewise, reductions were observed of both DIN and NO_3^- concentrations in sites C4-5.

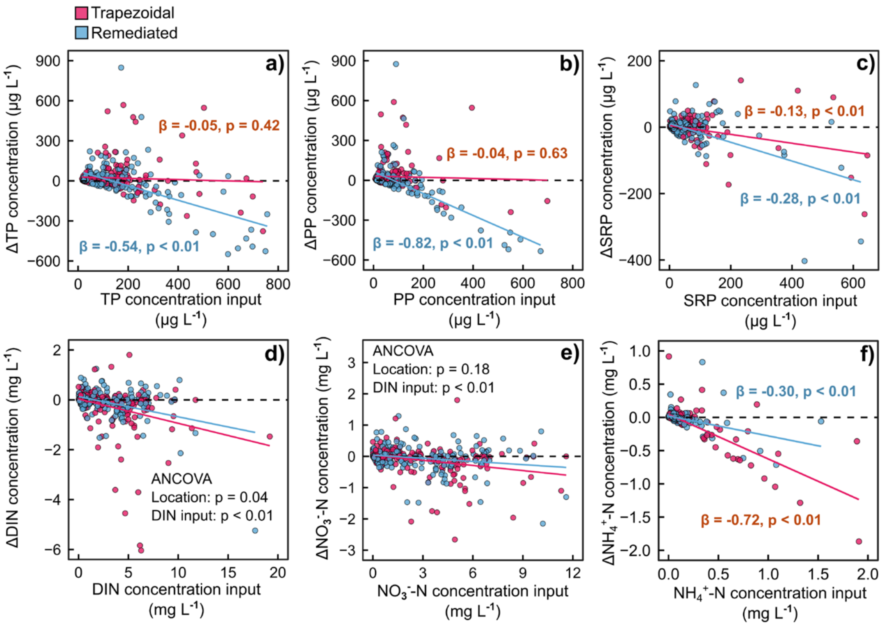


Figure 9. Stream water concentration changes of nine sites (C1-5, S6 and S8-10) along remediated (blue circles) and trapezoidal reaches (red circles), compared against concentration inputs at start of each reach. Concentrations of **a)** total phosphorus (TP), **b)** particulate phosphorus (PP), **c)** soluble reactive phosphorus (SRP), **d)** dissolved organic nitrogen (DIN), **e)** nitrate-nitrogen (NO_3^--N) and **f)** ammonium-nitrogen (NH_4^+-N). Water chemistry analyzed from monthly water samples collected between April 2020 and December 2023. Site S7 was excluded due to the lack of an upstream trapezoidal reference. P-values of linear regression and ANCOVA are shown within panels. β-values denote scaled regression coefficient. Adapted from papers I-III.

5.3 Metabolic linkages to nitrogen and carbon processing

The increased availability of high-frequency *in situ* sensors has enabled the study of dynamics in both pollutant supply and biological demand across the entire hydrological year. This section addresses objective IV by using stream metabolism rates to assess the capacity for modifying biogeochemical exports in a stream influenced by floodplain remediation, across a wide range of hydrological conditions (**Paper IV**).

The stream metabolism signal (O_2 turnover length) in site S8 originated from a reach extending on average 3.1 km upstream of the DS location, covering both remediated and trapezoidal reaches. The overall metabolic regime was dominated by negative net ecosystem productivity (-1.92 ± 1.79 g O_2 m^{-2} day^{-1}; NEP), but increases in GPP during summer 2023 resulted in positive NEP, peaking at 4.93 g O_2 m^{-2} day^{-1}. Daily GPP was predicted by light availability, which was restricted by riparian canopy that covered > 60 % of the modelled reach during summer months. This is consistent with a broad synthesis of stream metabolism, showing that light availability exerts a first order control on NEP (Bernhardt et al., 2022).

As supply of both NO_3^- and DOC greatly surpassed metabolic demand across the discharge range 0.1 – 4.4 m^3 s^{-1}, assimilation by algae, macrophytes and aerobic heterotrophs was insufficient to substantially modify downstream exports during hydrologically active periods. There was further no observable effect of autotrophic NO_3^- assimilation from water column during floodplain inundation, acknowledging that autotrophic demand integrated both the effect of trapezoidal and floodplain reaches.

In the growing season of 2023, reductions in discharge and NO_3^- concentrations coincided with increased metabolic activity, resulting in an unexpected near-limitation of NO_3^- (supply-demand ratio < 2). The concurrent dampening of daily NO_3^- cycling further supported the occurrence of NO_3^- limitation, which is rarely observed in agricultural streams when inferring limitation from nutrient stoichiometry (Jarvie et al., 2018b).

Storm events consistently triggered opposing responses in metabolism rates, where peak discharges suppressed GPP while stimulating ER. The correlation between response index of GPP and NO_3^- suggested that increased NO_3^- availability buffered GPP suppression during storm events. As NO_3^- was characterized by substantial dilution during peak discharges, temporal depletion in NO_3^- availability may have limited phytoplankton growth, as shown previously (Kelly et al., 2019). However, when comparing storm event responses in ER rates to NO_3^-, DOC and turbidity, there were no relationships for concurrent data. Concerning the controls for ER stimulation, this suggests a higher importance of changes in DOC quality as well as physical re-organization of benthic communities (Bertuzzo et al., 2022).

6. Impacts and management implications

6.1 Quantifying the success of stream remediation

In **papers I-III**, the selection of sites was guided by the priority of covering a wide range of remediated reaches with diverse designs and different catchment land use and soil texture distribution, together with active land manager interest and engagement. With the emphasis on spatial diversity, it was shown that not only floodplain designs but also differences in catchment pollutant pressures (determined by land use and soil texture distribution in the catchments) shaped the capacity for biogeochemical processing and reduction of pollutant exports. This suggests that high catchment pollutant inputs surpass instream processing capacity in many agricultural streams, such that their remediation alone is insufficient without in-field and edge-of-field measures. It is therefore important to set realistic expectations of retention capacities and acknowledge that an optimized engineered structure of a stream corridor still can be overwhelmed by excessive pollutant loads from past and present management. As previously shown with river restoration (Bernhardt & Palmer, 2011), the effect of any riparian intervention needs to be judged from the catchment-specific context it is placed in. In addition, the full potential of biogeochemical processing is not immediately realized after construction. It has been shown that N and P processing can increase with time after floodplain construction, requiring up to ten years before maximum efficacy is reached (Speir et al., 2020; Trentman et al., 2020). This is primarily dependent on vegetation establishment on floodplains, which gradually increases the often low organic matter content in recently excavated floodplain sediments (**Paper II**). Higher organic matter content can in turn enhance both denitrification

(**Paper II**; Speir et al., 2020) and SRP sorption capacity (Trentman et al., 2020). Likewise, sediment deposition on floodplains is also expected to increase over time with higher vegetation cover.

In each site, a control-impact design was chosen to evaluate the effect of floodplain remediation, pairing remediated reaches with upstream trapezoidal reaches of equivalent lengths. This was based on the assumption that paired reaches were subjected to similar hydrological and biogeochemical inputs, motivated by their drainage of comparable land use types and soil texture distributions. In particular, hydrochemical lateral inputs from tile drainage and groundwater were assumed to not differ between remediated and trapezoidal reaches. However, the choice to monitor hydrology only at start and end of remediated reaches did not allow for estimation of differences in water balance in trapezoidal reaches.

It has been suggested that a control-impact design, before and after implementation (BACI), is the ideal design for evaluating environmental impacts of stream interventions (Downes et al., 2002). This takes into account direct effects of remediation and eventual indirect effects on upstream control reach, but its reliance on extensive and costly monitoring strategies have restricted its use (Wohl et al., 2005). Similarly, the lack of monitoring before implementation of floodplain remediation in Sweden prevented a before and after approach in **papers I-III**. There is further a lack of field-measured stream geometry data after floodplain construction, which restricts the analysis of fluvial dynamics over time. Although planned stream cross-section geometries of remediated streams must be reported in Sweden (Larsson & Heeb 2016), final cross-section dimensions after excavation can differ and are therefore crucial for accurate assessment of geomorphic stability over time (D'Ambrosio et al., 2015).

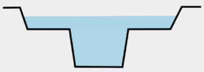
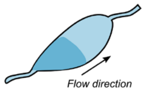
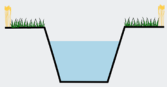
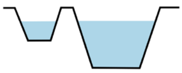
To fully assess the impact of stream interventions targeting eutrophication, it is necessary to couple water chemistry monitoring with ecological function metrics. Ecological recovery does not always correspond to reductions in pollutant loads, wherein long-term pollutant pressures shape ecological communities and their responses (Jarvie et al., 2013). Although the monitoring setup in **Paper IV** did not allow for control-impact or before-after study designs, it outlined the challenges in synchronizing metabolic assimilation peaks with floodplain connectivity. It further adds to the rapidly growing knowledge base of stream metabolism dynamics and regimes across stream networks and biomes, of aid to predict the effects of stream remediation on ecological functioning, besides water chemistry.

6.2 Comparison between floodplain remediation and other nutrient mitigation measures

Following in-field and edge-of-field measures, stream remediation represents the last point of mitigation before pollutants are exported from the catchment. Hence, stream remediation is complementary and does by no means substitute terrestrial mitigation measures which are often more proximate to pollution sources and can retain nutrients within the cropping system (Basu et al., 2022; Bol et al., 2018). For the comparison of floodplain remediation, the three complementary measures constructed wetlands, vegetated buffer strips and integrated buffer zones were chosen (Figure 10). These measures were selected based on the availability of comprehensive estimates of nutrient retention in Sweden and Northern Europe. Alternative stream corridor measures (e.g., lowered bank slopes, bank reinforcement, and channel meandering) were not compared due to a lack of available data from agricultural catchments.

Floodplain remediation showed the highest variability in load changes, with N, P and SS loads ranging from increases to reductions (Figure 10). The overall effect on loads along remediated streams indicates that differences in both background pressures and floodplain dimensioning results in highly variable downstream exports. However, it is important to note that estimated load changes integrate both the effect of floodplain remediation and background processes in stream and catchment. With the chosen monitoring setup, load changes in trapezoidal reaches could not be estimated and compared against remediated streams. In addition, while constructed wetlands and integrated buffer zones are spatially constrained and mainly target critical source areas, floodplain remediation also enables a wider spatial targeting of homogenously distributed sources.

Maximum NO_3^- load retention was five times higher in constructed wetlands compared to floodplain remediation (Figure 10), likely due to the combination of longer water residences times and higher denitrification and biotic assimilation in constructed wetlands (Vymazal, 2010). Constructed wetlands also showed higher TP load retention, explained by higher PP trapping.

Agricultural stream measure	Mode of operation	Load retention (kg ha^{-1} yr^{-1})	Maximum cost efficiency (SEK kg^{-1} ha^{-1} yr^{-1})
Floodplain remediation 	Interception of field runoff and instream transport Distributed measure Fluvial stabilization	[1]NO_3^--N: -14 800 – 2 400 [1]TP: -400 – 600 [1]SS: -713 000 – 283 200 (Sweden)	[1]NO_3^--N: 8 [1]TP: 170 [1]SS: 1
Constructed wetland 	Interception of field runoff and instream transport Localized measure	[2]NO_3^--N: -900 – 12 000 [2]TP: 0 – 200 (Sweden)	[2]NO_3^--N: 26 [2]TP: 241
Vegetated buffer strips 	Interception of field runoff Distributed measure	[3]TP: 0.6 – 1.4 (Sweden)	[3]TP: 1600
Integrated buffer zone 	Interception of field runoff Localized measure	[4]TN: 1 300 – 2 200 [4]TP: 20 – 30 (Sweden, Denmark and UK)	

[1]Papers I-III
[2]Djodjic et al., 2022
[3]Collentine et al., 2015
[4]Zak et al., 2019

Figure 10. Comparison of load retention (positive values denote retention) and cost efficiency of NO_3^--N, TP and SS with floodplain remediation, constructed wetlands, vegetated buffer strips and integrated buffer zones. Note that N retention of integrated buffer zones is reported as total N (TN).

Constructed wetlands also showed higher TP load retention, explained by higher PP trapping. However, the maximum TP retention of integrated buffer zones and vegetated buffer strips, that do not target instream erosion, was substantially lower compared to floodplain remediation. Maximum cost-efficiency of both NO_3^- and TP retention across 20 years resulted in lower costs per removed mass with floodplain remediation compared to constructed wetlands, explained by the lower loss of productive land with floodplain remediation.

6.3 Floodplain dimensioning and maintenance

6.3.1 Floodplain designs and dimensioning

Overall, floodplain designs and dimensions in the ten studied remediated streams covered a wide range in terms of reach lengths, floodplain elevations and widths, and were also implemented either as one- or two-sided floodplains (**Paper I-III**). The optimal dimensioning of floodplains depends on the objective with the remediation, be it flow dissipation and flood protection, erosion protection or N and P removal.

Floodplain elevation

The identification of inundation frequency as a primary control of floodplain biogeochemical activity (sediment deposition, denitrification and SRP exchange) demonstrates that careful consideration of floodplain elevation is necessary before implementation. Informed decisions should be made whether to prioritize or balance out higher N removal and PP deposition (elevation < 0.5 m; inundation frequency > 100 days yr^{-1}) against reduced risk of SRP release (elevation > 0.75 m; inundation frequency < 20 days yr^{-1}). Note that the thresholds of 0.5 and 0.75 m floodplain elevations are approximations and should not be taken as definitive cutoffs for high/low inundation frequency of all streams. Inundation frequency is also controlled by channel slope (Figure A2), which together with floodplain elevation can be used to estimate a target inundation frequency using a hydraulic model. In addition, pre-construction analyses of discharge and water chemistry together with bank soil texture and soil chemistry analyses form a solid basis for judging the appropriateness of targeting specific forms of nutrients and adjust floodplain elevation accordingly.

Although higher inundation frequencies have been predicted to reduce bank stability (Larsson & Heeb, 2016), there was no relationship between changes in PP or SS concentrations and inundation frequency (**Paper I**). Since channel-forming processes occur at decadal timescales (Simon & Rinaldi, 2006), it is still important to continue the monitoring and assessment of geomorphic stability beyond ten years after construction.

Floodplain length and width

Hydraulic modelling, comparing remediated and trapezoidal reaches, has shown that floodplain reaches > 1 km are required to achieve effects on flow dissipation and lowered surface water table in up- and downstream locations beyond the remediated reach (Lindmark et al., 2013). However, only four studied sites (C3, S7-8 and S10) in **papers I-III** met this criterion. Concerning floodplain widths, it has been recommended that the ratio between flooding width and channel width should surpass 3 to reduce mean water velocities and shear stress, compared to trapezoidal reaches (Ward et al., 2008). All of the studied sites had flooding:channel width ratios < 3, which together with limited reach lengths could explain the mixed responses in runoff reductions during floodplain inundation (Figure A1). Likewise, the restricted floodplain area may also contribute to the lack of impact on nutrient reductions by sediment deposition on floodplains and denitrification in floodplain sediments.

Current subsidy payments for floodplain remediation in Sweden are based on compensation per m length of remediated reach (Jordbruksverket, 2024), which incentivize longer reaches at the expense of floodplain width. It can therefore be assumed that the limited floodplain widths in Sweden are the result of the subsidy scheme. An alternative approach that can support wider floodplain widths would be to use areal-based subsidies similar to constructed wetlands.

Two-sided vs. one-sided floodplains

Originating from the theory of channel evolution in unmanaged streams, where two lateral floodplains are formed on both sides of the channel (Schumm et al., 1984; Landwehr & Rhoads, 2003), two-sided floodplains is the most common design of remediated streams. To account for obstructions such as riparian trees, boulders or high field elevations, one-sided floodplains have been implemented as a compromise, with double the width of two-sided floodplains. In **paper I**, the lack of reduction in PP export of one-sided designs, compared to two-sided designs, demonstrates that one-sided designs are unsuitable for mitigating erosion and PP losses. This is further corroborated by previous simulations of water velocity that predicted increased shear stress on banks without adjacent floodplains (Lindmark et al., 2013) and field observations of scouring in one-sided designs (D'Ambrosio et al., 2015). When targeting stream erosion protection and P mitigation, it is therefore recommended to prioritize two-sided floodplain designs and to identify location where these can be established.

6.3.2 Maintenance of channel and floodplains of remediated streams

Perceptions diverge on the role of streams either as interfaces for sediment exchange and nutrient cycling (Cluer & Thorne, 2013) or fluvially balanced sediment transport pathways (Simon, 1989). The different perspectives have emerged from the conflicting objectives of promoting ecological function, nutrient mitigation and hydraulic capacity. Although stream management in agricultural catchments is required to adhere to the securing of food production, it is important to identify areas for remediation where sediment deposition can be used to improve water quality without putting field drainage at risk.

Floodplain remediation was suggested by geomorphologists as a solution to repeated dredging of agricultural trapezoidal channels (Landwehr & Rhoads, 2003; Powell et al., 2007). However, in **paper I**, remediated streams with fine-textured beds and low slopes (C1-5, S7 and S10), showed no evidence for the process of a self-cleaning channel and redistribution of sediments to floodplains, as previously observed (D'Ambrosio et al., 2015; Krider et al., 2017). This implies a continued need for occasional dredging of channel sediments to maintain drainage function. However, the greater cross-section volume and thereby conveyance capacity of remediated streams allows for higher sediment accumulation before impairing drainage function which likely reduces the frequency of sediment removal, compared to trapezoidal channels. In sites with hard gravel beds and higher slopes (S6 and S8-9), ratios between channel and floodplain sediment deposition could not be estimated but visual observations and stream power estimates suggested low sediment deposition in channels. It is therefore feasible that streams with channel slopes > 0.3 % can support channel sediment self-cleaning. In addition to channel slope, Krider et al. (2017) reported that channel width can affect sediment redistribution to floodplains. By narrowing channel widths using excavated bank soil in a remediated stream with 0.2 % channel slope, they showed that it was possible to reduce channel deposition and redistribute sediments to floodplains.

Floodplain vegetation cover, consisting mainly of reed and herbaceous macrophytes, increased both sediment deposition (**Paper I**) and denitrification rates (**Paper II**) and is therefore an integral component in the biogeochemical functioning of remediated streams. Floodplain vegetation biomass was higher compared to biomass in channels, resulting in an increased ratio of organic C content in floodplains compared to channel sediments over time. Although cutting and removal of floodplain macrophytes reduce the potential for sediment trapping, this maintenance intervention can prevent the recirculation of nutrients stored in biomass to water column.

Macrophyte coverage in channels was in general high across all sites except C2 and S9. Establishment of instream vegetation is primarily a consequence of limited flow accumulation and lack of riparian canopy shading during summer and autumn, along with high nutrient content in sediments. Instream macrophyte growth is therefore common in many agricultural headwaters. Despite the higher potential for transient nutrient retention, instream macrophytes are often perceived as a nuisance by increasing the risk of drainage impairment (Levavasseur et al., 2014). Here, increased stream power through channel narrowing has also been shown to suppress the establishment of instream vegetation (Krider et al., 2017).

In light of the increasing interest in maintenance of agricultural streams in Sweden (Greppa näringen, 2023), it is important to recognize that some of these needs can be addressed by floodplain remediation. Thus, stream corridor interventions are important to include in agricultural water management strategies, allowing for new ways of understanding and perceiving agricultural stream functioning and the utility of natural fluvial processes to our benefit. Beyond ensuring a functional drainage of agricultural cropping systems, also animal habitats, human recreation and ultimately clean drinking water rely on the success of agricultural stream management.

7. Conclusions and future perspectives

The overarching aim of this thesis was to evaluate the biogeochemical functioning and potential for reducing eutrophication and erosion with floodplain remediation in streams across agricultural catchments in Sweden. By the use of interdisciplinary methods from catchment science, hydrology, geomorphology, biogeochemistry, aquatic ecology and statistical modelling, this thesis provided empirical evidence and decision support to realize water quality improvements through the implementation of floodplains in agricultural streams.

Based on the papers included in this thesis, it is concluded that floodplain activation via inundation and stream corridor dimensioning are master variables for controlling biogeochemical processes in floodplains, such as sediment deposition, denitrification and SRP release. Fluctuating anoxic conditions and higher vegetation cover gave rise to a biologically active riparian interface that increased beneficial denitrification rates but also production rates of the greenhouse gas N_2O of remediated streams. Thus, the additional N_2O production from floodplains constitutes an environmental trade-off between eutrophication and climate mitigation. It was further revealed that the risk of SRP release increased concomitantly with denitrification in floodplain sediments. The choice of floodplain elevation is therefore of critical importance for balancing dissolved N, P and greenhouse gas fluxes.

The observed reductions in both PP and TP exports with two-sided floodplains demonstrates that this design reduces stream erosion and improves downstream water quality with regards to P, as opposed to one-sided floodplains and trapezoidal channels. However, floodplain remediation did not support sediment self-cleaning in channels at locations with low channel slopes and fine-textured sediments. Hence, maintenance needs persist in the form of channel dredging and vegetation removal, albeit at longer return intervals compared to trapezoidal channels.

Despite the increased biogeochemical processing with floodplains, the lack of observed impacts on solute N and P implies that the additional floodplain activity is insufficient to modify high catchment exports of solutes, in particular NO_3^-. Estimates of stream metabolism that integrated trapezoidal and remediated reaches showed that load supply and metabolic demand of NO_3^- and DOC were decoupled across hydrologically active periods. This highlighted the challenge of matching high N assimilation rates of floodplain vegetation with floodplain inundation.

Interdisciplinary approaches are a prerequisite for capturing the overall effect of stream interventions and management on immediate and downstream aquatic ecosystems. Agricultural streams are subjected to a vast array of environmental pollutants and pressures, presenting challenges to achieve overall remediation success. Thus, it is only through the integration of biological, chemical and physical stream processes that a comprehensive understanding of underlying drivers can be realized and inform management towards water quality improvements.

7.1 Future perspectives

Based on the findings presented in this thesis, several areas were identified for refining and expanding the evidence-base for floodplain remediation as a suitable management practice for reducing eutrophication. In particular, the influence of the areal floodplain dimensioning, specifically length and width proportions, on hydrological and biogeochemical processes needs to be better understood. Currently, there are no estimates of any minimum flooding:channel width ratio required for nutrient removal processes to exert an effect on water quality. Coupled hydraulic and process-based modelling of the effect of floodplain dimensions on biogeochemistry should therefore be pursued to provide support for optimal sizing of length and width. This approach can further be used to explore stream network effects with different floodplain designs. However, while recommendations for floodplain dimensioning do exist based on hydraulics and flow dissipation, these are generally not followed.

In remediated streams, tile drainage outlets are often adjusted to discharge drainage water onto floodplains and not directly into channel surface water. This has been proposed to support additional water purification benefits. However, floodplain sediments can saturate with nutrients over time and also form gullies that rapidly transport tile drainage effluents to stream surface water. It is therefore important to further investigate the capacity for floodplain interception of nutrients from tile drainage effluents. Besides nutrient processing, there are further prospects to investigate the potential for floodplains to intercept and degrade other anthropogenic pollutants, such as pesticides, microbial pathogens, antimicrobial resistance genes and microplastics.

Targeted implementation of remediated streams to erosion-sensitive reaches is currently limited by the lack of systematic identification of streams with high internal SS and P loading. Despite the availability of nation-wide soil erosion risk maps, the identification of unstable agricultural streams with erosion-prone banks is confounded by background soil erosion. It is therefore necessary to intensify efforts to quantify specific erosion rates from banks in conjunction with soil erosion. Moreover, the capacity for improving bank stability with floodplain remediation should be compared to less costly and invasive riparian interventions, such as lowered channel bank slopes. To further strengthen the evaluation of erosion protection and environmental impacts of any stream mitigation measure, it is important to recognize the value of pre-construction monitoring of hydrology and water quality. To also allow for long-term evaluations of fluvial stability, it is crucial to measure final cross-section geometries after construction.

Amidst a changing climate, changes in weather patterns and increased frequencies in extreme events are projected to both increase flooding and drought spells and in turn accelerate nutrient and sediment losses. The responses in biogeochemical functioning and water quality with floodplain remediation should therefore be evaluated with scenarios that take not only past and present but also future hydrochemical pressures into consideration.

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Popular science summary

The main objective of this thesis was to aid the management of agricultural streams to support their function as water quality regulators while maintaining drainage capacity of agricultural soils. Water resources from streams to rivers and seas have been put under high pressure from nutrient and sediment inputs, often originating from agriculture. Enhanced nutrient loads in water environments, called eutrophication, drive rapid growth of algae and other water vegetation during summer season. This can in turn lead to increased production of harmful substances from algae but also reduce the diversity of other organisms present in water, suffocating from lack of oxygen. Eutrophication has restricted the use of water as a resource for drinking and recreation, but also shrunken available habitats for animals.

In agricultural fields, just the right amount of water is needed for crop growth - too little water makes crops wilt while too much water suffocates their roots. To achieve the right balance, the hydrology of agricultural landscapes has been heavily altered through field drainage and irrigation systems. In Sweden, removing excessive water has been the dominating issue, resulting in the expansion of tile drainage and open waterways. Unfortunately, agricultural streams and ditches are also efficient conduits for nutrients and sediments from agriculture, to the detriment of downstream water quality. The intimate contact between streams and agricultural soils can support high biological activity, which means that these streams have a high potential for retaining nutrients and sediments. However, the current design of many conventional agricultural streams, with narrow stream corridors and steep streambank slopes, prevents the full potential for both water and pollutant retention. To improve water retention and purification within these streams, floodplains can be constructed along the main channel in existing conventional streams. Floodplain remediation is a relatively new

measure in the Swedish agricultural landscape and its overall impact on stream processes and downstream water quality is therefore unknown across landscapes with different soil and land use properties.

In this thesis, the potential for reducing nutrient and sediment exports in agricultural streams was investigated in ten Swedish agricultural streams with floodplain remediation. To fully understand the impact of floodplains, a suite of biological, chemical and physical processes was studied, together with hydrology and water quality monitoring. The export of phosphorus bound to suspended particles was reduced with floodplains on both sides of the channel, compared to floodplains on one side of the channel and conventional streams. This further demonstrated that two-sided floodplains reduced erosion from streambanks and channel. The capacity for removing nitrogen from stream water through microbial denitrification (the conversion of nitrate to nitrogen gases, emitted to the atmosphere) in floodplain sediments accounted for 33 % of total denitrification in the streams. Denitrification increased with more frequent floodplain overflow, which contributed to oxygen-limited conditions needed for denitrification. However, frequent stream water overflow on floodplains also increased the release of dissolved phosphorus from sediments. Thus, the beneficial nitrogen removal but negative dissolved phosphorus release into stream water showed that water connectivity to floodplains affects nutrient forms differently. Despite this, the greater reduction in particulate phosphorus contributed to a net reduction in total phosphorus. When comparing changes in nutrient and sediment concentrations in floodplain streams to conventional streams, the impact was variable for different nutrients, with greater potential for reducing phosphorus compared to nitrogen. The measured activity of processes (sediment deposition, denitrification and sediment phosphorus release) did not exclusively lead to a reduced transport of different nutrients along the streams. This mismatch was explained by high pollutant inputs as well as limitations in floodplain designs (limited floodplain width, length and high floodplain elevations).

The work in this thesis contributes with new knowledge on the capacity for reducing nutrient and sediment exports using floodplain remediation, as well as the underlying controls by water and sediment processes. This knowledge can further be used to support decisions for stream water management in agricultural landscapes, aimed to reduce downstream eutrophication while maintaining necessary field drainage capacity.

Populärvetenskaplig sammanfattning

Det huvudsakliga målet med denna avhandling var att stödja förvaltningen av jordbruksvattendrag och förbättra vattenmiljön, utan att ge avkall på vattendragens förmåga att dränera jordbruksmark. Vattenförekomster från diken till åar och hav har i över 70 år utsatts för höga utsläpp av näringsämnen och jordpartiklar, som ofta härstammar från jordbruk. Förhöjda halter av näringsämnena kväve och fosfor i vattenmiljöer, kallat övergödning, driver på snabb tillväxt av alger och andra vattenlevande växter. Detta kan i sin tur leda till ökad produktion av skadliga substanser från alger men även minska mångfalden av vattenlevande organismer. Övergödning har begränsat användningen av vatten som dricksvattenresurs och dess roll för rekreation, men även krympt tillgängliga habitat för djur.

I jordbruksmarken behövs precis rätt mängd vatten för att grödor ska tillväxa, för lite vatten gör att grödorna vissnar medan för mycket vatten kväver deras rötter. För att nå den rätta balansen har vattnets väg i jordbrukslandskapet kraftigt förändrats med hjälp av fältdränering och bevattningssystem. I Sverige har bortförsel av vattenöverskott varit den dominerande frågan, vilket resulterat i expanderingen av täckdikning och öppna vattendrag. Jordbruksvattendrag och diken är tyvärr även effektiva ledare av näringsämnen och jordpartiklar från jordbruksmark, till nackdel för nedströms vattenkvalitet. Den täta kontakten mellan vattendrag och jordbruksmark kan gynna hög biologisk aktivitet vilket betyder att dessa vattendrag har en hög potential för att uppehålla näringsämnen och jordpartiklar. Nuvarande utformning av konventionella jordbruksvattendrag, med smala vattendragskorridorer och branta släntlutningar, motverkar dock den fulla potentialen för både retention och rening av vatten, näringsämnen och jordpartiklar. För att förbättra retention och rening av vatten i dessa vattendrag kan tvåstegsdiken anläggas med utgrävda svämplan längs med vattenfåran i befintliga konventionella vattendrag. Tvåstegsdiken är en relativt ny åtgärd i det svenska jordbrukslandskapet och dess övergripande påverkan på vattenprocesser och vattenkvalitet är därför fortfarande okänd i landskap med varierande jordtyper och markanvändning.

I denna avhandling undersöktes möjligheterna för att minska transporten av näringsämnen och jordpartiklar i tio svenska jordbruksvattendrag med anlagda tvåstegsdiken. För att fullt ut förstå påverkan av tvåstegsdiken studerades en rad av biologiska, kemiska och fysiska processer, tillsammans med mätningar av hydrologi och vattenkvalitet. Transporten av fosfor bundet till suspenderade partiklar minskade med svämplan på båda sidor om vattenfåran, jämfört med svämplan på ena sidan av vattenfåran och konventionella vattendrag. Detta demonstrerade dessutom att tvåsidiga svämplan minskade erosionen av slänter och vattenfåra. Kvävebortförsel från vatten genom mikrobiell denitrifikation (omvandlingen av nitrat till kvävgaser, som avgår i luften) i svämplanssediment stod för 33 % av total denitrifikation i vattendragen. Denitrifikationen gynnades av ökad översvämning av svämplan vilket bidrog till syrebegränsning i sediment, nödvändigt för denitrifikation. Frekvent översvämning av svämplan ökade dock även utsläpp av löst fosfor från sediment. Den fördelaktiga kvävebortförseln men negativa frigivningen av löst fosfor visade därför att vattnets kontakt med svämplan påverkar lösta näringsämnen olika. Trots detta bidrog den kraftigare minskningen av partikulärt fosfor till en nettominskning av totalfosfor. Vid jämförelsen av förändringar av närings- och partikelkoncentrationer i tvåstegsdiken gentemot konventionella vattendrag varierade effekten för olika näringsämnen, med större potential för minskning av fosfor jämfört med kväve. Den uppmätta aktiviteten av processer (sedimentering, denitrifikation och fosforutsläpp från sediment) ledde dock inte uteslutande till minskad transport av olika näringsämnen längs vattendragen. Denna skillnad förklarades av höga närings- och partikelbelastningar samt begränsningar i utformandet av tvåstegsdiken (begränsad svämplansbredd och längd samt höga svämplanshöjder).

Arbetet i denna avhandling bidrar med ny kunskap om tvåstegsdikens kapaciteten för att minska exporten av näringsämnen och jordpartiklar, samt hur olika vatten- och sedimentprocesser styr detta. Denna kunskap kan vidare användas för att stödja vattenförvaltning i jordbrukslandskap, med syfte att minska övergödning samtidigt som nödvändig dräneringskapacitet i fält bibehålls.

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Appendix

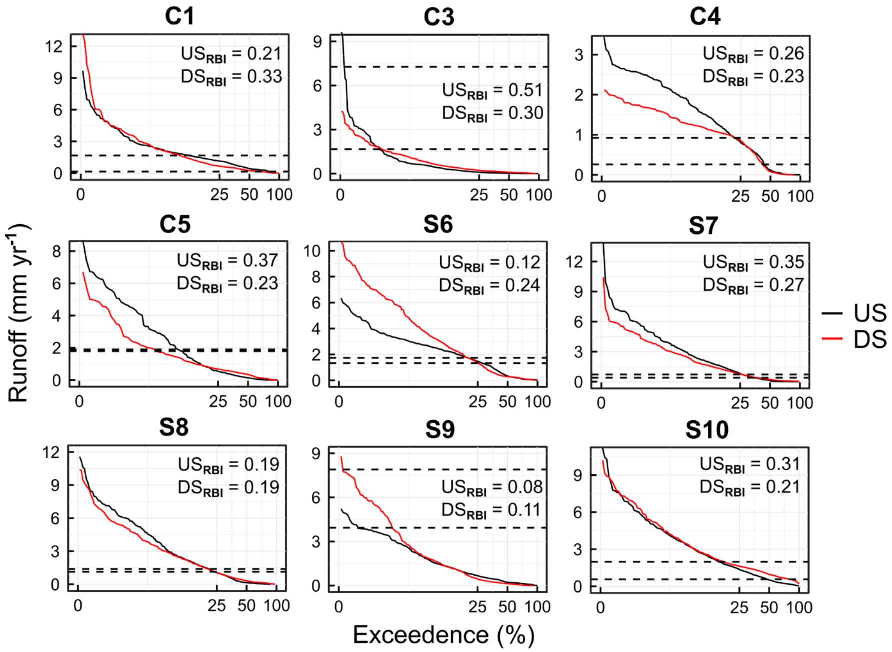

Figure A1. Runoff duration curves at the locations upstream (US) and downstream (DS) of remediated reaches (C1, C3-5, S6-10) between April 2020 and December 2022. Exceedence % denotes the probability of a certain runoff value to occur in the time series. Dashed lines show upper and lower threshold of floodplain inundation. RBI = flashiness index. Adapted from papers I-III.

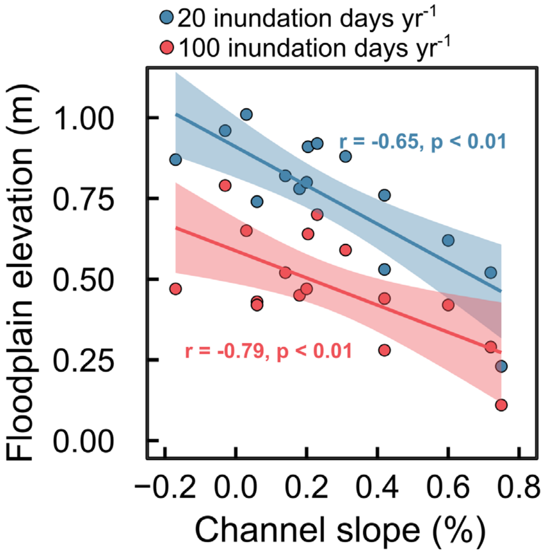

Figure A2. Relationship between floodplain elevation and channel slope for targeting floodplain inundation frequencies of 20 (blue circles) and 100 days yr^{-1} (red circles). Inundation frequencies predicted from up- and downstream stage measurements of sites C1-5 and S6-10, except upstream location of site S8 which was considered an outlier and therefore excluded. Adapted from papers I-III.

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Phosphorus supply and floodplain design govern phosphorus reduction capacity in remediated agricultural streams

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Abstract. Agricultural headwater streams are important pathways for diffuse sediment and nutrient losses, requiring mitigation strategies beyond in-field measures to intercept the transport of pollutants to downstream freshwater resources. As such, floodplains can be constructed along existing agricultural streams and ditches to improve fluvial stability and promote deposition of sediments and particulate phosphorus. In this study, we evaluated 10 remediated agricultural streams in Sweden for their capacity to reduce sediment and particulate phosphorus export and investigated the interplay between fluvial processes and phosphorus dynamics. Remediated streams with different floodplain designs (either on one side or both sides of the channel, with different width and elevation) were paired with upstream trapezoidal channels as controls. We used sedimentation plates to determine seasonal patterns in sediment deposition on channel beds and floodplains and monthly water quality monitoring. This was combined with continuous flow discharge measurements to examine suspended sediment and particulate phosphorus dynamics and reduction along reaches. Remediated streams with floodplains on both sides of the channel reduced particulate phosphorus concentrations and loads ($-54\,\mu g\,L^{-1}$, $-0.21\,kg\,ha^{-1}\,yr^{-1}$) along reaches, whereas increases occurred along streams with one-sided floodplains ($27\,\mu g\,L^{-1}$, $0.09\,kg\,ha^{-1}\,yr^{-1}$) and control streams ($46.6\,\mu g\,L^{-1}$). Sediment deposition in remediated streams was five times higher on channel beds than on floodplains and there was no evident lateral distribution of sediments from channel to floodplains. There was no effect from sediment deposition on particulate phosphorus reduction, suggesting that bank stabilization was the key determinant for phosphorus mitigation in remediated streams, which can be realized with two-sided but not one-sided floodplains. Further, the overall narrow floodplain widths likely restricted reach-scale sediment deposition and its impact on P reductions. To fully understand remediated streams' potential for reductions in both nitrogen and different phosphorus species and to avoid pollution swapping effects, there is a need to further investigate how floodplain design can be optimized to achieve a holistic solution towards improved stream water quality.

1 Introduction

Past and present inputs of bioavailable nutrients for agricultural production continue to negatively impact stream water quality and ecology across aquatic ecosystems globally (Sharpley et al., 2013; Smith, 2003). In particular, phosphorus (P) losses to surface waters play a key role in eutrophication as it is often the limiting element for algal and cyanobacterial growth in freshwater recipients (Correll, 1998). The transport of P from land to rivers and lakes is disproportionately influenced by headwater streams, representing entry points for sediments and nutrients to stream networks. Therefore, the capacity of headwater streams for the processing and removal of pollutants is critical for regulating downstream export (Wollheim et al., 2018). Headwaters in flat agricultural landscapes (< 2 % slopes) commonly comprise artificially straightened and deepened trapezoidal channels (Fig. 1b) that effectively drain excess water from fields and enable crop growth. However, their short water residence time and geomorphic instability also enhance downstream export of suspended sediments (SS) and associated P. Therefore, an improved understanding of the capacity for sediment and nutrient reductions within agricultural stream networks is critical, and increasingly so with a changing climate that accelerates pollution to freshwaters (Ho et al., 2019; Ockenden et al., 2017).

High export of sediments and P from trapezoidal channels, either historically converted from natural streams or artificially introduced, results from altered transport capacity and an imbalance between the catchment's supply and the stream's transporting power (Magner et al., 2012; Simon and Rinaldi, 2006). Internal erosion from trapezoidal channels can therefore contribute with higher sediment and P loads compared with distal sediment sources in catchments (Simon and Rinaldi, 2006). Closely interlinked with sediment conveyance in channels is the instream transport and storage of P, governed by fluvial processes (deposition/resuspension of sediment-bound P; Ballantine et al., 2009; Noe et al., 2019), redox processes (adsorption/desorption of soluble reactive P [SRP] to SS; Sandström et al., 2021) and geomorphic stability (streambank erosion; Fox et al., 2016). Channel bed sediments can store high amounts of P (Ballantine et al., 2009) and although deposition has been found to dominate in headwater reaches owing to stream power limitation (Worrall et al., 2020), sediment-bound P becomes susceptible to remobilization during high flow events (Bowes et al., 2003). Streambank erosion due to bank failures in agricultural streams can also be a significant source of P export, particularly in P-rich bank soils (Fox et al., 2016; Kronvang et al., 2013; Landemaine et al., 2015), but the efficacy of management strategies to counter this remains underexplored.

To reduce erosion and nutrient losses while maintaining drainage capacity, two-stage ditches with lateral floodplains excavated along existing channels have been tested as a stream remediation measure (Powell et al., 2007), with either two narrower or one single wider floodplain (Fig. 1). Here, floodplains refer to 2 to 6 m wide benches that are inundated upon higher flows (D'Ambrosio et al., 2015; Mahl et al., 2015; Powell et al., 2007). The purpose of two-stage designs is to establish a fluvial equilibrium by mimicking natural floodplain formation. Their design can support stable floodplains and banks, aid flood management, and reduce the need for routine dredging and bank restoration (D'Ambrosio et al., 2015; Krider et al., 2017). During higher flows, stream power dissipates as water inundates vegetated floodplains, allowing for reductions in SS and P concentrations by sedimentation (Davis et al., 2015; Mahl et al., 2015), sediment adsorption of SRP (Trentman et al., 2020), and nitrate removal through denitrification (Hallberg et al., 2022; Mahl et al., 2015; Speir et al., 2020). Reductions in SS and P concentrations and loads along remediated streams with a two-stage design are argued to be driven by lateral distribution of sediments from the inset channel onto floodplains during higher flows (Fig. 1), based on monitoring studies (D'Ambrosio et al., 2015; Krider et al., 2017; Mahl et al., 2015) and flume experiments (Bai and Zeng, 2019). To date, studies on sediment delivery in remediated streams have solely relied on either geometric survey data or grab water sampling, with emphasis on investigating its potential for improving geomorphic stability. Therefore, there is a need to explicitly link net P deposition rates and fluvial dynamics in field conditions with observed water quality impacts.

We measured net sedimentation rates and concentrations of P fractions in channel bed and floodplains of ten remediated streams (Fig. 2a) with two-stage design (Fig. 1) with the aim of investigating the spatial and temporal distribution of sediments and the effect on P export and long-term storage. Changes in stream water P concentrations along remediated streams and upstream trapezoidal control streams were compared to determine their reduction efficacy (Fig. 2b). Our objectives were to determine (1) if remediated streams with two-stage design can reduce SS and P export, (2) how floodplain designs (elevation and one- vs. two-sided floodplains) affect reduction capacity, and (3) what factors of transport, erosion and deposition control SS and P reductions. Two types of floodplain designs were compared: one-sided and two-sided floodplains along with a gradient in floodplain elevations and thus inundation frequencies.

2 Materials and methods

2.1 Site description

Ten study sites were selected with established two-stage ditches (referred to as remediated streams) in Central East (C1–5) and South Sweden (S6–S10; Fig. 2a). The study sites are located in low slope gradient and tile drained agricultural catchments, dominated by winter and spring sown cereal crops and ley grass. A control-impact design was used to compare remediated reaches (0.3 to 1.7 km reach lengths) with upstream, mostly trapezoidal reaches, hereafter referred to as control streams (see Figs. S6–S9 for comparison of profiles). An exception is site C4 where the erosion of stream bank has created a natural floodplain (Fig. S7). Control stream reaches were selected with equivalent lengths (where this was possible) and similar channel slopes and agricultural land use % in sub-catchments (Table S1). For detailed monitoring design see Figs. S1–S5. Site S7 was not paired with a control reach as the remediated stream originates from a wetland. Additionally, sites S7 and S8 are nested in the same stream network (Fig. 2a), with approximately 10 km stream length in between. Site C1 has two tributaries draining into the remediated stream reach.

In remediated streams, floodplains have been constructed either symmetrically on two sides (C1–3 and S8) or on one side (C4, S6–7 and S9–10) of the inset channel, site C5 being the exception with both designs along the reach. One-sided floodplains with double widths were constructed in streams

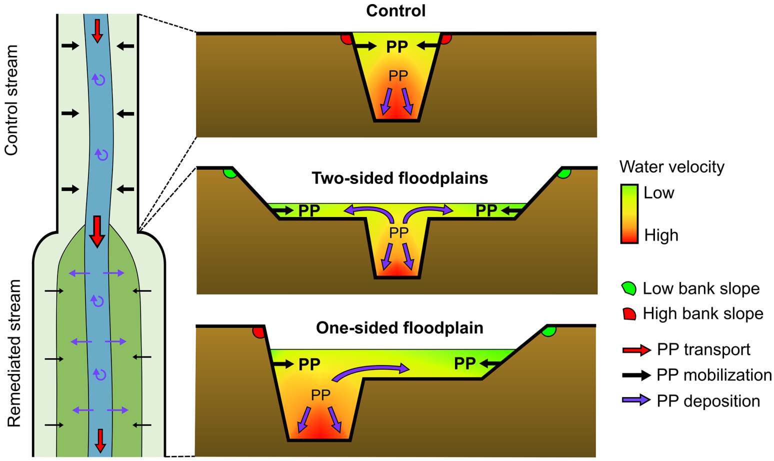

Figure 1. Conceptual model of fluvial processes that determine particulate phosphorus (PP) export in remediated vs. control streams. Deposition of PP occurs on channel beds predominantly during lower flows when transport capacity is lower than sediment delivery. During inundation, PP deposition dominates on floodplains, owing to lower water velocities. In two-sided floodplains, streambanks are stabilized during higher flows owing to their lower slopes and lower shear stress from water forces than the inset channel. In one-sided designs, one streambank is steeper and exposed to higher water velocities in the channel, resulting in higher mobilization and transport of PP.

where riparian obstructions (e.g., trees, boulders, and high field elevation) restricted two-sided floodplain excavation. Floodplain elevations range between 0.25 and 0.96 m in relation to the lowest point of the inset channel and the ratio of total flooded widths to channel top widths ranges between 1.3 and 2.9. Remediated streams were constructed between 2013 and 2019, with the purpose of preventing flooding of adjacent fields and improving fluvial stability and downstream water quality (Brink et al., 2012; Wiström and Lindberg, 2016; Hedin and Kivivuori, 2015).

The sites are variable in their catchment characteristics and hydrology (Table 1). Central East catchments (C1–5) are generally characterized by lower agricultural land use density and higher clay content in soils than in South catchments (S6–10). Overall, the remediated streams are dominated by base flows, with median discharge ranging between 0.01 and $0.32\,m^3\,s^{-1}$.

2.2 Sediment deposition rates: field sampling and analysis

To collect net sediment deposition rates, $0.16\,m^2$ square wooden fiber plates were installed in September 2020 and anchored with a 1 m metal rod through the center (Fig. S10). Plates with a smooth surface coating were chosen to allow for resuspension of settled sediments but also to enable rapid *in situ* sediment collection. Sedimentation plates were installed on channel beds and on floodplains along each remediated ditch at the three locations upstream (US), midstream (MS) and downstream (DS; Fig. 2b), except at sites S6 and S8–9, where no plates could be placed in channels owing to impenetrable gravel and pebble bed substrates. Over the course of 21 months, deposited sediments were collected from the plates three times, approximately biannually (Fig. 2c) and with a total sample count of 41 (channel) and 73 (floodplains). To collect intact sediment samples from submerged plates in channels, a cylindrical frame was fitted onto plates before retrieval to prevent losses of sediments to the water column (Fig. S10). Before sampling, plates were lifted up from the channel beds to dewater sediments and the sediments were then measured for depth to estimate the total volume. Sediments were only collected from the inner $0.04\,m^2$ square of the plates to avoid edge effects, and were subsequently stored in airtight plastic bags. Sediment plates with high volumes ($n = 32$) were subsampled from the $0.04\,m^2$ square, with approximately 1 kg of fresh weight sediments collected after homogenization in the field. The fresh weight per $0.04\,m^2$ of subsampled sediments was then estimated from the product of the sediment volume on the plate and the bulk density of fresh sediments, measured per 100 mL. In the case of missing sediment depth values in samples ($n = 13$), the depth was estimated based on a linear relationship between sediment fresh weight ($kg\,m^{-2}$) and sediment depth (mm) of known samples ($n = 90$; $r = 0.99$, $p < 0.01$). All samples were oven dried at 105 °C to determine dry matter and then analyzed for TP with ICP-OES (Avio 200, PerkinElmer, USA). The TP content in sediments was nor-

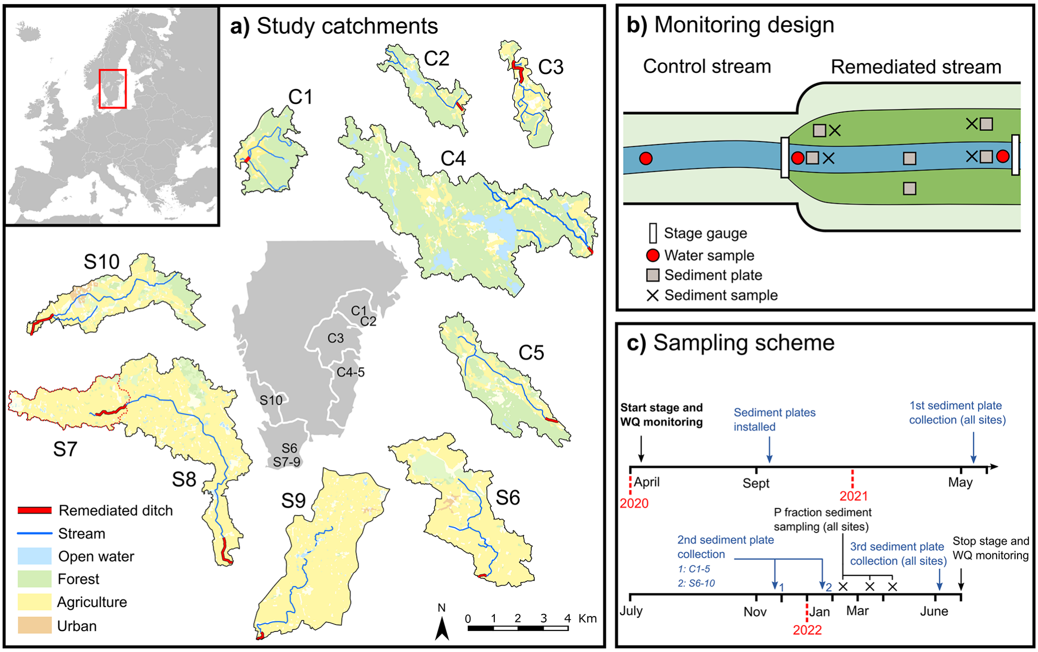

Figure 2. **(a)** Location of the ten catchments with remediated streams in Central East (C1–C5) and South Sweden (S6–S10). **(b)** Control-impact monitoring design. Sediment samples and water grab samples were collected at the start, middle, and end of remediated reaches and water grab samples were also collected at the start of control reaches. Water stage sensors were placed at the start and end of remediated reaches. **(c)** Sampling scheme over time of sediments deposited on plates and composite sediments for sequential P fractionation. Land use maps: © Lantmäteriet.

malized to deposition rates per area over six months as well as one year.

2.3 Composite sediment sampling and sequential P fractionation in sediments

In addition to newly deposited sediments on plates, composite sediments that represent the integration of both short- and long-term sediment deposition were sampled with a trowel in spring 2022 to determine P fractions. Sediments in channel bed and floodplains were sampled at US and DS for sequential P fractionation analysis (Fig. 2b, c). Sediments were sampled down to a depth of 5 cm with a trowel as 5 cm^3 cubes. At each channel location, five pseudo-replicates were sampled at a distance of 10 m along reaches and pooled into one sample. On floodplains, ten pseudo-replicates were sampled at a distance of 5 m. Sediments were placed in airtight plastic bags and stored in coolers during field transportation and at 4 °C back at the laboratory before dry matter was determined by oven drying at 105 °C.

To determine the forms of P present in composite sediment samples, a sequential P fractionation method was used, developed from Psenner and Puckso (1988) and Hupfer et al. (1995, 2009). The analyzed P species (as operationally defined) were water-soluble P (H_2O–P), redox-sensitive P adsorbed to iron and manganese (Fe–P), OH^- exchangeable P adsorbed mainly to aluminum (Al–P), P bound in organic compounds (Org–P), and calcium-bound P (Ca–P). Here, H_2O–P and Fe–P are also referred to as labile P and Al–P, Org–P, and Ca–P as recalcitrant P. The residual nonreactive P after the aforementioned fractionation (refractory P) was not determined; cumulative concentrations of analyzed P fractions are referred to here as TP. Fresh sediment samples were sequentially extracted with Milli-Q water (H_2O–P), buffered dithionate solution (Fe–P), NaOH (Al–P), NaOH after 30 min persulfate digestion at 120 °C, and subtracted with NaOH (Org–P) and HCl (Ca–P). All extractions were analyzed for unfiltered SRP using the molybdenum blue method (Murphy and Riley, 1962). Concentrations of P fractions in sediments were calculated as g P kg^{-1} DM for comparison with TP concentrations in newly deposited sediments.

Table 1. Characteristics of catchment, hydrology, and ditch geometry of remediated streams. Sites are ordered from North to South. Flow discharge and floodplain inundation were recorded between April 2020 and June 2022. Discharge in site C3 was recorded between 2018 and 2022, interrupted between December and March each year. Ditch and floodplain geometry were measured in March and April 2021 with a GPS leveling device (E600 GNSS Receiver, E-survey, China).

		Catchment properties				Hydrology						Stream geometry					
Site	County	Area (km^{-2})	Precipitation (mm)[a]	Soil texture	Agricultural land use (%)	Q_{50} ($m^3 s^{-1}$)	Baseflow index (BFI)	Flashiness index (RBI)	Unit stream power ($W m^{-2}$)[b]	Channel max flow ($m^3 s^{-1}$)	Floodplain inundation ($d yr^{-1}$)	Year of construction	Remediated length (m)	Floodplain height (m)	Flooding-channel ratio	Channel slope (−)	Floodplain design
Central East																	
C1	Södermanland	9.75	594	Silty clay	15	0.02	0.80	0.43	0.05	0.04	107	2012	340	0.55	1.6	0.04	2-sided
C2	Södermanland	7.91	594	Silty clay loam	27	0.03	0.93	0.13	0.16	0.20	17	2012	730	0.96	2.2	0.04	2-sided
C3	Östergötland	6.63	551	Clay loam	70	0.01	0.82	0.30	0.38	0.03	17	2014	1500	0.53	2.5	0.30	2-sided
C4	Kalmar	45.54	627	Clay loam	16	0.32	0.91	0.12	1.29	0.56	187	2019	320	0.65	1.3	0.10	1-sided
C5	Kalmar	16.32	627	Clay loam	38	0.06	0.83	0.30	1.09	0.21	53	2012	780	0.65	1.5	0.14	1-sided/2-sided
South																	
S6	Skåne	22.65	707	Loam	77	0.09	0.83	0.35	1.43	0.13	100	2016	400	0.48	2.9	0.17	1-sided
S7	Skåne	10.84	569	Loam	81	0.04	0.81	0.30	0.23	0.07	159	2013	1960	0.55/0.25	2.8	0.09	1-sided
S8	Skåne	42.41	569	Loam	81	0.22	0.86	0.23	10.90	0.68	93	2013	1770	0.48	2.4	0.47	2-sided
S9	Skåne	31.02	569	Loam	86	0.11	0.90	0.09	15.15	2.18	6	2019	630	0.71	2.3	0.44	1-sided
S10	Halland	16.38	823	Sandy loam	58	0.20	0.89	0.20	1.04	0.34	123	2014	1760	0.34	2.2	0.09	1-sided

[a] Mean annual rainfall 2016–2021 (https://www.smhi.se/data/meteorologi/nederbord, last access: 20 March 2023), [b] Unit stream power at the point of inset channel overflow.

2.4 Hydrological monitoring and stream power calculation

Discharge was estimated by establishing stage-discharge rating curves from discharge measured manually on four to eight occasions and continuous water stage data from pressure sensors, as described in Hallberg et al. (2022). To minimize overestimation of high flows outside the range of flow measurements, out-of-range flows were estimated for all sites except for C2 and C4, using the product of (1) polynomial regressions of stage-cross-section area and (2) logarithmic regressions of stage-mean velocity (Herschy, 2014). To determine the effect of floodplain inundation on sediment deposition, geometrical cross-section surveys were conducted and floodplain inundation events were subsequently estimated using stage data (Hallberg et al., 2022).

The power generated by a stream is a function of the amount of water that flows down a slope in a confined channel. To compare the stream power and its influence on bed sediments between streams of different sizes (Bagnold, 1973; Reinfelds et al., 2004), the total stream power can be normalized by channel width to unit stream power (ω) at a given cross-section using:

$$\omega = \frac{\rho g Q S}{w}, \tag{1}$$

where ρ is the density of the water ($kg\,m^{-3}$), g is the acceleration due to gravity ($m^2\,s^{-1}$), Q is the flow discharge ($m^3\,s^{-1}$), S is the dimensionless channel bed slope and w is the channel width (m), here measured as the top width of the inset channel. With unit stream power, the force exerted per surface area and thus potential sediment transport can be derived, irrespective of system size. Unit stream power was calculated at the point of floodplain overflow, using derived flow discharge and channel widths from geometrical surveys, representing the maximum force of water when restricted to the inset channel.

2.5 Water quality sampling and vegetation survey

Water grab samples were collected monthly between April 2020 and July 2022 to determine water chemistry at the four locations upstream control (TD), US, MS, and DS (Fig. 2b). At site S9, sampling after June 2021 was excluded from analysis as the control stream was then converted into a remediated stream. Water samples were analyzed for TP (SS-EN ISO 6878:2005), with and without 0.45 µm filtration, and suspended solids (SS; SS-EN 872:2005). Particulate P was calculated as the difference between unfiltered and filtered TP. Existing TP, PP, and SS water chemistry from a location 1 km downstream of site C3 (Kyllmar et al., 2014) were used for subsequent validation of the flow-weighted mean concentration (FW) load estimation method. Two types of data were used: (1) Annual loads estimated from fortnightly flow-proportional composite sampling and (2) fortnightly discrete concentration samples. In addition, vegetation cover was visually estimated on floodplains using three $1\,m^2$ ($1 \times 1\,m$) plots at each of the three locations US, MS, and DS. Average percentage vegetation cover was calculated from two surveys in May and June 2021.

2.6 Statistical data analysis

All statistical analyses were performed in R version 4.2.1 (RStudio Team, 2022). The packages' hydrostats (Bond, 2022) and ContDataQC (Leppo, 2023) was used to analyze the hydrological regimes. Base flow index (Gustard et al., 1992) were calculated using *baseflows* and flashiness index (Baker et al., 2004) using *RBIcalc*. Two outliers in TP and PP concentrations and one outlier in SS concentrations were removed from further analysis; these data points had exceptionally high concentrations (studentized residuals > 10, i.e., regression model residual divided by its adjusted standard error) and were measured in stagnant water during summer months, with minimal influence on total loads.

Annual loads of TP, PP, and SS per ha were calculated at US and DS locations using the FW method (Elwan et al., 2018). The accuracy of annual load estimation was tested downstream of site C3. Here, loads calculated from existing fortnightly flow-proportional composite sampling represent "accurate" loads (Dialameh and Ghane, 2022). At the same location, FW loads were calculated for the study period using water samples from two to three different days within each month to compare with true loads (Fig. S11). At this location, annual FW loads were underestimated across all parameters (TP: −28 %, PP: −29 %, and SS: −52 %). Daily loads of TP based on monthly water chemistry were also calculated by multiplying with daily flow. Changes in TP, PP, and SS concentrations were calculated as the difference between US and TD (control streams) and the difference between DS and US (remediated streams), hence negative values represent reductions in concentrations along reaches. In the case of non-normal sample distributions of changes in TP, PP, and SS concentrations (i.e., slightly right-skewed and with tails of outliers to both left and right), we used the approach of nonparametric permutation of t-tests ($R = 10\,000$), using MKinfer package (Kohl, 2022). Permutation tests do not assume an underlying distribution and solely rely on the assumption of exchangeability, such as the absence of temporal autocorrelation (Good, 2000). To test for autocorrelation, concentration data were regressed against date and, if significant ($p < 0.05$), it was excluded from subsequent permuted t-tests. Differences in changes in TP, PP, and SS concentrations between remediated and control streams as well as differences in P deposition rates across inundation classes and floodplain sides were tested with permuted unpaired two-tailed t-tests (*perm.t.test*). When testing differences in concentration changes for each individual site, paired two-tailed t-tests (*t.test,* stats package; "base" R) were used for normally distributed data and permuted paired two-tailed t-tests (*perm.t.test*) were used for non-normal data. Seasonal differences in P deposition rates were tested with one-way ANOVA (*anova*, stats).

The effects of P deposition and stream water P on labile P in composite sediments were quantified with Pearson correlation coefficients, using *cor.test* in the stats package. No permutation of Pearson correlations was necessary owing to met assumptions of homoscedasticity and normal distribution of residuals. To visualize the controls for P sedimentation, sample distribution of predictor variables (catchment and reach properties) was analyzed using scaled principal component analysis (PCA; *rda*) on two separate matrices for channels and floodplains, using the Vegan package (Oksanen et al., 2022). Vectors of P deposition rates were fitted to the PCAs using *envfit* with 10 000 permutations.

3 Results

3.1 Suspended sediment and P reduction in stream water

Absolute concentrations of TP and PP in stream water were highly variable across both seasons and sites, peaking in summer months during low flows and ranging from below the detection limit to $3940\,\mu g\,TP\,L^{-1}$ and $3557\,\mu g\,PP\,L^{-1}$. Overall, the proportion of PP out of TP ranged from 28 % to 86 % but PP was the dominant form across seven sites (Table S2). Concentrations of SS ranged between < 1 to $1200\,mg\,SS\,L^{-1}$ and correlated with PP concentrations at each site (Table S3), but to a lesser degree in site C1 and S9 at high concentrations. Changes in TP concentrations along upstream control reaches correlated weakly with base flow index ($r = 0.27$, $p < 0.01$) and flashiness index ($r = -0.25$, $p < 0.01$), but there was no general correlation along remediated reaches and thus little influence of different hydrological regimes on P concentration dynamics.

When grouping remediated streams by floodplain design, two-sided floodplains showed higher reductions in absolute TP and PP concentrations along reaches compared with control streams (Fig. 3). Concentration changes along remediated streams with one-sided floodplains were not different than control streams. Correspondingly, there was a similar pattern to changes in TP and PP loads along remediated streams: two-sided floodplains retained on average $0.17 \pm 0.26\,kg\,TP\,ha^{-1}\,yr^{-1}$ and $0.21 \pm 0.21\,kg\,PP\,ha^{-1}\,yr^{-1}$ whereas one-sided floodplains lost $0.07 \pm 0.10\,kg\,TP\,ha^{-1}\,yr^{-1}$ and $0.09 \pm 0.10\,kg\,PP\,ha^{-1}\,yr^{-1}$. Concentration reductions in TP and PP among sites with two-sided floodplains were mostly explained by site C2 but also S8 (Fig. 3). Changes in SS concentrations were not reduced along two-sided floodplains and site C3 significantly increased SS concentrations along remediated reach.

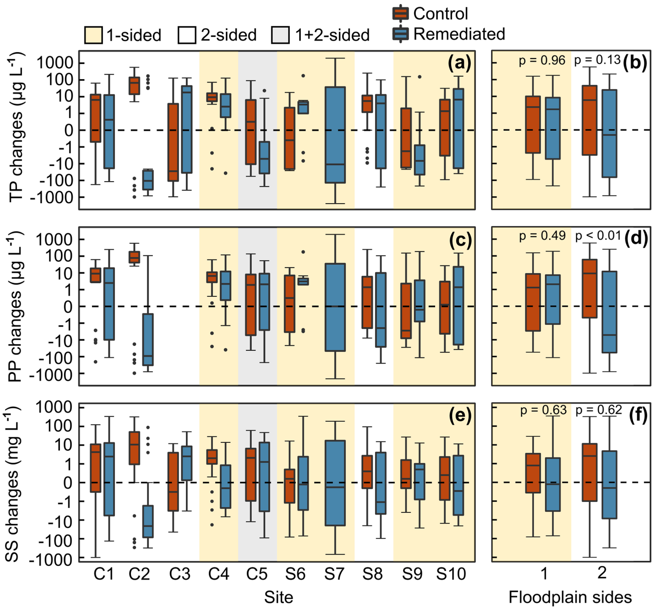

Figure 3. Changes in TP, PP, and SS concentrations (note $\log_{10}$ y axis) between the start and end of control streams (red boxes) and the start and end of remediated streams (blue boxes) from April 2020 to June 2022. Changes in concentrations are shown in **a, c, e** by each site (C1–5 and S6–10) and **b, d, f** sites grouped by number of floodplain sides along the inset channel. When comparing floodplain sides, two sites were excluded owing to a lack of control stream (S7) and a combination of one- and two-sided floodplains along the reach (C5). P values of permuted unpaired t-tests are shown within the panels. P values of parametric and, when necessary, permuted paired t-tests are shown within the panels. P values = NA refers to non-normal and autocorrelated data that prevent reliable computation of t-tests. Bold font denotes significant difference ($p < 0.05$) between locations.

Based on monthly water chemistry data, TP loads ($\mathrm{g\,d^{-1}\,ha^{-1}}$) were reduced during inundation events in site S8, compared with base flows (permuted t-test, $p = 0.02$). This effect was not observed at any other site. Higher incoming loads in two-sided floodplains led to greater load reduction, whereas higher loads in one-sided floodplains led to greater load increases along the remediated reaches (Fig. 4).

3.2 Deposition of sediments and P in channels and on floodplains

Sediment deposition was five times deeper on channel beds ($86 \pm 72\,\mathrm{mm\,yr^{-1}}$) than floodplains ($16 \pm 23\,\mathrm{mm\,yr^{-1}}$) at sites C1–5, S7, and S10 (Fig. 5a), showing that sedimentation on floodplains, despite inundation frequencies up to $200\,\mathrm{d\,yr^{-1}}$, contributed little to the reaches' total sediment budgets. Deposition rates of P correlated with sediment depth ($r = 0.96$, $p < 0.01$) and averaged $29.9 \pm 29.3\,\mathrm{g\,TP\,m^{-2}\,yr^{-1}}$ on channel beds and $7.0 \pm 10.4\,\mathrm{g\,TP\,m^{-2}\,yr^{-1}}$ on floodplains. Combined channel bed and floodplain deposition rates did not correlate with stream water P concentration changes, neither between remediated and control streams and irrespective of the number of floodplain sides (Fig. S12). Ratios of deposited P : sedimentmass did not differ between channel and floodplains (unpaired t-test, $p = 0.83$). There was a seasonal pattern of higher P deposition rates on channel beds during the more hydrologically active winter and spring seasons than during summer and fall (ANOVA, $F_{2,38} = 2.86$, $p = 0.07$).

On channel beds, P deposition rates increased with lower unit stream power ($r = -0.43$, $p = 0.03$; Fig. S13), representing the maximum stream power per $\mathrm{m^{-2}}$ bed surface area when flow was confined to the inset channel. Deposited P correlate with neither the average of stream water TP loads for each 6 month sampling period ($r = -0.16$, $p = 0.42$; Fig. S13) nor with the ratio of unit stream power : TP loads ($r = 0.28$, $p = 0.32$), indicating that unit stream power alone during lower flows was the primary driver for P sedimentation on channel beds.

On floodplains, P deposition rates fitted to a PCA of environmental predictors were primarily controlled by inundation frequency (Fig. S13b), in accordance with lin-

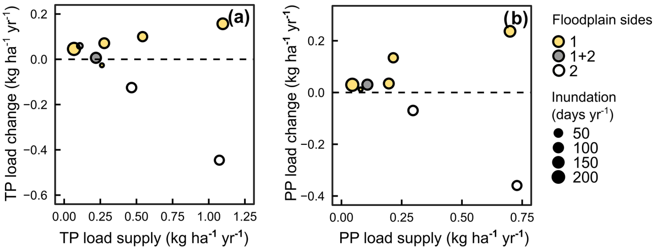

Figure 4. Changes in **(a)** TP and **(b)** PP loads along remediated streams compared with P supply loads at the start of remediated streams. Remediated streams grouped as one-sided (yellow circles), one- and two-sided (gray circles), and two-sided floodplains (white circles). Size of circles denote average floodplain inundation $\mathrm{d\,yr^{-1}}$. TP and PP loads of site C2 were excluded owing to a lack of flow data at downstream location, and PP loads in site C3 was excluded as this was not measured.

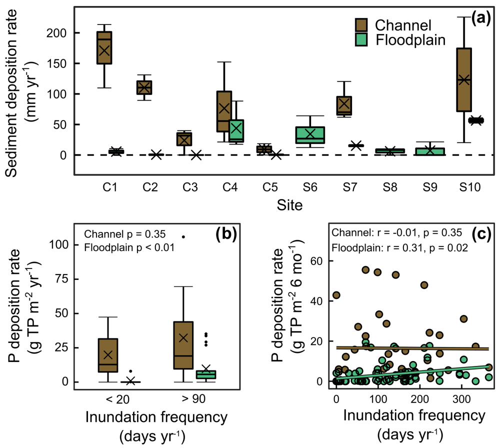

Figure 5. (a) Sediment deposition rate in depth on channel beds and floodplains across sites (C1–5 and S6–10) from sediment plates deployed between September 2021 and May 2022. **(b)** Annual P deposition rate grouped by floodplain inundation frequency classes and **(c)** biannual P deposition rate regressed against inundation frequency during each 6 month sampling occasion. *P* values of permuted unpaired *t*-tests and permuted Pearson correlations are shown within panels.

ear regression (Fig. 5c). Vegetation cover was also dependent on inundation frequency, but when excluding data points < 70 inundation $\mathrm{d\,yr^{-1}}$, there was an effect of higher P deposition rates with vegetation cover, independent of inundation (adj. $R^2 = 0.05$, Vegetation $p = 0.05$, Inundation $p = 0.87$). In contrast to channel beds, unit stream power did not influence P deposition rates on floodplains. Grouping of sites by mean inundation frequency revealed that P deposition rates on floodplains were 10.7 times higher during > 90 inundation $\mathrm{d\,yr^{-1}}$ compared with < 20 inundation $\mathrm{d\,yr^{-1}}$ (Fig. 5b). However, the process of lateral distribution from channel to floodplains could not be confirmed as P deposition rates on channel beds were not influenced by inundation.

3.3 Influence of sediment deposition on P stocks

There was an increase in TP content of composite sediments with construction age of remediated streams, both in channel beds ($r = 0.48$, $p = 0.04$) and floodplains ($r = 0.54$, $p = 0.02$), which was best explained by increases in Al–P fraction. Labile P fractions (H_2O–P and Fe–P) accounted for on average 19 % of TP; hence, the recalcitrant P fractions (Al–P, Org–P, and Ca–P) were the dominant forms of P in sediments (Fig. S14). There was a trend in higher labile P content (paired t-test, $p = 0.06$) in channel sediments compared with floodplain sediments but no differences for TP or recalcitrant P content (paired t-test, TP: $p = 0.12$, Recalcitrant P: $p = 0.23$).

Total P in newly deposited sediments on plates was 1.6 times higher than in composite sediments, i.e., sediments down to a depth of 5 cm outside of the plates. Newly deposited P further correlated with higher labile P content in composite sediments of channels but not floodplains (Fig. 6a). However, deposited P had no effect on recalcitrant P content, meaning that deposited P mainly contributed to the bioavailable P pool in channel sediments (Fig. 6b). Stream water concentrations of TP and PP predicted higher labile P content in channel and floodplain sediments equally well (Fig. 6c, d), indicating that P in the water column supplies bioavailable P on floodplains rather than sediment deposition.

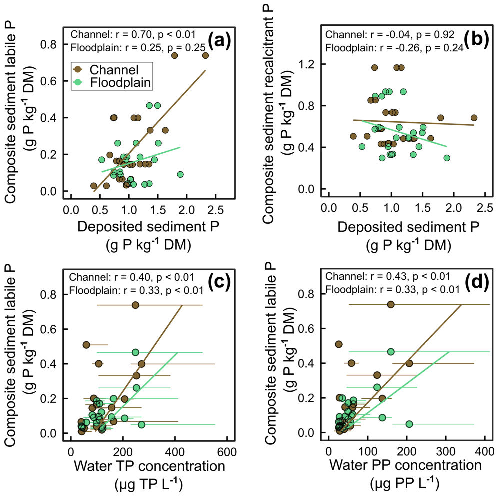

Figure 6. Linear regressions between deposited sediment TP and composite sediment **(a)** labile P and **(b)** recalcitrant P in channel sediments (blue circles) and floodplain sediments (red circles). Linear regressions of composite sediment labile P by **(c)** stream water TP concentrations and **(d)** stream water PP concentrations. Deposited sediments were collected between September 2021 and May 2022, composite sediments sampled in spring 2022 and monthly water grab samples between April 2020 and July 2022. Zero values from plates without deposited sediments were removed from analysis. Labile P represent P fractions H_2O–P and Fe–P. Pearson r and p values are shown within each panel and error bars show standard deviations of stream water P.

4 Discussion

4.1 Influence of floodplain designs on P reductions in streams

The observed reductions in TP and PP concentrations and loads along remediated streams show that this measure has the potential to improve downstream water quality compared with control streams. The effect was exclusive to two-sided floodplain designs; one-sided floodplains did not differ from control streams, which implies that a floodplain on only one side might be less suitable for P mitigation. However, the effect of two-sided designs was highly influenced by site C2, wherein a perched culvert at the outlet could have contributed to additional P removal. The combination of stagnant water conditions at the lower end of the reach and stream bank stabilization resulted in highly favorable conditions for P trapping, without the risk of bank overflow. After removing site C2 from the analysis, the effect of the two-side floodplain was visible, although not statistically significant. Differences in P loads between the two floodplain designs were even more pronounced in sites with higher catchment P supply, as two-sided floodplains responded as active P sinks, whereas one-sided floodplains behaved as passive conduits. However, TP load data were only available at three sites (C1, C3, and S8) with two-sided floodplains, which limits the certainty of this floodplain design being the main control responsible for P load reductions, warranting further study of floodplain designs. Further, flow-weighted average concentration load estimation from monthly water samples is generally conservative (Elwan et al., 2018), which was also the case in the proximity of site C3, where loads based on monthly samples were 30 % lower than loads from biweekly flow-proportional sampling. Remediated streams with lower catchment P loads, due to less agricultural land use, did not reduce P loads irrespective of floodplain design. Thus, to realize their P reduction potential, the results indicate that remediated streams should be targeted in areas with P loading exceeding 0.25 kg P ha^{-1} yr^{-1}.

Reduction in PP loads is particularly important in clay-dominated catchments, associated with high PP delivery from soil erosion to streams (Sandström et al., 2020), as in site C2. The highest reductions in both TP (30 %) and PP (34 %) concentrations occurred along the remediated stream in site C2, characterized by silty clay loam soil texture. Although the majority of the studied catchments were dominated by PP, site S8, which (in contrast to PP-dominated C2) was characterized by equal proportions of PP and SRP and loam soil texture, also showed a tendency toward reductions in TP, PP, and SS concentrations. This implies that remediated streams can be effective across diverse catchments with contrasting soil textures and P forms, if the criteria of high P loading is met. In addition, site S8 showed that remediated streams with frequent inundation and without additional ponding, as in C2, can improve downstream water quality. Site S8 has also been subject to re-meandering, which could account for increased water residence times and settling of particulates, in addition to its floodplains. However, among two-sided floodplain sites, C1 and C3 did not reduce P and SS concentrations, implying that solely implementing two-sided floodplains is insufficient and that further requirements such as frequent inundation, floodplain stability, and reach lengths must be met by proper design of two-sided sites. For instance, site C1 is characterized by a short and straight reach with a low flooding-to-channel ratio and unstable floodplains that prevent P and SS reduction.

We assumed that lateral inputs (tile drains and groundwater inflow) along both remediated and control reaches were comparable and thus not influencing the comparison between the two reaches. All paired reaches received tile drain inputs from adjacent fields with identical crop cultivation and flat topography with predominant subsurface flow pathways. There were no large deviations in loads between up- and downstream and we do not therefore have any reasons for suspecting significant lateral hotspots.

Although reductions in P loads were predicted during inundation events (Bai and Zeng, 2019; D'Ambrosio et al., 2015; Krider et al., 2017) this was only observed in site S8. However, the lower number of water samples collected during these events limited the comparison between base flow and inundation conditions, highlighting the need for high-resolution monitoring to resolve P responses to storm events. The lack of P reductions along streams with one-sided floodplains was likely connected to higher erosion from banks next to inset channels. These banks had higher slopes and resembled trapezoidal banks, compared with banks adjacent to floodplains (data not shown; Fig. 1). Higher bank slopes in loam and clay soils are known to reduce the cohesive strength and increase the vulnerability toward erosion of bank material to the water column (Thorne and Tovey, 1981). In one-sided designs, banks next to inset channels were also exposed to higher stream velocities; the greater water depth in channels than in floodplains led to higher water velocities that exerted greater shear stress on banks adjacent to the channel. Accordingly, we suggest that these processes that drive bank erosion together can explain the lack of P reductions with one-sided floodplains, consistent with previous observations of increased scour and deposition with this floodplain design (D'Ambrosio et al., 2015).

4.2 Influence of sediment deposition on P dynamics and channel morphology

Contrary to our initial assumption, higher P deposition rates on floodplains did not reduce P concentrations in stream water and could therefore not be confirmed as the sole process responsible for SS and P reduction in remediated streams. Instead, there was an average increase in TP concentrations of $1.6\,\mu g\,TP\,L^{-1}$ per deposited $g\,TP\,m^{-2}\,yr^{-1}$, suggesting that floodplain deposition co-occurred with other processes contributing to P losses, e.g., resuspension. Deposition rates of sediments and P on floodplains were controlled by inundation frequency and were substantially higher in this study than reported rates from five US sites (0.5–13 mm sediment yr^{-1}; D'Ambrosio et al., 2015) and three Swedish sites (0.04–1.2 $g\,TP\,m^{-2}\,yr^{-1}$; Lacoursière and Vought, 2020).

The relative importance of vegetation cover for trapping sediment-bound P on floodplains could only be assessed in locations with > 70 d of inundation yr^{-1}, without collinearity between the two variables. Provision of water and nutrients from inundation events increases biomass growth on floodplains. It can therefore be assumed that sediment deposition is promoted directly by inundation, which provides sediments to be settled, and indirectly by its effect on vegetation, which also traps sediments. However, the smooth surfaces of sediment plates that were used to measure deposition likely underestimated sediment trapping on densely vegetated floodplains.

Despite the relatively high sediment yields on floodplains, particularly at sites C4, S6, and S10, its lack of effect on P concentrations observed in our study contradicts previous findings, reporting floodplains as consistent P traps (D'Ambrosio et al., 2015; Davis et al., 2015; Mahl et al., 2015). An explanation for this could be that in our study sites, flooding to channel width ratios were significantly smaller than the recommended ratio of 3 to 5 (Powell et al., 2007) and therefore not contributing to sufficient reach-scale P deposition needed to impact P in the water column. The unexpected increase in P concentrations with floodplain deposition could be due to coinciding activation of erosion processes in channels, banks, or floodplains that override reductions from sedimentation. Total P losses during increasing floodplain deposition were primarily explained by mobilization or increased transport of PP, accounting for 87 % of TP losses, and not SRP. Overall, surface erosion in catchment soils did not drive stream sediment losses. Modeled maximum erosion rates of SS from surface runoff (Djodjic and Markensten, 2019) were generally low for all sites (10–50 $kg\,ha^{-1}\,yr^{-1}$), except for site S7 (100–250 $kg\,ha^{-1}\,yr^{-1}$), which likely contributed to SS and PP loading in that particular site. It is further possible that rapid resuspension of newly deposited sediments during higher flows leads to net losses of stream water P. As monitoring with sediment plates only accounted for net deposition, we could not quantify losses of sediments co-occurring with deposition.

Based on sediment deposition rates on plates, there was no indication of lateral sediment distribution from channel beds to floodplains in the seven sites (C1–5, S7, S10) where both channel beds and floodplains were evaluated. Instead, channel beds represented the dominant pool of sediment storage in remediated streams, independent of floodplain deposition rates. The ongoing aggradation on the channel beds indicates that these systems are still in fluvial disequilibrium, which is likely an indirect effect of floodplain inundation that also lowers water velocities in channels. However, we could not show if this also occurred in control streams as no channel beds were monitored in these reaches. The lack of correlation between channel deposition and reductions in P concentrations in stream water could be explained by the bi-annual sediment sampling interval, being too coarse to capture periodic P reduction from sedimentation, which has been reported to occur during lower flow conditions (Bowes et al., 2003). In sites S6 and S8–9, where channel slopes ranged between 0.2 % and 0.6 %, channel bed material was dominated by gravel and pebble substrates. Their impenetrable channel beds prevented plate installation and measurement of sedimentation rates but based on visual observations, no substantial deposition of fine sediments occurred over the study period. Hence, these channels appeared to have sufficient stream power in relation to SS loads to transport SS, driven by their higher longitudinal channel slopes.

4.3 Variability in sediment P stock

The dominance of P deposition on channel beds compared with floodplains was also reflected in the higher labile P fractions (H_2O–P and Fe–P) in composite channel bed sediments, which increased with P deposition rates. Although labile P fractions only accounted for 20 % of TP in composite sediments, Fe–P (the largest fraction of labile P) is sensitive to reducing conditions and subsequent re-release as SRP into the water column (Jarvie et al., 2005). The contribution of labile P from sediment deposition can therefore have negative implications for P export, supplying the system with easily available P rather than providing a persistent P trap. Labile P in composite sediments also correlated with stream water P concentrations, but here the causation was less clear owing to the reciprocity between sediment-bound P and stream water P; higher P concentrations in stream water can increase sediment P (deposition and sorption) whereas high sediment P can also lead to higher stream water P (resuspension and desorption).

The range of TP content in channel bed sediments was consistent with other agricultural streams in Sweden (0.07–1.57 $g\,P\,kg^{-1}$, Lannergård et al., 2020; 0.35–1.85 $g\,P\,kg^{-1}$, Sandström et al., 2021), indicating that remediated streams do not promote P enrichment in channel beds compared with streams without floodplains. However, the proportions of labile P fractions were overall lower in our study, either explained by less deposition of these fractions or a more rapid turnover of labile P. In a comparable system, sediments were also dominated by recalcitrant fractions that were shown to be stable over seasons (Kindervater and Steinman, 2019). The increase in Al–P with construction age was partly explained by higher clay content in catchment soils, which often are rich in Al silicates and hydroxides that increase the sorption of P to these surfaces (Gérard, 2016). However, the effect on TP with construction age was independent from clay content, suggesting a true accumulation of P in sediments over time in both channel and floodplain sediments. This post-construction maturation process indicates a system reset to low P content in sediments following construction, as previously shown for denitrification and microbial respiration rates in floodplain sediments (Speir et al., 2020).

The higher vegetation coverage on floodplains than on channel beds was not reflected in higher organic P fraction in sediments. Although soil organic carbon has been found to accumulate more over time in floodplain sediments than in channel sediments (Hallberg et al., 2022; Speir et al., 2020), the lack of increase in organic P in floodplain sediments with construction age suggests that the build-up of organic P forms, despite the presence of vegetation, is slow and requires more than 10 years to occur. In this study, P in vegetation biomass was not considered and although it could contribute as a seasonally active P sink, Kindervater and Steinman (2019) reported that channel and floodplain sediments on average held 18 times more P mass than biomass on a reach-scale.

4.4 Floodplain management toward P reduction and fluvial equilibrium

Currently in Sweden, the recommended dimensions of remediated streams with two-stage design (1 km length, 3–5 flooding to channel width ratio) are solely based on hydrology and do not take into account impact on sediment or nutrient reduction (Lindmark et al., 2013; Powell et al., 2007). Moreover, subsidies from Swedish authorities compensate for the stream length when constructing remediated streams and there is no mechanism for incentivizing the implementation of wider floodplains, resulting in narrower than recommended widths in remediated streams. We suggest that wider floodplains and shorter lengths can have larger effects on P reductions by sedimentation, improving cost-efficiency while maintaining the same total area. It is, therefore, also necessary to evaluate this within a modeling framework that can isolate a range of dimensions from other confounding, site-specific factors.

A one-sided floodplain design is sometimes preferred among landowners when constructing floodplains owing to the lower cost, easier implementation, and future maintenance accessibility, especially in locations where riparian obstructions (e.g., trees, boulders, and high bank elevation) inhibit the two-sided design. However, our results show clearly that this design is insufficient to reduce P concentrations and loads compared with control streams and therefore should be avoided. Instead, priority should be given to establishing two-sided floodplains in critical source areas with high P loading, where riparian conditions as well as channel slope allow this.

As higher inundation frequency (provided by lower floodplain elevations) increased P deposition, it is possible that lower and wider floodplains can increase reach-scale P trapping and thereby reduce P exports. However, lower floodplains can be exposed to higher shear stress and the risk of toe erosion of floodplains that can explain the lack of observed P reductions with deposition. When choosing the optimal floodplain elevation, multiple pollutants (e.g., nitrogen; Hallberg et al., 2022) should be considered to avoid potential pollution swapping (Bieroza et al., 2019).

Combining floodplains with partial damming, as in site C2, can lead to higher reductions in SS and P concentrations. Littlejohn et al. (2014) reported 45 % TP load reductions with two low-grade weirs in streams with constructed floodplains, comparable with that of site C2. Thus, raising culverts or installing low-grade weirs can complement remediated streams. The increased bankfull volume with constructed floodplains reduces the risk of flooding when restricting outflows; however, hydraulic calculations should be made to ensure that large flood volumes are confined within the bankfull volume when installing impoundments. Although the combination of re-meandering and floodplains resulted in lower SS and P export in site S8, stable meanders can be difficult to establish without additional protective measures to avoid erosion and migration (Wohl et al., 2015). Results on P mitigation with re-meandered channels are scarce but it has been shown to improve stream habitat quality (Lorenz et al., 2009). However, meanders rely on a certain freedom of channel migration that can infringe on productive agricultural land, lead to higher sediment (and thus P) exports, and are therefore less suitable in narrow riparian buffers with limited room for channel evolution. Sediment accumulation on channel beds is widely perceived as negative by landowners, requiring routine channel dredging to avoid drainage impairment and to reduce the risk of flooding. However, in the absence of maintenance activities, the natural formation of floodplains in agricultural streams has been observed to develop from sediment deposition in channels, facilitated by over-widened channels that limit stream power and bed load transport (Jayakaran and Ward, 2007; Landwehr and Rhoads, 2003). As such, the predominant accumulation of channel bed sediments in our studied sites could indicate that there is an ongoing adjustment toward floodplain formation, in addition to existing floodplains. The space to allow channel sedimentation before it puts conveyance capacity at risk is also greater in remediated streams and could motivate abstention from dredging to allow stabilization of fluvial equilibrium and adjustments of floodplains to the streams' fluvial conditions. Enhancing this potential by providing more space and designing wider floodplains is therefore important to effectively reduce P exports, rather than resorting to frequent channel dredging that systematically resets the fluvial imbalance. As future climate will bring higher frequencies of hydrological extremes and consequently higher P exports (Mehdi et al., 2015; Ockenden et al., 2017), remediation of agricultural streams can offer improved flood control, together with enhanced water purification.

5 Conclusions

Human pressures on freshwater resources, exacerbated by climate change, require urgent adjustment in agricultural streams to improve their capacity to transport and process large water and pollution fluxes. To date, nutrient mitigation strategies focus largely on in-field and point measures (e.g., soil management and wetlands), ignoring the enormous potential of headwater agricultural streams and their corridors as interfaces for reducing multiple pollutants (SS, N, and P) while also providing flood control. Evaluating the capacity for stream remediation in agricultural landscapes to reduce SS and P export, as carried out in this study, is therefore a critical step in improving our understanding of how agricultural streams can be modified to reduce downstream water quality impacts.

In this study, we demonstrated that reductions in stream water P concentrations and loads along remediated streams can be achieved with floodplains constructed laterally on both sides of the channel, compared with one-sided floodplains and control streams (trapezoidal channels). Higher catchment supply of P, driven by agricultural land use, also increased P reduction capacity in two-sided remediated streams. By linking fluvial dynamics with P storage and water quality impacts, our results showed that with narrow floodplains, PP reductions were controlled by protection against erosion rather than by the promotion of deposition. In this sense, remediated streams operate according to a different principle than flow-through wetlands, with the focus on reducing erosion and near-stream P inputs rather than actively trapping upstream-derived sediments. However, by extending the width of floodplains, the importance of P deposition for PP reductions is expected to increase. The fact that heavily impacted catchments in this study delivered high loads of N and P, as is typical in agricultural streams, underscores the need to adopt management strategies that can maximize reductions of multiple pollutants and minimize the risk for pollution swapping. Although the optimal design for PP reduction also has the potential to promote N removal, there is a further need to understand the effect on SRP dynamics when promoting high inundation frequencies and deposition of sediments with labile P.

Data availability. The water and sediment chemistry and hydrology measurements from the studied sites are provided as a Supplement.

Supplement. The supplement related to this article is available online at: https://doi.org/10.5194/hess-28-341-2024-supplement.

Author contributions. LH: conceptualization, methodology, data curation, formal analysis, visualization, writing – original draft, writing – review and editing. FD: methodology, formal analysis, writing – review and editing. MB: conceptualization, methodology, formal analysis, writing – review and editing, funding acquisition.

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Supplementary information of:

Phosphorus supply and floodplain design govern phosphorus reduction capacity in remediated agricultural streams

1. Catchment characteristics and spatial monitoring design

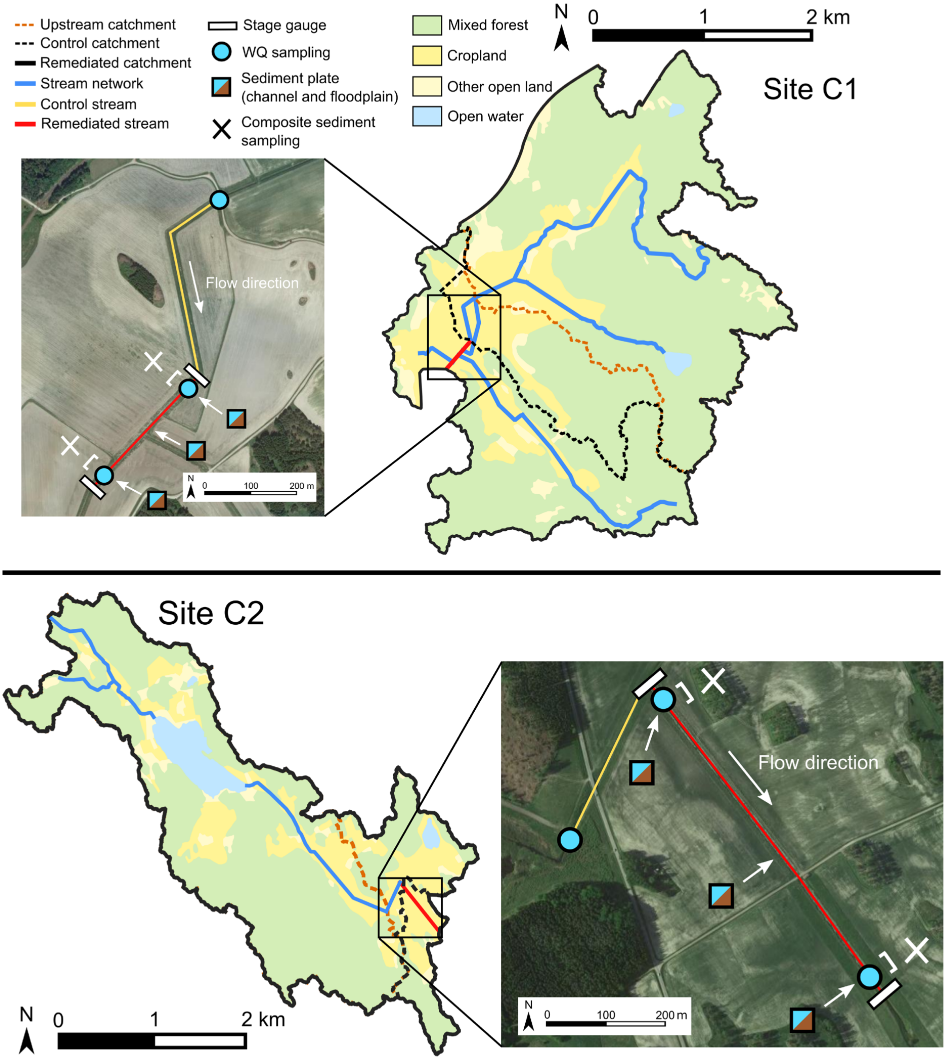


Fig. S1. Land use cover and catchment boundaries of remediated and control streams (C1 and C2). Location of stage gauges, water sampling, sediment plates and sediment sampling. Satellite images: Google, ©2023 Maxar Technologies. Land use maps: ©Lantmäteriet.

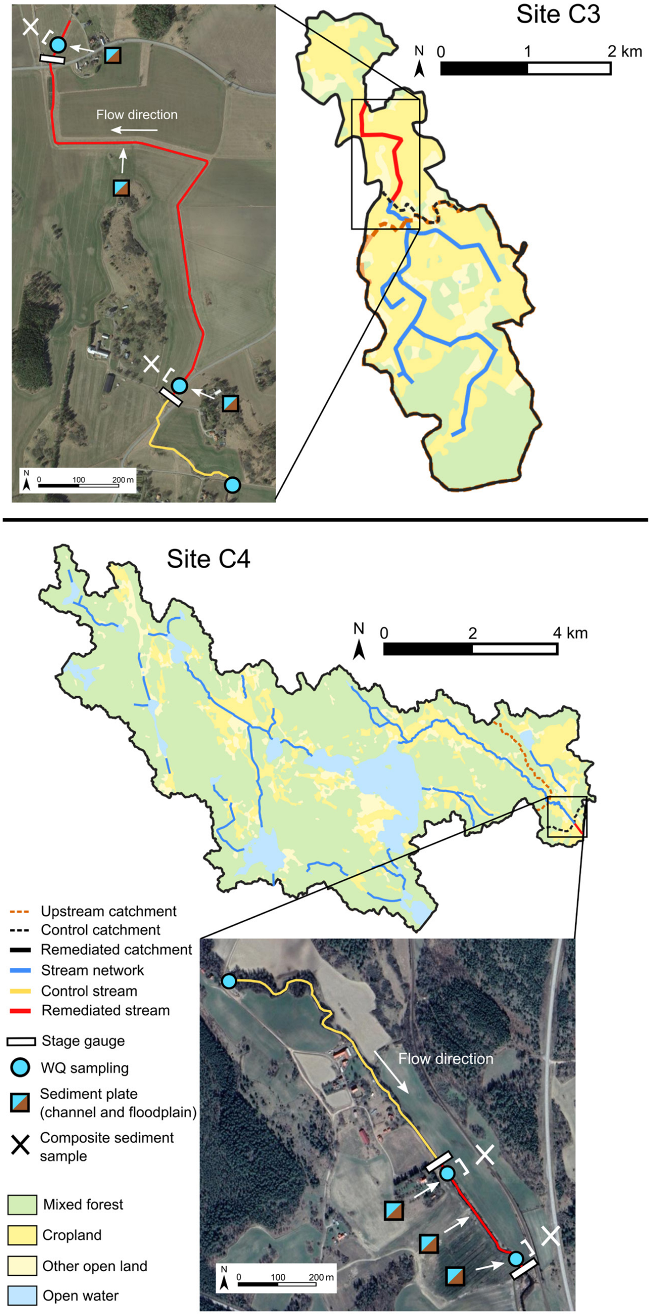


Fig. S2. Land use cover and catchment boundaries of remediated and control streams (C3 and C4). Location of stage gauges, water sampling, sediment plates and sediment sampling. Satellite images: Google, ©2023 Maxar Technologies, ©2023 CNES / Airbus. Land use maps: ©Lantmäteriet.

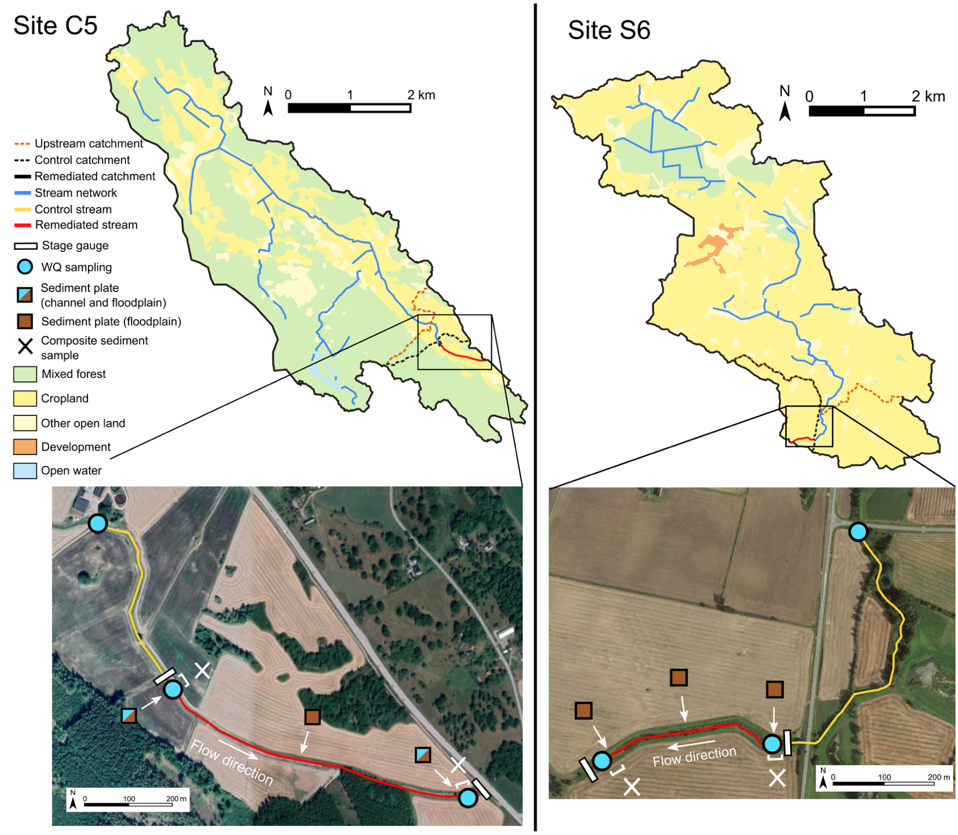


Fig. S3. Land use cover and catchment boundaries of remediated and control streams (C5 and S6). Location of stage gauges, water sampling, sediment plates and sediment sampling. Satellite images: Google, ©2023 CNES / Airbus. Land use maps: ©Lantmäteriet.

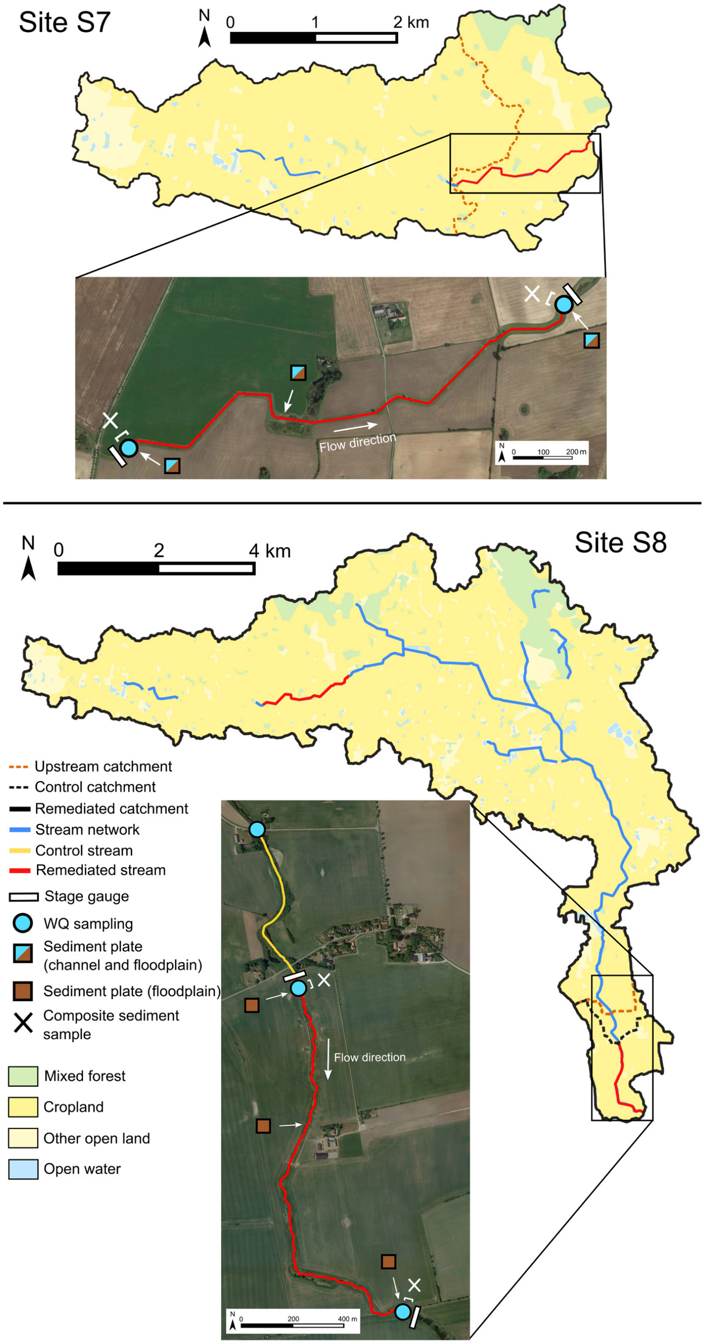


Fig. S4. Land use cover and catchment boundaries of remediated and control streams (S7 and S8). Location of stage gauges, water sampling, sediment plates and sediment sampling. Satellite images: Google, ©2023 Landsat / Copernicus. Land use maps: ©Lantmäteriet.

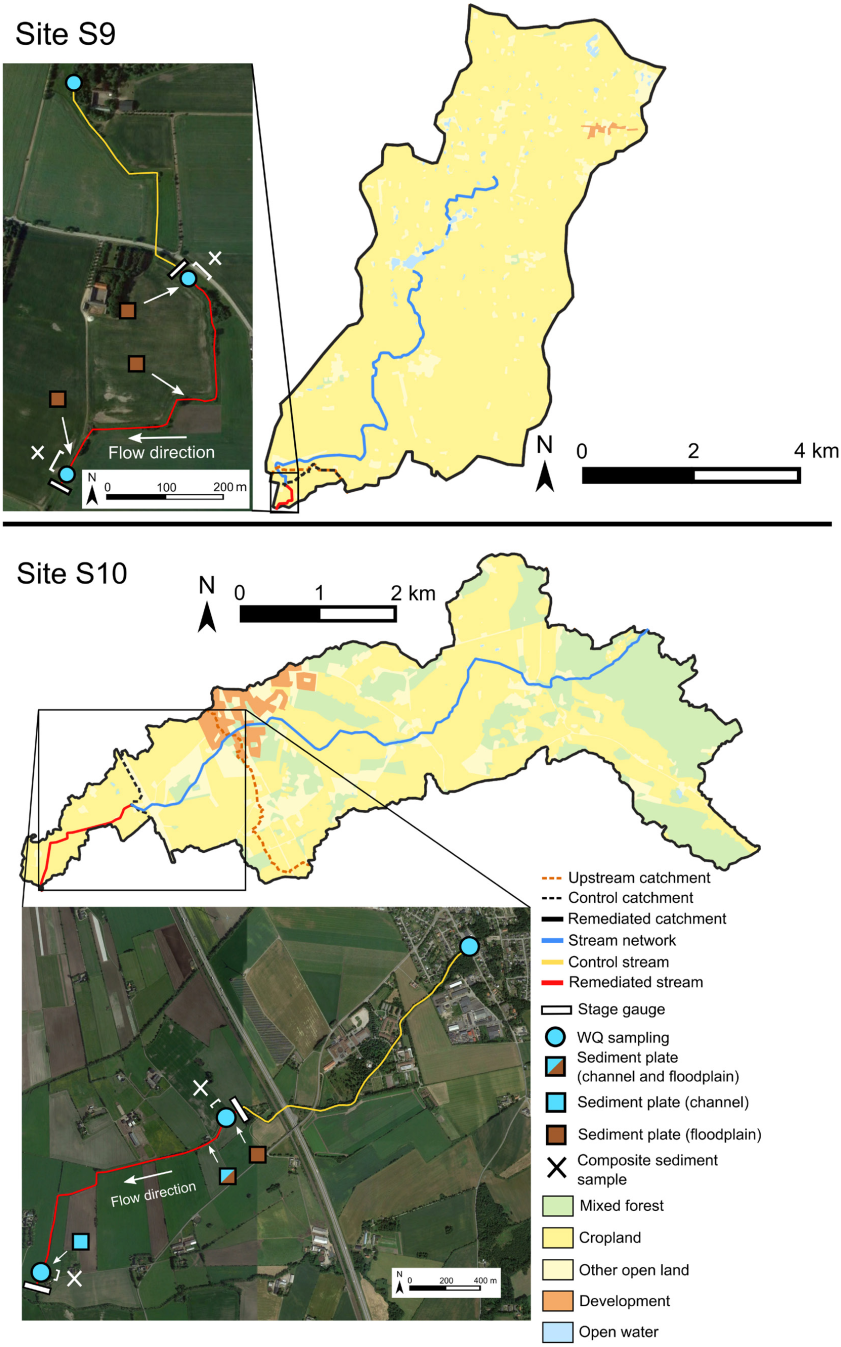


Fig. S5. Land use cover and catchment boundaries of remediated and control streams (S9 and S10). Location of stage gauges, water sampling, sediment plates and sediment sampling. Satellite images: Google, ©2023 Landsat / Copernicus. Land use maps: ©Lantmäteriet.

Table S1. Characteristics of sub-catchments draining control and remediated reaches (upstream catchment excluded) and channel geomorphology of control and remediated streams.

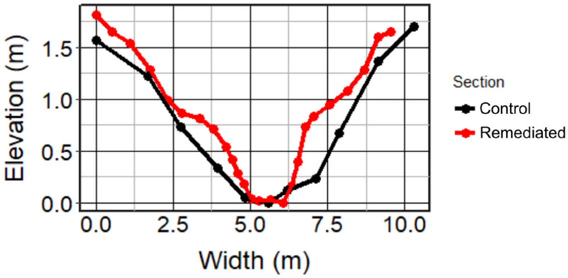
Site	Reach	Sub-catchment area (ha)	Agricultural land use (%)	Reach length (m)	Channel bed width (m)	Bank elevation (m)	Channel slope (-)
C1	Control	127	24	440	2.39	1.67	0.08
	Remediated	294	22	340	1.03	1.77	0.04
C2	Control	100	37	290	1.71	1.79	0.08
	Remediated	54	46	730	0.87	1.70	0.04
C3	Control	16	77	450	0.89	1.27	0.77
	Remediated	202	65	1500	1.01	2.24	0.30
C4	Control	326	35	900	-	-	0.37
	Remediated	46	46	320	2.63	1.41	0.10
C5	Control	52	60	450	1.57	2.32	0.37
	Remediated	184	14	780	1.08	1.79	0.14
S6	Control	236	84	620	1.74	1.70	0.71
	Remediated	102	92	400	0.95	1.62	0.17
S7	Remediated	298	82	1960	0.86	1.97	0.09
S8	Control	46	81	650	-	-	0.68
	Remediated	200	90	1770	1.30	1.33	0.47
S9	Control	13	93	450	-	-	0.25
	Remediated	42	90	630	1.22	1.37	0.44
S10	Control	234	61	1780	-	-	0.27
	Remediated	100	83	1760	6.00	2.17	0.09

2. Remediated and control stream profiles

C1

Remediated stream
(upstream looking downstream)

Control stream
(upstream looking downstream)



C2

Remediated stream
(downstream looking upstream)

Control stream
(upstream looking downstream)

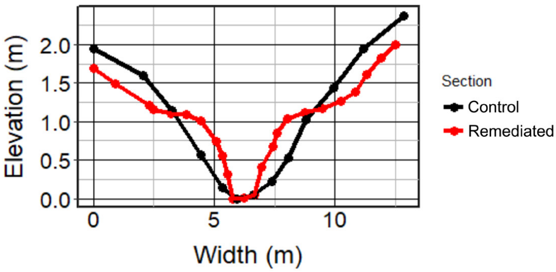
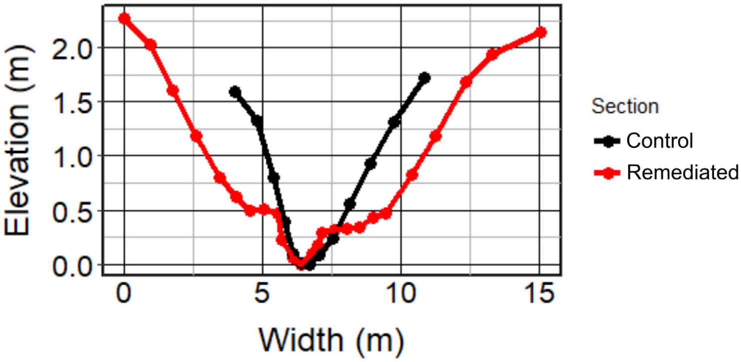


Fig. S6. Photographs and cross-section profiles of remediated and control streams at site C1 and C2.

C3

Remediated stream
(midstream looking downstream)

Control stream
(upstream looking upstream)



C4

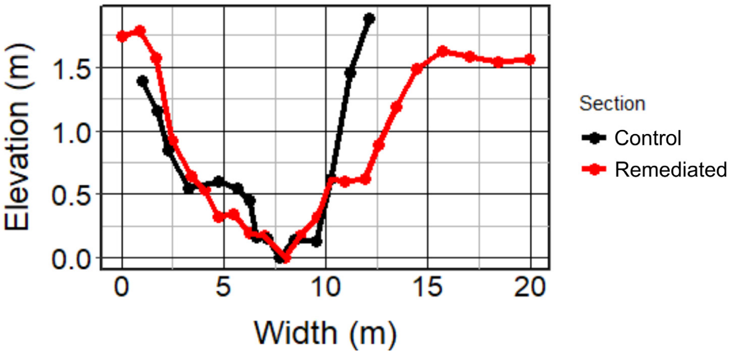
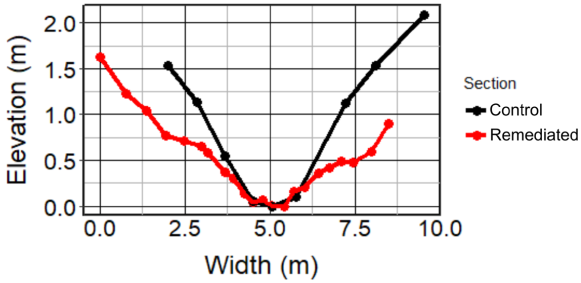


Fig. S7. Photographs (site C3) and cross-section profiles of remediated and control streams at site C3 and C4. No photographs taken of site C4 reaches.

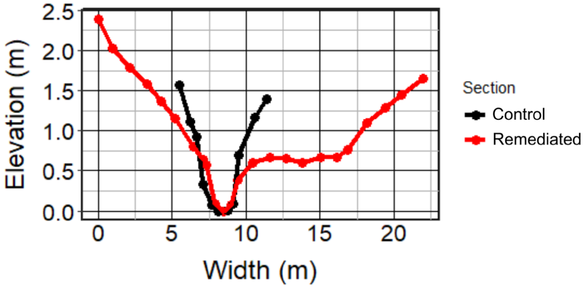
C5

Remediated stream
(upstream looking downstream)



S6

Remediated stream
(midstream looking upstream)



S7

Remediated stream
(upstream looking downstream)

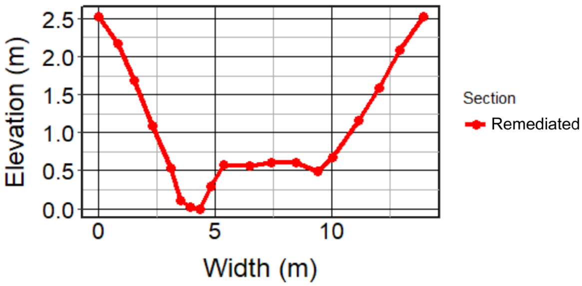
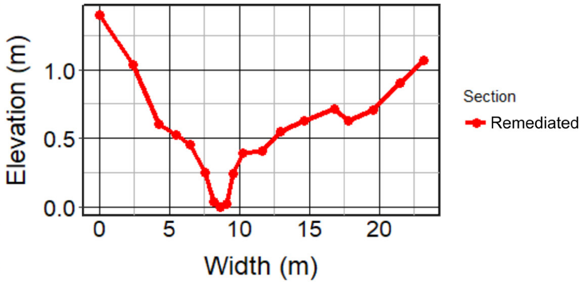


Fig. S8. Photographs and cross-section profiles of remediated and control streams at site C5, S6-7. No photographs taken of control streams (C5, S6) and no control stream exist for site S7.

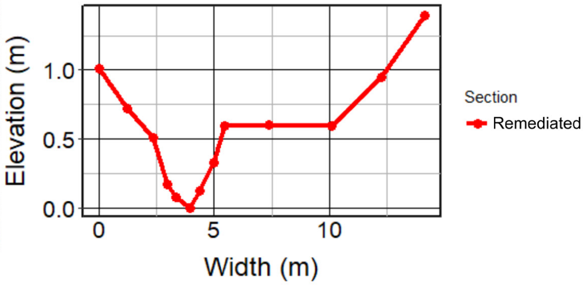
S8

Remediated stream
(upstream looking downstream)



S9

Remediated stream
(midstream looking upstream)



S10

Remediated stream
(downstream looking upstream)

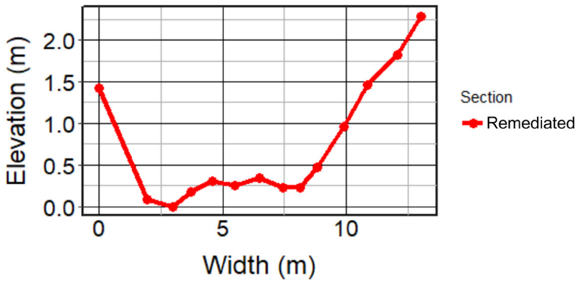


Fig. S9. Photographs and cross-section profiles of remediated streams at site S8-10. No photographs taken and no cross-section measured of control streams in site S8-10.

2. Sedimentation plates

Fig. S10. Photographs of sedimentation plates. Sedimentation plate deployed on floodplain surface (left). Sediment yield after retrieving sedimentation plate placed in channel, using a cylindrical frame to prevent sediment losses to water (right).

3. Validation of load estimation

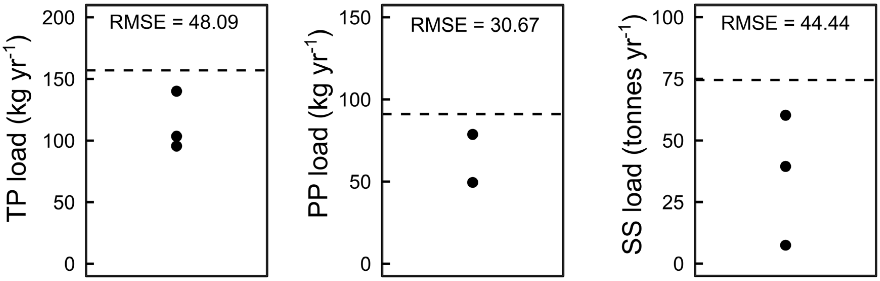


Fig. S11. Variation in calculated loads from a location 1 km downstream of site C3. Horizontal dashed lines denote accurate annual loads from fortnightly flow-proportional sampling during April 2020 to July 2022. Black circles denote annual loads calculated with flow-weighted mean concentration using water samples from two to three different days within each month over the study period. Root mean square error (RMSE) is a measure of bias and standard deviation between flow-weighted mean concentration and accurate loads.

4. Mean concentration and annual loads of P

Table S2. Mean concentrations ± one standard deviation and loads of total phosphorus (TP), particulate phosphorus (PP) and suspended sediments in control streams (TD), upstream remediated streams (US) and downstream remediated streams (DS). Samples were collected between April 2021 and June 2022. PP in site C3 was calculated as the difference of unfiltered TP and unfiltered reactive P. Longitudinal differences in water quality parameters were tested for each site with one-way ANOVA. Bold font denotes significant difference ($p < 0.05$) between locations.

Site	Location	Total phosphorus		Particulate phosphorus		Suspended sediments	
		Concentration ($\mu g\ L^{-1}$)	Load ($kg\ ha^{-1}\ yr^{-1}$)	Concentration ($\mu g\ L^{-1}$)	Load ($kg\ ha^{-1}\ yr^{-1}$)	Concentration ($mg\ L^{-1}$)	Load ($tonnes\ ha^{-1}\ yr^{-1}$)
C1	TD	98.86 ± 70.92		67.88 ± 59.07		21.60 ± 13.34	
	US	102.30 ± 53.65	0.47	72.18 ± 41.70	0.30	38.85 ± 42.77	0.11
	DS	112.27 ± 73.74	0.34	88.82 ± 66.11	0.23	54.89 ± 100.91	0.09
C2	TD	**339.50 ± 322.77**		**205.39 ± 253.12**		**57.32 ± 93.46**	
	US	**385.21 ± 216.26**	0.51	**293.34 ± 200.56**	0.36	**78.04 ± 83.77**	0.11
	DS	**206.24 ± 104.88**		**122.35 ± 61.95**		**24.87 ± 22.04**	
C3	TD	360.53 ± 346.56		170.51 ± 379.24		15.27 ± 14.18	
	US	240.82 ± 206.44	0.11	58.58 ± 103.47		14.13 ± 22.27	0.01
	DS	212.20 ± 155.07	0.17	63.74 ± 75.73		21.78 ± 32.34	0.01
C4	TD	**32.21 ± 13.55**		20.83 ± 12.04		8.69 ± 6.19	
	US	**47.44 ± 22.25**	0.07	31.80 ± 18.00	0.05	12.23 ± 6.98	0.02
	DS	**62.47 ± 54.62**	0.11	42.84 ± 46.27	0.07	11.19 ± 10.53	0.02
C5	TD	111.69 ± 51.08		**59.27 ± 28.91**		11.59 ± 5.97	
	US	131.90 ± 92.27	0.22	**84.06 ± 85.06**	0.11	30.22 ± 47.09	0.03
	DS	105.73 ± 50.86	0.23	**72.87 ± 51.16**	0.14	31.79 ± 34.28	0.05
S6	TD	88.19 ± 52.06		27.46 ± 10.31		20.17 ± 49.75	
	US	93.19 ± 55.09	0.54	26.79 ± 8.70	0.22	13.60 ± 24.33	0.08
	DS	104.48 ± 83.77	0.64	48.62 ± 64.62	0.35	40.10 ± 109.49	0.31
S7	TD	-		-		-	
	US	433.86 ± 774.73	1.10	309.48 ± 707.19	0.70	103.62 ± 209.82	0.31
	DS	474.22 ± 606.19	1.25	398.79 ± 589.92	0.94	54.40 ± 82.07	0.34
S8	TD	117.15 ± 59.18		59.80 ± 54.54		18.33 ± 35.60	
	US	145.67 ± 86.31	1.07	84.44 ± 78.59	0.73	24.50 ± 37.29	0.32
	DS	128.88 ± 51.59	0.63	61.84 ± 40.91	0.37	17.43 ± 19.32	0.14
S9	TD	149.62 ± 141.82		63.29 ± 98.83		10.94 ± 10.99	
	US	145.93 ± 111.35	0.26	60.35 ± 81.40	0.08	11.95 ± 14.36	0.02
	DS	143.55 ± 158.12	0.23	55.26 ± 58.55	0.09	9.91 ± 15.62	0.02
S10	TD	55.44 ± 48.29		38.19 ± 35.47		7.16 ± 8.86	
	US	54.00 ± 33.97	0.28	37.02 ± 27.41	0.20	9.08 ± 9.13	0.05
	DS	66.56 ± 48.56	0.35	45.28 ± 44.27	0.23	13.55 ± 28.34	0.05

5. Correlation between PP and SS concentrations

Table S3. Linear correlation between particulate phosphorus (PP) and suspended sediments (SS) stream water concentrations across control streams and remediated streams. Correlation assessed by coefficient of determination (R^2) and statistically significance of regression slope ($p < 0.05$). Samples were collected between April 2021 and June 2022. Correlation of site C3 was based on samples from upstream of control stream, being the only location with measured filtered and unfiltered TP for this site.

PP ~ SS correlation		
Site	R^2	P value
C1	0.46	< 0.01
C2	0.61	< 0.01
C3	0.51	< 0.01
C4	0.68	< 0.01
C5	0.59	< 0.01
S6	0.81	< 0.01
S7	0.65	< 0.01
S8	0.84	< 0.01
S9	0.23	< 0.01
S10	0.62	< 0.01

6. Changes in P concentrations across sediment P deposition rates

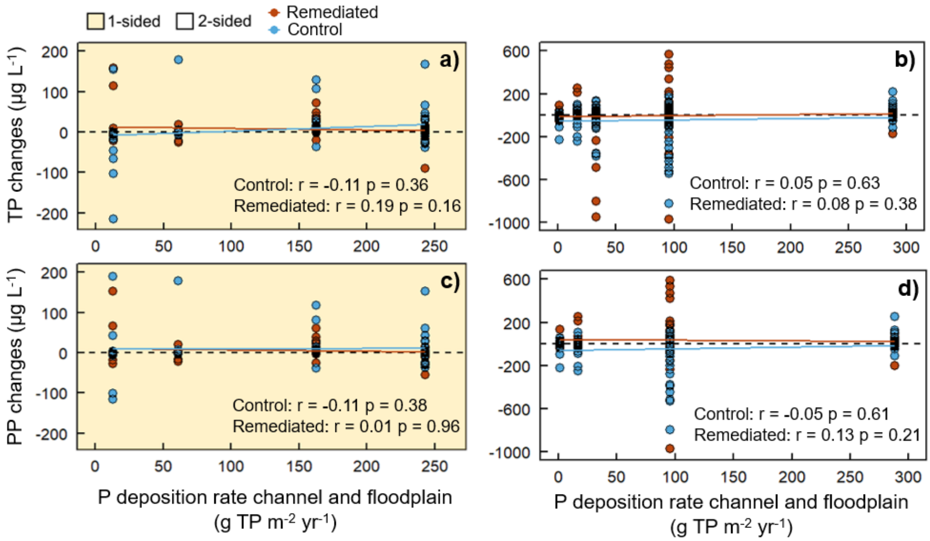


Fig. S12. Correlations between stream water P concentration changes along remediated and control streams and P deposition rates on channel beds and floodplains of remediated streams. Changes in TP concentrations along a) one-sided floodplains and b) two-sided floodplains and changes in PP concentrations along c) one-sided floodplains and d) two-sided floodplains. For sites without P deposition monitoring on channel beds (S6, S8–9), only floodplain deposition is shown. Site S7 was excluded due to no control stream. P-values of Pearson correlations are shown within each panels.

7. Catchment drivers for sediment P deposition

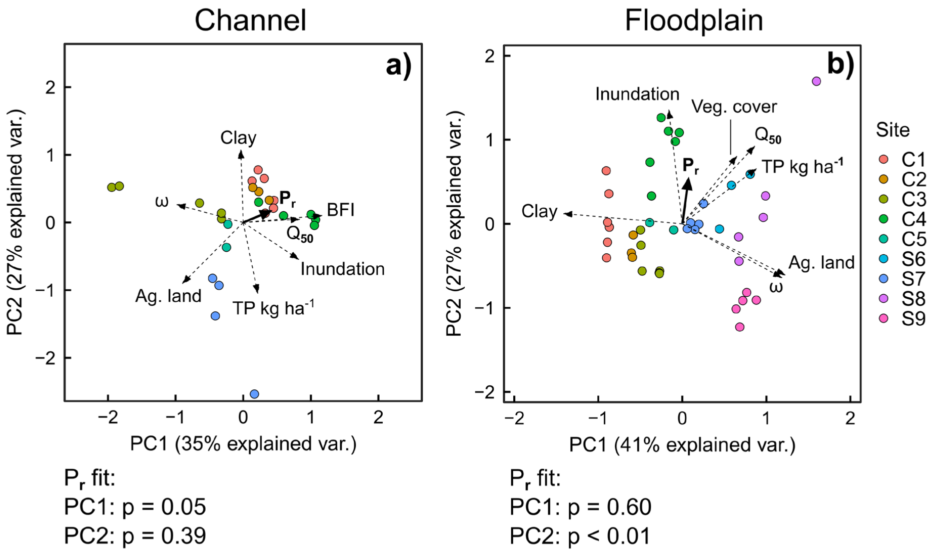


Fig. S13. Principal component analysis (PCA) of predictor variables (water chemistry, catchment and channel properties). Correlation with P deposition rates (P_r) on a) channel beds and b) floodplains. The influence of predictor variables on sample distributions are indicated by dashed vectors. The P_r was significantly correlated ($p < 0.05$) with the ordinations and is shown as blue vectors, with lengths proportional to the strength of the correlation. Circle color denote samples from 9 sites. Sample sites with missing variables were removed from the analyses and descriptor variables were standardized to equal standard deviations. ω = unit stream power, Q_{50} = median flow discharge, BFI = base flow index, SS load = 2.5 yr mean of suspended sediments loads (tonnes yr^{-1}).

8. Phosphorus fractions and content in composite sediments

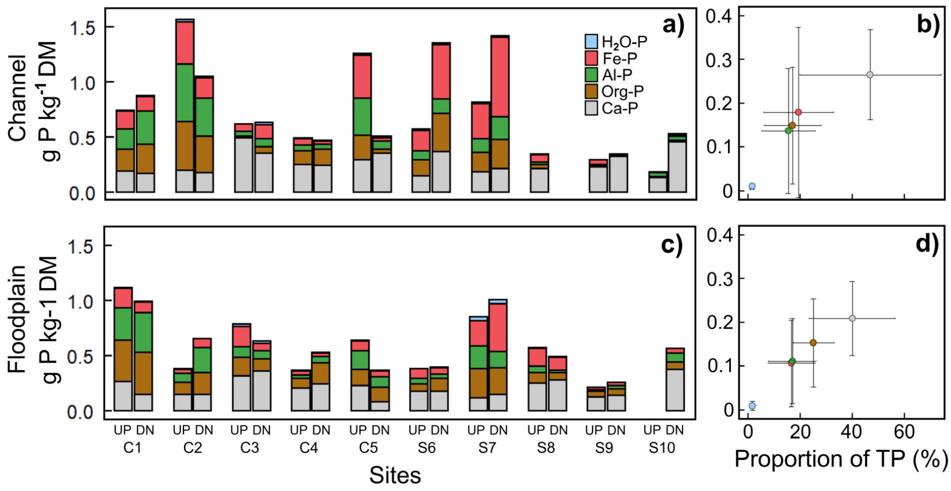


Fig. S14. Mass of P fractions in composite sediments in a) channel beds and c) floodplains of remediated streams across all studied sites. The relation between P fraction mass and proportions of total P in b) channel bed and d) floodplain sediments, with standard deviations shown as error bars. H_2O-P = water soluble P, Fe-P = P adsorbed to iron and manganese, Al-P = P adsorbed mainly to aluminum, Org-P = organic-bound P and Ca-P = calcium-bound P.

II

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Catchment controls of denitrification and nitrous oxide production rates in headwater remediated agricultural streams

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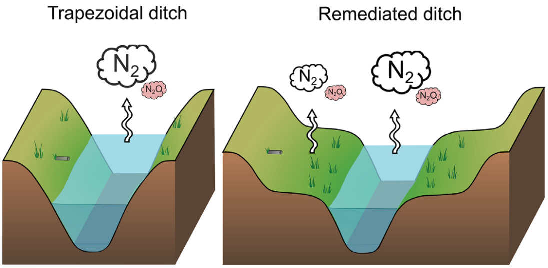
HIGHLIGHTS

- Floodplains increase total system denitrification by 50 %.
- Nitrous oxide production was lower in floodplains than in stream.
- Nitrate-rich catchments promote denitrification and suppress nitrous oxide yields.
- Catchment hydrological processes can override reach-scale nitrate retention.

GRAPHICAL ABSTRACT



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ABSTRACT

Heavily modified headwater streams and open ditches carry high nitrogen loads from agricultural soils that sustain eutrophication and poor water quality in downstream aquatic ecosystems. To remediate agricultural streams and reduce the export of nitrate (NO_3^-), phosphorus and suspended sediments, two-stage ditches with constructed floodplains can be implemented as countermeasures. By extending hydrological connectivity between the stream channel and riparian corridor within constructed floodplains, these remediated ditches enhance the removal of NO_3^- via the microbial denitrification process. Ten remediated ditches were paired with upstream trapezoidal ditches in Sweden across different soils and land uses to measure the capacity for denitrification and nitrous oxide (N_2O) production and yields under denitrifying conditions in stream and floodplain sediments. To examine the controls for denitrification, water quality was monitored monthly and flow discharge continuously along reaches. Floodplain sediments accounted for 33 % of total denitrification capacity of remediated ditches, primarily controlled by inundation and stream NO_3^- concentrations. Despite reductions in flow-weighted NO_3^- concentrations along reaches, NO_3^- removal in remediated ditches via denitrification can be masked by inputs of NO_3^--rich groundwaters, typical of intensively managed agricultural landscapes. Although N_2O production rates were 50 % lower in floodplains compared to the stream, remediated ditches emitted more N_2O than conventional trapezoidal ditches. Higher denitrification rates and reductions of N_2O proportions were predicted by catchments with loamy soils, higher proportions of agricultural land use and lower floodplain elevations. For realizing enhanced NO_3^- removal from floodplains and avoiding increased N_2O emissions, soil type, land use and the design of floodplains need to be considered when implementing remediated streams. Further, we stress the need for assessing the impact of stream remediation in the context of broader catchment processes, to determine the overall potential for improving water quality.

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1. Introduction

Sustained anthropogenic inputs of nitrogen (N) and phosphorous (P) to agricultural soils have fundamentally altered nutrient fluxes to aquatic ecosystems and resulted in water quality deterioration, eutrophication and hypoxia of freshwater and marine environments and higher costs of drinking water production (Andersen et al., 2017; Fowler et al., 2013; Sharpley et al., 2013). Phosphorus mitigation has been the long-standing focus for combating eutrophication (Schindler et al., 2016), but increasing attention is drawn to N as an important co-limiting nutrient in aquatic ecosystems (Conley et al., 2009; Glibert, 2017; Shatwell and Köhler, 2019; Vahtera et al., 2007). Despite raised ambitions in response to extensive policy (e.g. Water Framework Directive and Green Deal; Bieroza et al., 2021) poor water quality status still persists globally. This exposes critical gaps in our understanding of hydrological and biogeochemical functioning of streams and their catchments (Van Meter et al., 2018) and the important role of legacy N in agricultural soils and groundwaters (Basu et al., 2022). Agricultural N sources mainly enter aquatic ecosystems via the interface of headwaters (Abbott et al., 2018; Wollheim et al., 2018), representing ecosystem control points with a complex interplay between hydrological delivery and N removal via denitrification - the microbial reduction of nitrate (NO_3^-) to gaseous N under anoxic conditions (Tank et al., 2021). In these streams, abundant NO_3^- and labile organic carbon (C) sources converge, creating a high potential for denitrification activity. Headwaters in flat and intensively managed agricultural landscapes in temperate regions have been extensively modified into open ditches, characterized by a trapezoidal shape and minimal riparian connectivity, with the overarching purpose of draining excess water from arable land. As a consequence, dissolved NO_3^- bypasses much of the biogeochemically reactive land-water interface of riparian and hyporheic zones and is rapidly moved downstream during high flows, exacerbating eutrophication (Baker et al., 2012; Smith et al., 2008).

Since a large proportion of solutes enter the stream network through headwater agricultural streams and ditches, much focus is now directed towards improving their capacity for biogeochemical nutrient removal; a task that requires comprehensive mitigation strategies and innovative countermeasures (Lintern et al., 2020). To increase denitrification capacity in trapezoidal ditches while maintaining sufficient drainage capacity, two-stage ditches (Powell et al., 2007) have been proposed as cost-efficient stream remediation measures (Roley et al., 2016; Västilä et al., 2021). Two-stage ditches comprise an inset channel flanked by laterally connected floodplains and lower gradient banks with vegetation (Fig. 1a), as an adaptation from natural streams with lateral depositional bars formed by natural fluvial processes (Landwehr and Rhoads, 2003). The larger transect volume of two-stage ditches allows stream water to spread on to floodplains during high flows, relying on riparian connectivity to enhance reach-scale denitrification. Constructed floodplains have been shown to prolong water residence time and remove 2–13 % of stream NO_3^- loads (Mahl et al., 2015; Roley et al., 2012; Speir et al., 2020).

However, denitrification can terminate with nitrous oxide (N_2O) and potentially impose an environmental trade-off between water quality improvements and greenhouse gas emissions from constructed floodplains (Dee and Tank, 2020). Fertilization is a driver of N_2O emissions (Lebender et al., 2014) and elevated NO_3^- concentrations in streams have been linked to higher proportions of sediment denitrification terminating with N_2O (Schade et al., 2016). Previous studies of remediated ditches have focused on the temporal interplay between hydrology and NO_3^- and C substrate limitation of N_2 and N_2O production (Dee and Tank, 2020; Mahl et al., 2015; Roley et al., 2012; Speir et al., 2020). As the implementation of remediated ditches has expanded to a greater diversity of catchments with different dynamics in solute delivery, we need to improve our understanding of N removal capacities and the risk for N_2O emissions across a range of geological settings, soil types, land uses and specific floodplain designs.

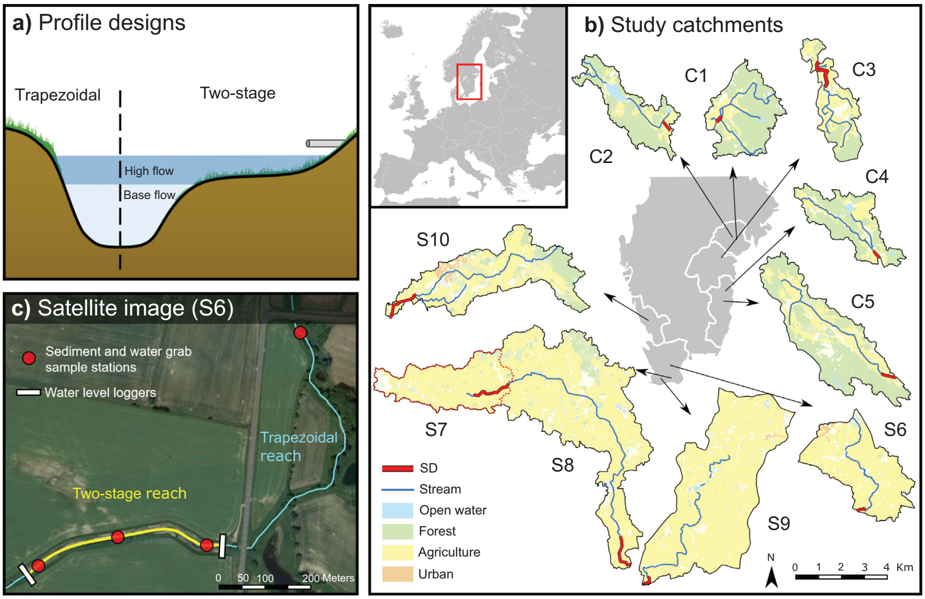

Fig. 1. a) Cross section of trapezoidal (TD) and two-stage (SD) profiles. b) Location of the ten catchments with SD ditches in Central East (C1-C5) and South Sweden (S6-S10). c) Satellite image of one reach in site S6 showing monitoring setup in upstream TD and downstream SD reach (flow direction from North to South). Sediment samples and water grab samples were collected at start of TD reach and at start, middle and end of SD reach, denoted by red circles. Water level loggers were placed at start and end of SD, denoted by white lines.

Here, potential denitrification rates and the fraction ending as N_2O, i.e. N_2O yield, were compared between stream and floodplain sediments of remediated ditches with two-stage design and conventional trapezoidal ditches in ten different catchments, with a focus on understanding catchment controls and the role of floodplain designs (Fig. 1b). Our overarching assumption was that high NO_3^- inputs increases both potential denitrification and N_2O rates. To determine the limiting factors and capacity for reach-scale N removal and N_2O production, we measured sediment and water chemistry, hydrology and catchment characteristics and hypothesized that:

1. Floodplains contribute to reductions in NO_3^- concentrations compared to trapezoidal ditches by increasing reach-scale denitrification activity.
2. Floodplains with lower elevations in relation to the stream channel bed and higher vegetation cover yield higher denitrification rates and suppress N_2O emissions due to prevailing anoxic conditions.

2. Materials and methods

2.1. Site description

Ten agricultural streams and ditches that had been modified into two-stage ditches (Fig. 1a), hereafter referred to as remediated ditches, were selected in Central East (C1–5) and South Sweden (S6–10). Each remediated ditch was paired with an upstream, trapezoidal ditch, equivalent in length (Fig. 1c). The S7 and S8 ditches share the same stream network, as S7 drains an upstream subcatchment nested within the S8 catchment. Study sites are located in tile drained agricultural headwater catchments, ranging between 8 and 42 km^2 in size and dominated by cultivation of both winter and spring sown cereal crops and ley (Fig. 1b). The geology of C1-C5 and S10 catchments is characterized by crystalline bedrock, and in S6-S9 catchments by limestone bedrock. Bedrock is overlaid by quaternary sediments with depths between 10 and 50 m. All studied ditches are of the 1st Strahler order, except C4 which is a 2nd order stream. The lengths of the remediated ditches range between 0.3 and 1.7 km, with floodplains constructed either on one side (C4, S6–7 and S9–10) or two sides (C1–3, C5 and S8) of the inset channel at elevations ranging from 0.25 to 0.94 m in relation to the stream bed. The ratio of floodplain width in relation to channel width ranges from 0.3 to 2.3. The remediated ditches were constructed between 2013 and 2019 by landowners and other stakeholders, aiming to reduce flooding of the nearby fields and improve water quality (Brink et al., 2012; Wiström and Lindberg, 2016; Hedin and Kivivuori, 2015).

Sampling was carried out in 2020–2021, which showed a higher annual precipitation in comparison with five year precipitation averages (SMHI, 2021; Table 1), although S7–9 in South were notably drier. The sites differ in precipitation, soil texture, agricultural land use and floodplain designs (Table 1). Catchments in Central East (C1–5) have lower annual precipitation as well as a lower percentage of agricultural land use and higher clay content in soils compared with catchments in South (S6–10). Floodplain inundation, the overflowing of stream water on floodplains, occurred predominantly in winter and spring at sites with lower floodplains (C4, S6–8). Along each remediated ditch, sampling was carried out in three locations: upstream (UP), mid (MD) and downstream (DN; Fig. 1c) and as paired reference, we sampled upstream of trapezoidal ditches (TD).

For each location along the floodplains in remediated ditches (UP, MD and DN), three 1 m^2 (1 × 1 m) plots were used to visually estimate vegetation cover and classify macrophytes into four functional groups: Reeds (genus *Phragmites* and *Typha*), Grasses, Herbs and Bryophytes. Vegetation observations occurred in May and June 2021. Stream benthic cover was further assessed for all sediment sampling locations in June 2021 by walking 50 m in a zig-zag pattern parallel to the stream and visually determining the stream bed material every 5 m, following the Wolman pebble count method (Wolman, 1954).

2.2. Sediment and water sampling

Stream bed and floodplain sediments were collected at different flow conditions, with and without floodplain inundation from all sites on four occasions (September 2020, November 2020, March 2021 and May 2021; Fig. 2) as well as monthly water grab samples between April 2020–June 2021 to determine potential denitrification and N_2O production rates as well as water and sediment chemistry. Additionally, stream sediments in remediated ditches were collected from CE2 and CE3 at four occasions in 2019 (June, July, August and September) for the same purpose. Stream and floodplain sediments and water samples were collected at the four locations TD, UP, MD and DN. Reference ditch for site S7 was excluded since the ditch originates from a wetland. During November 2020, no samples were collected from sites S7–9 due to logistical constraints. In total, 279 sediment samples were collected between 2019 and 2021. Sediments were sampled down to 3 cm depth with a trowel as 3 cm^3 cubes. For each location, three pseudo-replicates were sampled within 1 m^2 and pooled into one sample. Sediments were placed in airtight plastic bags and stored in coolers during field transportation and at 4 °C back at laboratory.

2.3. Denitrification assays on sediments

Potential denitrification rates, hereafter referred to as denitrification, were measured in 268 sediment samples (11 samples were excluded due to measurement errors) under anoxic conditions with the acetylene inhibition technique, without chloramphenicol (Hellman et al., 2019). To determine potential N_2O production rates, 133 sediment samples were incubated without acetylene (C_2H_2) in parallel with C_2H_2-amended samples, in September 2020 and March 2021. The denitrification and N_2O production rates were determined 1–7 days after sediment sampling and there was no effect of storage time on denitrification rates (ANOVA, $F_{1,\,245} = 0.5342, p = 0.47$). Microcosms were established with 10 g of homogenized fresh sediments and 20 ml distilled water in 125 ml flasks sealed with septum caps. The microcosms were flushed and purged with 1 atm N_2 5 times before adding 10 ml C_2H_2. We equilibrated the microcosms on a shaker at 170 rpm for 30 min before adding 1 ml substrate to reach final concentrations of 6 mg NO_3^--N L^{-1} (KNO_3) and 7 mg C L^{-1} (as a mixture of glucose, acetate and succinate). The microcosms were incubated in 25 °C with agitation (170 rpm) and gas samples were taken at 30, 75, 120, 150 and 180 min. The N_2O concentrations in gas samples were quantified using a gas chromatograph (Clarus-500, Elite-Q PLOT phase capillary column, PerkinElmer, U.S.) with a ^{63}Ni electron-capture detector. The mass of N_2O was corrected for temperature, partial pressure and the exchange of dissolved N_2O gas with the liquid phase, using the Bunsen coefficient 0.545 for N_2O, calculated according to Breitbarth et al. (2004). Denitrification and N_2O production rates were determined by linear or quadratic regression of N_2O mass over time (3–5 time points) and scaled to μg N g^{-1} DM h^{-1} and μg N g^{-1}C h^{-1}. The N_2O yield (%) was determined by dividing N_2O rates from incubations without acetylene with N_2O rates from incubations with C_2H_2.

2.4. Sediment and water chemistry analyses

Fresh weights of sediment subsamples were measured and dry weights were re-measured after oven drying (105 °C) to determine sediment dry matter (DM). Sediments were analyzed for C and N by dry combustion on a Leco CN analyzer (Leco, TruMac, India).

Bottled water samples were analyzed for NO_3^--N (ISO 15923-1:2013), ammonium-nitrogen (NH_4^+-N; ISO 15923-1:2013), dissolved organic carbon (DOC; SS-EN 1484) and suspended sediments (SS; SS-EN 872:2005). Dissolved oxygen (DO), pH and specific conductivity (SPC) were measured in situ, monthly to bimonthly, using optical and electrode-based sensors mounted to a handheld device (ProDSS, YSI, U.S.).

Dissolved organic matter (DOM) properties in collected water grab samples were determined optically in the laboratory to calculate fluorescence

Table 1
Characteristics of catchment, hydrology, floodplain properties and inundation pattern of remediated ditches. Sites are ordered from North to South. Flow discharge and floodplain inundation were recorded in April 2020–June 2021. Discharge in C3 was recorded in 2018–2021, interrupted between December–March each year. Ditch and floodplains dimensions were measured in March and April 2021 with a GPS levelling device (E600 GNSS Receiver, E-survey, China).

Site	County	Catchment properties					Discharge					Ditch dimensions					Floodplain inundation		
		Area (km^2)	Mean temperature (°C)*	Precipitation (mm)*	Soil texture	Agricultural land use (%)	Q50 ($m^3 s^{-1}$)	Baseflow Index (BFI)	Flashiness Index (RBI)	Runoff (mm yr^{-1})	Year of construction	Ditch length (m)	Floodplain Height (m)	Floodplain width (m)	Channel width (m)	Floodplain design	Frequency (day yr^{-1})	Mean Duration (day)	Max Duration (day)
Central East																			
C1	Södermanland	9.73	8.9 ± 7.5	594	Silty clay	16	0.026	0.875	0.269	206	2012	340	0.60	3.1	2.9	Two-sided	72	8	44
C2	Södermanland	7.91	8.9 ± 7.5	594	Silty clay loam	27	0.032	0.942	0.094	206	2012	730	0.94	3.5	2.8	Two-sided	31	7	16
C3	Östergötland	6.63	8.4 ± 7.6	551	Clay loam	70	0.013	0.807	0.330	114	2014	1500	0.53	3.2	2.3	Two-sided	27	6	20
C4	Kalmar	8.12	8.4 ± 7.2	627	Clay loam	35	0.486	0.944	0.070	1790	2019	320	0.51	1.7	5.4	One-sided	199	56	155
C5	Kalmar	16.32	8.4 ± 7.2	627	Clay loam	38	0.023	0.828	0.302	150	2012	780	0.53	1.4	2.6	One − /Two-sided	100	13	51
South																			
S6	Skåne	13.09	8.8 ± 6.9	707	Loam	84	0.086	0.856	0.303	575	2016	400	0.4	5.7	2.7	One − /Two-sided	149	41	126
S7	Skåne	10.84	9.2 ± 6.6	569	Loam	81	0.022	0.799	0.323	179	2013	1960	0.55/0.25	4.0	2.2	One-sided	124	20	129
S8	Skåne	42.41	9.2 ± 6.6	569	Loam	81	0.117	0.798	0.355	272	2013	1770	0.49	6.8	4.6	One − /Two-sided	98	31	76
S9	Skåne	31.02	9.2 ± 6.6	569	Loam	86	0.075	0.868	0.064	91	2019	630	0.85	–	–	One-sided	0	0	0
S10	Halland	16.38	9.7 ± 7.2	823	Sandy loam	58	0.121	0.720	0.580	723	2014	1760	0.34	2.8	7.4	One-sided	–	–	–

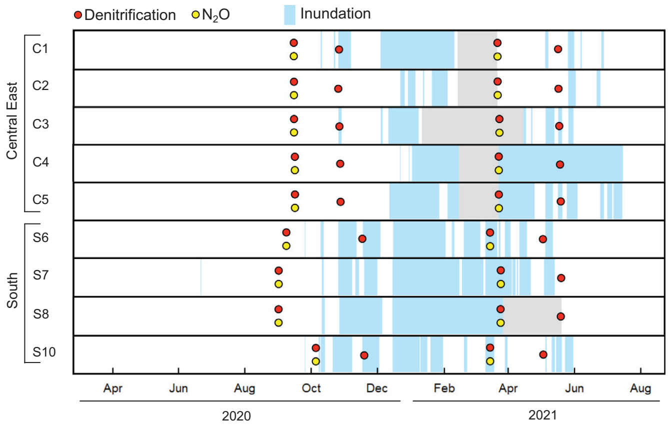

Fig. 2. Inundation of floodplains occurred predominantly in winter and spring across all sites. Blue color denotes recorded inundation events where water stage exceeded floodplain elevation in relation to stream bed from April 2020 to August 2021. Gray color denotes periods of missing water stage data and red and yellow circles show sampling dates for denitrification and N_2O production/N_2O yield, respectively. The figure shows either upstream or downstream inundation in remediated ditches for each site. The location with longest inundation periods chosen to show the onset of inundation in the ditches.

index (FI; McKnight et al., 2001), biological freshness index (BIX; Huguet et al., 2009), humification index (HIX; Ohno, 2002), specific absorbance at 254 nm, normalized for DOC ($SUVA_{254}$; Edzwald et al., 1985), ratio of specific absorbance at 250 nm and 365 nm (E2:E3; Peuravuori and Pihlaja, 1997) and absorbance spectral ratio of slopes between 275 and 295 and 350–400 nm (S_R; Helms et al., 2008), using Matlab (R2020a). These indices are well-established proxies for fingerprinting DOM origin and quality. Water samples were analyzed with a spectrophotometer (Aqualog, Horiba, Japan) equipped with a 150 W Xenon arc lamp, using a high precision quartz cuvette with 1 cm light path. Scans were acquired using excitation-emission matrices (EEM) at excitation and absorbance wavelengths between 240 and 600 nm and emission wavelengths between 212 and 620 nm, at 1 s integration time and 2 nm scan width. Distilled water blanks were scanned each day prior to analysis and the blank signal was subtracted from sample EEM scans to correct for Raman scattering. Raman peak intensities in blank samples were also recorded daily to normalize sample EEM scans and account for the variation in Raman intensities over time. The sample EEM scans were further corrected for inner-filter effect (McKnight et al., 2001) and signal intensities of first and second order Rayleigh scatter lines were removed by applying a Rayleigh masking filter. Instrument bias associated to optical components was automatically corrected in the Aqualog software after each spectral scan. All pre-processing corrections were performed in the Aqualog software.

2.5. Hydrology and geometry surveys

Pressure sensors (HOBO, Onset Computer Corporation, U.S.) were installed to log water stage every 10 min at UP and DN for all ditches. At the same locations, flow discharge was measured with the mid-section method using an acoustic doppler velocimeter (Flowtracker 2, SonTek, U.S.) at 4–8 occasions during different flow conditions in 2020 and 2021. Continuous flow discharge was derived by correlating discharge and corresponding stage data (Fig. S1), using:

$$Q = K(h + a)^p \tag{1}$$

where Q is discharge ($m^3\ s^{-1}$), h *is* stage (m), a is stage at zero flow (m) and K and p are constants (Rantz, 1982).

To determine the effect of floodplain inundation on denitrification, the geometry of selected cross-sections were measured every 50–100 m along all remediated ditches except S9. For this purpose a global GPS device (E600 GNSS Receiver, E-survey, China) with 2 mm accuracy were used, operating in real-time kinematic mode together with a mapping interface (SurPad 4.0, Geoelectron, China). Profile dimensions were surveyed in spring 2021 by walking across chosen cross-sections with the GPS instrument, to obtain length, width and elevation of channels and floodplains. Water stage data from UP and DN were combined with the derived floodplain elevations to calculate the frequency and duration of inundation. Water stage in UP locations was applied to the mean floodplain elevation of the upstream half of the remediated reach and stage data from DN to the downstream half. When water stage exceeded the mean of floodplain elevation it was recorded as an inundation event. The accuracy of recorded events was further cross-validated with field observations of inundation events.

2.6. Statistical analyses

All statistical analyses were performed in R version 1.2 (RStudio Team, 2019). The packages hydrostats and leppott/ContDataQC were used to analyze the hydrological regimes. Analysis of variance (ANOVA) and Pearson correlations were permuted 10,000 times using permuco and RVAideMemoire packages, and the resampling approach was chosen to account for non-normal distributions in the dataset. Base flow index (Gustard et al., 1992) and median flow discharge (Q_{50}) were calculated using *baseflows* and flashiness index (Baker et al., 2004) using *RBIcalc*. Differences in base flow index, flashiness index and Q_{50} between upstream and downstream of remediated ditches were tested using one-way ANOVA (*aovperm*). To test the differences in denitrification rates and N_2O yields in sediments between locations (TD, UP, MD and DN) and profiles (stream and floodplain), two-way ANOVA was used. Differences in denitrification rates and N_2O yield between regions were tested with one-way ANOVA.

To determine the effects of predictor variables (stream and sediment chemistry as well as catchment and ditch properties) on denitrification and N_2O yield, Pearson correlation coefficients were calculated (*perm.cor.test*). The sample distribution of predictor variables was analyzed by scaled principal component analysis (PCA; *rda*) on 4 separate matrices (denitrification and N_2O yield respectively from stream and floodplains respectively) using the Vegan package. Vectors of denitrification rates or N_2O yield were fitted to the PCAs using *envfit* with 10,000 permutations and vectors that were significantly correlated to principal component axes (p < 0.05) were included in ordination plots. Grouping of samples were tested statistically between regions (Central East and South), sites and stream vs. floodplains with permutational multivariate analysis of variance (PERMANOVA) using *adonis*.

To analyze changes in stream water quality (NO_3^--N, NH_4^+-N, DOC, DO and DOM indices) in trapezoidal and remediated ditches, two-way ANOVA was used with location and either site or region as factors. Water samples with NO_3^--N and NH_4^+-N concentrations below detection limit (< 2 % of observations) were assigned to detection limit concentrations divided by 2 (Hornung and Reed, 1990). Flow-weighted concentrations of NO_3^--N were calculated by dividing absolute concentrations (mg L^{-1}) with flow (m^3 s^{-1}) to determine the influence of hydrology on concentrations between UP and DN.

Reach-scale retention of NO_3^--N and NH_4^+-N concentrations in % was calculated as $C_{Ret} = (C_{Upstream-} - C_{Downstream}) / C_{Upstream}$ for trapezoidal and remediated ditches, where $C_{Upstream}$ denotes the upstream concentration in mg L^{-1}, and $C_{Downstream}$ the downstream concentration in mg L^{-1}. The samples were further classified into three flow regimes: $Q_{0–25}$, Q_{25}-Inundation and Inundation. The differences in concentration retention % were tested between trapezoidal and remediated under different flow regimes using two-way ANOVA.

3. Results

3.1. Hydrology and floodplain inundation frequency

Median discharge (Q_{50}) at catchment outlets was overall higher in South compared to Central East (Table 1). All remediated ditches were predominantly base flow driven and received inputs from groundwater and subsurface tile drains, with base flow indices ranging between 0.72 and 0.94 and no differences between upstream and downstream (Table S1). However, flashiness index indicated that discharge peaks were reduced along remediated ditches in Central East but enhanced in South.

Floodplain inundation ranged from 31 to 199 days yr^{-1} with a mean duration of 6–56 days per event (Table 1). Floodplain elevation was the main controlling factor for inundation as lower floodplains promoted higher inundation frequency ($r = -0.56$, $p < 0.05$) and longer duration ($r = -0.25$, p < 0.05). In South, the onset of floodplain inundation started in September to October and receded in May to June, while in Central East, inundation patterns were less uniform and started from October to December and receded in May to July (Fig. 2).

3.2. Denitrification rates in sediments

Denitrification rates in stream sediments were not statistically different between remediated ditches and trapezoidal ditches (ANOVA, $F_{1,157} = 1.24$, $p = 0.27$; Fig. 1a). In remediated ditches, denitrification in floodplain sediments averaged 3.01 ± 3.78 μg N g^{-1} DM h^{-1}, which was significantly lower compared to denitrification in stream sediments (Fig. 3a), averaging 5.85 ± 7.09 μg N g^{-1} DM h^{-1}. Denitrification rates in floodplain sediments were significantly higher in the South sites than the Central East sites, with stream sediments following the same trend, although non-significant (Fig. 3b). There were no longitudinal differences in denitrification rates for either stream or floodplain sediments (Fig. 3a). Over the course of September 2020 to May 2021, denitrification in floodplain sediments varied significantly between the tested months (ANOVA, $F_{4,104} = 2.91$, $p = 0.03$; Fig. S2). The lowest denitrification rates were measured in September (1.81 ± 2.08 μg N g^{-1} DM h^{-1}) followed by an increase in March (4.87 ± 6.12 μg N g^{-1} DM h^{-1}). By contrast, denitrification rates in stream sediments did not show any seasonal trends (ANOVA, $F_{8,120} = 1.32$, $p = 0.24$; Fig. S2). The floodplain age had a positive effect on denitrification, which increased with year since construction (ANOVA, $F_{4,104} = 6.94$, $p < 0.01$; Fig. S3). Rates in floodplains constructed in 2013 were >3 times higher than in floodplains constructed in 2019, apart from C2 constructed in 2012, which had the lowest denitrification rates. For stream sediments, there were no effects of construction age on denitrification (Fig. S3).

Denitrification rates were also expressed as per C content in the sediment, but this did not alter the denitrification patterns for stream and floodplain, seasonality or longitudinal reach effects as observed with rates expressed per dry matter (Fig. S4). In stream sediments, N % increased significantly from trapezoidal to remediated ditches (ANOVA, $F_{3,126} = 2.88$, $p = 0.04$; Fig. S5) and C % followed the same trend (ANOVA, $F_{3,126} = 2.43$, $p = 0.07$), indicating that remediated stream beds promote higher organic matter accumulation. This was further corroborated by increases in the cover of fine benthic organic material from trapezoidal to remediated ditches (Fig. S5).

3.3. Predictors of denitrification

The combined composition of water and sediment chemistry as well as catchment and ditch properties were significantly different between sites in Central East and South, both for stream and floodplains (PERMANOVA, $p < 0.05$; Fig. 4a, b). Higher land use % and sand content in combination with higher NO_3^--N concentrations characterized South sites compared to Central East sites (Fig. 4). C and N content in sediments were the strongest predictors for denitrification in stream sediments (C $r = 0.51$, p < 0.05 and N $r = 0.49$, p < 0.05; Fig. 4a), indicating that OM is limiting denitrification in stream sediments. Denitrification in floodplain sediments also correlated with C ($r = 0.26$, p < 0.05) and N content (r = 0.26, $p < 0.05$) but this relationship was weaker compared to stream sediments (Fig. 4b). Water chemistry showed contrasting responses for denitrification. Stream denitrification was not correlated with NO_3^--N, DOC, DO and pH, SS (Fig. 4a; Fig. S6), whereas floodplain denitrification increased with higher NO_3^--N and lower DOC concentrations (Fig. 4a) as well as higher DO concentrations and pH (Fig. S6). There were no correlations between denitrification and sediment C:N or stream DOC:NO_3^--N with either linear (Fig. S6) or non-linear regression (not shown).

Floodplain elevation relative to the stream bed influenced denitrification, with lower floodplains promoting higher denitrification rates ($r = -0.21$, $p < 0.05$) and peaking at elevations between 0.25 and 0.55 m. The PCA indicated that denitrification also increased with higher inundation frequencies (Fig. 4c). However, when testing directly for the effect of hydrology on denitrification, there were no significant correlations with denitrification and floodplain inundation frequency, duration or flow discharge (Fig. S6). Total vegetation cover was also positively correlated with denitrification (Fig. 4b; Fig. S6). Reed and grass species cover were identified as opposing correlates with denitrification (Fig. 4b; Fig. S6a), but both variables were more strongly predicted by inundation frequency on floodplains (Reed: $r = 0.67$, $p < 0.05$; Grass: $r = -0.49$, p < 0.05).

3.4. N_2O production rates and yields and its predictors in sediments

There were no difference in N_2O production rates (ANOVA, $F_{1,73} = 2.81$, $p = 0.08$; Fig. 3c) or N_2O yields (ANOVA, $F_{1,73} = 0.47$, $p = 0.50$; Fig. 3e) in stream sediments between trapezoidal and remediated ditches. Production rates of N_2O were lower in floodplain sediments compared to stream sediments (Fig. 3c). However, N_2O yields were higher in floodplain sediments (mean 0.38 ± 0.36), compared to stream sediments (mean 0.28 ± 0.25; Fig. 3c). Notably, regional characteristics influenced both N_2O rates and yields in the opposite direction as for denitrification rates, where the lowest N_2O rates and yields were observed in the South sites for both stream and floodplain sediments (Fig. 3d, f). Sites C1–2 and C4 showed a distinct shift towards N_2O as end-product of denitrification in floodplains. Older remediated ditches significantly suppressed N_2O yields in both stream (ANOVA, $F_{4,54} = 3.06$, $p = 0.03$) and floodplain sediments (ANOVA, $F_{4,53} = 3.00$, p = 0.03), with elevated N_2O yields in the youngest remediated ditches (C4 and S9; Fig. S3). As observed for denitrification, the C2, being the oldest remediated ditch, was the exception with high N_2O yields (Fig. S3).

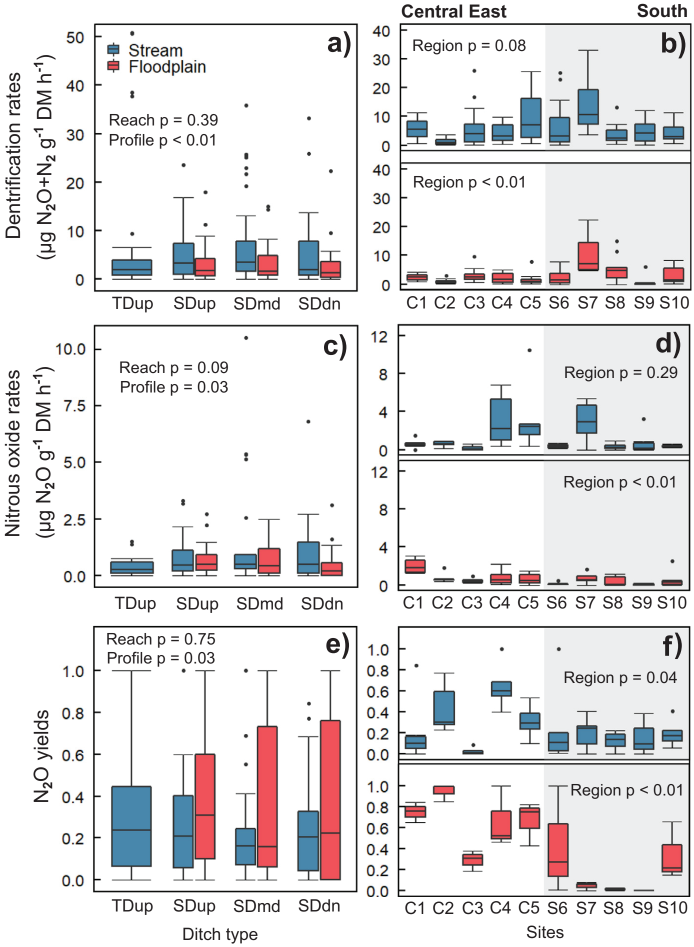

Fig. 3. Potential denitrification (PD) rates, potential N_2O rates and N_2O yields (N_2O:denitrification ratios) from sediments sampled between September 2020 and May 2021. PD rates are shown a) longitudinally in trapezoidal ditch (TDup) and upstream (SDup), middle (SDmd) and downstream (SDdn) of remediated ditches and for b) sites in Central East (C1-C5) and South regions (C6-S10). N_2O rates are shown c) longitudinally and for d) sites in Central East and South regions and N_2O yields are shown e) longitudinally and for f) sites in Central East and South regions. Box colors denote stream and floodplain sediments and *p*-values of two-way or one-way ANOVAs are shown within each panel.

The N_2O rates in stream sediments correlated with C and N content in sediments, but were not predicted by stream chemistry, similarly to denitrification (Fig. 4c; Fig. S6). The proportion of N_2O as an end-product was explained by smaller DOM particle-sizes in stream water (Fig. S6), and was not correlated with the composition of predictor variables (Fig. 4c). In floodplains, N_2O rates also correlated with high C and N content in sediments, and that higher total vegetation and grass cover suppressed N_2O rates (Fig. 4d; Fig. S6). Nitrous oxide yields were predicted by changes in stream chemistry, especially low NO_3^--N and high DOC concentrations as well as high clay content and low proportions of agricultural land use (Fig. 4d; Fig. S6). Although floodplain elevation did not affect N_2O rates, the N_2O yields increased with higher floodplain elevations, in direct contrast to the effects on denitrification (Fig. S6). Despite the positive correlation between N_2O yields and floodplain elevation, there was no significant effect from inundation frequency (Fig. S6). Allochthonous C (FI), pH and DO also predicted higher N_2O yields in floodplain sediments, suggesting that C availability and redox states play an important role for regulating N_2O yields.

3.5. Potential impact of denitrification on water quality

Flow-weighted NO_3^--N concentrations were reduced from upstream to downstream of remediated ditches (ANOVA, $F_{1,220} = 4.84$, $p = 0.03$), indicating that denitrification can contribute to N removal. The reduction was significant both for all estimated flows and lower flows within the range of measured discharge and stage (Fig. S7). The pattern of increasing Q_{50} from up- to downstream also confirmed that additional drainage water entered along remediated reaches (Table. S1). However, there were no

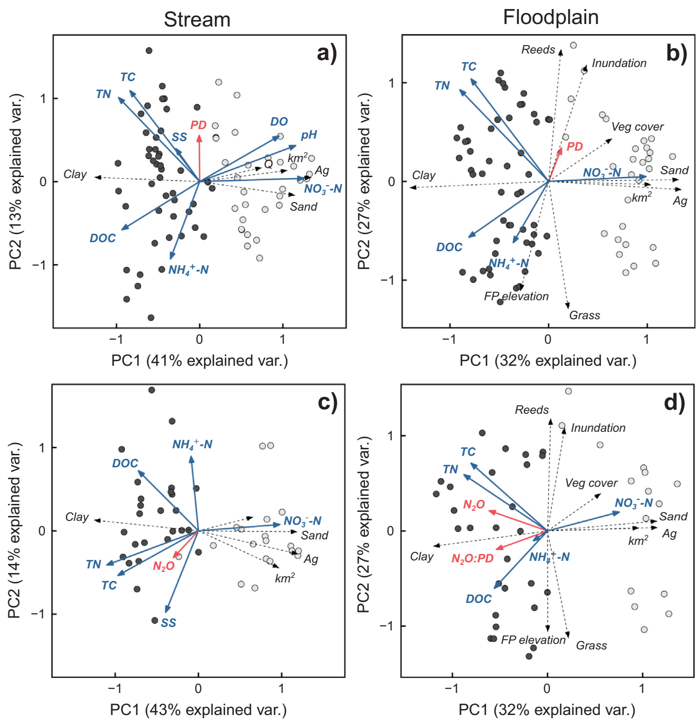

Fig. 4. Principal component analysis (PCA) of predictor variables (water and sediment chemistry, catchment and ditch properties) in stream (a, c) and floodplain sediments (b, d). Correlation with denitrification rates (PD) in a) stream sediments and b) floodplain sediments. Correlation with N_2O rates and N_2O yields (N_2O:PD) in c) stream sediments and d) floodplain sediments. The influence of predictor variables on sample distributions are indicated by blue (water, sediment) and dashed (catchments properties) vectors. PD, N_2O and N_2O:PD that are significantly correlated ($p < 0.05$) with the ordinations are shown as red vectors, with length proportional to the strength of the correlation. Circle color denote samples from Central East (black circles) and South (white circles). Sample sites with missing variables were removed from the analyses and descriptor variables were standardized to equal standard deviations. TC = total carbon, TN = total nitrogen, SS = suspended sediments, DOC = dissolved organic carbon, DO = dissolved oxygen, FP elevation = floodplain elevation, Ag = agricultural land use %, km^2 = Catchment area.

differences in absolute concentrations (Fig. 5a, c) and retention % (Fig. 5b, d) of NO_3^--N and NH_4^+-N in stream water, indicating that additional NO_3^--rich inputs along remediated ditches can mask the effect from denitrification.

Despite higher variation in retention of NO_3^--N and NH_4^+-N concentrations during low flow ($Q_{0–25}$) and base flow (Q_{25}-Inundation), there were no significant differences between trapezoidal and remediated ditches for any of the three flow regimes between April 2020 and June 2021 (Fig. 5b, d). The only site with significantly higher retention along remediated ditches compared to trapezoidal ditches was site S9 (ANOVA, $F_{1,23} = 9.02$, $p < 0.05$).

4. Discussion

4.1. Regional differences in catchment controls of denitrification

The higher capacity for denitrification in South confirmed that catchments with the highest NO_3^--inputs had the largest NO_3^- removal. This pattern can be attributed to the combination of high agricultural land use proportions and naturally well-drained soils (loam and sandy loam) in South, compared to the mixed land use and clay dominated soils in Central East. These soil related differences expand on previous studies, studying effects of denitrification, which have focused on catchments with loamy soils (Mahl et al., 2015; Roley et al., 2012; Speir et al., 2020; Powell and Bouchard, 2010). Although N_2O production rates followed denitrification in both stream and floodplain sediments, the relative production of N_2O did not increase under higher stream NO_3^- concentrations, observed particularly in the South. Contrasting responses in N_2O yields to high NO_3^- concentrations have been reported previously, with either no influence (Beaulieu et al., 2011) or positive effects on N_2O yields (Schade et al., 2016). The measured potential N_2O yields were higher than previous in situ and potential estimates (Beaulieu et al., 2011; Dee and Tank, 2020; Weigelhofer et al., 2013). This could be due to methodological differences in incubation period and substrate concentrations. For example, shorter incubation times can prematurely terminate N_2O reduction, resulting in overestimated N_2O rates (Weier et al., 1993). Further, potential denitrification rates overestimate in situ production rates in sediments (Mahl et al., 2015; Roley et al., 2012), which thereby restricts accurate scaling of these rates to ecosystem level. Nevertheless, substrate amended incubations provide a comparable basis for delineating the influence of sediment and water chemistry on N removal. Regardless of the differences in relation to in situ rates, our potential rates demonstrate a consistent differentiation between stream and floodplains as well as regions, driven by catchment-specific controls.

There was a complex interplay between denitrification and NO_3^-, C availability and redox states, where low NO_3^- concentrations coincided

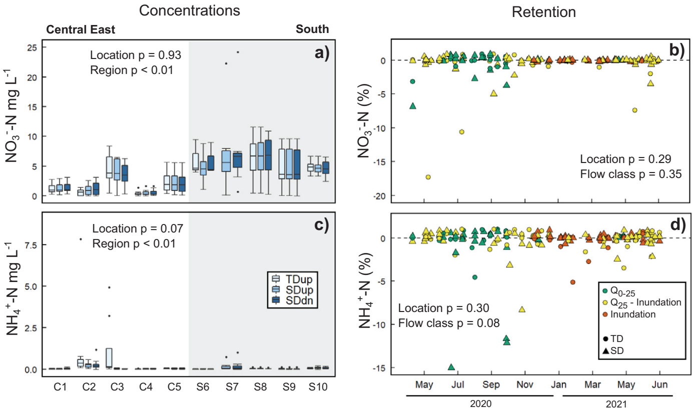

Fig. 5. Water concentrations of nitrogen species and nitrogen removal from April 2020 to June 2021. a) NO_3^--N and b) NH_4^+-N concentrations of all sites at upstream the trapezoidal ditches (TDup), and upstream (SDup) and downstream (SDdn) remediated ditches. Seasonal retention of concentrations of c) NO_3^--N and d) NH_4^+-N concentrations along trapezoidal ditches (TD) and remediated ditches (SD). Retention of concentrations of nitrogen species was calculated as $C_{Ret} = (C_{Upstream} - C_{Downstream}) / C_{Upstream}$ and was divided into the three flow regimes, Q_{0-25}, Q_{25}-Inundation and Inundation. *P*-values of two-way ANOVAs are shown within each panel.

with recalcitrant DOM compounds, low pH and oxic conditions in floodplains of Central East that concomitantly suppressed denitrification and increased N_2O rates and yields. The negative correlation between N_2O yields and FI in floodplain sediments, together with higher ratios of humified (HIX) and allochthonous (FI) DOM in Central East ditches, indicated that a shift from labile DOM towards recalcitrant DOM increased N_2O yields. This builds on previous findings that labile DOM fractions control denitrification activity (Barnes et al., 2012), and suggests that labile DOM do not only limit denitrifiers but also increase incomplete denitrification. Similar controls were found for N_2O rates, although non-significant, which suggests that N_2O yields are not only a product of denitrification efficiency, but are also regulated by the capacity for N_2O production. This is consistent with recent work showing that the production rate of N_2O increased in an estuary when the ratio of allochthonous DOM and NO_3^- were higher (Aalto et al., 2021). Also differences in the composition of microbial communities could explain differences in N_2O yield or N_2O production rates (Jones et al., 2022; Philippot et al., 2011), but this was not considered in the present study. Although the effect of land use and soil type covaried, we conclude that the catchment characteristics are important controls for both of NO_3^- and labile C availability, which ultimately promote higher N removal in remediated ditches without the expense of higher N_2O production.

4.2. Stream denitrification

The small temporal variation in denitrification rates in stream sediments were mainly limited by C and N content in the sediments and decoupled from hydrology and water chemistry. Although in situ denitrification rates in agricultural stream sediments are commonly limited by stream NO_3^- (Mahl et al., 2015; Mulholland et al., 2009; Roley et al., 2012), our results suggests that the enzymatic activity of denitrifiers is instead controlled by the organic matter content and its delivery of C compounds, similar to what has been observed in an estuary (Aalto et al., 2021). A dependency on the mineralization of organic matter and thereby release of electron donors could also explain the small temporal variation observed in denitrifications rates in these sediments. The increase in organic matter quantity and FBOM substrates along remediated stream beds suggests that the true effect size of denitrification is in fact higher in remediated ditches compared to trapezoidal ditches, as C consistently limited denitrification in the stream sediments. The C accumulation in remediated ditches can be explained by reduced water velocities in remediated ditches that promote primary production and increased sedimentation of senescing vegetation. The trend of decreasing flashiness from upstream to downstream of remediated ditches further indicates a higher potential for sediment deposition in lower parts of the reaches. Although this may result in higher stream sediment denitrification, instream deposition of fine sediments is expected to negatively affect drainage capacity (Landwehr and Rhoads, 2003) and benthic fauna (Piggott et al., 2015). These findings imply that the inset channels of the remediated ditches have not reached fluvial equilibrium, as observed in other remediated ditches (Krider et al., 2017; Mahl et al., 2015), and may require occasional channel dredging to maintain drainage function.

4.3. The role of floodplains in denitrification

Denitrification and N_2O production were concomitantly limited by organic matter, but lower C:N ratios in stream water decreased the relative proportion of N_2O production. Denitrification in floodplain sediments contributed to on average 33 % of total denitrification in remediated ditches, reflecting the potential when floodplains are activated by anoxic conditions (Dee and Tank, 2020; Speir et al., 2020). Although denitrification in floodplain sediments, as opposed to stream sediments, responded to changes in water chemistry and increased in spring when inundation was more pronounced, the rates did not exceed that of stream sediments even during inundation. The proportion of denitrification in floodplain vs. stream sediments from our study falls within the range of reported ratios between 12 and 60 % (Mahl et al., 2015; Roley et al., 2012; Speir et al., 2020; but see Powell and Bouchard, 2010), which underlines the large variation in N removal capacity of floodplains across seasons and sites. Moreover, the observed increase in denitrification with floodplain age up to eight years supports previous findings that floodplain sediments mature over time via organic matter accumulation and higher C availability (Speir et al., 2020). Yields were consistently higher in floodplains and exceeded that of stream sediments by >50 %, as the relative proportion of N_2O production increased with lower total denitrification activity. This shows that constructed floodplains with fluctuating redox conditions and lower potential for denitrification can favor incomplete denitrification and a higher proportion of harmful N_2O emissions, as previously demonstrated (Dee and Tank, 2020). On the other hand, the production rates of N_2O were lower in floodplain sediments, meaning that they contribute less N_2O to the atmosphere in absolute quantities in relation to stream sediments. Nevertheless, remediated ditches as a system composed of both stream and floodplain sediments emit more N_2O and therefore have a higher impact on climate warming than trapezoidal ditches.

In contrast to comparable studies with substrate limited denitrification assays (Mahl et al., 2015; Roley et al., 2012; Speir et al., 2020), our potential denitrification rates in the floodplain sediments were better predicted by stream NO_3^- than sediment C, suggesting that denitrifier activity was controlled by periodic inundation events that deliver NO_3^- rather than long-term C accumulation in the sediment. However, reed species cover predicted denitrification activity, as in other wetlands (Ruiz-Rueda et al., 2009) where reeds fuel denitrification by providing labile C through rhizodeposition (Hernandez and Mitsch, 2007). The negative correlation between both N_2O rates and yields and vegetation cover further suggests that the presence of labile C, in contrast to more recalcitrant C, promote complete denitrification and suppress N_2O production, especially in combination with higher NO_3^- availability and prevalent anoxic conditions in the sediments. Here, floodplain elevation was a more reliable proxy for anoxic conditions than inundation frequency and discharge. In addition to stream overflow, the water content in floodplain sediments is controlled by the groundwater table and since floodplain elevation indicates the distance to this level, it also indicates the probability of water saturation in the absence of inundation. Nonetheless, continuous inundation is not favorable as sustained anaerobic conditions can shift NO_3^- reduction towards dissimilatory nitrate reduction resulting in recycling instead of removal of N and thereby hindering improvement of water quality (Aalto et al., 2021). Since N removal in floodplains and the relative suppression of N_2O depend on both anoxic conditions and the delivery of NO_3^- and C compounds, we emphasize the importance of floodplain elevations that can accommodate sufficient periodic inundation (between 0.25 and 0.55 m in this study) as a key control to enhance denitrification.

4.4. Implications for water quality management at catchment scale

The additional denitrification activity in remediated ditches from floodplain sediments can lead to reductions in flow-weighted NO_3^- concentrations, but due to the absence of upstream flow data we could not test if this was true also for trapezoidal ditches. No reductions were observed in absolute concentrations which suggests that there was a substantial input of NO_3^--rich groundwaters along remediated ditches. This was corroborated by consistently high base flow indices across all ditches, implying a predominant mode of groundwater-driven hydrology due to increased hydrological connectivity between floodplains and the hyporheic zone. Due to limited flow measurements during higher flows, there were uncertainties in extrapolated discharge-stage rating curves in these ranges. However, all ditch hydrology was dominated by base flows which reduced the sensitivity towards over- or underestimation of high flows. Previously, floodplains have mainly been conceptualized as reservoirs for stream water overflow (Mahl et al., 2015; Speir et al., 2020), but as they extend the riparian corridor, it is important to also acknowledge their function as conduits for groundwater recharge from the hyporheic zone. As NO_3^- concentrations in groundwater often exceed that of stream water due to long-term crop fertilization (Puckett et al., 2011; Schilling and Zhang, 2004), it is likely that NO_3^--rich groundwater inputs masked the effect of NO_3^- removal from sediment denitrification. Due to the high dependency on catchment-governed lag times of NO_3^- delivery (Basu et al., 2022), we emphasize that remediated ditches themselves are not able to remove legacy NO_3^-, and other in-field measures are required to target these sources. In these catchments, the improvements in water quality can take time to be realized. Although the magnitude of drainage and groundwater NO_3^- inputs may greatly exceed that of denitrification removal rates, floodplains still remain relevant as proximal drivers for denitrification in the riparian zone, permanently removing NO_3^- from both the water column and groundwater (Sigler et al., 2022).

Our measured retention of NO_3^- concentrations were consistent with previous reported changes between −5 to 3 % (Davis et al., 2015; Hodaj et al., 2017; Roley et al., 2012). In contrast, reach-scale estimates of N removal based on denitrification rates in remediated ditches ranged between 27 and 70 % (Hanrahan et al., 2018; Speir et al., 2020), which highlights the challenges associated with accurate N flux mass balancing, especially accounting for tile drainage and groundwater inputs and contributions from legacy N stores.

In combination with catchment typology, the floodplain elevations also impacted denitrification and N_2O yield, suggesting an optimal elevation between 0.25 and 0.55 m to ensure sufficient anoxic conditions and thereby enhancing denitrification while suppressing the relative contribution of N_2O. As lower floodplain elevations were mainly designed in South catchments, we argue that there is a synergy between hydrologically connected floodplains together with high concentrations of NO_3^- and labile C that promotes complete denitrification. The higher floodplain elevations in Central East, with lower inundation frequencies, was motivated by a stronger focus on P mitigation and to maintain floodplain stability (Lindmark et al., 2013). Accordingly, Trentman et al. (2020) indicated that floodplain inundation lasting <8 days maintains oxidative conditions in sediments and thereby promotes P adsorption. However, this also reveals a potential trade-off between N and P mitigation with remediated ditches: lower floodplains and prolonged inundation can lead to sustained reducing conditions and an increased risk for the release of chemically adsorbed P (Trentman et al., 2020). As a measure for targeting N removal, it is important to be aware that the implementation of remediated ditches in clay catchments can increase the risk for N_2O emissions.

The role of denitrification as the primary N sink in aquatic ecosystems is debated and has been argued to be of only minor importance, activated mainly in hotspots of low-order streams during base flow and rarely exceeding 10 % of total N retention (Weigelhofer et al., 2013). Yet, others have shown that longer transient storage times can play a crucial role for denitrification to increase N removal and also suppress N_2O emissions (Quick et al., 2016; Zarnetske et al., 2011). Despite the wealth of information about reactivity rates, our knowledge is still limited about how the extended river corridor, e.g., through constructed floodplains, influences hyporheic exchange and transient storage of solutes, needed to determine both water and N residence times. In addition, tracing of isotopic N species (Zarnetske et al., 2011) and high-frequency monitoring of stream metabolism and NO_3^- (Jarvie et al., 2018) could offer an improved mechanistic understanding of the balance between denitrification and autotrophic uptake as well as the role of autotroph mediation of denitrification in floodplains. Modeling studies that account for both NO_3^- and water mass balance could further be used to disentangle catchment background effects, such as travel time for legacy N processes, from mitigation-specific effects (Chang et al., 2021; Ilampooranan et al., 2019).

5. Conclusion

Constructed floodplains of remediated ditches contributed to higher reach-scale NO_3^- removal, but did not reduce absolute NO_3^- concentrations. This was due to confounding catchment processes such as drainage and groundwater inputs along reaches that likely obscured reductions in stream NO_3^- concentrations downstream of remediated ditches. Remediated ditches in high NO_3^--input catchments, associated with loamy soils and high agricultural land use proportions, had the highest potential for both reducing NO_3^- export and N_2O emissions. We further confirmed that lower floodplain elevations and higher vegetation cover enhanced denitrification while suppressing N_2O yields. Overall, floodplains had lower N_2O production rates compared to stream sediments, but the periodic inundation of floodplains imposes a risk for elevated N_2O yields by suppressing total denitrification.

To realize the potential for NO_3^- removal with remediated ditches, it is critical to engineer the dimensions of floodplains for sufficient inundation as well as selecting appropriate placement in high NO_3^--input catchments. In addition, the multiple water quality benefits of remediated ditches, for example, flood prevention, bank stability and P mitigation also need to be considered during implementation to achieve a holistic solution for reducing eutrophication and erosion. Headwater agricultural catchments are under increasing pressures from legacy N stores and more frequent spells of flooding and drought. This leads to accelerating N losses and reduces the efficacy of single mitigation measures, such as remediated ditches. Thus, we further recommend a suite of complementary measures in these landscapes and an evaluation of potential water quality-climate impact trade-offs before their implementation.

CRediT authorship contribution statement

Lukas Hallberg: Conceptualization, Methodology, Data curation, Formal analysis, Visualization, Writing – original draft, Writing – review & editing. **Sara Hallin:** Methodology, Formal analysis, Writing – review & editing. **Magdalena Bieroza:** Conceptualization, Methodology, Formal analysis, Writing – review & editing, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personalrelationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.156513.

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Appendix A. Supplementary data

1. Discharge-stage rating curves

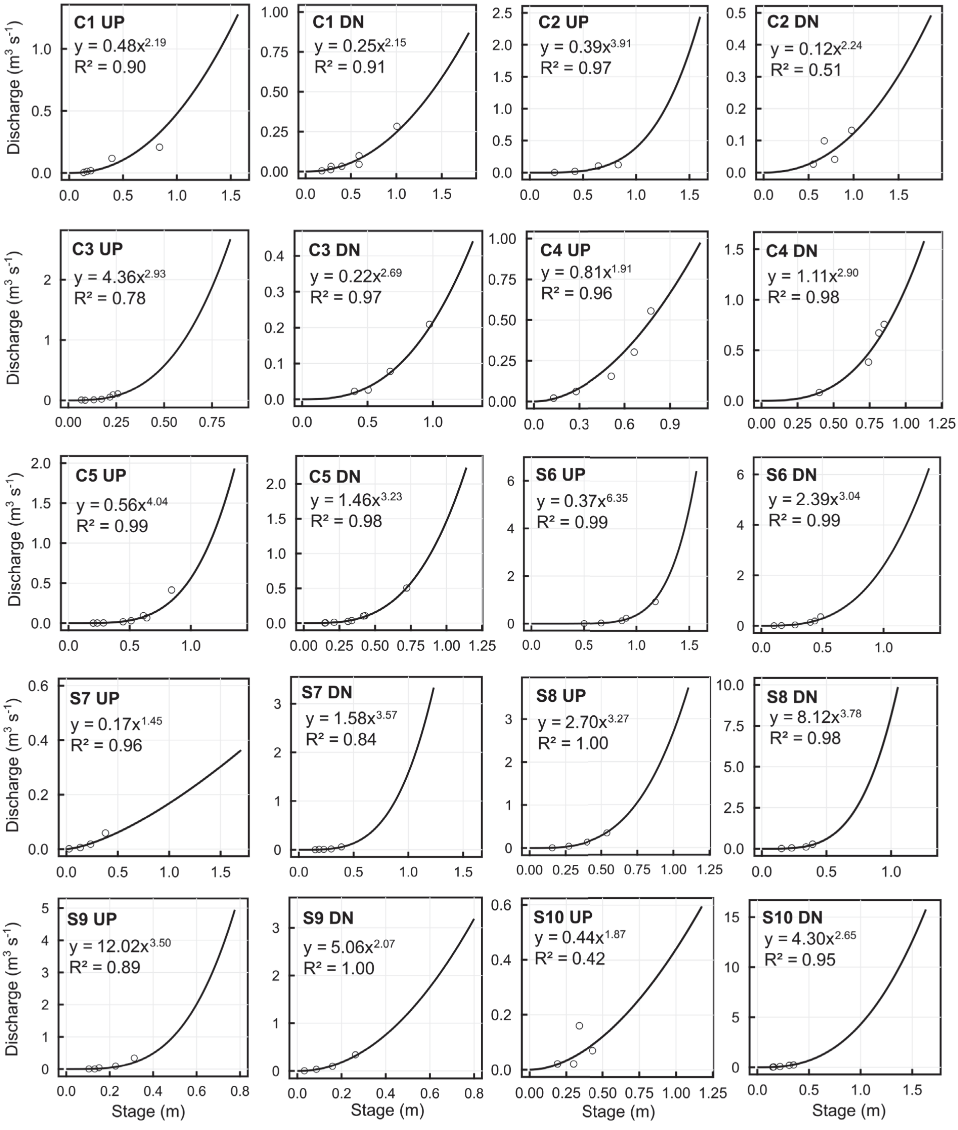


Fig. S1. Relationships between discharge ($m^3\ s^{-1}$) and stage (m) fitted with power function $Q = K(h+a)^p$ where Q is discharge ($m^3\ s^{-1}$), h is stage (m), a is stage at zero flow (m) and K and p are constants. Regressions are extrapolated to the highest recorded stage at each site. Sites (C1-C5 and S6-S10), location (UP, DN), estimated regression coefficients and R^2 are shown within each panel.

1. Hydrology along remediated ditches

Table. S1. Annual means of base flow index (BFI), flashiness index (RBI) and median flow discharge (Q_{50}) in remediated ditches at upstream (SDup) and downstream (SDdn). Flow measured between April 2020-July 2021.

Site	BFI SDup	BFI SDdn	RBI SDup	RBI SDdn	Q_{50} SDup	Q_{50} SDdn
East Sweden						
C1	0.84	0.87	0.35	0.27	0.01	0.03
C2	0.90	0.94	0.22	0.09	0.02	0.03
C3	0.56	0.81	0.66	0.33	<0.01	0.01
C4	0.95	0.94	0.05	0.07	0.09	0.49
C5	0.73	0.83	0.36	0.30	0.01	0.02
South Sweden						
S6	0.89	0.86	0.21	0.30	0.11	0.09
S7	0.88	0.80	0.17	0.32	0.02	0.02
S8	0.87	0.80	0.18	0.36	0.05	0.12
S9	0.81	0.87	0.08	0.06	0.02	0.08
S10	0.89	0.72	0.17	0.58	0.09	0.12

2. Seasonal denitrification rates

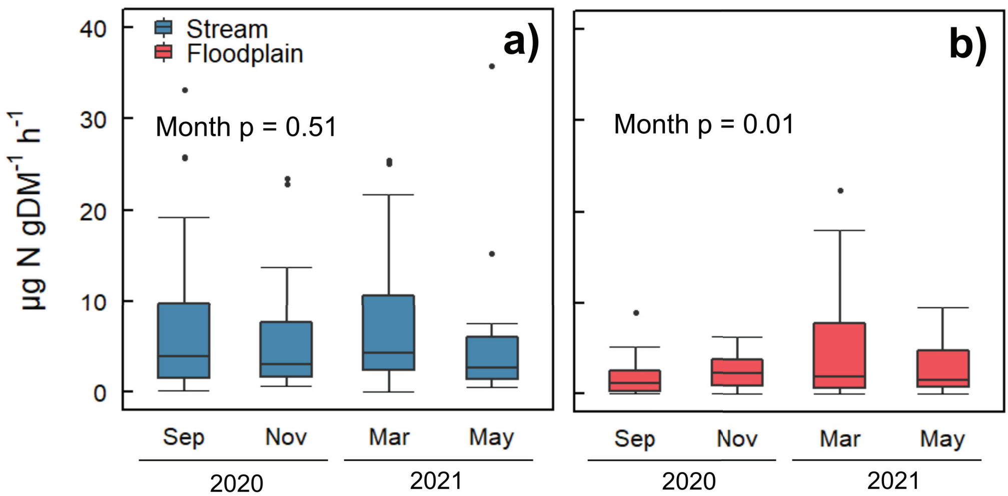


Fig. S2. Seasonal variation in potential denitrification (PD) rates measured between September 2020 and May 2021. Box colors denote **a)** stream and **b)** floodplain sediments and p-values of a one-way ANOVA are shown within each panel.

3. Denitrification rates with construction age

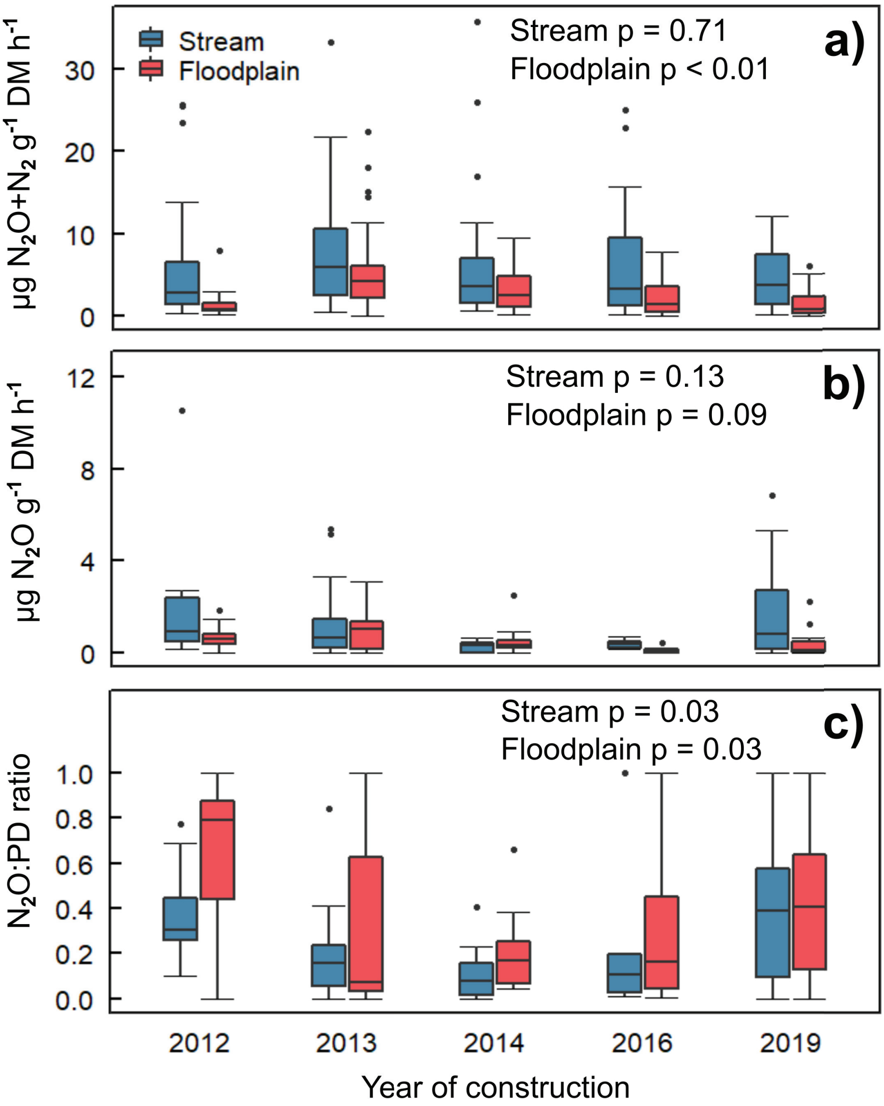


Fig. S3. **a)** Potential denitrification (PD) rates in floodplain sediments and **b)** N_2O yields (N_2O:PD) in stream and floodplain sediments since year of construction of remediated ditches. Box colors denote stream and floodplain sediments and p-values of a one-way ANOVA are shown within each panel.

4. Denitrification rates expressed per carbon mass

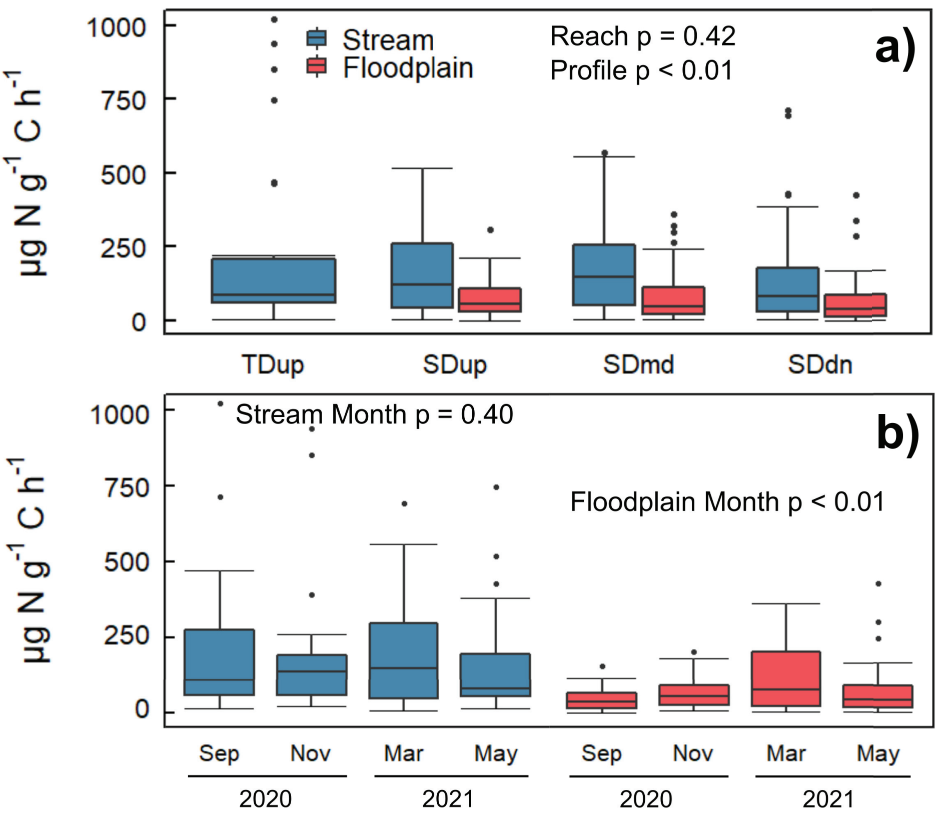


Fig. S4. Potential denitrification (PD) rates measured between September 2020 and May 2021. **a)** Longitudinal variation in PD rates per g C in trapezoidal ditches (TDup) and remediated ditches (SDup, SDmd and SDdn). **b)** Seasonal variation in PD rates per g C. Box colors denote stream and floodplain sediments and p-values of two-way or one-way ANOVAs are shown within each panel.

5. Stream bed benthic composition

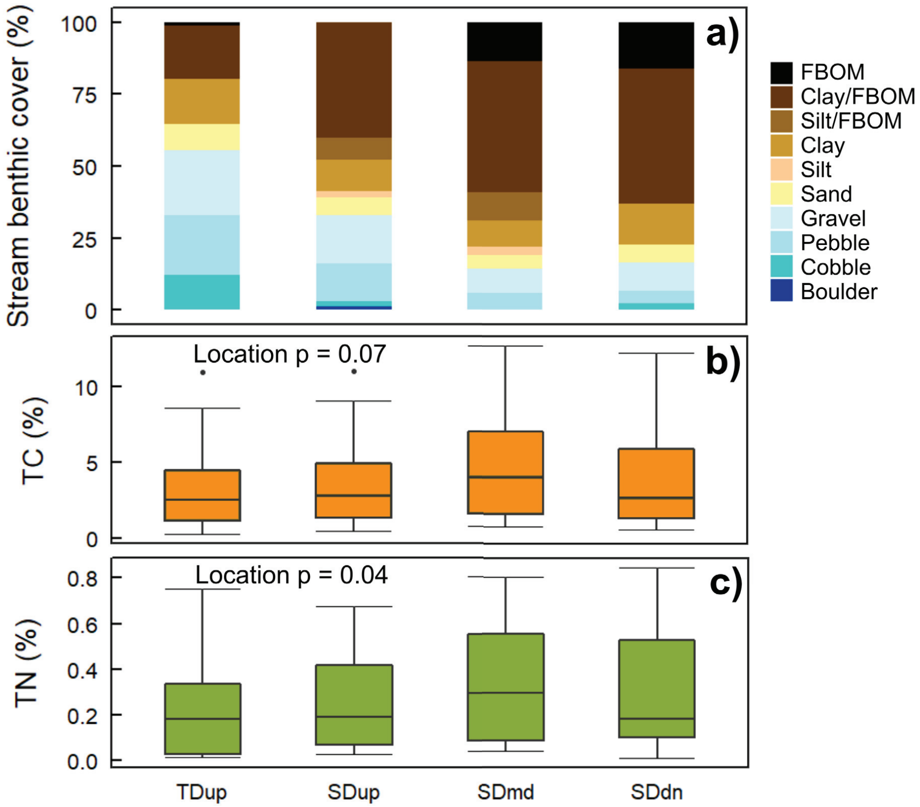


Fig. S5. **a)** Composition of stream benthic cover and **b)** total carbon (TC) and **c)** total nitrogen (TN) content in stream sediments in trapezoidal ditches (TDup) and remediated ditches (SDup, SDmd and SDdn). FBOM = fine benthic organic matter. P-values of a one-way ANOVA are shown within each panel.

6. Denitrification and nitrous oxide yield predictors

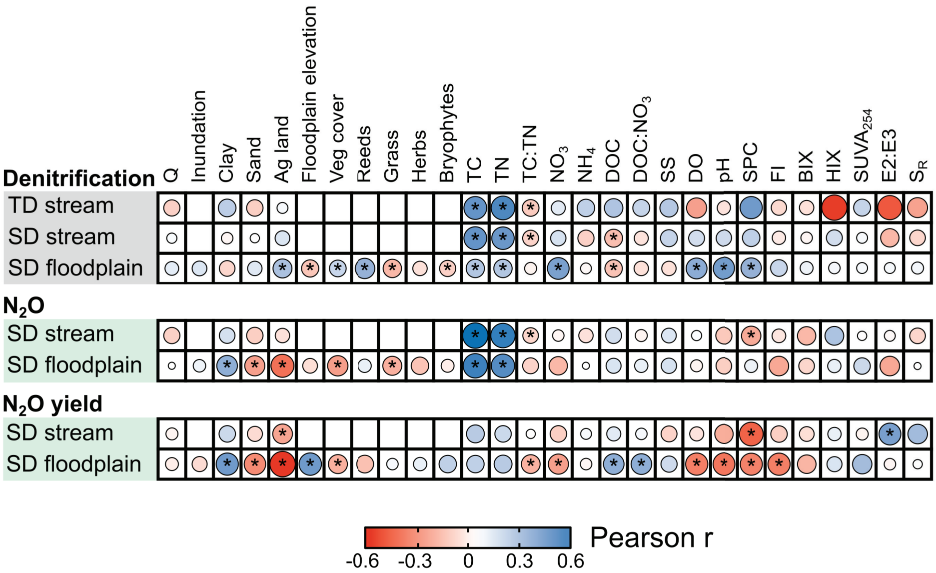


Fig. S6. Pearson correlation coefficients (r) for denitrification rates, N_2O rates and N_2O yields in trapezoidal ditches (TD stream) and remediated ditches (SD stream, SD floodplains) against catchment characteristics and water and sediment chemistry. Data shown for denitrification samples from 4 sampling occasions and N_2O rates and yields from 2 sampling occasions in September 2020 to May 2021. Vegetation cover represent the mean of inventories in May and June 2021. Statistically significant coefficients are denoted with an asterisk ($p < 0.05$).

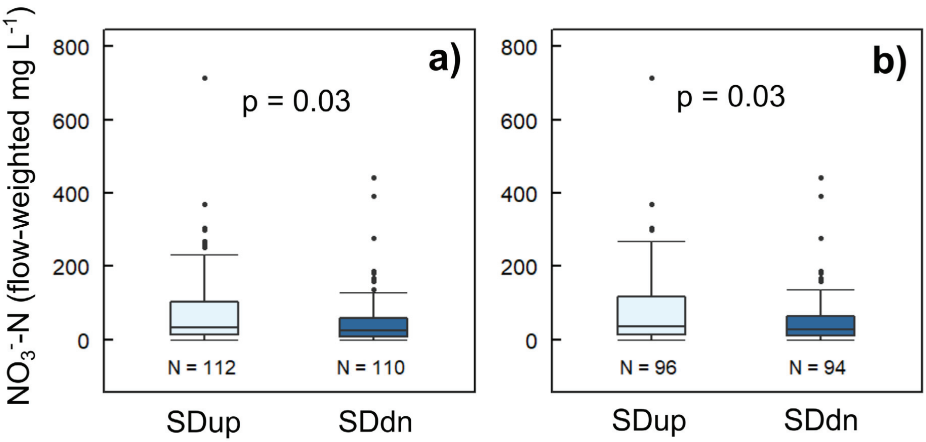
7. Changes in water quality along reaches

Table S2. Mean values of water quality parameters ± one standard deviation in trapezoidal ditches (TDup), upstream remediated ditch (SDup) and downstream remediated ditch (SDdn). Samples were collected between April 2021 and June 2021. Longitudinal differences in water quality parameters were tested for each site with one-way ANOVA and between regions (Central East and South) with two-way ANOVA. P-values and F-values are shown at the top.

		NO_3		NH_4		DOC		DO		FI		BIX		HIX		$SUVA_{254}$		E2:E3		S_R	
Two-way ANOVA	**Location**	$F_{2,206} = 0.06$	$p = 0.94$	$F_{2,206} = 2.71$	**$p = 0.05$**	$F_{2,206} = 0.75$	$p = 0.47$	$F_{2,174} = 0.76$	$p = 0.75$	$F_{2,134} = 0.81$	$p = 0.45$	$F_{2,134} = 0.31$	$p = 0.74$	$F_{2,134} = 1.84$	$p = 0.16$	$F_{2,134} = 2.33$	$p = 0.10$	$F_{2,134} = 2.20$	$p = 0.11$	$F_{2,134} = 0.54$	$p = 0.59$
	Region	$F_{1,206} = 91.60$	**$p < 0.01$**	$F_{1,206} = 2.17$	$p = 0.15$	$F_{1,206} = 66.33$	**$p < 0.01$**	$F_{1,174} = 5.07$	**$p = 0.03$**	$F_{1,134} = 40.83$	**$p < 0.01$**	$F_{1,134} = 35.03$	**$p < 0.01$**	$F_{1,134} = 6.62$	**$p = 0.01$**	$F_{1,134} = 10.32$	**$p < 0.01$**	$F_{1,134} = 2.40$	$p = 0.12$	$F_{1,134} = 8.51$	**$p < 0.01$**
Site	**Location**	**Mean (mg L^{-1})**	**n**	**Mean (mg L^{-1})**	**n**	**Mean (mg L^{-1})**	**n**	**Mean (mg L^{-1})**	**n**		**n**		**n**		**n**		**n**		**n**		**n**
C1	TDup	0.96 ± 0.88	14	0.06 ± 0.08	14	13.08 ± 4.00	14	11.07 ± 3.24	15	1.22 ± 0.04	6	0.59 ± 0.04	6	0.92 ± 0.03	6	4.85 ± 0.69	6	3.38 ± 0.31	6	1.60 ± 0.22	6
	SDup	0.98 ± 0.92	14	0.06 ± 0.11	14	14.20 ± 3.79	14	11.14 ± 3.13	15	1.22 ± 0.04	6	0.62 ± 0.08	6	0.91 ± 0.03	6	5.10 ± 0.79	6	3.14 ± 0.48	6	1.57 ± 0.10	6
	SDdn	1.05 ± 0.92	14	0.05 ± 0.05	14	13.62 ± 3.04	14	11.22 ± 2.99	17	1.24 ± 0.04	6	0.66 ± 0.13	6	0.88 ± 0.04	6	5.02 ± 0.50	6	3.14 ± 0.29	6	1.67 ± 0.10	6
C2	TDup	0.36 ± 0.53	14	0.48 ± 0.51	13	**35.86 ± 11.95**	14	4.38 ± 3.01	13	1.27 ± 0.05	10	0.62 ± 0.04	10	0.89 ± 0.04	10	4.96 ± 1.36	10	3.55 ± 0.42	10	1.46 ± 0.16	10
	SDup	0.61 ± 0.84	14	0.22 ± 0.17	14	**27.24 ± 6.47**	14	4.93 ± 3.58	14	1.26 ± 0.04	11	0.62 ± 0.03	11	0.90 ± 0.03	11	5.16 ± 1.03	10	3.76 ± 0.84	11	1.61 ± 0.31	11
	SDdn	0.78 ± 1.09	14	0.21 ± 0.30	14	**27.62 ± 6.08**	14	5.57 ± 4.02	14	1.27 ± 0.04	10	0.62 ± 0.03	10	0.91 ± 0.04	10	4.78 ± 2.12	10	4.17 ± 0.85	10	1.67 ± 0.38	10
C3	TDup	3.48 ± 2.88	11	**1.39 ± 2.04**	10	13.20 ± 5.37	11	7.04 ± 5.17	8	1.31 ± 0.05	3	0.69 ± 0.07	3	0.91 ± 0.02	3	3.64 ± 1.08	5	3.54 ± 0.16	5	1.51 ± 0.24	5
	SDup	3.34 ± 2.36	16	**0.03 ± 0.02**	16	10.29 ± 2.87	16	9.08 ± 4.96	9	1.33 ± 0.05	10	0.67 ± 0.05	10	0.90 ± 0.09	10	4.50 ± 1.15	9	3.50 ± 0.62	10	1.63 ± 0.25	10
	SDdn	2.80 ± 2.18	17	**0.04 ± 0.05**	17	11.15 ± 2.75	17	8.17 ± 5.61	9	1.33 ± 0.04	6	0.67 ± 0.04	6	0.92 ± 0.02	6	4.05 ± 0.57	6	3.66 ± 0.24	6	1.60 ± 0.10	6
C4	TDup	0.28 ± 0.34	15	0.02 ± 0.02	15	12.77 ± 2.54	15	9.70 ± 1.56	17	1.21 ± 0.02	11	0.64 ± 0.03	11	0.91 ± 0.02	11	**3.20 ± 0.19**	11	4.31 ± 0.39	11	2.37 ± 0.46	11
	SDup	0.33 ± 0.43	15	0.03 ± 0.02	15	12.74 ± 2.35	15	9.62 ± 3.20	20	1.22 ± 0.03	11	0.65 ± 0.06	11	0.90 ± 0.03	11	**3.40 ± 0.73**	11	4.04 ± 0.37	11	2.16 ± 0.37	11
	SDdn	0.49 ± 0.57	10	0.03 ± 0.02	10	13.75 ± 2.21	10	9.38 ± 2.95	17	1.24 ± 0.04	6	0.62 ± 0.04	6	0.90 ± 0.04	6	**5.29 ± 2.65**	6	3.57 ± 0.64	6	1.83 ± 0.36	6
C5	TDup	2.06 ± 1.90	10	0.05 ± 0.03	10	19.25 ± 6.03	10	8.74 ± 2.63	14	1.26 ± 0.04	8	0.61 ± 0.04	8	0.93 ± 0.02	8	4.38 ± 0.80	8	3.83 ± 0.41	8	1.54 ± 0.06	8
	SDup	1.83 ± 1.85	11	0.04 ± 0.03	11	17.59 ± 5.97	11	8.82 ± 3.20	18	1.26 ± 0.04	9	0.61 ± 0.04	9	0.92 ± 0.02	9	4.86 ± 1.67	9	3.47 ± 0.83	9	1.55 ± 0.09	9
	SDdn	1.75 ± 1.80	11	0.03 ± 0.03	11	16.96 ± 5.76	11	8.87 ± 2.38	19	1.26 ± 0.05	10	0.61 ± 0.05	10	0.93 ± 0.02	10	4.37 ± 1.16	10	3.74 ± 0.52	10	1.56 ± 0.01	10
S6	TDup	4.10 ± 2.63	7	0.04 ± 0.03	7	11.21 ± 3.96	7	10.06 ± 3.24	4	1.24 ± 0.03	5	0.61 ± 0.03	5	0.95 ± 0.01	5	3.94 ± 0.33	5	3.76 ± 0.13	5	1.60 ± 0.22	5
	SDup	3.56 ± 2.50	8	0.04 ± 0.02	8	10.55 ± 3.71	8	9.59 ± 2.10	7	1.26 ± 0.03	6	0.62 ± 0.04	6	0.94 ± 0.01	6	3.92 ± 0.28	6	3.76 ± 0.07	6	1.64 ± 0.22	6
	SDdn	4.07 ± 2.42	7	0.03 ± 0.02	7	10.76 ± 3.99	7	9.57 ± 2.04	7	1.24 ± 0.03	5	0.61 ± 0.04	5	0.94 ± 0.01	5	3.89 ± 0.27	5	3.67 ± 0.14	5	1.62 ± 0.23	5
S7	TDup	-		-		-		-													
	SDup	2.59 ± 3.05	12	0.26 ± 0.29	13	10.97 ± 2.25	12	9.28 ± 6.03	7	1.32 ± 0.03	11	**0.70 ± 0.01**	11	0.91 ± 0.05	11	3.88 ± 1.62	11	3.51 ± 0.90	11	2.00 ± 0.35	11
	SDdn	2.85 ± 3.04	12	0.15 ± 0.27	13	11.01 ± 2.04	13	5.74 ± 4.12	6	1.31 ± 0.05	11	**0.66 ± 0.04**	11	0.91 ± 0.05	11	4.25 ± 1.28	11	3.59 ± 0.62	11	1.84 ± 0.31	11
S8	TDup	3.75 ± 3.99	13	0.06 ± 0.04	13	12.27 ± 1.98	13	10.59 ± 1.69	7	1.30 ± 0.02	10	0.67 ± 0.01	10	0.94 ± 0.01	10	3.37 ± 0.32	10	4.44 ± 0.52	10	2.03 ± 0.18	10
	SDup	3.81 ± 4.05	13	0.04 ± 0.03	13	12.21 ± 1.92	13	10.15 ± 1.71	7	1.30 ± 0.02	10	0.67 ± 0.01	10	0.93 ± 0.02	10	3.85 ± 1.52	10	4.08 ± 0.89	10	1.99 ± 0.29	10
	SDdn	3.77 ± 4.07	13	0.04 ± 0.02	13	12.15 ± 2.05	13	9.60 ± 2.89	7	1.30 ± 0.02	10	0.67 ± 0.01	10	0.93 ± 0.01	10	3.50 ± 0.56	10	4.34 ± 0.49	10	1.97 ± 0.18	10
S9	TDup	2.95 ± 3.45	13	0.04 ± 0.04	13	10.33 ± 1.19	13	10.74 ± 3.49	6	1.32 ± 0.02	11	0.70 ± 0.01	11	0.93 ± 0.01	11	3.11 ± 0.18	11	4.51 ± 0.25	11	2.32 ± 0.16	11
	SDup	3.02 ± 3.39	13	0.04 ± 0.03	13	10.35 ± 1.22	13	10.25 ± 3.29	7	1.31 ± 0.04	11	0.70 ± 0.01	11	0.92 ± 0.01	11	3.08 ± 0.22	11	4.44 ± 0.45	11	2.28 ± 0.30	11
	SDdn	2.89 ± 3.44	13	0.03 ± 0.03	13	10.41 ± 1.25	13	11.45 ± 3.17	7	1.31 ± 0.02	11	0.71 ± 0.01	11	0.92 ± 0.01	11	3.00 ± 0.20	11	4.42 ± 0.31	11	2.39 ± 0.13	11
S10	TDup	4.81 ± 0.98	10	0.07 ± 0.03	10	8.44 ± 3.15	10	9.78 ± 1.14	7	1.29 ± 0.03	6	0.63 ± 0.02	6	0.94 ± 0.01	6	4.19 ± 0.73	6	3.44 ± 0.32	6	1.36 ± 0.12	6
	SDup	4.64 ± 1.06	10	0.09 ± 0.07	10	8.64 ± 2.82	10	9.86 ± 1.37	7	1.30 ± 0.01	7	0.63 ± 0.01	7	0.93 ± 0.05	7	4.25 ± 0.34	7	3.51 ± 0.29	7	1.40 ± 0.15	7
	SDdn	4.41 ± 1.28	10	0.08 ± 0.06	10	8.79 ± 3.05	10	9.55 ± 3.12	7	1.29 ± 0.02	5	0.63 ± 0.01	5	0.94 ± 0.01	5	4.22 ± 0.39	5	3.55 ± 0.16	5	1.52 ± 0.13	5

8. Flow-weighted nitrate concentrations

Fig. S7. Flow-weighted NO_3^--N concentrations in stream water at upstream (SDup) and downstream (SDdn) of remediated ditches. **a)** Flow-weighted concentrations across the entire estimated flow range. **b)** Flow-weighted concentrations excluding high flows that were extrapolated with power regressions beyond measured flow and stage. N denotes number of samples.



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Agricultural headwater streams are important pathways for nutrients and sediments, sustaining harmful eutrophication and poor water quality. In this thesis, instream nutrient and sediment mitigation by floodplain remediation was evaluated. Floodplain processes support reductions in particulate phosphorus and higher nitrogen removal. These findings provide decision support to improve current floodplain management and designs towards viable and healthy aquatic ecosystems.

Lukas Hallberg received his graduate education at the Department of Soil and Environment at the Swedish University of Agricultural Sciences. He received his M.Sc. degrees in Biology and Agricultural Science from the same university.

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