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Impacts of fish ponds on freshwater pearl mussel (*Margaritifera margaritifera*) streams

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“The whole is more than just the sum of its parts”

Aristotle

Preface

The aim of this dissertation was to characterize and evaluate the impact of fish ponds on a variety of habitat quality parameters of pearl mussel streams in Upper Franconia over multiple sales. Chapter 1 presents an overview of the habitat requirements for the FPM, focusing mainly on the juvenile habitat. Then, potential impacts of fish ponds on these habitats are reviewed based on the current scientific literature. Chapter 2 introduces the study streams and gives an overview on the methods used to characterized the impact of the fish ponds on the study streams.

On the catchment scale, impacts of the ponds on the hydrologic regime are assessed using hydrological modeling with the Soil and Water Assessment (SWAT), in order to gain insights on the cumulative effects of the ponds. Furthermore, the impact on the temperature regime is investigated at the local scale below selected ponds. Water temperature is an important factor for the development of juvenile FPM as well as the occurrence of the salmonid host fish (Chapter 3).

Another important factor determining habitat suitability for juvenile FPM is the quality of the interstitial habitat within and the fine sediment deposition on the stream bed. The juveniles rely on a well-oxygenated stream bed throughout their development, governed by a good connectivity with the free wave, which is impaired by high fine sediment deposition rates and the resulting clogging of the stream bed. Therefore, the (fine) sediment dynamics of the study streams are investigated in Chapter 4 at a high spatiotemporal resolution and a focus on the prevailing discharge conditions. This general information on the stream characteristics is later used to assess the impact of pond drainage.

Since pond drainage for fish harvest is proposed to release high amounts of suspended particles and nutrients, this period needs to be studies in detail. In Chapter 5, the impact of more than 20 different pond drainage operations is compared to the background fine sediment deposition rates and nutrient concentrations described previously. Drainage operations with different measures to prevent the release of suspended solids (suspended solid mitigation measures, SSM) are compared to drainage operation without such measures in order to evaluate their efficiency. Management implications are derived from the results to minimize pollution of the receiving water bodies during pond drainage.

As physical settling structures like settling ponds were proved an efficient measure to decreases fine sediment deposition rates below drained ponds, the mechanisms related to these effects were investigated in detail at one pond facility. The temporal patterns of water, suspended solids and nutrients discharge into the adjacent stream were studies over the drawdown and fishing of two ponds. One of them was emptied into a settling pond, while the other discharged directly into the receiving FPM stream. Different patterns for particulate and soluble pollutants were observed and used to derive management implications to address either fine sediment or nutrient outputs (Chapter 6).

In Chapter 7, the observed impacts of the fish ponds on the different parameters and over the multiple spatial and temporal scales as well as their interactions are discussed in order to develop a holistic evaluation of the pond impacts on the FPM habitat. Pond management implications are derived that also address the expected pressures from the ongoing climate change in order to mitigate negative effects and increase the benefits of fish ponds in FPM conservation.

The thesis demonstrates the strength of combining multiple methodologies to assess pond impacts over a large variety of temporal and spatial scales. The derived management implications highlight the chance of integrating a thoughtful pond management into the conservation of freshwater pearl mussel streams to minimize negative impacts and profit from the water retention offered by the ponds.

Summary

Fish ponds are essential landscape elements within the catchments of five streams that contain the largest remaining freshwater pearl mussel (*Margaritifera margaritifera*, FPM) populations in Northern Bavaria. The once abundant FPM is now one of the most endangered freshwater species. It relies on a complex life cycle, including a parasitic phase of its glochidia larvae on a salmonid host fish and a juvenile phase buried within the stream substrate for up to five years. Enhanced fine sediment and nutrient inputs through anthropogenic land use intensification are considered the key issues responsible for unsuccessful recruitment in these populations. In particular, the juvenile phase of the FPM relies on a permeable, well-oxygenated interstitial zone within the stream bed. High fine sediment deposition rates and biofilm formation due to high nutrient levels clog the pores within gravel substrates and impair the infiltration of oxygen-rich surface water, leading to oxygen depletion within the interstitial habitat and the loss of juvenile FPM. The remaining, overaged populations are experiencing an ongoing decline throughout their Central European distribution. Numerous attempts have been made to decrease fine sediment and nutrient inputs from agricultural land and municipal sewage in the study area. At the same time, the role of the hundreds of fish ponds within the catchments remained uncertain and controversial. The core objective of this thesis was to characterize the impact of the ponds on different parameters related to FPM habitat quality over multiple scales to evaluate their relevance in relation to other land uses.

(Fish) ponds are connected to their adjacent streams directly via surface in- and outflow channels and indirectly via the groundwater. The impact of the more than 150 ponds within one of the two study catchments on the hydrologic regime, including groundwater pathways, was investigated by hydrological modeling using the Soil and Water Assessment Tool (SWAT). The ponds were integrated into the catchment-wide model, and the resulting runoff was compared to the same SWAT model without ponds. The direct impact via the pond outlet channels was assessed by comparing the water temperature (Tw) measured upstream and downstream of outlet channels of selected ponds (Chapter 3). The ponds caused a 1.5% decrease in peak flows and a 4.5% increase in baseflow levels, indicating a moderate stabilizing effect on the hydrologic regime by retention of excess water during floods and groundwater recharge through the pond bottom. Discharge of heated pond water increased stream temperatures by up to 5.5 °C directly below ponds close to the main FPM streams. Catchment land use was another important driver of Tw, with impacts from pond effluents compensated along forested stream stretches, while Tw further increased along open stream stretches. The salmonid host fish might avoid stream stretches with high Tw during glochidia release, resulting in low infestation rates. On the other hand, a rise in Tw might increase juvenile mussels' growth rates. Therefore, the impacts on FPM populations vary throughout the year. These results demonstrate the potential to use ponds to retain water in the landscape to mitigate extreme low flow conditions that will occur more often with climate change. However, thoughtful effluent management is crucial to avoid negative impacts on the temperature-sensitive salmonid host fish.

Temporal variations were not only evident in the hydrologic and temperature regimes of the study catchments. Fine sediment deposition rates also strongly depended on the occurrence of certain discharge events throughout the year. An assessment of the spatiotemporal variation in (fine) sediment dynamics of the study streams (Chapter 4) revealed a strong difference between low and high flows and, again, differences between high discharges resulting from snow melt and strong rain. Deposition samples consisted of more than 80% fine sediments during low flow conditions, resulting in an ongoing clogging of the stream substrate. In contrast, high flows during the snow melt remobilized the entire stream bed, including large particles. The substrate was cleaned from fines, which only accounted for a proportion of 40% of the deposition samples after the snow melt. Strong rain events also mobilized larger particles, but the high input of fine sediments from soil erosion concealed the flushing effects. Differing catchment land use resulted in contrasting longitudinal fine sediment deposition patterns along the three study stream systems, with increased fine sediment deposition related to agricultural land use, while lower fine sediment deposition rates were observed in forested areas. Impacts from the fish ponds were variable. Several ponds had nearly no effects, while others caused increased fine sediment deposition rates downstream of the inflows of fish pond outlet channels into the main stream, particularly in autumn after the pond drainage for fish harvest.

The impact of pond drainage on fine sediment deposition and inputs of suspended solids and nutrients was studied for more than 20 different drainage operations and compared to the background fine sediment deposition rates under high and low flow conditions (Chapter 5). Pond drainage without any measures to mitigate the release of suspended solids resulted in maximum turbidities of more than 460 NTU, particularly during the fish harvest. Resulting fine sediment deposition rates were similar to or even higher than those measured after high discharge events. Three different types of suspended solid mitigation measures (SSM) were shown to strongly differ in their efficiency: Physical settling structures like settling ponds or harvesting fish by seining with the outlet closed strongly decreased the release of suspended solids and fine sediment depositions rates. In contrast, filtering the effluent through a bale of straw did not decrease turbidities and fine sediment deposition rates compared to standard drainage operations without SSM.

A detailed analysis of suspended particles and nutrients released from pond drainage with and without SSM at a high temporal resolution (Chapter 6) revealed again the strong potential of sufficiently large physical settling structures to prevent inputs of fines and nutrients into the adjacent FPM streams. Without SSM, around one-third of the released particles were concentrated within the last 1% water released during the fish harvest. Preventing these outputs by fishing with the outlet closed is, therefore, an efficient mitigation measure if the construction of physical settling structures is not feasible. However, soluble nutrients are continuously released during the entire drainage process and therefore need to be addressed either directly in the fish pond or in an additional treatment and settling pond before the effluent is discharged into the receiving stream.

The integrated approach of combining hydrological modeling of the whole catchment with extensive field studies on individual ponds demonstrated the potential impact of these small water bodies on a wide range of habitat parameters and over different temporal and spatial scales, also in relation to larger land use elements. While the general and cumulative effects on the catchment scale were evident from the SWAT model, the highly diverse pond management revealed variable impacts on the local scale, which are particularly relevant for the fragmented population structure of the remaining FPM. In areas with extensive land use, fish ponds can be responsible for increased Tw, fine sediment deposition rates, and nutrient concentrations, which degrade the FPM juvenile habitat. However, the studies also demonstrated that pond management could easily be adjusted to minimize the negative impacts and still benefit from the pond’s water retention potential to increase stream resilience and mitigate threats related to climate change.

Zusammenfassung

Fischteiche sind wesentliche Landschaftselemente in den Einzugsgebieten der letzten Flussperlmuschelgewässer in Nordbayern. Die Flussperlmuschel (*Margaritifera margaritifera*, FPM) ist eine der am stärksten bedrohten Süßwasserspezies und auf einen komplizierten Lebenszyklus angewiesen. Er umfasst eine parasitische Phase ihrer Glochidienlarven auf einem lachsartigen Wirtsfisch und eine bis zu fünf Jahre lang im Gewässersubstrat vergrabene juvenile Phase. Ein erhöhter Eintrag von Feinsedimenten und Nährstoffen durch die anthropogene Intensivierung der Landnutzung wird als Hauptursache für das Scheitern der Rekrutierung dieser Populationen angesehen. Insbesondere die juvenile Phase der FPM ist auf ein durchlässiges, gut mit Sauerstoff versorgtes Interstitial im Gewässerbett angewiesen. Hohe Feinsedimentdepositionsraten und die Bildung von Biofilmen aufgrund hoher Nährstoffgehalte verstopfen die Poren in Kiessubstraten und beeinträchtigen die Infiltration von sauerstoffreichem Oberflächenwasser. Der resultierende Sauerstoffmangel im hyporheischen Interstitial führt zum Absterben der juvenilen FPM. Die verbleibenden, überalterten Populationen sind in ihrem gesamten mitteleuropäischen Verbreitungsgebiet von einem anhaltenden Rückgang betroffen. Im Untersuchungsgebiet wurden bereits zahlreiche Versuche unternommen, den Eintrag von Feinsedimenten und Nährstoffen aus landwirtschaftlichen Flächen und kommunalen Abwässern zu verringern. Die Rolle der Hunderte von Fischteichen in den Einzugsgebieten war dagegen umstritten und ungesichert. Das Hauptziel dieser Arbeit bestand darin, die Auswirkungen der Teiche auf verschiedene FPM Habitat-Parameter und auf mehreren Ebenen zu untersuchen, um ihre Bedeutung im Vergleich zu anderen Landnutzungsformen zu bewerten.

Die (Fisch-)Teiche sind mit den angrenzenden Gewässern direkt über oberirdische Zu- und Abflusskanäle und indirekt über das Grundwasser verbunden. Die Auswirkungen der mehr als 150 Teiche in einem der untersuchten Einzugsgebiete auf das hydrologische Regime wurden mit Hilfe hydrologischer Modellierung analysiert. Dazu wurden die hydrologischen Prozesse im Einzugsgebiet einschließlich der Teiche mit dem Soil and Water Assessment Tool (SWAT) modelliert und die resultierenden Abflüsse mit demselben SWAT-Modell nur ohne die Teiche verglichen. Die direkte Auswirkung über die Ablaufgräben der Teiche wurde durch den Vergleich der Wassertemperatur (Tw) ober- und unterhalb der Zuläufe aus ausgewählten Teichen untersucht (Kapitel 3). Insgesamt führten die Teiche zu einer Verringerung des Spitzenabflusses um 1,5 % und zu einer Erhöhung des Niedrigwasserabflusses um 4,5 %, was auf eine mäßige stabilisierende Wirkung auf das hydrologische Regime hinweist. Bei Hochwasser wird überschüssiges Wasser zurückgehalten, außerdem findet Grundwasserneubildung über den Teichboden statt. Die Einleitung des erwärmten Teichwassers erhöhte die Fließgewässertemperaturen um bis zu 5,5 °C direkt unterhalb der Teichabläufe. Die Landnutzung im Einzugsgebiet war ein weiterer wichtiger Faktor für die Tw. Die Auswirkungen der Teichabläufe wurden entlang bewaldeter Abschnitte kompensiert, während die Tw entlang offener Fließgewässerabschnitte weiter zunahm. Die lachsartigen Wirtsfische könnten Fließgewässerabschnitte

mit Muschelbänken aufgrund erhöhter Tw meiden, was die Infektionsrate verringern könnte. Andererseits erhöht ein Anstieg der Tw die Wachstumsraten der Jungmuscheln, so dass die Auswirkungen auf die FPM-Populationen im Jahresverlauf variieren. Diese Ergebnisse zeigen, dass Teiche dazu genutzt werden können, Wasser in der Landschaft zurückzuhalten, um extreme Niedrigwasserbedingungen abzumildern, die im Zuge des Klimawandels häufiger auftreten werden. Dabei muss ein durchdachtes Ablaufmanagement durchgeführt wird, um negative Auswirkungen auf die temperaturempfindlichen Wirtsfische zu vermeiden.

Zeitliche Schwankungen gab es nicht nur im Abfluss- und Temperaturregime in den untersuchten Einzugsgebieten. Auch die Feinsedimentdepositionsraten hingen stark vom Auftreten bestimmter Abflussereignisse im Jahresverlauf ab. Die Untersuchung der räumlich-zeitlichen Variabilität der (Fein-) Sedimentdynamik in den untersuchten Einzugsgebieten (Kapitel 4) ergab einen starken Unterschied zwischen Niedrig- und Hochwasserabflüssen und wiederum Unterschiede zwischen hohen Abflüssen infolge von Schneeschmelze und Starkregen. Während die Depositionsproben bei Niedrigwasser zu mehr als 80 % aus Feinsedimenten bestanden, was zu einer anhaltenden Verstopfung des Bachbetts führte, remobilisierten hohe Abflüsse während der Schneeschmelze das gesamte Gewässerbett, darunter auch grobe Partikel. Dadurch wurde Feinsedimente aus dem Gewässerbett gespült, die nur einen Anteil von 40 % an den Depositionsproben ausmachten. Starkregenereignisse mobilisierten ebenfalls größere Partikel, doch der hohe Eintrag von Feinsedimenten durch Bodenerosion überdeckte die Spülwirkung. Die unterschiedliche Landnutzung im Einzugsgebiet führte zu gegensätzlichen longitudinale Mustern der Feinsedimentdeposition entlang der drei untersuchten Fließgewässersysteme. Es wurde ein Zusammenhang zwischen erhöhter Feinsedimentdeposition und landwirtschaftlicher Nutzung festgestellt, während in bewaldeten Gebieten geringere Feinsedimentdepositionsraten beobachtet wurden. Die Auswirkungen der Fischteiche waren variabel. Einige Teiche hatten nahezu keine Auswirkungen, während andere erhöhte Feinsedimentdepositionsraten unterhalb der Zuflüsse von Fischteichabläufen verursachten, insbesondere im Herbst nach dem Abfischen der Teiche.

Die Auswirkungen des Abfischens der Fischteiche auf die Feinsedimentdeposition und den Eintrag von Schwebstoffen und Nährstoffen wurden für mehr als 20 verschiedene Abfischungen untersucht und mit der Hintergrunddeposition bei hohem und niedrigem Abfluss verglichen (Kapitel 5). Eine Teichentwässerung ohne Maßnahmen zur Verringerung der Freisetzung von Schwebstoffen führte zu maximalen Trübungswerten von mehr als 460 NTU, insbesondere während der Fischentnahme, und zu ähnlichen oder sogar höheren Feinsedimentdepositionsraten als bei hohen Abflüssen. Es wurden drei verschiedene Arten von Maßnahmen zur Verringerung der Schwebstoffeinträge (SSM) untersucht, die sich in ihrer Effizienz stark unterschieden: Während Absetzbauwerke wie Absetzbecken, oder die Entnahme von Fischen mittels Zugnetz bei geschlossenem Ablauf die Freisetzung von Schwebstoffen und die Feinsedimentdepositionsraten stark verringerten, führte die Filtration des Ablaufwassers durch einen Strohballen nicht zu einer Verringerung der Trübungswerte und der Feinsedimentdeposition im Vergleich zu einem normalen Abfischen ohne SSM.

Eine detaillierte Analyse der während des Ablassens mit und ohne SSM freigesetzten Schweb- und Nährstoffe in hoher zeitlicher Auflösung (Kapitel 6) zeigte erneut das große Potenzial ausreichend großer Absetzstrukturen zur Verhinderung des Eintrags von Schweb- und Nährstoffen in die angrenzenden FPM Gewässer. Ohne SSM konzentrierte sich etwa ein Drittel der freigesetzten Partikel im letzten 1% des Wassers, das während der Abfischung freigesetzt wurde. Das Abfischen bei geschlossenem Ablauf ist daher eine wirksame Maßnahme zur Verringerung von Feinsedimenteinträgen, wenn der Bau von Absetzbauwerken nicht möglich ist. Die Freisetzung gelöster Nährstoffe erfolgt jedoch kontinuierlich während des gesamten Ablassens und muss daher entweder direkt im Fischteich oder in einem zusätzlichen Klär- und Absetzteich angegangen werden, bevor das Ablaufwasser in den Vorfluter eingeleitet wird.

Der integrierte Ansatz, bei dem die hydrologische Modellierung des gesamten Einzugsgebiets mit umfangreichen Feldstudien an einzelnen Teiche kombiniert wurde, ermöglichte es, die potenziellen Auswirkungen dieser kleinen Wasserkörper auf eine breite Palette von Lebensraumparametern und über verschiedene zeitliche und räumliche Maßstäbe hinweg zu bewerten, auch im Vergleich mit größeren Landnutzungselementen. Während die allgemeinen und kumulativen Auswirkungen auf der Einzugsgebietsebene aus dem SWAT-Modell ersichtlich waren, führte die sehr unterschiedliche Teichbewirtschaftung zu variablen Auswirkungen auf der lokalen Ebene, die für die fragmentierten Populationen der verbleibenden FPM besonders relevant sind. In Gebieten mit extensiver Landnutzung können Fischteiche zu erhöhter Tw, Feinsedimentdeposition und Nährstoffkonzentrationen führen, die den Lebensraum für juvenile FPM verschlechtern. Die Studien haben jedoch auch gezeigt, dass die Bewirtschaftung der Teiche leicht angepasst werden kann, um die negativen Auswirkungen zu minimieren und dennoch von ihrem Wasserrückhaltepotenzial zu profitieren, um die mit dem Klimawandel verbundenen Gefahren zu mindern.

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1. General Introduction

1.1 Habitat requirements and threats to the freshwater pearl mussel

Freshwater bivalves are amongst the most threatened species worldwide (Lopes-Lima et al. 2018), with a prominent example, the freshwater pearl mussel *Margaritifera margaritifera* (L., FPM) being critically endangered throughout most of this Central European range (Geist 2010). The foremost abundant species is declining dramatically since the beginning of the 20th century (Lopes-Lima et al. 2017) with declines affecting up to 90% or even 100% of certain populations (e.g. Bauer 1988). Anthropogenic habitat degradation through flow regulation (Addy et al. 2012), eutrophication and pollution (Cosgrove et al. 2000), land use intensification, and siltation and following colmation (Geist and Auerswald 2007; Österling et al. 2010) are seen as the main reasons for the decreasing numbers. The species is long-lived, with a life-span between 80 years in Central Europe until up to 280 years in Scandinavian populations (Geist 2010; Lopes-Lima et al. 2017) and remains reproductively active and highly fertile independent of age (Bauer 1987). The species' life cycle is complex; it involves a parasitic phase on the gills of a suitable salmonid host fish. The only two host fish species used in Europe are Atlantic salmon (*Salmo salar* L.) and brown trout (*Salmo trutta* L.) (Skinner et al. 2003). After female mussels are fertilized by sperm ingested from the water column, the so-called glochidia larvae develop in the females' gill demibranchs for about 5-7 weeks (Lopes-Lima et al. 2017). Glochidia are usually released into the water column between July and August and need to attach to a suitable host fish within few days (Jansen et al. 2001). The glochidia encyst themselves on the host gills and further grow and develop until they complete their metamorphosis to juvenile mussels after up to 10 months (Thomas et al. 2010). After the metamorphosis, post-parasitic juveniles burry into the substrate where they stay in the interstitial zone for 5-7 years. As sub-adults, the mussels emerge from the substrate to start filtering in the free wave and finally mature after 10-15 years (Lopes-Lima et al. 2017) The post-parasitic phase seems to be the most critical life stage, as most European population completely lack sub-adult mussels younger than 30-50 years (Geist 2010). Within the substrate, juveniles rely on high oxygenation, which can only be achieved by infiltration of oxygen-rich surface water into the interstitial. The main reason for recruitment failure are siltation caused by high amounts of fine sediments within in the stream bed (Boon et al. 2019; Geist and Auerswald 2007; Österling et al. 2010) as well as high nutrient levels that cause biofilm formation and oxygen depletion in the substrate (Bauer 1988; Patzner and Müller 2001). If pore spaces between larger substrate particles are clogged by fine particles, oxygen supply to the interstitial is limited, which leads to high mortality of juveniles.

The evaluation of habitat requirements is often based on the comparison between functional and non-functional FPM habitats (e.g. in Geist and Auerswald (2007)). In general, the species requires cool, oligotrophic streams with low calcium levels and a gravelly substrate. The amount of fine sediments < 1.0 mm should not exceed 30% (Geist and Auerswald 2007). Different guidelines on the federal and national levels as well as the CEN standard for monitoring FPM populations give target values for certain water quality parameters (Table 1). They all state the very low level of nutrients and suspended particles/turbidity necessary for functional FPM population. The target values for nitrate, ammonium and phosphate do not indicate acute toxicity when exceeded but should be seen as indicators for the oligotrophic conditions found in functional FPM streams. Substrate quality can be assessed by comparing parameter values measured in the free wave and the interstitial habitat: the smaller the difference between the two habitats, the better the connectivity between them (Boon et al. 2019). Nowadays however, suitable conditions are scarce and can in most instances only be found in the headwater areas of their original Central European distribution (Baer 1995). Since particularly the juvenile stage seems to be critically sensitive to increasing anthropogenic pollution, the remaining overaged populations are often supported by artificially reared juveniles (Gum et al. 2011).

Table 1: Examples of target values for functional freshwater pearl mussel (FPM) populations in different countries/ federal states.

	Bavaria (Germany) (Sachteleben et al. 2004)	Saxony (Germany) (Erdbeer et al. 2021)	Czech Republic (Simon et al. 2017)	Sweden (Degerman et al. 2009)	EU CEN Standard (Boon et al. 2019)
O_2 concentration	-	> 8.5 mg/l	-	-	Saturation near to 100%
Temperature [°C]	-	Tw summer: 14.5 – 18.0; max 21.0	< 20.0	< 25.0	0.0 – 25.0; max ≤ 20.0
Redox potential [mV]	-	-	-	> 300	> 300; Δ between surfaces and interstitial < 20%
pH	>6.1 < 8.0	> 6.8 < 7.5	6.0 – 7.1	>= 6.2	6.2 – 7.5
Conductivity [µS/cm]	< 150	> 120 < 300	< 70	-	-
BSB_5 [mg/l]	< 3.0	< 1.6	-	-	< 1.0
Nitrate [mg/l]	NO_3-N: < 1.700	NO_3-N: < 3.000	NO_3^-: < 2.500	NO_3^-: < 0.125	NO_3-N: 0.125 – 0.500
Ammonium [mg/l]	NH_4-N: < 0.100	NH_4-N: < 0.05l	NH_4^+ < 0.100	-	NH_4-N: 0.010 – 0.050
Phosphorus [mg/l]	Ortho PO_4-P: < 0.060	Total P > 0.040 < 0.080	Total P < 0.020-0.035	Total P < 0.005-0.015	-
Turbidity	-	> 4.00 < 15.00 NTU	-	< 1.00 FNU	< 0.30 -0.96 NTU, peaks < 10.00 NTU
Proportion of particles < 1.0 mm	<25%	> 10 < 25%	-	<25%	-

Flow conditions shape the aquatic environment and are particularly relevant for the grain size distribution and substrate composition as well as the transport of excess fine sediments (Hauer 2015). The FPM requires a substrate that is both stable under high flow conditions to avoid detachment and downstream drift of individuals (Hastie 2000, Hauer 2015) as well as permeable enough to ensure a sufficient supply of oxygen-rich surface water into the juvenile habitat (Geist and Auerswald 2007). E.g. large boulders that accumulated finer material at their downstream side are expected to provide a suitable habitat for juveniles (Hauer 2015; Vannote and Minshall 1982).

Catchment land use is an important factor controlling the habitat quality and variability for aquatic organisms at the reach and microhabitat scale (Allan 2004; Bierschenk et al. 2019; Hauer 2015; Knott et al. 2019), in particular by governing the flow regime (Bari and Smettem 2004; Burt and Slattery 2006; Neitsch et al. 2011), and fine sediment and nutrient inputs (Delkash et al. 2018; Parsons 2005). Anthropogenic land use changes, such as the increase of area used for agricultural production, forest clear-cutting and logging, and urbanization generally resulted in an increase of fine sediment inputs into streams (Owens et al. 2005; Wood and Armitage 1997). Bare agricultural land is prone to rain erosion during thunderstorms and strong rain events (Cerdan et al. 2010) while the closed vegetation cover of extensive grasslands and forests can protect the soil from being flushed into adjacent streams. Monocultures of coniferous spruce trees can have negative effects as they lower the soil pH and increase the in situ subsurface weathering of rocks (Hauer 2015), therefore causing an increase in fine sediment inputs. The resulting fine sediment deposition degrades the interstitial habitat via siltation and colmation, leading e.g. to changes in the benthic algae and macroinvertebrate community (including freshwater mussels) as demonstrated in numerous studies (e.g., Aldridge et al. 1987; Bylak and Kukuła 2022; Extence et al. 2013; Graham 1990; Jones et al. 2012; Knott et al. 2019; Mathers et al. 2017; Wood and Armitage 1997) as well as the degradation of spawning grounds for lithophilic fishes (Argent and Flebbe 1999; Reiser and White 1988; Sternecker et al. 2013). Increased suspended solid concentrations (SSC) can impair periphyton growth due to decreased light availability, while increased nutrient concentrations might boost the growth of certain species (Rosemond et al. 2000), altering species composition and based on that, entire food webs (Henley et al. 2000). As mostly sessile species, freshwater mussels such as the FPM are less able to escape adverse habitat conditions as they are largely immobile and therefore have to cope with decreasing habitat quality. Various studies associate the dramatic declines in freshwater mussel populations to changes in land use: Österling and Högberg (2014) found a negative correlation between juvenile mussel density and the proportion of clear-cutting within a 50 m buffer strip in Swedish FPM streams. Hornbach et al. (2021) observed declines in species numbers and individuals in catchments in the U.S. that were subject to intensive wetland-drainage to generate arable land. And a study by Ma (2016) correlated lower mussel abundance in Scottish FPM streams with cropland within a 5 m riparian buffer, exhibiting higher nutrient concentrations than within forested buffers. All these land use changes are thought to increase fine sediment and/or nutrient inputs into mussel streams, degrading the habitat of this sensitive species (Bauer 1988; Österling et al. 2010). Catchment land use therefore seems to be one of the most important factors determining a suitable mussel habitat.

1.2 The role of ponds within (pearl mussel) catchments

Important yet often neglected catchment elements are small, standing water bodies like ponds. Although they account for 30% of the global standing water surface, represented by millions of individual water bodies < 1 km² (Downing et al. 2006), studies on the impact of ponds on adjacent streams systems remain scarce (Downing 2010; Ebel and Lowe 2013). Besides natural ponds and wetlands, man-made ponds are often constructed to serve multiple reasons, varying from water collection of irrigation over storm water detention ponds and recreational uses to fish production, offering provisioning, regulating and supporting ecosystem services (Biggs et al. 2017; Moore and Hunt 2012). Water is supplied to the ponds either by in-pond groundwater springs and precipitation alone (in Germany, these are called “sky-ponds”) or by abstracting water from nearby streams. Water can be abstracted in two ways: by blocking the water flow in a stream channel by an impoundment, creating a standing water body around the former channel, termed “on-stream ponds” by Ebel and Lowe (2013). Alternatively, water can be divided from a stream through a pipe or an open channel and directed it into a pond set next to the channel, termed “off-stream pond”. Constructed ponds are usually connected to the neighboring stream system by an outflow to be able to drain excess water, e.g. to facilitate fish harvest. Drainage can be controlled by a standpipe, consisting of multiple pieces that are removed subsequently to allow for a stepwise lowering of the water level to avoid downstream flooding. Fish ponds are often equipped with a so-called “monk” to allow drainage from both the surface or the bottom of the pond and facilitate fish harvest (for a detailed graphical description of the monk, please refer to Figure 23 in Chapter 6). Through these outlets, ponds are connected to the adjacent stream systems, however, the impact of such small water bodies on the receiving waters remains largely unstudied (Sayer 2014). While conservation managers seem to have gained interest in managing ponds as hotspots for biodiversity (Biggs et al. 2017; Lemmens et al. 2013; Sayer et al. 2012), the effects on downstream ecosystems and aquatic communities also needs to be understood in detail (Biggs et al. 2017). This holds particularly true for the pond-scape in Upper Franconia, in the north of Bavaria, Germany, where a high number of extensive fish ponds co-occur with the endangered FPM. The impact of the fish ponds on the FPM habitat has been controversially discussed as multiple effects have been proposed by various stakeholders: Ponds likely have an impact on the hydrological regime and might serve as water retention structures helping to mitigate drought conditions (David and Davidova 2015; Rao et al. 2017). On the other hand, water abstractions to supply water to the ponds during dry summers might lead to extreme low flows in the donor stream. (Fish) ponds can serve as sediment traps, retaining particles in the standing water body and therefore decreasing the inputs of fine sediments into the FPM streams (Berg et al. 2016; Schmadel et al. 2019). Sedimentation ponds are widely used in various contexts (Verstraeten and Poesen 2000). Nutrient turnover within the bottom sediments of such retention ponds can reduce nitrate concentrations and remove excess nitrogen to treat polluted runoff, as applied in storm water retention ponds (Persson and Wittgren 2003). However, such retention ponds are not managed for fish production, so fish stocks are neglectable. Critics fear that the additional feeding of fish to increase yields leads to a deterioration of the pond water quality, potentially leading to a degradation of the water quality downstream of fish ponds, as has been shown in several studies, e.g. by Mauch and Wittling (1991). As fish ponds are regularly drained to harvest the fish, large quantities of fine particles and nutrient rich water that accumulated in the pond over the growth period, might be released in a short time period. High concentration in the effluent might exceed the transport capacity of the receiving stream, increasing fine sediment deposition downstream of the pond. The discharge of heated effluents from standing waters, warmed up by intensive solar radiation, could lead to stream warming, which might have positive effects on growth of some aquatic species but also poses the threat of exceeding temperature tolerances of cold-stenothermic species such as salmonids. To evaluate the impact of the fish ponds on the FPM habitat and develop mitigation strategies, if necessary, all these aspects need to be taken into account, as well as habitat requirements of its host fish. Two studies examined the impact of ponds/small lakes and wetland on FPM distribution in Scandinavia and came to contrasting results: While ponds and small lakes seem to favor FPM recruitment in Swedish catchments (Österling and Högberg 2014), Gosselin et al. (2022) found a negative relationship between the proportion of wetland cover and FPM recruitment. This gives a hint towards the variable impact, ponds can have on adjacent FPM streams. Moreover, a holistic assessment needs to consider different temporal und spatial scales, as pond impacts might vary with distance and throughout the year. Because ponds are an integrated part of the landscape, the impact of ponds need to be evaluated with regard to other catchment features that have impacts on the studied parameters, such as land use. Therefore, background conditions that serve as a baseline should be measured over multiple seasons and discharge events to assess the pond-specific effects in comparison with other catchment features. The following paragraphs will give an overview on how different catchment features might influence FPM habitat parameters.

1.3 Hydrologic regime and spatiotemporal patterns of fine sediment and nutrient dynamics of ponds

Flow is one of the most important drivers shaping the aquatic environment and its communities. The hydrologic regime describes the variation in stream flow over time and can be defined after Poff et al. (1997) by the five components magnitude, frequency, duration, timing and rate of change. These can also be used to describe flow events such as floods or low flows (Zeiringer et al. 2018). The hydrologic regime, is determined by the water cycle that ensures the constant transfer of water via evaporation and transpiration from land or water bodies into the atmosphere, where it is stored and eventually released as precipitation. The precipitation might infiltrate into soils, where it is either taken up by plants, percolates into the groundwater aquifer or flows into the next water body as subsurface flow, or accumulates and flows overland as surface runoff into a nearby stream or lake. In the Soil and Water Assessment Tool (SWAT), a model used to simulate hydrological processes within catchments, the hydrologic cycle is described by the following equation:

$$SW_t = SW_0 + \sum_{i=1}^{t}(R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}) \qquad \textit{(Equ. 1)}$$

where SW_t is the final soil water content (mm H_2O), SW_0 is the initial soil water content on day i (mm H_2O), t is the time (days), R_{day} is the amount of precipitation on day i (mm H_2O), Q_{surf} is the amount of surface runoff on day i (mm H_2O), E_a is the amount of evapotranspiration on day i (mm H_2O), w_{seep} is the amount of water entering the vadose zone from the soil profile on day i (mm H_2O), and Q_{gw} is the amount of return flow on day i (mm H_2O) (Neitsch et al. 2011).

These parameters largely depend on soil type, land use (in particular in small catchments (Ashmore and Church 2001)), slope and climatic factors such as precipitation form (rain vs snow), intensity and timing, air temperature and wind speed. Soil parameters such as texture influence the soil hydraulic conductivity, water infiltration, and storage capacity, while land use parameters such as vegetation control water uptake from the soil and the degree of evapotranspiration. The retention capacity of the various water storages within a catchment govern the hydrologic response to precipitation events and peak-runoff generation. Ponds and wetlands offer water storage directly within the water body itself or by increasing local groundwater storages (Juszczak et al. 2007; Lhotský 2010; Martinez-Martinez et al. 2014), potentially decreasing downstream peak flows and delaying the release of the excess water. This lag time might help to increase stream flow during dry periods (Wu et al. 2023). The proportion of streamflow that is sustained during dry periods, when no precipitation falls to support stream runoff, is called baseflow (Price 2011). During such periods, water is delivered to the stream through subsurface inputs from groundwater and drainage of near-surface storages of riparian soils and wetlands or ponds (Morley et al. 2011). Due to the water retention potential of ponds, they are said to stabilize the water balance (LAWA 2023), because they can reduce peak flows on the one hand, while increasing low flows on the other hand. However, ponds are often neglected in large-scale hydrological modeling approaches. Jalowska and Yuan (2019) reviewed 30 studies using one of the three possibilities to incorporate impoundments (reservoirs, wetlands, ponds) into SWAT models and found only 30% of them offered details on impoundment parametrization. Reservoirs were the majority of impoundments modeled, while ponds were only integrated into two of the reviewed studies. In general, ponds are rarely considered in hydrologic studies, which poses the risk of overlooking important processes in the water, sediment and nutrient cycles of the studied catchments. The poor representation of small- and meso-scale conditions in such models might be due to neglecting the role of small, local features, such as ponds (Baldan et al. 2021b).

A pond's water balance is influenced by the inputs it receives via inflow from the catchment or water abstraction from the adjacent stream system, and precipitation and the losses from evaporation, seepage and outflow to the receiving water body. In SWAT, the water balance of a pond is calculated using the following equation:

$$V = V_{stored} + V_{flowin} - V_{flowout} + V_{pcp} - V_{evap} - V_{seep} \qquad \text{(Equ. 2)}$$

where V is the volume of water in the pond at the end of the day (m³ H_2O), V_{stored} is the volume of water stored in the water body at the beginning of the day (m³ H_2O), V_{flowin} is the volume of water entering the water body during the day (m³ H_2O), $V_{flowout}$ is the volume of water flowing out of the water body during the day (m³ H_2O), V_{pcp} is the volume of precipitation falling on the water body during the day (m³ H_2O), V_{evap} is the volume of water removed from the water body by evaporation during the day (m³ H_2O) and V_{seep} is the volume of water lost from the water body by seepage (m³ H_2O). The volume of water entering the pond is subtracted from the surface runoff, lateral flow and groundwater loadings to the main channel (Neitsch et al. 2011).

Within the pond's water body, flow velocity is drastically reduced, which leads to a settling of particles (Verstraeten and Poesen 2000) that are delivered either from the inflowing water or via overland flow from the catchment. Trap efficiency, the proportion of incoming sediment that is retained in the pond (Heinemarm 1981), depends on the pond characteristics and the sediment load of the runoff it receives. Larger particles will settle more quickly after entering the pond, while fine particles will need a longer residence time (Verstraeten and Poesen 2000). Moreover, bottom-feeding fish like carp constantly disturb the pond bottom over the growing season, leading to a remobilization of fine sediments and adhered nutrients (Matsuzaki et al. 2007; Ritvo et al. 2004; Scheffer et al. 2003), potentially decreasing the trap efficiency and causing higher outputs of suspended particles.

Within the stream, transport of particles also strongly depends on the grain size as the critical (mean) flow velocity needed to initiate the movement of particles varies (Hjulström 1935). Larger particles need higher flow velocities to be mobilized and to be actively transported over longer distances. Fine sediments, in particular sand and fine gravel, are the most mobile fractions under even low flow conditions. However, if flows are not strong enough to mobilize even the smaller grain sizes and suspended particle concentrations exceed transport capacities, fine sediments are deposited on the stream bed, infiltrate into the substrate. This results in a clogging of the pores between gravels, filling up those spaces and decreasing the connectivity between interstitial habitat and the water column. The ongoing process of constant fine sediment deposition gradually filling up the inter-gravel spaces and sealing the stream bed is termed colmation, described by Schälchli (1992). It builds up under low flow conditions and increases stream bed compaction and resistance to higher shear stress until a bed mobilizing flood destroys the clogged layer and fines are resuspended and transported downstream. Therefore, the substrate quality in FPM habitats is depending on fine sediment inputs and on discharge conditions, as well as their interaction and variation in space and time. E.g., prolonged low flows during droughts not only pose the direct threat of drying out streams and mussel beds but also indirectly reduced habitat quality of juveniles by increasing the risk of colmation even at low inputs of suspended particles.

Ponds not only accumulate particles but are often also enriched in soluble nutrients. Nutrients are delivered to the pond either with the inflowing water and terrestrial inputs or are added by the fish farmer as fish feed or fertilizers to increase pond productivity. Nutrients like nitrogen (N) and phosphorous (P) compounds can either be bound to particles and settle to the pond bottom or are transformed via different microbial processes at the pond bottom-water-interface. Within the pond bottom, oxygen availability quickly decreases and the sediments become more anoxic (Boyd and Thunjai 2002). Oxygen delivery from the water column is limited due to the low permeability of pond soils and its consumption is increased through the degradation of organic matter accumulated in the pond sediments. As oxygen is depleted in deeper sediment layers, other electro acceptors are used in nutrient cycling. The most important pathways for nitrogen turnover in ponds and wetland are denitrification, anaerobic ammonium oxidation (anamox) and dissimilatory nitrate reduction to ammonium (DNRA). Anamox and the denitrification of nitrate (NO_3^-) over NO_2^-, NO and N_2O to gaseous N_2 lead to a complete removal of N from the system, while DNRA reduces NO_3^- to NH_4^+, so the N is only recycled and stays in the system (Burgin and Hamilton 2007; Rahman et al. 2019). All three processes lead to reduced nitrate concentrations in the pond water (Castine et al. 2012). Making use of these processes, fish-free ponds are used to treat polluted storm water runoff (Rahman et al. 2019), aquaculture effluents (Jackson et al. 2003) or waste water. However, fish ponds might receive additional nutrient inputs in the form of fertilizers and feeds. They are either eaten and processed by the fish that then excreted waste products as ammonium via the gills or as feces, or sink to the ponds bottom uneaten where their mineralization by microorganisms represents another ammonium source (Hargreaves 1998). Ammonium is in equilibrium with its un-ionized form, ammonia, which is more toxic to fish. Transition between the ionized and un-ionized form depends on temperature, salinity and pH, with a higher proportion of un-ionized NH_3 at increasing pH values. NH_4^+ is highly bioavailable and usually quickly processed (Tank et al. 2018; Webster et al. 2003), however, if available in concentrations exceeding the uptake capacity, it can cause eutrophication of downstream reaches. Excess nutrients in pond effluents might deteriorate water quality in the receiving water bodies (Crab et al. 2007). The amount of excess nutrients released from fish ponds depends on the production intensity, N-removal processes might prevail in extensive carp ponds with low feed inputs (Kestemont 1995), while on average 75% of feed nitrogen content in intensive production systems are not incorporated in fish biomass (Crab et al. 2007; Hlaváč et al. 2014) but accumulate in the pond. Concerning phosphate, another important nutrient in aquatic systems, its free ortho-phosphate form is highly reactive and therefore usually taken up immediately by phytoplankton (Conley et al. 2009). However, most of the total phosphate (TP) found in aquatic ecosystems is bound to particles and therefore unavailable for algal uptake (Baken et al. 2016). Particle-bound phosphate entering stagnate aquatic systems therefore settles to the bottom and is retained within the sediments where it can build up in high concentrations over time. From there, ortho-phosphate can be released into the water column, in particular under anoxic conditions (Selig 2003) as found in the pond bottom sediment. This process, also known as ‘internal loading’, has e.g. been observed in a hypertrophic fish pond by Potužák et al. (2016). This effect might be accelerated by carp foraging and digestive activities (Lamarra 1975). The function of fish ponds as sinks or sources of nutrients varies depending on ponds characteristics such as water residence time, production intensity and trophic state as well as the quality of the intake water and the receiving stream. Gál et al. (2016) found 23 semi-intensive cyprinid ponds retained on average 48% of the total nitrogen and 62% of the total phosphorus supplied by the intake water but discharged 78% more organic matter into the receiving streams. Potužák et al. (2016) found differing P retention capacities in a study of nine fish ponds: while semi-intensive ponds often acted as a source for phosphorous, extensive ponds used only for angling showed a high retention efficiency for P from intake waters.

Even if extensive carp ponds serve as a nutrient and fine sediment trap over most of the pond management cycle, these substances are likely being released during the pond drainage, conducted to gather the fish near the pond outflow to facilitate fish harvest (Banas et al. 2002; Banas et al. 2008; Kestemont 1995; Masson et al. 2005; Soongsawang and Boyd 2012). When the water level decreases, increasing water drag and the movements of fish and fishermen remobilize the pond bottom and increase the suspended solid concentration. Solids and nutrients accumulated in the pond over the production period might be released within a short period of time, potentially exceeding the transport capacity of the receiving water body. This can cause increased fine sediment deposition, colmation and oxygen depletion within the sediments downstream of the pond due to enhanced stream metabolism resulting from excessive inputs of nutrients and organic matter. Pond drainage of a 2 ha cyprinid pond released more than 2800 kg/ha suspended particles, 36.5 kg/ha total nitrogen and 5.1 kg/ha total phosphorous in one drainage operation during heavy rainfall (Banas et al. 2002). In particular, a high proportion of organic matter (Dróżdż et al. 2020) in the effluents has detrimental effects on the receiving water body, such as increased biological oxygen demand (BOD). Different measures are proposed to minimize inputs from pond drainage. For suspended solids, this includes the use of physical settling structures such as vegetated strips (Ghate et al. 1997) or settling ponds (Bohl 1988; Soongsawang and Boyd 2012; Tucker et al. 2008). The efficiency of such structures strongly depends on their size and retention time in relation to the amount of particles in the pond effluent. Soongsawang and Boyd (2012) estimated a 6240 m² large, 1 m deep settling basin with a resident time of 24 h to remove suspended mineral soil particle < 10 µm from drainage effluents of 0.5 ha pond area, including an additional removal of some soluble nutrients. The pond area was designed particularly generous and their equation included enough retention time to prevent rapid siltation of the settling pond and allow for a settling of small particles. In a field-based study, Frimpong et al. (2004) found suitability of 100 m long ditches to remove suspended solids from pond effluents highly variable, depending on ditch characteristics such as amount of vegetation. Another option proposed for Alabama channel catfish farming, where huge settling basins would be needed to treat effluents, is the reduction of drainage volume and improvement of drainage water quality (Boyd and Queiroz 2001) as well as the reduction of overall drainage numbers. Lin et al. (2001) suggest seining a pond without drainage to prevent the release of polluted water to the receiving water bodies, which is also mentioned by Bohl (1988). However, a complete renunciation of the draining of water might stand in conflict with fish welfare

The harvest technique seems to be an important target for the mitigation of negative effects from polluted effluents during pond drainage regarding suspended solids as well as soluble nutrients. The drainage management needs to be adapted to the specific pond and conditions in the receiving water body.

1.4 The thermal regime of streams and ponds

The thermal regime of a water body describes the patterns of variation in water temperature (Tw), including seasonal and daily variations, and spatial effects (Caissie 2006). As poikilothermic animals, most aquatic organisms strongly rely on Tw for their growth and metabolic processes and different organisms have specific temperature ranges and optima (Larsson 2005; Sylvester 1972; Wehrly et al. 2003). Tw therefore controls their distribution over multiple spatial scales, from the overall distribution along climate zones, over regional patterns to the small scale use of cold-water spots within stream stretches (Ebersole et al. 2001; Taeubert et al. 2014). Freshwater mussel growth is strongly dependent on Tw (Bauer 1992; Buddensiek 1995), with populations of the FPM in Scandinavia showing extremely slow growth rates and long life span of more than 200 years (Geist 2010), while Spanish populations have a much higher growth rate and shorter lifespan of only 35 years (San Miguel et al. 2004). Increased temperatures lead to enhanced growth in juvenile freshwater mussels in rearing facilities (Carey et al. 2013) and in field experiments, growth of juvenile FPM exposed in mesh cages showed a positive relationship with water temperature (Černá et al. 2018). Furthermore, different mussel life stages might exhibit different thermal tolerances (Benedict and Geist 2021) and unionid mussels do not only rely on their own thermal requirements but also on the thermal tolerances of their host fish (Pandolfo et al. 2012). Increasing Tw due to heated effluents or the ongoing climate change might therefore be a threat to the FPM that rely on cold-water adapted salmonid host fish (Smialek et al. 2021). Salmonids like brown trout actively search for cold-water thermal refuges during high Tw (Ebersole et al. 2003). Such cold-water patches can be associated with cold-water tributaries (Ebersole et al. 2015), side channels, alcoves, lateral seeps, floodplain springbrooks (Ebersole et al. 2003) and deep pools (Nielsen et al. 1994). If exceedance of their thermal limits causes the hosts to migrate to cooler habitats during the mussel spawning season, they might get out of the range of the mussels releasing glochidia.

Tw is influenced by multiple factors related to atmospheric conditions, topography, stream bed conduction and hyporheic exchange with the groundwater, and stream discharge (Caissie 2006). At the local and regional scale, factors such as incoming tributaries (Ebersole et al. 2015), thermal effluents from power plants (Sylvester 1972) or water abstractions also play a role, together with land use/ land cover effects from riparian shading or urban development. Tw in small stream is more affected by riparian land use and groundwater contribution (Hofmeister et al. 2015) than in larger streams where heat exchange due to solar radiation at the air-water-interface dominates over shading and streambed-water heat exchange (Caissie 2006). Riparian vegetation is seen as a major factor in headwater streams as shown in many studies on the effects of logging and clearcutting around forested streams reviewed by Poole and Berman (2001) and Caissie (2006) and later contributions e.g. from Garner et al. (2017);

Garner et al. (2015); Monk et al. (2013) and Timm et al. (2021). Canopy cover shields solar radiation from reaching the water surface and reduces convection and advection of heat energy due to reduced wind speed (Poole and Berman 2001), which is why deforestation often leads to increased Tw. Impoundments have been shown to be another anthropogenic perturbation of Tw. Due to their larger and more exposed surface, impounded water bodies like ponds receive more solar radiation and therefore exhibit much higher Tw then streams (Dripps and Granger 2013). The effect of water warming in ponds is the only reason, why carp production is feasible in the temperate zone of Central Europe (Balon 1995). If heated surface water is discharged into cool headwater streams, Tw in the receiving stream can be increased up to 10 °C (Shetter and Whalls 1955) and elevated water temperatures often persist for long distances. Dripps and Granger (2013) assessed stream heating downstream of three impounded lakes between 1.7 and 4.5 ha and found average summer stream temperatures were increased between 4.5 and 6.0 °C directly below the lakes and the elevated Tw persisted over several hundred meters. Ponds were also found to alter thermal signatures by increasing summer stream temperature (Seyedhashemi et al. 2021). These increases in Tw might be too high to be fully mitigated by measures such as riparian tree planting (Marteau et al. 2022) and might therefore be an important stressor for aquatic communities, potentially increased by the ongoing climate change.

Due to its high spatiotemporal variability and complexity it is difficult to describe the thermal regime of a stream. Arismendi et al. (2013) suggest using a combination of several factors related to magnitude, variability (daily, seasonal, annual), frequency, timing and duration of thermal events to cover several aspects of the thermal regime, which also allows to include the thermal requirements of target species. For example, short-term extreme temperatures might limit a species distribution at a given time and for a certain time period but might go unattended if considering only mean temperatures over a specific time period. Looking at patterns in a multitude of parameters offers the chance detect changing trends and attribute them to changing atmospheric conditions, land use and other anthropogenic factors.

Variation in stream thermal regimes is expected to increase through the ongoing climate change (Arismendi et al. 2013; IPCC 2022). Besides changes in weather patterns and increasing occurrence of extreme events such as strong rainfalls, the increasing air temperature also affects the temperature of water bodies (Mohr et al. 2021; Webb et al. 2003; Webb et al. 2008). Particularly during low flow conditions, mostly occurring during the hot summer months with low precipitation levels, already decreased water volumes with a reduced thermal capacity will heat up disproportionately high due to the high air temperatures and strong solar radiation (van Vliet et al. 2011). Ponds are particularly sensitive to increases in air temperature and precipitation patterns as they strongly depend on inflow from precipitation and water losses through evaporation (Matthews 2010). The predicted changes in precipitation patterns might be outweigh by increased evapotranspiration, causing decreased water levels more susceptible to Tw increases. Heated discharges from ponds are therefore likely to further increase the threat of thermal pollution of adjacent water bodies and their fauna.

1.5 Objectives

As demonstrated, multiple impacts of fish ponds on adjacent pearl mussel streams are expected, reaching from hydrology over temperature effects to nutrient and fine sediment dynamics. Therefore, a detailed understanding of the complex interplay of pond processes on multiple spatial and temporal scales, and the water and substrate quality of the FPM habitat (Figure 1) is essential to include the ponds into FPM conservation plans. Due to the complex life cycle, suitable FPM streams or stream stretches also need to ensure favorable conditions for the salmonid host fish. In order to evaluate the proposed effects of fish ponds and their interactions, four studies were carried out between 2018 and 2021 in two FPM catchments in North Bavaria, Germany. The five streams studied contain both more than 500 pond facilities as well as some of the largest remaining FPM populations in Central Europe. A combination of hydrologic modeling and field studies covering a wide range of discharges and sampling sites was conducted to cover as many relevant aspects as possible. The obtained results were used to propose management implications to maximize beneficial effects while minimizing negative effects of fish ponds. A special focus was laid on the pond drainage and fishing process, with one study quantifying fine sediment and nutrients released during pond drainage under different mitigation measures and comparing them to baseline conditions in the study streams. A second study on the details of suspended particle and nutrient release at a fine temporal scale was conducted to gain insight into the mechanistic of pond effluent quality. The following key research questions were addressed:

i) What are the cumulative effects of the ponds on the hydrologic regime and how do individual ponds affect the thermal regime of adjacent streams (Chapter 3)? This information will give an overview on the general effects of the ponds and via which pathways they are connected to the adjacent streams.

ii) How variable are the sediment dynamics and habitat conditions along the five study streams under different discharge conditions (Chapter 4)? This information is needed to evaluate the impact of the ponds in relation to other land uses.

iii) How much fine sediment and how many nutrients are released from pond drainages using different drainage and fishing techniques and how effective are different mitigation measures applied during pond drainage (Chapter 5)? It is essential to investigate this particular time point in the pond management cycle, when discharge of pond effluents into the receiving streams and the resulting effects are highest.

iv) Which processes affect fine sediment and nutrient cycling in fish ponds and what does that mean for the mechanisms of release of suspended solids and soluble nutrients during pond drainage (Chapter 6)? This will give insights into the processes in the pond that are responsible for the effluent quality during pond drainage.

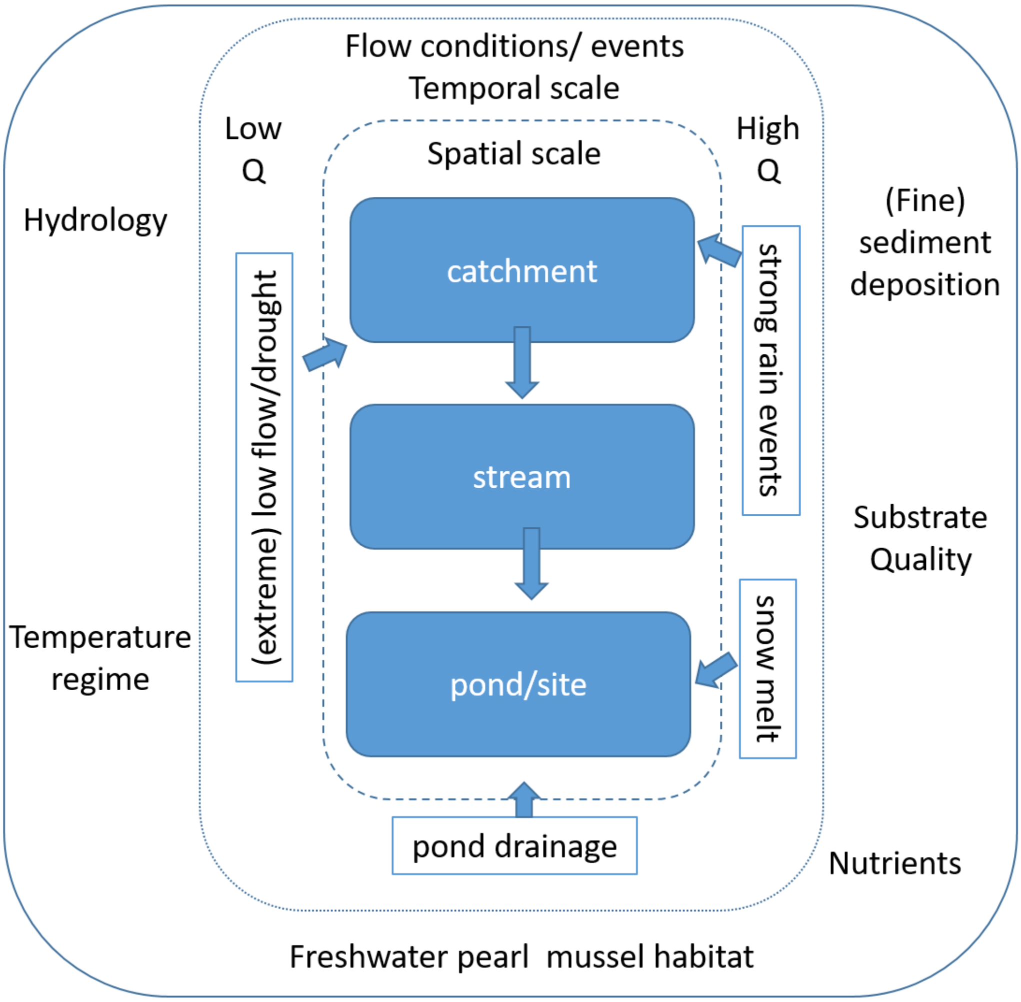

Figure 1: Conceptual overview of factors influencing the FPM habitat over multiple spatial and temporal scales, different discharges and the effects of fish ponds.

2. Material and Methods

In the present thesis, multiple methodologies were used to gain as holistic an understanding as possible of the mechanisms responsible for shaping the overall conditions within the study stream over time and space, with a particular focus on the impact of fish ponds. Theoretical approaches like hydrologic modeling were applied together with field-based monitoring campaigns and automatic data loggers.

2.1 Study area

The studies in this thesis were conducted in the catchments of the “Südliche Regnitz” (SR) and the “Schwesnitz” (SCHW), two subcatchments of the “Sächsische Saale” within the Elbe drainage (Figure 2). The area is located in the so-called “Dreiländer-Eck Bayern-Sachsen-Tschechien” at the border region between Germany (with the federal states Bavaria and Saxony) and the Czech Republic. Here, some of the largest remaining freshwater pearl mussel (FPM) populations can be found in the SR and its major tributary, the “Zinnbach” (ZB), as well as in the “Höllbach” (HB) and its major tributary, the “Mähringsbach” (MB), that after the confluence with the “Perlenbach” (PB) forms the SCHW. The only remaining relict FPM population of the former densely populated PB can be found in the “Bocksbach” (BB) tributary. Beyond the valuable FPM populations, there are more than 500 pond facilities with a total area of 85.57 ha in the two catchments, mostly used for extensive carp production. These ponds were mostly built rather recently, in the mid-20th century, in contrast to the centuries-lasting tradition of carp farming that is practiced since the 11th - 12th century in Central Europe (Balon 2004). Currently, most of the ponds are managed by private owners, except for one commercial facility in the upstream region of the BB, whose origins can be traced back to the 16th century, the fish breeding to the late 19th century.

Climate in the region is temperate, with a mean annual precipitation sum and temperature at the climate measurement station “Hof” (ID 2261) of 757.6 mm and 7.1 °C, respectively. The soils in the upper part of the two subcatchments, where the FPM streams are located, consists mainly of quartzite slate, with a more diverse geology in the downstream regions. Land use/ land cover (LULC) beyond the ponds differs slightly between the two study catchments: While LULC in the SR catchment in nearly equally distributed between forest (30.0%), pasture (35.5%) and agricultural use (33.0%), the SCHW catchment is dominated by coniferous forest (50.4%) followed by agricultural land use (22.6%) and pasture (18.8%). A gradient with increasing land use intensity exist along the stream courses, allowing for a comparison of less intensively used upstream sections to more intensively used downstream sections.

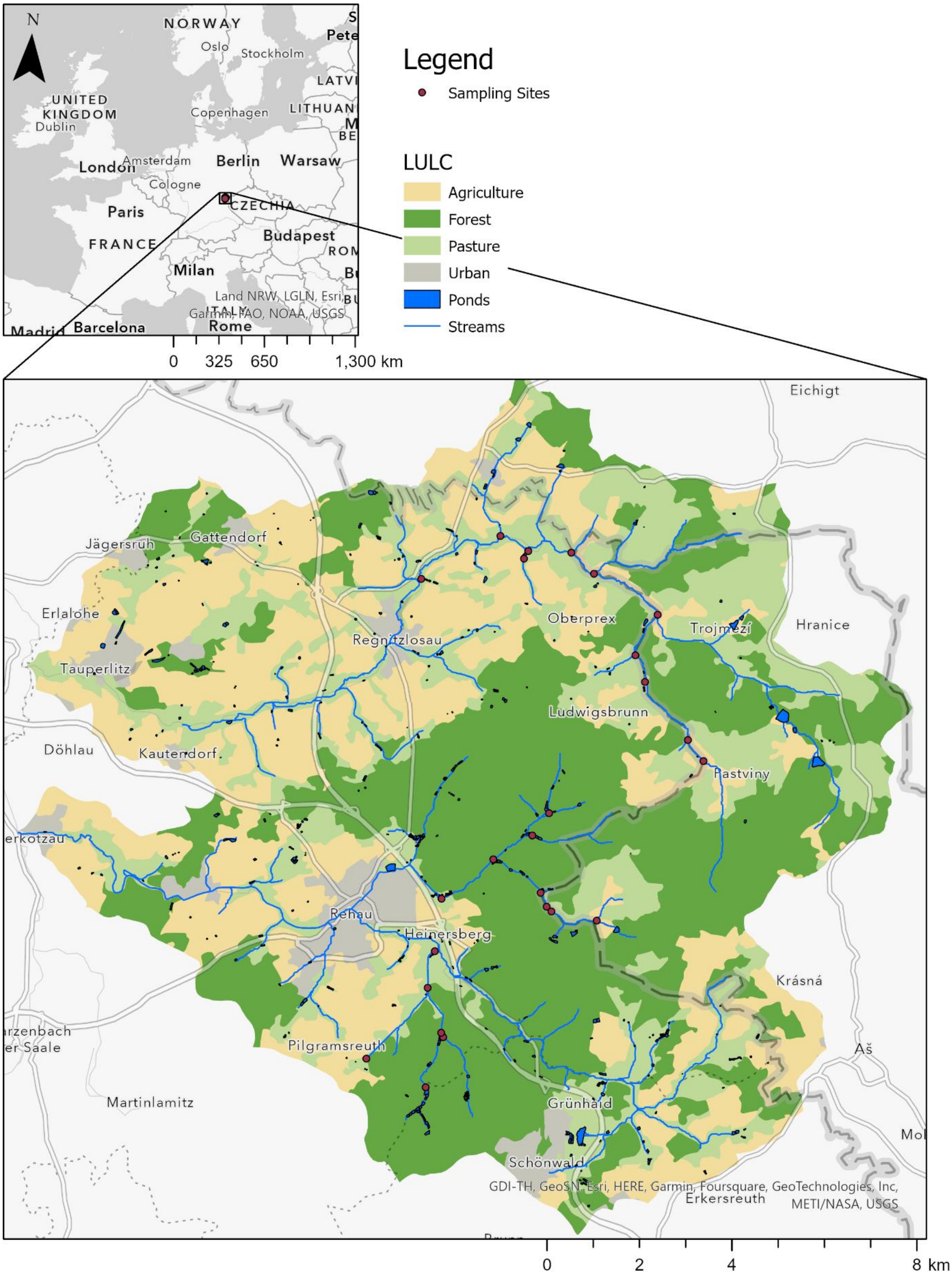

Figure 2: Map of the study area, the extend of the two studied catchments is represented by colored areas representing different land use types. Blue lines represent streams, while blue areas represent ponds; red dots represent positions of sampling sites; the map was created using ArcGIS Pro, using the "Human Geography Basemap" (Version 3.0).

2.2 Hydrological modeling

The Soil and Water Assessment Tool (SWAT) was chosen to model the influence of the ponds on the hydrology of the SCHW catchment. The model uses physically based hydrological modeling of the water balance on the catchment level and offers a simplified model description of erosion and sediment transport processes. A digital elevation model (DEM), data on land use, soil composition and regional climate are required for the initial model setup in "ArcSWAT" (2012, v10.2), which allows to merge the source data in form of raster data, shapefiles and tables within ArcGIS to build the model. The stream courses are delineated from the elevation differences in the DEM and the entire catchment is divided into smaller sub-basins. Based on individual combinations of land use and soil types, as well as slope class, the sub-basins are further divided into so-called HRUs (hydrological response units), for each of which all relevant hydrological processes are modeled using station-based climate data. The resulting surface runoff is then routed through the hierarchically structured stream network.

Input data on land use, soil composition, slope, and a DEM with a resolution of 5 x 5 m for the SCHW catchment were collected from several sources in Bavaria and the Czech Republic and merged into suitable maps using ArcMap 10.5. (ESRI, Inc., Redlands, CA, USA). Pond location and area were based on aerial imagery. SWAT allows a variety of adjustments to specific conditions and management options such as water withdrawals and land management at different levels: Basin, sub-basin and HRU. Ponds can be included at sub-basin level, but only one pond can be defined per sub-basin. For sub-basins with multiple ponds, they were combined into a single pond, following the hydrological equivalent wetland (HEW) concept (Wang et al. 2008). These ponds were integrated into the model following the procedure described by Jalowska and Yuan (2019) for incorporating water storage (reservoirs, ponds, and wetlands). The model was then calibrated using the SWAT CUP software to adjust parameter values to reduce uncertainty due to estimated or generalized values to better match simulated runoff and runoff measured at a gauging station. The calibration process was performed over multiple iterations, where the result of each iteration were compared to the observed values to narrow down the parameter range used in the next iteration. Several model evaluation criteria were considered to rate the performance (Moriasi et al. 2015) until a satisfactory result was achieved. The model performance of the calibrated model was further verified by a validation using a new data set of observed runoff data. Once the best parameter values were obtained, they were used in a second model, using the identical setup besides all ponds were excluded. This procedure allowed for a comparison of a model representing the current situation (with ponds) to a hypothetic model with changed conditions (without ponds). High, peak and low flow conditions were differentiated to account for the different hydrological processes involving ponds in the hydrological cycle.

The SWAT model was not only used to analyze the impact of the ponds on hydrology but also to get an insight into the overall functioning of the catchment. The baseflow index (BFI) was calculated using the hydrograph simulated by the model with ponds for the years 2018-2020, to compare the groundwater contribution during different discharge condition during the field study period, following the recommendations of the World Metrologic Organization (WMO 2008).

2.3 Assessing (fine) sediment dynamics, water and substrate quality

Water quality, in particular nutrient concentrations, is an important parameter to evaluate habitat suitability for sensitive species such as the FPM, as it is adapted to cool, oligotrophic conditions and population declines are often associated with a decline in water quality (Boon et al. 2019; Denic and Geist 2015; Stoeckl et al. 2020). Furthermore, increased fine sediment loads and resulting deposition on the stream bed degrade the juvenile habitat (Geist and Auerswald 2007). The habitat quality of the interstitial as well as (fine) sediment dynamics are therefore important factors to assess, in order to identify fine sediment sources and to develop methods to reduce inputs and restore the juvenile habitat.

2.3.1 General sampling design

To investigate the effect of the ponds on the FPM habitat, multiple sampling sites were established along each of the five study streams, each at the inflow from a pond system into the FPM streams (Figure 3). This allowed for comparison of conditions upstream and downstream of the confluence to examine the effect of several pond outlet channels and tributaries on the mainstream. At each sampling site, abiotic water parameters in the open water and interstitial, nutrient concentrations, substrate quality, sediment deposition and hydromorphological parameters were investigated at three points: 'us' (upstream of the confluence), 'out' (directly within the outlet channels), and 'ds' (approximately 20-30 m downstream of the confluence).

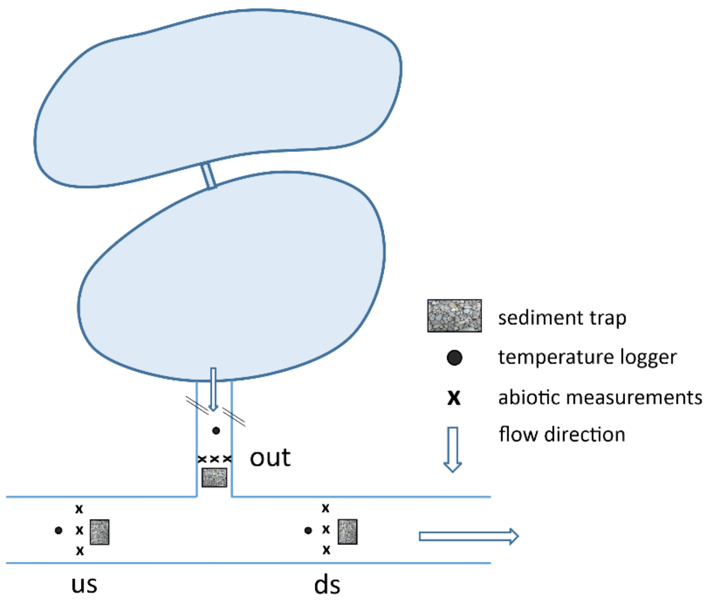

Figure 3: Sampling design used to assess the impact of fish ponds on sediment dynamics, water and substrate quality and water temperature.

Sampling was conducted once per month from June to November at all sampling sites during low flow conditions, as well as during specific events such as heavy rains, winter snowmelt, and spring to examine the effect of different discharge conditions, the pond management cycle and seasonal effects. The focus was on summer, the main growth phase of the FPM, with the greatest deficits due to reduced discharge, high water temperatures and low oxygen concentrations, as well as on the pond drainage and fish harvest in autumn.

2.3.2 Sediment dynamics

Sediment deposition over a weekly interval was determined using sediment traps. This methodology collects particles delivered by vertical deposition from the free wave over time and has been already used to assess spatiotemporal variation of sediment deposition in FPM streams (Denic and Geist 2015; Pander et al. 2015) and small, agriculturally used catchments (Knott et al. 2019). The traps were filled with an allochthonous 16/32 mm gravel mixture from the alpine foothills, so that the substrate of the traps could be easily distinguished from the natural substrate of the study streams. One trap per monitoring point was buried within the stream bed and left open to collect particles transported during different discharge conditions (Figure 4). Particles deposed during one week were separated by wet sieving using a sieving tower (Retsch GmbH, Haan, Germany) into the following grain size fractions: > 20 mm, 6.3 - 20 mm, 2.0 - 6.3 mm, 0.85 - 2.0 mm, and < 0.85 mm. The individual fractions were dried at 102 °C and the weight was determined to the nearest 0.5 g. The fraction < 0.85 mm is defined as 'fine sediment', as particles < 0.85 mm have a detrimental effect on the permeability of the aquatic substrate (McNeil & Ahnell, 1964) and consequently on benthic life stages such as fish eggs and juvenile FPM (e.g., Reiser & White, 1988; Argent & Flebbe, 1999).

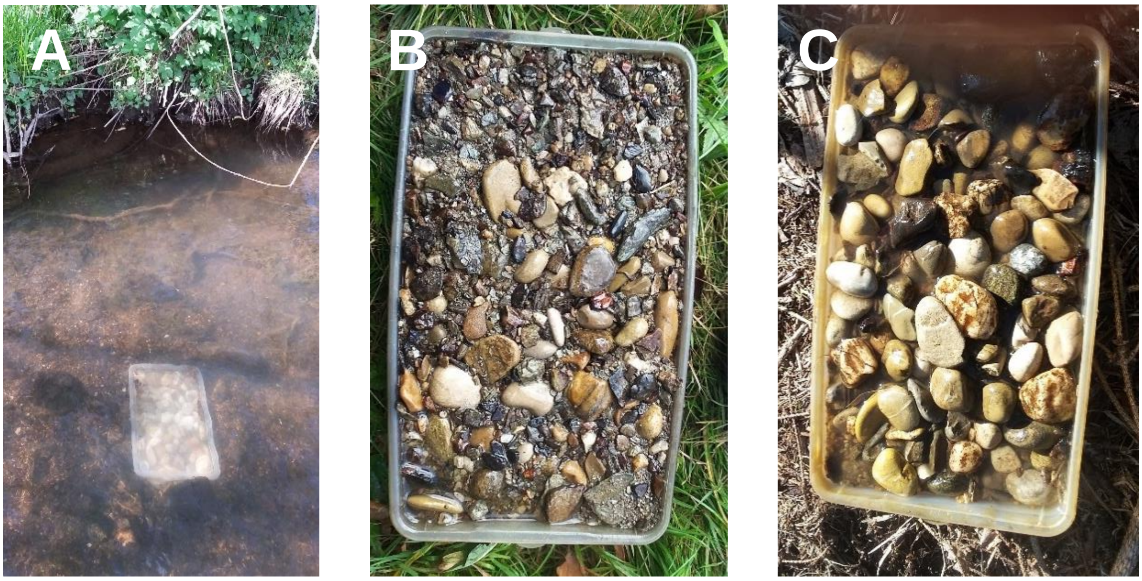

Figure 4: Photographs of sediment traps, A) inactive buried within the stream bed, B) after a high discharge interval, C) after a low discharge interval.

In addition, hydromorphological parameters were recorded 10 cm upstream of the sediment trap at the beginning and the end of each interval. For this purpose, the flow velocity at the surface (vo, m/s), as well as 2 cm above the bottom of the water body (vu, m/s) were measured at the right and left corner, respectively, as well as in the center of the sediment trap with a Flowtherm NT (Höntzsch, Waiblingen, Germany). The wetted width as well as the depth at the three locations was recorded with a measuring stick (± 1 mm).

2.3.3 Water and substrate quality

Abiotic measurements of the water quality were made at each sampling site in the free wave and at 5 cm depth in the substrate. O_2 concentration (O_2, mg/l), temperature (T, °C), pH, and conductivity (Lf, µS/cm, referenced to a T of 20 °C) were measured once in free water and from three samples of interstitial water using a Multi-3430 G portable meter (WTW, Weilheim, Germany). Interstitial water was sampled from 5 cm depth 10 cm above the sediment trap at three points along the stream cross-section: right, middle, and left, according to Pander et al., (2015). Redox potential (Eh, mV) in the free wave and in the interstitial was measured *in-situ* according to Geist & Auerswald, (2007) using a pH 3110 meter (WTW, Weilheim, Germany) and a platinum electrode against an $Ag/AgCl_2$ reference electrode. Substrate quality was assessed as the difference between values in the free wave and the interstitial as suggested by the CEN Standard on FPM monitoring (Boon et al. 2019). Stream bed penetration resistance was recorded using a penetrometer (Ejikelkamp Agrisearch Equipment, Giesbeek, The Netherlands) at five randomly distributed points.

Concentrations (mg/l) of NO_3-N, ortho PO_4-P and NH_4-N were obtained from water samples filtered through filters with a pore size of 22 µm and then analyzed via ionchromatography in two ICS 1100 ion chromatographs (Thermo Fisher Scientific, Dreieich, Germany).

2.4 Investigating the thermal regime

Water temperature (Tw, °C) was measured hourly using temperature loggers employed at each sampling point from June 2018 to September 2020. Since the effect of Tw on aquatic ecosystem is highly variable, looking at only single (mean) values at one specific site is not sufficient, thus the thermal regime should be monitored over different temporal and spatial scales. The thermal regime of a stream or river can be described in terms of magnitude, variability, frequency, duration and timing of thermal events (Arismendi et al. 2013, Figure 5).

An EL-USB-1 temperature logger (Lascar Electronics, Salisbury, UK) was installed at each sampling point and recorded the temperature (± 0.5 °C) in the open water every 60 min. Loggers were collected every three months, the data downloaded, and deployed again one week later. The data sets were pooled to establish one nearly continuous record for each monitoring point. The data sets were then used to calculated several indices to describe different aspects the temperature regime important for the target species FPM using an updated version of the StreamThermal R Package (Tsang et al. 2016). They were then used to classify ponds/sites according to their thermal regime and examined for general (seasonal) trends, differences between sites upstream and downstream of the outlet channel inflow, and differences between sites along the stream course.

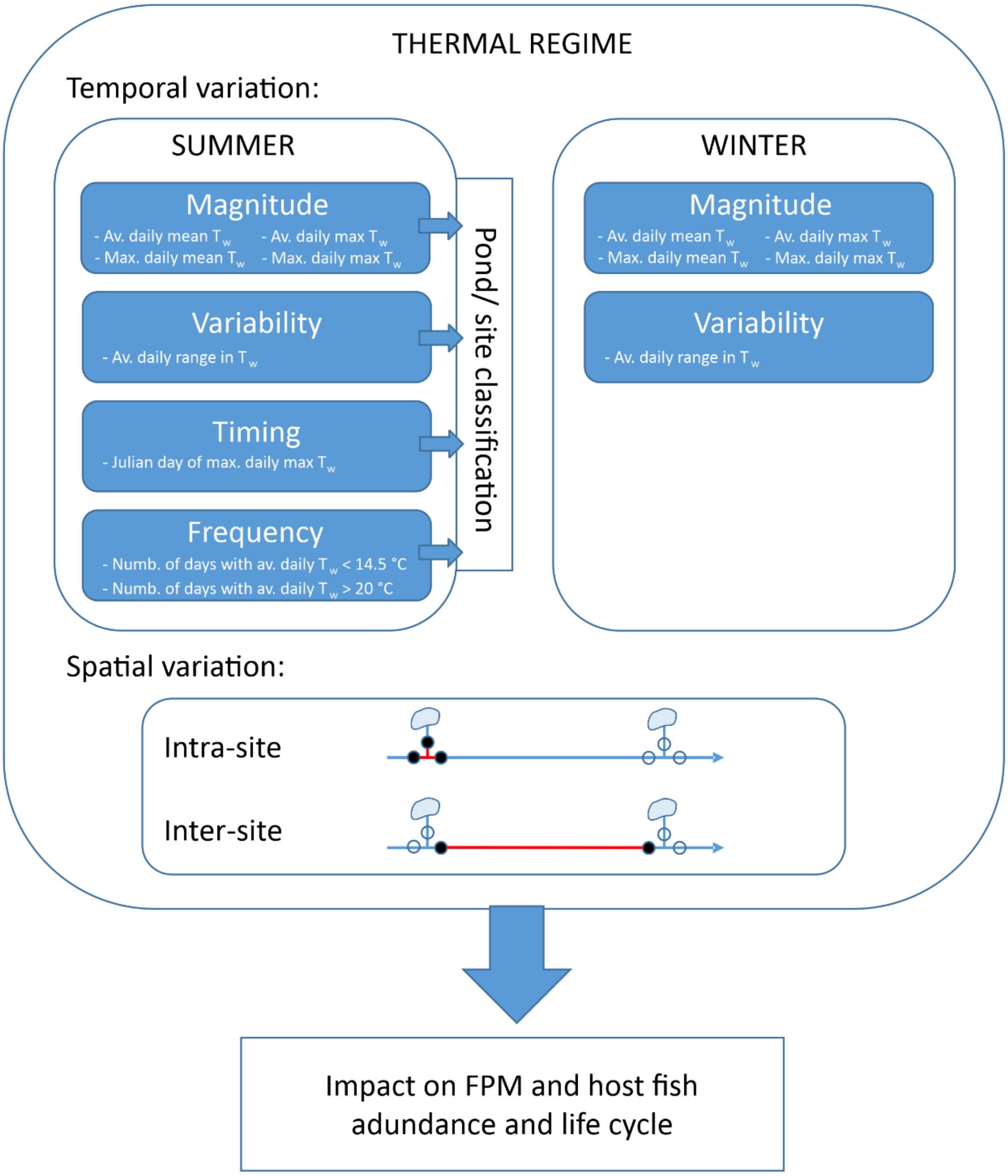

Figure 5: Assessment of the thermal regime.

2.5 Monitoring of suspended solids and nutrient concentrations during pond drainage

Pond drainage for fish harvest is the major event during the pond management cycle. Water is slowly released from the pond in multiple steps, which usually take several days, so that the fish can follow the decreasing water level and gather close to the outlet. Drainage is controlled using either a simple stand pipe composed of multiple pieces that are removed one after another, or through a traditional structure called ‘monk’. This is a stone or concrete tower with several wooden boards to block the water flow until a certain water level is reached in the pond. Depending on which wooden board is removed, water can be drained from the bottom or the surface of the pond. With decreasing water level, the risk of pond bottom material to be remobilized either by the water drag or the activity of fish increases. The maximum remobilization is expected to occur during the final fish harvest and strongly depends on the harvesting method: Fishing “in front of the monk”, meaning fishermen entering the pond and catching the fish in front of the outlet is supposed to cause higher sediment outputs than fishing “behind the monk”, where fish are flushed through the outlet pipe into a concrete fishing pit, where they can be easily collected. Another technique would be to harvest the fish using a seine net, either with the outlet open and still draining water to facilitate the process, or closed during the seining (Lin et al. 2001). The overall drainage management as well as the fish harvest methods are likely to cause different degrees of fine sediment outputs into the adjacent stream.

Therefore, a high number of pond drainage events were sampled as they were conducted by the pond owners, without intervening into their management practice. This allowed for sampling realistic conditions. Several pond owners took measures to prevent the output of suspended particles, while other did not. The measures applied to mitigate outputs, so-called suspended-solid-mitigation (SSM, Figure 6) measures could be group in three categories: Filtering, physical settling structures and seining with closed outlet.

Straw bales were used to filter effluent water during pond drainage by placing them behind the pond outlet. Physical settling structures included settling ponds the water was drained into before reaching the receiving stream, and long, shallow, overgrown outlet channels. A third method, the so-called “settling traps” were constructed by the water authorities board by widening the outlet channels to create three pool-like structures with decreased flow velocity to enhance particle settling, not only during fish harvest but also high discharge events. When fish were harvested with a seine net while the outlet was closed, a larger than usual proportion of water was retained in the pond before the net was pulled through.

Besides a monitoring of the sediment deposition resulting from pond drainage following the sampling design described above, turbidity (TURB, NTU), total and organic suspended solids (total suspended solids = TSS, organic suspended solids = oTSS, mg/l), ammonium as NH_4-N, nitrate as NO_3-N and orthophosphate as ortho PO_4-P at the ‘ds’ site were investigated during the pond drainage process. Turbidity was measured directly from three replicates of stream water, while the other parameters were taken from water samples that were stored frozen until further processing. Measurements were taken every half hour or when a change in turbidity could be observed visually. Samples for suspended solids were filtered using vacuum-filtration through pre-weight, ash-free filter papers with a retention capacity > 2 µm, dried and weight and related to the initial sample volume. The filters were afterwards combusted at 500 °C to determine the proportion of organic carbon as loss by ignition. Turbidity and TSS values taken at the same sites and time were used to establish the relationship between turbidity and TSS. Afterwards, turbidity measures taken from water samples using a TURB 355 T (WTW, Weilheim, Germany) or by an MPS-D8 automatic logger (SEBA Hydrometrie GmbH & Co. KG, Kaufbeuren, Germany) could be used as a proxy for the TSS concentration. Nutrient concentrations were analyzed by ion chromatography, as described above.

Figure 6: Suspended solids mitigation (SSM) measures sampled in the study : A) none; B) filtering through a bale of straw; C) overgrown settling channel; D) settling pond; E) seining with closed outlet; F) settling trap (picture source F: https://www.wwa-ho.bayern.de/fluesse_seen/massnahmen/perlmuschel/index.htm, accessed on the 04.10.2021).

3. Impact of fish ponds on hydrology and temperature regime of a small catchment containing the endangered freshwater pearl mussel

A similar version of this chapter was published: Hoess R., Generali K. A, Kuhn J. & J. Geist, 2022. Impact of Fish Ponds on Stream Hydrology and Temperature Regime in the Context of Freshwater Pearl Mussel Conservation. *Water*. 14: 2490. DOI: 10.3390/w14162490

Authors contributions: R.H. conceptualized the study and was responsible for project administration together with J.G.; R.H. developed the methodology in cooperation with K.A.G. and J.G. and was responsible for the software and formal analysis together with K.A.G.; R.H. did the main investigation, data curation and wrote the original draft; R.H. developed the visualization together with J.K. and J.G. For further information on the authors contributions please refer to Chapter 9. All authors have read and agreed to the published version of the manuscript.

3.1 Abstract

Conservation of endangered, cold-stenothermic species, such as the freshwater pearl mussel (FPM) and its salmonid host fish, are particularly challenging in headwater streams as their last refuge areas. Understanding the impact of anthropogenic catchment features such as fish ponds on the hydrology and the temperature regime of such streams is, therefore, important. In this study, runoff in a FPM catchment with more than 150 small ponds was simulated using SWAT and compared to a scenario without ponds. Additionally, water temperature was monitored hourly along three streams over 2.5 years, at sites upstream and downstream of the inflow of pond outlet channels. Temperature metrics were related to land use within a 180 m corridor along the streams. Peak flows were reduced by 1.5% with ponds, while low flows were increased by 4.5%. In summer, temperature in pond effluents was higher than in the receiving stream, depending on the proximity of the inflow points. Discharge from close-by ponds increased summer stream temperature directly downstream of the inflow by up to 5.5 °C. These increased temperatures were partly compensated by groundwater contribution in forested areas. In contrast, stream temperature significantly further increased along stretches flowing through open land, persisting independently of pond inflows. We suggest incorporating this knowledge on pond- and land use-dependent effects on stream temperature regimes into the conservation management of FPM and other cold-stenothermic species, as well as into climate change mitigation strategies targeting an increased resilience against temperature extremes.

3.2 Introduction

Increasing global air temperature due to anthropogenic climate change has affected freshwater ecosystems worldwide (van Vliet et al. 2013), making the management of flow and temperature regimes—and particularly of cold-water spots in streams and rivers during summer—a key priority for the sustainable management of cold-stenothermic species (Arismendi et al. 2012; Casas-Mulet et al. 2020; Ebersole et al. 2003; Ebersole et al. 2015; Kuhn et al. 2021). An intensification of the hydrological cycle can have pronounced effects on waterbodies, causing chances in regional patterns of evapotranspiration and precipitation, increasing the probability of extreme events such as floods and droughts (IPCC 2022). In particular, low flows can have adverse effects on stream biodiversity (Lake 2003), by reducing water and habitat quantity (Capon et al. 2021) which might be mitigated by increased water retention at the landscape scale (Auerswald et al. 2019b; Doriean et al. 2019; Geist and Auerswald 2019). Small waterbodies such as ponds and wetlands influence evaporation, groundwater recharge and flood retention (Acreman and Holden 2013; Khilchevskyi et al. 2020), offering an option for drought mitigation (Ameli and Creed 2019; Cai et al. 2015). On the other hand, they may contribute to the warming of receiving streams, potentially counteracting their effects of buffering low flows. To help incorporating the multiple, interacting effects of ponds and other catchment features such as land use (LU), hydrological modeling at the catchment scale can help evaluating effects and assist with the planning of pond construction. Modeling tools such as the Soil and Water Assessment Tool (SWAT) can be used to model whole catchments and then compare water or sediment yield between certain LU or climate scenarios (Akbas et al. 2020; Al Sayah et al. 2019; Makhtoumi et al. 2020; Neupane et al. 2018). Nonetheless, since hydrological modeling needs to generalize many parameters at a broad scale to ensure efficient computing, it can be challenging to analyze the impact of individual ponds through this catchment-scale approach. Field measurements on specific ponds of interest might be needed to ensure an accurate representation of pond effects at the local scale. Water temperature (Tw) measurements are ideally suited for this purpose, simultaneously providing additional information on ecological impacts of pond effluents, as hydrology and temperature regimes are coupled. Temperature surveys using automatic temperature loggers have become an easy and cost-efficient measure to obtain data at a high spatial and temporal resolution (Webb and Nobilis 2007). Numerous studies used this approach to investigate groundwater contribution to streamflow (e.g., Rau et al. (2010)), the impact of heated effluent discharge (Coulter et al. 2014), or the impact of impoundments and ponds (Seyedhashemi et al. 2021) at the reach scale. Moreover, the thermal regime of streams is, by itself, an important parameter shaping aquatic communities. Climate change might impact the hydrologic regime, with increasing periods of low flow conditions resulting in reduction or complete loss of habitat for cold-stenothermic species such as salmonids (Kuhn et al. 2021). Furthermore, rising air temperatures directly cause increased warming of surface water bodies (van Vliet et al. 2011) and changes in stream thermal regimes (Steel et al. 2017). Increasing Tw has direct impacts on stream organisms by lowering oxygen solubility (Piatka et al. 2021) and thus the physicochemical habitat quality as well as by enhancing the metabolic rate of poikilothermic species. It can also shift competitive advantages of invasive versus native species (e.g., Pander et al. (2022)). Tw therefore plays a key role in structuring aquatic communities and in determining species distribution (Bond et al. 2011; Capon et al. 2021; Davis et al. 2015; Warren et al. 2012; Wehrly et al. 2003). However, it is difficult to interpret in an ecological context as single or mean values might be insufficient to determine ecological impacts. The thermal regime accounts for diel, daily, seasonal and annual variation as well as spatial patterns along the stream course (Caissie 2006; Maheu et al. 2016). It might not only be relevant if a certain temperature threshold is exceeded, but also if the threshold is exceeded during a critical time in a target species life cycle (Souchon and Tissot 2012). Predicted increases in global Tw vary between 1.0 and 4.0 °C by 2050, depending on climate change scenario (Collins et al. 2013; Marteau et al. 2022; Punzet et al. 2012). Prolonged low flow periods additionally enhance stream warming by decreasing thermal capacity (van Vliet et al. 2013). This is mainly a threat to cool-stenothermal organisms adapted to cool thermal regimes such as the highly endangered freshwater pearl mussel *Margaritifera margaritifera* (FPM). This species and its fish host require summer Tw < 20 °C in most of their Central European and Scandinavian range (Boon et al. 2019). Higher Tw are associated with low numbers of gravid females in non-recruiting populations (Österling 2015). Moreover, successful reproduction of this species strongly depends on the presence of its salmonid host fish, the brown trout *Salmo trutta* or the Atlantic salmon *Salmo salar*, in close proximity to mussel beds during the spawning period, since the FPM life cycle includes a parasitic phase of its glochidia larvae on the host fish's gills (Geist 2010). Salmonids such as the brown trout are adapted to cold water temperatures with a temperature optimum between 12–18 °C (Jobling 1981) and with the upper critical temperature limit between 22–25 °C (Elliott and Elliott 2010). They were shown to avoid reaches with summer Tw over 21 °C and search for thermal refugia with cooler Tw (Alabaster and Downing 1966; Hitt et al. 2017). In Central Europe, the FPM usually release their glochidia between June and September, when Tw are at their maximum (Hastie and Young 2003). If the highly sensitive brown trout avoids reaches populated by FPM due to Tw above its thermal tolerance, infestation with glochidia cannot take place and natural reproduction ceases. Moreover, the survival and attachment ability of mussel glochidia also strongly depends on ambient Tw (Benedict and Geist 2021) and it has been shown that Tw is crucial for the performance and timing of excystment of metamorphosed juvenile mussels from their fish hosts (Taeubert et al. 2014; Taeubert et al. 2013). Insufficient levels of natural reproduction are the major cause of the strong declines in the last remaining FPM populations within the species Central European distribution (Geist 2010). However, Tw that are too cold also impair or delay reproduction, with a sufficient number of days with Tw > 10-15 °C needed for maturation (Hastie and Young 2003) and sufficient growth of juveniles (Buddensiek 1995; Gum et al. 2011). The European drought during the summers of 2018 and 2019 (Moravec et al. 2021) showed the urgent need to find ways to increase the resilience of headwater streams to drought. In particular, in north-eastern Germany as well as in north-east Bavaria low soil moisture levels and stream water levels threatened the last remaining FPM populations in the area, which represent some of the largest remaining in Central

Europe. To prevent reaches inhabited by the FPM from falling dry, water was discharged from nearby ponds and deep wells that are usually used for drinking water abstraction. This development is critical because the small headwater streams concerned are the last refuges of cold-water species in this area as they are dominated by groundwater inflow during summer (Kaule and Gilfedder 2021). The presence of more than 150 ponds in the catchment of the FPM streams raised the question to what extent they have an impact of the runoff and Tw of these valuable ecosystems. While most of the attention concerning the effect of heated effluents is focused on the impact of power plants (Raptis et al. 2016; Sylvester 1972) and damming (Ahmad et al. 2021; Sinokrot et al. 1995; Zaidel et al. 2021), the impact of pond effluents on water temperature of adjacent streams has received little attention, despite the high number of ponds all over the globe (Downing 2010). Various impacts of ponds on hydrological and temperature regime are suggested, depending on multiple parameters as well as on other LU features, but there is still limited understanding of LU-pond-stream interactions (Ebel and Lowe 2013). In this study, the impacts of multiple ponds within a small catchment with FPM were assessed through a catchment-wide hydrological modeling using SWAT as well as a field survey of Tw at multiple sites and compared with other LU features within the catchment. To our knowledge, this is the first study combining an evaluation of the impact of ponds on the hydrological and temperature regime at the catchment as well as the local scale with a focus on the critically endangered FPM. The following hypothesis were tested:

i) Ponds have a significant influence on the hydrological cycle at the catchment scale, increasing flood retention during high flows and buffering low water levels during low flow conditions.

ii) Ponds have an impact on the temperature regime in small, cool, headwater streams, with significant increase in stream temperature through ponds effluents in summer and nearly neutral effects during winter.

iii) Effects on hydrological and temperature regime accumulate with increasing number of ponds draining into a stream.

3.3 Material and Methods

3.3.1 Study area

The catchment of the “Schwesnitz” river, a tributary of the Sächsische Saale of the Elbe stream was chosen for this study as it contains more than 150 individual ponds, as well as populations of the endangered FPM, making it a priority area of conservation. It is formed by the two major tributaries “Perlenbach” coming from the south and “Höllbach” coming from the Czech Republic in the east (Figure 7). The mean annual precipitation sum and temperature at the meteorological station “Hof” (ID 2261) are 757.6 mm and 7.1 °C, respectively. The long-term mean annual discharge at the gauging station

"Schwesnitz" (50.245433°N, 12.019682°E) is 0.67 m^3/s. The discharge regime is dominated by higher flows in winter and in particular after the snow melt, and lower flows during summer and autumn, when groundwater dominates the baseflow (Kaule and Gilfedder 2021). The total catchment area of 90.82 km^2 is dominated by coniferous forest, primarily spruce (50.4%), followed by agricultural LU (22.6%) and pastures (18.8%) according to the satellite-derived CORINE 2012 Land Cover map (EU 2012). 150 earthen ponds are located within the study catchment, with an average size of 1384.6 m^2 (range: 39.5-46,101.4 m^2). They are mostly used for extensive carp production, meaning they remain filled year-round except for a period of about one week, when water is drained for fish harvest (Hoess and Geist 2021). Some of the ponds are managed as flow-through ponds, with a constant small amount of water discharged to the receiving streams, while in some ponds, water is only supplied to compensate for evaporation losses and the outflow is closed during low discharge conditions in summer.

3.3.2 Hydrological Model Setup

To model the hydrological regimes of the catchment, the Soil and Water Assessment Tool (SWAT, (Neitsch et al. 2011)) was used through geoinformatic software ArcMap 10.5.0 (Esri, Redlands, CA, USA) using the software extension ArcSWAT (Arnold et al. 2012). The SWAT model is a physically based model, relying on a digital elevation model (DEM) to delineate the stream network and the associated catchment, which is then divided into smaller subbasins. The model further incorporates LU and soil maps, as well as slope and climate data. Each subbasin is further divided into so-called hydrological response units (HRUs), each consisting of a unique combination of LU, soil and slope categories that are used to model hydrological processes for each HRU. The surface runoff generated in each HRU is then routed through the stream network of the different subbasins until it reaches the catchment outlet. The model was set up using the following input data: The DEM used in this study was merged from two sources, a 1 x 1 m DEM covering the German part of the catchment provided by the water authorities board (WWA) in Hof and a 5 x 5 m DEM covering the Czech part of the catchment provided by the Czech Office for Surveying, Mapping and Cadaster. Both raster data sets were merged into one data set with a 5 x 5 m resolution. The complete DEM was used to delineate the stream network and subbasins based on a 50-ha threshold to achieve the maximum possible channel resolution. Four slope categories were derived based on the DEM. LU was obtained from the CORINE Land Cover CLC 2012 data set (EU 2012, Figure 7) and converted into the SWAT LU classification. The soil map was obtained by merging the soil map BÜK25 provided by the Bavarian Environmental Agency for the German part and the soil map CR50 provided by the Czech geological survey. Hydrological soil groups for the German part were provided by the WWA Hof. German and Czech soil classes were harmonized based on particle size. Based on these data sets, an initial SWAT model was set up for the 90.82 km^2 Schwesnitz catchment, comprised of 87 subbasins and 1889 HRUs, based on a threshold of 3% for each LU, soil and slope category. Daily precipitation data for the period between 2008–2021 was obtained from four meteorological stations of the German National Meteorological Service within or close to the study catchment (Station “Hof”, ID: 2231; Station “Selb”, ID: 4548; Station “Rehau”, ID: 4109; Station “Regnitzlosau”, ID: 4107; Figure 7). Data on daily minimum and maximum air temperature and relative humidity was only available for the stations at “Hof” and “Selb” and daily wind speed data was only available for the “Hof” station. Solar radiation data was simulated within the SWAT weather generator, since no small-scale observation data was available for this parameter. Therefore, the Hargraves method was chosen for calculating potential evapotranspiration (PET), which is only based on daily minimum and maximum temperature (Neitsch et al. 2011). The initial model was run with a warm-up period of two years. SWAT allows the integration of one pond or wetland per subbasin (Jalowska and Yuan 2019; Neitsch et al. 2011). Spatial data on the location and surface area of the ponds found within the catchment was obtained from orthophotos and field observations. When multiple ponds occurred within one subbasin, they were combined to form a hydrological equivalent wetland (HEW, Wang et al. (2008)). In total, 68 of the subbasins contained at least one pond. The pond parameter PND_FR, quantifying the proportion of the subbasin area draining into the pond, was obtained from the DEM using the Flow Path Tracing tool of ArcHydro. The other parameters describing the ponds were derived from the pond map, field observations or literature and are listed in Table 2.

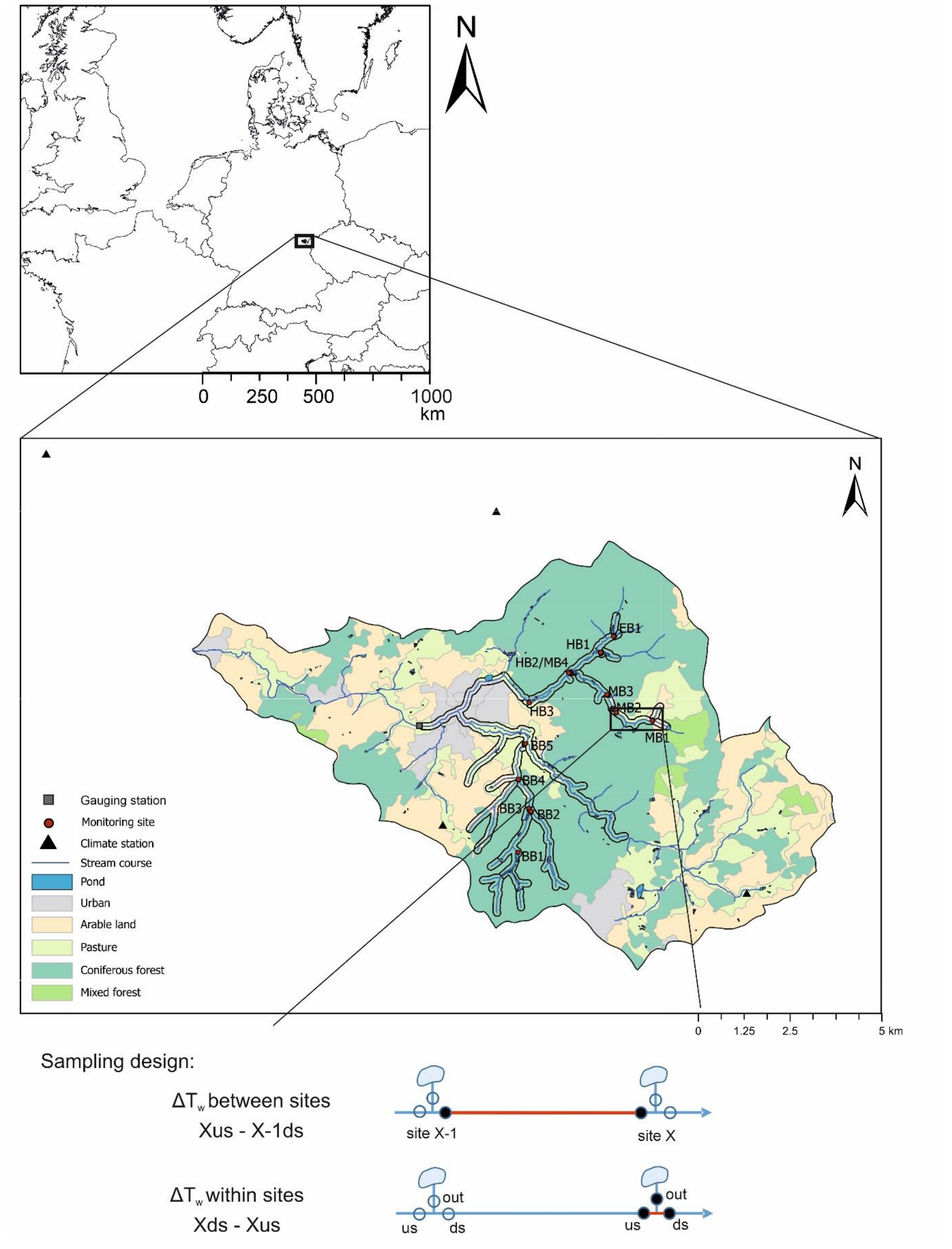

Figure 7: Map of the study catchment including stream courses as delineated in SWAT, ponds, temperature monitoring sites, climate stations, and different LU categories obtained from CORINE Land Cover Classification ; the black line around the study streams represents the 180 m corridor around the streams that was used to evaluate the effect of LU on stream temperature; sampling design with the position of temperature loggers within sites and delta Tw (ΔTw) calculated between and within sampling sites is shown below.

Table 2: Initial input variables used to integrate ponds into the SWAT model: PND-parameters, definition and source.

Parameter	Definition	Source
PND_FR	Fraction of subbasin area that drains into ponds (0–1)	DEM
PND_PSA	Surface area of ponds when filled to principal spillway [ha]	Shape file based on orthophotos
PND_PVOL	Volume of water stored in ponds when filled to the principal spillway [10^4 m³ H_2O]	Surface area x depth (=0.8 m, mean depth derived from field observations)
PND_ESA	Surface area of ponds when filled to emergency spillway [ha]	PSA × 1.1 (derived from field observations)
PND_EVOL	Volume of water stored in ponds when filled to the emergency spillway [10^4 m³ H_2O]	PSA x depth (=1.0 m, derived from field observations)
PND_VOL	Initial volume of water in ponds [10^4 m³ H_2O]	=PVOL
PND_SED	Initial sediment concentration in pond water [mg/L]	39 (derived from water samples from fish-free pond)
PND_NSED	Equilibrium sediment concentration in pond water [mg/L]	96 (derived from water samples from stocked pond)
PND_K	Hydraulic conductivity through the bottom of ponds (mm/h)	1 (after Baldan et al. (2021b))
IFLOD1	Beginning month of non-flood season	0
IFLOD 2	Ending month of non-flood season	0
NDTARG	Number of days needed to reach target storage from current pond storage [d]	5 (derived from field observations)
PND_D50	Median particle diameter of sediment [µm]	10 (default)

3.3.3 Calibration and validation

The calibration and uncertainty analysis software SWAT CUP (V 5.2.1.1, 2w2e GmbH, 2019) was used to calibrate the initial models' parameters to accurately simulate the runoff values at subbasin 32 compared to those observed at the gauging station "Schwesnitz" (provided by the Bavarian Waterscience Service, available at www.gkd.bayern.de (accessed on 12 April 2022)). A sensitivity analysis was performed prior to the calibration to identify the relevant parameters. These were included into the calibration process, together with several parameters chosen after a literature review. For the pond parameters, only PND_K showed a significant effect and was included into the calibration. The parameters used in the calibration process can be found in Table 3. Five iterations with 300 simulations each were performed using the SUFI-2 algorithm on daily runoff data for the period from 01/01/2015 to 12/31/2021. The results were validated for the period from 01/01/2010 to 12/31/2014. To evaluate the effect of the ponds on the hydrologic regime, daily runoff at subbasin 32 was compared between the existing scenario with ponds (used for calibration) and a scenario without ponds, where the calibrated parameters were used in the simulation but PND_FR was set zero in all subbasins, which turns off the pond function (Jalowska and Yuan 2019). To differentiate between effects at high, peak, and low flows, runoff conditions were classified using long-term statistical thresholds (see also Hoess and Geist (2020)), as well as the maximum flow for each high discharge event.

Table 3: Parameters used in the calibration with SWAT CUP, their initial calibration range and the value fitted after calibration, small letter before the parameters represents the mode of adapting the parameter during the calibration (r = relative, v = replace, a = add).

Parameter	Definition	Initial calibration Range	Fitted value
r__CN2.mgt	Initial SCS runoff curve number for soil moisture condition II	−0.5–0.5	−0.302425
v__ESCO.hru	Soil evaporation compensation factor of HRU	0.0–0.5	−0.207090
r_SOL_AWC(#).sol	Available water capacity of the soil layer (#) (mm H_2O/mm soil)	−1.0–0.5	−0.404389
r__SOL_BD(#).sol	Moist bulk density of the soil layer (#) (mg/m³)	0.0–0.7	0.640855
a__CANMX.hru	Maximum canopy storage (mm H_2O)	80–180	82.587418
r__SOL_K(#).sol	Saturated hydraulic conductivity of the soil layer (#) (mm/h)	−0.2–0.8	0.755521
a__GW_REVAP.gw	Groundwater ”revap” coefficient	0.00–0.18	0.061701
a__GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur (mm H_2O)	−1000–2000	−500.878265
a__REVAPMN.gw	Threshold depth of water in the shallow aquifer required for ”revap” or percolation to the deep aquifer to occur (mm H_2O)	−750–0	−373.519958
r__SLSUBBSN.hru	Average slope length (m)	−0.5–1.0	0.580487
a__GW_DELAY.gw	Groundwater delay time (days)	100–350	41.244041
a__OV_N.hru	Manning’s „n“ for overland flow	0–100	47.643097
v__PND_K.pnd	Hydraulic conductivity through bottom of ponds (mm/h)	0–1	0.602785
v__ALPHA_BF.gw	Baseflow alpha factor (1/days)	0–1	0.799400
v__RCHRG_DP.gw	Deep aquifer percolation fraction	0.0–0.4	0.231797

3.3.4 Temperature measurements

To obtain Tw for assessment of the temperature regime upstream and downstream of the inflow of pond outlet channels, a total of 31 temperature loggers (EL-USB-1, Lascar Electronics, Salisbury, UK) with accuracy of Tw ± 0.5° C and logging interval of 60 min were employed between spring 2018 and autumn 2020. At twelve monitoring sites (Figure 7), loggers were installed upstream (‘us’) and downstream (‘ds’) of the inflow of several pond outlet channels, following the study design described by (Hoess and Geist 2021). Where possible, an additional logger was installed directly within the pond outlet channel. Five sampling sites were established in the BB, one tributary of the “Perlenbach”. The upper part of the stream is dominated by a big fish farm consisting of more than 25 ponds. Along its course through a mostly forested area, multiple pond facilities drain into the stream. Towards the confluence with the “Perlenbach”, the catchment LU around the BB changes to open, agriculturally used lands. In contrast, the MB, a tributary of the “Höllbach”, originates from an open wetland area, then flows through coniferous forest. Multiple ponds drain into the MB along its course, where three sampling sites were established until the confluence with the “Höllbach”. Along the course of the “Höllbach” (HB), four sampling sites were established, including one at a tributary (EB) and one the confluence with the MB. At EB1, the main stream received inflow from two pond facilities from two different side channels. Furthermore, the monitoring site EB1 is close to a deep well, from which groundwater was discharged during summer 2018 and 2019 to support streamflow. Moreover, due to the drought conditions water discharge from several ponds was decreased to sustain conditions needed for fish farming, causing the complete cessation of discharge from the pond MB1.

3.3.5 Data analysis

The hourly measured Tw values were compared between the monitoring points at each site. Furthermore, delta Tw (ΔTw) was calculated between subsequent temperature loggers ($X – 1 < X < X + 1$) by subtracting the value measured by the logger at the position upstream (X - 1) from the value measured at position T at a given date and time once within sites (site X ds—site X us) as well as between sites (site X us—site X- 1 ds; Figure 7). The hourly measurements of Tw were also summarized as daily mean, minimum and maximum data and then used to calculate 12 metrics describing the general temperature regime and aspects of interest within the context of FPM at the respective monitoring points using an adapted version of the StreamThermal package in R (Table 4). Delta values of these metrics between subsequent monitoring points were calculated as described above. Pearson's correlation test was used to analyze the relationship between temperature metrics and pond or LU characteristics, assuming a linear relationship. Multivariate analysis of the summer thermal regime was conducted using an agglomerative hierarchical clustering analysis of monitoring points based on the standardized/normalized summer metrics. The cluster analysis was performed in PRIMER 7 Version 7.0.17 (Plymouth Marine Laboratory, Plymouth, UK) on Euclidean distance using the Group Average joining algorithm following Rivers-Moore et al. (2013).

To assess the potential influence of LU on Tw, the proportion of LU types were calculated within a 180 m corridor around the streams using ArcMap 10.5.0. (Esri, Redland, CA, USA, Figure 7) using pond spatial data and CORINE data (summarized into two categories: forested area: coniferous and mixed-forest, open land: pasture and arable land). Distance between the pond outlets and the 'out' monitoring sites was calculated and grouped into three categories: 'close': pond < 50 m from the monitoring site; 'medium far': pond 400–600 m from the monitoring site and 'far': pond > 1000 m from the monitoring site. A multiple linear regression model including the proportion of pond and forested area (the proportion of open land was excluded due to high collinearity with the proportion of forested area), the reach length and the interaction between all three variables was run to evaluate the impact of the variables on the difference in average daily mean temperature in summer (ΔADM_su) between two subsequent monitoring points. Model performance was evaluated by standard graphical validation (Zuur et al. 2009) and the 'DHARMa' diagnostic package. A significance level of alpha < 0.05 was set for all statistical analysis. For analyzing patterns of Tw changes along the stream course, a complete set of

ΔTw for all monitoring points within one stream was prepared. Periods for the three streams were chosen to maximize the time span of all loggers within one stream to deliver continuous data. This resulted in the study period for BB being from the 6/11/2019–9/1/2020, for MB being from 06/18/2018–05/21/2019 and for HB being from 11/24/2018–08/20/2019.

Table 4: Temperature metrics used to describe the thermal regime at the monitoring points, and their ecological relevance.

	Metrics	Relevance
Magnitude		
ADM_su	Average daily mean T_w in summer	Physiological response, development/growth rates, concept of degree-days
ADM_wi	Average daily mean T_w in winter	
MaxD_su	Maximum daily mean T_w in summer	Potential thermal limit for aquatic organisms
MaxD_wi	Maximum daily mean T_w in winter	
AMax_su	Average daily maximum T_w in summer	
AMax_wi	Average daily maximum T_w in winter	
MaxT	Maximum daily maximum T_w in summer	
Variability		
Range_su	Average daily range in T_w in summer	Dial variation
Range_wi	Average daily range in T_w in winter	
Timing		
Jdmax	Julian day of MaxT in summer	Possible shift in timing of life history transitions
Frequency		
b14_5	Number of days in summer with average daily T_w < 14.5 °C	T_w > 14.5 °C needed to achieve sufficient growth in FPM
a20	Number of days in summer with maximum daily T_w > 20 °C	Host fish (brown trout) will migrate from stream reach at T_w > 21 °C

Radon was used as a tracer for groundwater discharge in the study catchment between 2019 and 2020 (Kaule and Gilfedder 2021). Since groundwater discharge has a considerable influence on Tw (Bloomfield et al. 2021; Chu et al. 2008; Stanfield et al. 2009), in particular during low flows, the output of the SWAT model was used to calculate the baseflow index (BFI) in 2018, 2019 and 2020 for subbasins representing the stretches between monitoring sites, where applicable. BFI was calculated using the lfstat package in R, as recommended by the WMO (2008). Since the BFI for the subbasins representing the stretches along the MB that matched the stretches in the radon study (Kaule and Gilfedder 2021) showed a similar pattern as the radon-based groundwater discharge, the BFI was used as a proxy for groundwater discharge in all other subbasins of the present study (Table S 1). Differences in BFI between stream sections were assessed in R Version 4.1.0 (www.r-project.org, 2021) using Kruskal-Wallis with post-hoc Mann-Whitney U-test with Bonferroni correction, as the data were not normally distributed and variances inhomogeneous.

3.4 Results

3.4.1 Hydrologic regime

The model evaluation parameters for the calibrated model are given in Table 5. The calibrated model with ponds was classified as “good” or “very good” for five of six widely used evaluation criteria (Moriasi et al. 2015) for daily stream flow, with NSE of 0.71 and 0.77 for the calibration and validation period, respectively. For the validation period, only PBAIS of 16.2 was “not satisfactory” due to an overestimation of low flows (Figure 8). Since the field campaign took place during the calibration period, when all three parameters classified the model as “good” or “very good”, the stream flow modeled for the subbasins of interest were considered valid for further analysis.

Table 5: Calibration output; classification of evaluation parameters after Moriasi et al. (2015) for daily stream flow data at subbasin 32.

	NSE	PBIAS	R^2
Calibration 2015/01/01–2021/12/31	0.71 “good”	3.3 “very good”	0.71 “good”
Validation 2010/01/01–2014/12/31	0.77 “good”	16.2 “not satisfactory”	0.79 “good”

Within the 68 subbasins containing a pond, the average proportion of subbasin area draining into a pond (PND_FR) was 41.4%, representing 26.3% of the total catchment area. Mean HEW size was 0.094 ha and varied between 0.059–0.151 ha. When the ponds were included, the model simulated stream flow that was on average 3.2% higher than in the scenario without ponds. Separating between high and low flow conditions (based on the long-term statistics of the gauging station) revealed a more differentiated pattern: Over all higher flows, the stream flow with ponds was 0.2% lower than without ponds while the peak flows were decreased by 1.5% with ponds. In contrast, stream flow under low flow conditions was on average 4.5% higher with ponds than without them. The ponds therefore moderately increased low flows while the buffering effect during the peak flows was relatively low. Concerning the timing of floods, no differences could be observed between the two models. In the scenario without ponds, 16 days were additionally classified as days with high flow conditions over the whole simulation period, which represented a shift from 20.3 to 21.0% compared to the scenario with ponds.

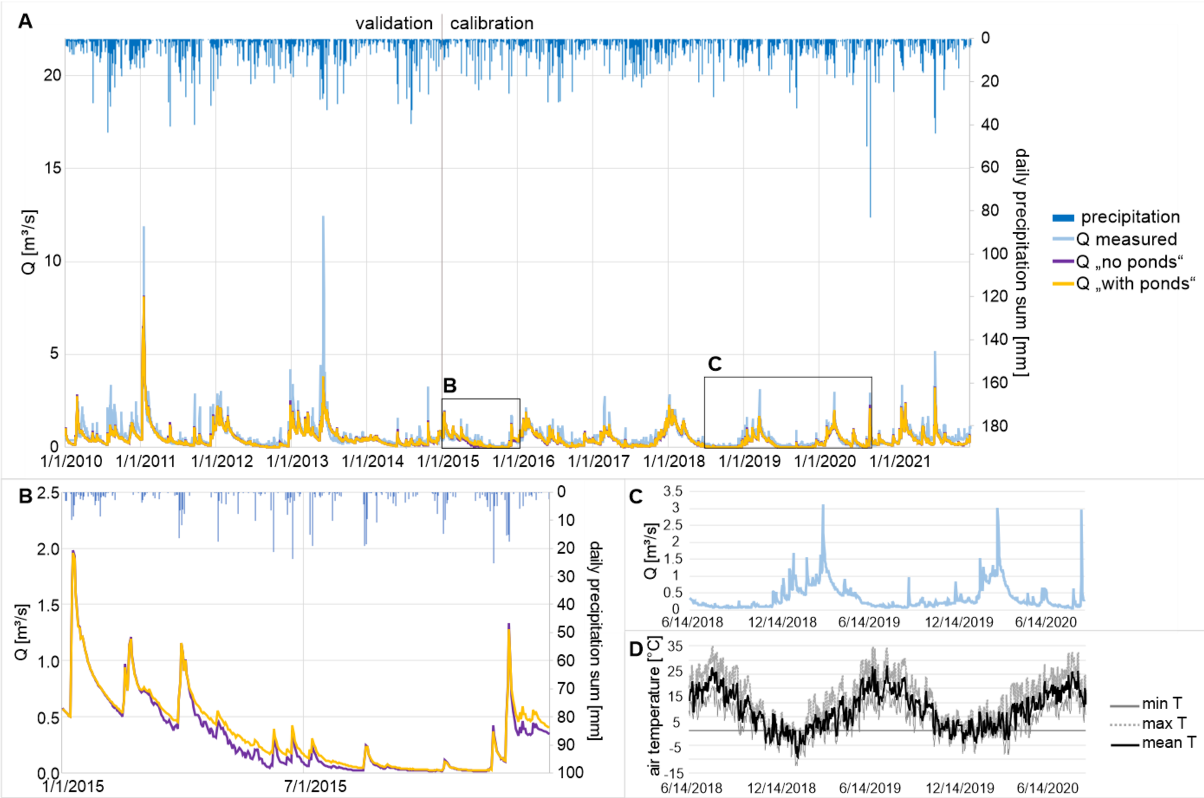

Figure 8: Discharge, precipitation and air temperature of the study area: (A) observed stream flow (Q measured, light blue line), modeled stream flow Q for scenarios with (yellow line) and without ponds (violet line) and precipitation (blue bars) at the climate station "Rehau" for the whole time period not including warm-up; dark line separated the calibration from the validation period. (B) close-up into the simulated streamflow with and without ponds in 2015; (C) close-up into the measured streamflow for the period of the temperature study; (D) mean (black line), minimum (solid grey line) and maximum (dotted grey line) air temperature at the climate station "Hof" for the period of the temperature study.

The BFI values were calculated using the hydrographs simulated by the model with ponds in 2018, 2019 and 2020 for 13 subbasins matching the temperature monitoring campaign. (Table S 1). Average BFI over all reaches was 0.79 ± 0.07 over the whole modeled period. Considering BFI in the temperature-study stretches in 2018–2020 revealed a strong temporal differentiation: While groundwater contribution in 2018/19 was on average 91 ± 9% during the extreme low flow conditions, it decreased to 54 ± 8% in 2020. Highest BFIs were observed in the upstream regions of the MB and BB, with slightly lower values towards the downstream reaches and in the HB. However, the values did not differ significantly between upstream and downstream reaches of the respective streams in neither of the years (Mann-Whitney U-test, p-values > 0.05).

3.4.2 Temperature regime

In 2018, after a dry and warm spring, water levels dropped below the mean low water levels from the end of July until mid-November. Mean daily air temperatures at the climate station "Hof" reached values > 20 °C during in July and August, with a maximum of 26.6 °C and only minimal precipitation occurring in summer and autumn (Riedel et al. 2021). In the subsequent winter, low temperatures (mean daily air temperatures down to below −7.0 °C) and increased discharge were accompanied by snowfall, but this period of rather high discharges was followed by another dry spring and extremely hot summer in 2019. Water levels in the study streams decreased to even lower values, with stream segments drying out completely in a neighboring stream system (see Hoess and Geist (2020)) and stream flow in the study streams only sustained due to intervention by the water authorities discharging water from adjacent ponds and pumping from deep wells. Increased precipitation alleviated the situation from September onwards, moving towards a period of higher precipitation rates and well sustained stream flow levels during autumn, winter and the subsequent spring and summer 2020, when mean daily air temperatures > 20 °C were only reached during August.

The average summer temperature regime differed between the monitoring points at the inflow of ponds effluents. ADM_su among monitoring points within the pond outlet channels was 16.8 ± 2.4 °C and therefore higher than in the main stream, where mean ADM_su at 'us' monitoring points was 15.2 ± 1.5 °C and 15.5 ± 1.0 °C at 'ds' monitoring points. The mean difference between 'us' and 'ds' was 0.5 ± 1.3 °C, indicating increased Tw downstream of the pond outlet channel inflow. Average MaxD_su at 'out' was 20.7 ± 2.5 °C, with AMax_su reaching 18.4 ± 2.1 °C. The highest ever measured Tw within the study period was 27.5 °C at the site BB5_out. Average MaxD_su and AMax_su at 'us' sites were 18.6 ± 1.8 °C and 17.0 ± 1.7 °C and 19.1 ± 1.2 °C and 17.3 ± 1.1 °C at 'ds', respectively. The mean range_su was 3.0 ± 0.7 °C for 'out', 3.5 ± 0.6 °C for 'us' and 3.4 ± 0.6 °C for 'ds' monitoring points. MaxT was reached on average on day 209 ± 24 for 'out', on day 209 ± 20 for 'us' and on day 207 ± 25 for 'ds' monitoring points. On average, Tw was below 14.5 °C for 21 ± 21 of the days of summer at 'out', for 33 ± 22 days at 'us' and for 26 ± 16 days at 'ds' monitoring points. Tw of 20 °C or higher was reached at 24 ± 21 of the days of summer at 'out', at 9 ± 12 days at 'us' and at 8 ± 8 days at 'ds' monitoring points.

In winter, ADM_wi was 2.8 ± 1.0 °C at 'out', 2.9 ± 0.6 °C at 'us' and 2.8 ± 0.6 °C at 'ds' monitoring points. MaxD_wi at 'out' was 5.6 ± 0.7 °C, with Amax_wie reaching 3.3 ± 1.0 °C. Average MaxD_wi and AMax_wi at 'us' sites were 5.8 ± 0.4 and 3.5 ± 0.6, and 5.7 ± 0.4 and 3.4 ± 0.6 °C at 'ds', respectively.

However, all metrics showed a high site variability (Table 6, Figure 9) with the impact of the pond discharge on the receiving stream ranging from an increase of ADM_su downstream of up to 2.6 °C to a decrease of 2.9 °C. Therefore, an analysis of the monitoring sites was conducted via cluster analysis to identify underlying patterns.

Table 6: Average temperature metrics at the three monitoring points per site (codes after Table 4, su = summer, wi = winter) as well as the difference between the value calculated at 'us' compared to 'ds' (delta value) over the whole study period.

	ADM [°C]		MaxD [°C]		Amax [°C]		Range [°C]		MaxT [°C]	Jdmax	b14_5	a20
	su	wi	su	wi	su	wi	su	wi	su	su	su	su
BB1	19.2	4.4	22.5	6.7	20.2	4.6	1.9	0.5	23.8	218	4.0	44.0
BB2 out	15.9	2.9	20.1	5.9	17.6	3.4	3.2	1.0	22.3	209	17.7	7.7
BB2 us	13.5	3.0	16.5	5.6	14.9	3.5	2.6	0.9	17.8	207	62.0	0.0
BB2 ds	14.8	3.2	18.4	5.6	16.2	3.5	2.7	0.7	20.5	204	35.3	2.0
Delta BB2	1.3	0.1	1.9	0.0	1.3	0.0	0.0	−0.1	2.7	−4	−26.7	2.0
BB3 out	19.2	2.4	22.8	5.0	20.6	2.6	2.7	0.4	25.0	211	2.3	47.0
BB3 us	14.9	2.9	18.6	5.9	16.4	3.4	2.9	1.0	20.7	208	33.7	1.0
BB3 ds	16.2	2.8	19.5	5.6	17.8	3.3	3.0	0.9	22.2	229	15.7	10.3
Delta BB3	1.3	−0.1	0.9	−0.2	1.4	−0.1	0.1	−0.1	1.5	22	−18.0	9.3
BB4 out	13.4	4.2	16.3	6.4	15.7	4.9	4.1	1.4	19.5	229	65.0	0.0
BB4 us	15.9	2.7	19.3	6.0	17.8	3.5	3.8	1.4	22.2	226	19.3	11.3
BB4 ds	15.6	3.0	19.0	6.0	17.4	3.7	3.5	1.4	21.7	205	23.0	6.7
Delta BB4	−0.3	0.2	−0.3	0.0	−0.4	0.2	−0.3	0.0	−0.5	−22	3.7	−4.7
BB5 out	19.0	3.1	23.5	5.7	20.3	3.4	2.7	0.6	25.7	205	4.3	41.3
BB5 us	15.9	3.0	19.5	6.3	18.2	3.9	4.2	1.6	22.5	204	19.7	14.3
BB5 ds	16.3	3.0	19.9	6.1	18.5	3.8	4.1	1.5	23.2	211	16.7	18.3
Delta BB5	0.4	0.0	0.5	−0.1	0.3	0.0	−0.1	−0.1	0.7	7	−3.0	4.0
MB1 out	15.6	2.7	19.1	5.1	17.2	3.3	3.0	1.0	21.0	215	27.0	3.0
MB1 us	12.6	3.5	15.9	6.2	14.2	4.3	3.1	1.5	17.5	218	74.0	0.0
MB1 ds	13.6	3.3	17.0	5.9	15.1	4.0	3.1	1.3	19.2	224	60.7	0.3
Delta MB1	1.0	−0.1	1.5	−0.2	0.9	−0.2	−0.1	−0.2	1.0	3	−13.5	0.0
MB2 us	14.0	2.8	17.2	5.6	16.0	3.4	4.0	1.1	20.0	211	51.0	0.5
MB2 ds	15.8	3.0	19.7	5.8	18.1	3.7	4.3	1.3	23.5	214	20.7	15.7
Delta MB2	2.2	−0.2	3.0	0.0	2.8	−0.1	0.6	0.1	4.5	−2	−37.5	20.5
MB3 out	15.8	1.9	19.4	5.4	17.2	2.5	2.6	1.1	23.0	154	21.0	8.0
MB3 us	15.3	2.5	18.9	5.8	17.4	3.2	4.1	1.3	21.5	204	27.7	7.3
MB3 ds	14.9	2.5	18.3	5.8	16.9	3.2	3.7	1.3	20.8	207	33.3	2.3
Delta MB3	−0.3	0.0	−0.6	0.0	−0.6	0.0	−0.4	0.0	−0.8	2	5.7	−5.0
EB1 out	14.0	2.1	18.3	5.0	15.8	2.6	3.2	1.0	23.7	210	44.7	7.0
EB1 us	17.9	2.0	21.8	4.8	19.8	2.4	3.5	0.6	24.2	201	4.0	37.0
EB1 ds	15.8	2.7	19.9	5.0	17.7	3.2	3.3	1.0	22.3	186	23.7	14.0
Delta EB1	−2.1	0.0	−1.9	0.0	−2.1	0.0	−0.2	0.0	−1.8	−15	19.7	−23.0
HB1 out	18.0	2.1	22.0	4.8	19.5	2.4	3.0	0.5	24.3	220	8.7	35.7
HB1 us	15.3	2.5	18.7	5.7	17.1	3.0	3.5	1.0	21.0	206	27.0	6.0
HB1 ds	16.5	2.6	20.0	5.2	18.1	3.0	3.2	0.7	22.0	203	9.7	11.3
Delta HB1	1.2	0.1	1.3	−0.6	1.0	−0.1	−0.3	−0.3	1.0	−3	−17.3	5.3
HB2 out	14.7	2.7	18.2	6.0	16.5	3.4	3.7	1.3	20.5	205	37.7	4.7
HB2 us	15.1	3.0	17.8	5.5	17.0	3.4	3.7	0.8	20.8	206	31.5	3.5
HB2 ds	15.5	2.7	18.7	5.7	17.3	3.2	3.7	1.1	20.8	204	22.3	5.7
Delta HB2	0.8	0.2	1.0	0.4	0.6	0.2	−0.1	0.1	0.3	−5	−15.5	3.0
Delta MB4	0.8	−0.1	0.5	−0.2	0.8	−0.1	0.0	−0.1	0.3	−1	−15.3	1.0
HB3 ds	15.8	2.4	19.5	5.6	17.4	3.0	3.2	1.1	21.5	208	20.7	6.0

Of the 11 monitoring points within the pond outlet channels, six were categorized as 'close' to the pond with a mean distance between pond outlet and the 'out' monitoring site of 25.5 ± 12.8 m, three were categorized as 'medium far' with a mean distance of 536.0 ± 62.0 m and the remaining two as 'far' with a mean distance of 1757.9 ± 209.0 m between the pond outlet and the 'out' monitoring point.

The multivariate patterns of temperature metrics calculated for the summer months indicated a clear distinction between monitoring points in the outlet channels of close-by ponds (Figure 9) from those of far-away ponds and most of the monitoring points in the main streams. The cluster analysis generated two main branches, one consisting of four monitoring points within pond outlet channels close to the pond in BB, HB and EB as well as the monitoring point EB1us, also located close to a pond. These monitoring points did no differ significantly in summer temperature regime (SIMPROF test, $p > 0.05$). In the second branch, the monitoring point MB3out was significantly different from the remaining points. Within the remaining monitoring points, mainly comprised of ‘us’ and ‘ds’ points, as well as the medium and far-away ‘out’ points, one cluster with monitoring sites with a similar, rather cool summer temperature regime was separated, including two far-away ‘out’ points, together with the three most upstream sites in the MB (SIMPROF test, $p > 0.05$). The rest of the monitoring points did not show a consistent clustering between ‘us’, ‘ds’ or ‘out’ or within streams or sampling sites.

The impact of the distance between the pond and the inflow of the outlet channels was further supported by the significant strong negative relationship between the ADM_su of ‘out’ monitoring points and the distance between the monitoring point and the pond (Pearson’s Test, $R(8) = -0.67$; $p < 0.05$). ADM_su at points that were only 50 m from the pond outlets was 18.6 °C, with a maximum daily mean temperature of 22.5 °C and an average maximum daily temperature of 19.9 °C. In contrast, when the monitoring point was more than 1000 m away from the pond outlet, average daily mean temperature in summer was 15.3 °C with a maximum of 19.2 °C and an average daily maximum temperature of 17.1 °C.

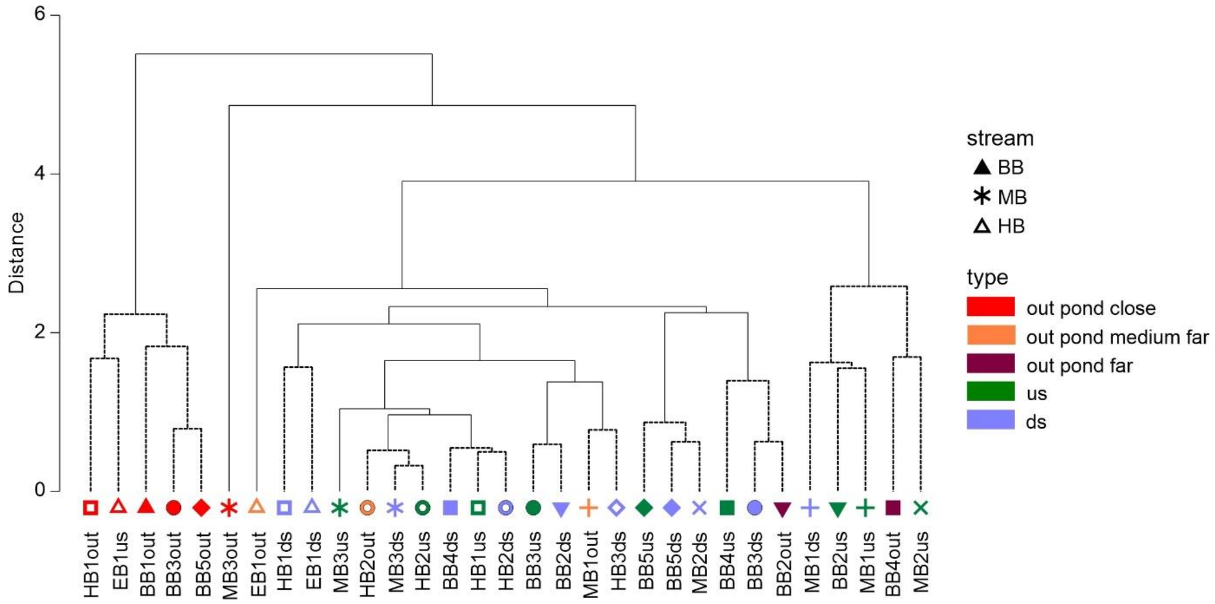

Figure 9: Result of the cluster analysis based on average normalized values for all temperature metrics in summer for monitoring points in pond outlet channels close to the pond (red), medium (orange) and far from the pond (violet), and monitoring points upstream (green) and downstream (blue) of the pond outlet channels into the main streams; similar symbols represent monitoring points at the same sampling site; filled symbols represent sites in BB, cross/star symbols represent sites in MB; open symbols represent sites in HB; solid lines represent a significant separation of the two branches, dashed lines represent that the separation is not significant, according to SIMPROF test.

Summer temperature regime at the inflow of close-by ponds, such as at site BB3 and HB1 (Figure 10), was clearly influenced by the heated pond effluents. Summer T_w within the outlet channels reached high mean and maximum values, with daily maximum values over 20 °C for more than 60% of the monitored summer days, when water was discharged from the ponds. Compared to the main stream above the inflow, Tw was elevated by 3–4 °C. Discharge of this effluents led to an increase in water temperature downstream of the pond inflow, with a maximum increase of 5.5 °C from 16.5 °C at HB1us to 22.0 °C at HB1ds. Therefore, the number of days when the daily mean Tw was below 14.5 °C was reduced from 31 at BB3us to 16 at BB3ds in 2018 and from 25 to 7 in 2019, while values > 20 °C were reached downstream in 11 instead of 2 days in 2018 and 18 instead of 1 day in 2019. Due to the overall higher temperatures, the effect strength was higher during the hot and dry years, but a similar pattern could be observed in 2020. In winter, water discharged from the ponds was slightly cooler than in the receiving stream. Differences between 'us' and 'ds' sites were marginal. The mean daily temperature range in winter was decreased at BB3 out with 0.4 °C compared to 1.0 °C and 0.9 °C at BB3us and BB3ds, while it was similar between all three sampling sites in summer.

The impact of a longer distance from the pond and groundwater contribution was obvious when analyzing the temperature regime of far-away ponds and site EB1. Summer temperature regime at BB4, where the pond was located approximately two kilometers upstream, was the opposite of those observed at the close-by ponds. Here, summer Tw at the outflow channel was cooler than in the main stream, causing a maximum decrease at BB4ds of 4.5 °C compared to BB4us. The number of days with average daily Tw < 14.5 °C increased by 4 days, while the number of days with maximum daily Tw > 20 °C decreased between 1 and 8 days. In winter, this pattern was reversed and daily mean Tw in the outlet channel were on average 1.4 °C higher than upstream in the receiving stream, causing a slight increase of Tw downstream. Summer Tw was also strongly decreased downstream of the outlet channel at EB1, which was used to discharge groundwater from a deep well during summer 2018 and 2019. This overwrote the impact of the inflow from EB1us, which clustered together with the "close ponds" in the cluster analysis and had a pond located 95 m upstream. During 2018/19, mean ADM_su at EB1out was 13.1 °C while average ADM_su was 18.2 °C at EB1us, with Tw reaching > 20 °C on 43 and 54% of the summer days in 2018 and 2019. Below the groundwater discharge, daily Tw decrease by 2.8 °C with a maximum decrease of 8.5 °C. During the more moderate summer of 2020, Tw at EB1us still reached an average ADM_su of 17.4 °C but without the groundwater discharge from the well, ADM_su at EB1out was 15.6 °C and 16.6 °C at EB1ds, following the patterns of a close-by -pond-site, which also accounted for the winter temperature regime.

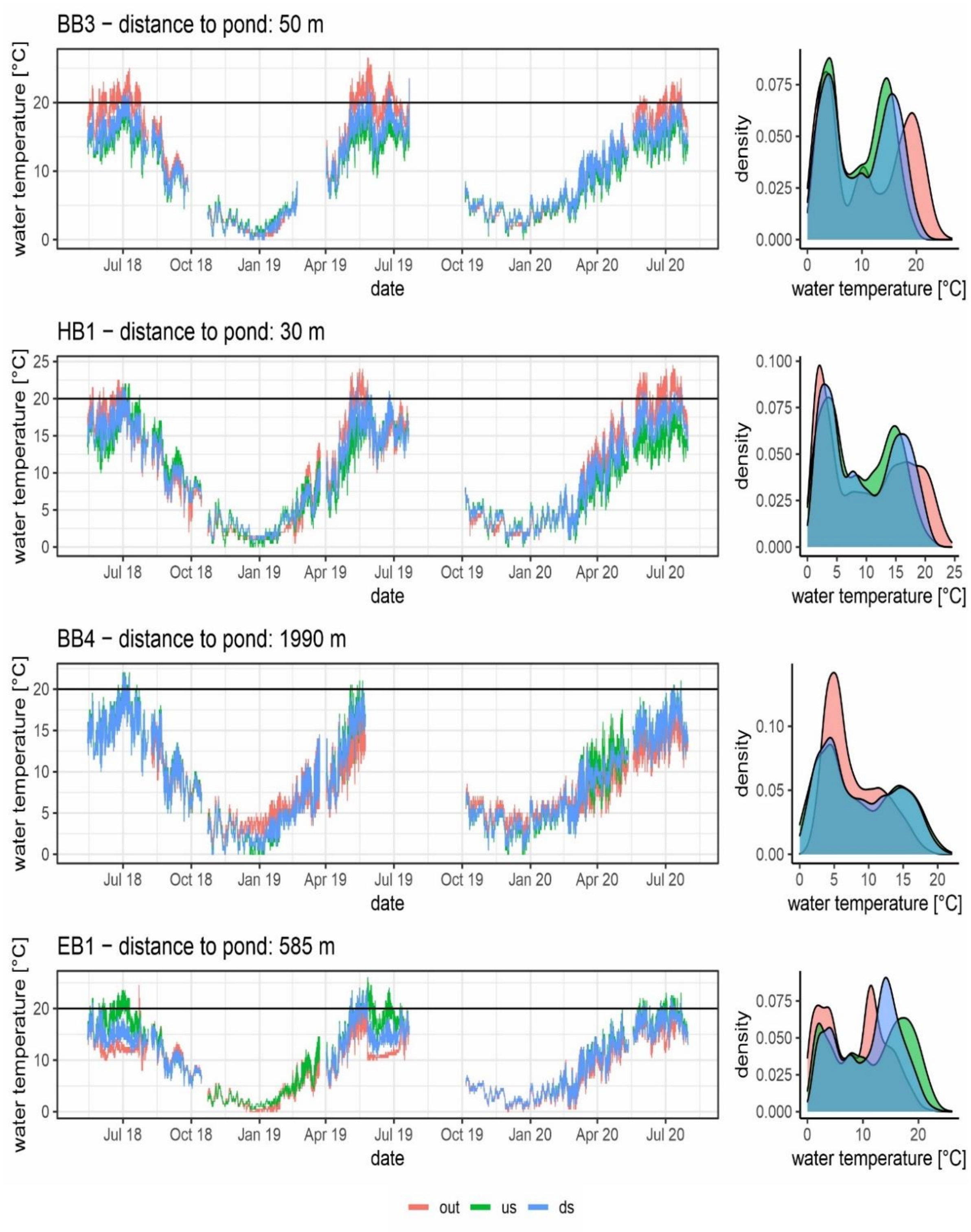

Figure 10: Hourly stream temperature at four sampling siteswith water temperature measured at the 'out' (red), 'us' (green) and 'ds' (blue) monitoring points, solid line represents the limit of 20 °C, the density plot on the right shows the proportion of a certain temperature measured at the monitoring points over the respective time period.

The proportion of corridor area between monitoring points that was covered by pond area, by forested land and by open land is given in Table 7. The upstream section of BB is dominated by forest and several ponds until site BB3, when open, more agriculturally used land becomes more important. The catchment around the EB/HB system was almost exclusively surrounded by forest, which is mixed with open land only at the very down-stream section. The upstream section of MB is surrounded by both open and forested land, while the downstream part is dominated by forest.

Table 7: Proportion of different LU categories within the 180 m corridor along the reach between the subsequent monitoring points, reaches are differentiated between short stretches covering the distance between monitoring points upstream and downstream of a channel inflow (site X) and longer stretches covering the distance between two subsequent monitoring points (point X–point X + 1); within a sampling site, this refers to the reach between 'us' and 'ds' monitoring points, between sampling sites, this refers to the reach between 'ds' point of the upstream and 'us' point of the subsequent site.

Reach	Reach length (km)	% Pond area	% Forested area	% Open land	Total area [ha]
BB1	0.77	12.5	87.5	0.0	42.49
BB1-BB2	1.44	0.8	99.2	0.0	71.58
BB2	0.06	5.2	94.8	0.0	114.07
BB2-BB3	0.11	0.0	100.0	0.0	1.34
BB3	0.02	16.3	83.7	0.0	5.51
BB3-BB4	1.16	0.0	68.1	31.9	16.39
BB4	0.02	0.1	47.2	52.7	62.27
BB4-BB5	1.16	0.8	5.0	94.2	34.44
BB5	0.03	12.9	0.0	87.1	2.73
EB1	0.02	3.3	96.7	0.0	3.62
HB1	0.06	7.1	92.9	0.0	12.24
HB1-HB2	1.18	2.6	97.4	0.0	32.43
HB2	0.03	50.4	49.6	0.0	2.03
HB2-HB3	1.75	2.3	47.8	40.0	34.38
MB1	0.03	8.7	8.2	83.0	9.07
MB1-MB2	1.16	0.3	59.8	40.0	22.86
MB2	0.16	7.8	57.2	35.0	3.17
MB2-MB3	0.39	0.0	68.2	31.8	6.63
MB3	0.02	25.4	74.6	0.0	1.48
MB3-MB4/HB2	1.35	2.6	97.4	0.0	26.39

The multiple regression model set up to investigate if a certain LU type or the distance between two monitoring points was responsible for changes in ADM_su along the stream course, yielded only the interaction between all three parameters as significant ($F_{df} = (7, 46) = 3.44$; $p < 0.05$), explaining 34.4% of the variation. Therefore, temperature changes along the stream course and their associated LU changes were analyzed in detail (Figure 11).

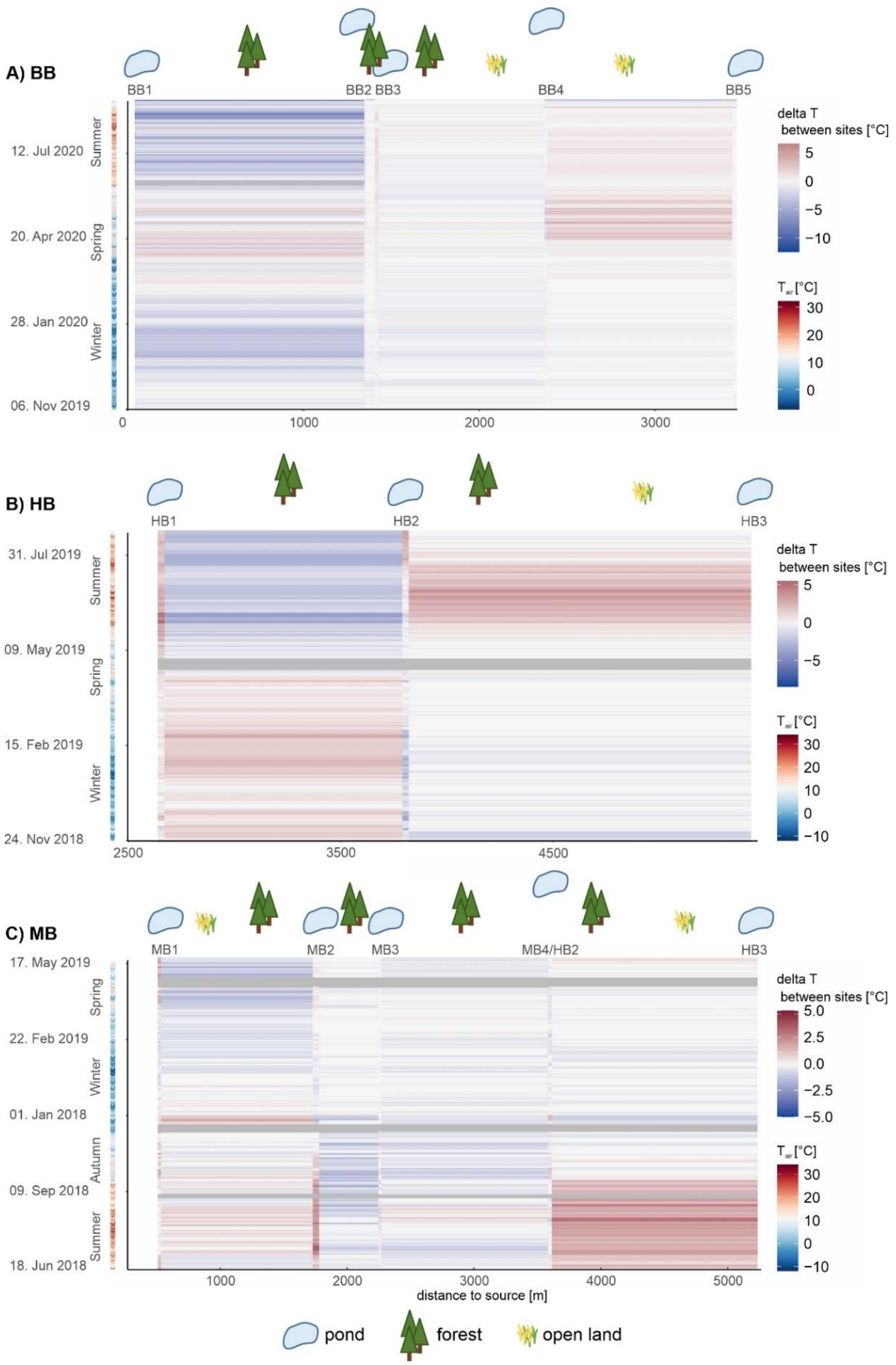

Figure 11: Temperature difference between two subsequent monitoring points (point X–point X + 1) along the three study streams, within a sampling site (this refers to the reach between us and ds monitoring points) and between sampling sites (this refers to the reach between ds point of the up-stream and us point of the subsequent site); Line color gives the difference in water temperature; matching air temperature (T_{air}) at the climate station "Hof" is represented on the left side; pictograms represent the main LU features along the respective reach.

Temperature changes along the BB between winter 2019 and summer 2020 revealed a general cooling effect of relatively warm water released from the pond facility at the very upstream part while the stream course passed through forested land. The effect varied slightly over the seasons, where an inversed pattern of increasing Tw in spring could be observed. Within the forested stretch, the inflow from the close-by pond facility at BB3 caused an increase in Tw with increasing air temperatures during summer (see also Figure 10), that was more or less sustained when the proportion of forest decreased between BB3 and BB4. After BB4, were the inflow from a drainage channel with a far-away pond (see Figure 10) had a slight cooling effect, Tw strongly increased, in particular in spring and early summer when flowing through mainly open land. In total, Tw along the 3.5 km of stream course decreased on average by 4.2 °C from 19.3 at BB1ds to 15.1 °C at BB5ds in summer, and by 1.7 °C from 5.2 to 3.5 °C in winter.

Along the HB, the discharge from a close-by pond at HB1 in the upstream section caused an increase of Tw over a short distance of 30 m all over the year that was strongest during the early summer, when it increased up to 5.5 °C. During winter and early spring, the increase in Tw below the inflow from the pond was moderate. Along the sub-sequent stream stretch flowing through coniferous forest for 1.1 km, ΔTw showed contrasting seasonal patterns with a decrease during summer, while a further increase could be observed during winter and early spring. An increase below the discharge from a second pond facility at HB2 that made up more than 50% of the surrounding corridor area, could only be observed in the second half of summer, while Tw decreased below it during winter. The subsequent stream segment went along a LU change from forested to open land, which made up 40% of the corridor, as well as inflows of several small, open ponds. During winter, Tw remained relatively constant between HB2 and HB3 but increase by up to 2.5 °C in summer. In total, Tw along the 2.8 km of stream course increased on average by 1.8 °C from 14.6 at HB1us to 16.4 °C at HB3ds in summer, and decreased by 0.1 °C from 2.0 to 1.9 °C in winter.

At MB1, an increase in Tw downstream the inflow of the medium-far but with 1.1 ha one of the largest ponds in the study was apparent from mid-spring on, reaching a maximum increase of 3.5 °C. From July on, no difference in Tw could be observed between MB1us and ds, despite strongly increasing air temperatures. This was due to the fact, that no more water was discharged from the pond and the outlet channel was dry until the end of September. Coniferous forest was along the left banks of the subsequent stream segment, while the right banks were more dominated by open land. Tw decreased along this segment from mid-winter to early summer, while the values increased moderately during late summer and autumn. Inflow from the pond facility very close to MB2 in-creased Tw in summer by up to 5.0 °C from 17.0 °C at MB2us to 22.0 °C 50 m downstream at MB2ds in August 2018. Between autumn and spring, Tw were only moderately affected by discharge from the pond facility. In summer, the high Tw at MB2 ds decreased again over the subsequent stream segment covered for 0.5 km by coniferous forest, e.g., by 1.5 °C from 22.0 °C to 20.5 °C at MB3us and in maximum by 3.0 °C from 19.0 to 16.0 °C. The pond facility at MB3, draining into a small wetland before flowing towards the main stream, did not cause any changes in Tw from upstream to downstream over the year. The next stream section of 1.3 km contributed to stream warming for several weeks in late summer, while Tw decreased along the forested stretch for most of early summer and autumn and showed a variable pattern of slight increases, decreases or neutral behavior over the rest of the year. Inflow of the following far-away pond caused moderate decreases, except of the autumn period, when Tw slightly increased. A major increase in the summer Tw was again apparent for the stream section between MB4/HB2 and HB3, when LU changed to open land. In summer 2018, Tw along this stretch rose on average by 1.3 °C from 14.8 to 16.1 °C and in maximum from 19.5 to 22 °C. In total, Tw along the 4.7 km of stream course increased on average by 3.1 °C from 13.0 °C at MB1us to 16.1 °C at HB3ds in summer, and decreased by 1.4 °C from 3.3 °C to 1.9 °C in winter.

3.5 Discussion

Comparing the SWAT models with and without ponds showed that the cumulative effect of the ponds on stream flow is rather moderate but still noticeable in the catchment hydrology, particularly regarding baseflow support and therefore increased resilience to drought. Without the ponds, discharge levels would have been about 4.5% lower during the extreme drought in 2018/19 which might have been the final strike for the already extremely low water levels to sustaining sufficient flow around FPM beds. The already existing threat of extreme low flows in the study area became obvious during the summers of 2018 and 2019, when mussels had been translocated during the drying-out of a neigh-boring FPM stream system. In a future under climate change, extreme situations of dry and hot conditions during summertime become more likely and an understanding of the effects of ponds on the hydrology and temperature regime will facilitate taking management decisions on these systems with endangered species. On the other hand, FPM are also threatened by high peak flows with high shear stress destabilizing the substrate and causing mussel downstream drift (Baldan et al. 2020; Hastie et al. 2000; Morales et al. 2006; Strayer 1999). The elevated baseflow levels, together with the increased retention capacity for high flows, indicates that ponds had an overall buffering effect by increasing extremely low water levels and slightly reducing peak flows, both being beneficial for the target species. These patterns are consistent with studies on the modeled impact of wetland loss (Ameli and Creed 2019) or the predicted effect of wetland reconstruction (Javaheri and Babbar-Sebens 2014), but the overall effect for the studied ponds were lower compared to the effects of wetlands. The different values for hydraulic conductivity of pond and wetland bottom soils are likely the cause for the different effect strength between these two types of water bodies: While wet-land soil hydraulic conductivity ranged from 3.1 to more than 50 mm/h in studies by Ameli and Creed (2019) and Babbar-Sebens et al. (2013), the calibrated PND_K in the present study was only 0.6 mm/h, therefore limiting recharge to the shallow aquifer and the groundwater storage capacity. However, concerning the function of ponds for fish production, this value seems realistic as seepage through the pond bottom should be avoided to ensure sufficient and stable pond water levels (Stone and Boyd 1989). The generally low effect on hydrology, in particularly high flows, might also be due to small size of the ponds compared to overall catchment area (Ebel and Lowe 2013). Baldan et al. (2021b) also simulated comparably low reduction of high flow magnitude for small ponds with a retention volume of 50 m³ per ha subbasin area.

The impact of the ponds concerning high flows was only marginal, causing no delay of peak flow timing or flood duration, as suggested by Javaheri and Babbar-Sebens (2014) and Acreman and Holden (2013) for wetlands. The reason is likely that the ponds, used for extensive fish production, are usually filled all over the year, except for several days during the fish harvest (Hoess and Geist 2021). Storage capacity of the rather shallow ponds was therefore already utilized, and the remaining volume played a minor role in flood mitigation. When ponds are redesigned to serve as additional storage basins, depth could be increased to generate a higher storage capacity, as suggested for wetland restoration (Babbar-Sebens et al. 2013). It would also be an option to not fill them up completely to allow for higher flood retention.

The impact of the ponds on supporting low flows was higher than for the buffering of high flows, most probably due to an increased groundwater recharge through the pond bottom that resulted in higher baseflow levels in the streams. This effect seemed to exceed the effect of an increased water loss through evaporation from the pond surface (Jalowska and Yuan 2019; Rains et al. 2016), which, on the contrary would lead to reduced stream flow. Since pond area was very low compared to the total catchment area (0.6%), such evaporation effects had a marginal effect in the model. Hydrologically, the ponds are connected to the stream via their inlet and outlet, from where water is directly abstracted from or released into the stream as well as through subsurface flow paths (Rains et al. 2016). The presence of a pond, wetland or even beaver dam can increase the local water table (Ameli and Creed 2019; Majerova et al. 2015), increasing the storage capacity of the shallow aquifer (Juszczak et al. 2007). When the groundwater storage is influenced by ponds or wetlands, the stored water is released as baseflow with a time lag, which can be extremely important during periods of drought and prolonged lack of precipitation, therefore increasing drought resilience of the system (Ameli and Creed 2019). This does only account if the pond bottom consists of natural soils with a certain hydraulic conductivity to allow pond-aquifer exchange as is the case for all ponds in the study area. If pond bottoms as sealed using impermeable pond foil, such effects are likely to decrease.

However, the use of the SWAT model to accurately assess pond impacts has significant limitations. First, the representation of ponds in the model as HEW limits the model accuracy. It was not possible to model all ponds individually, e.g., in two subbasins 25 individual ponds that were spread over the whole subbasin area had to be technically combined into one “big pond”. This limits a realistic representation of hydrologic functions based on the area-to-circumference-ratio, as small ponds have proportionally stronger effects on groundwater retention than larger ponds (Juszczak et al. 2007). In addition, several authors demonstrated that the location of ponds and wetland relative to the stream course plays a role in defining small-scale hydrological processes (Ameli and Creed 2019; Rains et al.

2016). The limited possibility of de-tailed integration of pond configuration and location into the SWAT model will certainly increase the uncertainty of the real pond effects in the study catchment, whereas the representation of ponds as permanently filled waterbodies next to the stream network matches the fish pond management practice in the study area. The majority of pond owners refill their ponds directly after fish harvest. However, the usual pond management in big fish farms includes a so-called “wintering” of ponds, leaving them dry after fish harvest for disinfection and nutrient mineralization. Such pond management practice might not be modeled correctly by the standard SWAT approach.

In addition, the overall model evaluation showed an overestimation of low flow during the validation period resulting from the unevenly distributed occurrence of (extreme) low flow conditions in the calibration (including 2018 and 2019), while the two highest flood events occurred during the validation period. This can be a major issue in model calibration for periods affected by the ongoing climate change, when conditions are shifting towards increasing extreme events. Including a longer calibration/validation period of observed stream data could help to improve the model performance. While a hydrological model can be used to assess cumulative effects at the regional scale, in particularly for small catchments, the spatial resolution might not be sufficient to simulate streamflow at a high spatiotemporal scale (Ameli and Creed 2019; Rains et al. 2016; Stanfield et al. 2009). This is due to the often-insufficient representation of small scale climate conditions through climate stations and the high variability of natural flow conditions.

Relying on hydrological modeling alone might therefore lead to incorrect representation of the effect of single pond facilities at the local scale, which needs to be addressed using a different approach. In the present study, analyzing the temperature regime along the stream course in a field study was chosen due to the relatively easy and cost-efficient use of automatic temperature loggers at a higher spatial resolution and measurements at a smaller time scale (Webb and Nobilis 2007).

The temperature monitoring indeed revealed strong small-scale effects of discharge from ponds during the summer. The strength of the effect was highly dependent on the distance between pond and adjacent stream, with summer thermal regime in the outlet channels of close-by ponds being clearly differentiated from far-away ponds and the main stream. Standing water bodies such as ponds receive heat inputs through solar radiation, which can lead to substantial warming due to their larger area, limited cover and shading by riparian vegetation, and restricted heat exchange with cooler stream water at regulated inflow (Van Buren et al. 2000). Common carp production in Central Europe can only be feasible through the warmer temperature in shallow ponds, as this fish species is usually adapted to warmer water in its original south-eastern range (Balon 1995). The effect of heated effluents on stream temperature was apparent in the higher temperature metrics directly downstream of close-by ponds compared to upstream, often representing medium values between ‘us’ and ‘out’ and increasing the number of days with MaxT > 20 °C, in the most extreme case by up to 20 days. Seyedhashemi et al. (2021) found a similar thermal response in stream reaches with a high portion of ponded catchments. They found a distinct “thermal signature” based on the relation with air temperature by which it was possible to identify anthropogenic influences on the natural thermal regime. To evaluate the ecological impact of such elevated summer Tw on the FPM as target species is complex, as it depends not only on its specific requirements for growth and maturation but also on the thermal tolerance of its host fish (Pandolfo et al. 2012). On the one hand, both species rely on nutrient-poor, oligotrophic conditions, which are at present only to be found in the remaining almost natural headwaters. However, re-production and growth in these habitats might be constrained by the cold temperatures, indicating that the current species distribution represent the upper-most limit. In spring and early summer, elevated temperatures below pond discharges might actually improve growth rates of juveniles and fecundity of adult FPM. On the other hand, high summer temperatures might prevent brown trout from visiting sites close to mussel beds, and thus from infestation with mussel larvae, which depend on the attachment to the fish host for their further development. Even after successful infestation of the host fish, temperature still plays a key role in the host-parasite interaction of freshwater mussels. Highest metamorphosis success in the thick-shelled river mussel *Unio crassus* could be observed at 17 °C in the laboratory, while excystment rates decreased at lower or higher temperatures (Taeubert et al. 2014). In natural environments, spatial variation, e.g., through pond effluents, plays an important role in this interaction, as demonstrated in the case of the MB: In the upstream regions the MB displayed a cold temperature regime strongly dominated by groundwater contribution. Here, a high proportion of days with ADM_su below 14.5 °C might prevent sufficient growth of adult and juvenile FPM. Heated effluents from ponds might therefore support growth rates in such areas, potentially enhanced through nutrient input, if the heating is not too high to exceed host thermal limits around the period of glochidia re-lease. At site MB2, the inflow of heated effluent from the close-by pond facility strongly decreased the proportion of days with insufficient Tw for growth but at the same time in-creases the proportion of days with MaxT > 20 °C.

However, the temperature effect of pond effluents is often locally restricted and was found to be strongly dependent on the distance between pond and adjacent stream. Particularly the case of BB4 demonstrated that warmer Tw can be compensated along longer distances, even causing a contrasting effect of cooling of the main stream after the inflow of a side channel. Inflow from such cool-water sources can provide thermal refuges for sensitive species such as the brown trout and contribute to the overall thermal heterogeneity (Ebersole et al. 2015; Kuhn et al. 2021). Cool summer Tw are usually caused by strong groundwater contribution as could be observed in the case of EB1: mixing the pond effluent with groundwater from the deep well in 2018 and 2019 to support stream flow, yielded an average daily summer Tw of 11.6 °C, causing a substantial cooling in the downstream reach. On the reach scale, longer distances allow for a stronger interaction of the effects of pond effluents and other catchment LU features as well as groundwater upwelling, potentially buffering the impact of point sources such as pond discharge. This was also indicated by the regression model and has also been suggest by other authors (e.g., Monk et al. (2013)). Groundwater indeed dominated summer stream flow during the low flow period in 2019, as demonstrated by Kaule and Gilfedder (2021) and the BFI values derived from the modeled hydrographs, indicating a strong impact on summer stream temperatures. However, since BFI along the stream course remained constant but Tw increased over a larger distance with transitions of LU from forested to open land along the BB and HB longitudinal temperature patterns could also be related to LU change. Riparian vegetation had an important effect on longitudinal temperature patterns. In the BB, the combined effect of high groundwater contribution and reduced heat inputs through shading likely caused the cooling of the water heated by the ponds at BB1. The cooling effect of riparian vegetation cover is attributed mostly to reduced energy in-puts from solar radiation (Garner et al. 2014), or both groundwater inflows and shading (Story et al. 2003). Mitigation of increased Tw through riparian shading has been proven by multiple studies (e.g., Garner et al. (2017); Garner et al. (2015); Imholt et al. (2010); Malcolm et al. (2008); Roth et al. (2010)). Stretches covered by coniferous forest even yielded cooler summer Tw than mixed-deciduous forest, with Tw being reduced for both forest types when compared to open land (Dugdale et al. 2018). Similar effects could be observed in the present study along stretches with coniferous vegetation in all three study streams. They even compensated the cumulative impact of pond effluents within forested stretches. Along open stretches lacking thermal refuges for salmonids in sufficient quantity and quality, restoration measures such as the creation of holding pools should be considered, to create habitat with lower temperature during drought and temperature stress, particularly in the headwater areas prone to low summer runoff levels (Ebersole et al. 2001; Elliott 2000).

Beyond the observed inter-annual effects, with higher summer Tw values and stronger pond effects in drought years, compared to the more normal conditions in 2020, seasonal patterns could be detected. While effluents from close-by pond caused a significant warming during summer, Tw in winter was slightly lower at ‘out’ monitoring points of close-by ponds than in the main stream. In contrast, Tw monitoring at groundwater dominated BB4out yielded higher temperatures, as was expected fact that groundwater temperature is equal to the mean air temperature, therefore being higher in winter. Effects of pond effluents on the main stream in winter remained more or less neutral, most likely due to the overall higher flows, reducing the importance of effluents contributing to the stream flow. In winter, riparian cover from coniferous trees limits radiative losses and re-emit longwave energy towards the stream surface (Dugdale et al. 2018), therefore leading to a reversed effect of warming of the concerned stretches in winter. In contrast, open stretches without such a vegetation cover, displayed a cooling gradient. This demonstrated again that LU plays a significant role for longitudinal temperature patterns on the reach scale.

3.6 Conclusion

The SWAT model revealed moderate impacts of the ponds on hydrology and drought mitigation, indicating a buffering effect increasing baseflow and slightly decreasing peak flows on the catchment scale. Both effects are likely beneficial for the FPM which is threatened by increasing drought periods during summer and high shear stress at peak flows, mostly during winter. The impacts on the thermal regime were pronounced at the local scale, mainly for close-by ponds, while the interaction between ponds, reach length, LU and groundwater contribution caused more variable effects on the reach scale. Above all, the present study provides implications for pond management in FPM streams concerning hydrologic and temperature regime. On the catchment scale, ponds can be seen as a measure to retain water in the landscape by retaining peak flows and elevating baseflow levels during summer. However, their use for drought mitigation might be compromised through careless pond management practices such as water abstraction from the stream to sustain pond water levels during extreme low flow. This effect could not be included into the SWAT model but represented a common practice in the study region, making the water authorities issue a general order prohibiting any water deprivation below a critical gauge height. If ponds are in close proximity to large mussel beds, effluent discharge might be stopped during the most critical period of glochidia release to lower the risk of the host fish avoiding the specific reach. This also applies to situations when pond dis-charge is used to maintain water levels in the receiving streams. Timing of the FPM life cycle and the thermal capacity of the receiving stream ecosystem should be taken into consideration, to avoid additional stress. Therefore, the timing of water release could be adapted to summer Tw or tree plantation around ponds could be used to increase shading and decrease warming of pond water. In addition to adapted pond management, an assessment on available thermal refugia for the salmonid host fishes can be recommended.

4. Spatiotemporal variation of streambed quality and fine sediment deposition in five freshwater pearl mussel streams, in relation to extreme drought, strong rain and snow melt

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4.1 Abstract

Oxygenated streambeds are considered a key requirement for the successful recruitment of stream fauna, including highly endangered freshwater pearl mussel *Margaritifera margaritifera*. Excessive amounts of fines impede exchange between open water and interstitial, leading to colmation and low oxygen levels in the juvenile habitat. Understanding the dynamic relationship between sediment delivery, transport, deposition and remobilization in relation to anthropogenic drivers is still poorly understood, yet is essential for conservation and restoration.

This study analyzed spatiotemporal sediment dynamics and interstitial habitat quality in five pearl mussel streams at the border region between Bavaria, Saxony and the Czech Republic during 2018 and 2019, comparing extremely dry periods with higher discharge events caused by snow melt and rainfall. Physicochemical habitat conditions within the streambed and sediment deposition were recorded in high spatial resolution along the stream courses, with a particular focus on the effects of tributaries and outflows of man-made fishponds. Habitat conditions were unsuitable for juvenile pearl mussels at the majority of sites, indicated by pronounced differences in physicochemical parameters between open water and the substrate, independent of discharge conditions. Sediment deposition varied markedly between discharge events, in terms of both the quality and quantity of deposits. Snow melt resulted in the highest sedimentation rates, but the smallest proportion of fine particles. During low flow conditions, fine sediment deposition was highly variable, ranging from 0.048 to 4.170 kg/week/m^2, mostly independent of flow velocity. High spatiotemporal variation was observed within and amongst stream systems, revealing different longitudinal patterns of fine sediment deposition, with catchment land use

as the main driver. Temporal variability in sediment deposition was mainly associated with the discharge condition while abiotic parameters varied mainly with season.

The high site-specificity of sedimentation rates and substrate conditions in response to different discharge events highlights the importance of an adapted conservation management which considers anthropogenic effects at the local scale.

4.2 Introduction

Freshwater mussels are amongst the most threatened animal species worldwide (Lopes-Lima et al., 2018), with species such as the European freshwater pearl mussel (*Margaritifera margaritifera,* L.) having become target species for conservation and aquatic ecosystem restoration (Boon et al. 2019; Geist 2010; Geist and Hawkins 2016). Reasons for freshwater mussel decline are mostly driven by man-made changes including degradation, fragmentation and pollution of aquatic habitats, as well as land use intensification (Dudgeon et al. 2006; Geist 2011). The freshwater pearl mussel (FPM) is particularly susceptible to anthropogenic modification of stream ecosystems and the surrounding catchment (Addy et al. 2012; Cosgrove et al. 2000; Denic and Geist 2015; Geist 2010; Geist and Auerswald 2007; Horton et al. 2015; Ma 2016; Österling and Högberg 2014). The species is adapted to clear, cool, fast-flowing streams with low nutrient concentrations (Bauer 1988; Geist 2010; Hastie et al. 2000). Its complex life cycle includes a parasitic phase on salmonid fish (mostly *S. trutta* and *S. salar*), lasting up to 10 months, as well as a post-parasitic juvenile phase buried within the substrate for up to five years (Taeubert et al. 2010). Mortality during the juvenile phase due to deficient substrate quality is considered to be the main reason for the ongoing lack of sufficient recruitment for 30–50 years in most of its central European populations (Geist 2010). Even the largest remaining pearl mussel populations in Germany, in Upper Franconia, have lacked recruitment of juveniles for decades (Bauer 1988; Denic and Geist 2017; Stoeckl et al. 2020), mostly due to excessive amounts of fine sediments in the streambed (Denic and Geist 2015).

A stable, well-oxygenated streambed containing only a small proportion of fine sediment is required throughout the entire juvenile development (Buddensiek 1995; Geist and Auerswald 2007; Hastie et al. 2000). The clogging of the streambed by fine-grained particles filling up the voids between larger pieces of gravel, known as colmation, causes degradation of the hyporheic habitat, as it physically impedes the infiltration of oxygen-rich surface water into the interstitial zone (Brunke 1999; Schälchli 1992). This dynamic process builds up under low flow conditions, until the occurrence of a high discharge event provides enough shear stress to mobilize larger bed particles (Hjulström 1935; Wharton et al. 2017). Beyond these theoretical assumptions, natural high flow events might vary in their impact on sediment dynamics, depending on their duration, strength and timing (Navratil et al. 2012). Fine sediment loads in streams have increased due to human activities such as deforestation and agricultural intensification (Collins et al. 2009; Davies et al. 2009; Kemp et al. 2011; Knott et al. 2019), and now often exceed the natural transport capacity. In-channel sources like bank erosion and remobilization of deposed fines (Kronvang et al. 2013; Naden et al. 2016; Wood and Armitage 1997) in combination with the structural diversity of the streams (Braun et al. 2012) also play a role in fine sediment dynamics.

The effects of man-made fishponds on substrate quality in pearl mussel streams remain controversial, particularly since such ponds may have positive effects, such as the trapping of fine sediments originating from terrestrial erosion, as well as negative effects such as the release of peak concentrations of fines, e.g. during water release for fish harvest. Along pearl mussel streams in their central European core area of distribution at the border between Bavarian, Saxony and the Czech Republic, about 200 fishponds have been built, yet there is a lack of assessment of their effects on pearl mussel habitats. This information is therefore crucial for their operation in the context of pearl mussel conservation. To improve streambed quality, considered the most important target of conservation (Denic and Geist 2015; Geist and Auerswald 2007; Horton et al. 2015; Österling et al. 2010), both the reduction of fine sediment inputs from anthropogenic activities in the catchment, as well as instream processes, need to be considered (Auerswald and Geist 2018).

The complex processes of sediment dynamics and streambed quality are rarely studied at a high spatial resolution and in the context of variable discharge conditions. Tributary inflows might lead to abrupt increases in the proportion of fines, disrupting the continuous transport downstream (Rice et al. 2001). Intra-annual variations might occur due to natural or anthropogenic changes in sediment delivery processes and variations in stream flow.

In 2018, a long-lasting period of extremely high temperatures and low precipitation occurred from April to mid-October in central Europe, followed by an even hotter period in summer 2019. This resulted in extremely low groundwater levels and stream flow in the study region, and caused negative effects for ecosystems all over Europe (Buras et al. 2020). Studies on the response of sedimentation patterns to low flow and drought conditions are underrepresented compared to flood events (Lake 2003), although the negative effects of sediment deposition are expected to increase under such conditions. With the ongoing climate change, droughts are likely to occur more frequently (Vogel and Olivier 2019) which poses a challenge to adapt management plans (Denic and Geist 2017; Geist 2010) on the basis of a systemic process understanding of such effects.

The core objective and novelty of this study was to use this unprecedented opportunity of extreme conditions to analyze the spatiotemporal variation of streambed quality and fine sediment deposition in five freshwater pearl mussel streams from three catchments in the context of extreme drought, heavy rainfall and snow melt, conjoint to anthropogenic effects related to land use and fishponds. Specifically, we hypothesized that:

i) Drought – connected with precipitation shortfall, low water levels and the highest temperatures – results in the lowest streambed quality.

ii) Heavy rain and snow melt, resulting in increased flow, provide crucial ecological functions to pearl mussel streams, such as the washing out of fine sediments and increasing the exchange rate between open and interstitial water.

iii) Streambed quality and fine sediment deposition reveal a high degree of spatiotemporal variation, with highly different baseline levels depending on the stream.

4.3 Material and Methods

4.3.1 Study area and sampling design

The study area included five streams from three sub-catchments of the Saechsische Saale, located within the main distribution area of freshwater pearl mussels in central Europe, at the border region between Germany and the Czech Republic (Figure 12). All streams are oligotrophic, siliceous headwater streams of the Elbe drainage (Denic and Geist 2015). The average annual discharges of these streams range between 0.36 m^3/s in the Bocksbach, 0.18 m^3/s in the Hoellbach/Maehringsbach system and 0.90 m^3/s in the Suedliche Regnitz/Zinnbach system. Natural recruitment of juvenile FPMs has been absent in all these streams for several decades, leaving an overaged population of around 27,000 individuals in the whole study area (Stoeckl et al. 2020). Land use varies within and between the sub-catchments, with spruce forest making up 80% of land use around the Maehringsbach (MB), Hoellbach (HB) and Bocksbach (BB) (Schmidt et al. 2014). The upper reach of the BB is dominated by the outflow from a large commercial trout farm, and is surrounded by coniferous forest, the lower part is more intensively used for maize and crop production. The catchment of Zinnbach (ZB) consists mainly of extensively used grassland and arable land. The catchment of the Suedliche Regnitz (SR) is mostly used for agriculture, with the degree of intensification increasing towards the lower reach. Several small villages and settlements also influence both streams, although a nature conservation area was established around them in 2001. The catchments of all five streams are used for fish production in earthen ponds. The most common fish species produced in the ponds are cyprinids such as the common carp (*Cyprinus carpio*), and to a lesser degree rainbow trout (*Oncorhynchus mykiss*). In total, around 200 ponds, mostly managed by private owners, drain into the pearl mussel streams.

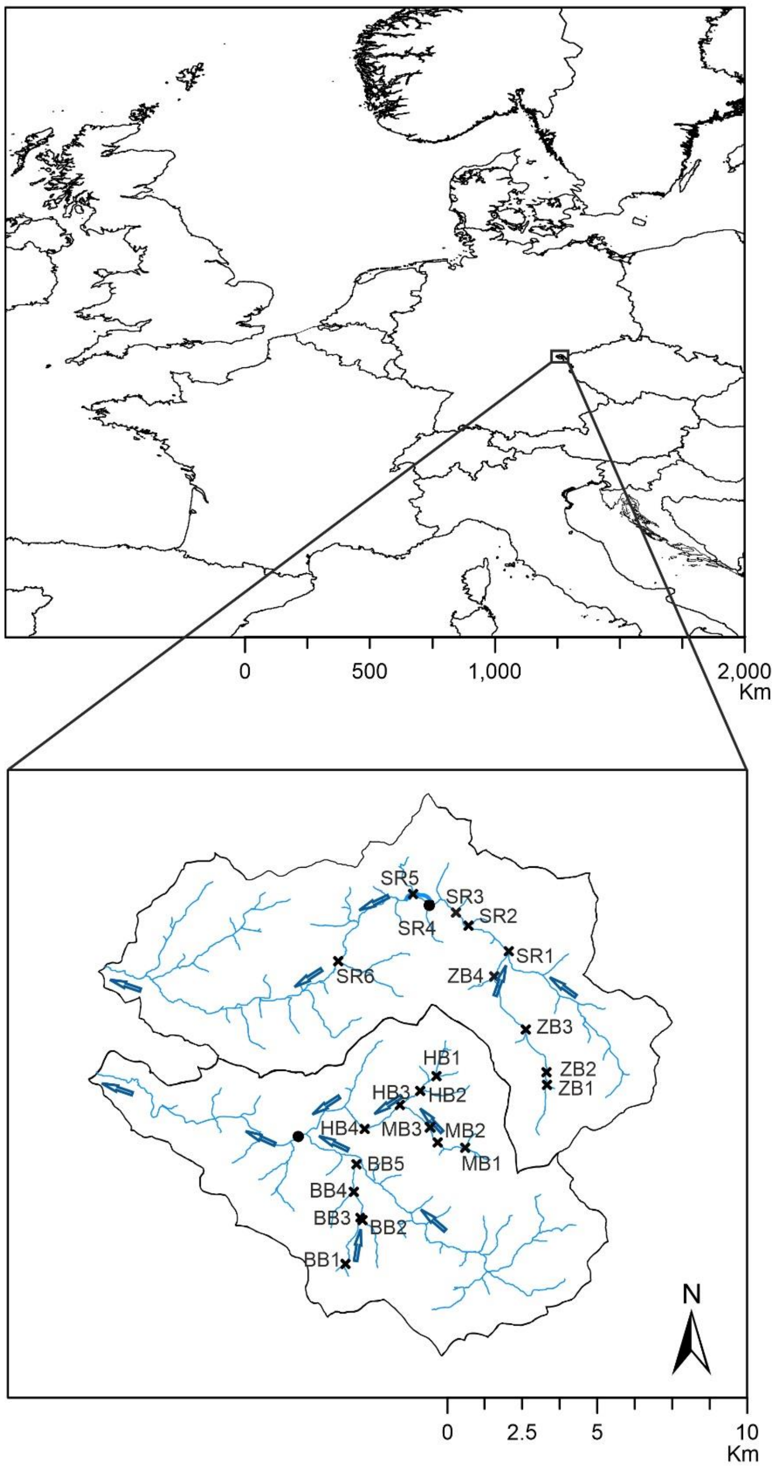

Figure 12: Map of the study area showing the sampling sites in the study streams (crosses) as well as the two main gauging stations (circles): gauge' Suedl. Regnitz', close to site SR4, integrating discharge from Zinnbach (ZB) and Suedliche Regnitz (SR), gauge 'Schwesnitz' integrating discharge from Bocksbach (BB), Hoellbach (HB) and Maehringsbach (MB) (see also Figure 13); arrows indicate flow direction.

Overall, 22 locations were selected within the streams to account for longitudinal influences. They were set at confluences of small tributaries with pond facilities located upstream. At each confluence, two sampling transects were established and equipped with sediment traps: one upstream ('us') and one downstream ('ds') of the confluence. At major confluences of two streams, e.g. the Maehringsbach and the Hoellbach at site HB3 and the Zinnbach and Suedliche Regnitz at SR1, an additional transect was established within the tributary (‘out’, also representing the out-flow of a pond facility, Figure 12). Five sites were located in BB, four in HB, three in MB, four in ZB and six in SR. Most of the sites in BB, HB and MB were directly downstream of fish pond facilities (av. distance of 25 m, range 10–60 m), except for one site at BB and one in the MB (nearest pond 1960 m and 450 m away, respectively). The average distance to the nearest pond in SR and ZB was 718 m, and ranged from 30 to 1750 m.

The sampling was carried out in summer and autumn 2018, as well as in winter, spring, summer and autumn 2019 (Table 8), to cover different discharge conditions. Five different types of discharge events were classified according to the average discharge, the type of precipitation, and the timing (Table 8 and Figure 13). Average discharge was below the long-term low flow threshold for eight of the thirteen sampling intervals, which were classified as “extreme low flow” events. Two intervals had average discharge exceeding this threshold, but still below the long-term average flow, classified as “low flow” events. The discharge exceeded the low flow threshold after increasing temperatures caused “snow melt” during the sampling interval in February 2019. Total rainfall of 43.4 mm during the sampling in April 2019 caused a high discharge event classified as “spring rain”, whereas a total of 58.3 mm of precipitation was observed during the “autumn rain” event in September 2019.

Table 8: Characterization of the 13 sampling intervals; mean values are given separately for the ungauged BB, HB and MB (discharge integrated in the gauge 'Schwesnitz') and ZB and SR, (discharge integrated in gauge 'Suedl.Regnitz').

Sampling interval	Discharge event	Sum precipi-tation [mm]		Av. Discharge [m³/s]	Range discharge [m³/s]	Av. flow velocity [m/s]	Av. water depth [cm]	Av. T [°C]	Av. total sediment deposition [kg/week/m²]
June 18	low flow	0.40	BB/HB/MB	0.350	0.040 – 1.050	0.33 ± 0.11	9.4 ± 3.3	15.8 ± 1.7	1.295 ± 1.563
			ZB/SR	NA	NA	0.41 ± 0.12	16.8 ± 6.9	13.3 ± 1.5	0.629 ± 0.274
July 18	extreme low flow	8.20	BB/HB/MB	0.114	0.049 – 0.201	0.08 ± 0.06	7.0 ± 2.6	16.7 ± 2.9	0.639 ± 0.409
			ZB/SR	0.028	0.014 – 0.054	0.09 ± 0.06	11.4 ± 5.4	15.0 ± 2.4	0.402 ± 0.262
August 18	extreme low flow	2.00	BB/HB/MB	0.082	0.040 – 0.119	0.06 ± 0.05	7.0 ± 2.5	15.9 ± 1.9	0.501 ± 0.311
			ZB/SR	0.013	0.010 – 0.016	0.03 ± 0.03	9.9 ± 4.5	14.5 ± 1.8	0.283 ± 0.314
September 18	extreme low flow	26.30	BB/HB/MB	0.111	0.040 – 4.410	0.03 ± 0.04	6.6 ± 2.6	13.0 ± 3.4	1.085 ± 0.784
			ZB/SR	0.022	0.011 – 0.131	0.05 ± 0.05	11.8 ± 5.6	14.6 ± 1.3	1.066 ± 1.536
October 18	extreme low flow	15.90	BB/HB/MB	0.150	0.040 – 0.350	0.04 ± 0.03	9.7 ± 4.4	10.3 ± 0.8	0.806 ± 0.531
			ZB/SR	0.021	0.008 – 0.086	0.13 ± 0.07	18.0 ± 6.9	9.4 ± 1.3	0.800 ± 1.816
November 18	extreme low flow	3.40	BB/HB/MB	0.103	0.060 – 0.162	0.05 ± 0.04	7.3 ± 2.6	5.0 ± 1.4	0.620 ± 0.642
			ZB/SR	0.035	0.031 – 0.041	0.10 ± 0.10	15.0 ± 12.5	6.8 ± 0.9	0.187 ± 0.113
February 19	Snow melt	10.20	BB/HB/MB	0.752	0.284 – 1.790	0.25 ± 0.11	12.8 ± 4.3	2.4 ± 1.3	5.131 ± 4.286
			ZB/SR	0.919	0.100 – 3.930	0.41 ± 0.11	32.6 ± 13.7	2.3 ± 0.5	16.289 ± 7.330
April 19	spring rain	43.40	BB/HB/MB	0.553	0.382 – 1.230	0.23 ± 0.09	12.6 ± 3.8	15.2 ± 1.5	5.061 ± 4.770
			ZB/SR	0.150	0.062 – 0.447	0.25 ± 0.09	21.7 ± 7.4	14.3 ± 1.8	2.714 ± 3.709
June 19	low flow	11.70	BB/HB/MB	0.206	0.177 – 0.263	0.14 ± 0.08	7.7 ± 2.7	17.9 ± 2.2	1.172 ± 1.047
			ZB/SR	0.031	0.012 – 0.052	0.21 ± 0.09	14.4 ± 5.9	16.5 ± 1.6	0.557 ± 0.359
July 19	extreme low flow	17.60	BB/HB/MB	0.120	0.060 – 0.203	0.08 ± 0.06	5.9 ± 2.3	14.3 ± 1.9	0.941 ± 0.526
			ZB/SR	0.009	0.005 – 0.016	0.06 ± 0.05	10.5 ± 4.9	13.7 ± 2.1	0.466 ± 0.526
August 19	extreme low flow	9.70	BB/HB/MB	0.096	0.032 – 0.215	0.11 ± 0.08	5.7 ± 2.7	12.4 ± 2.1	0.655 ± 0.449
			ZB/SR	0.004	0.002 – 0.012	0.05 ± 0.04	8.2 ± 4.6	14.3 ± 1.8	0.492 ± 1.055
September 19 (1)	autumn rain	58.30	BB/HB/MB	0.246	0.050 – 1.780	0.16 ± 0.11	7.7 ± 3.2	14.0 ± 1.8	6.427 ± 6.873
			ZB/SR	0.042	0.002 – 0.545	0.12 ± 0.08	12.2 ± 5.4	12.6 ± 2.4	3.649 ± 3.199
September 19 (2)	extreme low flow	1.90	BB/HB/MB	0.156	0.086 – 0.215	0.14 ± 0.10	6.2 ± 2.6	12.7 ± 1.1	1.364 ± 1.895
			ZB/SR	0.008	0.005 – 0.007	0.12 ± 0.10	10.9 ± 4.9	10.9 ± 1.6	0.253 ± 0.154

4.3.2 Abiotic habitat parameters

The physicochemical habitat parameters dissolved oxygen (O_2, in mg/ l), temperature (T, in °C), pH, and electric conductivity (cond., in µS/cm, related to 20 °C) were measured using a handheld Multi-3430 G Probe (WTW, Weilheim, Germany) once in the open water as well as in three samples of interstitial water taken at a substratum depth of 5 cm on the left, middle and right side of the streambed (see Geist and Auerswald (2007)). Redox potential (Eh, in mV) was measured in situ in the open water and at three points in the interstitial (5 cm depth) as the water samples. For the redox measurements, a handheld pH 3110 probe (WTW, Weilheim, Germany) was used together with a platinum electrode and an $Ag/AgCl_2$-reference electrode (Geist and Auerswald 2007). Flow conditions were characterized by water depth, measured with a graduated rod (± 0.1 cm) and current speed (v, in m/s) measured 2 cm above the substratum and at the surface, using a handheld measurement unit with a vane wheel (Flowtherm NT, Höntzsch, Waiblingen, Germany).

4.3.3 Sediment deposition

Plastic boxes of two different sizes (19 × 33 × 11 cm or 19 × 16.5 × 9 cm, ROTHO clear boxes, ROTHO Kunststoff AG, Würenlingen, Switzerland, cf. Denic and Geist (2015); Knott et al. (2019)) were used as sediment traps, depending on the stream width. Comparability of the sediment deposition per area was confirmed during an unpublished pre-experiment in the study streams for all but the biggest grain size (> 20 mm). The sediment traps were filled with a standardized 16–32 mm gravel mixture of allochthonous gravel from the alpine upland, which was easily distinguishable from the autochthonous sediment of the pearl mussel streams. Exposure of the traps mimics the introduction of clean gravel as commonly applied as a spawning-site restoration measure (Denic and Geist 2015; Pander et al. 2015). Each transect was equipped with one sediment trap, placed at the middle of each stream section, covering between 6 and 80% of the wetted width of the channel at all sites. A second, empty trap of the same size was buried into the streambed as a placeholder, to a depth that ensured the opening of the sediment trap level with the substratum surface. Before each weeklong sampling interval, the substrate was cleaned to ensure constant starting conditions. When collecting the samples, the whole content of the sediment trap was pre-processed in the field by sieving it through a 10 mm sieve. Bigger particles of native stream sediment were identified by eye and placed with the sample. The sediment trap was refilled with the same allochthonous substrate after each sampling, and put back into the same spot. Sediment samples were fractionated in the laboratory by wet sieving, using a wet-sieving tower (Fritsch, Idar-Oberstein, Germany) of decreasing mesh size (20, 6.3, 2.0 and 0.85 mm). The fractions retained on each sieve were dried at 102 °C and weighed to the nearest gram. Sediment deposition was expressed in kg/week/m^2. The percentage of each grain fraction was determined (as % wt), and the geometric mean particle diameter (dg, in mm) was calculated according to Sinowski and Auerswald (1999). The fraction < 0.85 mm is defined as ‘fine sediment’ throughout this paper.

4.3.4 Data analysis

All field measurements, as well as the weight of the sediment fractions, were entered into a Microsoft Access database. To assess the quality of the interstitial, independently of seasonal variations in temperature and pH, the exchange between the surface water and the interstitial habitat was expressed as the difference in physicochemical parameters measured in both ecotones and included in the analysis of abiotic parameters.

Testing for significant differences in univariate physicochemical parameters and sediment deposition between discharge events or streams was conducted using the open-source software R (R core team, 2016, www.rproject.org). Measured values were tested for normality and homogeneity of variance using the Shapiro-Wilk test and the Levene test, respectively. Parameters meeting both requirements were compared using t-test for two group- and ANOVA with host-hoc Tukey test for multi–group-comparison. For not-normally distributed parameters with inhomogeneous variances, the Wilcoxon-Rank test was applied for two-group comparison, whereas the Kruskal-Wallis test with the post-hoc pairwise Mann-Whitney U-test with Bonferroni-correction was used for multiple groups. The relationship between fine sediment deposition and measured flow velocity was tested using Spearman’s correlation test, since both parameters were not normally distributed. Multivariate analysis of abiotic parameters and of the composition of sediment deposits were performed using the statistical software PRIMER v7 (Plymouth Marine Laboratory, Plymouth, UK). The variables were normalized by subtracting the mean and dividing by the standard deviation to account for different scales and then used to perform Principal Component Analysis (PCA) on Euclidian distances between all samples. These resemblance matrixes were used to test for significant differences between the discharge events and streams via ANOSIM (ANalysis Of SIMilarity). A significance level of $\alpha < 0.05$ was applied for all statistical tests to decide if differences were significant.

4.4 Results

4.4.1 Discharge conditions

Discharge patterns in 2018 and 2019 were strongly influenced by the lack of rainfall during the exceptionally hot and dry summer months of the respective years, with higher flows from snow melt and rain in the intervening period in winter and spring 2019. Snow melt caused the highest peak discharge, and high flow resulting from snow melt lasted on average 2.5 times longer than after the rain events (Table 9), although nearly no precipitation was observed during this time. In contrast, the two rain events resulted in a short but steep rise in the hydrograph that only persisted for several hours. Although all streams lie within a close geographical range, differences in the discharge curves were observed (Figure 13) in terms of the strength and persistence of peak flows following rain events. In particular, the SR/ZB system responded quickly and strongly to precipitation and snow melt, as is evident from the high flashiness compared to the MB/HB system, where the hydrograph was more buffered, and generally lower discharge levels and peak flows were observed.

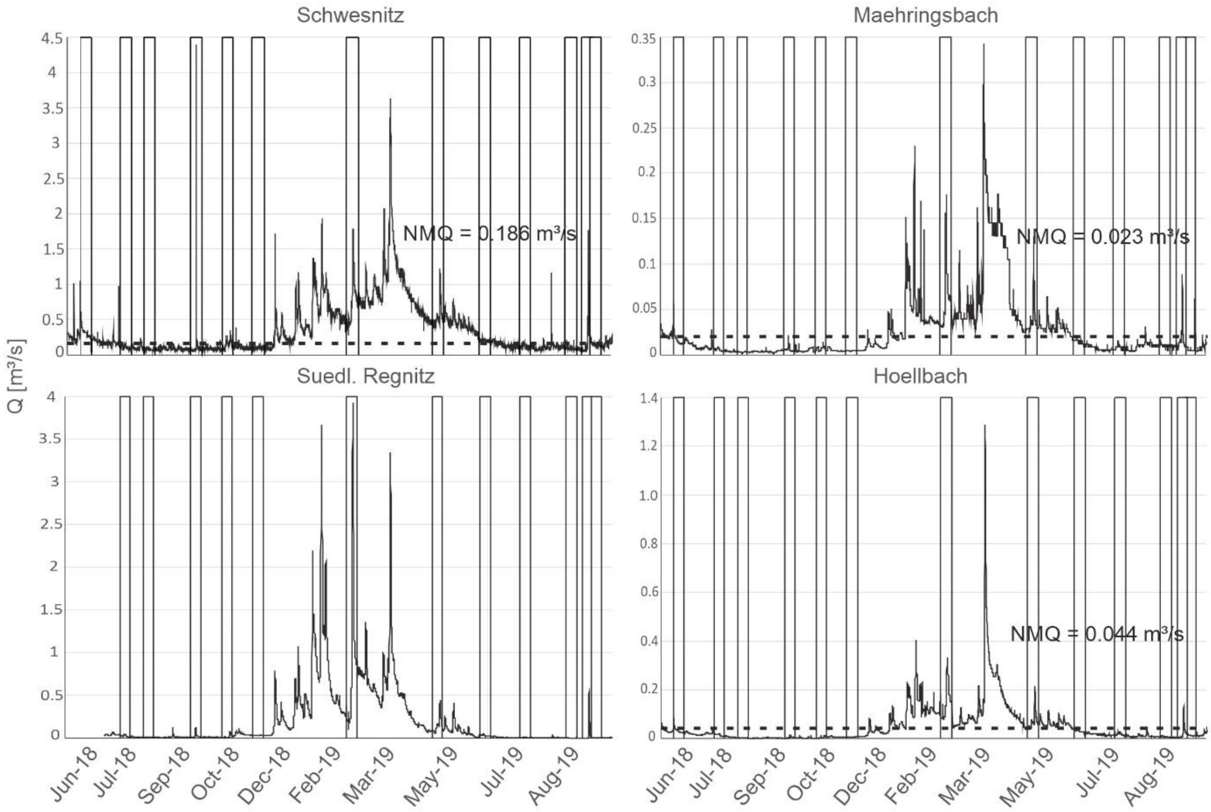

Figure 13: Hydrographs of four gauging stations on the studied streams: "Schwesnitz" integrates flows from BB, HB and MB; "Maehringsbach" is located in the downstream part of the MB, close to site "HB3 out"; "Hoellbach" is located at the very downstream part of the MB/HB system, downstream of the site "HB4";" Suedl. Regnitz" is located in the downstream part of the study area at ZB/SR, close to the sampling site SR4; sampling intervals are shown as bars, NMQ (long-term low flow discharge) values (where available) are expressed as dotted lines.

Table 8 summarizes the hydromorphologic and climatic conditions measured at the 13 sampling intervals. Seasonal patterns in temperature were most obvious, with highest mean and maximal values reached in the summer months. Decreasing water levels and flow velocities during summer and autumn 2018 and summer 2019 accompanied the development of drought conditions.

Table 9: Characteristics of the sampled flood events at four gauging stations; Q_{mean} = mean flood discharge, Q_{peak} = maximum flood discharge, 'Dur' refers to the time needed to reach peak discharge; Fl = "flashiness", rate of change in Q during the rising limb of the hydrograph (as per Buendia et al. (2014).

		Schwesnitz	Maehringsbach	Hoellbach	Suedl. Regnitz
Spring rain	Q_{mean} [m^3/s]	0.75	0.07	0.11	0.23
	Q_{peak} [m^3/s]	1.23	0.13	0.22	0.45
	Dur_{peak} [h]	18	13	15	17
	Fl [m^3/s/h]	0.038	0.008	0.011	0.020
Autumn rain	Q_{mean} [m^3/s]	0.52	0.05	0.07	0.15
	Q_{peak} [m^3/s]	1.78	0.09	0.13	0.55
	Dur_{peak} [h]	10	16	11	12
	Fl [m^3/s/h]	0.163	0.005	0.011	0.045
Snow melt	Q_{mean} [m^3/s]	0.96	0.07	0.19	1.43
	Q_{peak} [m^3/s]	1.79	0.12	0.34	3.93
	Dur_{peak} [h]	42	35	43	36
	Fl [m^3/s/h]	0.032	0.004	0.006	0.103

Concerning oxygen supply into the interstitial habitat, a corresponding pattern of lower O_2 concentrations in the interstitial at higher temperatures during low and extreme low flow could be observed (Table 8 and Table 10). O_2 levels in the interstitial were comparably low in the summer months (Jun-Sep) in both years (pairwise Mann-Whitney U-test, p-values > 0.05), with means ranging between 5.40 and 6.43 mg/l, then increasing significantly to 7.84 mg/l on average (pairwise Mann-Whitney U-test, p-values < 0.05) with falling temperatures from October 2018 to April 2019. In contrast, redox values measured in the streambed did not show a significant correlation with temperature (Spearman's correlation test, $p > 0.05$). Mean weekly sediment deposition during all study intervals was 1.981 ± 3.986 kg/week/m^2, ranging over three orders of magnitude from 0.032 to 34.472 kg/week/m^2. The lowest deposition rates occurred during the extreme low flow conditions, being even significantly lower than during the low flow events (pairwise Mann-Whitney U-test, $p < 0.001$), whereby the highest occurred during the snow melt (Table 8, Figure 14 A).

Table 10: Mean and range of current velocity and fine sediment deposition and mean of abiotic parameters measured in surface water (FW) and in 5 cm substratum depth (INT) measured in all five streams during the five discharge events.

Discharge event	Mean v [m/s]	Range	Mean deposition of fine sediment < 0.85 mm [kg/week/m²]	Range	Mean O_2 FW [mg/l]	Mean O_2 INT [mg/l]	Mean Eh FW [mV]	Mean Eh INT [mV]
Low flow	0.17	0.02 – 0.37	0.613	0.241 – 2.095	8.76	6.26	511.6	354.8
Extreme low flow	0.08	0.00 – 0.46	0.510	0.048 – 4.170	8.82	5.88	513.4	338.2
Spring rain	0.24	0.04 – 0.39	1.796	0.530 – 7.844	9.78	7.94	535.0	387.5
Autumn rain	0.15	0.00 – 0.90	2.406	0.578 – 7.367	9.67	6.18	428.8	335.3
Snow melt	0.32	0.04 – 0.55	3.135	0.285 – 6.634	12.73	10.74	494.6	346.4

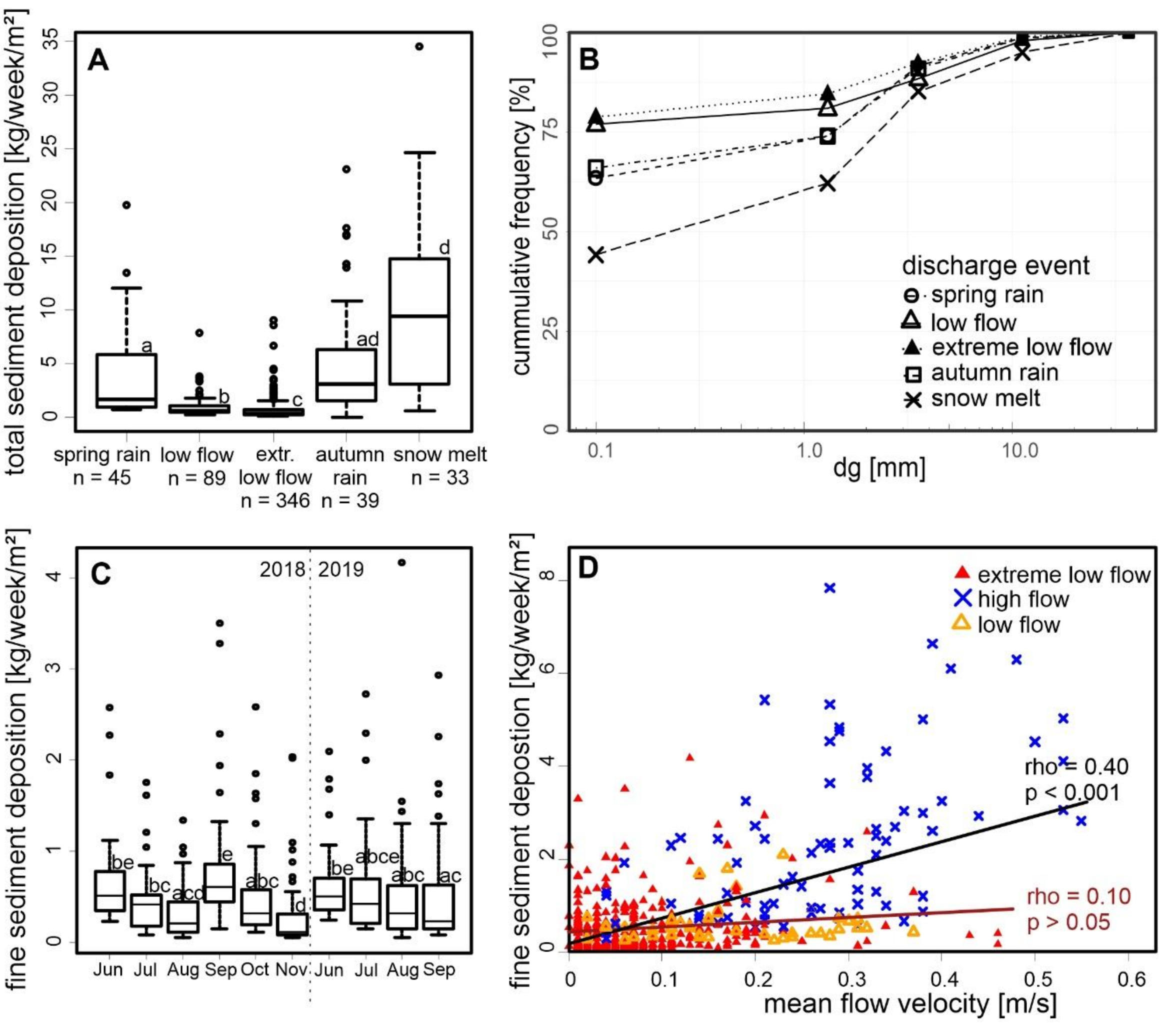

Figure 14: Temporal variation in (fine) sediment deposition : (A) Box and whisker plot comparing the amount of sediment deposition in the sediment traps over weekly periods during the five discharge events, (B) Cumulative texture lines, showing the distribution of different grain size fractions in deposits from the five discharge events, (C) Box and whisker plot comparing the amount of fine sediment between the sampling intervals under low flow and extreme low flow conditions, (D) Correlation between mean flow velocity and the amount of fine sediment deposed into the sediment traps, during the extreme low flow events, the low flow events and high flow events snow melt and spring rain; black line shows linear regression between all measurements, red line shows linear regression for extreme low flow only; values next to the regression lines show result of Spearman's correlation test for the respective data set; different letters at the boxes represent significant differences between the arithmetic means based on pairwise Mann-Whitney U-tests.

Discharge had a pronounced effect on the size composition of sediment deposits collected in the sediment traps (Figure 15). Deposits during low and extreme low flow conditions mostly contained particles smaller than 0.85 mm (Figure 14 B). Higher discharges during snow melt or rain events caused deposition rates nearly 10 times as high, and a more heterogeneous grain-size composition. Grain-size distribution differed significantly between all events except for the two rain events and the two low flow conditions (Table 11), with the highest differences occurring between extreme low flow and snow melt conditions (ANOSIM, $R = 0.802$; $p < 0.001$). The mean grain diameter (dg) of sediment deposits was similar for the two low flow events (pairwise Mann-Whitney U-test, $p > 0.05$) with mean values around 1.40 mm. The mean dg of 1.65 mm at the two rain events was significantly higher than under extreme low flow conditions (pairwise Mann-Whitney U-test, p-values < 0.05), but significantly lower than after the snow melt with 2.29 mm (pairwise Mann-Whitney U-test, $p < 0.001$). The lowest proportion of fine sediments was deposited during the snow melt, contributing less than 50% to the total material collected during this event. The two rain events resulted in a higher proportion of fines in the deposits, but particles larger than 2 mm still occurred with a proportion of over 25%.

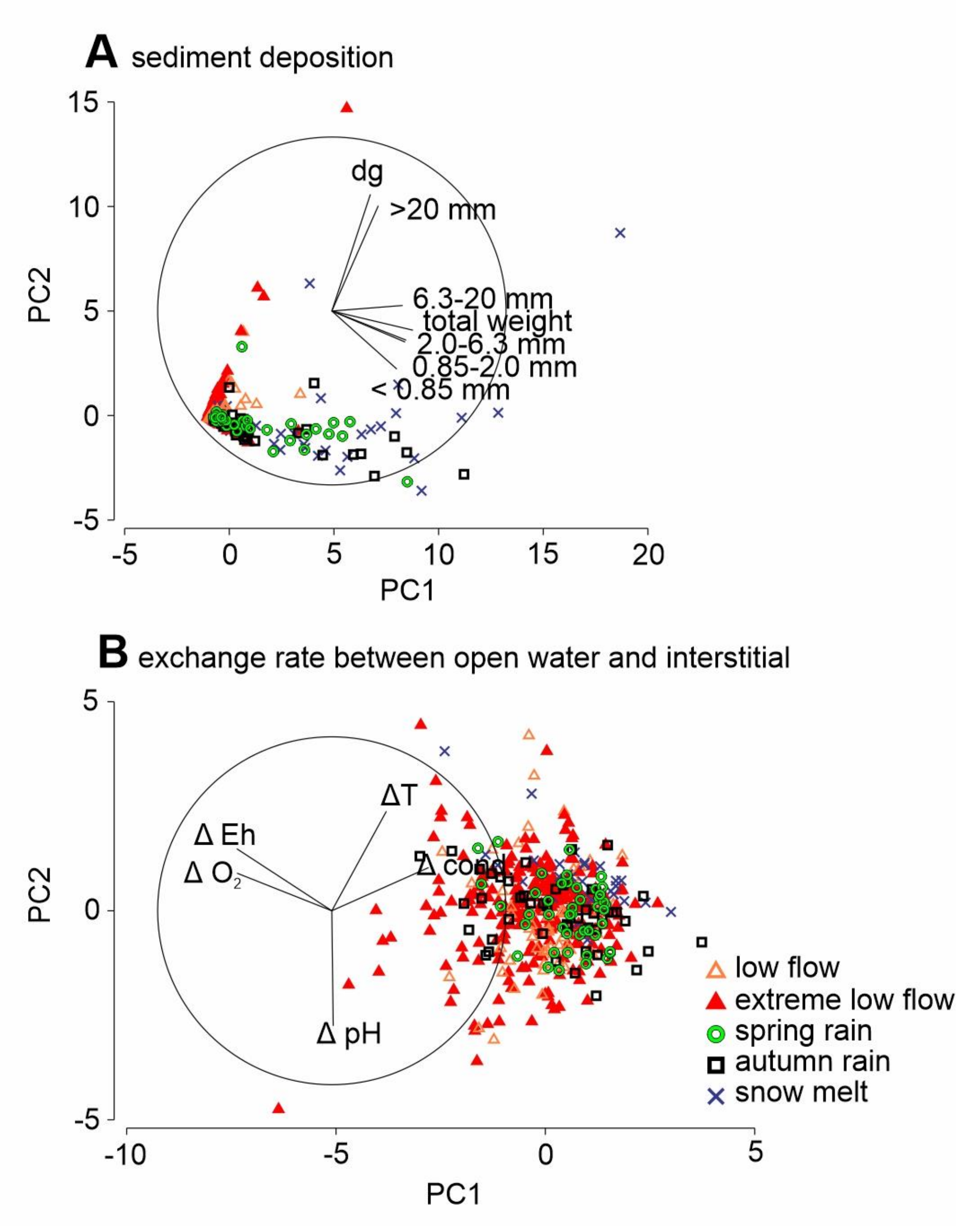

Figure 15: Principal Component Analysis (PCA) based on Euclidean distances performed on (A) Normalized values of weight of the grain size fractions in sediment deposition samples integrated for one week during the 5 discharge events and (B) Normalized deltas of abiotic parameters measured during the low discharge events "low flow" and "extreme low flow", before the "spring rain" to account for the "spring conditions" and after the high discharge events "autumn rain" and "snow melt".

In contrast to the clear effect that discharge had on the deposition of different grain sizes, delta values between open and interstitial water for all physicochemical parameters did not differ significantly between the discharge events (ANOSIM Global Test: R = -0.002, p > 0.05, Figure 15), indicating generally low exchange rates over all sampled conditions. Flow velocity measured directly on site ranged from 0.0 to 0.90 m/s during all sampling intervals. Mean values differed significantly between all discharge events (pairwise Mann-Whitney U-test, p-values < 0.001,Table 10).

Table 11: Results of ANOSIM comparing deposit composition between discharge events, Global R = 0.385, p < 0.001.

Groups	R statistic	p – value	Level of significance
extreme low flow – low flow	0.030	> 0.05	n.s.
extreme low flow – snow melt	0.802	0.001	***
extreme low flow – autumn rain	0.637	0.001	***
extreme low flow – spring rain	0.452	0.001	***
low flow – snow melt	0.679	0.001	***
low flow – autumn rain	0.493	0.001	***
low flow – spring rain	0.294	0.001	***
snow melt – autumn rain	0.065	0.01	**
snow melt – spring rain	0.208	0.001	***
autumn rain – spring rain	0.046	> 0.05	n.s.

Site-specific average flow velocity and fine sediment deposition had a significant positive correlation when considering the data set ranging over all measured velocities. (Spearman's correlation test, rho = 0.40; p < 0.001, Figure 14 D). At the extremely low flows, fine sediment deposition was highly variable, ranging over two orders of magnitude from 0.048 to 4.170 kg/week/m^2 over a range of flow velocities between 0.00 and 0.46 m/s. For this small range of velocities, correlation between flow and fine sediment deposition was no longer significant (Spearman's correlation test, rho = 0.10; p > 0.05, Figure 14 D).

Fine sediment deposition rates under low and extreme low flow conditions were not constant over the development of the drought conditions (Kruskal Wallis test, Chi2 = 83.07; df = 9; p < 0.001). A decrease was observed over the months without significant precipitation (Figure 14 C), a phenomenon which was reversed during periods of increased rainfall, such as in September 2018 and the winter and spring period, which led to deposition rates similar to those in June 2018 (pairwise Mann-Whitney U-test: Jun18-Sep18, p > 0.05; Jun18-Jun19, p > 0.05; Sep18-Jun19, p > 0.05). Fine sediment deposition then decreased once again as the drought developed during summer 2019.

4.4.2 Spatial differences

4.4.2.1 Abiotic parameters

Abiotic habitat conditions were significantly different in streams of the three sub-catchments (ANOSIM Global R = 0.342; p = 0.001). Differences between BB and the sites at the MB/HB were smaller than between BB and ZB/SR but similar to the difference between MB/HB and ZB/SR (ANOSIM: BB-MB/HB: $R = 0.3$, $p < 0.05$; BB-ZB/SR: $R = 0.5$; $p < 0.01$; MB/HB-ZB/SR: $R = 0.3$, $p < 0.01$). With regard to streambed quality, a comparison of redox values in the interstitial habitat revealed significantly different suitability for juvenile FPMs between stream systems (Table 12). The BB showed the most adverse interstitial conditions, with an average redox potential of only 249 mV for all measurements, with over 75% of all measurements below 300 mV, i.e. indicating anaerobic conditions. Mean redox values in the interstitial were significantly higher in MB/HB and ZB/SR (pairwise Mann-Whitney U-test, $p < 0.001$), with 362 and 356 mV, respectively. Redox values indicated aerobic conditions for more than 70% of all measurements in the MB/HB and in more than 60% in the ZB/SR. When comparing all measured abiotic parameters between sampling sites in a PCA (Figure 16) two independent gradients were apparent: One referring to redox values and O_2 concentrations in the interstitial, negatively correlated to the delta between open water and interstitial of these parameters, and the second one orthogonally pointing in the direction of increasing temperature, pH and electric conductivity in both surface and interstitial water (Figure 16). Sites within the BB were all grouped together, commonly sharing low values for interstitial oxygen concentration and redox potential. In contrast, most sites in the MB/HB system had higher redox and O_2 values in the interstitial. A longitudinal gradient became apparent within the MB, with interstitial oxygen supply decreasing towards the downstream part until the confluence with the HB. The HB system was characterized by an outlier with extremely low interstitial redox and oxygen values at the site HB2, directly below a large fishpond facility. Sites of the ZB/SR also lined up along a longitudinal gradient, with sites in the upstream part of the ZB grouping close to the MB/HB sites, whereas the values of pH and electric conductivity increased towards sites more downstream along the course, and at the confluence with the SR. At the last sampling site SR6 (Figure 12), the average conductivity in the open water was more than 3 times higher than at ZB1. Along the MB/HB system, conductivity values showed little variation, at around 100 µS/cm, the BB showed a moderate increase from 80 to 110 µS/cm (Table 12). Sites in the upstream part of the SR had higher oxygenated interstitial than in the lower reach.

Table 12: Abiotic parameters measured in the open water (FW) and at 5 cm substratum depth (INT) during all sampling intervals in the three sub-catchments, Δ refers to the difference between FW and INT values, with negative values indicating higher values in the INT.

	Parameter	Mean ± sd	Median	Range
BB	$O_{2\ FW}$ (mg/l)	9.21 ± 1.70	8.92	1.19 – 13.21
	$O_{2\ INT}$ (mg/l)	6.18 ± 2.42	6.42	0.13 – 11.79
	Δ O_2 (mg/l)	3.03 ± 2.24	2.81	-5.21 – 9.11
	T_{FW} (◦C)	13.1 ± 5.0	14.2	1.3 – 21.2
	T_{INT} (◦C)	14.2 ± 5.2	15.8	1.4 – 22.7
	Δ T (◦C)	-1.1 ± 1.4	- 1.0	- 10.5 – 2.3
	pH_{FW}	6.9 ± 0.6	7.1	4.6 – 7.9
	pH_{INT}	6.7 ± 0.6	6.8	4.5 – 8.6
	Δ pH	0.3 ± 0.4	0.3	-1.2 – 1.8
	$Cond._{FW}$ (µS/cm)	87 ± 24	78	58 – 185
	$Cond._{INT}$ (µS/cm)	104 ± 37	99	60 – 335
	Δ Cond (µS/cm)	- 18 ± 35	- 8	- 265 – 25
	Eh_{FW} (mV)	488 ± 74	505	202 – 581
	Eh_{INT} (mV)	249 ± 91	249	- 52 – 461
	Δ Eh (mV)	239 ± 69	249	20 – 359
	Fine sediment deposition (kg/week/m^2)	1.495 ± 1.460	0.967	0.096 – 7.844
MB/HB	$O_{2\ FW}$ (mg/l)	9.13 ± 1.64	8.86	0.16 – 13.19
	$O_{2\ INT}$ (mg/l)	5.90 ± 2.59	6.07	0.00 – 11.86
	Δ O_2 (mg/l)	3.22 ± 2.29	2.72	0.14 – 10.87
	T_{FW} (◦C)	12.0 ± 4.7	12.4	0.6 – 21.5
	T_{INT} (◦C)	13.1 ± 5.1	13.5	0.4 – 23.3
	Δ T (◦C)	-1.2 ± 1.2	- 1.0	- 6.2 – 1.6
	pH_{FW}	6.7 ± 0.6	6.8	4.3 – 8.1
	pH_{INT}	6.4 ± 0.5	6.5	4.5 – 7.3
	Δ pH	0.3 ± 0.4	0.3	- 0.9 – 1.3
	$Cond._{FW}$ (µS/cm)	115 ± 42	103	66 – 297
	$Cond._{INT}$ (µS/cm)	126 ± 37	120	60 – 272
	Δ Cond (µS/cm)	-11 ± 27	- 6	- 124 - 79
	Eh_{FW} (mV)	503 ± 83	531	267 – 609
	Eh_{INT} (mV)	362 ± 121	373	- 38 – 615
	Δ Eh (mV)	142 ± 111	140	- 179 - 428
	Fine sediment deposition (kg/week/m^2)	0.781 ± 0.895	0.517	0.080 – 6.634
ZB/SR	$O_{2\ FW}$ (mg/l)	9.23 ± 1.59	9.25	5.50 – 13.10
	$O_{2\ INT}$ (mg/l)	6.96 ± 2.24	6.81	0.50 – 12.64
	Δ O_2 (mg/l)	2.26 ± 1.59	1.98	- 0.45 – 9.57
	T_{FW} (◦C)	11.9 ± 4.2	12.8	1.6 – 18.8
	T_{INT} (◦C)	13.4 ± 5.1	14.4	1.5 – 22.9
	Δ T (◦C)	-1.6 ± 1.2	- 1.5	- 6.1 – 1.2
	pH_{FW}	7.2 ± 0.4	7.2	6.3 – 8.4
	pH_{INT}	7.0 ± 0.3	7.0	6.1 – 7.5
	Δ pH	0.3 ± 0.3	0.3	- 0.2 – 1.9
	$Cond._{FW}$ (µS/cm)	155 ± 59	150	66 – 479
	$Cond._{INT}$ (µS/cm)	171 ± 67	161	69 – 473
	Δ Cond (µS/cm)	-16 ± 40	- 6	- 297 – 89
	Eh_{FW} (mV)	492 ± 86	518	38 – 628
	Eh_{INT} (mV)	356 ± 87	358	111 – 540
	Δ Eh (mV)	139 ± 92	134	- 163 – 370
	Fine sediment deposition (kg/week/m^2)	0.730 ± 1.110	0.305	0.048 – 6.300

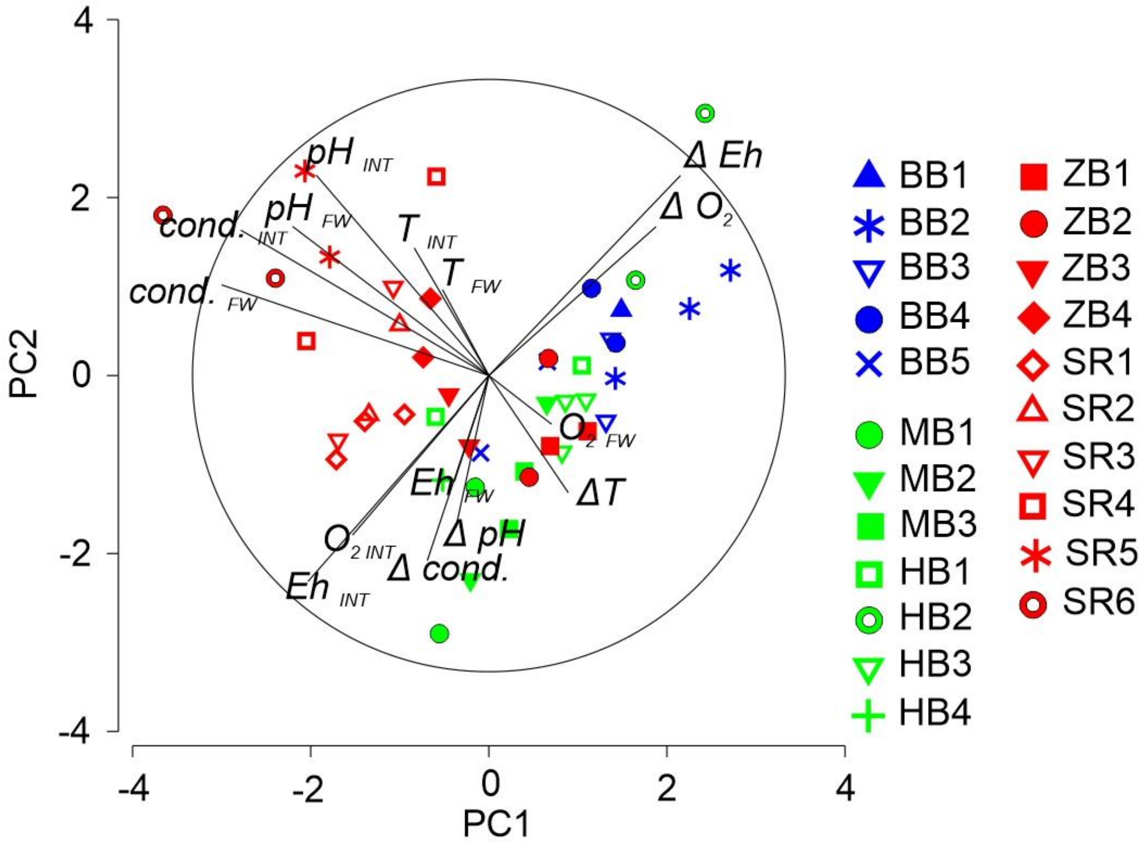

Figure 16: PCA based on Euclidean distances between sampling sites according to their normalized abiotic water parameters. Colors symbolize the three different stream systems. Vectors are based on Pearson's correlation indices showing the strength of the correlation of each variable to the first two PC axes (the circle represents 100% correlation).

Concerning the average fine sediment deposition per week, significant differences were evident between all three stream systems (pairwise Mann-Whitney U-test, $p < 0.001$, Table 12, Figure 17). The lowest mean fine sediment deposition rate was observed in the ZB/SR, followed by the MB/HB. The highest mean fine sediment deposition rate of 1.495 kg/ week/m^2 occurred in the BB where variance was also significantly higher (Levene test, $p < 0.05$).

4.4.2.2 Longitudinal patterns of fine sediment deposition

The longitudinal patterns of fine sediment deposition were completely different in the three stream systems (Figure 17): In the BB, low fine sediment deposition rates were measured close to its source. The inflow of effluents from the close-by fish farm did not cause relevant fine sediment inputs. Instead, fine sediment deposition steeply increased within a few kilometers of the point where the stream exits the forest, and flows into the more heavily used, agricultural part of the catchment. Here, several tributary inflows caused steep increases in fine sediment deposition, the most extreme effect caused by the inflow of a drainage channel at site BB4. Towards the downstream end of the stream, fine sediment deposition rates then dropped again between km 2.5 and 3.5, before increasing again at the most downstream site. The mean fine sediment deposition was most significantly impacted after the autumn rain, with a 4-fold increase compared to the low flow conditions in BB (pairwise Mann-Whitney U-test, $p < 0.001$), whereas the fine sediment deposition after the snow melt did not differ significantly from the low flow conditions (pairwise Mann-Whitney U-test, $p > 0.05$). The MB/HB system had a contrary pattern, with high fine sediment deposition rates at the very upstream part (with a maximum of 5.4 kg/week/m^2 deposited during spring rain), declining to very low weekly deposition rates at the downstream reach. A steep increase in fine sediment deposition at the first kilometer of the stream course corresponded to the inflow from a pond facility. Here, deposition rates increased by an average of 85% directly downstream compared to directly upstream, with a maximum increase of 300% during the sampling interval in November 2018. The ZB/SR system revealed the most uniform longitudinal fine sediment deposition pattern, with generally constant deposition rates along the stream course, despite multiple tributary inflows. The high increase after the inflow at ZB1 in 2019 was caused by a wild boar wallow directly next to the sediment trap. The notable increase of 1500% at the trap downstream of the most downstream site SR6 in October 2018 was the result of draining the fishponds located upstream. In contrast to the other two stream systems, fine sediment deposition in the ZB/SR system differed significantly between all high discharge events (pairwise Mann-Whitney U-test: spring rain – autumn rain: $p < 0.05$; spring rain – snow melt: $p < 0.001$; autumn rain – snow melt: $p < 0.001$), with snow melt causing highest deposition rates. This stream system was most severely affected by the prolonged drought conditions in 2019, leading to a partial drying-out of multiple river reaches.

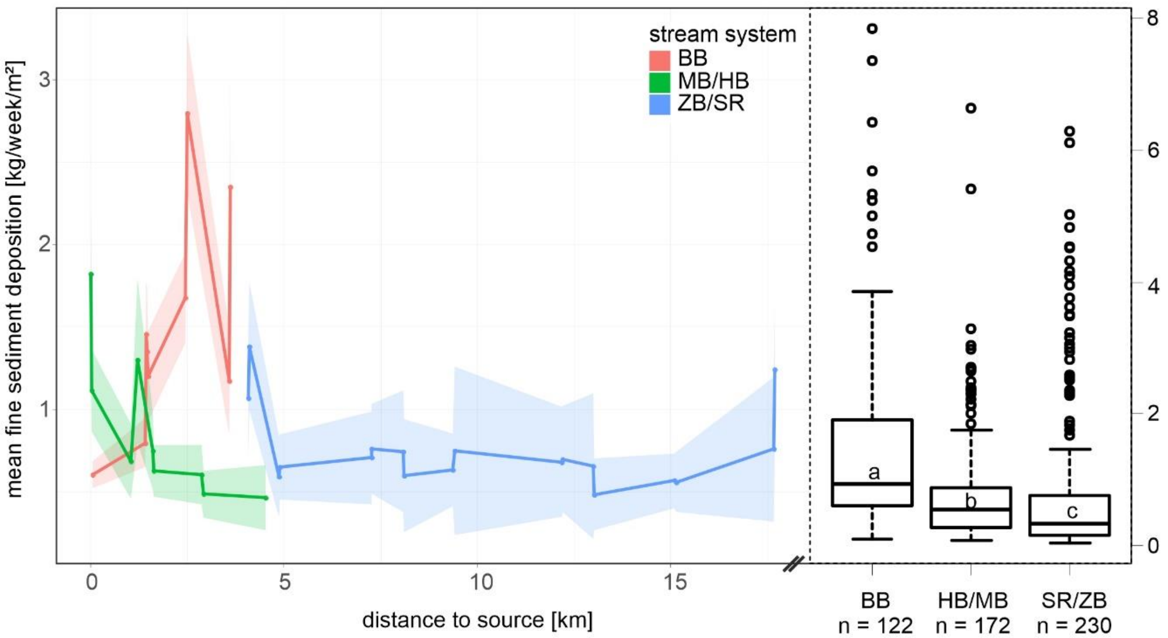

Figure 17: Longitudinal patterns of mean fine sediment deposition along the stream courses comprising data from all sampling intervals. Lines between sampling points were drawn to enhance readability; Standard errors are expressed as transparent ribbon; box and whisker plot show fine-sediment deposition in the three stream systems; different letters at the boxes represent significant differences between the arithmetic means of discharge events based on pairwise Mann-Whitney U-tests.

4.5 Discussion

The differences in the total amount and the particle composition of the sediment deposits observed after rain events and the snow melt revealed the influence of individual characteristics of events concerning erosion rates (e.g. Wade and Kirkbride (1998)) and the contribution of external or internal sediment sources, as stated by (Navratil et al. 2012). High discharges caused by precipitation yielded a higher proportion of fines deposited on the streambed compared to the snow melt, due to increased erosion rates. These external inputs mitigated the flushing effect of bed mobilization that was indicated by the higher proportion of larger grain sizes in the deposits resulting from rain events. During snow melt, higher mean and peak discharge led to an even more significant mobilization of the streambed, whereas lower amounts of catchment-derived inputs might have led to a reduced proportion of fines in the deposits. Snow melt represents an important aspect of the natural flow regime (Yarnell et al. 2010) that has not yet been considered in the context of FPM habitat quality. In light of climate change, with rising temperatures expected in future winters, decreases in snow cover might reduce the beneficial effects of snow melt (Schneider et al. 2013; Yarnell et al. 2010). A negative trend in maximal snow height as well as a decreasing number of days with fresh snow can already be observed for the study region in climatic records since the 1940s. Moreover, increasing temperatures might result in a higher amount of winter rain, changing the mode towards more external inputs due to a higher erosive power. Shifting the start of the snow melt might lead to an even longer low flow season, enhancing the negative effects for aquatic communities adapted to cool stream conditions.

Deposits during low and extreme low flows consisted almost exclusively of fine particles. Corresponding with the increasing temperatures and decreasing flow rates, oxygen concentration in the interstitial was lowest during the dry months. Multiple studies found increasing levels of fine sediment deposition under low flow conditions, with fines infiltrating into the pore spaces and covering the streambed (Buendia et al. 2014; Rovira and Batalla 2006; Wood and Petts 1999). The accumulation of fine sediments – due to decreasing transportation rates during the course of a drought – impeded the inflow of oxygen-rich surface water during a time of already low oxygen concentration, due to high temperatures and enhanced stream metabolism, representing worst conditions for juvenile FPMs. The lack of precipitation limited the supply of fine sediments emanating from external sources, which was evident from the decreasing weekly deposition rates over periods without rainfall. Stream discharge was no longer the main driver of fine sediment deposition, as is evident from the non-significant correlation with fine sediment deposition at low flows. This shifted the importance towards in-stream processes such as bank and in-channel erosion, as suggested by Fox et al. (2010) and Mosley (2015), as well as local streambed hydromorphology. An example of the disproportionate effect of local sediment sources at low transport capacities in this study was observed at ZB1 in 2019 where wild boar destroyed stream banks, which led to high deposition rates directly downstream of the wallow.

In contrast to the varying response of sediment deposition to the different discharge events, the difference between physicochemical parameters measured within the interstitial and in the open water revealed insufficient exchange between open water and interstitial habitat throughout the entire sampling period. The use of the delta values rather than looking at the absolute values was selected to evaluate the infiltration rates of surface water independently of the seasonal variations in abiotic parameters such as oxygen and pH, which are primarily velocities prevailing during the sampling intervals were apparently not sufficient to mobilize significant portions of the streambed which would be necessary to cleanse it of fines and restore it such that it is a well-oxygenated, suitable habitat for juvenile freshwater pearl mussels. This was indicated by the small proportion of particles larger than 20 mm which were deposited after the high discharge events. The co-occurrence of rain events, which increased the input of sediment from the surrounding catchment via erosion, counteracted the positive effect of sediment flushing. The crucial roles of flow regimes and internal stream processes has long been underestimated compared to catchment soil erosion (Auerswald and Geist 2018). If a flood is not severe enough to effectively remove material from the channel via overbank deposition, sediments are simply transported further downstream, and remain in the system (Pander et al. 2015). No overflown banks were observed during the sampling period.

The current substrate in the five streams did not meet the requirements for the successful development of juvenile FPMs, not even under the "optimal" conditions after snow melt, with higher discharge and higher oxygen availability due to lower temperatures and overall low stream metabolism (Denic and Geist 2015).

The different patterns observed in the three stream systems in response to the various discharge events can be linked to different catchment land use, which is believed to have distinct impact on runoff generation and sediment inputs (Heathwaite et al. 1990; Hester and Gooseff 2010; Hümann et al. 2011; Kang et al. 2001; Knott et al. 2019; Sutherland et al. 2002; Thompson et al. 2013), also reflected in biological community structure (e.g. Bierschenk et al. (2019). The discharge and fine sediment deposition in the MB/HB and BB system, both located (at least partly) within forested catchment areas, were less affected by the rain events and the snow melt. Forested areas are less susceptible to erosion due to a buffering effect of the tree canopy, and produce less input of suspended solids (Boix-Fayos et al. 2008). The ZB/SR system, lying within an open catchment consisting of grassland und crop fields, reacted quickly and strongly to runoff, and displayed the highest variability responding to the type and timing of precipitation events. These observations are in line with the common assumption that agriculturally used areas are generally more prone to erosion (Knott et al. 2019; Pimentel and Kounang 1998) and the lower infiltration capacity of cultivated soils results in the increased generation of surface runoff (Burt and Slattery 2006). Increasing anthropogenic influences along the stream course were also evident in the increase in abiotic parameters such as temperature, pH, and conductivity, following the shift in land use from extensive grassland to a higher proportion of crop production.

The observation of three clearly individual longitudinal patterns – despite the geographical proximity of the streams – supports the role of local catchment properties on fine sediment deposition and substrate quality. This highlights the need for site-specific monitoring, adapted to the research question, and the limitations of extrapolating results from a single gauge or monitoring station to whole stream systems or even catchments, as mentioned previously by Poff et al. (2006) and others (Englund and Cooper 2003; Larned et al. 2011; Poole et al. 1997; Snyder et al. 2003). Denic and Geist (2015) found a mean deposition rate in the MB of 2.6 kg/month/m^2, whereas the extrapolated average monthly fine sediment deposition rate from the present study was 4.12 kg/m^2. An explanation for this nearly two-fold increase can be the choice of sampling sites: Stoeckl et al. (2020) selected one site in the downstream part of the MB as a reference site, not accounting for the higher deposition rates in the upper part of the stream.

A special issue within the sampling region concerned the management of fish farming ponds: Most of the pond facilities did not cause large input of fine sediments during the sampling period, as indicated by comparison of deposition rates in sites upstream and downstream of the outflow. This can be attributed to the pond management cycle: For most of the year, little water is discharged from the ponds: Over the winter, most of them are either closed or even left dry. During the summer, most of the ponds have little outflow to boost fish production in warm, stagnant water which is the preferred habitat of carp. During this phase, the ponds accumulate sediments from the catchment or waste products of the fish. Exceptions were one pond facility at HB2, with the highest fine sediment deposition rates directly downstream, leading to a totally anaerobic interstitial, and at site MB2, where equally significant increases in fine sediment deposition were observed downstream of the outflow on all sampling occasions. Following the pond-management cycle, the water is released during the harvest process in autumn. Observations from the one pond fishing event sampled during the Sept 2018 interval at site SR6 revealed that the release of water led to the remobilization of the accumulated material and flushing into the SR, leading to a strong increase in fine sediment deposition directly downstream. These results support the assumption that effluents from individual fish ponds might increase fine sediment inputs into the pearl mussel streams, but they do not confirm a generally detectable impact throughout the year, either. Instead, a strong dependency on local conditions and timing was evident. The evaluation of the effect of fish ponds needs therefore to be pond-specific and include all stages of the production cycle.

4.6 Conclusion

Effective measures against streambed colmation need to consider all relevant processes which control fine sediment dynamics. There is a need to analyze the interaction of land use, climatic variations, hydrology and local hydromorphologic conditions, in order to uncover the drivers of FPM habitat degradation and to take effective conservation measures. These should depend on the current status of the interstitial habitat and the flow regime, and be adapted to the expected climatic changes. The years

2018 and 2019 foretold that this additional stress can be the tipping point for the already declining populations (DuBose et al. 2019; Hastie and Young 2003; Santos et al. 2015). A delayed or even complete failed recruitment was observed during the recent low flow periods, together with a higher mortality of adult mussels. With the predicted decreasing strength of snow fall, insufficient groundwater recharge in winter and spring, a higher probability of lower precipitation rates and higher evapotranspiration due to higher temperatures, an intact flow regime needs to be maintained to prevent the dying out of the last remaining FPM populations in the area of the present study (Geist and Hawkins 2016; Rolls et al. 2012). Groundwater abstraction for multiple human uses should be minimized and drainage systems dismantled, so as to retain water in the landscape and mitigate the negative effects of low flows. To achieve the long-term goal of restoring the natural reproduction of FPMs in these streams, a holistic approach must be applied, combining substrate restoration measures to provide suitable juvenile habitat at a local scale, with effective erosion protection and improved water retention at the catchment scale to maintain a high-quality substrate.

5. Effect of fish pond drainage on turbidity, suspended solids, fine sediment deposition and nutrient concentration in receiving pearl mussel streams

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Authors contributions: R.H. conceptualized the study and developed the methodology together with J.G., and conducted the investigations, wrote the original draft and reviewed and edited the manuscript in cooperation with J.G.; R.H. was responsible for data curation, formal analysis and visualization. For further information on the authors contributions please refer to Chapter 9. All authors have read and agreed to the published version of the manuscript.

5.1 Abstract

Extensive fish production in earthen ponds is a common aquaculture practice, which requires draining of the ponds for fish harvesting. Despite their value for biodiversity and water retention, the impact of fish ponds on the receiving streams as regards fine sediment and nutrient pollution remains controversial. This holds particularly true for streams with endangered freshwater pearl mussels, requiring a highly permeable streambed with low fine sediment content for successful juvenile development. This study quantified the amount of fine sediment, suspended solids and nutrients delivered to pearl mussel streams in relation to the pond characteristics, distance to the receiving stream and applications of measures to prevent the input of fines. Comparing fine sediment deposition above and downstream of the pond inlets after 21 pond drainage operations, as well as continuous measurements of the turbidity for 12 operations revealed varying effects of pond fishing on the receiving streams. Average fine sediment deposition was increased by nearly six-fold compared to upstream and maximum turbidity values for single drainage operations exceeded 460 NTU. Draining between 1% and 92% of the water volume of individual ponds resulted in additional loading of 0.07 - 4.6 t suspended particles. Physical mitigation structures that prevent mobilized material from reaching the receiving stream significantly reduced the fine sediment input and deposition rates. Harvesting methods that do not require complete drainage of the pond reduced the turbidity by ten-fold. Without mitigation measures, the impact of pond drainage operations on the fine sediment deposition was comparable to high discharge events. No significant increase in nutrient concentration was observed during most drainage operations. These results reveal remarkable effects of pond drainage on the aquatic

environment, as well as the possibility to minimize such impacts by switching to harvest methods that do not require complete pond drainage and installation of sedimentation structures.

5.2 Introduction

Fish production in earthen ponds has been practiced in Central Europe for centuries (Balon 1995). Major areas for the production of cyprinids include the border region of Bavaria, Saxony and the Czech Republic, where traditional, extensive pond aquaculture is common (Biermann and Geist 2019; Brämick 2020; FEAP 2017). The same region also holds one of the largest remaining freshwater pearl mussel (*Margaritifera margaritifera* L., FPM) populations in Central Europe, a species that is considered a target species for conservation of oligotrophic streams with intact streambeds (Geist 2010; Geist 2011). The current decline of freshwater mollusks (Lopes-Lima et al. 2018) in general and FPM in particular, has resulted in concern about the potential effects of earthen fish ponds on the mussel populations and their fish hosts. Juvenile FPMs burrow themselves in the streambed for a period of about 5 years where they depend on well-oxygenated substrates. This phase in their life cycle was identified as the main factor that limits their recruitment in Europe (Geist and Auerswald 2007), thus, fine sediment pollution through pond drainage could be a major problem. Similarly, the resulting colmation and clogging of interstitial spaces within the streambed can also affect the only host fish in the area, *Salmo trutta* (Geist et al. 2006; Sternecker and Geist 2010). Effluents from aquaculture facilities, particularly during the time of fish harvest requiring pond drainage, could be a source of suspended materials (Diana 2009; Kestemont 1995). In recirculating aquaculture systems, where suspended particles are almost exclusively derived from feed residues and feces, mean concentration of total suspended solids (TSS) of 35 mg/l, with a maximum concentration of about 70 mg/l can occur, when not managed for solids removal (Becke et al. 2019). Earthen fish ponds accumulate fine particles over the growing season, originating from the water supply, feed residues, fertilizers and fish waste products (Avnimelech et al. 1999). In ponds stocked with cyprinids, their dabbling feeding activities causes resuspension of the bottom soils, resulting in increased turbidity and nutrient concentrations of the water. During the production period, the impacts of stagnant warm-water ponds on the recipient streams are minimized, as long as only a little water is discharged from the pond. Under these conditions, they offer additional aquatic habitat, thus increasing local biodiversity (Lemmens et al. 2013; Wezel et al. 2013) and may act as sediment traps. Fish harvest reverses this effect, when the pond is partially or fully drained to accumulate fish at the outlet to facilitate capture. Water discharged from ponds contains considerable amounts of nutrients (Cao et al. 2007; Schwartz and Boyd 1994) and the resuspension of the pond bottom due to fish movement, fishing activities and drag of the discharging water increases the concentrations of suspended solids in the effluents (Banas et al. 2008; Lin et al. 2001; Schwartz and Boyd 1994; Szabó 1994). In previous studies, the mean concentration of TSS in pond effluents during drainage ranged from 84 mg/l (Hargreaves et al. 2005a) to 4308 mg/l (Vallod 2010), with peak concentrations of more than 40,000 mg/l (Hargreaves et al. 2005a). The impact remains highly pond-specific, depending on the management practice employed (Frimpong et al. 2004). Most studies focused on suspended solids and nutrient concentrations of receiving streams, however, the consequences on the streambed habitats, including the resulting fine sediment deposition, have not been clarified.

Although the sediment and nutrient accumulation are smaller in extensively-used ponds than in intensive aquaculture systems that enhance fish production by feeding and artificial aeration (Hlaváč et al. 2014), they might be high enough to harm the vulnerable stream systems needed for the development of juvenile FPM. Due to the high conservation value of these streams, many ponds are currently managed to reduce the output of suspended solids. These include measures recommended in the literature like physical settling structures (Schwartz and Boyd 1994; Shireman and Cichra 1994), vegetated outflow channels and the implementation of 'Better Management Practice' (BMP) including slow, stepwise drainage, drainage from the surface (Tucker et al. 2008), and the use of harvesting techniques that do not require pond drainage (Lin et al. 2001). The effectiveness of these measures compared to the 'common' practice remains unknown.

To the best of our knowledge, this study is the first to investigate the impact of pond drainage on FPM streams, including a comparison with the natural background conditions. This is rarely applied in studies of the pollution through pond effluents, yet crucial in determining the relative importance. Measures to prevent the input of suspended solids and nutrients during pond drainage are evaluated to develop management options that could minimize the negative effects. Specifically, the following hypotheses were tested

i) Pond drainage leads to increased turbidity, TSS and nutrient concentrations in the receiving streams.

ii) Drainage of fish ponds increases the fine sediment deposition rates clearly distinguishable from the background conditions.

iii) Effects of a drainage operation depend on the pond characteristics as pond size, drainage/fishing method, distance to the receiving stream and application of measures that prevent fine sediment output.

5.3 Material and Methods

5.3.1 Study sites and ponds

The impact of pond drainage was studied in five streams at the border region between Bavaria, Saxony, and the Czech Republic (Figure 18). These streams hold some of the biggest remaining FPM populations in Central Europe, thus are of high conservation value (Dobler et al. 2019; Stoeckl et al. 2020). The mean annual discharge in these small, cool, siliceous streams ranges between 0.18 and 0.90 m³/s. The populations of FPM in the streams have declined continuously due to the lack of reproduction caused mainly by streambed colmation resulting from excess fines (Denic and Geist 2015; Geist and Auerswald 2007).

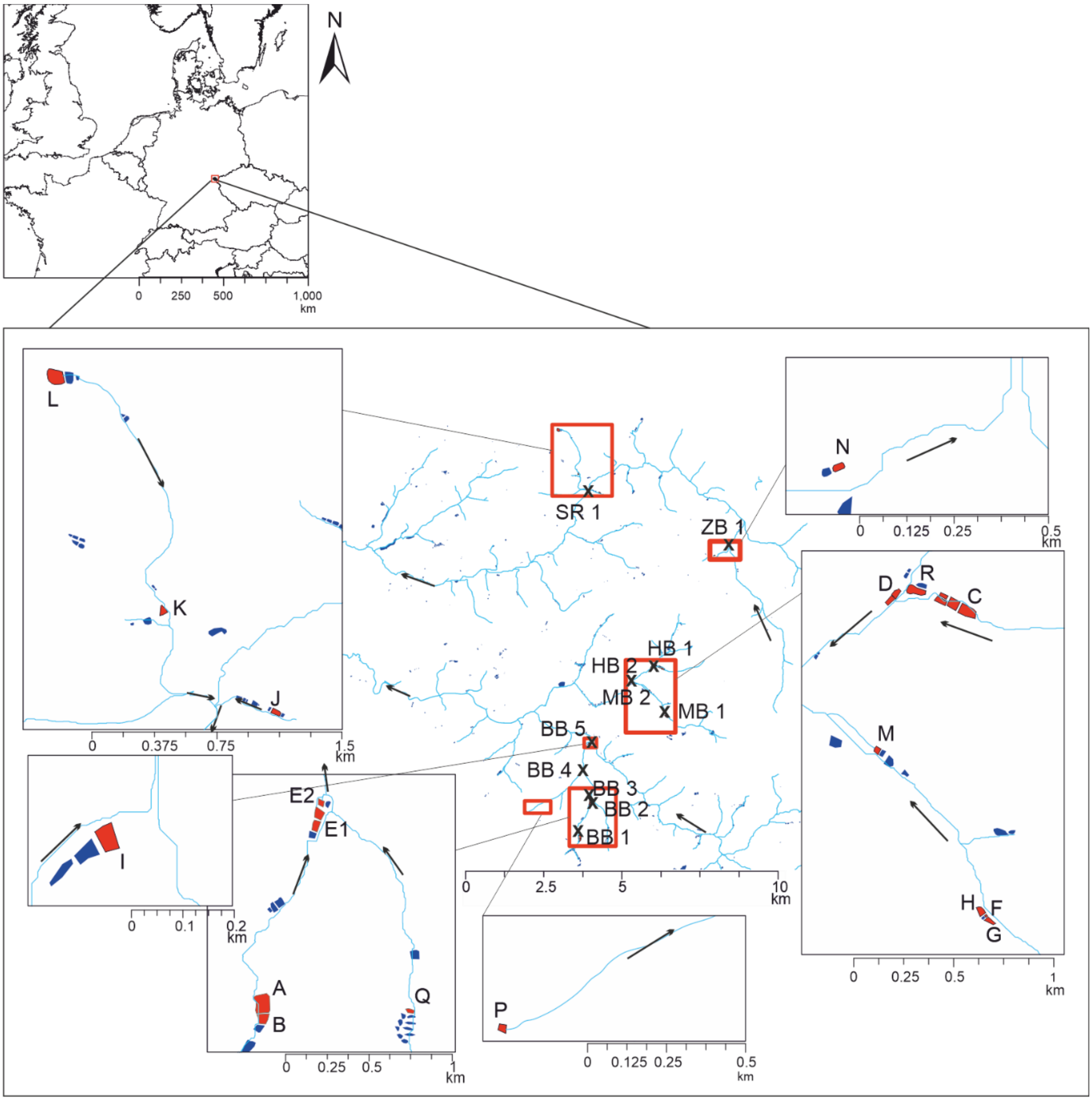

Figure 18: Map of the study area, showing the five study streams SR, ZB, HB, MB and BB. Ponds are shown dark blue, sampled ponds in red are labeled according to Table 13; sampling sites are symbolized by crosses; arrows symbolize flow direction.

In 2018 and 2019, the sediment deposition in 21 pond drainage operations was sampled at 12 sampling sites using sediment traps (Figure 18). Most of the studied ponds were extensively-used carp ponds owned by private individuals who manage the pond as a hobby, occasionally feeding some cereal or stale bread to increase the production. The size of the ponds ranged from 560 to 11,180 m^2, with an average of 3080 m^2. Ponds are usually built close to the streams and collect water from multiple sources in the catchment, groundwater sources inside the pond and through precipitation. For fish harvest in autumns, most ponds are drained completely over multiple steps. With the decreasing water levels, the fish gather around the outlets, where they are collected using dip nets. Usually, the outlet is left open during the fishing process to release the excess water. There are diverse outlet structures: Some ponds are drained through the traditional 'monk', a tower-like outlet structure containing multiple wooden boards to allow stepwise drainage, either from the lower, middle, or upper layer of the pond. Others are drained through stand pipes of various diameters (150 to > 300 mm) composed of multiple parts to allow for stepwise drainage from the surface. Several ponds are equipped with a physical settling structure such as a settling pond, an overgrown channel or a combination of both, a sediment trap, that the effluent passes before it reaches the receiving streams. The reduced flow velocity in these structures is meant to increase the settlement of suspended particles. Other methods of preventing excessive output of suspended solids included filtration of the effluent through a bale of straw fixed at the outlet structure and the use of a seine net to collect fish without complete drainage, closing the outlet during the process. The different techniques for reducing the impact of pond drainage on the receiving streams are summarized as 'suspended solids mitigation measures' (SSM). Detailed information on pond characteristics and fishing methods is given in Table 13.

Table 13: Pond characteristics of the drainage operations with observations of fine sediment deposition; 'Ds site' refers to the sampling site downstream of the inflow of the pond drainage channel into the receiving stream; 'SSM' refers to the application of measures to prevent to input of suspended solids into the receiving stream; the table is continued on the next page.

Pond	Drainage operation	Year	Ds site	Distance to ds site (m)	Area (m²)	Species	Drainage and fishing method	SSM
A	1	2018	BB1/BB2	30/1420	9040	Cyprinids	Full drain, slow, fishing behind the monk	None
	2	2019	BB1/BB2	30/1420	9040	Cyprinids	Full drain, slow, fishing behind the monk	None
B	3	2019	BB1/BB2	70/1460	3380	Trout	Full drain, slow, monk	None
C	4	2018	HB1	35	1520	Cyprinids	Full drain, slow, small tube	None
	5	2019	HB1	35	11,180	Cyprinids	Full drain, slow, small tube	None
C + D	6	2019	HB2	890	11,180 + 1200	Cyprinids	Full drain, slow, small tube	None
E1	7	2019	BB3	40	5810	Cyprinids	Full drain, monk	None
E2	8	2019	BB3	40	1950	Cyprinids	Full drain, small tube	None
F + G	9a/b	2019	MB1	45	171 + 1271	Cyprinids	Full drain, slow, small tube	Bale of straw
H	10	2019	MB1	25	1360	Trout	Full drain, slow, large tube	None
I	11	2019	BB5	130	2140	Cyprinids	Full drain, slow, small tube	None
J	12	2019	SR1	330	1290	Cyprinids	Full drain, slow, small tube	Settling pond
	13	2018	SR1	330	1290	Cyprinids	Full drain, slow, small tube	Settling pond
K	14	2019	SR1b	980	1400	Cyprinids	Full drain, slow, monk	Sediment trap
L	15	2019	SR1b	2750	8420	Cyprinids	Full drain, slow, small tube	Settling pond + sediment trap
M	16	2019	MB2	730	1190	Cyprinids	Full drain, slow, monk	Settling channel
N	17	2019	ZB1	730	560	Cyprinids	Full drain, slow, small tube	Sediment trap
P	18	2019	BB4	1990	910	Cyprinids	Full drain, fast, small tube	None
Q	19	2019	BB2 out	1490	1140	Cyprinids	Full drain, slow, small tube	Settling pond
R	20	2018	HB2	890	3460	Cyprinids	Full drain, slow, monk, seining	None
	21	2019	HB2	1000	3460	Cyprinids	Half drain, slow, monk, seining	Partial drain, seining with closed outlet

5.3.2 Sediment deposition, turbidity and nutrients

Sediment deposition into sediment traps was monitored at multiple locations along the pearl mussel streams, with a focus on the inflows of the drainage channels from the ponds. The sampling locations were equipped with sediment traps at three sites: Upstream of the confluence of the channel ('us', used as a reference), downstream of the confluence ('ds') and directly into the pond outflow close to the confluence ('out', Figure 19). Two different plastic boxes (330 x 190 x 110 mm or 160 x 190 x 90 mm depending on the channel width; Rotho Kunststoff AG, Wuerenlingen, Switzerland, see also Denic and Geist (2015); Knott et al. (2019)) filled with clean allochtonous gravel between 16 and 32 mm were buried leveled to the streambed surface and left open to collect all material deposited during one week.

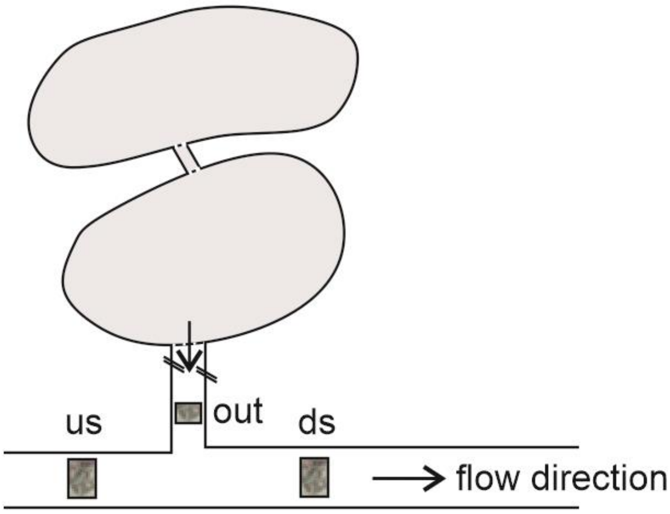

Figure 19: Sampling design for measuring sediment deposition; grey boxes symbolize sediment traps: one is installed as'reference' upstream of the confluence with the pond drainage channel (us), one directly in the drainage channel (out) and one downstream of the channel inflow (ds); arrows represent flow direction.

Sediment deposition was recorded after pond drainage in the autumns of 2018 and 2019. Additional samples were collected on multiple occasions from June 2018 to November 2019 covering mostly low-flow conditions as well as three high discharge events. These data provide the information about the natural background sediment deposition to compare the effects of pond drainage. For detailed information on the prevailing conditions during the background sampling see Hoess and Geist (2020). The deposited sediments were transferred to the laboratory to separate different grain size fractions via wet-sieving using a sieve of 0.85 mm mesh size. The fraction that was not retained by the 0.85 mm sieve was collected and is defined as 'fine sediment' throughout this paper. Particles < 0.85 mm have a harmful impact on streambed permeability (McNeil and Ahnell 1964), and consequently on biota inhabiting the streambed. These adverse effects were shown in multiple studies on the survival of salmonid eggs and larvae (e.g. Argent and Flebbe (1999); Reiser and White (1988)) that need similar interstitial conditions as juvenile FPM buried in the substrate. Moreover, previous studies on fine sediment impacts on FPM also focused on this fraction (e.g. Denic and Geist (2015)). The material was dried at 102 °C and weighed to the nearest gram. Fine sediment deposition rates are given in kg/week/m².

On 12 occasions, the pond drainage and fishing process was monitored by subsequent turbidity measurements (NTU) downstream of the pond inflow into the receiving stream every 30 min to 1 h or when a visible change in turbidity occurred. Since most of the ponds were partly drained one or two days before the actual fishing, the amount of water left in the ponds for drainage on the monitoring days ranged between 10% and 100%. Three 10 ml replicates were measured per sampling using a handheld measuring device TURB 355 T (WTW, Weilheim, Germany). The changes in water depth at the sampling locations were recorded using a measuring rod (precision 0.1 cm). Water samples of 1 L were taken before the pond drainage on the fishing day to serve as a reference and at maximum turbidity to analyze the TSS and nutrient concentrations. The water samples were frozen and stored at -18 °C before further processing. The samples were filtered through an ash-free filter paper (MN640d) with retention of particles > 2 mm (Macherey-Nagel, Düren, Germany) and dried at 102 °C. The papers were then weighed and the average weight of 10 filter papers rinsed with deionized water and dried at 102 °C was subtracted to obtain the amount of suspended particles, TSS (mg/l).

The NO_3-N, PO_4-P and NH_4^+ concentrations (mg/l) of the filtered water samples were determined via ion chromatography using two ICS 1100 ion chromatographs (Thermo Fisher Scientific, Dreieich, Germany). An AG-22 guard column followed by an AS-22 column was used to separate the anions eluted with a mixture of 1.8 mM di-sodiumcarbonate and 1.7 mM sodiumhydrogencarbonate, followed by the separation of the cations eluted in 20 mM methanesulfonic acid using a CG-12 guard and CS-12 separation columns.

5.3.3 Data analysis

Univariate statistical tests were performed in R, Version 3.6.3 (www.r-project.org, 2020). The data were tested for normal distribution and for homogeneity of variances using the Shaprio-Wilk and Levene's Tests, respectively. When both requirements were fulfilled, two groups were compared via *t*-Test and ANOVA with post-hoc Tukey-Test was performed for multi-group comparison. When the data were not normally distributed and the variances inhomogeneous, Wilcoxon-Rank Test or Kruskal-Wallis Test with post-hoc Mann-Whitney U-Test with Bonferroni correction were performed, respectively. A significance level of a < 0.05 was set in all tests. The Spearman's Correlation Test was performed to determine the relationship between the pond parameters and the environmental impacts, as well as between measured turbidities, TSS, and fine sediment deposition. To quantify the effect of pond drainage on downstream sites, delta values were computed by subtracting deposition values D_{us} from D_{ds}. High positive values indicate a remarkable increase in the fine sediment deposition below the pond inflow.

Additional loading of the receiving stream with suspended solids ($M_{surplus}$) for eight drainage operations was estimated using the Gauckler-Manning-Strickler equation, to calculate the flow (v_m, m/s) during reference conditions and pond fishing, which were then integrated over the time and relating the amount of water to the measured TSS concentrations:

$$v_m = R^{2/3} * J^{1/2} * k_{St} \qquad \textit{(Equ. 3)}$$

with R (hydraulic radius, m) calculated from the depth at the sampling time point and the wetted width, extrapolated from the relationship between the depth and the width during the background samplings via Linear Regression, J (energy gradient, m/m) determined from a digital elevation model in ArcMap 10.5. (ESRI, Inc., Redlands, CA, USA) and k_{St} (Strickler's roughness factor, $m^{1/3}$/s) estimated from field observations of the streambed, according to (Spreafico et al. 2001) and LfU-BW (2003). The resulting average flow velocities were compared to the data from the background samplings for validation. The discharge (Q, l/s) during the pond fishing process, consisting of the natural 'reference flow' of the stream and the surplus water drained from the pond, was estimated by multiplying v_m with the time (t, s) between two sampling intervals (i) which were then summed up:

$$Q = \sum_{i=0}^{n} v_{m,i} * (t_i - t_{i-1}) \qquad \textit{(Equ. 4)}$$

The amount of the 'reference flow' (Q_{ref}, in l/s) was determined by calculating the average flow under the reference conditions.

$$Q_{ref} = v_{m,ref} * (t_n - t_0) \qquad \textit{(Equ. 5)}$$

The difference between the reference and actual flows represents the amount of water discharged from the pond (Q_{pond}, l/s).

$$Q_{pond} = Q - Q_{ref} \qquad \textit{(Equ. 6)}$$

The TSS loading (M, kg) under the reference and fishing conditions was calculated by multiplying the discharge values with the respective mean TSS concentration (TSSC, mg/l).

$$M = Q * TSSC * 10^{-6} \qquad \textit{(Equ. 7)}$$

Finally, the amount of TSS under the reference conditions was subtracted from that measured during the fishing process, to gain the surplus loading.

$$M_{surplus} = M_{pond} - M_{ref} \qquad \textit{(Equ. 8)}$$

5.4 Results

5.4.1 Fine sediment deposition

The average fine sediment deposition into the sediment traps downstream of drained ponds was 2.41 kg/week/m^2, with a high variability (sd = 2.95; Table 14) and values ranging from 0.16 to 19.08 kg/week/m^2, indicating pronounced differences among ponds. In general, the mean deposition D_{out} and D_{ds} was significantly higher by a factor of almost six relative to D_{us} (pairwise Mann-Whitney U-Test, $p < 0.001$). Comparing D_{out} and D_{ds} after drainage to the background period, a significant increase after the pond fishing was evident (pairwise Mann-Whitney U-Test, $p < 0.001$). Considering the high and low background flow conditions separately, the average fine sediment deposition after drainage was nearly three times higher than that during low-flow (pairwise Mann-Whitney U-Test, p-value < 0.005). The deposition caused by the high discharge events was comparable to that caused by the drainage (pairwise Mann-Whitney U-Test, p-value > 0.05).

A significant negative relation was found between the fine sediment deposition and the distance from the pond to the subsequent sampling site, explaining 13.5% of variation (Spearman's Correlation Test, rho = 0.368; $p < 0.05$). In contrast, the pond size did not show any significant correlation with the amount of fine sediment deposited downstream (Spearman's Correlation Test, rho = 0.018; $p > 0.05$).

Another factor responsible for the site specificity was the application of SSM. Without such measures, D_{ds} was nearly three times higher (Wilcoxson-Test, W = 380, $p < 0.001$) compared to operations with SSM. The maximum value for D_{out} without SSM was about 10 times higher than that obtained when SSM was employed.

To account for the high site-specificity in regard to land use and pond management, the difference between D_{ds} and D_{us} was assessed for each drainage operation to evaluate the surplus deposition caused by pond fishing (Figure 20). This comparison yielded an average four-fold increase with SSM. In contrast, the deposition rate downstream was increased by nine-fold for operations without SSM. Under the background conditions, D_{ds} was increased by 1.7 relative to D_{us} in all sampling locations. The average delta values obtained when no SSM was applied were significantly higher compared to the delta values at both low and high discharge background conditions (pairwise Mann-Whitney U-Test, $p < 0.05$). In contrast, no significant difference was observed when comparing the deposition after operations with SSM to the background conditions (pairwise Mann-Whitney U-Test, $p > 0.05$).

Table 14: Fine sediment deposition in kg/week/m² after pond drainage and during the background sampling intervals at high and low discharge, for reference sites upstream of the inflow of drained ponds to the receiving stream (D_{us}), sites within the drainage channel (D_{out}) and downstream of the inflow to the receiving stream (D_{ds}) for drainage operations with and without SSM.

		Mean	sd	Range
Pond drainage				
D_{us}		0.41	0.47	0.11-2.20
D_{out}	With SSM	1.34	0.67	1.16 – 2.63
	Without SSM	4.01	4.56	0.43 – 19.08
D_{ds}	With SSM	0.79	0.69	0.16 – 2.31
	Without SSM	2.23	1.48	0.23 – 4.72
Background				
D_{us}	High Q	2.50	1.46	0.69 – 5.68
	Low Q	0.56	0.47	0.06 – 2.03
D_{out}	High Q	2.07	1.84	0.19 – 6.28
	Low Q	1.32	1.57	0.07 – 10.26
D_{ds}	High Q	3.07	2.24	0.63 – 7.84
	Low Q	0.75	0.63	0.06 – 2.94

Figure 20: Delta values between fine sediment deposition downstream and upstream of the inflow of pond drainage channels into the receiving pearl mussel streams after drainage of the respective ponds, labels according to Table 13. Values for drainage operations with measures to prevent suspended solids in pond effluents are displayed as yellow bars, operations without suspended solids mitigation measure are displayed as red bars. Mean delta values with standard errors are shown for the background sampling intervals under low (green bars) and high (blue bars) flow conditions at the respective sites. Positive values indicate increased values at ds sites compared to us sites (i.e. increased in fine sediment deposition in the receiving stream by the pond).

5.4.2 Turbidity, suspended solids and nutrients

Except for operation 9a and 18, the ponds were partially drained before the detailed monitoring on the fishing day. The surplus water that reached the sampling sites was between 1 and 92% of the pond volume. The results are shown in Table 15.

The release of the remaining water and the fish harvest increased the turbidity of the receiving stream compared to the reference values for all but the drainage operations 16 and 19. The maximum turbidities were attained during the final drainage and fishing steps (Figure 21).

Without the application of SSM, the highest turbidity values reflected an increase by 256.7 ± 150.9 NTU, compared to the reference conditions. One exception was operation 5, where the turbidity increased only by 4.9 NTU. The maximum turbidity of 464.6 NTU was measured during operation 18, which was more than 120 times the reference value. When SSM was applied, the maximum increase was on average 64.7 ± 83.2 NTU. As indicated by the high standard deviation, efficiency of various SSM in preventing outputs of TSS varied markedly. Draining the pond effluent through physical settling structures, caused a moderate average increase in turbidity of 14.8 NTU (Table 15). In contrast, the use of straw bales was less effective, causing mean turbidity increases of 183.0 ± 20.0 NTU. At pond R, the fishing/harvesting method was varied in the two sampling years, allowing a direct comparison of different mitigation measures at the same pond. In 2018, nearly all the water was drained before fishing with a seine net in operation 20, whereas in 2019, the last 25% of the water was retained in the pond and the outflow was closed before the seining during operation 21. This led to a decrease in the maximum turbidity by one order of magnitude from 332.0 NTU in 2018 to 32.1 NTU in 2019 (Figure 21).

The highly significant positive relationship between the maximum turbidity during the fishing operation and the fine sediment deposition (Spearman's Correlation Test, $R^2 = 0.966$; $p < 0.001$, Figure S 1) suggested that turbidity can be used as a proxy for fine sediment depositions

The TSS released during the drainage and fishing operations ranged from 37 to 1114 mg/l, representing a significant increase relative to the reference samples (Wilcoxon-Test, $W = 198$, $p < 0.05$). The trends of the TSS concentrations generally followed that of turbidity (Table 15). Correlation between maximum turbidity and TSS of corresponding water samples yielded a significant correlation coefficient of $R^2 = 0.45$ (Spearman's Correlation Test, rho = 0.67; $p < 0.001$, Figure S 1). The maximum values were attained during drainage operations 1 and 18, resulting in a maximum increase relative to the reference conditions of 780% and 8300%, respectively.

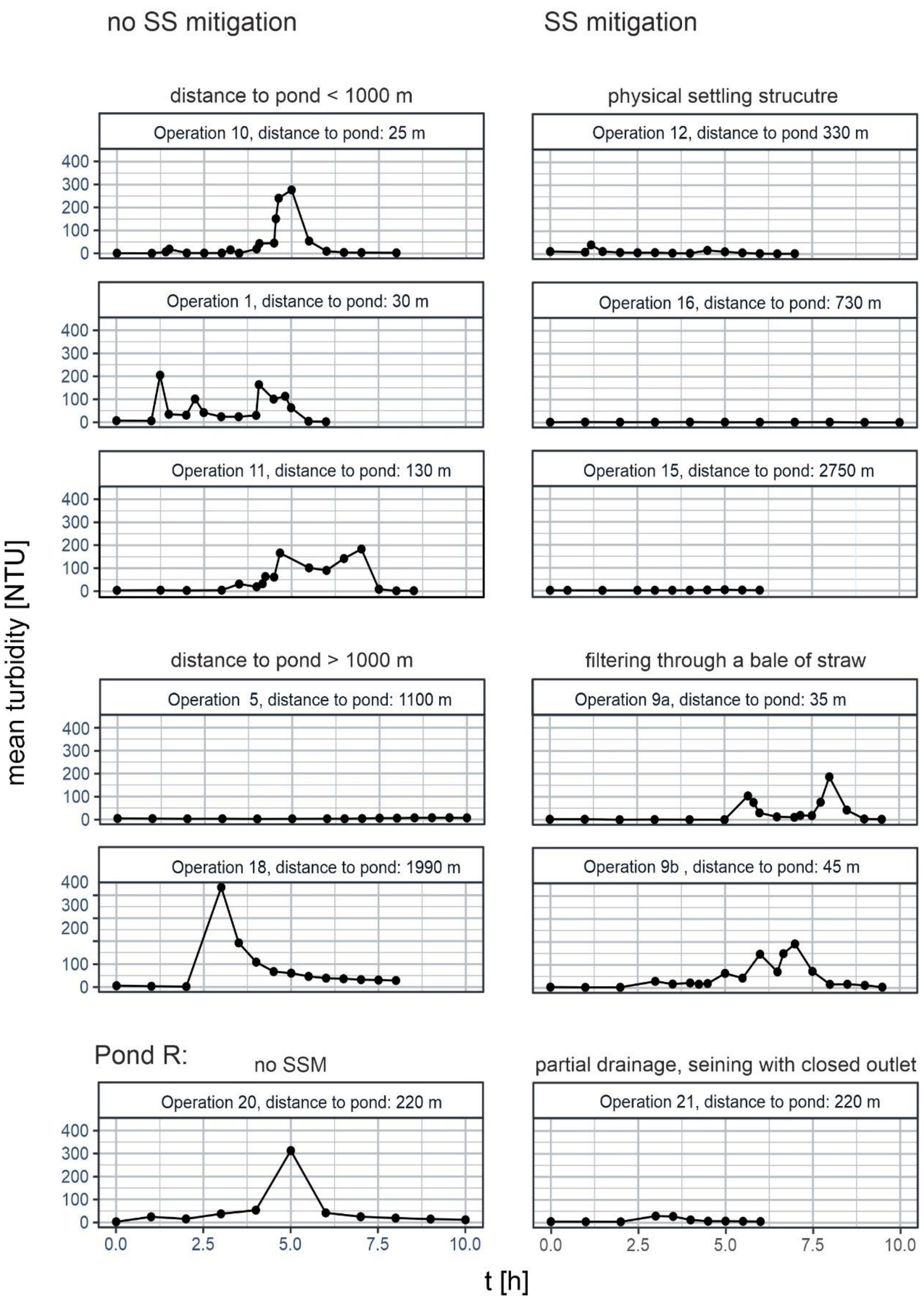

Figure 21: Turbidity values in NTU during 12 pond drainage operations; ponds on the right side were drained by applying some kind of measure to prevent input of suspended solids; operations without the application of SSM are categorizes according to the distance between pond and receiving stream, operations with SSM application are categorized according to the mitigation measure; Operations 20 and 21 at the bottom allow a direct comparison of different draining measure at the same pond (R).

The surplus loading of the receiving streams with TSS during the fishing process varied from 70 to 4600 kg of material per pond (Table 15). The maximum loadings were reached without SSM, exceeding 1000 kg in three of the five occasions. The amount of particles was strongly dependent on the proportion of water drained during the process and the pond size, as well as the distance between the pond and the sampling site. In site MB1, three different drainage operations could be compared: a very small pond was almost completely drained during operation 9a, but released only 70 kg of additional material. In operations 9b and 10, draining two ponds almost seven times as big, resulted in the release of an additional 190 and 340 kg, respectively. However, the amount of water released during operation 10 was above two times that released during operation 9b.

In contrast to the strong effects on the fine sediment deposition and turbidity, the drainage showed less pronounced effects on the nutrients (Table 15). Water release for fish harvest did not yield a significant increase in the NO_3-N concentrations in the receiving streams, as well as in comparison with the reference sample and the background samples (pairwise Mann-Whitney U Test, $p > 0.05$). On three occasions, the NO_3-N concentrations decreased during the fishing process. Nonetheless, the highest NO_3-N concentration of 16.28 mg/l corresponded to the highest turbidity measured during the operation 18. The maximum values around 7.00 mg/l in background samples were measured at the most downstream sites in the SR and BB (for background nutrient values, see Table S 2). Ammonium inputs varied highly between the different drainage operations. On two occasions, where no SSM was applied, ammonium concentrations during pond drainage rose significantly from 0.008 mg/l to more than 0.390 mg/l and from 0.001 to 0.045 mg/l. Filtration through a bale of straw did not seem effective in order to reduce ammonium input, as concentrations increased by nearly 10-fold during operation 9b. Ammonium concentrations in background samples were mostly below 0.001 mg/l, except for samples from November, were values over 0.1 mg/l could be detected at nearly all sites.

PO_4-P was detected in four water samples during the operations in BB, MB and HB with values ranging from 0.07 to 0.21 mg/l. For all other samples, the concentrations were below the limit of detection of 0.05 mg/l. Under background conditions, PO_4-P was only detected in the BB, either below the fish farm at site BB1 or below the inflow of an agricultural drainage ditch at BB3, with a mean concentration of 0.23 mg/l.

Table 15: Results of the detailed monitoring of 13 fishing operations; operation number (Op) matches the number in Table 13; the proportion of pond volume drained (V_{pond} drained) refers to the surplus amount of water that reached the sampling site on the fishing day divided by the pond volume; mean ± standard deviation (sd) is given for measurements during pond fishing and compared to the reference conditions before water release from ponds started (ref); for nutrients, maximum values reached during ponds drainage is compared to the reference conditions; Water samples to determine the TSS and nutrient concentrations were only taken for drainage operations in 2019 and only if turbidity increased substantially compared to the reference measurement.

				Turbidity [NTU]			TSS [mg/l]				NO_3-N [mg/l]		NH_4^+ [mg/l]		PO_4-P [mg/l]	
Op	Pond	SSM	V_{pond} drained [%]	Mean ± sd	Mean ref	Max	Mean ± sd	Mean ref	Max	$M_{surplus}$ [kg]	Fishing Max	Mean ref	Fishing Max	Mean ref	Fishing Max	Mean ref
1	A	No	28.4	59.2 ± 60.9	6.2	250.1	379.7 ± 250	92.2	718.9	1550	1.57	1.80	0.374	0.142	< 0.05	< 0.05
5	C	No	1.1	4.1 ± 1.6	3.3	8.2	98.2 ± 54	24.7	136.6	110	0.88	1.14	0.001	0.002	0.07	< 0.05
9a	F	Bale of straw	90.8	32.4 ± 50.4	0.9	169.8	299.9 ± 73	NA	379.3	70	2.92	NA	0.006	NA	0.21	< 0.05
9b	G	Bale of straw	21.0	46.9 ± 55.7	2.8	199.9	179.0 ± 58	142.0	219.8	190	2.40	1.97	0.088	0.009	< 0.05	< 0.05
10	H	No	48.9	45.2 ± 78.4	1.2	292.7	129.9 ± 134	86.8	323.2	340	3.05	1.91	0.389	0.008	< 0.05	< 0.05
11	I	No	21.5	54.2 ± 60.4	4.0	213.6	249.3 ± 211.2	53.5	399.1	1800	4.31	4.19	0.045	<0.001	0.13	< 0.05
12	J	Settling pond	6.2	17.9 ± 35.5	10.0	61.9	50.0	174.8	50.0	NA	3.10	2.95	0.059	0.039	< 0.05	< 0.05
15	L	Settling pond & sediment trap	4.1	2.5 ± 0.8	1.9	4.7	-	-	-	-	-	-	-	-	-	-
16	M	Settling channel	8.1	1.8 ± 0.5	1.9	2.4	-	-	-	-	-	-	-	-	-	-
18	P	No	91.7	77.7 ± 111.5	3.8	464.4	834.3 ± 395	13.3	1113.9	4600	16.28	7.09	0.002	<0.001	< 0.05	< 0.05
19	Q	Settling pond	1.7	7.8 ± 2.4	7.0	11.0	-	-	-	-	-	-	-	-	-	-
20	R	No	44.2	27.2 ± 61.5	2.7	332.0	-	-	-	-	-	-	-	-	-	-
21	R	Partial drain, seining, closed outlet	8.0	10.8 ± 9.3	4.4	32.1	39.3	0.001	39.3	950	1.24	1.32	0.016	0.014	0.09	< 0.05

5.5 Discussion

The findings of this study provide first insights into quantifying the effects of fish pond draining on the receiving streams with populations of the endangered FPM. The remarkable variations observed in the fine sediment deposition, turbidity and nutrient inputs indicate that no general ‘pond effect’ can be deduced. The strong linkages with operational procedures such as fish harvesting technique and SSM suggest that the management practice is of decisive importance in minimizing the negative effects of the ponds on the receiving streams.

As expected, most ponds did not have great impacts on fine sediment deposition during the growing season. A high variability in the background deposition could be observed between the different sampling sites, representing differences in catchment land use as well as flow conditions as described in Hoess and Geist (2020). Regarding the individual pond outflows, particularly under low-flow conditions, only small differences were observed between D_{us} and D_{ds}. In contrast, pond drainage during fish harvest released suspended solids, in particular when no SSM was applied, resulting in an average suspended particle loading of 1.7 t. The amounts discharged from these small water bodies in one day were comparable to the mean erosion rates of 1.9 t per ha estimated for Germany in one year (Cerdan et al. 2010). Since many of the ponds are located in forest areas, where erosion rates are generally low, these inputs represent a major source of fine sediment loading into the FPM streams. Although the TSS loading rates could not be estimated for the whole drainage process for all operations, they covered the most adverse steps of the procedure: Several authors suggest that the highest TSS concentration in surface drains is reached in the final step, when 20 - 40% of the suspended material is discharged with only 2 - 5% of the water (e.g. Banas et al. (2002); Vallod (2010)). These results therefore represent minimum loading rates and likely underestimate the effect of the drainage operations. Expanding the impact to warm-water pond production in general, should remind that the production methods employed in this study represent the most extensive farming method with very limited feeding. The effect of draining larger, more intensively-used ponds could be considerably stronger. The material released during the pond drainage threatens the aquatic environment, as it settles along the streambed. After pond drainage, the deposition rates were strongly increased and the effects reached up to 3 km downstream. In these precious ecosystems, the fine sediment deposition needs to be minimized, as successful FPM reproduction is associated with a maximum proportion of particles of < 1 mm of 38% (Geist and Auerswald 2007). Without applying SSM, the effects of the drainage were comparable to those of high discharges caused by rain events e when a much larger proportion of the catchment drains into the receiving streams. Moreover, the fish harvest coincides with the most critical time period in late summer and autumn: reduced precipitation and high temperatures result in minimal water levels, highest temperatures, and lowest oxygen concentrations (Denic and Geist 2015). The sudden and quick release of material accumulated in the ponds during the summer could exceed the transport capacity of the receiving stream, resulting in an enhanced fine sediment deposition, sealing of the streambed and oxygen depletion. Increased turbidities of above 1.9 NTU have been associated with FPM populations lacking recruitment for 10 years in 26 Swedish streams (Österling et al. 2010). The turbidity during pond drainage exceeded this value (up to 240-fold) when no SSM was applied. In contrast, some drainage operations with SSM only caused maximal turbidity between 2.4 and 11.0 NTU for a very short time, except for the use of the bale of straw. Also, only 4 of the 13 measurements before pond drainage were below 2.0 NTU, indicating that the turbidity already exceeds levels tolerated by FPM even without the drainage effluents. The average TSS concentrations of 250 mg/l varied remarkably over 2 orders of magnitude, with a maximum of over 1000 mg/l. Banas et al. (2008) measured mean values around 60 mg/l for the draw down step and 1150 mg/l for the final fishing step during drainage of extensive carp ponds between 2 and 620 ha, with maximum values exceeding 4000 mg/l. Another study on carp pond drainage in France reported mean TSS concentrations between 30 and 4300 mg/l during the drainage of a 24 ha pond (Vallod 2010). The comparable effect of reported TSS values and the present pond drainage operations, despite their much smaller size, show that the activities during the fish harvest rather than the pond size seem to be the main cause of high TSS concentrations in the effluents. This highlights the importance of the management practice employed during pond drainage in reducing the sediment output. This is further supported by the lack of relationship between the pond size and sediment deposition in this study.

The effluents of carp ponds can contain significant amounts of nutrients, with the potential to eutrophicate the receiving streams. However, Hlaváč et al. (2014) stated that excess nutrients result mostly from feed residues. In the extensive pond culture practiced within the study area, feed is seldom added, mostly in the form of cereal or stale bread to boost the production of natural food at the bottom of the pond. This could be the reason for the little effect of drainage on the ammonium and nitrate concentrations, whereas the three ponds where an increased ammonium output could be observed stood out due to a higher level of intensification. The life cycle assessment of the environmental impact of extensive carp pond production showed that the environmental pollution through nitrogen during the production of 1 kg of carp is 0.028 kg of N-equivalent over 3 years of production cycle (Biermann and Geist 2019) due to the extensive production method. However, since only soluble nutrients were measured, molecules attached to particles have not been considered, which might lead to underestimation of the nutrient loads (Marttila and Kløve 2012), particularly for ortho-phosphate. The detection of PO_4-P concentrations above the detection level in streams that never reached such a high concentration under the background conditions, shows the potential effect of pond water on the nutrient status of the oligotrophic FPM streams. Future studies should focus on all relevant forms of nitrogen and phosphate as the most limiting nutrient in freshwater systems that is also highly bound to suspended particles. Nutrient levels should also be monitored at a fine time scale during pond drainage, to avoid missing short term peak concentrations. The high concentrations measured in the reference samples reveal that the nitrate concentration in the streams has already exceeded the threshold values for recruiting FPM streams (Boon et al. 2019), ranging from 0.125 mg/l (Moorkens 2006) to 0.5 mg/l (Bauer

1988). Concerning ammonium, Moorkens and Killeen (2014) stated that ammonium-N should not exceed 0.01 mg/l, which was met in most of the background samples. It should be noted that the samples with ammonium values around 0.1 mg/l were all taken in late fall, after many of the ponds draining into the streams had been fished.

The results from this study reveal that physical settling structures have a great potential in reducing the amount of fine sediment delivered to the receiving streams during fish harvest. Drainage operations where effluents passed through a settling pond, a sediment trap or a vegetated channel did neither result in strong increases in turbidity or TSS compared to reference conditions, nor an increased sediment deposition relative to the background conditions. These results are consistent with the literature: Tucker et al. (2008) recommend the establishment of constructed wetlands, settling basins with a minimum retention time of 6 - 8 h or vegetated, low-gradient drainage ditches to remove large proportions of suspended particles from effluents. According to Boyd et al. (1998), settling structures might retain more than 75% of the TSS from drainage effluents of channel catfish ponds after a retention time of 8 h. Soongsawang and Boyd (2012) suggested the construction of a settling basin of 6240 m² to treat drainage effluents from a fish research station with a hydraulic retention time of 24 h. The benefits of such retention structures act not only during pond drainage but may also reduce the general impacts of pond effluents, e.g. during spilling due to high flood events (Schwartz and Boyd 1994).

To ensure the long-term functionality of such structures, a thought-out sediment removal management is needed in the long-term, as sediments would accumulate over time and reduce the efficiency to retain suspended particles. If settling ponds are drained to flush-out the material, the effects could be more intense than if the effluent had been discharged directly on multiple occasions. The settled material should be removed in a feasible time interval by measures that do not require drainage, such as mud-pumps and suction dredgers, as used in the restoration of a fish pond in the Czech Republic (Pokorný and Hauser 2002).

Although the construction of settling ponds of sufficient size is an effective SSM, this might be difficult to implement at already existing facilities. If a pond facility consists of multiple ponds, fish production could be shut down in the last pond to establish a retention pond. Where this is not feasible, seining the pond without complete drainage is another possible strategy to reduce negative impacts. Adapting to this fishing method reduced the turbidity by 10-fold at pond R. Various authors propose seining without pond drainage as the most efficient way to minimize the input of suspended solids (Lin et al. 2001; Tucker et al. 2008) without the need for (re-) construction. On the other hand, the use of straw bales to block fine particles as applied in operation 9a and 9b seems the least suitable SSM, as it resulted in turbidity values comparable to those obtained without SSM but the ponds were located further away from the receiving stream, as for operation 11.

Most of the ponds showing only small impacts of the receiving streams were at the end of longer channels, sometimes even overgrown with submerged vegetation. Particularly at a low slope, the low flow velocity and increased water residence time allows particles to settle (Hargreaves et al. 2005b). However, the drainage operation 18 demonstrates that this assumption does not necessarily hold for all situations. Although the relatively small pond was located almost 2 km away from the receiving stream, the highest impacts in all effluent parameters were recorded here. This may be attributed to the high stocking density and of the type of fishing. The pond was drained and fished on the same day in one step, without sufficient time for settling of the suspended material.

In addition to the observed effects of pond drainage, a significant fine sediment deposition takes place in the Upper Franconian FPM streams when the ponds are closed. Particularly during high discharge events, large amounts of fine particles are delivered to the streams resulting in increased deposition on the streambed which is further enhanced by altered flow regimes (Auerswald and Geist 2018). Measures to prevent inputs of fines from agricultural land use must be undertaken in addition to the adaptions concerning fish farming to sustainably restore the FPM populations. This study reveals that the construction of settling structures upstream of the inflow of highly polluted drainage channels into FPM streams could be a suitable conservation measure to reduce inputs from agricultural drainage systems that concentrate runoff from erosion prone crop fields.

6. Nutrient and fine sediment loading from fish pond drainage to pearl mussel streams – Management implications for highly valuable stream ecosystems

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Authors contributions: R.H. conceptualized the study and analyzed and interpreted the data together with J.G.; R.H. was responsible for data collection and visualization and wrote the original draft. For further information on the authors contributions please refer to Chapter 9. All authors have read and agreed to the published version of the manuscript.

6.1 Abstract

Man-made, drainable aquaculture ponds have the potential to affect the water quality in the receiving waters, but whether they act mainly as a source or sink of fine sediments and nutrients is still unclear. Particularly in oligotrophic streams containing populations of the highly endangered freshwater pearl mussel (*Margaritifera margaritifera*), even low additional inputs pose the threat of exceeding thresholds for downstream habitat quality. In this study, the effluent quality during the drainage of two extensively used cyprinid ponds with a size of 0.103 and 0.150 ha was monitored at a high temporal resolution, to characterize the nutrient and sediment loading into the receiving stream under two different management scenarios. The loading of total suspended solids (TSS) was disproportionally dominated by the final step of pond drainage during the fish harvest, when a proportion of 30% of the particles released over the entire drainage process was released with only 1% of the total water volume drained. The continuous release of the ponds' surface water resulted in an additional loading of 28.8 kg/ha of NO_3–N, 0.82 kg/ha of NH_4–N and 0.58 kg/ha of total-P that was not strongly enhanced by the fish harvest. Using a settling pond was an efficient measure to reduce the amount of suspended particles and excess ammonium and phosphorous reaching the receiving stream. Without such a measure, TSS concentrations in the receiving stream during the fish harvest were elevated to a maximum of >900 mg/l, representing a 20-fold increase compared to 45 mg/l upstream. However, about 1/3 of the released TSS were retained in the overgrown outflow ditch. The differences in loading and retention patterns of dissolved and particulate pollutants revealed the need for divergent approaches to address suspended or dissolved pollutants: Physical settling structures can be effective at reducing particulate inputs, but they might not be sufficient to mitigate the negative effects on oligotrophic streams without a specific design to

sustainably remove nutrients. This information on drainage management is not only relevant for minimizing the impacts of aquaculture ponds on downstream ecosystems, but also for the maintenance of nature conservation and flood retention ponds.

6.2 Introduction

The role of ponds in aquatic systems is often neglected due to their small size (i.e., defined as water bodies of max. 2 ha by the Pond Conservation Group (1993)) in relation to other catchment elements. Nonetheless, due to their worldwide high abundance, they contribute significantly to nutrient turnover, water supply and retention, and local biodiversity (Céréghino et al. 2014; Downing 2010; Schmadel et al. 2019). This applies to natural shallow waters and wetlands as well as to man-made ponds created to serve a certain purpose (Clifford and Heffernan 2018) such as storm water and sediment retention (Harrell and Ranjithan 2003; Tixier et al. 2011; Verstraeten and Poesen 2000), waste water treatment (Ockenden et al. 2014), and aquaculture production (Kestemont 1995; Ottinger et al. 2016; StMLU/ANL 1995). All types of ponds require a certain degree of management and maintenance to sustain their function and the services they provide. This includes sediment removal to prevent the loss of retention capacity by silting-up, or water treatment with chemical substances (Zamparas and Zacharias 2014). An important feature of artificial ponds is the possibility to drain water in a controlled manner and potentially empty the pond completely. This option of human interference is even used to differentiate natural from man-made ponds in the definition of small water bodies in Germany (StMLU/ANL 1995). Pond water is released through some kind of outlet structure, connecting the pond to the surrounding stream system. Therefore, ponds need to be recognized as integral parts of the landscape with an impact on the hydrology, as well as the nutrient and sediment dynamics of the adjacent streams (Ebel and Lowe 2013; Fairchild and Velinsky 2006). This holds particularly true for small, oligotrophic, undisturbed headwater streams with a gravel bed substrate, providing habitat for endangered species. Knowledge gaps still exist on the interaction and connectivity between different waterbodies in a landscape (Biggs et al. 2017; Sayer 2014) with regards to benefits and threats for the involved ecosystems as well as the possibilities of intervention through distinct management measures.

Ponds are often constructed to trap sediments via increased settling of particles in standing water bodies. The trapping efficiency depends on pond characteristics like size, retention time, outlet structure and age, as well as the runoff and sediment loading from the catchment (Verstraeten and Poesen 2000; Verstraeten and Poesen 2001). They are also used to treat water contaminated by dissolved nutrients or other harmful substances (Thiere et al. 2009; Thorslund et al. 2017). In particular, nitrate concentrations can be efficiently decreased via microbial activities in the anoxic layers of the pond bottom, including denitrification and dissimilatory nitrate reduction to ammonium (DNRA) (Burgin and Hamilton 2007; Céréghino et al. 2014; Erbanová et al. 2012; Hargreaves 1998). Interest is also emerging on the use of ponds to increase local biodiversity (Riley et al. 2018; Sayer et al. 2012; Williams et al. 2004), making them an important element of nature conservation. Their moderate size and manipulability makes them an ideal starting point for environmental management at a small, local scale (Schmadel et al. 2019). There is growing evidence on the need of adequate management, including frequent drainage to maximize species richness and the number of endangered species (Lemmens et al. 2013). In developing countries, integrated fish farming in earthen ponds can make an important contribution to the local supply with protein-rich food by simultaneously improving the resource use of agricultural waste products (Mathias 2006).

In addition to their beneficial functions and services, ponds may also be a source of substantial nutrient and fine sediment loads, e.g. when used for fish production. In aquaculture ponds, production is usually enhanced by high stocking rates and additional feed inputs to increase growth rates. Due to incomplete uptake of feeds (Hargreaves 1998), a considerable amount of unused nutrients can accumulate and potentially affect receiving streams. In addition to the inputs through excretion, benthivorous species like the common carp (*Cyprinus carpio*, L.) stir up the sediments while foraging, leading to increased turbidity (Scheffer et al. 2003) and causing a constant release of particle-bound nutrients like phosphorous into the water column (Huser et al. 2021). If ponds are drained for fish harvest, high amounts of the nutrient and particle-rich water are channeled to the receiving streams in a very short time, which might lead to a degradation of stream ecosystems downstream of fish ponds (Cao et al. 2007; Crab et al. 2007; Hoess and Geist 2021; Vsetícková et al. 2012).

In Germany, a significant portion of carp production occurs in Bavaria, with a high number of extensive warm water pond facilities in the northern region (Biermann and Geist 2019). These pond facilities partly overlap with catchments that hold the largest remaining populations of the highly endangered freshwater pearl mussel *(Margaritifera margaritifera* L., FPM). This species is restricted to cool, clear headwater streams with very low nutrient concentration (Denic and Geist 2015; Geist 2010) as well as minimal amounts of fine sediments in the juvenile habitat in the stream bed (Geist and Auerswald 2007). Therefore, a thoughtful management of the fish ponds is needed to mitigate any negative impact on these valuable streams.

This study focused on quantifying the pond drainage effluents released to the receiving stream with one of the last populations of the endangered freshwater pearl mussel in the area, comparing nutrient and suspended sediment loading at a high temporal resolution under two different management scenarios. This includes using a settling pond to collect effluents during a gradual, stepwise drawdown versus a rapid drawdown and release of effluents directly into the receiving stream. For both management scenarios, periods of the continuous drawdown of the water level and the final fish harvest were differentiated. The following hypotheses were tested:

i) Total suspended solid (TSS) and nutrient concentrations in drainage effluents during the drawdown step are lower than those released during the final step of fish harvest.

ii) The total nutrient and fine sediment loading depends on the management scenario – using a settling pond to treat the pond effluent results in a significant reduction of TSS and nutrients reaching the receiving stream.

6.3 Material and Methods

6.3.1 Study site

The current study analyzed the drainage of two small fish ponds within the catchment of the Bocksbach (BB), a small stream of the Elbe-catchment in northern Bavaria, Germany. Climatic conditions in the region are temperate, with an average total annual precipitation of 690 mm and mean temperatures of 6.9 ◦C. The study ponds are part of a pond chain (Figure 22) consisting of three ponds with a total catchment area of 10.1 ha. The catchment surrounding the ponds consisted of 44% forested area, 24% pasture and meadows, and 32% of arable land, used for crop and maize production. The three ponds are connected via subsurface tubes (diameter: 150 mm), so that water from the upper pond (pond A) drains into the middle pond (pond B), which then drains into the last pond (pond C), the only one with an outflow into the receiving stream. Excess water from pond C is drained through a stand tube of 150 mm diameter into an 85 m long ditch that drains into the stream (BB). The water level in pond B and pond C can be regulated at each outlet by a so called 'monk', a typical outlet structure consisting of several wooden boards that allow for a stepwise drainage from the surface (Figure 23). The pond area of pond A, pond B and pond C is 605, 1,030 and 1,500 m^2, respectively. Pond A was not stocked with fish and later used to maintain fish harvested from the two other ponds. Pond B and C were stocked with cyprinid fish at a low intensity (stocking rates of 340 and 300 fish/ ha) and fish production was extensive without additional feeding or fertilization. The ponds are drained for fish harvest once every one to two years. This represents the common practice in this region, where mostly small ponds (around 140 ponds with an area < 1 ha) are managed by individual owners, mainly for self-supply.

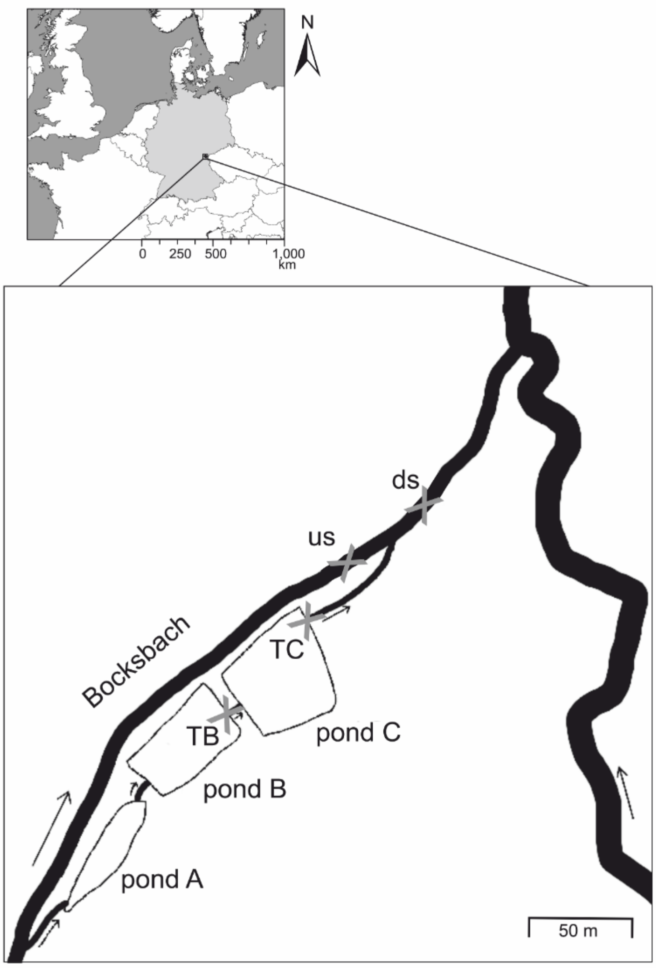

Figure 22: Location in Bavaria, Germany and map of the study site ; gray crosses represent sampling points for taking water samples, and turbidity and depth measurements: one sampling site was at the outflow of each pond (TB and TC), one sampling site upstream ('us') and one downstream ('ds') of the inflow of the ponds into the receiving stream; arrows indicate flow direction.

6.3.2 Drainage operations

Two different management scenarios were conducted for pond B and C: Drainage of pond B represented a best-management practice of slowly lowering the pond water level in multiple steps and discharging the excess water into pond C used as a settling pond, before it reached the receiving stream. In contrast, during the drainage of pond C, the water was released directly into the Bocksbach in one single step.

The drainage process started on October 19, 2020 by opening the outlet of pond B, to allow effluent water to continuously flow into pond C. Since the two ponds were connected, the drainage process was controlled by the monk at pond C: The water level was lowered in multiple steps by removing one wooden board at a time. The next wooden board was removed only after an adequate retention time was given for the incoming water from pond B. This procedure corresponded to the "stepwise drawdown into a settling pond" scenario. A sufficient reduction of the water level in pond B was reached on October 21 after 45 h and the fish were harvested from pond B. After that, the outlet of pond C was opened by removing all remaining wooden boards from the monk to increase water discharge. The drainage of pond C was stopped during the night of October 21 after 11:00 p.m. to prevent an early drop of the water level during the night to minimize stress for the fish. Therefore, until 08:00 a.m. on October 22, no water was released from the pond. On the morning of October 22, the remaining wooden boards were removed again to drain the excess water until a sufficiently low water level for the fish harvest was reached after 70 h. This procedure corresponded to the “rapid drawdown and direct release” scenario. The fish were harvested in both ponds using a seine net dragged through the nearly empty pond (about 12% of the water left) to gather the fish near the outlet and collect them with dip nets. No rainfall occurred during the entire study period. A second pond fishing event was noticed 2.85 km upstream of the study facility during the late afternoon hours on October 21.

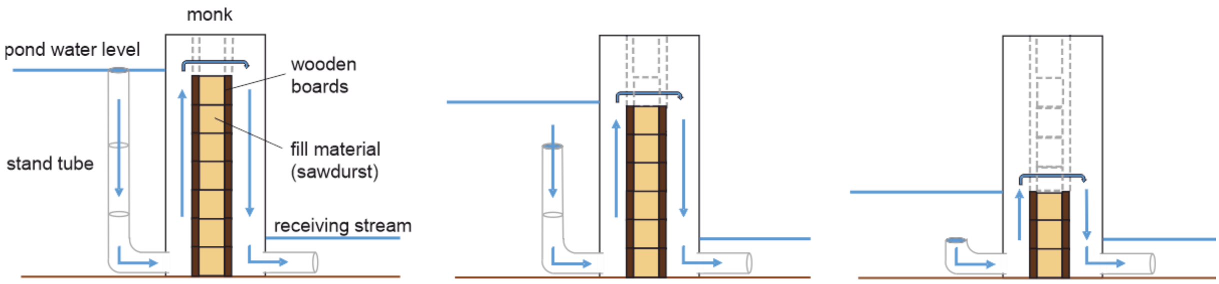

Figure 23: Control of the pond water level and stepwise drainage using a so-called ‘monk’ at the outlets of pond B and C: surface water is released through a stand pipe, the water level is controlled by the number of wooden boards (sealed with a filling material like sawdust); for pond drainage, first a part of the stand tube is removed, a stepwise decrease of the water level is controlled by the monk by removing one wooden board at a time; the water level in the pond will represent the water level inside the monk.

6.3.3 Sampling design

Measurements were taken at four different sampling sites (Figure 22). Water samples (1 l) were collected from the effluent of pond B (TB) and pond C (TC), and at two sites in the receiving stream: site ‘us’, located upstream of the outlet ditch, about 200 m downstream of the water inlet into pond A, was used as reference site for the stream baseline values and quality of the ponds inflow water (for a comparison of values from the inlet of pond A and ‘us’, see Table S 3). Site ‘ds’ was located downstream of the inflow of the outlet ditch and 40 m downstream of ‘us’. Sampling occurred at the following time steps: One sample was taken before the outflow was opened for the first time. After a drainage step was initiated by manipulating the outflow tube or removing a wooden board, samples were taken manually every 5 min for the first hour, then every 30 min. This period of constant water release from the surface was defined as the “drawdown step”. For both drainage operations, samples were also taken every 5 min during the release of the last 10% of the water volume drained and the fish harvest, which was defined as the “fishing step”. From all water samples, the turbidity (in NTU) was measured on-site from three 10 ml replicates using a handheld measuring device TURB 355 T (WTW, Weilheim, Germany), whereas the remaining sample volume was frozen and stored at -18 °C until further processing in the laboratory. An automatic sampling device asp-port a (Endress + Hauser, Reinach BL, Switzerland) was used to collect samples at the 'ds' site during night times. Additionally, dissolved oxygen (in mg/l), pH, electric conductivity (in µS/cm), water temperature (in °C) and turbidity (in NTU) at the 'ds' site were measured with an automatic multi-probe MPS-D8 (SEBA Hydrometrie GmbH & Co. KG, Kaufbeuren, Germany) every 15 min during the whole process.

The water levels in pond C as well as at the 'us' and 'ds' sites were recorded using measuring rods (±0.1 cm) installed at the pond outflow (deepest part) and mid-stream of the respective stream sites with every measurement.

6.3.4 Sample processing

Three 15 ml aliquots from each water sample at each time point were used for the analyses of nitrate (NO_3–N), ammonium (NH_4–N), total phosphorous (TP) and orthophosphate (ortho-PO_4-P) concentrations (in mg/l). TP was measured photometrically after a digestion of the unfiltered sample with sulfuric acid and peroxodisulfate to convert all the present phosphorous compounds into orthophosphate. The digested samples were then treated with molybdate ions in sulfuric solution to form molybdophosphoric acid, which was further reduced using ascorbic acid to form phosphomolybdenum blue, which was determined using a PhotoFlex Turb portable photometer (WTW, Weilheim, Germany), according to USEPA Standard Methods (USEPA 1978) NH_4–N, NO_3–N and ortho-PO_4-P of filtered samples were measured using Ion Chromatography with modifications from Hautman and Munch (1997), using a Metrohm IC 861 with a suppression unit (suppression solution consisting of deionized water, 0.1 mol/l H_2SO_4 and 0.02 mol/l $C_2H_2O_4$) and an eluent from 1.8 mmol/l Na_2CO_3; 1.7 mmol/l $NaHCO_3$ for the anions and a Metrohm IC 761, with the eluent consisting of 0.7 mmol/l Dipicolin; 1.7 mmol/l HNO_3 for NH_4–N (Metrohm AG, Herisau, Switzerland).

To obtain TSS concentrations in relation to turbidity, 100 ml aliquots of 97 samples, selected based on their turbidity to reflect the whole range of turbidities obtained from the various sampling points (including low values during the drawdown steps, as well as all samples during the fishing steps), were filtered via vacuum filtration through dried and pre-weighed MN 640 d ash-free cellulose filter papers (retention capacity >2 µm, Macherey-Nagel, Düren, Germany) and dried at 102 °C. The filters were then weighed again and the initial filter weight subtracted to determine the amount of material retained. Following, the filters were combusted at 500 °C and the remaining ashes was weight again to obtain the amount of inorganic material. The amount of organic particles (oTSS) was calculated from the weight difference before and after combustion. The proportion of organic carbon was calculated as % loss on ignition.

6.3.5 Data analysis

The TSS values for the remaining samples were estimated by relating them to the maximum turbidity values ($TURB_{max}$ measured at that time point via linear regression ($F (1, 74) = 240.3$; $p < 0.001$; $R^2 = 0.76$), yielding the following equation:

$$TSS = 2.9962 * TURB_{max} + 12.01 \qquad \textit{(Equ. 9)}$$

To assess the effect of effluent inflow on the receiving stream compared to the baseline conditions, the values measured at the 'us' site at a specific time point were subtracted from the values of the 'ds' site. Positive delta values indicated an increase below the inflow of the effluent, whereas negative delta values indicated a decrease due to dilution by the effluent.

The volume of water drained from pond C during subsequent time steps was calculated from the depth difference multiplied by the pond area. To assess the flow rate, this volume was related to the time between the two depth measurements. The average TSS and nutrient concentrations in mg/l between the two measurements were then multiplied by the amount of water drained, to calculate the TSS and nutrient loadings to the receiving stream from the pond.

The surplus water and the amount of TSS and nutrients reaching the receiving stream were estimated as described in Hoess and Geist (2021) based on the discharge, obtained using the Gauckler-Manning-Strickler equation ((Equ. 10), and the water level increase through the effluent inflow, and comparing the measured TSS and nutrient levels at 'ds' to those taken at 'us'.

$$v_m = R^{2/3} * J^{1/2} * k_{St} \qquad \textit{(Equ. 10)}$$

The Gauckler-Manning-Strickler equation was used to calculate the flow (*vm*, in m/s) based on the hydraulic radius, *R* (in m), the energy gradient, *J* (in m/m) and the Strickler's roughness factor, *kSt* (in $m^{1/3}/s$).

All the data were processed in Microsoft Excel (2016) and univariate statistical analysis were performed in Rstudio (Version 1.2.5042, http ://www.rstudio.com/, 2020). The data were tested for normal distribution and homogeneity of variances using the Shapiro-Wilk and Levene's tests, respectively, followed by *t*-test for two-group and ANOVA with a post-hoc Tuckey's test for multi-group comparison if both requirements were fulfilled. The data that did not follow normal distribution with inhomogeneous variances were tested using the Wilcoxon- Rank test or Kruskal-Wallis test with post-hoc Mann-Whitney *U* test with Bonferroni correction, respectively. A significance level of $\alpha < 0.05$ was set for all tests.

6.4 Results

Mean dissolved oxygen of 10.5 mg/l (9.9–11.0 mg/l), mean pH of 7.0 (6.9–7.1), mean electric conductivity of 113.5 µS/cm (102.8–122.0 µS/ cm) and mean water temperature of 8.0 °C (6.2–10.7 °C) were measured at 'ds' over the entire sampling period. Turbidity, TSS, oTSS, NO_3–N, NH_4–N, TP and ortho-PO_4-P measurements taken during the drainage process at all sites are summarized in Table 16.

Table 16: Summary of the measurement results during drawdown (values directly after an outlet manipulation were excluded) and fish harvest (fishing) at the sites 'us' (baseline values above the effluent inflow, values during elevated turbidity due to pond drainage upstream were excluded), TB (effluent from pond B), TC (effluent from pond C), and 'ds' (below the effluent inflow, fish harvest includes only data for TC, since turbidity values, TSS and nutrient concentrations at 'ds' did not differ between the drawdown and the fishing in TB); data for TP censored with a detection limit of <0.05 mg/l, with (fraction (%) <0.05) in brackets.

		us	TB		TC		ds	
		baseline	drawdown	fishing	drawdown	fishing	drawdown	fishing TC
Turbidity	Mean	7.2	4.16	493.8	4.6	524.0	7.8	35.7
(NTU)	Min	3.9	4.16	21.7	2.2	47.9	3.6	6.4
	Max	12.2	4.16	>1000	17.33	>1000	34.0	179.9
	n	32	1	9	57	11	97	20
TSS (mg/l)	Mean	47.8	26.5	3533.0	27.8	3944.7	36.5	116.3
	Min	24.4	26.5	75.2	19.7	176.7	21.4	31.9
	Max	130.3	26.5	19,134.9	71.0	10.507.6	118.9	912.3
	n	22	1	9	58	11	97	20
oTSS	Mean	14.7	-	2740.0	5.7	3126.2	10.6	96.0
(mg/l)	Min	1.0	-	70.8	0.0	97.6	0.0	6.5
	Max	30.2	-	14,525.9	20.9	8068.2	44.3	726.9
	n	15	0	9	9	11	9	15
Prop.	Mean	30.7	-	80.9	13.9	76.1	18.9	49.5
organ. C	Min	3.0	-	75.9	0.0	55.2	0.0	9.1
(%)	Max	59.0	-	94.1	38.1	81.2	51.9	80.7
	n	15	0	9	9	11	9	15
NO_3-N	Mean	7.916	5.356	3.633	4.422	2.448	7.683	9.101
(mg/l)	Min	6.343	5.356	1.932	1.552	0.901	4.620	7.972
	Max	9.283	5.356	5.966	8.331	3.179	12.676	12.141
	n	14	1	9	48	11	66	20
NH_4-N	Mean	0.126	0.122	0.749	0.097	0.722	0.054	0.075
(mg/l)	Min	0.000	0.122	0.456	0.000	0.164	0.000	0.000
	Max	0.287	0.122	0.949	0.326	1.892	0.691	0.209
	n	14	1	9	48	11	66	20
TP (mg/l).	Mean	0.07	<0.05	0.50	0.16	0.31	0.08	0.09
censored	Min	< 0.05 (35%)	<0.05	<0.05 (20%)	<0.05	0.10	<0.05	<0.05 (15%)
(prop. <	Max	0.13	(100%)	1.94	(52%)	0.55	(50%)	0.23
0.05 mg/l)	n	14	<0.05	9	2.06	11	0.56	20
			1		48		62	
ortho	Mean	0.033	0.000	0.017	0.063	0.028	0.031	0.023
PO_4-P	Min	0.000	0.000	0.000	0.000	0.004	0.000	0.000
(mg/l)	Max	0.139	0.000	0.061	1.435	0.101	0.340	0.234
	n	14	1	9	48	11	66	20

In measurements obtained in the receiving stream at 'us', upstream of the pond facility, turbidity ranged from 3.9 to 12.8 NTU with a mean of 7.2 NTU. TSS concentration ranged from 24.42 to 130.27 mg/l, with an average of 47.82 mg/l. Combustion of these samples yielded an average proportion of organic carbon of 30.7%. Concerning nutrients, the average NO_3–N concentration was 7.92 ± 1.11 mg/l, NH_4-N was 0.13 ± 0.10 mg/l, and average TP and ortho-PO_4-P concentrations were 0.07 ± 0.03 and 0.03 ± 0.05 mg/l, respectively.

Drainage of a pond facility 2.85 km upstream of the study site on October 21 yielded mean turbidity values of 30.7 ± 3.6 NTU, with values ranging from 26.7 to 36.6 NTU over a period of 4 h at the 'us' site (Figure 24). These values were excluded from the 'us' dataset.

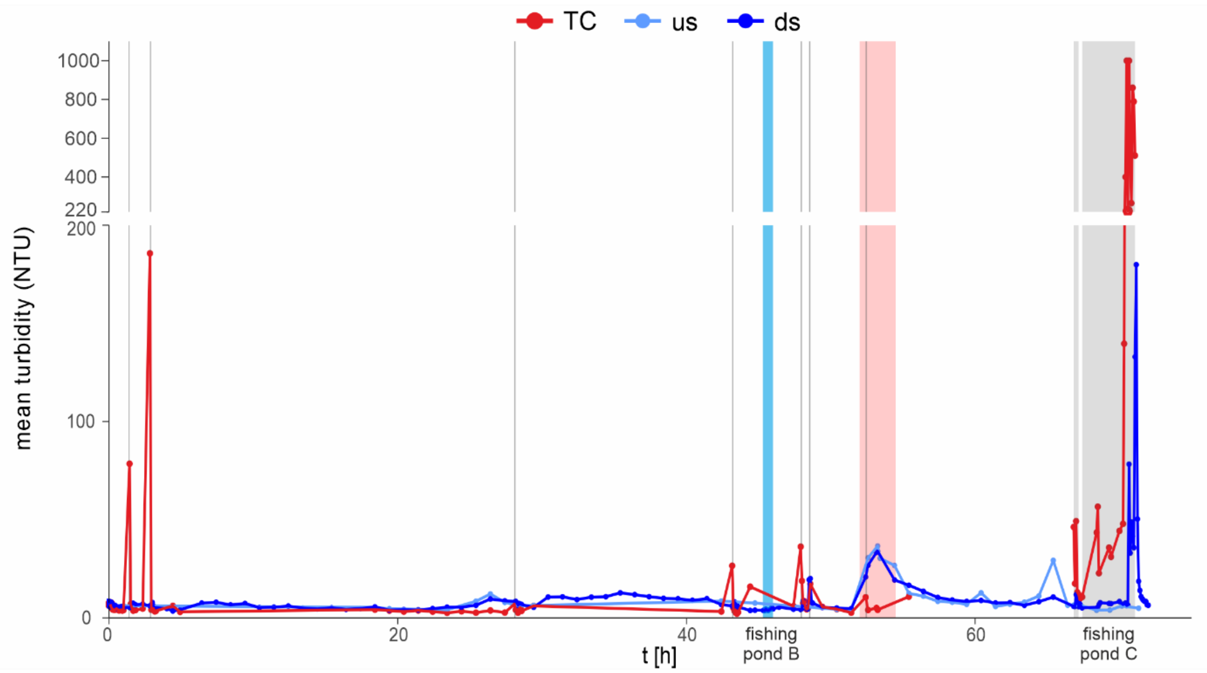

Figure 24: Mean turbidity in the effluents of pond C (TC) and at the sampling sites upstream ('us') and downstream ('ds') of the effluent inflow over the entire drawdown and fishing process ; to better resolve the low values during the first 70 h, the lower part of the graph has been magnified, values are <200 NTU; dots represent individual measurements, the gap in the connecting line for TC marks the night from Oct. 21 to 22, when the monk was closed to prevent premature drainage overnight; gray areas mark manipulations at the outlet and final fishing at TC; the blue area marks fishing at TB; the red area marks the period when measurements were affected by fishing further upstream.

Comparing these data to the values measured in the effluent of pond C revealed significant differences for nearly all water quality parameters (Table 16, Figure 25). Mean values for turbidity, TSS and NO_3-N concentrations were significantly lower in the drawdown effluents at TC than at the 'us' site (turbidity: Wilcoxon test, p-value < 0.001; TSS: Wilcoxon test, p-value < 0.001; NO_3–N: *t*-test, p-value < 0.001). Average NO_3–N concentrations of 4.42 mg/l in the TC samples were nearly two times lower than in the receiving stream at the 'ds' site. The amount and the proportion of organic particles, NH_4–N, TP, and ortho-PO_4-P during the drawdown did not differ significantly at TC and at 'ds' in the receiving stream (pairwise Mann-Whitney *U* test, p-values < 0.05). Nonetheless, short term peaks in turbidity (Figure 24), TP, ortho-PO_4-P, and NH_4–N values could be observed at TC after outlet manipulations to initiate the next drawdown step, reaching maximum values of 185.7 NTU and 0.12 mg/l, 0.34 mg/l and 0.47 mg/l, respectively. These increases were rarely strong enough to still be detected at 'ds' (Figure 26).

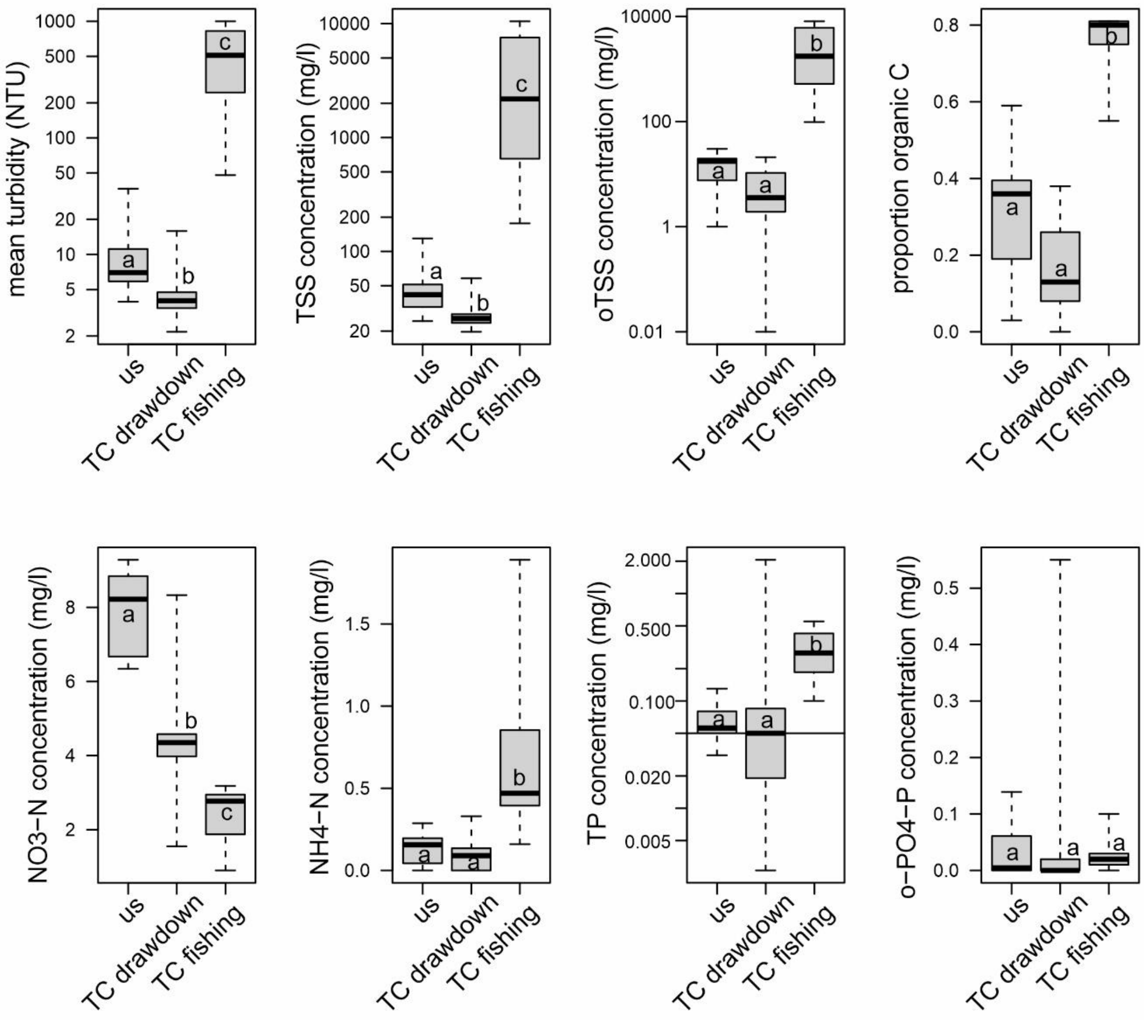

Figure 25: Box-Whisker-Plots of the measured parameters at the point upstream ('us'), in the effluents during drawdown (TC drawdown) and during fish harvest of pond C (TC fishing); data for TP as censored boxplot to account for the detection limit of 0.05 mg/l, indicated by the horizontal line; logarithmic y-axes of mean turbidity, TSS, oTSS and TP concentration for better differentiation; different letters represent significant differences.

Contrary to the minimal effects during drawdown, fishing in both ponds caused strong increases in the pond effluents in all variables except for ortho-PO_4-P and NO_3–N. Effluent quality in pond B and pond C did not differ significantly between the two fishing events for all variables (pairwise Mann-Whitney *U* test, p-values > 0.05), except for NO_3–N, which was significantly higher at TB than at TC (pairwise Mann- Whitney *U* test, p-values < 0.05). The maximum values of TSS at TB of about 20,000 mg/l were double the maximum of 10,500 mg/l reached during the fishing of pond C. Compared to the baseline values in the receiving stream, the average turbidity of pond effluents during fishing of 510.4 NTU (based on all individual measurements from TB and TC) was more than 70-fold that of the value of 7.2 NTU at ‘us’. During both fishing events, turbidity exceeded values of 1,000 NTU in some of the effluents. TSS concentrations were increased by 74-fold and 83-fold in the effluents at TB and TC, respectively. They consisted of 81 and 76% of organic particles, whereas the average proportion of organic carbon at ‘us’ was only 30%. The concentrations of NH_4–N and TP were also elevated in the fishing effluents at TC, by about 6-fold and 4-fold, respectively. In contrast, average NO_3–N concentrations in the effluents of 2.5 mg/l at TB and 3.6 mg/l at TC were lower than at ‘us’, where average concentrations of 7.9 mg/l were obtained.

Despite the highly elevated turbidity, particle and nutrient values at TB during fishing, these effects could neither be detected at TC nor in the receiving stream at ‘ds’ (Figure 24). In contrast, during the fishing in pond C, the particles and nutrients released could be detected at ‘ds’ with some delay. Strongly elevated values of turbidity, TSS, NH_4–N, TP, and ortho- PO_4-P were measured during the entire fishing process, lasting 50 min.

In the receiving stream, NH_4–N and ortho-PO_4-P concentrations at the ‘ds’ site increased significantly 15 min after the activities in the pond started, whereas increased TSS and TP concentrations, as well as turbidity, were detected only after 25 min (Figure 24 and Figure 26). NH_4–N, and TP, TSS and turbidity values remained elevated even after the monk was closed after 50 min, reaching their peak values of 0.209 mg/l, 0.23 mg/l, 912.3 mg/l and 179.9 NTU after 50 and 55 min, respectively. It took another 50 min for turbidity, TSS and NH_4–N values at the ‘ds’ site to return to their original levels.

Impacts on the receiving stream concerning dissolved substances were highly variable (Figure 26), depending on the nutrient and timing. The impact of the fish harvest on dissolved substances was not as pronounced as for the particulate substances. NO_3–N concentrations at ‘ds’ were mostly lower than at ‘us’, probably due to the lower concentration in the effluent. While differences between the ortho-PO_4-P concentration at ‘us’ and ‘ds’ were negligible during most of the fishing period, besides one peak after 15 min, the TP concentrations at ‘ds’ were clearly increased after the onset of the fishing in TC, reaching up to 0.23 mg/l. In contrast, ‘us’ values were below the detection limit of 0.05 g/l for the entire period. Similar effects could be observed for the NH_4–N concentration, which was elevated up to one order of magnitude at ’ds’ compared to ’us’ (Figure 26).

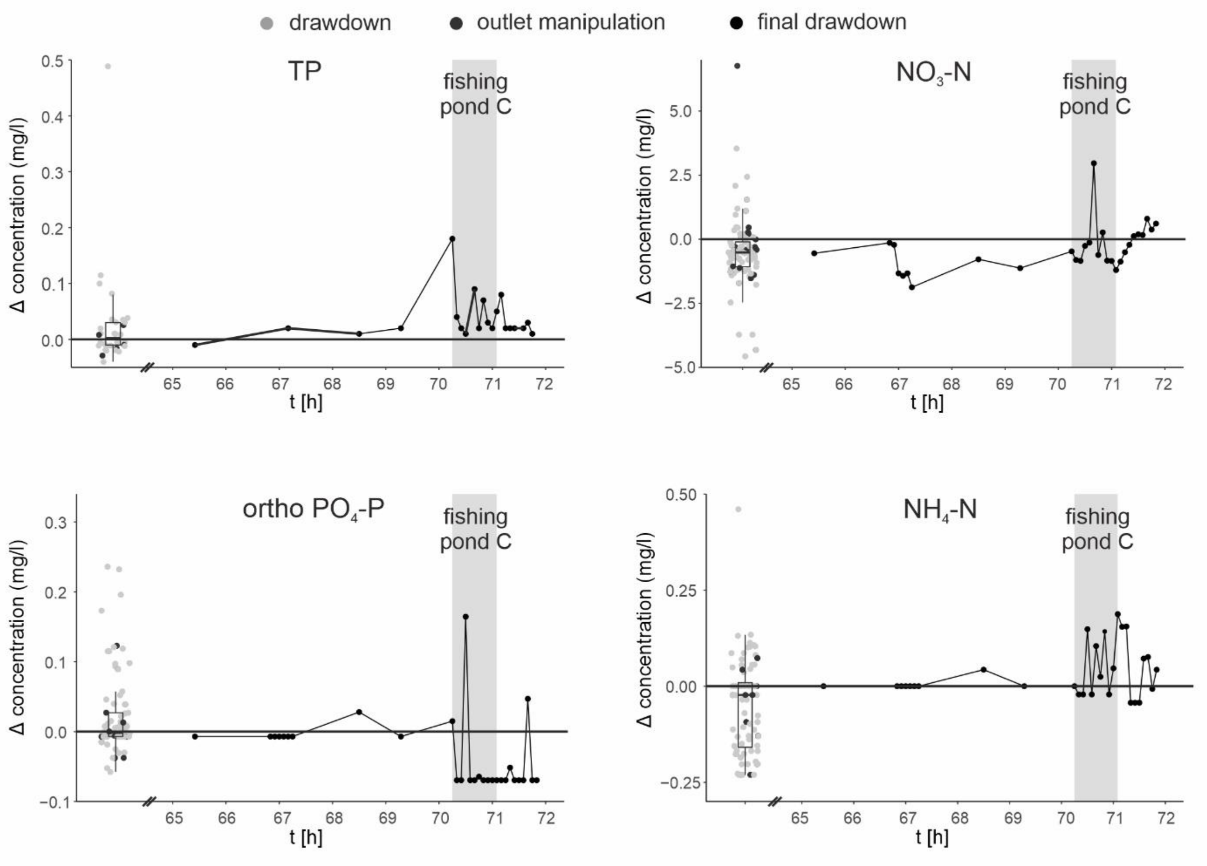

Figure 26: Differences in nutrient concentration between samples taken downstream and upstream of the effluent inflow (Δ 'ds' – 'us') during the whole process, the values of the first three days are summarized as Box-Whisker-Plots, with jittered points representing individual samples; light gray points indicate sampling during the continuous drawdown, dark gray points indicate samples taken after manipulation of the outlet. Data for the last day and fish harvest at pond C is shown in detail over time; black, connected points indicate samples taken after the start of the final drawdown; the gray bar represent the time periods of fishing activities at pond C.

In order to calculate the additional load on the receiving stream caused by the drained pond water, the measured concentrations at the respective time points were related to the drained water volume. The total volume of water drained during the 72-h process was estimated to be 1,090 m³ (Table 17). The amount of water drained during the actual fish harvest was less than 1% of that total volume. A total of nearly 93.5 kg of suspended sediment was released from pond C, of which about 60% was released during the drawdown process and 30% was released throughout the last 1% of the water drained during fishing. Of these approximately 31 kg released from pond C, only slightly less than 70% reached the 'ds' site, with the remainder being retained in the outlet ditch. Different patterns were observed for soluble nutrients: They were released constantly and proportionally to the amount of water; therefore, most of the input here (93–98%) occurred during drawdown. Also, no significant reduction of the additional load due to retention in the outlet ditch was evident for the soluble nutrients. In total, nearly 4.32 kg NO_3–N, 0.12 kg NH_4–N, 0.09 kg TP (of which 0.01 kg ortho-PO_4-P) were released during drawdown and fishing of the two ponds, which corresponded to loading rates of 28.8 kg/ha NO_3–N, 0.82 kg/ha NH_4–N, 0.58 kg/ha TP and 0.09 kg/ha ortho-PO_4-P.

Table 17: Amount of material released from pond C during the entire drainage process and divided between drawdown and fishing processes, as well as the additional load detected in the receiving stream at site ds during fish harvest at TC.

	Water (m^3)	TSS (kg)	oTSS (kg)	NO_3-N (kg)	NH_4-N (kg)	TP (kg)	Ortho-PO_4-P (kg)
Total	1,090.7	93.48	–	4.32	0.12	0.087	0.0133
Drawdown process	1,081.8	62.06	–	4.25	0.11	0.082	0.0128
Fishing process	8.9	31.43	26.06	0.07	0.01	0.005	0.0005
Surplus reaching ‘ds’ during fish harvest of TC	8.0	23.16	17.41	0.05	0.85	0.004	0.0001

6.5 Discussion

The results of this study indicated different loading patterns for particulate and dissolved substances released during pond drainage. In aquaculture ponds, elevated TSS concentrations are common, originating from feed residues and fish feces, as well as from fish re-suspending settled particles during foraging (Avnimelech et al. 1999; Badiou and Goldsborough 2015), particularly at high feed input and stocking rates (Vseticková et al. 2012). Natural fish ponds also accumulate inputs from external sources such as eroded soil particles and nutrients delivered during runoff events (Willis et al. 2010), depending on the local land use and slopes (Knott et al. 2019; Zhang et al. 2020), as well as on the shoreline length and catchment size of the pond (Brainard and Fairchild 2012). However, in the two study ponds, the mean TSS concentration during the drawdown step was only around 25 mg/l and slightly lower than in the receiving stream upstream of the ponds. This indicated that at low stocking and production intensities, the capacity to remove particles from the water column via settling, prevails over the sediment remobilization by cyprinids. Short-term turbidity peaks in the effluent after manipulation of the outlet were caused by resuspension of bottom particles due to the increased drag of the water, but could not be detected in the receiving stream downstream of the ponds. Therefore, the release of surface water during the drawdown did not cause significant loadings into the receiving stream.

In contrast, a disproportionally high input of particles could be observed for the fishing period. One third of the total suspended sediment load was released within the last one percent of the total drainage volume. Such high loading rates in a short time will likely exceed the transport capacity of the receiving stream, leading to increased deposition rates on the stream bed (Hoess and Geist 2021). TSS values exceeding 10,000 mg/l were caused by resuspension of settled particles by fish and fishermen. Pond bottom sediment is the basis for pond productivity, so it is desired to be rich in nutrients and managed accordingly through feed and fertilizer inputs (Fairchild and Velinsky 2006). The longer retention time further supports the transformation of coarse particulate organic matter into fine particulate organic matter (Ebel and Lowe 2013). Therefore, the proportion of organic carbon of particles released during the fishing was more than two times higher than in the receiving stream. When released, these particles have the potential to shift energy dynamics of the adjacent ecosystem, which goes beyond the effect of the loading with dissolved substances. Streams with recruiting freshwater pearl mussels are associated with very low turbidity values. Österling et al. (2010) found non-functional populations in Sweden associated with turbidity values > 1.9 NTU at baseflow conditions. This threshold was already exceeded in all measurements at 'us', indicating an already degraded habitat quality due to high fine sediment loads even without the addition of pond drainage effluents.

Physical settling structures are a common measure to prevent particle loading to the receiving waters during pond fishing (Hoess and Geist 2021; Soongsawang and Boyd 2012; Tucker et al. 2008). Particles remobilized during the fishing in pond B were efficiently retained in pond C, as the highly elevated TSS concentrations measured in the effluent of pond B could not be detected in the effluent of pond C. Furthermore, during fishing in pond C, the lag time between the observation of elevated turbidity levels in the effluent and the receiving stream, as well as the difference in loading and surplus rates demonstrated that about one third of the material released from TC during fishing was retained in the shallow, overgrown outlet ditch and did not reach the receiving stream.

Concerning the nutrients, due to the low stocking intensity and the absence of additional feed inputs, the levels measured in the study ponds were either similar to the baseline values of the inflow water or, in the case of NO_3–N, even lower. These observations are only representative of very extensively used fish ponds that are typical for this area (Biermann and Geist 2019). Since most of the excessive organic nitrogen in aquaculture facilities originates from feed inputs and fish feces (Hlaváč et al. 2014), loading rates are likely to be higher for drainage of semi-intensive or intensive production ponds. Nonetheless, the pond drainage released a significant amount of nutrients to the receiving stream that would otherwise have been retained in the pond. Compared to the particulate substances, patterns of nutrient loading over the drainage process were more complex. They reflected substance-specific internal processes, mostly driven by interactions at the pond bottom/water interface.

Baseline values for nitrate measured at 'us' already exceeded the target value of 1.7 mg/l NO_3–N set by the guidelines for freshwater pearl mussel protection of Bavaria (Sachteleben et al. 2004). The more recently published international CEN standard on the monitoring of FPM populations and their environment (Boon et al. 2019) states that NO_3–N concentrations in sustainable FPM populations in Central Europe should be even below 0.5 mg/l. The baseline values measured in this study presumably originate from diffuse inputs from the catchment area. This indicated a potentially high nitrogen loading into the pond through the inflow water even without additional feed inputs. However, reduced NO_3–N concentrations, similar to those measured in the present study, are commonly reported in studies on pond water chemistry, e.g. Fairchild and Velinsky (2006) observed a decrease in NO_3–N downstream of 13 ponds in southeastern Pennsylvania. Multiple processes are responsible for nitrate removal in ponds, including uptake by phytoplankton and other primary producers, denitrification and DNRA. Denitrification processes occur in the anoxic zone of the substrate - therefore they are more pronounced in stagnant water bodies than in streams. During this process, NO_3–N is reduced to N_2O and N_2 that are released into the atmosphere (Chen et al. 2017). This process is also used in waste water treatment to permanently remove nitrogen (Ebel and Lowe 2013). However, the effect of denitrification in pond sediments may have been overestimated in the past (Gold et al. 2019). Only recently, researchers have become aware of alternative pathways of nitrate removal in aquatic ecosystems, such as DNRA. Since this process transforms nitrate into biologically available ammonium rather than N_2, it does not represent a permanent removal of nitrate out of the system (Burgin and Hamilton 2007). DNRA is thought to be dominant in environments with higher proportion of labile organic carbon compounds, such as pond sediments (Gruca-Rokosz et al. 2009; Nogaro et al. 2010). Moreover, studies in storm water wet ponds have shown that net nitrogen fixation might exceed denitrification rates during summer if nitrate delivery to the sediment is restricted by thermal stratification (Gold et al. 2017; Hohman et al. 2021). Thermal stratification also increases DNRA processes relative to denitrification in lakes due to more reducing conditions in the hypolimnion (Nizzoli et al. 2010). However, stratification is less prevalent in shallow ponds and may therefore be less relevant. The proportion of alternative nitrogen cycling processes in relation to denitrification and the factors favoring one process over the other in shallow freshwater ponds need to be further investigated, to draw management implications for their use for a permanent nitrate removal (Burgin and Hamilton 2007; Gold et al. 2019).

Ammonium is formed as a waste product of protein metabolism and, depending on the pH, is in equilibrium with the highly toxic NH3. Under aerobic conditions, it oxidizes via nitrite to nitrate. Therefore, it accumulates in the reducing environment of the pond bottom. The charged ammonium cation is weakly adsorbed to clay particles, its desorption representing the source of a continuous supply to the water column (Gold et al. 2017). For channel catfish ponds, Hargreaves (1998) estimated that 25-33% of the ammonia supplied to the water column originated from the pond sediments due to the rapid mineralization of settled organic nitrogen to ammonia. Ammonium fluxes from the sediment can be enhanced by bioturbation through benthivorous fish, causing elevated concentrations in the pond water. The increased NH_4–N levels in the last proportion of drainage effluent represent a further release through pond bottom mobilization during fish harvest. Ammonium target values for functional FPM streams in Bavaria should be below 0.1 mg/l (Sachteleben et al. 2004). In the CEN standard, (Boon et al. 2019) suggest even lower target values between 0.01 and 0.05 mg/l of NH_4–N. Both these values were exceeded in effluents at TC during drawdown, as well as at 'us' and 'ds' sites in some measurements. However, the high NH_4–N concentrations in effluents during the fishing of pond C, on average 0.7 mg/l, demonstrated the potential of NH_4–N pollution from pond drainage.

Free ortho-phosphate is highly reactive and normally taken up immediately by phytoplankton (Withers and Jarvie 2008). This holds true for ortho-PO_4-P in stream water as well as in the pond, as was demonstrated by the similar levels at 'us', 'ds' and in the pond effluents at any point in time. They were below the target value of 0.06 mg/l stated in the FPM conservation guidelines of Bavaria (Sachteleben et al. 2004). However, the largest portion of phosphorous in aquatic systems is bound to particles. In this form, it is no longer immediately available and settles on the pond bottom. In contrast to the solids, nutrients bound to the pond bottom are always in equilibrium with their soluble forms, so that a certain amount is constantly released to the water column. This process is called 'internal loading' and can be a significant contributor to continuous eutrophication in lakes (Nürnberg 2009; Song et al. 2015; Steinman and Spears 2020). It is enhanced under anoxic conditions (Taguchi et al. 2020), particularly in sediments with a high content of organic matter, as in fish ponds, as reported in a study of nine mostly eutrophic fish ponds by (Potužák et al. 2016). During fish harvest, the particulate phosphate was remobilized and released together with the suspended matter, resulting in maximum concentrations of TP.

These findings reveal an important difference between dissolved and particulate inputs. NH_4–N and TP were constantly released with the drainage water, resulting in an additional loading of 12.3 kg of ammonium and 8.6 kg of phosphorous to the receiving stream. In contrast to the suspended particles, the soluble nutrients were constantly released proportional to the amount of water over the entire drawdown. Therefore, despite the maximum nutrient concentrations that also occurred during the fish harvest, the amount of nutrients released during fishing did not contribute disproportionally to the total loading, as in the case of the particles. The total loading of soluble nutrients is therefore more dependent on the pond size and the amount of water drained. The two study ponds were relatively small compared to the mean pond size in Bavarian aquaculture of 2.70 ha (LfL 2020). The amount of nutrients delivered from the drainage of such ponds is likely to be higher.

In contrast to particulate substances, dissolved nutrients as NH_4–N and ortho-PO_4-P will not be retained in short-term retention structures such as the overgrown outlet ditch. Therefore, before water is released from structures such as sedimentation basins, a sufficient retention time is necessary, to allow for an actual turnover of nutrients. Whereas a sufficient settling of particles requires one to three days (Soongsawang and Boyd 2012; Tucker et al. 2008), this is typically too short to efficiently remove nutrient inputs into receiving water bodies.

6.6 Implications for management

Two direct implications for the management of pond drainage to minimize impacts on downstream receiving ecosystems can be drawn from this study:

Firstly, an efficient retention of particles in the pond can be achieved by adapting the drainage process. As long as surface water is drained slowly, without significant remobilization of the pond bottom, loading rates can be tolerable. In contrast, bottom drains should be avoided, as well as the release of highly turbid water during fish harvest or sediment removal. As demonstrated in this study, physical settling structures can substantially reduce inputs into the receiving stream. The structures need to be maintained regularly, otherwise, settled material might be released accidently or over a longer period of time.

Secondly, if the target is to substantially reduce nutrient inputs from eutrophic water bodies, it is insufficient to only address the last step of the drainage process. The nutrient rich pond bottom constantly releases ammonium and phosphorous, which are dissolved and equally distributed within the water column. This causes elevated nutrient levels in the effluent, leading to a constant loading over the entire drainage process. Larger ponds will therefore release a higher total amount of nutrients. To minimize ammonium and phosphorus inputs, internal pond measures should be taken to generally reduce nutrient concentrations in pond water. Feed and fertilizer inputs should be minimized, while removing biomass in the form of fish and macrophytes can further reduce in-pond nutrient levels. Heavily polluted effluent water could also be drained into a well-designed treatment pond before it is released into the receiving water, if it is managed for efficient denitrification. This can be enhanced by providing sufficient light penetration and reduced shading to allow nutrient uptake by macrophytes, periphyton and phytoplankton as suggested by Crab et al. (2007). Sufficient aeration can increase nitrification of ammonium to nitrate (Hargreaves 1998), and help to reduce thermal stratification to promote denitrification and ensure permanent nitrogen removal (Gold et al. 2017). Drainage volume should be adapted to the capacity of the receiving waters to dilute the inputs.

With decreasing trends in cyprinid consumption in the population in the study region, many of the senior pond owners decided to quit fish farming and pond maintenance and are selling their properties. As natural ponds are hotspots of biodiversity (Gee et al. 1997; Hill et al. 2018), often populated by endemic and endangered species (Davies et al. 2009; Miracle et al. 2010), nature conservation organizations as well as local water authorities are interested in buying such ponds and in adapting their management. Usually, nature conservation ponds are managed to be preferentially fish-free and without any additional nutrient inputs (Lemmens et al. 2013) in order to secure or enhance local biodiversity. In the light of climate change, the idea of using small, natural water bodies as water retention structures for drought mitigation is also becoming more and more popular, including potential pond drainage to support stream flow (Baldan et al. 2021b; Vaclav and Davidova 2015). However, even ponds without fish production need to be maintained and managed to sustain their functions. They accumulate diffuse inputs from the catchment or inflow water, leading to higher nutrient levels, probably intensified by internal loading from historically nutrient-rich pond bottoms. Nutrient levels and their fluctuation over time and season should be monitored prior to drainage and considered in the planning of maintenance actions.

Sustainable nature conservation needs to balance targets to potential negative effects on the receiving stream (Sayer 2014), to maximize benefits and services provided by the ponds and simultaneously minimize the associated threats to the adjacent watercourses. For example, ponds in the study area need to be managed with regard to the endangered freshwater pearl mussel. The high baseline turbidity and nutrient levels, which greatly exceeded the target levels stated to be acceptable for pearl mussel streams (Boon et al. 2019; Österling et al. 2010; Sachteleben et al. 2004), indicated already poor suitability of the water body for the target species. In this case, completely abstaining from pond drainage to strictly minimize all possible inputs might outweigh the benefits of maintaining the pond ecosystem. This consideration must be made by local management authorities in respect to conservation policy and legal requirements.

7. General Discussion

The four studies demonstrate the importance of considering fish ponds as essential catchments elements in the conservation and restoration of adjacent streams. Ponds can act either directly via their effluents or indirectly through their connection with the shallow aquifer. They have effects on hydrologic and thermal regimes that are shaping habitat quality and therefore the distribution of aquatic communities. Ponds play a role in sediment dynamics, in particular during the pond drainage. Due to their relatively small size and artificial nature, ponds can be managed and controlled more easily than larger land use elements. A well thought-out pond management offers the chance to increase the overall resilience of valuable stream systems, such as the studied freshwater pearl mussel streams. Through their water retention capacity and removal of nitrate, the ponds offer regulating and supporting ecosystem services besides the provision of fish for human consumption (Biggs et al. 2017; Moore and Hunt 2012).

The multiple impacts of fish ponds were studied using a combined approach of hydrological modeling together with field studies. The catchment-wide SWAT model had the advantage of an easy integration of a large number of land use and soil parameters as well as the groundwater and considering a large time scale of more than 10 years (Baldan et al. 2021b). It was used to gain a general overview of the cumulative effects of the several hundred ponds on the catchment scale and to identify processes governed by the ponds, such as the groundwater recharge. However, model parameterization needs to summarize and simplify certain catchment components and processes to enable feasible computation time. This bares the threat of losing small-scale variability that is important for a realistic representation of pond effects (Blöschl and Sivapalan 1995). Concerning the ponds, the field studies showed highly variable effects for different ponds that could not be accounted for in the SWAT model. The very fact that individual ponds within a sub-basin had to be combined into one Hydrological Equivalent Wetland (HEW, Wang et al. (2008) to be integrated into the model, demonstrated that processes at the pond scale could only be represented to a limited extent. Therefore, studying the actual impacts on the FPM habitat in the field was the key to examining the impacts of individual ponds at the local scale, and provided an opportunity to consider their individual management. As shown in Chapters 5 and 6, the impact of ponds on adjacent streams was highly dependent on the pond conditions and drainage management. At the scale of a single pond, drainage could cause high TSS concentrations in the receiving stream or had almost no impact, depending on the application of SSM. Sampling at a high spatial and temporal resolution also considered the highly dynamic nature of stream systems (Geist 2015), including potential diffuse and point sources of pollution as well as different flow conditions. The extensive field studies allowed for comparison of pond impacts with background data to assess the influence of fish ponds in the context of other stressors affecting the FPM habitat. The information derived from hydrologic modeling can improve the interpretation of data obtained from field observations (Baldan et al. 2021b; Grayson et al. 1992). Combining the advantages of both methods allowed for a holistic evaluation of pond impacts on the FPM habitat at multiple scales.

High levels of fine sediment deposition on the stream bed and increased nutrient concentrations have been identified as the main reason for the unsuccessful recruitment of the freshwater pearl mussel in the study region (Bauer 1988; Denic and Geist 2015). As evident from Chapter 4, substrate conditions are still unsuitable for juvenile mussels due to enhanced fine sediment deposition rates. Flow conditions were more important for fine sediment dynamics than the impact of the pond effluents over most of the year. However, as demonstrated by the hydrologic modeling, the ponds themselves influence the flow regime, showing the complex interactions between different processes acting in and around the ponds.

7.1 Flow regime

As demonstrated in Chapters 3 and 4, high discharge events are essential in shaping the FPM habitat. As sedentary organisms, FPM rely on stream stretches that offer sufficient substrate stability and protection from high shear stress due to strong current velocities (Allen and Vaughn 2010). One example of the threat of major flood events on FPM populations was observed by Hastie et al. (2001) in 1998 in Scotland when a 100-year return flood in the River Kerry scoured more than 50,000 individuals. High discharges develop high current velocities, particularly within narrow, channelized streams, and can develop extreme shear forces. Peak flow reduction through water retention in ponds might mitigate extreme current velocities and prevent detachments of mussel specimens (Baldan et al. 2021b). Artificial structures, such as retention and smaller detention ponds, are used for flood protection worldwide (Clary et al. 2020; Janke et al. 2022; Yazdi et al. 2021), retaining excess water in depressions after high precipitation events. In particular, detention, or “dry ponds”, can effectively store excessive amounts of water during peak flows and release them with a delay (Seibert and Auerswald 2020). However, water retention capacity in storm water ponds differs from fish ponds, where high pond water levels need to be sustained over the production period to avoid low oxygen concentrations and other issues threatening animal welfare. The initial pond water level prior to storm water inflow is an essential factor in peak flow reduction, so detention ponds are often not permanently filled (Hancock et al. 2010). Efficient fish production, therefore, limits the additional retention capacity for flood discharges. This was reflected in the relatively low effect on peak flow reduction through the ponds shown in the SWAT model in Chapter 3. The modeled retention capacity was adjusted to the realistic conditions of fish production ponds by setting a low additional retention volume for the emergency spillway. A similar process of reduced storage capacity might cause wetlands saturated during wet winter months to actually contribute to a higher flood peak rather than retaining water while reducing peak discharge during dry periods (Acreman and Holden 2013). Moreover, the overall small size of the study ponds (average of 0.14 ha and a total of 53.56 ha, compared to the 9082.2 ha total catchment area) was insufficient to achieve a shift in the timing of peak flows, demonstrating their limited flood mitigation potential, compared to larger fish pond facilities. In a study by Lhotský (2010), a 276 ha fish pond transformed a flood wave of 44 m³/s to 12.6 – 17.1 m³/s, and similar flood peak reductions were observed by similar big ponds in the same region.

High discharge volumes and resulting high current velocity not only detach mussels but also remobilize the stream bed, including larger particles. This was demonstrated by the high proportion of larger grain sizes deposited after high discharge events in Chapter 4. Therefore, high discharge conditions play a major role in (fine) sediment dynamics (Allan et al. 2021) and the habitat quality for juvenile FPM. The effects of high discharge events on sediment transport processes depend on their duration, strength, and timing, as well as the (weather) conditions responsible for the increased flow rates (Navratil et al. 2012). The supply and the deposition of fine soil particles depends considerably on catchment land use and can vary throughout the year (Cerdan et al. 2010; Knott et al. 2019), as evident in Chapter 4. To evaluate the impact of fish ponds on fine sediment deposition in the adjacent streams, the overall background fine sediment deposition in relation to other catchment land uses must be considered.

Strong rain events, particularly during dry summers, can develop extreme erosivity, particularly over bare agricultural soils not protected by vegetation. Rain erosivity based on rain-radar data in Germany showed the highest average daily erosion index values during June, July, and August (Auerswald et al. 2019a), when vegetation cover over agricultural land is lowest and infiltration into dry soils limited, which increases the particle detachment through the force of falling rain (Moragoda et al. 2022). Therefore, fine sediment loads and fine sediment deposition after rain events can be considerably high, as demonstrated in Chapter 4, where the maximum fine sediment deposition rates occurred after the strong rain events. Erosion rates from pastures and forests are considerably lower than from arable land because the vegetation cover protects the soil from the erosive force of the rain drops and increased infiltration rates dampen the development of overland flow that could also detach soil particles (Cerdan et al. 2010).

In contrast to strong rain events, high discharges caused by snow melt are less prone to deliver fine sediments via erosion, as the impact of raindrops falling on soils is irrelevant for pure snowmelt events. Runoff from snow melt is usually higher than from precipitation events due to the higher amount of water stored in the snow pack and the decreased infiltration capacity of frozen soils. However, the increased overland flow over the frozen soil can also lead to increased fine sediment inputs during the snow melt (Ollesch et al. 2005). In the study area, snow melt played an important role in stream bed cleaning, developing the highest current velocities capable of mobilizing large proportions of the stream bed. The cleaning of spawning sites for the host fish and juvenile FPM habitats (Hauer 2015) might compensate for the negative effects of mussel detachment by bed-mobilizing floods. Olofsson (2017) hypothesized that due to the life cycle of the FPM, with millions of glochidia produced each year and the thousands of juveniles in functional populations, even a complete failure of one breeding cycle due to extreme events such as floods is no risk for the population. However, in the study presented in Chapter 4, the average amount of fine sediment deposited after the snow melt was higher than for the rain events, although its proportion was much lower. Apparently, fine sediment inputs from the catchment are still too high in the studied FPM streams, as already demonstrated by Denic and Geist (2015) and Pander et al. (2015). However, even if the substrate quality did not significantly increase after the snow melt, cleaning the stream bed through snow melt is likely to be an important driver to sustain interstitial habitat quality in restored stream sections, as in the Czech part of the ZB/SR catchment.

With the ongoing climate change, an increase in the pressures acting on the already declining FPM populations can be expected. The predicted changes in precipitation patterns towards an increase in short-term intense precipitation and an increase in winter rain over snow can already be observed (IPCC 2022; LfU 2021) and may lead to even higher erosion rates in the future. Due to the very low growth rates during the cold period, soil cover of winter crops is minimal and, therefore, particularly prone to erosive rains (Auerswald et al. 2019a). The importance of the snow melt on stream bed cleaning demonstrated in Chapter 4 needs to be included in the overall understanding of sediment dynamics in the study area and can help to adapt FPM conservation and justify stringent measures. Erosion protection measures, need to be enhanced throughout the study catchments, as it is most effective to prevent the transport of soil particles into the streams already at their source (Geist 2015; Pulley et al. 2019). On-field measures, like plantation of cover crops between production cycles or reduced tilling, were recommended, e.g. by Panagos et al. (2015) and implemented into the EU Common Agricultural Policy (CAP) reform in 2003. Buffer strips, as mandatory along Bavarian waterbodies since 2019, can help to reduce diffuse inputs from larger areas (Knott et al. 2019). Maintaining a closed plant cover throughout the year should become an essential requirement for farming in valuable FPM catchments to mitigate climate change impacts on soil erosion. Point sources for fine sediments, such as drainage ditches (Chapter 4), pond drainage events (Chapters 5 and 6), or roads, as identified in a study by Pulley et al. (2019) in a Scottish FPM stream, need to be addressed before they discharge into FPM streams.

Other projected climate change impacts include prolonged summer low flow conditions resulting from increased evapotranspiration at increased air temperatures. Such a development can already be observed in the study catchments, where increased low flow periods were observed in 2018-2020 and again in 2022, after one relatively wet year, 2021. In 2019, 2020, and 2022, stream sections inhabited by FPM fell dry rather quickly within a few days, and hundreds of individuals had to be evacuated and maintained in the nearby FPM rearing station. Such pulse disturbances can have dramatic consequences on stream biota, as shown in a meta-analysis by Sabater et al. (2023). In particular, invertebrate richness and biomass are negatively affected by flow cessation, and freshwater mussels are even less able to escape from drying stream section (DuBose et al. 2019; Mitchell et al. 2019; Mitchell et al. 2018). Even if mussels can survive in isolated pools or shaded wet spots, they rely on flowing water to transport their glochidia to suitable host fish. Therefore, recruitment and subsequent recovery from drought conditions might be decreased. Sustaining the needed water flow in the study streams in the future demands catchment scale measures to increase the resilience of these systems to droughts, e.g., through increasing water retention within the landscape. The water storage provided by the ponds, demonstrated in Chapter 3, can potentially mitigate some of the climate change-related threats. This accounts, on the one hand, for releasing water directly from the pond storage through controlled drainage. Pond drainage to sustain stream flow has already been applied in the study catchment during the extreme droughts in the summers of 2018 - 2020 and 2022. However, fish pond management might also threaten already low baseflows under drought conditions if stream water is used to sustain pond water levels. This practice has led to a general order that prohibits water extraction for pond filling during extreme low flows in the study area. Beyond the threat of a dry-out of stream stretches inhabited by FPM, low flows also decrease the transport capacity of streams (Chapter 4). Baldan et al. (2021a) predicted increased fine sediment deposition in a FPM stream in Austria, drastically decreasing the area of suitable habitat under RCP 8.5.

Concerning mitigating climate change impacts, groundwater recharge from ponds increases groundwater storage. Wetlands and stream systems that are driven mainly by groundwater recharge are said to be more resistant to climate-driven changes (Union of Concerned Scientists et al. 2005). The comparison of the SWAT models with and without ponds showed a moderate stabilizing effect of the ponds on stream discharge, particularly during low flow conditions. The effect of water loss from the open pond surfaces through evaporation on the basin water balance was exceeded by the increased groundwater storage. If water from snowmelt or precipitation can be stored within the catchment and released over a longer period, such delayed recharge might help sustain stream flow levels during precipitation-free periods (Carey et al. 2010). The increase in baseflow through ponds and functional wetlands might explain the positive effects of ponds within Swedish FPM catchments that seemed to favor mussel recruitment (Jensen 2007; Österling and Högberg 2014). In contrast, the proportion of wetland cover was negatively correlated with FPM recruitment in Norway (Gosselin et al. 2022). However, while ponds and wetlands were an indicator for extensive anthropogenic land use in the Swedish studies, most Norwegian wetlands are drained, which impairs their function as water and sediment retention features.

The ecological importance of elevated baseflow levels is not only the mere quantity of water within the stream and the maintenance of flow within aquatic habitats; it has broader implications for the habitat quality for aquatic organisms. This will become another critical issue in light of climate change when high Tw and low water levels increase stream metabolism and hence oxygen consumption (Whitehead et al. 2009). The hyporheic zone in the interstitial, the contact zone between groundwater and surface water, is the FPM juvenile habitat, and groundwater upwelling might help clean the interstitial from excess fine sediments clogging pore spaces. Increased baseflow levels are also relevant to diluting nutrient inputs and increasing nutrient transportation downstream, reducing the risk of eutrophication and biofilm formation. However, groundwater is usually low in oxygen content, and the oxygenation of the interstitial needed for the successful development of FPM juveniles depends on the infiltration of oxygen-rich surface water. The increased delta values between abiotic parameters in the free wave and the interstitial (Chapter 4), particularly during summer, indicated that flow in this direction, from the surface to the interstitial zone, was limited by the high fine sediment content of the stream bed in the study streams. Increased groundwater contribution through ponds is also important to sustain cool water temperatures (Tw), a critical factor in the distribution of cold-stenothermal fish species such as salmonids (Hayashi and Rosenberry 2002; Wenger et al. 2011). Maintaining a continuous supply of groundwater to headwater streams will become increasingly critical in the future (Kaule and Gilfedder 2021) to provide sufficient baseflow levels and thermal conditions suitable for headwater species.

7.2 Temperature regime

However, in contrast to the subsurface pathways through the aquifer, the discharge of heated water from nearby ponds increased Tw in FPM streams considerably during summer (Chapter 3). Summer is the season of prolonged low flows and warmer water temperatures, both in the streams and, even more pronounced, in the ponds. Efficient production of the warm-stenothermic carp in the temperate zone of Central Europe is only feasible because of the increased Tw in stagnant ponds (Balon 1995). Besides, the best practice in carp ponds management theoretically states that ponds should be kept closed over the production period and only supplied with fresh water to compensate for evaporation and seepage losses (O'Grady and Spillett 1985), in practice, most ponds owners released water from their ponds in summer. In some cases, these heated effluents considerably increased stream Tw directly below. Increased stream temperatures can boost oxygen consumption by microorganisms in biofilms, enhancing the pressure on juvenile mussel development. Therefore, an assessment of the FPM habitat should be conducted at least in summer, during the worst conditions (Boon et al. 2019).

Moreover, to sustain FPM populations, it is important to consider not only the mussels' thermal requirements but of the thermal optimum of the host fish (da Silva et al. 2022; Pandolfo et al. 2012). Although brown trout densities in functional FPM streams are often lower than in non-functional streams (Geist et al. 2006), a low FPM recruitment might also be attributed to very low host fish densities. The brown trout is more sensitive to increased Tw and is able to migrate from sections that are too warm (Elliott 2000). If sections over mussel beds are avoided by the host due to temperature stress during glochidia release, limited infestation rates will impair recruitment already at this early stage. During the parasitic phase, high Tw might also decrease the transformation success of glochidia on the hosts' gills since cool Tw is known to suppress the immune response of ectothermic organisms like fish (Roberts and Barnhart 1999). Taeubert et al. (2014) found shorter development time and decreased excystment rates for *U. crassus* from hosts maintained at high Tw, indicating impaired recruitment of juvenile mussels at high Tw. However, since the growth rates of poikilothermic organisms increase with higher Tw, an increased Tw might actually be beneficial for the development of post-parasitic FPM in very cool headwater stretches (Taeubert et al. 2014). Forest thinning around stream sections inhabited by FPM is a common restoration measure applied in the study area (Hruška 1992) to enhance juvenile growth. However, such measures will increase Tw on a large spatial scale (Chapter 3) and over the whole year and do not allow for adapted management considering the different temperature optima for the different mussel life stages described above. Water release from fish ponds can easily be controlled and adapted as a conservation measure to locally increase (juvenile) mussel growth during summer when it is ceased during the glochidia release and development time to ensure suitable conditions for the host fish around mussel beds.

7.3 Pond drainage

During most of the year, the effects of fish ponds on fine sediment dynamics compared to other land uses were rather moderate, although some ponds could be identified as point sources for fine sediments throughout the growth period. This was again due to the constant release of pond water, even during the summer. Resuspension of the pond bottom by the feeding activities of benthivorous carp increased TSS concentrations in effluents, occasionally leading to increased fine sediment deposition rates below the ponds. This is a major difference to fish-free settling ponds used to treat high TSS concentration, e.g., in wastewater treatment. Moreover, the pond drainage represented a major fine sediment and nutrient source under certain circumstances. As evident from Chapter 5, these included the fast and complete drainage of the ponds, collecting fish with individual dip nets rather than seining, and the lack of SSM like settling structures. The remobilization of accumulated fines during the short period of the fish harvest releases high TSS concentrations within a small amount of water (Chapter 6), limiting dilution and transportation downstream of the effluent inflow. This demonstrated the threat of man-made fish ponds to adjacent stream systems if poorly managed or over-exploited. Turbidities > 15 NTU should not occur in FPM streams over an extended period of time, and the proportion of fines < 1mm typically remains < 25% in functional FPM streams (Tabel 1). The fine sediment deposition rates in the FPM streams below the drained ponds were similar or even higher than after high discharge events when a much larger proportion of the catchment contributes to fine sediment inputs. Therefore, stream stretches with low fine sediment inputs from the surrounding catchment land use are particularly threatened by the inputs from fish ponds.

On the other hand, the ponds' artificial nature offers a high level of control that can be applied to mitigate such adverse impacts by rather easy management measures. Chapter 5 showed that the impact of fish ponds on TSS concentration in the effluent and following fine sediment deposition strongly depended on drainage management. The varying impacts of pond drainage on receiving water bodies have been described in various studies. Banas et al. (2002) related suspended sediment outputs between 10.8 to 36.5 kg/ha to drainage operations taking place during drought or heavy rainfall. Lin et al. (2001) compared the environmental impacts of four different fish harvest methods and found the method requiring no pond drainage to be the least harmful. Similar results were evident in Chapter 5, where one sampled fish harvest scenario included seining the pond with a closed outlet. The other efficient form of SSM was the use of settling structures to retain particles. The responsible in-pond processes were studied in Chapter 6: The low TSS and TP concentrations over most of the drawdown step resulted from an efficient settling of particles and particle-bound nutrients within the stagnant pond water body (Yazdi et al. 2021) during a time where carp are usually inactive due to cool temperatures. This resembles the sediment retention in a settling pond. For an efficient reduction of TSS concentrations and adhered nutrients like phosphates in effluents, newly constructed settling basins should be large enough to ensure a sufficient water residence time to allow also the settling of fine particles (Soongsawang and Boyd 2012). High water residence time also facilitates a microbial turnover of nitrate within the anaerobic layer of the pond bottom (Janke et al. 2022; Yazdi et al. 2021). This was evident from the reduced nitrate concentrations in pond effluents shown in Chapter 5 and 6. However, an actual removal of N from the system is only achieved by complete denitrification to N_2. The varying importance of alternative nitrate pathways has already been studied in storm water detention ponds (Gold et al. 2019). Research needs to be conducted to assess to what degree alternative nitrate pathways act within fish ponds and how the importance of the different processes varies temporally to verify their potential to remove nitrate permanently.

Dissolved nutrient inputs from pond drainage need to be considered differently than particulate pollutants. As they are released over the entire drainage process, elevated nutrient concentrations can hardly be mitigated by closing the outlet during fish harvest. If there is no possibility to construct settling structures with a sufficient water residence time to allow for N and P turnover, they need to be addressed directly within the pond. Reduced nutrient levels can be achieved by extensification and reducing the stock size and feed inputs. Such measures will directly decrease the general nutrient inputs, reduce the remobilization of settled nutrients from the pond bottom by benthivorous fish and allow for macrophyte growth. The capacity of P uptake by plants is limited in intensively stocked cyprinid ponds as benthivorous carps decrease the number of submerged macrophytes through feeding and increasing turbidity in the pond (Badiou and Goldsborough 2015). High nutrient concentrations in adjacent streams threaten the sensitive FPM, depending on oligotrophic conditions (see target values in Table 1). Increased BOD 5, nitrate, and phosphate concentrations have been associated with increased mortality of adult and juvenile FPM by Bauer (1988). Bílý and Simon (2007) observed a poor population status of FPM populations in the Czech Republic in streams with higher nutrient pollution. Nutrient concentration in pond effluents exceeded the target values set for functional FPM streams in most of the sampled drainage operations. The inflow of pond effluents has the potential to abruptly worsen the conditions in the receiving FPM streams at the local scale.

Gradual, longitudinal changes are a central feature of river systems over large spatial scales, as described for various parameters like discharge, channel width, and substrate composition by the “river continuum concept” (Vannote et al. 1980). However, at finer spatial scales, stream parameter changes seldom match a smooth gradient but are often strongly influenced by abrupt land use changes or the inflow of tributaries with different abiotic characteristics (Rice et al. 2001). Tributaries are therefore regarded as essential elements of river discontinuities (Perry and Schaeffer 1987). Discontinuities along the longitudinal stream gradient are usually associated with impoundments, as defined by the “serial discontinuity concept” by Ward et al. (1983). Since stream characteristics in unregulated rivers often also deviate from a smooth, gradual change due to the inflow of multiple tributaries, Rice et al. (2001) developed the “link discontinuity concept” to attribute an abrupt change in downstream ecosystem structure to tributary inflows. This includes changes in water volume, substrate composition, and water quality parameters like conductivity and nutrient concentrations. The impact on the adjacent stream is complex and strongly depends on the water volume discharged from the tributary channels and the quality and quantity of suspended particles and dissolved nutrients delivered by the tributary (Rice et al. 2001). The same accounts for the measured impact of the fish ponds, whose effect strength strongly depended on pond characteristics and management, proximity to the main stream, and timing. The studies described in the previous chapters show that the link discontinuity concept can be applied to inflows from (fish) ponds, mainly during pond drainage, and should be extended to the water temperature. The warming by inflows from ponds was stronger than from “non-pond” tributaries that sometimes even cooled main stream Tw if they were strongly influenced by groundwater (Chapter 3). As demonstrated in Chapters 3 and 4, impacts might be particularly pronounced if a close-by, intensively used pond discharges into a stream section with overall homogenous catchment properties, e.g., low diffuse fine sediment inputs along forested stretches.

The studied ponds are relatively small catchment elements, therefore, their effects on the overall hydrology and other studied processes were moderate. However, cumulative negative effects of multiple pond discharges along cool, oligotrophic stream stretches might be strong enough to exceed the stream's capacity to transport particles and dilute heat and nutrient inputs, leading to a degradation of habitat quality for sensitive species like brown trout and FPM. On the local scale, several fish ponds were shown to exhibit strong impacts on the habitat quality of specific stream stretches. Local conditions play an essential role in the current situation of small, fragmented FPM populations remaining in Central Europe. The exact location of a mussel bank might be highly relevant for their contribution to recruitment due to the very low numbers of remaining individuals distributed over the stream in scattered patches. The impact of the ponds draining into the FPM streams should therefore be included in management plans and the search for suitable sites for releasing captive-bred juveniles. Due to the imperiled status of this species and the enormous effort needed to propagate juveniles in rearing stations over multiple years, release sites must be chosen carefully to ensure the success of such reinforcement measures.

7.4 Conclusion

The results stated in the chapters above demonstrate the complex effects of fish ponds on receiving FPM streams that cannot be seen as clearly positive or negative. In light of climate change, drought mitigation will become more relevant in FPM conservation plans. Keeping (fish) ponds within the study catchments offers the possibility to increase stream resilience to droughts by improved water retention in the landscape. However, the future use of the ponds for fish farming might be questioned. The regulatory ecosystem services through increased water retention, groundwater recharge, and nutrient cycling might exceed the provisioning service of fish production, particularly at the low intensity level within the study area. The future of pond farming in Central Europe is threatened by climate change, as it depends on a sufficient supply of water of reasonable quality, which might become problematic (Whitehead et al. 2009). If fish ponds are abandoned, water and conservation managers should take the opportunity to buy them to preserve their function as water retention structures. The retention capacity of existing ponds might be increased via pond restoration to increase both the amount of water stored directly in the pond and groundwater recharge through the pond bottom to increase groundwater contribution under low flow conditions. If stocked with fish, releasing water from the pond should be limited, and effluent quality should be analyzed to prevent high nutrient or heat loading into the adjacent water body. If unavoidable, pond drainage must be done with the highest care to prevent the adverse effects of fine sediments and nutrients mobilized from the ponds. If ponds are drained as a conservation measure, either to increase Tw or to support baseflow levels, water should only be released from the surface. A certain pond water level should be sustained to prevent remobilization of the pond bottom and the release of excessive amounts of fine sediments. Extensive fish production might still be feasible if mitigation measures are taken, at least during the fish harvest. The pond owners in the study area are aware of the value of the remaining FPM populations, and the issue of potential fine sediment pollution is well-known among them. The mitigation measures sampled in Chapter 5 were realistic scenarios, which proves their feasibility in pond farming practices. This hopefully increases their acceptance with the local pond owners. Routing the effluent into (existing) ponds as settling structures, keeping submerged vegetation in outflow channels, or closing the pond outlet during the final fish harvest were efficient measures to prevent high TSS concentrations, turbidities, and fine sediment deposition rates in the main streams. Settling ponds should be fish-free to prevent remobilization of settled particles by benthivorous feeding. Straw filters were inefficient in reducing suspended solid concentration in effluents and should not be used as a single measure. Fish production should stay at the most extensive level to prevent high nutrient concentrations in effluents, as they are more challenging to treat outside the pond waterbody.

7.5 Outlook

The studies presented in this thesis proved the advantages of combining hydrological modeling with field studies to assess the impact of fish ponds on adjacent streams. The need to use a holistic approach to cover the many aspects and processes influenced by the ponds over multiple spatial and temporal scales was evident. In particular, integrating various catchment components and considering effects over different time points in the management cycle gave valuable insights into the pond impacts. However, such approach is seldom applied in studies of aquaculture effects. The methodology established in the four chapters can easily be transferred to other situations, e.g., to study the impact of larger ponds or more intensive aquaculture facilities. As an important and still growing agricultural sector worldwide, an increase in aquaculture production can intensify pressures on aquatic ecosystems. As evident from the studies presented above, even very extensive fish ponds can have negative environmental impacts on oligotrophic streams if poorly managed. Larger, more intensive pond farming will likely have more severe impacts. Several studies have investigated the TSS and nutrients released from single, intensive aquaculture facilities. Still, they lack an assessment of cumulative impacts on factors like sediment deposition, hydrology, and thermal regime over multiple scales and in relation to other land use elements. The methodology described in this thesis for assessing the impact of ponds as comprehensively as possible could be used to investigate the impacts of such aquaculture facilities beyond the scope of effluent quality. The more holistic understanding yielded by this approach is needed to derive management implications to ensure sustainable production of highly valuable fish protein.

Despite their demonstrated impact, fish ponds alone are not the main reason for habitat degradation in the studied FPM streams. They do not act separately but are an integrated part of the catchment. Therefore, addressing only the impacts of the fish ponds will not be sufficient to restore these streams. Fine sediment and nutrient concentrations were elevated above FPM target values throughout most of the studied stream sections, even at minimal pond discharge, as demonstrated in Chapters 5 and 6. Pond management can, therefore, only be one part of a catchment-wide approach, acting together with measures to increase the overall resilience to climate change, restoration of natural flow regimes, and erosion protection measures. This will likely include the issue of field drainage, which often affects large areas in the catchment. The methods used to study the pond impacts can be extended to non-pond point sources like tributary or drainage channels that could contribute significantly to the degradation of the condition in the main stream. The same accounts for a scientific monitoring of the efficiency of sedimentation structures constructed as a restoration measure, to ensure their suitability to restore stream conditions. Since fine sediment dynamics strongly depend on discharge conditions, such investigations should follow the design presented in Chapter 4 and be conducted during low and high flow conditions, ideally both during snowmelt and rain events. In the case of field drainages, where high fine sediment loads often arise together with high nutrient concentrations, additional measures might be required. Research on innovative SSM is needed to address this issue and integrate it into the catchment-wide approach needed for sustainable restoration of the threatened FPM population in Central Europe. The methodology developed in this thesis offers an ideal holistic approach to evaluate the efficiency of such measures.

8. Publication List

8.1 Publications related to this thesis

Hoess R. & J. Geist, 2020. Spatiotemporal variation of streambed quality and fine sediment deposition in five freshwater pearl mussel streams, in relation to extreme drought, strong rain and snow melt. *Limnologica*. 85: 125833. DOI: 10.1016/j.limno.2020.125833

Hoess R. & J. Geist, 2021. Effect of fish pond drainage on turbidity, suspended solids, fine sediment deposition and nutrient concentration in receiving pearl mussel streams. *Environmental Pollution*. 274: 116520. DOI: 10.1016/j.envpol.2021.116520

Hoess R. & J. Geist, 2022. Nutrient and fine sediment loading from fish pond drainage to pearl mussel streams – Management implications for highly valuable stream ecosystems. *Journal of Environmental Management*. 302: 113987. DOI: 10.1016/j.jenvman.2021.113987

Hoess R., Generali K.A., Kuhn J.& J. Geist, 2022. Impact of Fish Ponds on Stream Hydrology and Temperature Regime in the Context of Freshwater Pearl Mussel Conservation. *Water*. 14: 2490. DOI: 10.3390/w14162490

8.2 Further publications not included in this thesis

Geist J., Benedict A., Dobler A.H., **Hoess R,** & P. Hoos, 2023. Functional interactions of non-native aquatic fauna with European freshwater bivalves: implications for management. *Hydrobiologia*. DOI: 10.1007/s10750-022-05121-2

8.3 Oral presentation

Hoess R. & J. Geist, 2019. Impact of fish ponds on sediment deposition and habitat quality of freshwater pearl mussels. International Workshop Freshwater mussels: Search for suitable habitats and evaluation of protection measures, Dresden, Germany, March 2019

Hoess R. & J. Geist, 2019. Patterns of sediment deposition and habitat quality of three freshwater pearl mussel streams under drought conditions. Pearl mussel conference Restoration in pearl mussel habitat, breeding and natural food sources, Hof, Germany, November 2019

Hoess R. & J. Geist, 2021. Impact of fish pond drainage on fine sediment deposition and nutrient concentrations in receiving pearl mussel streams. Final Conference Interreg Malsemuschel, Český Krumlov, Czech Republic, October 2021

Hoess R. & J. Geist, 2022. Fischteiche an Flussperlmuschelgewässern – Einflüsse bei Niedrigwasser [Fish ponds at freshwater pearl mussel streams – Impacts under low flow conditions]. Fachtagung Muschelschutz in Bayern, Freising, Germany, March 2022

Hoess R. & J. Geist, 2022. Impact of fish ponds on temperature regimes of streams with endangered freshwater pearl mussel. 36th Congress of the International Society of Limnology, Berlin, Germany, 2022

Hoess R. & J. Geist, 2023. Fine sediment dynamics in pearl mussel streams of North Bavaria. International freshwater mussel conference - Conservation of freshwater mussel in light of climate change and human pressures, Selbitz, Germany, 2023

8.4 Poster presentation

Hoess R. & J. Geist, 2023. Impacts of fish ponds on freshwater pearl mussel streams. International freshwater mussel conference - Conservation of freshwater mussel in light of climate change and human pressures, Selbitz, Germany, 2023

9. Author contributions to the chapters

Chapter 3: Impact of fish ponds on hydrology and temperature regime of a small catchment containing the endangered freshwater pearl mussel

Conceptualization, R.H. and J.G.; methodology, R.H., K.A.G. and J.G., software, R.H. and K.A.G.; validation, J.K. and J.G.; formal analysis, R.H. and K.A.G.; investigation, R.H.; resources, J.G.; data curation, R.H.; writing—original draft preparation, R.H.; writing—review and editing, J.G., J.K. and K.A.G.; visualization, R.H., J.K. and J.G.; supervision, J.G.; project administration, R.H. and J.G.; funding acquisition, J.G. All authors have read and agreed to the published version of the manuscript.

Chapter 4: Spatiotemporal variation of streambed quality and fine sediment deposition in five freshwater pearl mussel streams, in relation to extreme drought, strong rain and snow melt

Conceptualization: J.G., Methodology: R.H., J.G., Investigation: R.H., Data curation: R.H., Formal analysis: R.H., Visualization: R.H., Writing - original draft: R.H., Writing - review & editing: J.G., Supervision: J.G., Funding acquisition: J.G.

Chapter 5: Effect of fish pond drainage on turbidity, suspended solids, fine sediment deposition and nutrient concentration in receiving pearl mussel streams

Conceptualization: J.G., R.H.; Methodology: J.G., R.H.; Investigation: J.G., R.H.; Data curation: R.H.; Formal analysis: R.H.; Visualization: R.H.; Supervision J.G; Writing - original draft: J.G., R.H.; Writing - review & editing: J.G., R.H.; Resources: J.G.

Chapter 6: Nutrient and fine sediment loading from fish pond drainage to pearl mussel streams – Management implications for highly valuable stream ecosystems

Conceptualization: RH, JG; Data collection: RH; Data analyses and interpretation: RH, JG; Visualization: RH; Writing- original draft: RH; Writing-editing: JG; Supervision: JG; Funding: JG.

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11. Supplemental Material

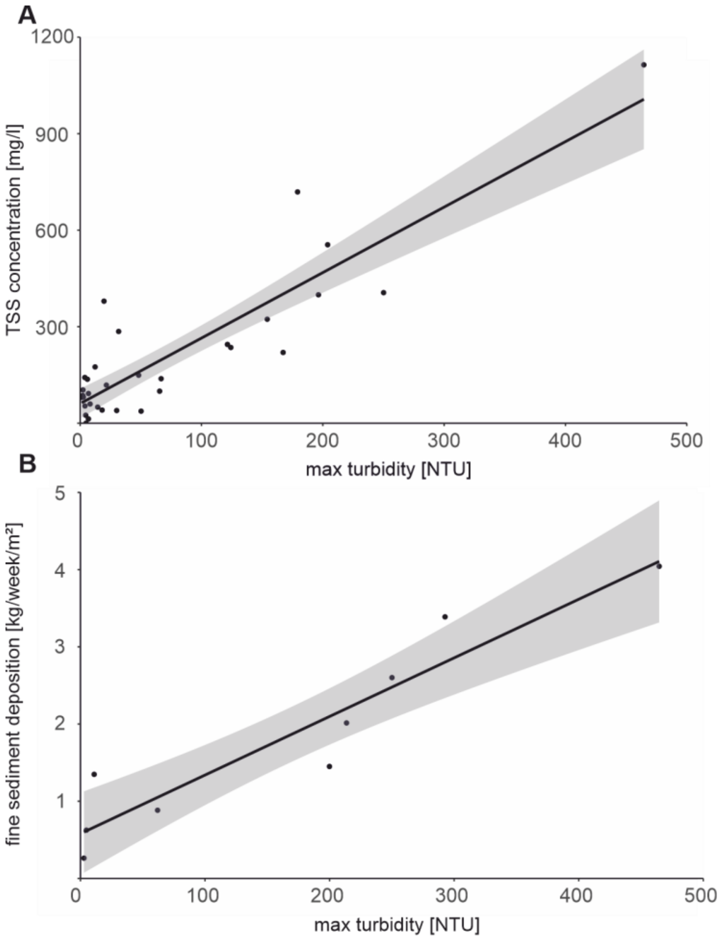

Figure S 1: correlation between the maximum turbidity [NTU] measured during pond drained and (A) concentration of total suspended solids TSS [mg/l] in the respective water sample and (B) the fine sediment deposition at the sampling site (only for nine sites where turbidity measurements were taken at the same site as the sediment trap).

Table S 1: Matching subbasins from the SWAT model with stretches from the temperature study, groundwater contribution after Kaule and Gilfedder (2021) (GW) where applicable and BFI calculated based on the simulated outflow for different time periods: mean over the whole period (excluding warm-up), and summer (su) 2018, 2019 and 2020.

Stretch	EB1	EB1	HB1	HB2-HB1	HB3-HB2	MB1	MB1	MB2-MB1	MB3-MB2	MB4-MB3	BB2-BB1	BB4-BB2	BB5-BB4
Subbasin	1	2	7	11	23	28	35	31	30	18	70	53	45
GW	-	-	-	-	low	high	high	high	high	low	-	-	-
Mean BFI	0.86	0.85	0.82	0.85	0.83	0.82	0.74	0.79	0.80	0.81	0.69	0.67	0.69
BFI su 2018	0.72	0.72	0.72	0.73	0.81	0.98	0.94	0.96	0.95	0.94	0.95	0.91	0.91
BFI su 2019	0.97	0.97	0.94	0.96	0.92	0.99	0.97	0.98	0.97	0.96	0.91	0.87	0.88
BFI su 2020	0.61	0.58	0.53	0.58	0.57	0.67	0.47	0.56	0.59	0.58	0.45	0.40	0.43

Table S 2: Nutrient concentration [mg/l] in water samples from background samplings.

Sampling site	Interval	NO_3-N [mg/l]	PO_4-P [mg/l]	NH_4^+ [mg/l]
SR1	July 2018	3.91	< 0.05	0.026
SR1	November 2018	3.66	< 0.05	0.102
SR1	April 2019	6.34	< 0.05	<0.001
SR1	July 2019	3.51	< 0.05	0.148
SR1	November 2019	7.06	< 0.05	<0.001
HB2	July 2018	1.22	< 0.05	<0.001
HB2	November 2018	1.22	< 0.05	<0.001
HB2	April 2019	< 0.5	< 0.05	<0.001
HB2	July 2019	1.78	< 0.05	<0.001
HB2	November 2019	1.42	< 0.05	<0.001
MB1	July 2018	3.40	< 0.05	0.039
MB1	November 2018	2.44	< 0.05	0.340
MB1	April 2019	2.73	< 0.05	<0.001
MB1	July 2019	3.31	< 0.05	<0.001
MB1	November 2019	2.46	< 0.05	<0.001
BB1	July 2018	0.77	< 0.05	<0.001
BB1	November 2018	1.23	< 0.05	0.11
BB1	April 2019	0.62	< 0.05	<0.001
BB1	June 2019	0.41	0.39	0.022
BB1	July 2019	1.27	0.27	<0.001
BB1	November 2019	1.31	< 0.05	0.101
BB4	July 2018	3.08	0.29	<0.001
BB4	November 2018	7.39	0.11	0.001
BB4	April 2019	5.11	< 0.05	<0.001
BB4	June 2019	1.38	< 0.05	0.009
BB4	July 2019	6.50	< 0.05	<0.001
BB4	November 2019	2.24	0.13	<0.001
BB5	July 2018	5.21	< 0.05	<0.001
BB5	November 2018	6.82	< 0.05	0.032
BB5	April 2019	7.26	< 0.05	<0.001
BB5	June 2019	1.83	< 0.05	0.002
BB5	July 2019	6.31	< 0.05	<0.001
BB5	November 2019	6.20	< 0.05	<0.001

Table S 3: pre-sampling data from a site (called BB3 ds) close to the inflow into pond A and site ‘us’.

	Date	Inflow pond A	us
NO_3-N [mg/l]	Nov-18	0.000	0.000
	Apr-19	0.000	0.000
	Jul-19	0.008	0.018
	Jun-20	0.059	0.045
NH_4-N [mg/l]	Nov-18	7.389	7.436
	Apr-19	5.614	6.096
	Jul-19	6.799	7.130
	Jun-20	3.508	3.966
ortho - PO_4-P [mg/l]	Nov-18	0.111	0.134
	Apr-19	0.000	0.000
	Jul-19	0.000	0.000
	Jun-20	0.013	0.016